WETLANDS AND URBANIZATION

Implications for the Future

Final Report of the Puget Sound Wetlands and Stormwater Management Research Program

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Edited by Amanda L. Azous and Richard R. Horner

Washington State Department of Ecology, Olympia, WA.

King County Water and Land Resources Division

and the

University of Washington, Seattle, WA.

Table of Contents

SECTION 1 OVERVIEW OF THE PUGET SOUND WETLANDS AND STORMWATER MANAGEMENT RESEARCH PROGRAM

by Richard R. Horner

INTRODUCTION

The Puget Sound Wetlands and Stormwater Management Research Program (PSWSMRP) was a regional research effort intended to define the impacts of urbanization on wetlands. The wetlands chosen for the study were representative of those found in the Puget Sound lowlands and most likely to be impacted by urban development. The program's goal was to employ the research results to improve the management of both urban wetland resources and stormwater.

This overview paper begins by defining the issues facing the program at its inception. It then summarizes the state of knowledge on these issues existing at the beginning and in the early stages of the program. The paper concludes by outlining the general experimental design of the study. Subsequent papers present the specific methods used in the various monitoring activities.

THE ISSUES

The PSWSMRP was inspired by proposals of stormwater managers and developers in the 1980s to store urban runoff in wetlands to prevent flooding and to protect stream channels from the erosive effects of high peak flow rates (see Athanas 1988 and McArthur 1989 for discussion of the use of wetlands for runoff quantity control). Stormwater managers were also interested in exploiting the known ability of wetlands to capture and to retain pollutants in stormwater, interrupting their transport to downstream water bodies (see Athanas 1988, Chan et al. 1981, Hickok 1980, Lakatos and McNemar 1988, Livingston 1988, and McArthur 1989 for discussion of the use of wetlands for runoff quality control).

In response to proposals to use wetlands for urban runoff storage, natural resources managers argued that flood storage and pollutant trapping are only two of the numerous ecological and social functions filled by wetlands. Among the other values of wetlands are groundwater recharge and discharge; shoreline stabilization; and food chain, habitat, and other ecological support for fish, waterfowl, and other species (Office of Technology Assessment 1984, Zedler and Kentula 1986). Resource managers further contended that using wetlands for stormwater management could damage their other functions (Livingston 1988; Newton 1989; Brown 1985; Canning 1988; ABAG 1986). They noted the general lack of information on the types and extent of impacts to wetlands used for stormwater treatment (Chan et al. 1981; Brown 1985; ABAG 1986; Canning 1988; Woodward-Clyde Consultants 1991).

Several researchers have suggested that findings about the impacts of municipal wastewater treatment in wetlands are relevant to stormwater treatment in wetlands (Chan et al. 1981; Silverman 1983). In some cases, wastewater treatment in wetlands has caused severe ecological disruptions (US EPA 1985), particularly when wastewater delivery is uncontrolled (Wentz 1987). A number of studies have raised concerns about possible long-term toxic metal accumulations, biomagnification of toxics in food chains, nutrient toxicity, adverse ecological changes, public health problems, and other impacts resulting from wastewater treatment in wetlands (Benforado 1981; Guntspergen and Stearns 1981; Sloey, Spangler, and Fetter 1978; Dawson 1989).

Other researchers have reported negative impacts on wetland ecosystems from wastewater treatment. Wastewater additions can lead to reduced species diversity and stability, and a shift to simpler food chains (Heliotis 1982; Brennan 1985). Wastewater treatment in natural northern wetlands tended to promote the dominance of cattails (Typha sp.) (R. H. Kadlec 1987). In addition, animal species diversity usually declined. Discharge of wastewater to a bog and marsh wetland eliminated spruce and promoted cattails in both the bog and marsh portions (Stark and Brown 1988). Thirty years of effluent discharge to a peat bog caused parts of the bog to become monoculture cattail marsh (Bevis and Kadlec 1978). Application of chlorinated wastewater to a freshwater tidal marsh reduced the diversity of annual plant species (Whigham, Simpson, and Lee 1980). These findings on the effects of wastewater applications to wetlands have probable implications for the use of wetlands for stormwater treatment.

Despite the controversy over use of natural wetlands for stormwater treatment, it became apparent in early discussions on the subject that wetlands in urbanizing watersheds will inevitably be impacted by urbanization, even if there is no intention to use them for stormwater management. For example, the authors of a U.S. Environmental Protection Agency (US EPA) handbook on use of freshwater wetlands for stormwater management (US EPA 1985) stated that the handbook was not intended to be a statement of general policy favoring the use of wetlands for runoff management, but acknowledged that some 400 communities in the Southeast were already using wetlands for this purpose. Moreover, directing urban runoff away from wetlands in an effort to protect them can actually harm them. Such efforts could deprive wetlands of necessary water supplies, changing their hydrology (McArthur 1989) and threatening their continued existence as wetlands. In addition, where a wetland's soil substrate is subsiding, continuous sediment inputs are necessary to preserve the wetland in its current condition (Boto and Patrick 1978). Directing runoff to wetlands can help to furnish nutrients that support wetland productivity (McArthur 1989).

In its early years, the PSWSMRP focused on evaluating the feasibility of incorporating wetlands into urban runoff management schemes. Given this objective, the researchers initially viewed the issues more from an engineering than a natural science perspective. However, in later years, an appreciation of the fact that urban runoff reaches wetlands whether intended or not led the researchers to shift their inquiry to more fundamental questions about the impact of urbanization on wetlands. Thereafter, the Program's point of view ultimately merged natural science and engineering considerations. The information yielded by the Program will, therefore, be useful to wetland and other scientists, as well as to stormwater managers.

IMPACTS OF URBANIZATION ON WETLANDS

Urbanization impacts wetlands in numerous direct and indirect ways. For example, construction reportedly impacts wetlands by causing direct habitat loss, suspended solids additions, hydrologic changes, and altered water quality (Darnell 1976). Indirect impacts, including changes in hydrology, eutrophication, and sedimentation, can alter wetlands more than direct impacts, such as drainage and filling (Keddy 1983). Urbanization may affect wetlands on the landscape level, through loss of extensive areas, at the wetland complex level, through drainage or modification of some of the units in a group of closely spaced wetlands, and at the level of the individual wetland, through modification or fragmentation (Weller 1988). Over the past several decades, it has become increasingly apparent that untreated runoff is the primary threat to the country's water quality. There has, consequently, been substantial research about the relationship between urbanization and runoff quality and quantity. However, the PSWSMRP focused primarily on the impacts of runoff on wetlands themselves, and not on the effects of urbanization on runoff flowing to wetlands.

Runoff can alter four major wetland components: hydrology, water quality, soils, and biological resources (US EPA 1993; Johnson and Dean 1987). Because impacts to wetland components are not distinct from one another but interact (US EPA 1993), it is difficult to distinguish between the effects of each impact or to predict the ultimate condition of a wetland component by simply aggregating the effects of individual impacts (Hemond and Benoit 1988). Moreover, processes within wetlands interact in complex ways. For example, wetland chemical, physical, and biological processes interact to influence the retention, transformation, and release of a large variety of substances in wetlands. Increased peak flows transport more sediment to wetlands that, in turn, may alter the wetlands' vegetation communities and impact animal species dependent on the vegetation.

SOURCES OF IMPACTS TO WETLANDS

Brief consideration of how urbanization affects runoff illustrates the potential for dramatic alteration of wetlands. Hydrologic change is the most visible impact of urbanization. Hydrology concerns the quantity, duration, rates, frequency and other properties of water flow. It has been called the linchpin of wetland conditions (Gosselink and Turner 1978) because of its central role in maintaining specific wetland types and processes (Mitsch and Gosselink 1993). Moreover, impacts on water quality and other wetland components are, to a considerable degree, a function of hydrologic changes (Leopold 1968). Of all land uses, urbanization has the greatest ability to alter hydrology. Urbanization typically increases runoff peak flows and total flow volumes and damages water quality and aesthetic values. For example, one study comparing a rural and an urban stream found that the urban stream had a more rapidly rising and falling hydrograph, and exhibited greater bed scouring and suspended solids concentrations (Pedersen 1981).

Pollutants reach wetlands mainly through runoff (PSWQA 1986; Stockdale 1991). Urbanized watersheds generate large amounts of pollutants, including eroded soil from construction sites, toxic metals and petroleum wastes from roadways and industrial and commercial areas, and nutrients and bacteria from residential areas. By volume, sediment is the most important nonpoint pollutant (Stockdale 1991). At the same time that urbanization produces larger quantities of pollutants, it reduces water infiltration capacity, yielding more surface runoff. Pollutants from urban land uses are, therefore, more vulnerable to transport by surface runoff than pollutants from other land uses. Increased surface runoff combined with disturbed soils can accelerate the scouring of sediments and the transport and deposition of sediments in wetlands (Loucks 1989; Canning 1988). Thus, there is an intimate connection between runoff pollution and hydrology.

INFLUENCE OF WETLAND AND WATERSHED CHARACTERISTICS ON IMPACTS TO WETLANDS

Watershed and wetland characteristics both influence how urbanization affects wetlands. For example, impacts of highways on wetlands are affected by such factors as highway location and design, watershed vulnerability to erosion, wetland flushing capacity, basin morphology, sensitivity of wetland biota, and wetland recovery capacity (Adamus and Stockwell 1983). Regional storm patterns also have a significant influence on impacts to wetlands (US EPA 1993). Hydrologic impacts are affected by such factors as watershed land uses; wetland to watershed areal ratios; and wetland soils, bathymetry, vegetation, and inlet and outlet conditions (Reinelt and Horner 1990; US EPA 1993). It is apparent that any assessment of the impacts of urbanization on a wetland should take into account the landscape in which the wetland is located. Whigham, Chitterling, and Palmer (1988), for example, suggested that a landscape approach might be useful for evaluating the effect of cumulative impacts on a wetland's water quality function. The rationale for such an approach is that most watersheds contain more than one wetland, and the influence of a particular wetland on water quality depends both on the types of the other wetlands present and their positions in the landscape.

IMPACTS OF URBANIZATION ON WETLANDS

Hydrologic Impacts

The direct impacts of hydrologic changes on wetlands are likely to be far more dramatic, especially over the short term, than other impacts. Hydrologic changes can have large and immediate effects on a wetland's physical condition, including the depth, duration, and frequency of inundation of the wetland. It is fair to say that changes in hydrology caused by urbanization can exert complete control over a wetland's existence and characteristics. A SWMM model run reported by Hopkinson and Day (1980) predicted that urbanization bordering a swamp forest would increase runoff volumes by 4.2 times. Greater surface runoff is also likely to increase velocities of inflow to wetlands, which can disturb wetland biota and scour wetland substrates (Stockdale 1991). Increased amounts of stormwater runoff in wetlands can alter water level response times, depths, and duration of water detention (US EPA 1993). Reduction of watershed infiltration capacity is likely to cause wetland water depths to rise more rapidly following storm events. Diminished infiltration in wetland watersheds can also reduce stream baseflows and ground water supplies to wetlands, lengthening dry periods and impacting species dependent on the water column (Azous 1991).

Water Quality Impacts

Direct Water Quality Impacts -- Prior to the PSWSMRP study, there was very little information specifically covering the impacts of urban runoff on water quality within wetlands (Stockdale 1991). On the other hand, there have been extensive inquiries into the effects of urbanization on runoff and receiving water quality generally. See, e.g., US EPA 1983, summarizing the results of the Nationwide Urban Runoff Program. Much of this information undoubtedly is suggestive of the probable effects of urban runoff on wetland water quality. There have also been numerous "before and after" studies evaluating the effectiveness of wetlands for treatment of municipal wastewater and urban runoff. See, e.g., ABAG 1986; Brown 1985; Chan et al. 1981; Dawson 1989;

Franklin and Frenkel 1987; Hickok et al. 1977; Hickok 1980; Lynard et al. 1980; Martin 1988; Morris et al. 1981; and Oberts and Osgood 1988. Many of these studies have focused on the effectiveness of wetlands for water treatment rather than on the potential for such schemes to harm wetland water quality.

Nevertheless, data on the quality of inflow to and pollutant retention by wetlands are likely to give some indication of the effects of urban runoff on wetland water quality. Studies on the effects of wastewater and runoff on other wetland components, such as vegetation, also may provide indirect evidence of impacts on wetland water quality. See, e.g., Bevis and Kadlec 1978; Brennan 1985; Chan 1979; Ehrenfeld and Schneider 1983; Isabelle et al. 1987; Morgan and Philipp 1986; Mudrock and Capobianco 1979; Stark and Brown 1988; Tilton and Kadlec 1979; and Whigham, Simpson, and Lee 1980. A number of researchers have warned of the risks of degradation of wetland water quality and other values from intentional routing of runoff through wetlands (see ABAG 1986; Brown 1985; Canning 1988; Chan et al. 1981; Galvin and Moore 1982; and Silverman 1983). Subsequent papers in this monograph describe the results of water quality impact studies performed by the program.

Hydrological Impacts on Water Quality -- Hydrology influences how water quality changes will impact wetlands. Hydrologic changes can make a wetland more vulnerable to pollution (Harrill 1985). Increased water depths or frequencies of flooding can distribute pollutants more widely through a wetland (Stockdale 1991). How wetlands retain sediment is directly related to flow characteristics, including degree and pattern of channelization, flow velocities, and storm surges (Brown 1985). Toxic materials can accumulate more readily in quiescent wetlands (Oberts 1977). In a study on use of wetlands for stormwater treatment, Morris et al. (1981) found that wetlands with a sheet flow pattern retained more phosphorus, nitrogen, suspended solids, and organic carbon than channelized systems, which were ineffective.

Changes in hydroperiod can also affect nutrient transformations and availability (Hammer 1992) and the deposition and flux of organic materials (Livingston 1989). Fries (1986) observed higher phosphorus concentrations in stagnant than in flowing water. In wetland soils, the advent of anaerobic conditions can transform phosphorus to dissolved forms (US EPA 1993). Lyon et al. (1987) reported that anaerobic conditions in flooded emergent wetlands increased nutrient availability to wetland plants, compared to infrequently flooded sites.

Impacts to Wetland Soils

Hydrologic Impacts to Wetland Soils -- Flow characteristics within wetlands directly influence the rate and degree of sedimentation of solids imported by runoff (Brown 1985). If unchecked, excessive sedimentation can alter wetland topography and soils, and ultimately result in the filling of wetlands. Alternatively, elevated flows can scour a wetland's substrate (Loucks 1989), changing soil composition, and leading to more channelized flow. Materials accumulated over several hundred years could, therefore, be lost in a matter of decades (Brinson 1988).

Water Quality Impacts to Wetland Soils -- The physical, chemical, and biological characteristics of wetland soils change as they are subjected to urban runoff (US EPA 1993). The physical effects of runoff on wetland soils, including changes in texture, particle sizes distributions, and degree of saturation are not well documented (US EPA

1993). However, a wetland's soil can be expected to acquire the physical characteristics of the sediments retained by the wetland.

Suspended matter has a strong tendency to absorb and adsorb other pollutants (Stockdale 1991). Sedimentation, therefore, is a major mechanism of pollutant removal in wetlands (Chan et al. 1981; Silverman 1983). Chemical property changes in wetland soils typically reflect sedimentation patterns (ABAG 1979; Schiffer 1989). Materials are often absorbed by wetland soils after entering a wetland, as well (Richardson 1989).

When nutrient inputs to wetlands rise, temporary or long-term storage of nutrients in ecosystem components, including soils, can increase (J.A. Kadlec 1987). Rates of nutrient transfer among ecosystem components and flow through the system may also accelerate. When chlorinated wastewater was sprayed onto a freshwater tidal marsh, surface litter accumulated nitrogen and phosphorus (Whigham, Simpson, and Lee 1980). However, although wetland soils can retain nutrients, a change of conditions, such as the advent of anaerobiosis and changed redox potential, can transform stored pollutants from solid to dissolved forms, facilitating export from the soil. (US EPA 1993). The capacity of wetland soils to retain phosphorus becomes saturated over time (Richardson 1985; Nichols 1983; R.W. Beck and Associates 1985). If the soil becomes saturated with phosphorus, release is likely.

Wetland soils can also trap toxic materials, such as metals (US EPA 1993). Horner (1988) found that there were high toxic metals accumulations in inlet zones of wetlands affected by urban runoff. Mudrock and Copobianco (1979) observed increased sediment metals concentrations in several locations in a wetland receiving wastewater. The quantity of metals that a wetland can absorb without damage depends on the rate of metals accretion and degree of burial (US EPA 1985). If stormwater runoff alters soil pH and redox potential, many stored toxic materials can become immediately available to biota (Cooke 1991).

Water quality impacts on wetland soils can eventually threaten a wetland's existence. Where sediment inputs exceed rates of sediment export and soil consolidation, a wetland will gradually become filled. Filling by sediment is a particular concern for wetlands in urbanizing areas (Stockdale 1991). Many wetlands have an ability to retain large amounts of sediment. For example, Hickok (1980) reported that a wetland captured 94% of suspended solids from stormwater. Oberts and Osgood (1988) observed that a stormwater treatment wetland lost 18% of permanent storage volume and 5% of total storage volume because of high rates of solids retention.

Impacts to Vegetation

Impacts on wetland hydrology and water quality can, in turn, affect wetland vegetation. Horner (1988) stated that emergent zones in Pacific Northwest wetlands receiving urban runoff are dominated by an opportunistic grass species, Phalaris arundinaceae, while non-impacted wetlands contain more diverse groupings of species. Ehrenfeld and Schneider (1983) observed marked changes in community structure, vegetation dynamics, and plant tissue element concentrations in New Jersey Pine Barrens swamps receiving direct storm sewer inputs, compared to swamps receiving less direct runoff. However, human impacts on wetland ecosystems can be quite subtle. For example, Keddy (1983), upon reconsidering data from two prior studies of ecological changes in wetlands, concluded that human influences, and not natural succession, as originally believed, were the principal causes of change in the vegetation of two New England wetlands.

Hydrologic Impacts on Vegetation -- Hydrologic changes can have significant impacts on the livelihood of the whole range of wetland flora, from bacteria to the higher plants. Hickok et al. (1977) observed that microbial activity in wetland soils correlated directly to soil moisture. However, surface microbial activity decreased when soils were submerged and became anaerobic (Hickok 1980). To a greater or lesser degree, wetland plants are adapted to specific hydrologic regimes. For example, Bedinger (1978) observed that frequency and duration of flooding determined the distribution of bottomland tree species. Flood plain terraces with different flooding characteristics had distinct species compositions. Increased watershed imperviousness can cause faster runoff velocities during storms that can impact wetland biota (Stockdale 1991). However, as watersheds become more impervious, stream base flows and groundwater supplies can decline. As a result, dry periods in wetlands may become prolonged, impacting species dependent on the inundation (Azous 1991; US EPA 1985). Changes in average depths, duration, and frequency of inundation ultimately can alter the species composition of plant and animal communities (Stockdale 1991).

There have been numerous reports on the tolerance to flooding of wetland and non-wetland trees and plants. See, e.g., Green (1947); Brink (1954); Ahlgren and Hansen (1957); Rumberg and Sawyer (1965); Minore (1968); Gill (1970); Cochran (1972); Teskey and Hinckley (1977a, b, c, d); Bedinger (1978); Whitlow and Harris (1979); Davis and Brinson (1980); Walters et al. (1980); McKnight et al. (1981); Chapman et al. (1982); Jackson and Drew (1984); Kozlowski (1984); Thibodeau and Nickerson (1985); and Gunderson, Stenberg, and Herndon (1988). While flooding can harm some wetland plant species, it promotes others (US EPA 1993). There is little information available on the impacts of hydrologic changes on emergent wetland plants, although Kadlec (1962) identified several species that can tolerate extended dry periods. Rumberg and Sawyer (1965) reported that hay yields in native wet meadows increased with the length of flood irrigation if depths remained at 13 cm or less and declined if depths stayed at 19 cm for 50 days or longer.

Plant species often have specific germination requirements, and many are sensitive to flooding once established (Niering 1989). The life stage of plant species is an important determinant of their flood tolerances. While mature trees of certain species may survive flooding, the establishment of saplings could be retarded (Stockdale 1991). Where water levels are constantly high, wetland species may have a limited ability to migrate, and may be able to spread only through clonal processes because of seed bank dynamics (van der Valk 1991). The result may be reduced plant diversity in a wetland. However, anaerobic conditions can increase the availability of nutrients to wetland plants (Lyon, Drobney, and Olsen 1986).

Hydrologic impacts on individual plant species eventually translate into long-term alterations of plant communities (US EPA 1985). Changes in hydroperiod can cause shifts in species composition, primary productivity (US EPA 1985), and richness (Cooke 1991). Ehrenfeld and Schneider (1983) theorized that changes in hydrology were among the causes of a decline of indigenous plant species and an increase in exotic species in New Jersey Pine Barrens cedar swamps. In general, periodic inundation yields more plant diversity than either constantly wet or dry conditions (Conner et al. 1981; Gomez and Day 1982). However, early results of the PSWSMRP indicated that wetlands with wider water level fluctuations have lower species richness than systems with lower water level fluctuations (Azous 1991, Cooke and Azous 1992). Monitoring in a Cannon Beach, Oregon wastewater treatment wetland revealed little change in herbaceous and shrub plant cover after two years of operation, except in channelized and deeply flooded portions, where herbaceous cover decreased (Franklin and Frenkel 1987). Slough sedge cover increased slightly in a shallowly flooded area. In 1986, flooding stress was observed in red alder trees in deeper parts of the wetland. Thibodeau and Nickerson (1985) examined a wetland, part of which was drained and part of which was impounded to a greater depth. Vegetation in the drained portion became more dense and diverse, but there was a marked decline in the number of species in the flooded portion after three years.

Please see Hydrologic Effects on Vegetation Communities, later in this volume, for the results of the PSWSMRP study on the effects of water level changes on wetland vegetation.

Water Quality Impacts on Vegetation -- High suspended solids inputs can reduce light penetration, dissolved oxygen, and overall wetland productivity (Stockdale 1991). However, inflow containing high concentrations of nutrients can promote plant growth. Tilton and Kadlec (1979) reported, for example, that in a wastewater treatment wetland, plants closer to the discharge point had greater biomass and higher concentrations of phosphorus in their tissues, and the cattails were taller. When nutrient inputs to wetlands increase, they may be stored either temporarily or over the long-term in ecosystem components, including vegetation (J.A. Kadlec 1987). Rates of nutrient movement, by transfer among ecosystems components and through the system, may accelerate.

Toxic materials in runoff can interfere with the biological processes of wetland plants, resulting in impaired growth, mortality, and changes in plant communities. The amount of metals absorbed by plants is, for some species, a function of supply. Ehrenfeld and Schneider (1983) reported that, in cedar swamps in the New Jersey Pine Barrens, plants took up more lead when direct storm sewer inputs were present than when runoff was less direct. The degree to which plants bioaccumulate metals is highly variable. Chan (1979) stated that pickleweed (Salicornia sp.) concentrated metals, especially zinc and cadmium, more than mixed marsh and upland grass vegetation. However, plants in a brackish marsh that had received stormwater runoff for more than 20 years did not appear to concentrate copper, cadmium, lead, and zinc any more than plants in control wetlands not receiving storm water (Chan et al. 1981).

While toxic metals accumulate in certain species, such as cattails, without causing harm, they interfere with the metabolism of other species (Stockdale 1991). Toxic metals can harm certain species by interfering with nitrogen fixation (Wickcliff et al. 1980). Metals can also impinge on photosynthesis in aquatic plants, such as water weed (Elide sp.) (Brown and Rattigan 1979). Portele (1981) reported that roadway runoff containing toxic metals had an inhibitory effect on algae. Marshall (1980) found in a bioassay study of the effects of stormwater on algae, that nutrients did not stimulate growth as much as predicted because of the presence of metals in the stormwater. Isabelle et al. (1987) found that the germination rates of wetland plants exposed to roadside snow melt in several concentrations varied inversely with snow melt concentration.

Changes in plant community composition may be the major impact of pollution in wetlands. Morgan and Phillip (1986) stated that the major effect of residential and agricultural runoff with high pH and nitrate concentrations was to cause indigenous aquatic macrophytes of the New Jersey Pine Barrens to be replaced by non-native species. Ehrenfeld and Schneider (1983) also reported marked changes in plant community structure and vegetation dynamics in Pine Barrens cedar swamps where direct storm sewer inputs were present. Isabelle et al. (1987) found that, where wetland plants had been exposed to roadside snow melt in several concentrations, community biomass, species diversity, evenness, and richness after one month of growth varied inversely with snow melt concentration. Impacts were not as severe where runoff was less direct.

Impacts to Wetland Fauna

Hydrologic Impacts on Wetland Fauna -- Hydrologic changes can have as great an effect on wetland animal as on plant communities. Nordby and Zedler (1991) reported that, in two coastal marshes, animal species richness and abundance declined as hydrologic disturbance increased. Shifts in plant communities as a result of hydrologic changes can have impacts on the preferred food supply and cover of such animals as waterfowl.

Increased imperviousness in wetland watersheds can reduce stream base flows and groundwater supplies, prolonging dry periods in wetlands and impacting species dependent on the water column (Azous 1991). Many amphibians require standing water for breeding, development, and larval growth. Amphibians and reptile communities may experience changes in breeding patterns and species composition with changed water levels (Minton 1968 in Azous 1991). Because amphibians place their eggs in the water column, the eggs may be directly damaged by changes in water depth. Alterations in hydroperiods can be especially harmful to amphibian egg and larval development if water levels decline and eggs attached to emergent vegetation are exposed and desiccated (Lloyd-Evans 1989 in Azous 1991). Water temperature changes that accompany shifting hydrology may also impact egg development (Richter et al. 1991).

Hydrologic changes have implications for other wetland animals, as well. Alterations to water quality and wetland soils caused by hydrologic changes may negatively affect animal species. For example, increased peak flows that accelerate sedimentation in wetlands or cause scouring can damage fish habitat (Canning 1988). Mortality of the eggs and young of waterfowl during nesting periods may rise if water depths become excessive. (US EPA 1993). Johnsgrad (1956) reported that water level fluctuations resulting from an artificial impoundment in eastern Washington State caused a redistribution of bird populations. Flooding of potholes by the impoundment reduced waterfowl production and forced breeding waterfowl into the remaining smaller potholes. Hydrologic changes may impact mammal populations in wetlands by diminishing vegetative habitat and by increasing the potential for proliferation of disease organisms and parasites as base flows become shallower and warmer (Lloyd-Evans 1989). There is concern about maintaining habitat around wetlands that are receiving stormwater in order to permit free movement of animals during storm events (US EPA 1993).

Water Quality Impacts to Wetland Fauna -- Pollutants can have both direct and indirect effects on wetland fauna. Portele (1981) reported that road runoff containing toxic metals had an inhibitory effect on zooplankton, in addition to algae. Azous (1991) reported a significant negative correlation between water conductivity, a general indicator of dissolved substance concentrations, and amphibian species richness. Aquatic organisms, particularly amphibians, readily absorb chemical contaminants (Richter and Wisseman 1990). Thus, the status of such organisms is an effective indicator of a wetland's health. The degree of bioaccumulation of metals in wetland animals varies by species. In a brackish marsh that had received storm runoff for 20 years, there was no observed bioaccumulation of metals in benthic invertebrates (Burstynsky 1986). However, a filter-feeding amphipod (Corophium sp.), known for its ability to store lead in an inert crystal form, accumulated significant amounts of lead. Water quality changes can indirectly harm fish and wildlife by reducing the coverage of plant species preferred for food and shelter (Mitsch and Gosselink 1993; Weller 1987 and Lloyd-Evans 1989 in Azous 1991).

Please see the discussions of amphibian, emergent aquatic insect, bird, and small mammal communities in relation to watershed development and habitat conditions, later in this volume, for the results of the PSWSMRP study on the effects of hydrologic and water quality changes on wetland animals.

Use of Wetlands for Stormwater Treatment

Impacts from intentional use of wetlands for stormwater management could be more harmful than those that would occur with incidental drainage from an urbanized watershed. For example, raising the outlet and controlling the outflow rate would, in general, change water depths and the pattern of rise and fall of water. Structural revisions to improve pollutant trapping ability would increase toxicant accumulations, in addition to the direct effects of construction. On the other hand, stormwater management actions could be linked with efforts to upgrade wetlands that are already highly damaged.

PUGET SOUND WETLANDS AND STORMWATER MANAGEMENT RESEARCH PROGRAM DESIGN

Representatives of the stormwater and resource management communities in the Puget Sound area of Washington State formed a committee in early 1986 to consider how to best resolve questions concerning wetlands and stormwater runoff. Committee members came from federal, state, and local agencies; academic institutions; and other local interests. The Resource Planning Section of the government of King County, Washington, coordinated the committee's work. The committee's initial effort was to enumerate the wetland resources that are implicated in urban stormwater management decisions and to identify the general types of effects that runoff could have on these resources. The committee members also oversaw the preparation of a literature review, designed to determine the extent to which previous work could address the issues before them, and a management needs survey.

Literature Review and Management Needs Survey

The principal activity of the Program's first year was a comprehensive literature review, which concluded with a report (Stockdale 1986a) and an annotated bibliography (Stockdale 1986b) covering the reported research and observations relevant to the issue of stormwater and wetlands. The review was updated in 1991 (Stockdale 1991). These reviews concentrated on what was known and what was not known about these issues at the time. Best known was the performance of wetlands in capturing pollutants, mostly derived from studies on their ability to provide advanced treatment to municipal wastewater effluents. Only a small body of information pertained to stormwater. The greatest shortcoming of the literature concerned the ecological impacts to wetlands created by any kind of waste stream. The literature reviews also made clear the dearth of research on any aspect of Pacific Northwest wetlands, in contrast to some other areas of the country. Many detailed aspects of the subject of stormwater and wetlands were very poorly covered, including the relative roles of hydrologic and water quality modifications in stressing wetlands and the transport and fate of numerous toxicants in wetlands.

On the basis of their discussions and the literature review, the committee members participated in a formal survey designed to identify the most important needs for reaching the goal of protecting wetlands in urban and urbanizing areas, while improving the management of urban stormwater. The survey involved rating a long list of candidate management needs with respect to certain criteria. Computer processing of the ratings led to the following list of consensus high priority management needs:

- Definition of short and long-term impacts of urban stormwater on palustrine wetlands;
- Management criteria by wetland type;
- Allowable runoff storage schedules that avoid or minimize negative effects on wetlands and their various functions; and
- Features critical to urban runoff water quality improvement in wetlands.

Research Program Design

After completion of the literature review and management needs survey, the committee and staff assembled by King County turned to defining a research program to serve the identified needs. The program they developed included the following major components:

- Wetland survey;
- Water quality improvement study;
- Stormwater impact studies; and
- Laboratory and special field studies.

The purpose of the wetland survey was to provide a broad picture of freshwater wetlands representative of those in the Puget Sound lowlands. The survey covered 73 wetlands throughout lowland areas of King County. One important goal of the survey was to identify how urban wetlands differ from those that are lightly affected by human activity. The survey's design, results, and conclusions were reported by Horner et al. (1988) and Horner (1989). The survey results assisted in designing the remainder of the research program.

The water quality improvement study was an intensive, two-year (1988-1990) effort to answer remaining questions about the water quality functioning of wetlands. Reinelt and Horner (1995) discuss its methods and findings.

The results from the various portions of the Program were used to develop extensive guidelines for coordinated management of urban wetlands and stormwater. These guidelines have been continuously updated and refined throughout the program, as more information became available.

Wetlands Impacted by Urbanization in the Puget Sound Basin

The research program focused primarily on palustrine wetlands because urbanization in the Puget Sound region is impacting this wetland type more than other types. Palustrine wetlands are freshwater systems in headwater areas or isolated from other water bodies (Cowardin et al. 1979). They typically contain a combination of water and vegetation zones. Some palustrine wetlands consist of open water with only submerged or floating plants, or with no vegetation. Others include shallow or deep marsh zones containing herbaceous emergent plants, shrub-scrub vegetation, and/or a forested community.

Two “poor fens” being impacted by urban development were also monitored during the study. Poor fens, commonly confused with true bogs, are a special wetland type that is of considerable interest in northern regions. Under natural conditions, water supply to poor fens consists only of precipitation and groundwater. The lack of surface water inflow restricts nutrient availability, resulting in a relatively unusual plant community adapted to low nutrition and the attendant acidic conditions. Such a community is vulnerable to increased nutrient supply and buffering by surface water additions.

Stormwater Impact Studies

The stormwater impact studies formed the core of the program. This field research was supplemented by the laboratory and special field studies, which allowed investigation of certain specific questions under more control than offered by the broader field studies.

A special effort was made to ensure that research was conducted according to sound scientific design, so that the results and their application in management would be defensible. In order to approximate the classic "before/after, control/treatment" experimental design approach, the impact study included “control” and “treatment” wetlands. The stormwater impact study was conducted in 19 wetlands in King County, approximately half treatment and the remainder control sites. Figure 1 displays these 19 sites and four others, including three in Snohomish County to the north, where special studies were conducted.

The treatment wetlands, located in areas undergoing urban development during the course of the study, were monitored before, during, and after urbanization. The goals of studying these wetlands were to characterize preexisting conditions and to assess the consequences of any changes accompanying urbanization and modification of stormwater inflow. Not all of the treatment watersheds developed as much as anticipated at the outset of the study. The watersheds of the control wetlands ranged from no urbanization to relatively high levels. However, the watersheds of all of these wetlands were characterized by relative stability in land use during the study. The use of control sites made it possible to judge whether observed changes in treatment wetlands were the result of urbanization or of broader environmental conditions affecting all wetlands in the region. Control wetlands were paired with treatment sites on the basis of size, water and plant zone configuration, and vegetation community types. In recognition of the imperfect matches that occur in pairing natural systems, data analyses were performed for various groupings of sites and not just with respect to paired wetlands.

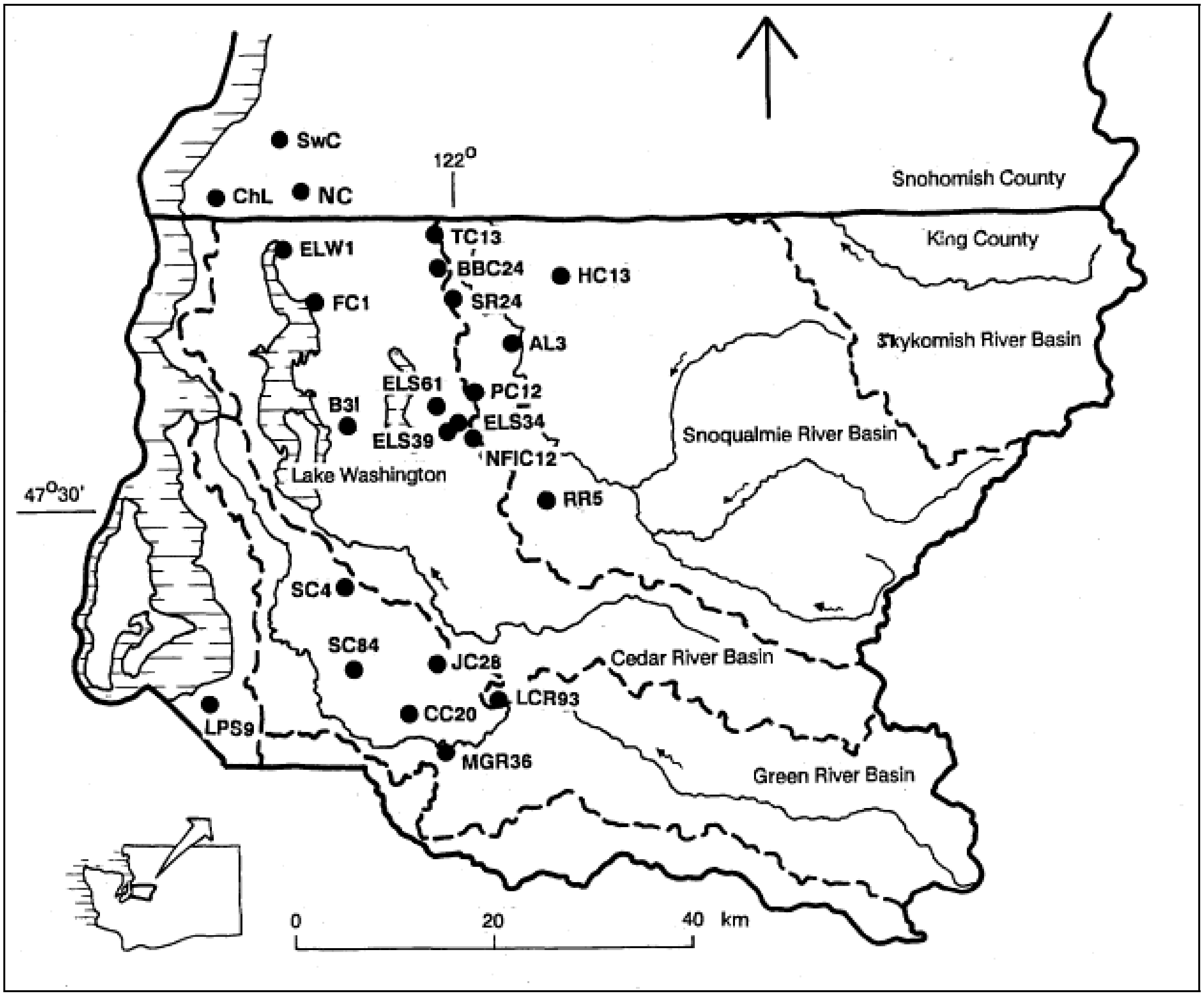

Figure 1. Puget Sound Wetlands and Stormwater Management Research Program study locations.

Because the program was interested in long-term as well as short-term effects, the impact monitoring was continued for eight years. Research in 1988 and 1989 generally provided the baseline data for the treatment wetlands. Data from 1990 reflected the early phase of urbanization in these wetlands. Monitoring resumed in 1993, generally shortly after a phase of building in the watersheds ended. Monitoring in 1995 was intended to document effects that took longer to appear.

Figure 2 illustrates the conceptual framework of the designs of the specific sampling programs pursued in the stormwater impact study and analyzing and interpreting the resulting data. The two blocks on the left of the diagram represent the driving forces determining a wetland's character (Watershed and Surrounding Landscape Conditions and Wetland Morphology). The term "surrounding landscape" signifies that not only a wetland's watershed (the area that is hydrologically contributory to the wetland) but also adjacent land outside of its watershed can influence the wetland. The surroundings include the wetland buffer, corridors for wildlife passage, and upland areas that provide for the needs of some wetland animals. Wetland morphology refers to form and structure and embraces shape, dimensions, topography, inlet and outlet configurations, and water pooling and flow patterns.

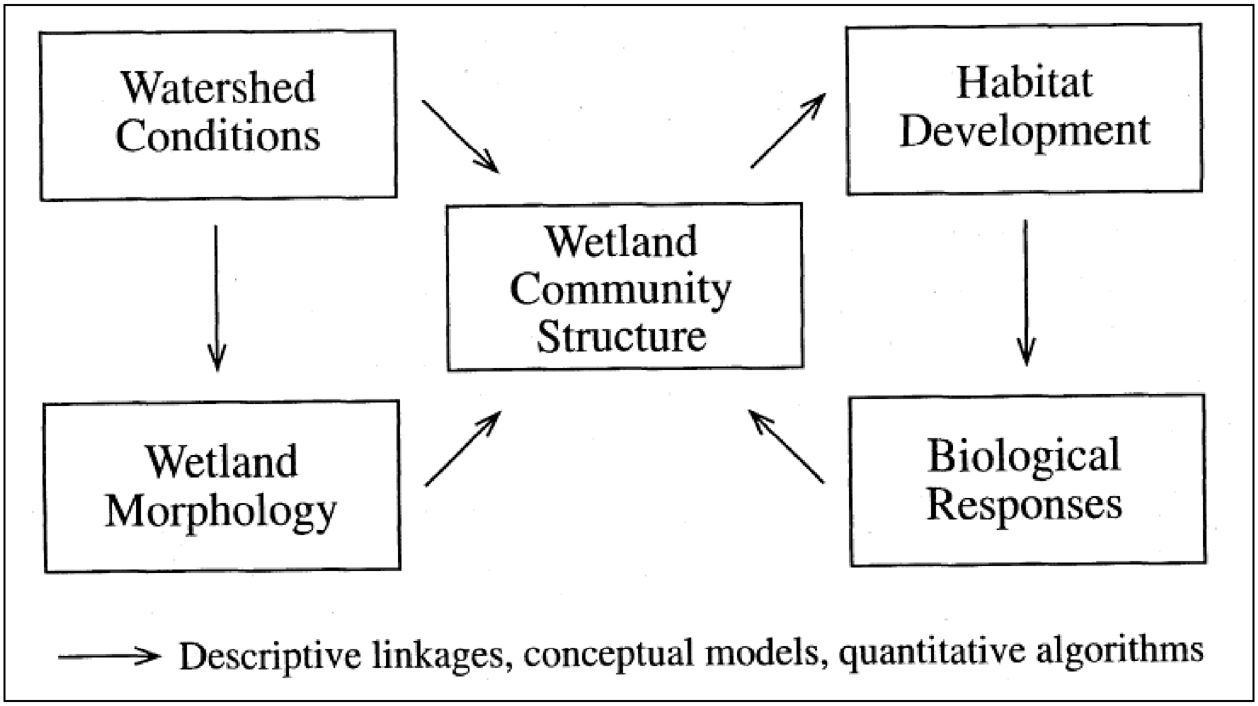

Figure 2. Puget Sound Wetlands and Stormwater Management Research Program experimental strategy.

The central block (Wetland Community Structure) represents the physical and chemical conditions that develop within a wetland and constitute a basis for its structure. Included are both quantity and quality aspects of its water supply and its soil system. Together these structural elements develop various habitats that can provide for living organisms, represented by the block at the upper right of the diagram. Biota will respond depending on habitat attributes, as illustrated by the block at the lower left. It is a fundamental goal of the Puget Sound Wetlands and Stormwater Management Research Program to describe these system components for the representative wetlands individually and collectively.

Connecting lines and arrows on Figure 2 depict the interactions among the components. It is a second fundamental goal of the program to understand and be able to express these interactions, toward the ends of advancing wetlands science and the management of urban wetlands and stormwater. Expression could come in the form of qualitative descriptions, relatively simple conceptual models, or more comprehensive mathematical algorithms. The extent to which definition of these interactions can be developed will determine the thoroughness with which management guidelines and new scientific knowledge can be generated by this research program.

The stormwater impact study examined the five major structural components of wetlands: (1) hydrology, (2) water quality, (3) soils, (4) plants, and (5) animals. Figure 3 presents a typical plan for monitoring of these components. A crest stage gage was used to register maximum water level since the preceding monitoring occasion, and a staff gage gave the instantaneous water level. These readings provided the basis for hydrologic analysis, as detailed in the paper on Morphology and Hydrology in Section 2.

Samples for water quality analysis were taken from the water column in an open water pool, and soil samples were collected at either three or four locations (see Water Quality and Soils in Section 2). Plant cover by species was determined along one or more transect lines, depending on wetland size and complexity of water and vegetation zones. Foliar tissue was sampled for analysis of metals content, and plant standing crop was cut for measurement of biomass gravimetrically. For more on the methods used in these monitoring activities refer to the Vegetation Community paper. Adult insect emergence was continuously monitored using triplicate emergence traps (see Emergent Aquatic Insect Community in Section 2). Amphibian breeding success was monitored along transects (labeled Herp. A, B in Figure 3). Adult amphibians as well as small mammals were live-trapped along other transects (labeled Mammal line A, B). The Section 2 papers titled Amphibian Community and Small Mammal Community elaborate on the methods. Birds were censused at one station as described by the Bird Community paper.

Definition of Watershed and Surrounding Landscape Characteristics

Essential to understanding the relationships between urban stormwater discharge and wetlands ecology was definition of the characteristics of wetland watersheds and surrounding landscapes. Each land use includes distinctive features, such as imperviousness and vegetative cover, that directly affect wetland conditions (Taylor 1993). Use of geographical information in the analysis of the effects of urbanization on wetlands allows the linking of effects with specific land use changes associated with urban development.

To this end, the program used a geographical information system (GIS) to inventory land uses in the watersheds of the study wetlands (Taylor 1993) (see Table 1). The GIS furnished quantitative and graphical representations of land use patterns. Study sites were located on U.S. Geographical Survey 7.5 minute series topographic maps

and the maps were used to locate wetland and watershed boundaries. Aerial photographs from 1989 were digitized into a computer data base and used to delineate wetland boundaries on the basis of wetland vegetation and open water. Land uses were classified according to a standard land use classification scheme. The GIS provided the areas of watersheds, wetlands, and land uses. These data were expressed in three ways: (1) wetland and watershed areas in hectares; (2) watershed land uses and vegetative cover as percentages of watershed areas; and (3) ratios of the areas of watersheds, land uses, and vegetative cover to wetland areas. The most important quantities yielded by the third method were the ratios of watershed and wetland areas (wetland areas were subtracted from their watershed areas in calculating these ratios). The method also was used to determine the ratios of impervious and forested areas to wetland areas. The 1989 GIS data were updated through manual examination of 1995 aerial photographs. In addition, in 1996, the same information was developed for 1000-meter wide bands of the surrounding landscapes using 1995 satellite images.

With regard to calculating watershed imperviousness, the program found that the relevant literature generally did not provide the level of detail necessary to establish the relationships between imperviousness and the land use definitions used in the GIS inventory. The program, therefore, relied on a variety of sources linking specific land uses to imperviousness levels. Estimates of imperviousness were made by using values from the literature for similar land uses (Alley and Veenhuis 1983; Prych and Ebbert 1986; Taylor 1993) and adjusting them according to best professional judgment.

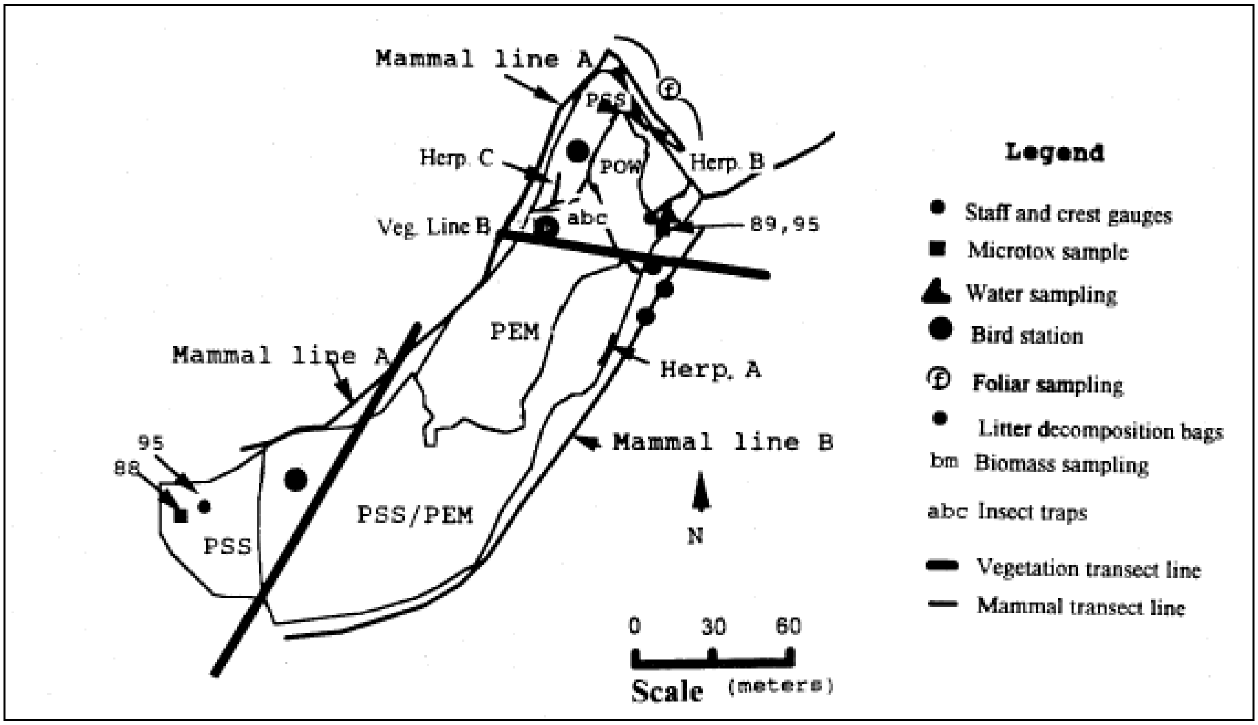

Figure 3. Typical monitoring plan (Patterson Creek 12 wetland).

Table 1. Landscape data for program wetlands.

Site	Watershed Area	Wetland Area	T/C	% Urban Cover 1989	% Urban Cover 1995	% Urban Cover Change	% Forest Cover 1989	% Forest Cover 1995	% Forest Cover Change	% Impervious Cover 1989	% Impervious Cover 1995	% Impervious Cover Change
AL3	47.35	0.81	C	13.3	13.3	0.0	73.9	73.9	0.0	4.1	4.1	0.0
B3I	183.73	1.98	C	74.7	75.2	0.5	0.0	0.0	0.0	54.9	55.4	0.5
ELW1	54.63	3.84	C	56.6	56.6	0.0	0.0	0.0	0.0	19.9	19.9	0.0
FC1	357.34	7.28	C	81.2	81.2	0.0	14.7	14.7	0.0	30.8	30.8	0.0
HC13	359.36	1.62	C	1.5	1.5	0.0	76.6	75.1	-1.5	3.6	3.6	0.0
LCR93	198.22	6.09	C	12.8	11.0	-1.8	44.1	13.0	-31.1	5.8	6.1	0.3
LPS9	183.32	7.69	C	69.8	73.8	4.0	0.0	0.0	0.0	21.8	21.6	-0.2
MGR36	45.73	2.23	C	4.1	4.1	0.0	88.8	88.8	0.0	2.9	2.9	0.0
RR5	64.35	10.52	C	2.4	2.4	0.0	62.4	62.4	0.0	3.4	3.4	0.0
SC4	3.64	1.62	C	12.5	12.5	0.0	46.1	46.1	0.0	11.8	11.8	0.0
SC84	193.04	2.83	C	77.8	78.2	0.4	20.1	19.7	-0.4	18.5	17.0	-1.5
SR24	88.22	10.12	C	0.0	0.0	0.0	100.0	100.0	0.0	2.0	2.0	0.0
TC13	11.74	2.06	C	0.0	0.0	0.0	100.0	89.7	-10.3	2.0	2.3	0.3
BBC24	38.45	2.10	T	10.5	52.7	42.2	89.5	47.4	-42.1	3.4	10.6	7.2
ELS39	69.20	1.74	T	88.8	87.9	-0.9	18.5	10.8	-7.7	24.6	24.2	-0.4
ELS61	27.11	2.02	T	23.9	34.4	10.5	2.5	3.7	1.2	5.1	10.6	5.5
JC28	296.64	12.55	T	54.7	64.9	10.2	34.4	19.8	-14.6	20.0	20.6	0.6
NFIC12	3.24	0.61	T	0.0	100.0	100.0	100.0	0.0	-100.0	2.0	40.0	38.0
PC12	84.58	1.50	T	23.5	34.0	10.5	75.2	64.7	-10.5	5.1	6.8	1.7

a T=treatment wetlands; C=control wetlands.

b ELS39 developed was approximately 15% urban in 1988, before GIS analysis.

Effective Impervious Area (EIA) represents the impervious area that is actually connected to constructed drainage systems. This value was estimated as a proportion of Total Impervious Area (TIA) according to the formula EIA = 0.15 * TIA1.41 (Alley and Veenhuis 1983). This equation was developed in Denver and its accuracy (correlation coefficient = 0.98 and standard error = 0.075) probably varies in other areas. However, Alley and Veenhuis's estimates were compatible with those in Puget Sound lowland hydrologic models (PEI 1990; SCS 1982). After determining EIA and TIA values for each land use. EIAs for entire watersheds were determined using the formula EIADB = $\Sigma 1 \rightarrow k$ (EIAk * LUk), where EIADB is the percentage of watershed area that is effectively imperviousness, k corresponds to the land uses inventoried in the basin, EIAk is the percentage of watershed area associated with land use k, and LUk is the percentage of the watershed classified as land use k. TIAs were calculated using the same formula.

ORGANIZATION OF THE MONOGRAPH

The papers that follow trace the major areas of progress in filling in the conceptual framework presented in Figure 2. The first series of papers provides a descriptive ecology of the palustrine wetlands of the central Puget Sound lowlands, organized according to the major structural components monitored during the program. The next series of papers assesses the effects of urban stormwater and other influences of urbanization observed during the study. The final series makes recommendations for managing urban stormwater and the wetlands subject to it.

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Section 2 Descriptive Ecology of Freshwater Wetlands in the Central Puget Sound Basin

CHAPTER 1 MORPHOLOGY AND HYDROLOGY

by Lorin E. Reinelt, Brian L. Taylor and Richard R. Horner

INTRODUCTION

This chapter provides an overview of the morphologic and hydrologic characteristics of palustrine (isolated or depressional freshwater) wetlands and their watersheds in the central Puget Sound Basin. Natural and anthropogenic factors that affect wetland morphology and hydrology are discussed with particular attention to the effects of development (typically, the conversion of forested lands to urban areas) on changing watershed and wetland hydrology. It was concluded that wetland water level fluctuation (WLF) estimates, measured with staff and crest-stage gages, provide a good overall indicator of wetland hydrologic conditions. Analysis methods and materials used in the PSWSMRP are also presented.

Wetlands are ecosystems that develop at the interface of aquatic and terrestrial environments when hydrologic conditions are suitable. Wetlands are recognized as biologically productive ecosystems offering extensive, high-quality habitat for a diverse array of terrestrial and aquatic species, as well as multiple beneficial uses for humans, including flood control, groundwater recharge and water quality treatment. However, as urbanization of natural landscapes occurs, some or all of the functions and values of wetlands may be affected. Some wetlands may be impacted by direct activities such as filling, draining or outlet modification, while others may be affected by secondary impacts, including increased or decreased quantity and reduced quality of inflow water.

The morphology of a wetland and the wetland's position within the landscape greatly influences its' characteristics. Morphology is used here to describe the wetland's physical shape and form. As a result of a wetland's shape, it may contain significant pooled areas with little or no flow gradient (termed an open-water system), or alternatively, it may show evidence of channelization and contain a significant flow gradient (termed a flow-through system). In some instances, a wetland may also form in a local or closed depression (termed a depressional system).

The outlet condition of a wetland, as defined by the degree of flow constriction, has a direct effect on wetland hydrology and hydroperiod. Finally, a wetland's position in the landscape is also a key factor affecting wetland hydrologic conditions. Palustrine (isolated, freshwater) wetlands usually have relatively small contributing watersheds and often occur in areas with groundwater discharge conditions.

Hydrology is probably the single most important determinant for the establishment and maintenance of specific types of wetlands and wetland processes (Mitsch and Gosselink 1993). Water depth, flow patterns, and the duration and frequency of inundation influence the biochemistry of the soils and are major factors in the selection of wetland biota. Thus, changes in wetland hydrology may influence significantly the soils, plants and animals of particular wetland systems. Precipitation, surface water inflow and outflow, groundwater exchange and evapotranspiration, along with the physical features noted above, are the major factors that influence the hydrology of palustrine wetlands.

PUGET SOUND WETLANDS AND STORMWATER MANAGEMENT RESEARCH PROGRAM

The Puget Sound Wetlands and Stormwater Management Research Program was established to determine the effects of urban stormwater on wetlands and the effect of wetlands on the quality of urban stormwater. There are two primary components of the research program: (1) a study of the long-term effects of urban stormwater on wetlands, and (2) a study of the water quality benefits to downstream receiving waters as urban stormwater flows through wetlands. In both studies, the hydrologic and morphologic conditions of the wetlands had a direct effect on observations involving water quality, soils, and the plant and animal communities.

This paper presents hydrologic information gained from a broad overview of the hydrology of 19 wetlands (representing a variety of watershed development conditions) studied from 1988-95, and specific information on the hydrology of two wetlands (one each in an urban and nonurban area) intensively studied from 1988-90 (B3I and PC12, respectively. (Study site locations are shown in Section 1, Figure 1).

WETLANDS IN URBANIZING AREAS

Wetlands have received increased attention in recent years as a result of continuing wetland losses and impacts resulting from new development. In urbanizing areas, the quantity and quality of stormwater can change significantly as a result of land-use conversion in a watershed. Increases in the quantity of stormwater may result from new impervious surfaces (e.g., roads, buildings), installation of storm sewer piping systems, and removal of trees and other vegetation. On the other hand, decreased inflow of water can result from modifications in surface and groundwater flows. For cases where wetlands are the primary receiving water for urban stormwater from new developments, it is hypothesized that the effects of watershed changes will be manifested through changes in the hydrology of wetlands.

Wetland hydrology is often described in terms of its hydroperiod, the pattern of fluctuating water levels resulting from the balance between water inflows and outflows, topography, subsurface soil, geology, and groundwater conditions (Mitsch and Gosselink, 1986). Wald and Schaefer (1986) referred to seasonal water level changes as the "heartbeat" of Pacific Northwest palustrine systems.

WETLAND HYDROLOGIC FUNCTIONS

Wetlands provide many important hydrologic, ecological, and water quality functions. Specific hydrologic functions include flood protection, groundwater recharge, and streamflow maintenance. Wetlands provide flood protection by holding excess runoff after storms, before slowly releasing it to surface waters. While wetlands may not prevent flooding, they can lower flood peaks by providing detention of storm flows.

Wetlands that are connected to groundwater or aquifers provide important recharge waters. Wetlands retain water, allowing time for surface waters to infiltrate into soils and replenish groundwater. During periods of low streamflow, the slow discharge of groundwater maintains instream flows. The connection of wetlands with streamflows and groundwater make them essential in the proper functioning of the hydrologic cycle.

HYDROLOGY OF PALUSTRINE WETLANDS

The hydrology of palustrine wetlands is governed by the following components: precipitation, evapotranspiration, surface inflow, surface outflow, groundwater exchange, and change in wetland storage (Figure 1-1). In a hydrologic balance, these components are represented by the following equation (Reinelt et al., 1993):

$$P + I +/- G +/- \quad S = ET + O \qquad (1)$$

where P = precipitation; I = surface inflow; G = groundwater exchange, S = change in wetland storage, ET = evapotranspiration; and O = surface outflow.

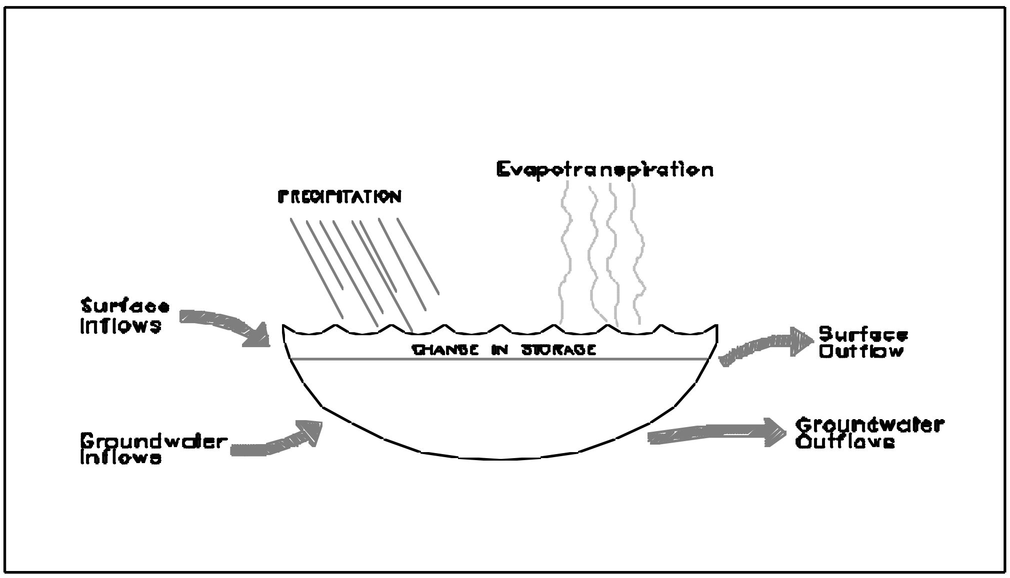

Figure 1-1. Wetland water budget components

Precipitation

Precipitation is determined by regional climate and topography. Approximately 75 percent of the total annual rainfall occurs from October to March during a well-defined wet season in the Puget Sound region. Generally, annual precipitation totals across central Puget Sound increase further east with increasing elevation. Rainfall tends to be more uniform geographically during the wet season, and more variable and intense during short dry-season cloud bursts.

Surface Inflows

Surface inflows result from runoff generation in the wetland's watershed. The quantity of surface inflows are determined by watershed land characteristics such as cover (e.g., impervious surface, forest), soils, and slope, as well as the wetland-to-watershed ratios. The rate of water delivery to the wetland is also affected by the predominant flow type in the watershed (e.g., overland or sheet flow, subsurface flow or interflow, concentrated flow). Generally, as a watershed becomes more developed, with more constructed storm drainage systems, the more rapid the hydrologic response in the watershed.

Groundwater

The role and influence of groundwater on wetland hydrology are highly variable. The exchange of water between the wetland and groundwater is governed by the relative elevations of surface water in the wetland and surrounding groundwater, as well as soil permeability, local geology and topography. Numerous studies have discussed the importance of groundwater in maintaining wetland hydrology (Winter, 1988, Surowiec, 1989, Mitsch and Gosselink, 1986). Wetlands can be discharge or recharge zones for groundwater, or both depending on the time of year. The palustrine wetlands studied in this research are predominantly groundwater discharge zones (i.e., water discharges from groundwater to the wetland).

Groundwater flow to wetlands can be quantitatively estimated using Darcy's Law, an empirical law governing groundwater flow:

$$Q = K * (dH/dL) * A \qquad (2)$$

where K = hydraulic conductivity, dH/dL = the hydraulic or piezometric gradient and A = cross-sectional area or control surface across which groundwater flows.

In the detailed study of two wetlands, shallow and deep piezometers were installed at both wetlands to estimate the horizontal and vertical components, respectively, of groundwater flow to the wetlands.

Change in Wetland Storage

Wetland storage changes seasonally and in response to storm events. The water storage can be estimated as the mean water depth of the wetland multiplied by the areal extent of the wetland (Mitsch and Gosselink, 1986). Seasonal changes in wetland storage are attributable to the local patterns of precipitation and evapotranspiration.

Duever (1988) asserted that the prime factor controlling seasonal fluctuation is drainage basin topography and that wetland water levels generally coincide with regional groundwater levels. Surowiec (1989) noted that steep slopes adjacent to a wetland can also lead to increased groundwater inputs, particularly on a seasonal basis.

Event changes in wetland storage result from increased surface or ground water inputs associated with precipitation. Reinelt and Horner (1990), Azous (1991), and Taylor (1993) referred to this as water level fluctuation, and estimated it for an occasion i as the difference between a crest-stage measurement (peak water level since the previous sampling occasion) and the instantaneous staff-gage measurement.

Evapotranspiration

Evapotranspiration (ET) consists of water that evaporates from wetland water or soils combined with the water that passes through vascular plants that is transpired to the atmosphere. Solar radiation, temperature, wind speed and vapor pressure are the main factors influencing evaporation rates (Linsley et al., 1982).

The ratio of ET to evaporation varies widely depending on vegetation type and site conditions. Reported ET ratios vary between 0.67 and 1.9 (Dolan et al., 1984; Boyd, 1987; Koerselman and Beltman, 1988). Generally, emergent wetland vegetation transpires more than woody vegetation; however, factors such as plant density also effect transpiration rates. Evapotranspiration is greatest from May to August (exceeding 100 mm per month) and least from November to March.

Surface Outflow

Surface outflows are affected by all the hydrologic factors noted above. For wetlands with relatively large watersheds, outflows are often comparable in magnitude to inflows. The physical features that affect surface outflows include outlet conditions, wetland-to-watershed ratios, and wetland morphometry.

RESEARCH METHODS AND WETLAND DESCRIPTORS

Many of the methods and materials used for morphologic, hydrologic, and watershed data collection were previously reported in PSWSMRP papers (Reinelt and Horner, 1990; 1991; Taylor, 1993). This paper provides a summary of the methods, with additional information on data processing and analysis.

Wetland Morphology

Three different measures of wetland morphology that influence the hydrology and hydroperiod of wetlands were defined by Reinelt and Horner (1990): wetland shape/type (open water, flow through, depressional), outlet condition, and wetland-to-watershed ratio. Wetlands were classified as open-water systems if significant open water pools were present and surface water velocities were predominantly low (less than 5.0 cm/s). Wetlands were classified as flow-through systems if there was evidence of channelization and significant water velocities. All depressional wetlands are also open water wetlands.

Outlet conditions were defined by level of constriction (Reinelt and Horner, 1991; Taylor, 1993) as high (e.g., undersized culvert, closed depression, confined beaver dam), or low to moderate (e.g., overland flow to stream, oversized culvert, broad bulkhead or beaver dam). The wetland-to-watershed ratio was determined by the wetland and contributing watershed areas. Watershed areas were delineated based on USGS quadrangle map contours and wetland areas were obtained from the King County Wetlands Inventory (1992). The hydroperiod of wetlands with low wetland-to-watershed ratios (less than 0.05) tends to be dominated by surface inflows, whereas wetlands with higher ratios are more influenced by regional groundwater conditions.

Watershed Characteristics

Changes in land use ultimately affect wetlands receiving water from an urbanizing drainage basin. Different land uses have unique combinations of factors that directly affect watershed hydrology, such as imperviousness and vegetative cover. By collecting information about drainage basin land use, it is possible to link wetland hydroperiod characteristics to specific land uses, as well as general changes associated with urban development.

A geographic information system (GIS) was developed to manage land use data for the watersheds of the study wetlands, and to facilitate quantitative and graphical analysis of land-use patterns. Land-use classifications, based on a national standard (Anderson, 1976), were determined from 1989 aerial photographs and subsequently digitized into Arc/Info (Reinelt et al., 1991). For each study site, the GIS contained information about total watershed and wetland area, and the area and percent of watershed area for each land use type (e.g., urban, agriculture, forest).

Watershed Imperviousness

The literature consistently identifies hydrologic effects of urbanization with increased impervious areas within the watershed (Schueler, 1994). Impervious area increases within a watershed reduce evaporation and infiltration, and as a result of forest clearing for urban conversion also result in a loss of vegetative storage and decreased transpiration (Lazaro, 1979).

Imperviousness was estimated from aerial photos and empirical relationships between land uses and percent impervious cover (Table 1-1, Gluck and McCuen, 1975; Alley and Veenhuis, 1983; KCSWM, 1990). This estimation technique was found to produce results consistent with values used in Puget Sound lowland hydrologic models (PEI, 1990; SCS, 1982). Effective impervious areas (impervious surfaces connected to a storm drainage system) were also estimated according to a formula reported by Alley and Veenhuis (1983) based on drainage basins in the Denver area:

EIA = 0.15 TIA1.41 (R 2 = 0.98, standard error = 7.5%) (3)

where EIA and TIA = percent effective and total impervious area, respectively.

Table 1-1. Impervious and effective impervious areas associated with land uses.

CODE	NATIONAL STANDARD	IA%	EIA%	Reference
111	Low Density SFR (<1 unit/acre)	<15	4	a
112	Med. Density SFR (1-3 unit/acre)	20	10	a
113	High Density SFR (3-7 units/acre)	40	25	a
114	Mobile Homes	70	60	b
115	Low Density MFR (>7 units/acre)	80	72	b
120	Commercial (general)	90	85	b
121	Retail sales and services	80	72	b
123	Offices and professional services	75	66	b
124	Hotels and Motels	75	66	c
131	Light Industrial	60	48	d
132	Heavy Industrial	80	72	c
144	Freeway Right-of-way	100	99	b
151	Energy Facilities	80	72	c
152	Water Supply Facilities	80	72	c
155	Utility Right-of-way	5	1.5	c
160	Community Facilities (general)	75	66	c
161	Educational Facilities	40	27	b
162	Religious Facilities	70	60	c
171	Golf Courses	20	10	b
172	Parks	5	1.5	b
190	Open Land (general)	2	1	c
192	Land being developed	50	37	c
193	Open space - designated	2	1	c
200	Agricultural Land	5	1.5	c
300	Grassland	2	0	c
400-430	Forest Lands	2	0	c
440	Clearcut areas	5	0	c

REFERENCES:

a. King County Surface Water Management (1990)
b. PEI (1990)
c. Estimate based on similar land uses
d. Alley and Veenhuis (1983)

WATERSHED SOILS

The Soil Conservation Service (SCS) Soil Survey for King County (Snyder et al., 1973) was used to evaluate the drainage characteristics of the soils in each of the 19 drainage basins. Two soil parameters were reviewed to determine which would be an appropriate index of the soil hydrologic characteristics relevant to the analysis: permeability and general drainage characteristics.

Soil permeability, is measured as a range of infiltration rates, the units of which are distance/time. Soil permeability for the majority of the soils found in the watersheds was in the range of 2.0 - 6.3 inches/hour. The drainage class is a more general description of the soil characteristics such as “Moderately well-drained” or “Somewhat excessively drained.” Many soils in the Pacific Northwest (e.g. Alderwood series) are underlain by glacial till, a hardpan layer that limits the ultimate depth of percolation and plays an important role in routing subsurface flow. Drainage class was therefore thought to be a better estimator of the hydrologic role of the watershed soils than permeability because it represents the effects of the multiple soil horizons characterized as a particular soil type; an infiltration rate is based on the top soil layer.

For this study, a watershed soils index (WSI) was calculated as an area weighted mean of soil drainage classes found in the study basins. Each of six drainage classes described by the SCS was assigned a number that ranged from 1 to 6, with lower numbers representing poorly drained soils (Table 1-2). The range of the WSI corresponded with the SCS Hydrologic Soil groups, which are used in the Curve Number method of runoff estimation. The WSI was preferred for the analysis because it describes soil drainage to a finer level than the hydrologic soil group.

Table 1-2. Soil drainage classes and watershed soils index (WSI).

Drainage Class	WSI	SCS Hyd. Group (1)	Examples
Very poorly drained	1	(D)	(Muck)
Poorly drained	2	D	Norma, Bellingham
Somewhat poorly drained	3	(D)	(Oridia, Renton)
Moderately well drained	4	C	Alderwood, Kitsap
Well drained	5	B, C	Ragnar, Beausite
Somewhat excessively drained	6	A	Everett, Indianola

(1) Parenthesis indicates soils that were not found in any of the wetland watersheds.

WETLAND HYDROLOGY

Wetland Water Level Measurements and Fluctuation

Water level measurements in wetlands can be made using a variety of gages or instruments. Readings can be either instantaneous, continuous, or representative of a peak or base level since the last site visit. In the PSWSMRP, we utilized staff and crest stage gages to record instantaneous water levels and peak occasion water levels, respectively, during each site visit. At two wetlands (Bellevue 3I and Patterson Creek 12), continuous water levels were recorded over a two-year period (1988-90) using automatic data recorders. Gages were placed in open water areas or areas of channelized flow where water level measurements could be attained throughout most of the year.

The crest-stage and staff gage data were used to estimate wetland water level fluctuation. To estimate the water level fluctuation at a wetland site, two factors were considered: (1) the water level prior to the storm event, hereafter referred to as the base water level, and (2) the water level change resulting from the event. Four methods of calculating water level fluctuation were investigated in a preliminary analysis before choosing a preferred method to use in the analysis (Azous 1991, Taylor, 1993). The methods differed primarily in how the base water level prior to the stormwater influx was estimated. The fluctuation was then calculated as the difference between the maximum and base water levels. The selected method used the midpoint of the sampling interval to estimate the base water level:

$$WLF_i = C_i - 0.5(S_i + S_{i-1}) \quad (4)$$

where WLF_i , C_i , and S_i = the water level fluctuation, crest level, and base level, respectively for sampling occasion i, and S_{i-1} = the base level for occasion i-1.

The water level fluctuation data were used in three ways during the analysis. The data from each sampling occasion were used when evaluating the relationship between precipitation and water level fluctuation. Mean and maximum study period WLF values were used when assessing the effects of land use and wetland characteristics on the wetland hydroperiod.

Seasonal Fluctuation in Wetland Water Levels

Seasonal fluctuation in wetland water levels is probably the most important factor governing wetland development and functioning in the Pacific Northwest (Wald and Schaefer, 1986). A quantitative measure of seasonal WLF was developed based on an examination of the hydroperiod plots for the study wetlands.

May and October are months when the water level changes dramatically in those sites that undergo large seasonal fluctuations. Noting this, the dry season water level was estimated as the mean of staff gage measurements collected during the months June through September. Similarly, the wet season water level was estimated as the mean of staff gage measurements collected between November and May. Approximately equal sample sizes were used to calculate each of the seasonal mean water levels. The mean seasonal difference in water levels was calculated as the difference in these seasonal mean water levels. The data from the early study period (April 1988 - April 1991) were used to calculate seasonal WLFs.

A second measure of the seasonal WLF is the range of water levels observed. The water depth range was calculated as the difference between the study period maximum and minimum water levels. This measure, used with the mean seasonal water level difference described above, provided a picture of the wetland hydroperiod suitable for analysis, because both typical and extreme events were addressed.

Length of Summer Dry Period

The length of the summer dry period (as defined by the absence of surface water) was also analyzed; however, this dry period estimate was subject to the following limitations: (1) Estimating the length of the dry period was affected by the flow characteristics and topography within the wetland; that in turn determined which areas dry first. Because gages were placed in the wetland areas thought to be the last to dry out during the summer, a water level of 3 cm or less constituted “dry” in this analysis. (2) The exact length of the dry period was uncertain, because of the frequency of site visits. The approximate monthly sampling interval during the summer months did not allow for the determination of the date the water level reached “zero.” To compensate for this uncertainty, the transition from “wet” to “dry” (or vice versa) was assumed to occur at the midpoint of the sampling interval.

RESULTS AND THE CONCEPTUAL MODEL

Descriptive results of the morphologic and hydrologic analysis of the study wetlands are shown in Table 1-3. The various water level fluctuation patterns observed in the wetlands and a conceptual model relating wetland and watershed characteristics to wetland hydroperiod are also presented below.

Table 1-3. Wetland and watershed morphologic and hydrologic characteristics.

Wetland	Outlet Condition	Outlet Constriction	WLF Type	Dry in Summer?	System Type	% TIA 1989	% TIA 1995
AL3	None	high	FL	Y	OW/D	4	4
B3I	Culvert	high	SH	N	FT	55	55
BBC24	Beaver dam	low	SL	N	OW	3	11
ELS39	Culvert	high	FH	Y	OW	25	25
ELS61	Stream	low	FL	N	OW	5	11
ELW1	Lake	low	SH	N	FT	20	20
FC1	Beaver dam	moderate	S/FH	N	FT	31	31
HC13	Beaver dam	high	FL	N	OW	4	4
JC28	Stream	low	SL	Y	FT	20	21
LCR93	None	high	FH	Y	FT	6	6
LPS9	Drain inlet	high	FH	Y	FT	22	22
MGR36	Stream	low	SL	N	FT	3	3
NFIC12	None	high	FL	Y	OW/D	2	40
PC12	Beaver dam	high	FL	N	OW	5	7
RR5	Beaver dam	low	FL	N	OW	3	3
SR24	Road	low	FL	N	OW	2	2
SC4	Culvert	low	SL	Y	FT	12	12
SC84	Stream	low	FL	Y	OW	19	17
TC13	Drain inlet	moderate	FL	Y	OW	2	2

WATER LEVEL FLUCTUATION PATTERNS

Based on the water level fluctuation analysis, wetlands were classified into four distinguishable types of hydroperiods (Figure 1-2): (1) stable base water level with low event fluctuations (SL), (2) stable base water level with high event fluctuations (SH), (3) fluctuating base water level with low event fluctuations (FL), and (4) fluctuating base water level with high event fluctuations (FH). The four patterns were defined quantitatively using a threshold of 20 cm. Wetlands with a base water level range less than or greater than 20 cm were considered stable or fluctuating, respectively. Similarly, wetlands with event fluctuations less than or greater than 20 cm were considered low or high, respectively. Figure 1-2 shows the WLF pattern for the 19 study wetlands.

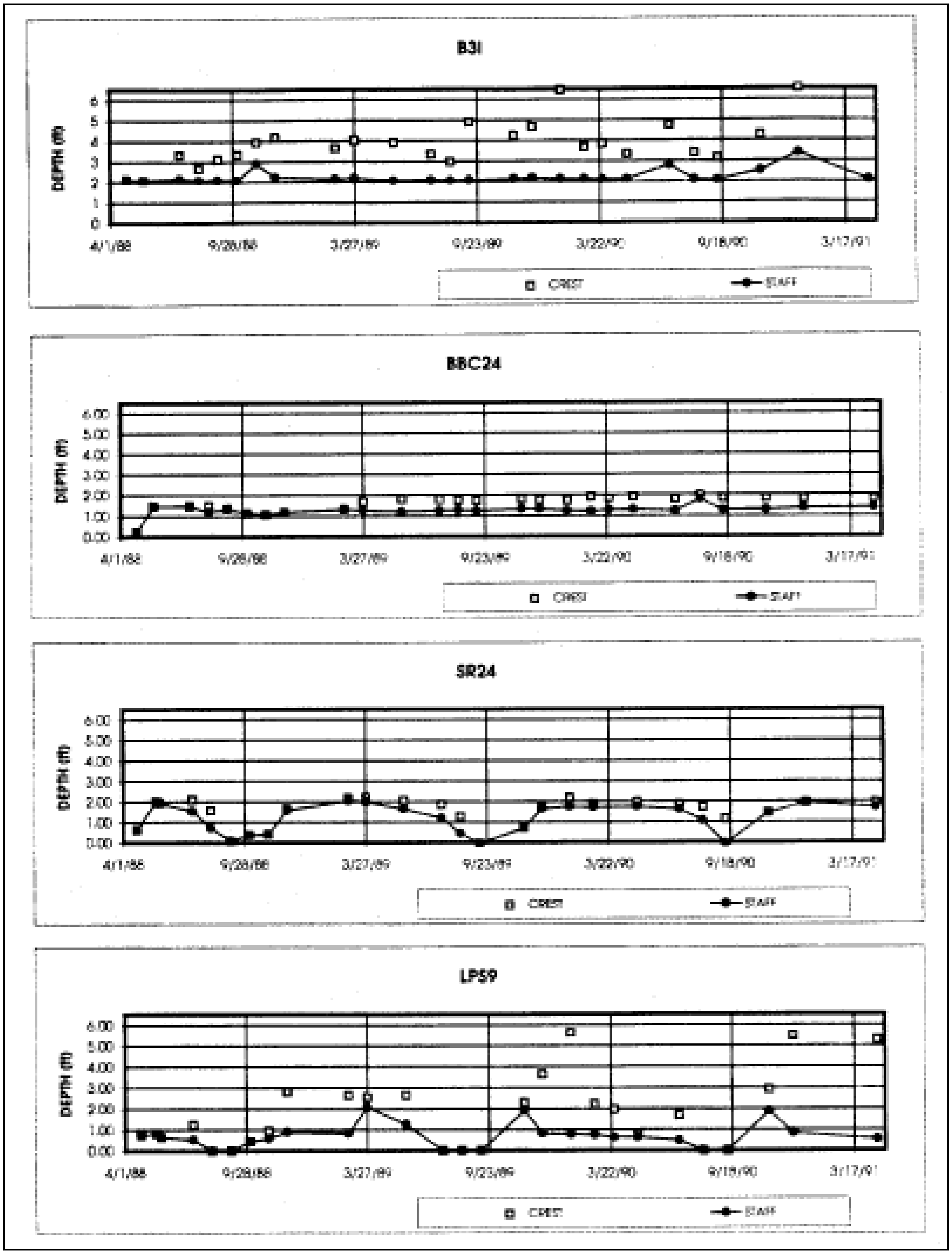

Figure 1-2. Four Water Level Fluctuation Patterns

CONCEPTUAL MODEL OF INFLUENCES ON WETLAND HYDROPERIOD

A conceptual model was developed by Taylor (1993) to characterize the relationships between watershed and wetland morphological characteristics and wetland hydroperiod (Figure 1-3). This model was used as a basis to examine, through application of a multivariate regression model, which wetland and watershed hydrologic processes, and factors governing these processes, had the greatest influence on wetland hydroperiod. Results from this analysis are presented in Section 3 of this report.

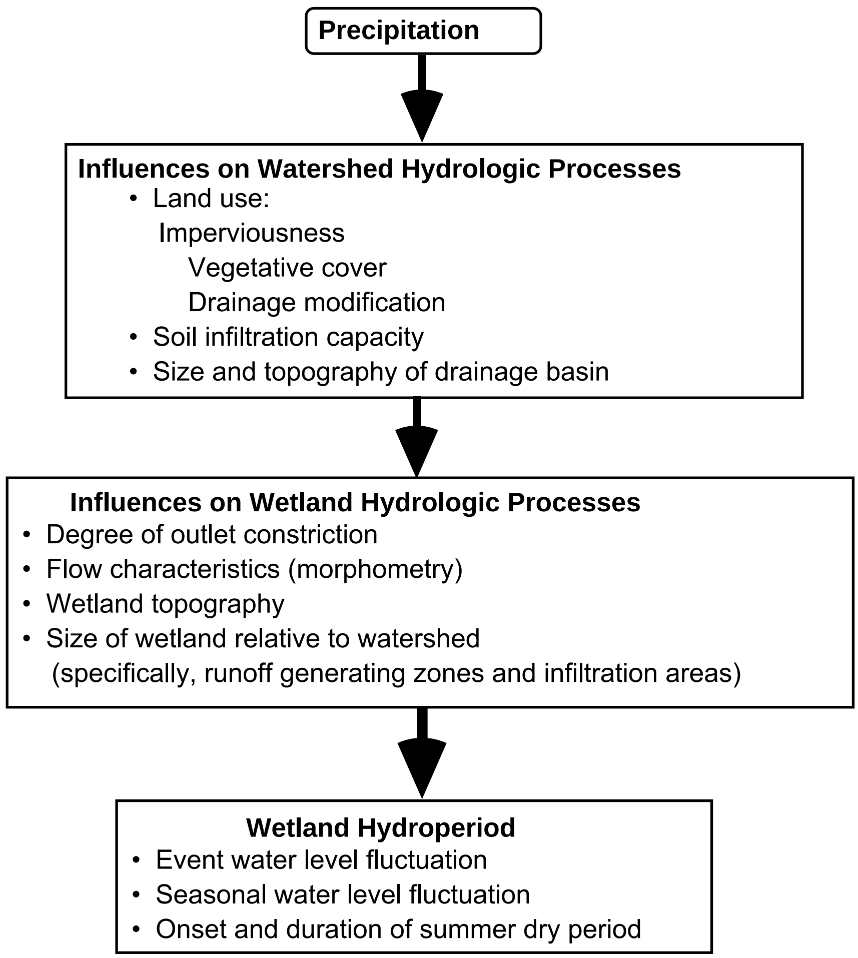

Figure 1-3. Conceptual model of influences on wetland hydroperiod.

CONCLUSIONS

There are many descriptive measures of the morphologic and hydrologic characteristics of freshwater wetlands in the central Puget Sound basin. This paper summarized those that were examined and utilized by the PSWSMRP. The physical shape or type of wetland (e.g., open water, flow-through), the wetland's position within the landscape (particularly as related to the wetland-to-watershed ratio), and the degree of outlet constriction were presented as key wetland characteristics affecting hydroperiods. The imperviousness, land cover, and soils of the watershed were also found to be important characteristics affecting surface runoff and wetland hydrology.

The quantity of stormwater entering many wetlands in the central Puget Sound region has changed as a result of rapid development in urbanizing areas. These changes may affect the functions and values of wetlands by impacting the hydrology, which in turn may affect the plant and animal communities. If the relationships between watershed and wetland changes and their impacts on wetland hydroperiod can be characterized and documented, it may be possible to mitigate these effects through improved watershed controls or development regulations.

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CHAPTER 2 WATER QUALITY AND SOILS

by Richard R. Horner, Sarah S. Cooke, Lorin E. Reinelt, Kenneth A. Ludwa and Nancy T. Chin

INTRODUCTION

This chapter emphasizes water and soil quality in wetlands without significant urbanization in their watersheds. Like other chapters in this section, its purpose is to characterize particular elements of Puget Sound Basin freshwater wetland ecology in a state relatively unaffected by human activity. The wetlands profiled in this group were those with < 4% impervious surface and ≥ 40% forested area in their watersheds. It is recognized that human influence is not entirely absent in these cases, but truly pristine examples do not exist in the lowlands of the Puget Sound Basin. While there are palustrine wetlands in the Pacific Northwest that are not directly affected by urbanization, it is difficult to locate wetlands that are completely unaffected by humans. Indeed, even where there is no human activity in a wetland's watershed, atmospheric pollutants from distant sources could still reach these "pristine" wetlands through rainfall. The wetlands considered here are regarded as representative of the closest to a natural state attainable in the ecoregion. Chapter 9 concentrates on water and soil quality in wetlands with watersheds that are moderately and highly urbanized, as well as those with watersheds that had new development during the years of the study.

It is important to reiterate that the research program concentrated on palustrine wetlands of the general type most prevalent in the lower elevations of the central Puget Sound Basin. The results and conclusions presented here are probably applicable to similar wetlands somewhat to the north and south of the study area, but may not be representative of higher, drier, or more specialized systems, like true bogs and "poor" (low nutrition) fens.

WATER QUALITY

Collection and Methods

Collection of samples for water quality analysis was performed in 1988-1990, 1993, and 1995. Sampling was concentrated during the wet and dry seasons, with fewer samples taken in the transition seasons between those periods. The reason for this scheduling was to concentrate effort when the most pollutants enter wetlands, during the runoff season, and when the decrease in surface water due to relatively low inflow and high evapotranspiration is expected to concentrate pollutants most.

In the last four of the five years' samples were collected in 19 wetlands on the following schedule: November 1-March 31--4 samples, April 1-May 31--1 sample, June 1-August 31--2 samples, and September 1-October 31--1 sample. Sampling occurred at about the same times each year in order to get a consistent view of seasonal water quality variation. The same general pattern was observed in 1988; but there were only 14 wetlands in the program at that time, sampling did not begin until May, and a total of seven instead of eight samples was taken. Some of the wetlands, in most years nine of the 19, had no surface water for varying lengths of time in the late spring, summer, and/or early fall and could not be sampled during those times.

Samples were taken from the largest open water pool in each wetland, if there was one. If not, they were collected near the outlet if surface water existed there or, otherwise downstream of the inlet. The standard grab method was generally used to collect the samples manually. A hand-pump-operated device (Horner and Raedeke 1989) was employed to take samples intended for dissolved oxygen analysis and in cases where shallow water prevented conventional grab sampling without entraining material from the bottom.

Temperature and pH were measured in the field, temperature either by mercury thermometer or electronic meter. The pH was determined with the electronic meter, in latter years a Beckman Model φ 11 instrument. Dissolved oxygen samples were stabilized in the field and transported on ice, along with samples for other analyses, to one of several laboratories used in the different years.

Water quality analyses varied somewhat from the beginning to the end of the program. Some analyses that did not produce much usable information in the early years were dropped. Analyses that were performed in all years and are the focus of this chapter are:

Temperature	Soluble reactive phosphorus (SRP)
pH	Total phosphorus (TP)
Dissolved oxygen (DO)	Fecal coliforms (FC)
Conductivity (Cond)	Total lead (Pb)
Total suspended solids (TSS)	Total copper (Cu)
Ammonia-nitrogen (NH3-N)	Total zinc (Zn)
Nitrate + nitrite-nitrogen (NO3+NO2-N)	

Among the analyses deleted after the early years were dissolved metals, which were usually below detection limits. It is probable that the use of exceptional methods would detect these constituents, but doing so was outside the objectives of this research. Enterococcus was dropped as a bacteriological measure because it did not yield the hoped for reduced variability often prevalent with fecal coliforms, and was never widely adopted as a standard analyte as had been anticipated 10 years ago. Oil and grease and total petroleum hydrocarbons were measured in a relatively small number of samples but were always present in the wetland water column in very small concentrations, with the exception of an isolated incident when an oil spill was suspected. In the final two years of sampling data became available on a number of metals in addition to the three of most interest since they were run routinely on the inductively coupled plasma-mass spectrometer (ICP-MS) used by the laboratory handling those samples.

Horner and Ludwa (1993) prepared a monitoring and quality assurance/quality control (QA/QC) plan that specifies in detail the sampling and analytical methods and QA/QC provisions for the last two years of the program, which were typical of all years. A report by Reinelt and Horner (1990) is the best source of detail on methods for the initial years. Water quality methods and results were also reported by King County Resource Planning Section (1988); Azous (1991); Reinelt and Horner (1991); Platin (1994); Ludwa (1994): Taylor, Ludwa, and Horner(1995); and Chin (1996).

Research Findings: A Portrait of Puget Sound Basin Wetland Water Quality

The main objective of this section is to develop a water quality profile of the least developed wetlands in the data set as presumably representative of the “best attainable” condition in the Puget Sound Basin lowlands. In developing this profile companion data are also presented for more urbanized cases, in part to allow some comparisons now and also for more extensive discussion of those cases in Chapter 9. Later in this chapter wetlands in the data set are classified according to morphological characteristics and again compared. These comparisons are performed with the use of basic summary statistics (primarily, means, standard deviations, and medians). For the most part, tests for statistical significance of differences and analyses of variance were not performed, because of lack of replication of conditions with any exactness, large natural variability, and relatively small sample sizes under any given set of conditions.

Table 2-1 gives a statistical summary of the water quality data gathered over the full project from wetlands whose watersheds did not experience significant urbanization change during that period (control wetlands) grouped by urbanization status. Chapter 9 takes up wetlands with watersheds that did change. Nonurban watersheds (N) were classed as those with both < 4% impervious land cover and ≥ 40% forest; highly urbanized watersheds (H) were considered to be those having both ≥ 20% impervious and ≤ 7% forest. Those not fitting either of the other categories were classified as moderately urbanized watersheds (M). Valentine (1994) developed this classification scheme for analysis of the probable origin of soil metals, and it is maintained here for water quality as well for consistency. Characteristics of the individual watersheds can be found in Section 1, Table 1. Indeterminate statistics (< or > a given value) are the result of some measurements being below detection or, in the case of FC, bacterial colonies too numerous to count in the dilutions analyzed in some very concentrated samples.

Examination of Table 2-1 reveals several general points about wetland water quality. First, excepting pH, concentrations were very variable, as indicated by the relatively high coefficients of variation (CV). The principal sources of water quality variability are examined later in the chapter. Fecal coliform was the most variable of the analytes overall, followed by TSS and NH3-N. Other than for pH, DO, and conductivity, medians were usually lower than arithmetic means, signifying the influence on means, but not on medians, exerted by a relatively few high values. This trend is consistent with a log-normal probability distribution of values, a distribution frequently observed in environmental data (Gilbert 1987).

Nonurban wetlands are the main focus of this chapter, and Chapter 9 further discusses the other categories. With a cursory comparison of the Table 2-1 medians, it can be seen that pH rose slightly and DO marginally declined with increasing urbanization. Conductivity and NH3-N increased substantially from nonurban to moderately urban wetlands but actually were a bit lower in highly urbanized cases. NO3+NO2-N and TP increased from N to M status but not further with H status. Cu showed little difference among categories, but many values were below detection. The remaining variables (TSS, SRP, FC, Pb, and Zn) all increased with each step up in urbanization level.

Table 2-1. Water quality statistics for wetlands not experiencing significant urbanization change (1988-1995).

Status	Statistic	pH		DO	Cond.		TSS		NH3-N		NO3+NO2-N		SRP	TP		FC		Cu		Pb		Zn
				(mg/L)	(µS/cm)		(mg/L)		(µg/L)		(µg/L)		(µg/L)	(µg/L)		(CFU/100 mL)		(µg/L)		(µg/L)		(µg/L)
N	Mean	6.38		5.7	72.5	<	4.6	<	59.9	<	368.2	<	17.6	52.3	>	271.3	<	3.3	<	2.7	<	8.4
	Maximum	7.65		11.3	230.0		73.0		1373.0		3200.0		414.0	850.0		6240.0		15.0		21.0		49.0
	Std. Dev.	0.53		2.6	63.8	>	8.5	>	129.3	>	484.6	>	47.6	86.6	>	1000.4	>	2.7	>	2.8	>	8.3
	CV	8%		45%	88%	>	185%	>	216%	>	132%	>	271%	166%	>	369%	>	80%	>	105%	>	99%
	Median	6.36		5.9	46.0		2.0		21.0		111.5		6.0	29.0		9.0	<	5.0		1.0		5.0
	n	162		205	190		204		205		206		200	206		206		93		136		136.0
M	Mean	6.54	<	5.5	142.4	<	9.2	<	125.7	<	598.2	<	31.5	92.5	>	2664.8	<	3.7	<	3.4	<	9.8
	Maximum	7.88		14.8	275.0		180.0		2270.0		7210.0		280.0	780.0		359550.0		7.0		13.0		33.0
	Std. Dev.	0.82	>	3.6	72.8	>	21.6	>	266.8	>	847.2	>	37.9	91.8	>	27341.7	>	1.9	>	2.7	>	7.2
	CV	13%	>	66%	51%	>	235%	>	212%	>	142%	>	120%	99%	>	1026%	>	51%	>	79%	>	73%
	Median	6.72		5.1	160.0		2.8		43.0		304.0		16.0	70.0		46.0	<	5.0		3.0		8.0
	n	132		173	161		175		177		177		172	177		173		78		122		122.0
H	Mean	6.73	<	5.4	150.9	<	9.2	<	68.3	<	395.4		31.2	109.5	>	968.6	<	4.1	<	4.5	<	20.2
	Maximum	7.51		10.5	271.0		87.0		516.8		1100.0		79.0	1940.0		38000.0		12.0		22.0		73.0
	Std. Dev.	0.57	>	2.9	85.5	>	15.1	>	104.4	>	239.4		15.7	233.5	>	4752.8	>	2.5	>	4.0	>	16.7
	CV	9%	>	53%	57%	>	164%	>	153%	>	61%		50%	213%	>	491%	>	62%	>	89%	>	83%
	Median	6.88		6.3	132.2		4.0		32.0		376.0		28.2	69.0		61.0	<	5.0		5.0		20.0
	n	52		67	61		66		67		67		65	67		66		29		44		44.0

N = wetlands with nonurban watersheds; M = wetlands with moderately urbanized watersheds;
H = wetlands with highly urbanized watersheds; OW = open water wetland; FT = flow-through wetland

A water quality portrait of Puget Sound Basin lowland palustrine wetlands relatively unaffected by humans, then, shows slightly acidic (median pH = 6.4) systems with DO often well below saturation, and in fact sometimes quite low (< 4 mg/L). Dissolved substances are relatively low (most conductivity readings < 50 µS/cm) but somewhat variable. Suspended solids are routinely low but quite variable, reflecting the strong influence of storm runoff events on TSS. Median total dissolved nitrogen concentrations (the sum of ammonia, nitrate, and nitrite) are more than 20 times as high as dissolved phosphorus, suggesting general limitation of plant and algal growth by P. Some of the fairly abundant TP would become available over time to support photosynthesis, but probably not enough to modify the general picture. The low median fecal coliform indicates that most readings are very low (< 10 CFU/100 mL), but a small number is so high that the mean is 30 times the median. Both mean and median heavy metals concentrations are in the low parts per billion range, with standard deviations just about identical to the means.

Wetland Water Quality in Context

To proceed with a descriptive picture of regional wetland water quality, it is useful to provide some context for the quantitative information. This portion of the chapter discusses the statistical data with respect to informal criteria for separating the data into groups that can be associated with various factors that may influence the magnitudes. The account also gives a sense of how water quality compares in regional wetlands versus streams.

Reinelt and Horner (1990) first presented the informal criteria based on several considerations; they were slightly modified for this paper. Some are regulatory standards applied to other water body types (water quality standards have not yet been adopted for wetlands in Washington). Others have generally recognized biological relevance, but some are simply arbitrary breakpoints in the data distributions. In all cases professional judgment was applied in adopting a numerical informal criterion.

Table 2-2 gives the distribution of wetlands, using median values, among the three urbanization categories relative to the informal criteria. It also repeats the medians and means for each category from Table 2-1.

Some water quality variables did not appear to depend on urbanization. One site in each category had median pH < 6, apparently as a consequence of some presence of peat in soils and peat-forming vegetation. Each group also had DO distributed among the three criteria ranges. As discussed later, it seems that DO depends more heavily on wetland morphology than on urbanization.

Several variables exhibited rising medians with urbanization; but when viewed in terms of the criteria, low concentrations predominated, suggesting relatively light pollutant loading from stormwater runoff. Most NH3-N median values were in the lowest range in all categories. Wetlands produce ammonia in decomposing the abundant organic matter internally produced (Mitsch and Gosselink 1993); and, absent an elevated source, concentrations would not necessarily be expected to follow urbanization. Most NO3+NO2-N medians were also in the lowest range in the N and M wetlands but not in the most highly urbanized. For zinc, the most frequently detected metal, no median in any urbanization class approached the chronic criterion for the protection of aquatic life. In fact, the chronic criterion was violated in only one of these wetlands, a highly urbanized one, in individual samples during the entire program. Although not shown in the table, the same general situation prevailed for copper but not for lead, which has a very low chronic criterion in these generally soft waters (3.2 µg/L). As can be seen in Table 2-1, H wetlands had Pb medians above that concentration, and M wetlands fell close to it.

TSS, conductivity, TP, and fecal coliforms exhibited a general tendency toward more sites in the higher criteria ranges with increasing urbanization. Still, TSS medians were very low. A total phosphorus concentration > 20 µg/L is often recognized as one sign that a lake is eutrophic, and > 50 µg/L as an indication of a hypertrophic state (Welch 1980). No wetland had a median below 20 µg/L, and the majority of M and H wetlands fell above 50 µg/L. Wetlands are recognized as systems more prone to eutrophication than lakes for a number of reasons (e. g., rapid nutrient cycling, often having the entire water column in the photic zone). Even those subject to little or no urbanization appear to have a rather high trophic state, and more urbanized systems are even higher. However, since wetlands flush more rapidly than lakes, these elevated TP concentrations may be a lesser concern in wetlands than they would be in lakes. All but three wetlands would meet the 50 CFU/100 mL fecal coliform standard that applies to lakes and the highest class streams in Washington on the basis of their means. Two moderately and one highly urbanized site could not meet even the least stringent standard. Of course, a number of individual values were far higher.

For the least urbanized wetlands, the following general statements can be made to characterize the water quality of Puget Sound Basin lowland palustrine wetlands in a fairly natural state:

- These wetlands are highly likely (83% of cases observed) to have median conductivity < 100 µS/cm, NH3-N < 50 µg/L, TP in the range 20-50 µg/L, fecal coliforms < 50 CFU/100 mL, and total Zn < 10 µg/L.

- These wetlands are also likely (68% of cases observed) to have median TSS in the range 2-5 mg/L and NO3+NO2-N < 100 µg/L.
- The pH and DO in these wetlands are unpredictable from consideration of urbanization status alone, being dependent on other factors.

Table 2-3 statistically summarizes water quality data from 50 locations on western King County streams collected by the Municipality of Metropolitan Seattle during 1990-1993. These data represent grab samples taken on a regular schedule, by chance most often under baseflow conditions. In these ways they are comparable to the wetland data produced by the PSWSMRP. Unlike Tables 2-1 and 2-2, though, Table 2-3 mixes results from streams with very different influences. Nevertheless, it is useful to show how regional wetland and stream water quality compare.

While most wetlands tended strongly to be slightly acidic, and some were rather more so, streams tended just as strongly to be slightly alkaline. This difference is very likely the result of organic acid production by plants that are virtually absent in lotic systems. As expected, flowing streams were observed to be better oxygenated than wetlands, with median DO about twice as high. Streams at the median level were similar to moderately and highly urbanized wetlands in conductivity, but the nonurbanized wetlands had a central tendency below even the minimum measured stream value. TSS median concentrations were generally similar in the two types of water bodies. NH3-N was generally higher in wetlands, reflecting the relatively high production rate of this species accompanying organic matter decomposition. On the other hand, NO3+NO2-N was for the most part lower in the wetlands, perhaps because of slower nitrification in the more oxygen-depleted environment. Median stream TP fell between the levels in the nonurbanized and more highly urbanized wetlands. Stream median fecal coliforms were higher than in any wetland category, but there were no extremely high values such as were measured in the wetlands. This observation suggests that coliform organisms are able to reproduce more successfully in rich, quiescent wetland environments once they enter. All in all, the two sets of results exhibit rough comparability, with most deviations mirroring the physical and biological differences in the two systems.

Table 2-2. Comparison of medians of water quality variables for wetlands not experiencing significant urbanization change (1988-1995) with informal criteria.

Variable	Criterion	Nonurbanized			Moderately Urbanized			Highly Urbanized		
		Median	Mean	%[a]	Median	Mean	%[a]	Median	Mean	%[a]
pH	5-6	6.4	6.4	16.7	6.7	6.5	14.3	6.9	6.7	50.0
	6-7			67.7			57.1			0.0
	7-8			16.7			28.6			50.0
DO	< 4	5.9	5.7	33.3	5.1	< 5.5	57.1	6.3	< 5.4	50.0
(mg/L)	4-6			33.3			14.3			0.0
	> 6			33.3			28.6			50.0
Cond	< 100	46	72	83.3	160	142	28.6	132	151	50.0
(µS/cm)	100-200			16.7			42.8			0.0
	> 200			0.0			28.6			50.0
TSS	<2	2.0	< 4.6	33.3	2.8	< 9.2	14.3	4.0	< 9.2	50.0
(mg/L)	2-5			67.7			71.4			0.0
	> 5			0.0			14.3			50.0
NH_3-N	< 50	21	< 60	83.3	43	< 126	71.4	32	< 68	100.0
(µg/L)	50-100			16.7			14.3			0.0
	>100			0.0			14.3			0.0
NO3+NO2-N	< 100	112	< 368	67.7	304	< 598	57.1	376	< 395	0.0
(µg/L)	100-500			16.7			28.6			50.0
	> 500			16.7			14.3			50.0
TP	< 20	29	52	0.0	70	92	0.0	69	110	0.0
(µg/L)	20-50			83.3			42.8			0.0
	> 50			16.7			57.1			100.0
FC	< 50	9	> 271	83.3	46	> 266	71.4	61	> 969	50.0
(CFU/100 mL	50-100			16.7			0.0			0.0
	> 100			0.0			28.6[b]			50.0[b]
Zn	< 10	5.0	< 8.4	83.3	8.0	< 9.8	85.7	20.0	< 20.2	50.0
(µg/L)	> 10[c]			16.7			14.3			50.0

[a] % of sites with this urbanization status (see Table 1 for definitions) fitting the criterion.
[b] Medians are > 200.
[c] Highest median is 21 µg/L, in comparison to the 59 µg/L chronic criterion for the protection of aquatic life with a water hardness of 50 mg/L as $CaCO_3$.

Seasonal Variation

Wetlands are highly variable systems with annual, seasonal, and diurnal variability in water chemistry. They often have several sources of water supply, each possessing a distinctive chemical blend that varies from year to year. Many water quality parameters exhibited clear seasonal fluctuations in the wetlands studied. DO concentrations were generally higher from mid-November to mid-May than during the remainder of the year

(Reinelt and Horner 1990). This pattern is not surprising considering that most precipitation and runoff and the coolest temperatures in the Pacific Northwest occur during this period, and cooler, more turbulent water absorbs more oxygen.

Conductivity and pH did not exhibit such variation in most wetlands monitored. However, some had higher conductivity from May to November, when wetland water levels drop and dissolved substances become more concentrated (Reinelt and Horner 1990). Many wetlands had substantially higher TSS concentrations during the winter and early spring, the period of greatest runoff and erosion. However, colonial algae can cause high TSS readings in the late summer as well (Reinelt and Horner 1990).

Table 2-3. Distribution of water quality data for baseflow samples from 50 stream sites (Municipality of Metropolitan Seattle 1994).

	Maximum	75th Percentile	50th Percentile	25th Percentile	Minimum
Conductivity (µmho/cm)	30,900	203	130	104	53
Suspended solids (mg/L)	12.6	4.8	3.4	2.8	1.6
Ammonia (µg/L)	190	24	15	13	5
Nitrate+nitrite (µg/L)	3,000	1,100	630	320	73
Temperature (°C)	13.5	11.1	10.6	10	8.0
Dissolved oxygen (mg/L)	11.4	11.0	10.4	9.6	5.8
Turbidity (NTU)	16.5	2.7	1.8	1.4	0.7
Fecal coliform (org/100 mL)	900	220	100	49	7
Enterococcus (org/100 mL)	410	170	53	22	5
pH	8.2	7.6	7.5	7.3	6.9
Total phosphorus (µg/L)	150	66	48	32	13

While many wetlands monitored by the program had lower concentrations of NH3-N, SRP, and TP from November to May, they had higher nutrient concentrations in the other part of year possibly as a result of greater fertilizer applications and lower water levels that concentrate nutrients. NO3+NO2-N values fluctuated greatly in the program wetlands, and tended to vary directly with DO (Reinelt and Horner 1990). This association is another sign that nitrification moderated by the degree of aerobiosis has a strong influence on how much NO3+NO2-N will be found in a wetland water column.

Medians and geometric means of fecal coliform (FC) and enterococcus bacteria were highly variable. Peak counts occurred most frequently in late August and September, and least often from mid-November through February (Reinelt and Horner 1990). The monitoring program found that while most water quality parameters varied seasonally, NH3-N, SRP, TP, FC, and enterococcus were especially changeable (Reinelt and Horner 1990).

Variation with Wetland Morphology

Wetland morphology refers to its form and physical structure and embraces its shape; perimeter length; internal horizontal dimensions; topography (also termed bathymetry), which is the pattern of elevation gradients; water inlet and outlet configurations; and water pooling and flow patterns. These factors establish zonation at early successional stages by determining the extent of inundation from place to place and the hydrodynamic characteristics of flow. From these structural zones stem vegetation composition, distribution, and productivity, and, ultimately, the same features of the animal communities. Of course, these biota in turn influence morphological development over time through detrital and sediment accretion and animal activities like burrowing and dam building by beavers. The various morphological characteristics entirely determine the flood-flow storage and alteration function of wetlands. Along with the friction produced by vegetation, they set the residence time of water within the wetland, which is a key regulator of sediment trapping, nutrient processing, and other water quality functions.

Early work in the program determined that one aspect of morphology in particular, water pooling and flow patterns, had a substantial influence on wetland water quality (Reinelt and Horner 1990, 1991). The wetlands in the study were classified as either open-water (OW) or flow-through (FT) types. The OW systems contain significant pooled areas and possess little or no flow gradient, while the FT wetlands are often channelized and have a clear flow gradient.

Using the first three years of data (Reinelt and Horner 1990), it was found, unsurprisingly, that temperatures ranged higher in wetlands characterized by relatively large open pools, especially from May to September. On an annual basis, the photosynthetic pigments chlorophyll a and phaeophytin a attained higher concentrations in wetlands characterized by large open pools, which have greater light exposure and longer residence times, and ranged much higher than in flow-through wetlands during the growing season. Dissolved oxygen tended to be significantly lower than in flow-through wetlands during these periods.

Table 2-4 summarizes statistics for the wetlands whose watersheds stayed relatively stable during the program broken down by urbanization and morphological status. Comparing open water versus flow-through wetlands in the N and M categories, it can be seen that medians were higher, often substantially so, for flow-through than for open water wetlands in both urbanization categories for pH, DO, Cond, NO3+NO2-N, SRP, FC, and Pb. In addition, the flow-through means were higher in moderately urbanized wetlands for TSS, NH3-N, TP, Cu, and Zn. Over all levels of urbanization flow-through wetland medians were higher for all water quality variables reported in the table.

It is clear from these results that flow-through wetlands strongly tend to be less acidic and better oxygenated than open water sites, as would be expected. Humic acid-producing vegetation thrives in an environment with low inflow, and attendant nutrient income. In these ponded systems oxygen renewal from the atmosphere is not as efficient as in flowing water, and they are warmer and hence have lower oxygen solubility. Also, more primary production and more oxygen-consuming organic decomposition occurs in the relatively long period of water residence. It is also clear that flow-through wetlands generally have higher pollutant concentrations, probably due to the greater loading of pollutants by the flow and reduced pollutant removal from the water column with the shorter hydraulic residence times.

Concentrations of NO3+NO2-N exhibited one of the greatest disparities between open water and flow-through wetlands. In addition to greater loading introduced by the flow, this phenomenon is probably partially due to higher oxygen levels in flow-through cases, which promote nitrification that converts ammonia to nitrite and then nitrate forms. In fact, ammonia differed between the two types of morphology much less than did NO3+NO2-N, suggesting that ammonia discharged to wetlands may be more effectively nitrified in flowing systems. Of course, these systems also support less decomposition by microorganisms and, thus, likely produce less ammonia internally than do open water wetlands.

Table 2-4. Water Quality Statistics for Wetlands Not Experiencing Significant Urbanization Change (1988-1995) Grouped by Urbanization and Morphological Status.

Status	Year	Stat.[a]	pH		DO	Cond.		TSS		NH_3-N		NO_3+NO_2-N		SRP		Tot P		FC		Tot Cu		Tot Pb		Tot Zn
					(mg/L)	(µS/cm)		(mg/L)		(µg/L)		(µg/L)		(µg/L)		(µg/L)		(CFU/100mL)		(µg/L)		(µg/L)		(µg/L)
N/OW	1995	Mean	33498.54		5.3	5	<	5.1	<	83	<	260	<	23		63	>	144	<	2.4	<	2.4	<	7.0
N/OW	1995	St. Dev.	935.94		2.6	3	>	7.8	>	154	>	487	>	59		103	>	702	>	3.7	>	3.1	>	8.4
N/OW	199500%	CV	3%		50%	50%	>	152%	>	186%	>	187%	>	258%		163%	>	488%	>	158%	>	131%	>	120%
N/OW	1995	Median	33092.00		5.3	5		2.4		35		49		6		31		5		0.9		1.0		2.5
N/OW	1995	n	136		135	135		109		104		107		121		136		115		23		21		65
N/FT		Mean	33443.64		6.6	7	<	6.6	<	65		672	<	11		31	<	327	<	1.6	<	5.3	<	4.4
N/FT		St. Dev.	919.99		2.3	2	>	11.7	>	133		417	>	19		27	>	855	>	2.6	>	0.0	>	5.8
N/FT		CV	3%		35%	35%	>	179%	>	206%		62%	>	165%		88%	>	262%	>	163%	>	0%	>	133%
N/FT		Median	33073.00		6.5	7		2.5		32		613		7		24		56		0.7		1.2		2.5
N/FT		n	70		70	70		54		46		70		62		70		65		8		6		29
M/OW	1995	Mean	33703.88	<	3.7	4	<	6.8	<	158	<	291	<	23		69	>	233	<	2.4	<	2.5	<	8.7
M/OW	1995	St. Dev.	851.14	>	2.7	3	>	14.7	>	371	>	401	>	41		69	>	769	>	2.0	>	2.2	>	6.7
M/OW	199500%	CV	3%	>	73%	74%	>	216%	>	235%	>	138%	>	178%		100%	>	331%	>	82%	>	88%	>	77%
M/OW	1995	Median	6.14		3.2	65		2.2		49		99		10		44		17		1.5		1.5		7.5
M/OW	1995	n	77		74	75		62		62		54		73		77		61		22		24		48
M/FT	1995	Mean	33505.61	<	7.0	7	<	12.4	<	127	<	912		39		111	>	4764	<	3.0	<	4.1	<	9.2
M/FT	1995	St. Dev.	918.20	>	3.6	4	>	26.4	>	200	>	989		35		103	>	37867	>	1.9	>	3.5	>	7.3
M/FT	199500%	CV	3%	>	51%	53%	>	214%	>	157%	>	108%		90%		93%	>	795%	>	62%	>	85%	>	79%
M/FT	1995	Median	33126.00		7.7	8		4.7		49		688		29		85		220		2.3		2.2		6.8
M/FT	1995	n	100		96	98		95		96		98		97		100		90		24		40		52
H/FT	1995	Mean	33436.94	<	5.4	5	<	10.5	<	81	<	401		31		110	>	1021	<	3.9	<	5.0	<	21.3
H/FT	1995	St. Dev.	914.79	>	2.8	3	>	15.8	>	111	>	236		16		233	>	4901.0	>	3.4	>	4.7	>	17.8
H/FT	199500%	CV	3%	>	52%	53%	>	150%	>	138%	>	59%		50%		213%	>	480%	>	88%	>	95%	>	84%
H/FT	1995	Median	33055.00		6.4	6		4.8		41		377		28		69		69		2.6		3.9		20.0
H/FT	1995	n	67		66	67		57		55		66		65		67		62		12		28		37
All		Median	6.16		4.6	39		2.3		39		70		7		35		8		1.1		1.4		5.2
OW		n	213		209	210		171		166		161		194		213		176		45		45		113
All		Median	7.00		6.9	183		4.2		42		510		21		60		110		2.3		3.0		6.9
FT		n	237		232	235		206		197		234		224	0	237		217		44		74		118

[a] See Table 1 for definitions of Status and Stat. abbreviations.

It must be noted that a preponderance of flow-through wetlands are in more urbanized areas, which certainly affects pollutant loading and may affect the strength of conclusions, although probably not the overall trends. It is possible that this skewed distribution is not just a coincidence but reflects the urban situation, in which higher peak runoff flows, wetland filling, and stream channelization favor flow-through over open water wetland conditions.

SOILS

Collection and Methods

Soil samples were collected once from each wetland during the months July-September in 1988-1990, 1993, and 1995. Soil sampling areas were selected 3 meters to the side of vegetation transect lines at every point where the soil type appeared either to be transitional or completely different. Small soil cores or signs of vegetation change were the basis for judgment. Two to five samples, most commonly four, were collected from each wetland. The number had a relationship to the size and zonal complexity of the wetlands. This coverage was considered to be adequate because a synoptic study of 73 urban and rural wetlands early in the program found that there were no significant differences among wetland zones (e.g., open pool, inlet, scrub-shrub, and emergent) with respect to soil texture, organic content, pH, phosphorus, and nitrogen (Horner at al. 1988). Because oxidation-reduction potential and one metal were significantly different near inlets as compared to other locations, the inlet zone was emphasized in as one spot in choosing sampling areas, however.

Soil samples were collected with a corer consisting of a 10-cm (4-inch) diameter ABS plastic pipe section ground to a sharp tip. The corer was twisted into the soil with a wooden rod inserted horizontally through two holes near the top. Coring depth was 15 cm (6 inches). Samples were inserted immediately into plastic bags, air was extruded, and the bags were sealed with tape. They were then transported to one of several laboratories used in the different years.

A standard 60-cm (2-ft) deep soil pit was dug at each sampling point not inundated above the surface. The pit was observed and notes were recorded for depth to water table (if within 60 cm of the surface), horizon definition (thickness of each layer and boundary type between), color (using Munsell notations), structure (grade, size, form, consistency, and moisture), and presence of roots and pores.

Soil core samples were analyzed for:

<u>General characteristics</u>	<u>Metals</u>
Particle size distribution (PSD)	Arsenic (As)
% organics as loss on ignition (LOI)	Cadmium (Cd)
pH	Copper (Cu)
Oxidation-reduction potential (redox)	Lead (Pb)
<u>Nutrients</u>	Zinc (Zn)
Total phosphorus (TP)	
Total Kjeldahl nitrogen (TKN)	

A report by King County Resource Planning Section (1988) provides detail on the general analytical methods, as well as the sampling program design. A method for PSD, also termed soil texture, was developed during this program for soils with more than 5% organics, as most wetland soils have. Texture is the measurement of the proportions of the various sizes of mineral particles in a soil, classified from largest to smallest as sand, silt, and clay (gravel, when significant, is also recognized in the texture classification). The analysis of any soil with more than 5 % organic content must include a step that removes the organic material. Failure to remove the organic component may cause clumping of particles and render the results inaccurate. The new PSD method, which is provided in Appendix A of this chapter, is considered to be accurate for soils with up to 25% organics. At higher levels it is not accurate because of sample loss during organic removal preparation, especially in the clay component.

Publications by Cooke, Richter, and Horner (1989); Richter et al. (1991); Cooke (1991); and Cooke and Azous (1993) are additional sources of detail on methods and findings from the initial years. Soils methods and results were also reported by Azous (1991), Cooke (1991), and Platin (1994).

Research Findings: A Portrait of Puget Sound Basin Wetland Soils

General Soil Characteristics and Nutrients

As with water quality, this discussion is conducted mainly with reference to descriptive statistics. Tests for statistical significance of difference and analyses of variance generally were not performed, for the same reasons stated earlier in the chapter. Table 2-5 statistically summarizes the soils data, excluding PSD, for wetlands that did not experience significant urbanization change during the program. Metals are discussed in a later subsection. Nonurban wetlands are the main focus of this chapter, and Chapter 9 further discusses the other categories, as well as wetlands with watersheds that did change.

Like water quality, soil quality exhibited extensive variability. As with most water quality variables, coefficients of variation for the majority of the soil variables were generally in the approximate range 75-150%, although in both soil and water cases some were higher. CVs for pH were considerably lower than for other analytes for both soils and water, usually about 10 $\pm$ 3%. Again like water quality variables, the quantities in Table 2-5 usually exhibited medians lower than the means, except for redox and sometimes pH. Therefore, most of these data also are far from normally distributed, and are probably log-normal.

Most wetlands had at least some pockets of peat of mainly sedge and grass origins (Cooke 1991), and their soils accordingly tended to be acidic. Among the different groupings of data in Table 2-5, median pH values were $\leq$ 6.1, except for highly urbanized cases, which were both flow-through . Overall, highly urbanized sites had the highest median pH, followed by moderately urbanized and then nonurbanized locations. Flow-through wetlands overall and in the N and M categories had higher median pH than open water types.

These wetlands frequently had anaerobic soils, as indicated by median redox values often < 250 mv, the approximate point at which oxygen is fully depleted. The median for the most highly urbanized case was lower than for the N wetlands, which themselves had lower median redox than the M cases. Open water wetlands overall had higher redox readings than flow-through ones. This result is somewhat surprising, in that open water wetlands are thought to host more oxygen-consuming decomposition and to have oxygen replenished from the atmosphere less efficiently than flow-through cases. It was found in the synoptic study of 73 wetlands at the beginning of the program that open water zones had the lowest redox readings (<100 mv) (Horner at al. 1988). Redox was below the level at which oxygen is generally depleted in the inlet, open water, and emergent zones but not in the scrub-shrub and forested zones. In another contrast with the more recent results, soils in the inlet zones of wetlands in nonurban wetlands had significantly ($P < 0.05$) lower redox than in urban wetlands.

The highest median TP occurred in moderately urbanized wetlands, and the highest TKN in nonurbanized ones. Open water systems were higher than flow-through cases in both nutrients overall, but that result was due to large differences between the two types of morphology in the M wetlands. This tendency was actually reversed in nonurbanized wetlands, which exhibited higher nutrients among the flow-through wetlands in that subset.

The N urbanization category had median organic content over 30%, although with extensive variation. The M and H sites overall had about half to two-thirds of that level. On the whole open water wetlands exceeded flow-through ones in organics, although again this tendency is due to the M wetland data. This finding is as expected, since ponded systems have more primary productivity and capability of settling solids.

Soil texture is important to the nutrition, structure, drainage, and erosion prevention characteristics of a soil. Nutrients are found in a soil attached to organic matter, clay particles, and metal oxides (especially iron oxides). Soils with a high portion of clay, organic material, or both adsorb water and nutrients much more readily than soils low in these components. Fine textured soils have a more compact structure, which may impede aeration of the soil. Clays adsorb water and if positioned lower down in the soil profile, can impede drainage, causing an impervious layer and creating a wetland. Sandy soils have very little cohesion and erode much more easily than silt- or clay-rich soils. One of the influences of urbanization on wetland ecosystems is deposition of sediments from development activities (clearing and grading).

Table 2-5. Soil quality statistics for wetlands not experiencing significant urbanization change (1988-1995) grouped by urbanization and morphological status.

Status [a]	Stat. [a]	pH	Redox.	TP	TKN	Org.	Cu	Pb	Zn	As
			(mV)	(mg/kg)	(mg/kg)	(%)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)
N/OW	Mean	5.43	143	743	5261	45.5	58	65	27	6.5
	Maximum	6.67	649	6579	27369	97.3	2221	418	154	20
	St. Dev.	0.64	322	1087	5983	34.2	300	96	29	5.4
	CV	12%	225%	146%	114%	75%	515%	147%	106%	84%
	Median	5.40	153	251	2866	47.0	16	24	18	4.4
	n	47	46	55	58	58	54	54	54	54
N/FT	Mean	5.73	113	839	3899	36.1	21	33	31	15
	Maximum	7.12	629	8882	14223	97.3	40	322	103	225
	St. Dev.	0.48	296	1698	3297	28.8	13	60	30	42
	CV	8%	262%	202%	85%	80%	61%	184%	97%	284%
	Median	5.70	184	342	3799	34.1	17	18	17	6.6
	n	27	27	27	29	29	27	27	27	27
All N	Mean	5.54	132	775	4807	42.4	46	54	28	9.3
	Maximum	7.12	649	8882	27369	97.3	2221	418	154	225
	Std. Dev.	0.60	311	1310	5261	32.6	245	87	29	25
	CV	11%	236%	169%	109%	77%	536%	159%	103%	267%
	Median	5.60	184	283	2885	35.8	16	20	17	6.0
	n	74	73	82	87	87	81	81	81	81
M/OW	Mean	5.31	317	1114	6354	35.4	12	32	22	6.2
	Maximum	6.72	656	3827	22517	92.3	24	101	92	16
	St. Dev.	0.90	227	1019	5670	29.9	7	30	24	4.4
	CV	17%	72%	91%	89%	85%	60%	95%	107%	70%
	Median	5.11	362	945	5783	23.7	14	21	15	4.8
	n	26	26	25	25	25	28	28	28	28
M/FT	Mean	5.95	165	756	3999	25.7	18	77	60	8.1
	Maximum	6.96	611	2743	27967	92.3	63	530	334	25
	St. Dev.	0.51	279	819	5942	28.1	12	89	62	5.2
	CV	9%	169%	108%	149%	109%	68%	115%	104%	65%
	Median	6.09	226	481	2021	12.6	17	57	48	7.7
	n	45	43	41	43	47	43	43	43	43
All M	Mean	5.71	222	892	4865	29.1	16	59	45	7.3
	Maximum	6.96	656	3827	27967	92.3	63	530	334	25
	Std. Dev.	0.74	269	909	5912	28.9	11	75	53	5.0
	CV	13%	121%	102%	122%	99%	69%	126%	119%	68%
	Median	5.78	306	537	2764	15.1	15	34	33	6.2
	n	71	69	66	68	72	71	71	71	71
H/FT	Mean	5.88	87	654	2703	31.6	31	89	103	13
	Maximum	6.97	640	2995	11282	80.9	63	273	456	40
	St. Dev.	1.01	291	784	2892	26.3	17	68	112	9.4
	CV	17%	335%	120%	107%	83%	56%	76%	109%	74%
	Median	6.46	91	314	2033	22.8	30	64	65	10
	n	32	32	34	36	35	32	32	32	32
All OW	Mean	5.39	206	859	5590	42.4	43	54	25	6.4
	Maximum	6.72	656	6579	27369	97.3	2221	418	154	20
	St. Dev.	0.74	302	1074	5877	33.1	244	81	27	5.1
	CV	14%	147%	125%	105%	78%	572%	151%	107%	80%
	Median	5.35	305	414	3141	34.2	15	24	16	4.5
	n	73	72	80	83	83	82	82	82	82
All FT	Mean	5.87	127	744	3540	30.3	23	69	66	11
	Maximum	7.12	640	8882	27967	97.3	63	530	456	225
	St. Dev.	0.70	286	1102	4449	27.8	15	78	80	22
	CV	12%	226%	148%	126%	92%	67%	113%	123%	199%
	Median	5.92	162	378	2274	16.1	20	46	45	7.70
	n	104	102	102	108	111	102	102	102	102

[a] See Table 1 for definitions of abbreviations.

Table 2-6 presents a comparison of soil textures in 1989 and 1995 for the wetlands that did not experience significant watershed change. As was hypothesized for these cases, little change occurred over these years regardless of urbanization or morphological status. The soils of the majority of these wetlands were dominated by silt-range particles, again irrespective of status. One N/OW and one M/FT site, located in different parts of the study area, had predominately sand. With two exceptions, the wetlands were found to have relatively little clay (≤ 20%). However, a wetland in north King County and one in south King County had about 30% and 50%, respectively. It bears noting that some of the samples contributing to these statistics had > 25% organic content, in the range where the analytical method is less accurate.

Table 2-6. Comparison changes in average particle size distributions from 1989 to 1995 in wetlands that experienced little urbanization change grouped by urbanization and morphological classifications.

Site[a]	Wetland Area (ha)	PSD 1989 (%sand/silt/clay)	PSD 1995 (%sand/silt/clay)
N/OW			
AL3	0.81	26/54/20	No data
HC13	1.62	47/47/6	45/37/18
RR5	10.52	74/15/11	68/21/11
SR24	10.12	1/89/10	6/75/19
N/FT			
LCR93	10.93	No data	30/50/20
MGR36	2.23	13/76/11	20/70/10
M/OW			
ELS39	2.02	35/49/16	15/69/16
SC84	2.83	4/81/15	11/73/16
TC13	2.06	30/41/29	38/32/30
M/FT			
ELW1	3.84	83/13/4	75/18/7
FC1	7.28	13/71/16	10/75/15
SC4	1.62	No data	No data
N/FT			
B3I	1.98	31/62/7	24/61/15
LPS9	7.69	30/16/54	32/20/48

[a]See Table 1 for definitions of abbreviations.

Analysis of the PSD measurements within individual wetlands indicates that PSD often varied substantially across wetlands and showed no trends with the amount of organic matter in the soil or the soil series. No association was seen between the total suspended solids in the surface water and changes in soil texture. However, soils located near the inlets of M and H wetlands were significantly ($P < 0.05$) more likely to have more sand than silt as compared to other locations.

METALS IN SOILS

Cadmium, lead, and zinc in wetland soils were observed to be highly variable from year to year, but copper and arsenic varied less. Overall, there was a declining trend in soil metal content over the years of the study. These results are somewhat surprising since the soil cores were 15 cm deep, representing soil horizons that would be expected to maintain fairly stable metals concentrations from year to year. Figure 2-1 shows that median concentrations of As, Cu, Pb, and Zn for all of the program wetlands generally declined each year. It is possible that metals enter and depart from wetland soils more easily than previously believed, permitting a rapid change in results in response to changes in inputs from the watershed. Declining metals pollution from vehicles and dissipating pollutants from industrial air pollution point sources, such as the closed ASARCO smelter in Tacoma, could explain the general decline of metals since the start of the program.

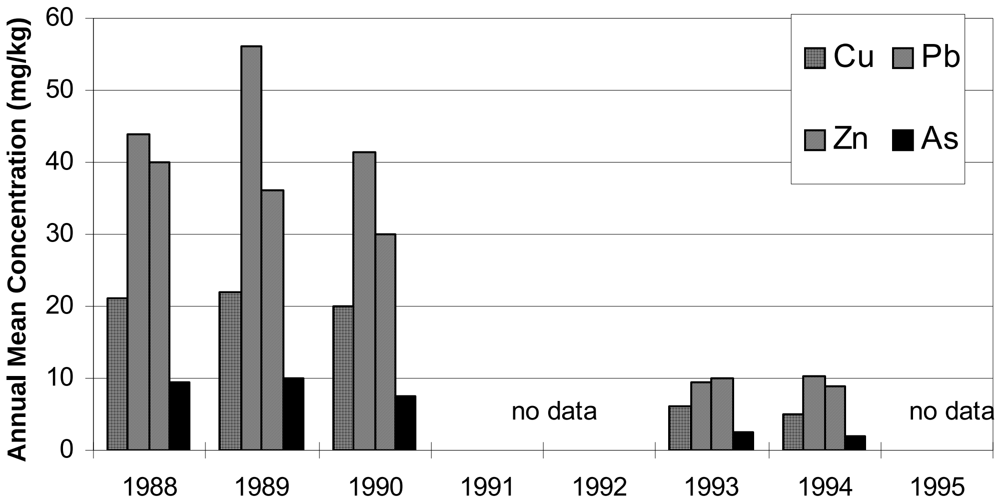

Figure 2-1. Annual mean metal concentration for all wetland samples in each year.

Cadmium was undetectable in the soils of most monitored wetlands, except in three that also had relatively high Pb. This result is consistent with the observation that metals often increase in tandem. Although the program detected substantial increases in Cd, Pb, and Zn at several wetlands between 1989 and 1990 (Richter et al. 1991), it is significant that there are no apparent common characteristics among these wetlands. They represent differing hydrology, ecology, and levels of watershed development.

It can be seen in reviewing the metals data in Table 2-5 that, like water quality and general soil characteristics, soil metals exhibited extensive variability. Again, too, medians were normally considerably less than means.

It is further apparent in Table 2-5 that median metals concentrations increased from nonurban to moderately urbanized and again from there to highly urbanized wetlands, except for a small drop in copper from N to M. Flow-through wetlands overall had higher median concentrations of all metals than did open water ones, although very marginally so for Cu. This tendency was again stronger for the moderately urbanized than the nonurbanized wetlands. For the most part, then, soils exhibited the same trend as water quality, with quantities considered to be pollutants higher in FT than in the OW wetlands. It was thought that water column contaminants might be lower in open water wetlands because of losses to the soil. However, this supposition was not borne out by the soil results. Still, having the two most developed sites in the FT group may be skewing the results.

The Washington Department of Ecology (1991) set metals criteria for freshwater sediments in terms of lowest effect and severe effect thresholds. The criteria are:

	Lowest Effect Threshold (mg/kg dry soil)	Severe Effect Threshold (mg/kg dry soil)
Copper	16	110
Lead	31	250
Zinc	120	820

No mean or median value exceeded the severe effect criteria, and very few individual readings surpassed them at any time during the program. However, lowest effect thresholds were exceeded by some Cu and Pb means and even medians. Many individual readings in wetlands in all urbanization categories were beyond these lower limits.

Even though there is a trend toward increasing soil metals with urbanization, it is a fact that soil in either urban or nonurban wetlands can have elevated metals. These contaminants could be entering wetlands outside of urban areas in a variety of ways. Possibilities include via precipitation and atmospheric dryfall, dumping of metal trash, and leaching from old constructed embankments. Roads and narrow-gage railroad beds were built using mine tailings to serve logging operations in the last century in the vicinity of some of the wetlands. This phenomenon suggests the need for site-specific inquiries into metals pollution in Pacific Northwest palustrine wetlands, rather than reliance on broad patterns.

SUMMARY OF SOILS CHARACTERISTICS OF WETLANDS WITH NONURBANIZED WATERSHEDS

A soils portrait of Puget Sound Basin lowland palustrine wetlands relatively unaffected by humans shows a somewhat acidic condition; pH is very likely to be in the range 5-6. With redox as a basis, soils at many times and places will be anaerobic, but with great variability. Phosphorus is likely to be somewhere in the vicinity of 300 mg/kg, with nitrogen (TKN) approximately an order or magnitude higher. Based on these results, most soil samples from nonurban wetlands can be expected to have > 25% organics, and > 10% is extremely likely. Texture appears to be more a function of local conditions than a function of urbanization, or lack of it.

The metals As, Cu, Pb, and Zn, can range over two orders of magnitude, from a minimum in the low parts per million (mg/kg) region, in the soils of these nonurban wetlands. Most commonly, they appear to have approximately equal amounts of Cu, Pb, and Zn, around 20 mg/kg, and about one-quarter to one-third as much As. This level and the observed variation around it is sufficiently high to exceed lowest effect threshold freshwater sediment criteria for Cu often and for Pb occasionally, but very rarely for Zn.

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CHAPTER 3   CHARACTERIZATION OF PUGET SOUND BASIN PALUSTRINE WETLAND VEGETATION

by Sarah S. Cooke and Amanda L. Azous

INTRODUCTION

Nineteen wetlands in the Puget Sound Basin in King County, Washington were studied for five years between 1988 to 1995. An additional seven wetlands were studied infrequently over the same period. Our study attempted to understand the character and structure of wetland plant communities and, in particular, if and how wetland communities respond to changing land use and hydrology. The vegetation communities of each wetland were sampled and compared with land use and hydrologic conditions in the watershed.

Results from the early years of the study had shown that many plant species were found in only one or a few of the wetlands surveyed, suggesting that, regardless of size, wetland plant communities often have a few relatively uncommon species present (Cooke and Azous 1992). Early work had also shown a hydrologic measure, mean annual water level fluctuation, was negatively correlated with plant richness in wetlands. This chapter re-examines those earlier findings.

Plant richness and composition in wetlands was compared with wetland area, urbanization in the watershed and water level fluctuation. Community structure was examined through analysis of species richness, composition and percent cover. Ordination and classification analyses were used to identify distinct plant communities and examine relationships with the presence, abundance and distribution of invasive species.

Finally, the wetland indicator status of several common plant species, as assigned by Reed (1988, 1993), was examined. Comparisons were made between the Reed indicator status and a status assigned using quantitative data on species occurrence and hydrologic regime collected for our study.

METHODS

The wetlands surveyed were inland palustrine wetlands ranging in elevation from 50 m to 100 m above mean sea level and characterized by a mix of emergent, scrub-shrub, aquatic bed, and forested wetland vegetation classes. Wetlands were selected so that approximately half would be affected by urbanization sometime after the baseline year. Sites that remained unaffected by urbanization were expected to be the controls for those wetlands receiving urbanization treatment. The wetlands were matched, wherever possible, as treatment (new urban disturbance) and control (no new urban disturbance) pairs on the basis of morphological characteristics and vegetation zones (Cooke et al.'s 1989a, b, c).

Unfortunately, not all of the watersheds developed as predicted. Only six watersheds developed beyond 10% of the baseline developed area. This hindered the ability to statistically compare differences in plant community structure due to stormwater and urbanization effects between control and treatment pairs. Instead, differences in plant communities related to stormwater and urbanization effects were identified by correlating conditions found at the wetlands with watershed conditions and analyzing all wetlands as a continuum. Together, over the study period, the wetlands represented a spectrum of watershed development conditions and hydrologic regimes that we were able to analyze and compare with respect to the plant communities observed.

Wetland sizes were estimated through analysis of USGS 7.5 minute series topographic maps and ranged from 0.4 to 12.4 hectares. Geographic Information Analysis (GIS) was also used to delineate land use and impervious areas within the watersheds (Taylor 1993). Land use classifications included agricultural lands, single and multiple family residential housing, commercial and industrial development, transportation corridors, and any other development within a watershed that reduced forest cover.

Plant communities in each wetland were characterized during a two to three week period during the active growing season between July and August, during the years 1988, 1989, 1990, 1993, and 1995. Plant community composition and percentage cover were sampled in permanent plots adjacent to linear transects established across the hydrologic gradients of each wetland. Species cover was recorded using a cover class system based on the Octave Scale (Barbour et al. 1987, Gauch 1982). Detailed protocols for the vegetation field work are documented in Cooke et al. (1989a). The data set also includes seven additional wetlands that were surveyed during the years 1993, 1994 and 1995 as part of several related studies.

Species were identified using Hitchcock et al. (1969) and were verified with specimens from the University of Washington Herbarium. Using the Cowardin classification system (Cowardin et al. 1979), sample plots were assigned a category based on the dominant structure of the vegetation community, such as aquatic bed (PAB), emergent (PEM), scrub-shrub (PSS), forested (PFO), upland, or some transition zone between them (e.g., PEM/PSS). The Cowardin classification system was selected because it is widely used in functional assessments, wetland protection, and mitigation criteria (Washington State Department of Ecology 1993). In some cases zones changed over time and were re-categorized as required. Upland zones were not included in the analyses of richness or disturbance, but were included in all other analyses.

The vegetation survey data were used to calculate the frequency with which plant species were observed among the wetlands surveyed. We also calculated total plant species richness for individual wetlands, and the total and average plant richness found in the different vegetation community zones of each wetland. Species were categorized according to wetland indicator status (Reed 1988, 1993) and included obligate (OBL), facultative wetland (FACW), facultative (FAC) and facultative upland (FACU) species. The indicator status in Reed is assigned based on qualitative expert experience of how frequently a plant species is found growing in wetland conditions (Table 3-1).

Table 3-1. Indicator status categories for wetland plant species.

Code	Designation	Wetlands Probability 1
OBL	Obligate wetland	> 99
FACW	Facultative wetland	67 to 99
FAC	Facultative	34 to 66
FACU	Facultative upland	1 to 33
UPL	Obligate upland	< 1
NI	No indicator status	

1Percent occurrence of plant found in a wetland

Community types were defined and described using ordination (DECORANA) and classification (TWINSPAN) comparisons (Hill 1979 a, b). Plant community data were tabulated in a two-way data matrix (species by cover). The classification method involved grouping similar vegetation units into categories (Cliffors and Williams 1973, Causton 1988). All of the species that composed more than 25 percent cover in the sample stations were included. Ordination was used to display the species data plots in graphical space where like-communities were plotted close together and dissimilar communities were plotted further apart (Hill 1979b, Gauch 1982). The frequency of species and the relative dominance of species were both described by the proportion of vegetation sampling plots in which the species were found.

Hydrologic measurements, including instantaneous water levels from staff gages and peak levels from crest gages, were recorded at least eight times annually while water was present in the wetlands (Reinelt and Horner 1990). Since we did not have a gage at each sample station, the hydrology at each vegetation sample station was calculated based on the elevation of the sample stations in relationship to the water levels measured at the wetland staff and crest gages. This method assumed that water levels were evenly distributed throughout the wetland varying only as elevation varied. In most cases this assumption was sufficiently accurate, however, the wetlands we studied were sometimes more hydrologically complex, so vegetation sample stations were field checked and eliminated if calculated water levels were inaccurate.

Each sample station was assigned an instantaneous water level and a maximum water level. Water level fluctuation (WLF) was computed as the difference between the peak level and the average of the current and previous instantaneous water levels for each four to six week monitoring period. Mean WLF was calculated by averaging all WLFs for a specific season, or the entire year. These data were averaged over the year and each of four seasons; the early growing (EG) (Mar 1-May 31), intermediate growing (IG) (June 1-August 30), senescence (Sept 1-Nov 15), and dormant (Nov 16-Feb 28) seasons.

The hydrologic data were used to compare the results of field measurements with Reed's categorization of wetland indicator plants. A status was assigned to each species based on the hydrologic regime observed at the vegetation sampling stations. If a station was inundated at any time during the year to within 30 cm of the surface of the sample station the station was considered wet and the plant categorized as growing in wet conditions. Water levels to within 30 cm of the soil surface at the station were used in order to account for saturated soil conditions. All occurrences of individual species were evaluated and, based on the proportion in wet stations versus dry, categorized according to indicator status using Table 3-1.

RESULTS

Community Structure and Composition

Two hundred and forty-two plant species were identified in 26 wetlands over the study period (the list of species is provided in Appendix Table 3-1). Most were obligate (OBL) species (28%), followed by FAC (23%), FACU (22%), and FACW (16%) species. The remaining 11% had no assigned indicator status.

Forty-five species (19%) were found in only one (4%) of the wetlands surveyed. Over 38 percent of plant species were found in less than three wetlands (12%). The distribution of plants according to wetland indicator status was similar to the overall distribution. Forty percent of OBL, 35% of FAC, and 39 % of FACU species were also found in three or fewer wetlands. FACW species were generally more widely dispersed among wetlands, with all species observed in at least eight wetlands.

Most of the species observed were shrubs (35%), followed by herbs (25%) and ferns and horsetails (14%). Least commonly found were rushes (2%), sedges (3%), grasses (3%) and trees (13%). All of the exotic plant species identified in the study wetland plots were either herbs, shrubs, or rushes.

Rubus spectablilis, Rubus ursinus and Polystichum munitum were observed in all 26 wetlands, however, Spirea douglasii was considered to be the most dominant species as it occurred in 25 of 26 wetlands and covered greater than 64% of the sample station in more than 21% of the stations in which it was observed. Alnus rubra, Athyrium filix-femina, and Salix scoulerleriana were also found in 25 of 26 wetlands but rarely dominated the sample station. Phalaris arundinaceae, an invasive weed, was considered the second most dominant species, being found in 18 wetlands (69%) and dominating the sample station in 19% of the plots in which it was observed. Other invasive wetland species were Ranunculus repens found in 65% (17) of wetlands, and Juncus effusus, observed in 58% (15) of the wetlands. Lythrum salicaria, an exotic considered highly invasive, fortunately, was found in only one wetland. Table 3-2 shows some of the most common and least common plants we found categorized by occurrence and cover dominance.

Table 3-2. Species occurrence for different categories of plant type and cover dominance.

Cover Dominance Category	High Occurrence (>80% wetlands)	Low Occurrence (<10% wetlands)
Usually dominant. Greater than 64% coverage in more than 19 percent of observations.	Phalaris arundinaceae Spirea douglasii	Juncus supiniformis Menyanthes trifoliata
Dominance in plots varies	Alnus rubra Athyrium filix-femina Kalmia microphylla Lonicera involucrata Polystichum munitum Pteridium aquilinum Ranunculus repens Rhamnus purshiana Rubus laciniatus Rubus spectablilis Rubus ursinus Salix pedicellaris Salix scoulerleriana Salix sitchensis Vaccinium parvifolium	Azola mexicana Brasenia schribneri Eriophorum chamissonis Hippurus vulgaris Hydrocotyl ranunculoides Hydrophyllum tenuipes Nymphaea odorata Polygonum amphibium Potentilla gramineus Rhynchospora alba Sparganium eurycarpum Sagittaria latifolia Scirpus acutus Veronica americana
Always less than 1% coverage	no species	Mimulus guttatus Myosotis laxa Potamogeton diversifolius Ranunculus acris Rorippa curvisiliqua Rumex obtusifolius Trillium ovatum Vaccinium ovatum Vaccinium uliginosum Vicia sativa

Plant Community Zone Characterization

Twenty-four wetland vegetation community zones were encountered in the 26 study wetlands. These include the four Cowardin types PAB, PEM, PSS, PFO (Cowardin et al. 1979), an additional two zones called BOG and UPL (upland transition), and combinations of each. Table 3-3 lists the frequency of occurrence of each zone out of a total 465 vegetation stations sampled over the study. Shrub-scrub and forested wetlands were the most common vegetation zones sampled, 26 and 16 percent of the samples respectively. Emergent communities were 13 percent of the samples, and bogs (Sphagnum moss systems) 4.5 percent. Mixed communities were found in about one third of the stations sampled, though mosaic type communities, which include more than three mixed community types, were fairly rare.

Communities of the different vegetation zones were evaluated with respect to the dominant plants and their associated Reed indicator status. Most species found in PAB zones were obligate (74%) or FACW (17%). Six percent of species were FAC, and no FACU species were observed in PAB zones. As would be expected, PAB zones were dominated by obligate herbs (67%), followed by obligate and FACW shrubs (15%), FACW herbs (3%), and rushes (3%).

Species most frequently observed in PEM communities were obligate herbs (31%) followed by FACW shrubs (8%) and FACU (8%) shrubs. Many more FAC (15%) and FACU (17%) species were present in emergent areas than were observed in aquatic bed areas.

Scrub-shrub zones were more evenly distributed between obligate (21%), FACW (22%), FAC (27%) and FACU (23%) species. PSS zones were comprised of 14% obligate herbs and 15% FACW shrubs. FAC and FACU shrubs were about 11% of the species observed in PSS communities.

Table 3-3. Plant community zone frequency of occurrence (descending order).

Plant Community Zone	Frequency	Plant Community Zone	Frequency
PSS	25.59%	PEM/PFO	1.29%
PFO	16.34%	PAB	1.29%
PEM	13.12%	PAB/PSS	1.08%
PEM/PSS	9.25%	PEM/UPL	0.86%
UPL	7.31%	PEM/BOG	0.22%
PSS/PFO	5.16%	PEM/PFO/UPL	0.22%
BOG	4.52%	PEM/PSS/PFO	0.22%
PFO/UPL	3.87%	PFO/PAB/PEM	0.22%
PAB/PEM	2.58%	PFO/PSS/UPL	0.22%
PSS/UPL	2.58%	PAB/PFO	0.22%
BOG/PSS	1.94%	PAB/PFO/UPL	0.22%
PAB/PEM/PSS	1.51%	UPL/PFO/PEM	0.22%

Shrubs made up about 36% of the species observed in PFO zones, and most of the shrubs observed were FAC (11%) or FACU (19 %) species. Upland tree species comprised about 17% of species observed in forested zones. FACW trees comprised less than 0.5% of the forested species observed. Thirteen percent of species observed in PFO zones were obligates, of which 9% were herbs.

A limited number of bog zones were also sampled. Obligate species made up 50% of the observations in bogs, and were mostly shrubs (21%) and herbs (18%). The remaining species found in bogs were primarily FACW shrubs (21%).

Wetland Plant Community Associations

Wetland vegetation sample plots were classified into eleven community types using TWINSPAN (Hill 1979a), on the basis of species composition and percent cover (Figure 3-1). The communities include the categories and species listed in Table 3-4.

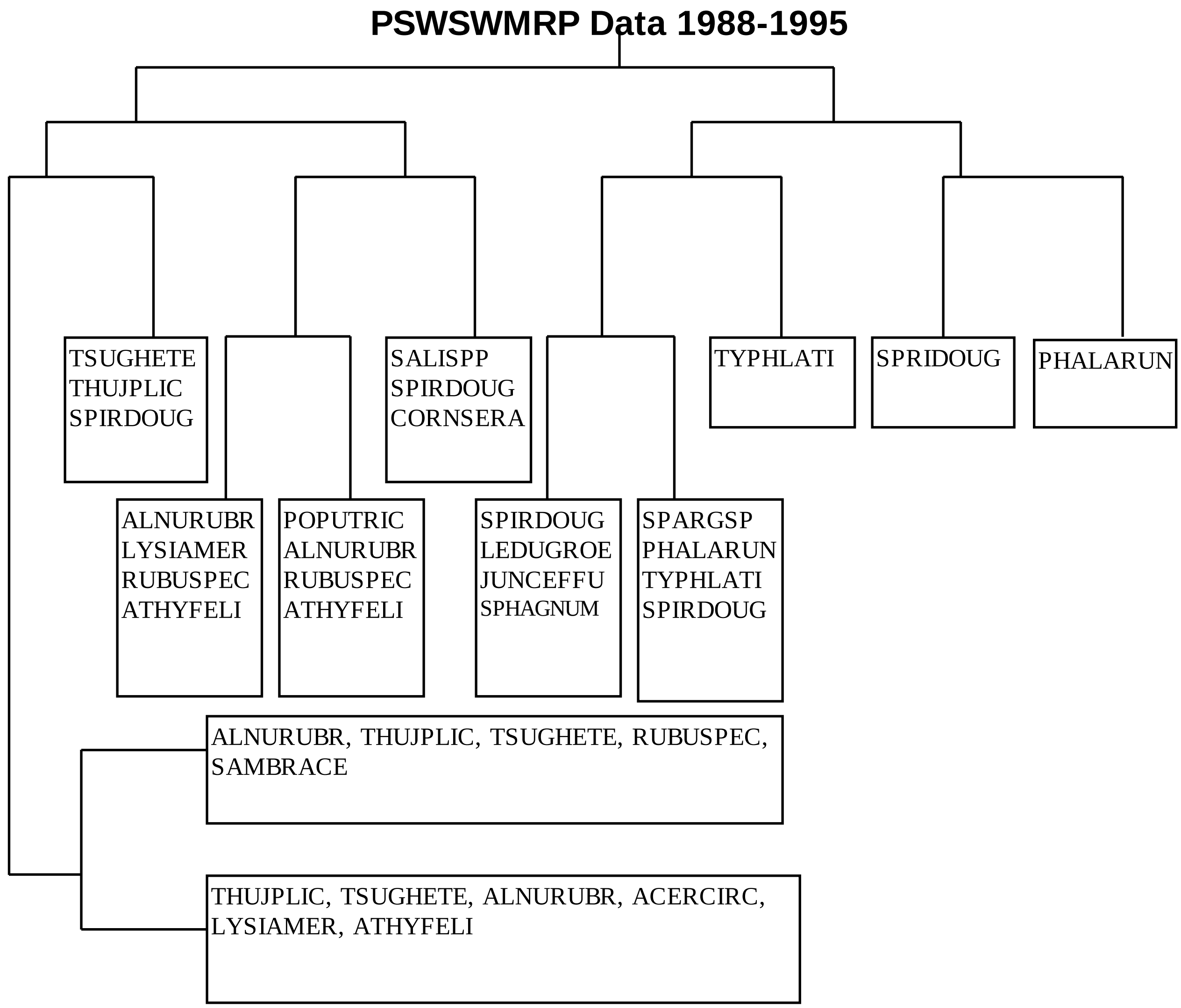

Figure 3-1 Classification of wetland community types present in the PSWSMRP study sites using TWINSPAN (Hill 1979a). (After Houck 1996).

All 24 of the community types listed in Table 3-3 are characterized by the dominant species associations shown in the of the TWINSPAN analysis (Figure 3-1), confirmed in the DECORANA analysis, and described in Table 3-4. These 11 basic community types were repeatedly observed in the study wetlands. Subdominant species changed and a single or a few uncommon species were sometimes present, but the dominant plant associations could be described by one of the eleven types. These communities may be used as a guide for understanding species composition and community structure in wetlands and are relevant to developing reference plant communities for palustrine wetlands in the Puget Basin.

Table 3-4. Wetland community type descriptions (Houck 1996).

Descriptive Name	Cowardin Community Type 1	Community Name 2	Dominant Species
Coniferous forest	PFO PFO/UPL PAB/PFO/UPL UPL	Tsuga-Thuja	Tsuga heterophylla Thuja plicata Spirea douglasii Gaultheria shallon Polystichum munitum
mixed coniferous-deciduous forest with shrub understory	PFO PSS/PFO PEM/PFO PEM/UPL PEM/PFO/UPL	Tsuga-Thuja-wet	Tsuga heterophylla Thuja plicata Acer macrophyllum Acer circinatum Lysichitum americanum
mixed coniferous-deciduous forest with little understory	PSS/UPL	Alnus-Thuja	Alnus rubra Thuja plicata Tsuga heterophylla Rubus spectabilis Sambucus racemosa
deciduous forest	PFO/PSS/UPL PFO/PSS	Populus	Populus balsamifera Alnus rubra Rubus spectabilis Athyrium filix-femina
deciduous forest	PEM/PSS/PFO PFO/PAB/PEM	Alnus	Alnus rubra Rubus spectabilis Cornus sericea Lysichitum americanum Athyrium filix-femina
mixed shrub scrub	PAB/PSS PAB/PFO	Salix-Spirea	Salix spp. Spirea douglasii Cornus sericea Cornus sericea Lonicera involucrata
bog	BOG, BOG/PSS	poor fen-shrub	Rhododendron groenlandicum (Ledum g.), Sphagnum Spirea douglasii
mixed emergent	PAB/PEM/PSS BOG/PEM	poor fen-marsh	Phalaris arundinacea Typha latifolia Rhododendron groenlandicum Sparganium spp Spirea douglasii
emergent	PAB, PAB/PEM	Typha	Typha latifolia Solanum dulcmara Lemna minor
emergent	PEM	Phalaris	Phalaris arundinaceae Solanum dulcmara Urtica dioica
scrub-shrub	PSS	Spirea	Spirea douglasii Salix sitchensis S. Alba

1- Community type used in Table 3-3   2- Community name used in Figure 3-2

The Abundance and Distribution of Invasive Plant Species

Patterns of invasive plant species distribution, dominance and abundance were compared among and within the wetland study sites (Houck 1996). The frequency of invasive species was found to be highly dependent on the conditions present, which varied for different species. For example, Phalaris arundinaceae, Rubus procerus and Solanum dulcamara were more abundant in urbanized watersheds, while Typha latifolia and Juncus effusus were generally more abundant in less urbanized watersheds (Houck 1996). Houck examined water level fluctuation, depth of flooding, and duration of inundation and found that only duration of flooding was associated with the abundance of some invasive species. Typha latifolia and Juncus effusus were generally more abundant in permanently flooded conditions, while Rubus procerus was found in sites where flooding seldom occurred.

Invasive species were most abundant in aquatic bed and emergent marsh communities. The species most frequently observed were Phalaris arundinaceae and Typha latifolia (Figure 3-2). Very few invasive species were found in coniferous forested communities in either the wetland or upland zones.

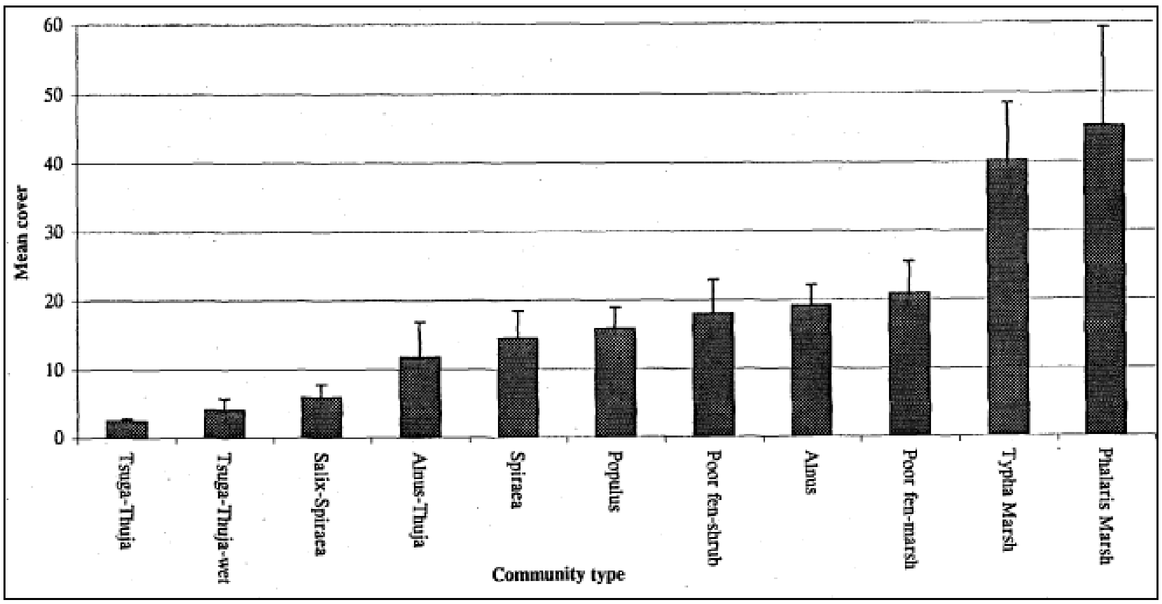

Figure 3-2. Abundance of invasive species within the community types found in the PSWSMRP study sites. Error bars are one standard error (Houck 1996).

Community Richness

At the completion of the study, total plant richness ranged from 35 to 109 species across the wetlands surveyed. Twelve of the wetlands had between 60 and 84 species. Seven had less than 60 species and seven had between 85 and 109 species.

Plant richness varied widely between and among the Cowardin vegetation types. Emergent type richness contained from two species to 33 and averaged 19 species per station overall. Scrub-shrub types ranged from four to 27 species and averaged 11 species per station. Forested types had from five to 31 species and averaged 19. Aquatic bed types had the fewest species, from one to eight and averaging about four among the sample stations. The highest total plant richness was found in wetlands with the largest number of Cowardin community types (Fisher's r to z (Frz), R = 0.41, p = 0.0001).

Plant richness in wetlands and in the Cowardin et al. (1979) vegetation Aquatic bed was compared with wetland area, impervious area in the watershed and water level fluctuation. Total plant richness of a wetland was found to have no significant relationship to wetland area (Figure 3-3), nor did average wetland plant richness within community types, such as PEM and PSS, have any relationship to wetland area.

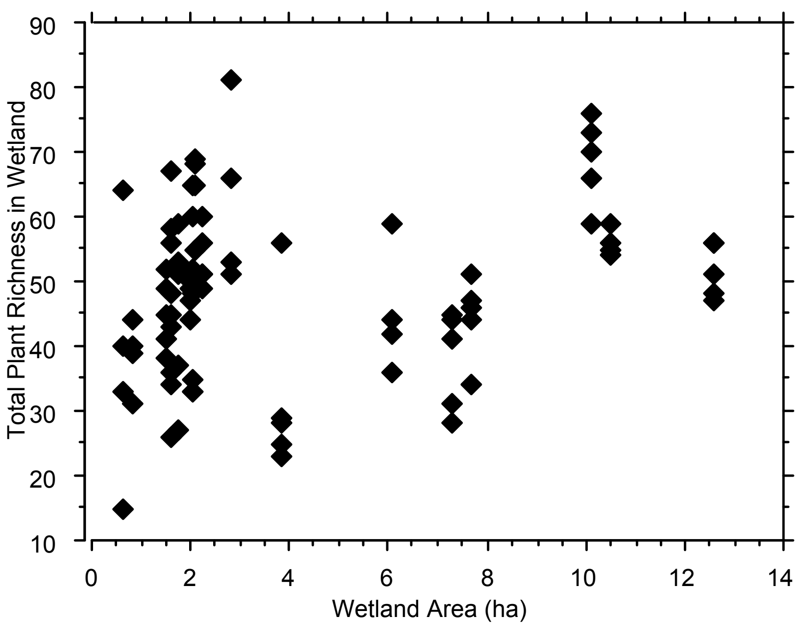

Figure 3-3. Plant richness and wetland area.

The percent of impervious area within the watershed was negatively correlated with average plant richness in the emergent zones and scrub-shrub community types (Figure 3-4). On average both types exhibited significantly lower species richness as the amount of impervious area in the watershed increased (Frz, PEM: R = 0.55, p = 0.002; PSS: R = 0.57, p = 0.001).

All years of data were examined for the relationship between mean annual WLF and plant richness with the following results. Total plant richness found in wetlands was unrelated to the degree of WLF. Average plant richness within the forested community type was also found to be unrelated to mean annual WLF. However, in both the emergent and scrub-shrub types, average plant richness was negatively correlated with mean annual WLF. Figure 3-5 shows plant richness in the these community types related to mean annual WLF for all years of data and all wetlands. The results showed a significant relationship in both types for all years combined (Frz, PEM: R = -0.38, p =

0.006; PSS: R=-0.5, p = .0001). When years were examined singly, both the emergent and scrub-shrub types showed significant negative correlations between plant richness and water WLF for three of the five years.

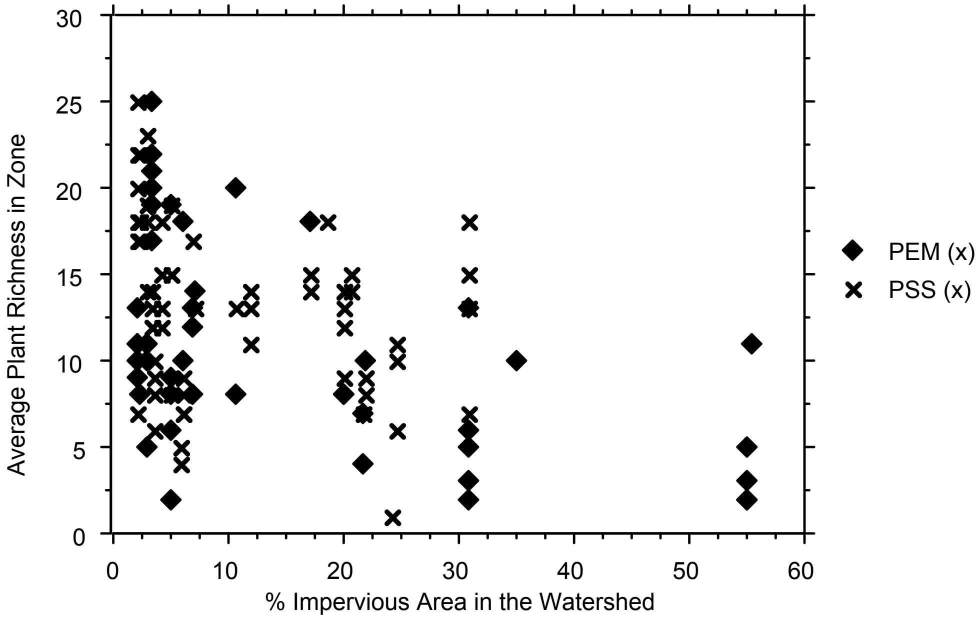

Figure 3-4. Richness in the emergent and scrub-shrub communities and impervious area in the watershed.

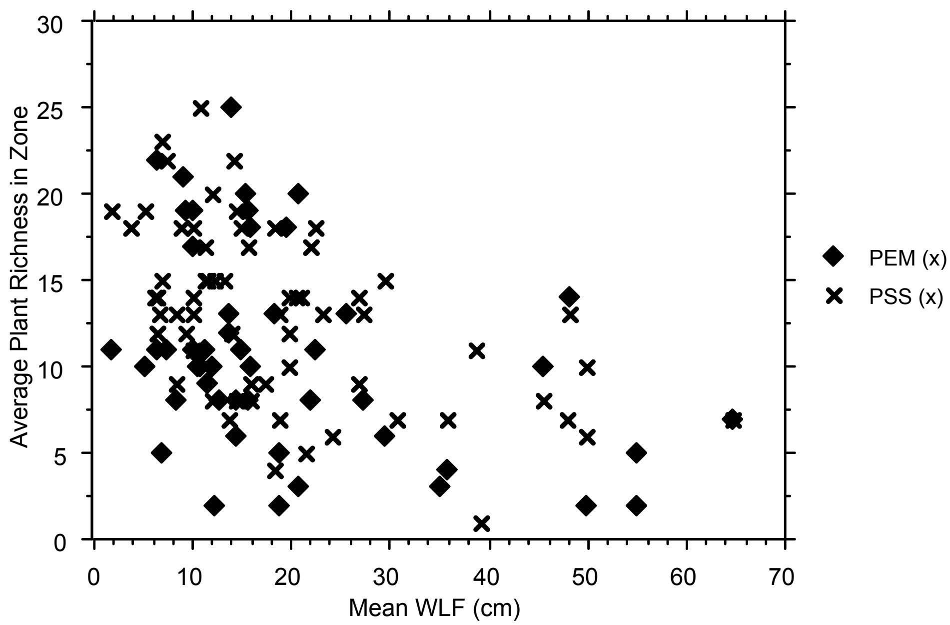

Figure 3-5. Plant richness in the emergent and scrub-shrub communities related to mean annual WLF.

Wetland Indicator Status

All 242 plant species found during our study were assigned an indicator status based on the hydrologic regimes observed at the study sample stations. Since upland zones represented only 18% of the stations sampled, it was expected that many transitional species would look wetter than indicated by Reed. However, for most species, the hydrology based assignments for indicator status matched Reed's assignments. Of the 93 that did not match, approximately 42% were eliminated because the number of observations was considered to be too low (less than 10) for an accurate assessment or because the water depths measured at the wetland gage did not accurately reflect the water depths at the vegetation sample stations. In addition, 27 species were eliminated as they were observed growing in conditions that were within 10% of the maximum and minimum range of observed frequency assigned by Reed (Table 3-1) and within the margin of error for measuring water levels. The remaining 27 species were selected as candidates for a change in the indicator status assigned by Reed and are shown in Table 3-5.

Interestingly, several species listed in the table are categorized as obligates by Reed but were found in wet zones in fewer than 88% of our observations. Most were observed in at least 25 sample stations. This study, however, was not able to measure hydrology at each plot but only calculate it based on events measured at the wetland gage station with respect to the elevation of the vegetation sample station. The remaining differences between the Reed categories and the hydrologically based categories may be due to the presence of hydrologic conditions not accounted for by our methods. Nevertheless, the number of observations and frequency of inundation recorded warrant a review of the species listed in Table 3-5.

Table 3-5. Comparison of indicator assignments for some plant species.

Plant Species	Form	Weed	Reed Wetland Indicator	Status Assigned Using PSWSMRP Study Data	Percent of Observations Plant Was Wet	Number of Observations
Epilobium ciliatum (watsonii)	herb	no	FAC	FACW	0.83	144
Acer circinatum	shrub	no	FACU	FAC	0.58	180
Dicentra formosa	herb	no	FACU	FAC	0.48	23
Dryopteris expansa (austriaca)	fern	no	FACU	FAC	0.62	143
Epilobium angustifolium	herb	no	FACU	FAC	0.6	15
Hypericum formosum	herb	no	FACU	FAC	0.65	17
Rubus procerus (discolor)	shrub	exotic	FACU	FAC	0.64	115
Sambucus racemosa	shrub	no	FACU	FAC	0.54	205
Stellaria media	herb	no	FACU	FAC	0.65	40
Tsuga heterophylla	tree	no	FACU	FAC	0.56	211
Rubus laciniatus	shrub	exotic	FACU	FACW	0.72	116
Carex hendersonii	sedge	no	none	FAC	0.33	21
Carex exsiccata (vesicaria)	sedge	no	OBL	FACW	0.85	13
Carex obnupta	sedge	no	OBL	FACW	0.67	72
Hypericum anagalloides	herb	no	OBL	FACW	0.8	25
Juncus acuminatus	rush	no	OBL	FACW	0.87	30
Ludwigia palustris	herb	no	OBL	FACW	0.75	24
Lycopus americanus	herb	no	OBL	FACW	0.74	50
Lycopus uniflorus	herb	no	OBL	FACW	0.7	44
Mimulus guttatus	herb	no	OBL	FACW	0.82	11
Myosotis laxa	herb	no	OBL	FACW	0.8	43
Salix pedicellaris	shrub	no	OBL	FACW	0.73	26
Scirpus atrocinctus	shrub	no	OBL	FACW	0.74	65
Scirpus microcarpus	shrub	no	OBL	FACW	0.74	42
Solanum dulcamara	herb	no	OBL	FACW	0.71	212
Veronica americana	herb	no	OBL	FACW	0.74	108
Veronica scutellata	herb	no	OBL	FACW	0.69	80

DISCUSSION

Wetland management regulations in the Puget Sound lowlands, for the most part, classify wetlands on the basis of area, the number and type of vegetation communities, and the presence of threatened or endangered species (King County 1990, Toshach 1991). Although one might rationalize that larger wetlands are more diverse ecosystems, we found that in the case of plants, wetland area is not directly related to the rarity or richness of the plant community.

Other factors, such as hydrologic regime and the kinds and frequency of disturbance, appeared to be more critical in determining the diversity and character of the wetland plant communities we studied. Generalized classifications of vegetation structure, such as forested, scrub-shrub and emergent, lend no insight into the presence or absence of unusual plant species, plant associations or the biodiversity value of a wetland to a region. Our results suggest that selecting for wetland size and certain types of wetland plant communities will not insure protection of regional wetland values and functions.

Eleven distinct wetland plant communities were identified that are typical of the region. These communities were mostly found in mixed assemblages interspersed throughout individual wetlands. Most of the wetlands studied were characterized by several wetland plant community types with transition zones between them. In general, when several community types were present, plant richness was higher within individual communities, as many species were observed to transition between community types.

The presence of these zones is highly dependent on the hydrologic gradients at work in a wetland. Wetlands with the richest and most diverse plant communities were typically characterized by more complex hydrology and more variable morphology, providing many surfaces at different gradients for plant species to inhabit. Wetlands with simpler vegetation communities were more frequently topographically uniform, resulting in simpler hydrologic patterns. These differences may be traced, to some extent, to patterns of disturbance, including water level fluctuation.

It is important to focus our management efforts toward understanding the conditions required for wetland plant and animal diversity and to comprehensively mitigate the functions lost when wetlands are disturbed. In addition to preserving large wetlands with diverse hydrologic zones, we should consider addressing land use and development constraints to limit the extent of increases in water level fluctuations occurring in wetlands due to increased impervious area. Limiting other types of disturbance and monitoring invasive species presence also provide reasonable tools for maintaining species richness and regional biodiversity.

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Appendix Table 3-1. List of plant species and frequency found among 19 Puget lowland palustrine wetlands.

Plant Species	Number of Wetlands	Percent of All Wetlands	Plant Species	Number of Wetlands	Percent of All Wetlands
Adiantum pedatum	1	0.04	Rorippa calycina	5	0.19
Agrostis scabra	1	0.04	Rosa pisocarpa	5	0.19
Aira caryophyllea	1	0.04	Sium suave	5	0.19
Anthoxanthum odoratum	1	0.04	Utricularia minor	5	0.19
Arbutus menziesii	1	0.04	Agrostis gigantea (alba)	6	0.23
Asaurum caudatum	1	0.04	Crataegus monogyna	6	0.23
Berberis aquifolium	1	0.04	Fraxinus latifolia	6	0.23
Brasenia schribneri	1	0.04	Galium aparine	6	0.23
Carex athrostachya	1	0.04	Hypericum formosum	6	0.23
Carex stipata	1	0.04	Ludwigia palustris	6	0.23
Chenopodium alba	1	0.04	Rubus leucodermis	6	0.23
Cladina rangiferina	1	0.04	Smilacena racemosa	6	0.23
Convolvulus sepium	1	0.04	blue-green algae	6	0.23
Cornus canadensis	1	0.04	Anaphalis margaritacea	7	0.27
Echinochloa crusgalii	1	0.04	Festuca rubra	7	0.27
Festuca pratensis	1	0.04	Galium cymosum	7	0.27
Fragaria virginiana	1	0.04	Glyceria borealis	7	0.27
Goodyeara oblongifolia	1	0.04	Holodiscus discolor	7	0.27
Hippurus vulgaris	1	0.04	Juncus bufonius	7	0.27
Hydrocotyl ranunculoides	1	0.04	Lycopus americanus	7	0.27
Hydrophyllum tenuipes	1	0.04	Myosotis laxa	7	0.27
Juncus supiniformis	1	0.04	Potamogeton natans	7	0.27
Lamium purpurea	1	0.04	Rumex crispus	7	0.27
Lythrum salicaria	1	0.04	Carex lenticularis	8	0.31
Melilotus alba	1	0.04	Dactylis glomerata	8	0.31
Nymphaea odorata	1	0.04	Geranium robertianum	8	0.31
Poa palustris	1	0.04	Glyceria elata	8	0.31
Poa pratensis	1	0.04	Juncus acuminatus	8	0.31
Potamogeton diversifolius	1	0.04	Juncus ensifolius	8	0.31
Potamogeton gramineus	1	0.04	Rhododendron groenlandicum (Ledum g.)	8	0.31
Ranunculus acris	1	0.04	Nuphar polysepalum	8	0.31
Rhinanthus crista-galli	1	0.04	Oplopanax horridus	8	0.31
Ribes bracteosum	1	0.04	Potentilla palustris	8	0.31
Rorippa curvisiliqua	1	0.04	Ribes lacustre	8	0.31
Rosa nutkana	1	0.04	Stachys cooleyae	8	0.31
Rosa rugosa	1	0.04	Symphoricarpos albus	8	0.31
Rumex acetosella	1	0.04	Tiarella trifoliata	8	0.31
Sagittaria latifolia	1	0.04	Carex hendersonii	9	0.35
Scirpus acutus	1	0.04	Circium arvense	9	0.35
Solanum nigrum	1	0.04	Dicentra formosa	9	0.35
Stellaria longifolia	1	0.04	Rosa gymnocarpa	9	0.35
Tanacetum vulgare	1	0.04	Scirpus atrocinctus	9	0.35
Trifolium pratense	1	0.04	Sorbus scopulina	9	0.35
Vaccinium ovatum	1	0.04	Sphagnum spp.	9	0.35
Vicia sativa	1	0.04	Salix pedicellaris	10	0.38
Adenocaulon bicolor	2	0.08	Scirpus microcarpus	10	0.38
Alnus sinuata	2	0.08	Agrostis capillaris (tenuis)	11	0.42
Alopecurus pratensis	2	0.08	Callitriche heterophylla	11	0.42
Azola mexicana	2	0.08	Epilobium angustifolium	11	0.42
Bromus ciliatus	2	0.08	Lycopus uniflorus	11	0.42

Appendix Table 3-1 continued. List of plant species and frequency found among 19 Puget lowland palustrine wetlands.

Plant Species	Number of Wetlands	Percent of All Wetlands	Plant Species	Number of Wetlands	Percent of All Wetlands
Claytonia lanceolata	2	0.08	Rubus parviflorus	11	0.42
Cornus nuttallii	2	0.08	Scutellaria lateriflora	11	0.42
Elytrigia repens (Agropyron repens)	2	0.08	Sparganium emersum	11	0.42
Eriophorum chamissonis	2	0.08	Torreyochloa pauciflora (Puccinellia p.)	11	0.42
Glecoma hederacea	2	0.08	Veronica scutellata	11	0.42
Gnaphalium uliginosum	2	0.08	Equisetum hyemale	12	0.46
Gymnocarpium dryopteris	2	0.08	Equisetum telmateia	12	0.46
Hercaleum lanatum	2	0.08	Holcus lanatus	12	0.46
Hypochaeris radicata	2	0.08	Menziesia ferruginea	12	0.46
Impatiens noli-tangere	2	0.08	Polygonum hydropiper	12	0.46
Lolium mulitflorum	2	0.08	Typha latifolia	12	0.46
Lotus corniculatus	2	0.08	Bidens cernua	13	0.5
Menyanthes trifoliata	2	0.08	Carex arcta	13	0.5
Pinus monticola	2	0.08	Galium trifidum	13	0.5
Polygonum amphibium	2	0.08	Ilex aquifolia	13	0.5
Rhynchospora alba	2	0.08	Picea sitchensis	13	0.5
Rumex obtusifolius	2	0.08	Sorbus americana	13	0.5
Salix hookeriana	2	0.08	Berberis nervosa	14	0.54
Smilacena stellata	2	0.08	Carex utriculata =(rostrata)	14	0.54
Sparganium eurycarpum	2	0.08	Maianthemum dilatatum	14	0.54
Streptopus roseus	2	0.08	Salix alba	14	0.54
Taraxacum officinale	2	0.08	Tolmiea menziesii	14	0.54
Trfolium repens	2	0.08	Corylus cornuta	15	0.58
Vaccinium uliginosum	2	0.08	Juncus effusus	15	0.58
Vallisneria americana	2	0.08	Lemna minor	15	0.58
Actea rubra	3	0.12	Prunus emarginata	15	0.58
Alisma plantago-aquatica	3	0.12	Stellaria media	15	0.58
Carex exsiccata (vesicaria)	3	0.12	Trillium ovatum	15	0.58
Circium vulgare	3	0.12	Cornus sericea (stolonifera)	16	0.62
Cytisus scoparius	3	0.12	Geum macrophyllum	16	0.62
Dulichium arundinaceum	3	0.12	Oenanthe sarmentosa	16	0.62
Elodea canadensis	3	0.12	Acer macrophyllum	17	0.65
Lolium perenne	3	0.12	Carex obnupta	17	0.65
Mimulus guttatus	3	0.12	Ranunculus repens	17	0.65
Montia siberica	3	0.12	Rubus procerus (discolor)	17	0.65
Nasturtium officinale	3	0.12	Urtica dioica	17	0.65
Phleum pratense	3	0.12	Equisetum arvense	18	0.69
Ribes sanguineum	3	0.12	Glyceria grandis	18	0.69
Nastutium officinale	3	0.12	Luzula parviflora	18	0.69
Taxus brevifolia	3	0.12	Phalaris arundinaceae	18	0.69
Utricularia vulgaris	3	0.12	Populus balsamifera	18	0.69
Viola glabella	3	0.12	Blechnum spicant	19	0.73
Azolla filiculoides	4	0.15	Carex deweyana	19	0.73
Eleocharis ovata	4	0.15	Malus fusca (Pyrus f.)	19	0.73
Eleocharis palustris	4	0.15	Salix sitchensis	19	0.73
Hieracium pratense	4	0.15	Acer circinatum	20	0.77
Hypericum anagalloides	4	0.15	Lysichitum americanum	20	0.77
Iris pseudacorus	4	0.15	Polypodium glycyrrhiza	20	0.77

Appendix Table 3-1 continued. List of plant species and frequency found among 19 Puget lowland palustrine wetlands.

Plant Species	Number of Wetlands	Percent of All Wetlands	Plant Species	Number of Wetlands	Percent of All Wetlands
Kalmia microphylla	4	0.15	Pseudotsuga menziesii	20	0.77
Linnaea borealis	4	0.15	Salix lucida var. lasiandra	20	0.77
Mentha arvensis	4	0.15	Solanum dulcamara	20	0.77
Myosotis scorpioides	4	0.15	Thuja plicata	20	0.77
Petasites frigidus	4	0.15	Tsuga heterophylla	20	0.77
Physocarpus capitatus	4	0.15	Veronica americana	20	0.77
Plantago lanceolata	4	0.15	Oemleria cerasiformis	21	0.81
Plantoga major	4	0.15	Gaultheria shallon	22	0.85
Populus tremuloides	4	0.15	Sambucus racemosa	22	0.85
Solidago canadensis	4	0.15	Dryopteris expansa (austriaca)	23	0.88
Spirodela polyrhiza	4	0.15	Epilobium ciliatum (watsonii)	23	0.88
Streptopus amplexifolius	4	0.15	Lonicera involucrata	24	0.92
Vaccinium oxycoccos	4	0.15	Pteridium aquilinum	24	0.92
Agrostis oregonensis	5	0.19	Rhamnus purshiana	24	0.92
Amelanchier alnifolia	5	0.19	Rubus laciniatus	24	0.92
Betula papyrifera	5	0.19	Vaccinium parvifolium	24	0.92
Circaea alpina	5	0.19	Alnus rubra	25	0.96
Convolvulus arvensis	5	0.19	Athyrium filix-femina	25	0.96
Digitalis purpurea	5	0.19	Salix scoulerleriana	25	0.96
Drosera rotundifolia	5	0.19	Spirea douglasii	25	0.96
Hedera helix	5	0.19	Polystichum munitum	26	1
Lonicera ciliosa	5	0.19	Rubus spectablilis	26	1
Ribes divaricatum	5	0.19	Rubus ursinus	26	1

CHAPTER 4 EMERGING MACROINVERTEBRATE DISTRIBUTION, ABUNDANCE AND HABITAT USE

by Klaus O. Richter and Robert W. Wisseman

INTRODUCTION

Macroinvertebrates—particularly insects, are diverse and abundant zoological components of freshwater aquatic systems. Of all invertebrates, the trophic diversity and numerical abundance of insects, and especially the Diptera (true flies), make this group the most important taxa in streams, lakes and other water environments. Unique adaptations have evolved in their life-history patterns (breeding, oviposition, hatching and development), morphological and physiological characteristics (respiration) and behavioral traits (lotic/lentic habitat affinities, functional feeding groups) to enable them to occupy most wetland habitats and trophic levels.

Recent research focusing on aquatic invertebrates in wetlands, indicates the importance of insects in energy and nutrient transfer within aquatic ecosystems (Rosenberg and Danks 1987). They furnish food for other invertebrates (e.g., predatory insects and arachnids such as mites and spiders) and comprise significant portions of the nutritional requirements of amphibians, water birds and small mammals. They are especially important to rearing fish (e.g., Salmonidae, game fishes), contributing to commercial and sport fisheries.

Diptera as well as other aquatic insects are pivotal components of complex food webs, significantly increasing the number of links in the web with their richness and abundance. As filter feeders, shredders and scrapers they convert and assimilate microorganisms and vegetation into biomass of aquatic insects providing significant production available to secondary and tertiary consumers. Alternately, insects are sometimes thought detrimental to human health. Dipteran families including Simuliidae (black flies) and Culicidae (mosquitoes) are vectors of disease and can be pests to humans, livestock and other mammals. Consequently, they may be of medical and economic importance (Courtney et al. 1996).

The distribution and abundance of macroinvertebrates in running waters and lakes have long been recognized as important tools in describing and assessing the condition of these aquatic ecosystems (Rosenberg and Resh 1993). However, it has been relatively recently that they were identified as providing an indication of the condition of palustrine environments (Ludwa and Richter, this volume), particularly wetlands of watersheds undergoing urbanization (Ludwa 1994, Hicks 1996). This is primarily because basic information regarding their spatial and seasonal distribution and abundances in palustrine wetlands is uncommon. Moreover, specific hydrologic, water quality and other habitat characteristics that may account for invertebrate, and specifically insects, remain unavailable.

Consequently, in this paper we characterize the emergent macroinvertebrates in palustrine wetlands in the Pacific Northwest by describing the distribution and abundance of taxa collected in emergence traps at 19 wetlands of the Puget Sound region. Moreover, we determine characteristics of wetlands and watersheds that may account for their occurrences.

METHODS

We collected adult macroinvertebrates (e.g., most often insects of minimum of 5mm in size and easy to see with unaided eye), encompassing a wide diversity of taxa using emergence traps (Figure 4-1). We used emergence traps rather than dip nets (e.g., sweep nets) or benthic sampling because captures in emergent traps represent the final component of insect production, allows quantification of cumulative production over variable time periods and presorts species on their ability to climb or fly into the collecting chamber facilitating identification procedures. In addition, emergence traps exhibit less sampling variability compared to sediment sampling.

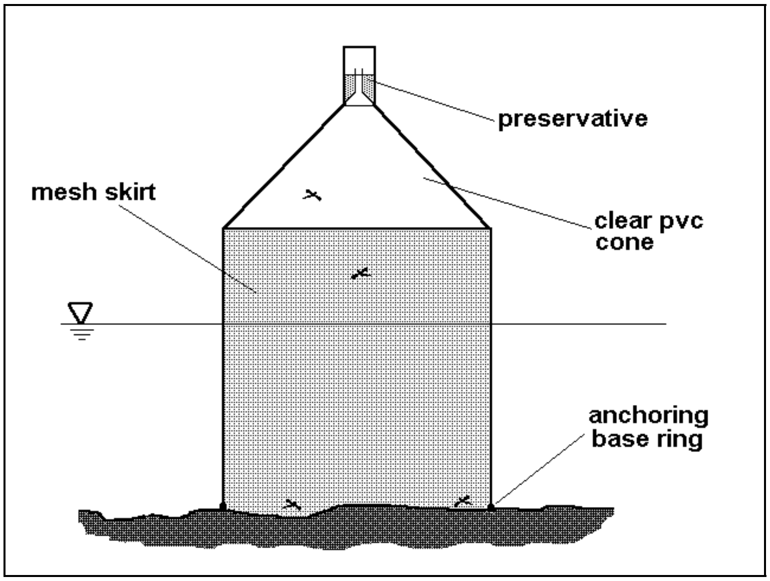

Figure 4-1. Side view cross-section of aquatic macroinvertebrate emergence trap.

The traps function by funneling emerging invertebrates upward into a glass jar at the top of the trap containing a liquid preservative. We placed three replicate traps, each covering a circular area of 0.25 M^2 within approximately one meter of each other in the deepest (maximum 1m deep) areas of wetlands that could be reached with chest waders during spring when invertebrates were first expected to emerge. Substrate and vegetation characteristics present at the trapping locations are described for individual wetlands in Table 4-1.

Traps were installed with base rings embedded in the substrate or flush to the ground. In September 1988 we installed traps in 14 wetlands with traps at five additional wetlands added in May 1989. We emptied traps semi-monthly. Traps were not emptied from mid-November 1988 to March 1989 and during other winter periods because of low or non-existent winter emergence. We attempted to empty traps within a three day period, although in some cases we took 19 days to collect samples. We captured and summarized macroinvertebrate data for all 19 wetlands in 1989 (including captures from September 15,1988 through September 31, 1989) and 1993 (including captures from April 10, 1993 through April 9, 1994). In 1995 we trapped in 18 wetlands (deleted TC13) from January 1, 1995 through October 30, 1995. Since winter emergence was low, our data essentially represents values for invertebrate years 1989, 1993 and 1995.

We identified emergent macroinvertebrates and placed them into broad groupings (adult arthropods, terrestrial arthropods, aquatic and semi-aquatic insects) for descriptive and statistical analysis. We use 1989 as our comprehensive description of macroinvertebrate distribution and abundance in Pacific Northwest wetlands in that we identified all invertebrates to the lowest taxa feasible—generally genus and species. In subsequent years we classified only to major taxa (e.g., Orders) except for Diptera, for which we identified captures to family and further limited our taxonomic efforts to the numerically dominant chironomid midges (suborder Nematocera, lower dipteran flies) with other Nematocera identified only to family. The suborder Brachycera (higher dipteran flies) were not identified because of the specialized expertise required for their taxonomy and because of the possibility they entered traps from adjacent areas during low water. We assigned the Dipterans to the aquatic group, since the vast majority of taxa within this order have larval stages developing in water or in saturated soils (Courtney et al. 1996). We identified wetland-associated terrestrial forms as species in which all life stages are found in terrestrial habitats.

The majority of the wetlands chosen for study are small palustrine systems ranging from several hectares to less than one hectare in size. We classified wetlands according to the level of development within their watersheds and flooding regime. Thus within watersheds of various levels of urbanization, wetlands were identified as to whether they were perennially or seasonally flooded during the monitoring year. Permanent wetlands exhibited standing water the entire year whereas seasonally flooded sites generally dried out between April and June and were re-flooded only after the onset of autumn rains in mid-October.

Table 4-1. Aquatic invertebrate emergence trap conditions (from Ludwa 1994).

	Flow		Substrate			Vegetation																					
Wetland	still water	discernible flow	gravel	sand	silt/mud	moss	periphyton	*Alnus rubra*	*Athyrium*	*Carex*	*Epilobium*	*Equisetum*	*Glyceria*	*Juncus*	*Lemna*	*Lysichtum*	*Nuphar*	*Oenanthe*	*Phalaris*	*Potentilla*	*Salix*	*Scirpus*	*Solanum*	*Sparganium emersum*	*Spirea*	*Typha*	*Veronica*	unidentified plants
AL3	×					×																						
B31		×	×	×	×			×		×		×							×			×						
BBC24	×																					×				×	×	
ELS39	×				×		×			×				×											×			
ELS61	×						×								×											×		
ELW1		×			×	×			×				×						×								×	
FC1		×			×		×						×		×				×									
HC13	×				×			×					×						×			×						
JC28	×				×			×								×												
LCR93		×															×	×							×	×		×
LPS9		×																×	×									
MGR36	×												×		×					×	×					×	×	
NFIC12	×				×	×	×							×														
PC12	×									×			×		×				×			×			×	×		
RR5	×									×				×								×						
SC4	×				×	×	×	×			×																×	
SC84	×					×	×											×					×				×	
SR24	×									×														×				
TC13	×				×	×	×											×				×				×	×	

We ran three Detrended Correspondence Analysis (DCA), iterations of the 1989 wetland data (Hill 1979, Hill & Gauch 1980) as follows; 1) all taxa; including terrestrial, aquatic and semiaquatic taxa identified to the lowest level reported, 2) all aquatic/semi-aquatic taxa, including the Brachycera, identified to the lowest level reported and 3) all chironomid midge taxa identified to the lowest level reported. Taxa abundances were log transformed prior to running the DCA program.

RESULTS AND DISCUSSION

Annual Overall Arthropod Richness and Abundance

Annual arthropod yield is presented in Table 4-2 (Table 4-2 and all subsequent tables may be found in Appendix 4-1). Terrestrial abundance was highest in 1989 (but see wetland NFIC12). Low numbers of arthropods were captured at wetlands both in 1993 and 1995. Total aquatic and semi-aquatic taxa richness and abundance varied widely between years and wetlands but were consistently dominated by Diptera in both categories. Aquatic and semi-aquatic abundance was highest in LCR93 with

21,501 invertebrates counted in 1995 and lowest in ELW1 with 256 animals tallied in 1995.

Terrestrial Arthropod Richness and Abundance

Arachnids and hexpods insects were the two terrestrial arthropod classes most frequently captured in emergence traps (Table 4-3). Arachnids are common predators (spiders) and parasites (mites) on aquatic insects and other invertebrates. Of the insects, we captured a total of nine terrestrial orders. Homoptera—particularly Aphididae (aphids), Coleoptera (beetles) and Hymenoptera (e.g., Parasitoid wasps) were represented in the greatest numbers. The captured taxa of these orders are often associated with emergent plant parts above water which were enclosed by the traps. That is probably why they were captured in our traps and not because they are obligate wetland species.

Total terrestrial arthropod richness ranged from a high of ten to a low of seven major invertebrate taxa in a single year (Table 4-3). Neuroptera were missing from eight wetlands (AL3, ELS39, ELW1, PC12, RR5, SC84, SR24 and TC13) and Hemiptera from five (AL3, BBC24, FC1, SR24 and TC13). Densities ranged from 56,439 M^2 in BBC24 for 1989 to a low of 9 M^2 at JC28 in 1993. The most abundant terrestrial taxa were Aphididae (e.g., aphids-Homoptera) mostly because of their reproductive characteristics, communal feeding and small size. Aphids frequently feed on exposed broad-leaved aquatic vegetation (personal observation) and therefore are abundant in open water wetlands that are characterized by water lilies such as found in BBC24. They are largely missing from forested and scrub-shrub wetlands without such plant species as for example JC28 and AL3.

Aquatic and Semi-Aquatic Insect Richness and Abundance

Five aquatic and semi-aquatic insect orders, Ephemeroptera (mayflies), Odonata (dragonflies/damselflies), Plecoptera (stoneflies), Trichoptera and Diptera, were collected within wetlands. Ephemeroptera were captured at 12 wetlands during the survey (Table 4-4). Their abundance was low (<25) except at LCR93 and JC28 at which maximum numbers were 232 and 206 individuals respectively. In the 1989 survey they represented only two taxa (Table 4-5). Ephemeroptera, in general, inhabit both lentic and lotic waters where adequate supplies of dissolved oxygen are found. The taxa we identified, *Callibaetis* and *Paraleptophlebia,* are also found mostly in perennial and seasonal wetlands respectively. Overall, Ephemeroptera richness and abundance were greater at perennial than in annually/seasonally flooded sites. Moreover, they were patchily represented in non-urbanized sites (AL3, SC4, HC13, LCR93, MGR36, SR24, TC13, PC12) and moderately urbanized (BBC24, ELW1, ELS61, ELS39, JC28, NFIC12, RR5, SC84, LPS9) sites but were not found at both highly urbanized sites (B3I and FC1).

Surprisingly, Odonata were captured at only three wetlands, and in low (<2 at ELS61 and LPS9, <25 at BBC24) numbers. A total of three species of damselflies were found at BBC24 and ELS61. Odonata require year-round standing water, and therefore are generally not found in temporary and seasonal wetlands.

Plecoptera, a lotic insect order, was encountered at eight wetlands. In 1989 this represented eight taxa including a new species, *Capni*. We found Plecoptera in large numbers (1576) at LCR93 in 1989, moderate numbers (101) at NFIC12 in 1995 and low numbers of 42 and 32 animals/ M^2 in 1995 at RR5 and 1989 at JC28 respectively (Table 4-4). In all other wetlands and years they were collected in low numbers (<10). Plecoptera was usually found in wetlands with flow-through channels.

Trichoptera taxa richness is relatively high with 24 taxa identified during 1989 alone (Table 4-6). Regardless of wetland, the majority of larvae belonged to the family Limnephilidae. Oxyethira, a hydroptilid, was common at BBC24. Numbers of Trichoptera were low (<200) at many wetlands with the exception of 1995 at LPS9.

Insect emergence was clearly dominated by Diptera (Table 4-2). The abundance of individuals within these taxa often varied widely with the highest number of Diptera being as much as 13 times greater than the lowest numbers (e.g., ELS61 versus ELW1). Most often variations between high counts are between two to six times the low counts. More extreme are abundance data for chironomids in which numbers in high years are as much as 190, 84 and 36 times the numbers found in low year counts as in RR5, MGR36 and NFIC12, respectively, whereas the ranges at most other wetlands differed by five to 20. Nevertheless, the relative ranking of taxa abundance by wetlands was often relatively constant with the same wetlands retaining their lowest or highest relative ranking from among all wetlands. B3I and ELS61, for example, ranked in the top three in Cecidomyiidae and Tipulidae abundance in at least two of three years. Other common dipteran families captured included the Psychodidae, Tipulidae, and Empididae.

Actual dipteran numbers ranged from a high of high of 20,781 M^2 in ELS61 in 1989 to a low of 256 M^2 in ELW1 in 1995. In fact, ELW1 consistently had the lowest number and ELS61 the highest number of Diptera during the three-years of monitoring. Low dipteran numbers of under 1,000 M^2 in two out of three years were also identified at JC28 and high numbers of 10,000 M^2 or more at LCR93 and NFIC12. As expected significantly fewer aquatic and semi-aquatic forms and numbers were present in the higher dipteran suborder Brachycera, than the largely aquatic Nematocera (longhorned flies).

In the Nematocera, members of the family Chironomidae (midges) were clearly represented by the greatest number of taxa and often also by numbers. Chironomid midges have been found to be one of the most abundant and diverse groups in other regions of North America (Wrubleski 1987) and therefore these findings were expected. In 1989 we identified a high of 42 taxa in BBC24 and counted a high total abundance of 11,925 animals at ELS61 (Table 4-6). Chironomid taxa richness was consistently high in the perennial, non-urbanized wetlands, and consistently low in the non-urbanized wetlands that dried out in the summer. Table 4-7 provides the abundance rankings of all Chironomid taxa within our 1989 wetland characterization scheme.

Non-chironomid families of numerical importance include the Sciaridae, Cecidomyiidae (gall gnats that sometimes live in the tissues of live aquatic vegetation), Dixidae (dixid midges), and Tipulidae (crane flies). Rarely found non-Chironomids included the Anisopodidae, Bibionidae, Scatopsidae, Simuliidae (black flies) and Trichoceridae. Certain Psychodidae are often found in water and sewage-treatment facilities (Courtney at al. 1996). Psychodidae, Ptychopteridae and Syrphidae are collector-gatherers feeding on decaying fine organic matter associated with microorganisms. Collector-filterers include most Culicidae and Simuliidae. Tipulidae and Ephydridae are considered shredders.

Taxa richness of semi-aquatic and aquatic insects was generally higher (>40 taxa) in persistent than seasonal wetlands that dried out in summer (but see LCR93; Table 4-6). Both high and low richness was found in wetlands whose watersheds were largely non-urbanized depending on whether they remained flooded or dried out in summer. Interestingly, the three wetlands in non-urbanized watersheds that dried out in summer 1989 (AL3, NFIC12 and TC13) exhibited the lowest overall richness values as did the wetlands that dried out in highly urbanized watersheds (ELS39, LPS9, and SC84). Richness values for these wetlands ranged from 20-30 total taxa, with only 8-16 Chironomid taxa.

In 1989 we identified a high of 62 non-Brachycera taxa at BBC24 with 8570 animals M^2. The lowest richness of one third this highest value was observed at NFIC12, AL3 and ELS39. Densities were lowest at 655 animals M^2 in ELW1.

Shannon and Pielou diversity indices (Shannon and Weaver 1949, Pielou 1966) calculated for the full compliment of aquatic/semiaquatic invertebrates as well as the chironomid communities (Table 4-6) indicate that most wetlands within highly urbanized watersheds have lower richness than those in less urbanized watersheds. Most wetlands characterized by water permanence generally also exhibited higher richness than those that dried out.

In all three analyses using DCA, permanent wetlands were clearly distinguished from summer dry sites along axis 1 (Figures 4-2, 4-3 and 4-4). Axis 1 is most easily interpreted as representing a gradient progressing form wetlands experiencing lengthy summer drying, to wetlands having year-round standing water. Also, those summer dry site communities which experience highly fluctuating water levels are found at the extreme of axis 1, for example B3I, with one of the most urbanized watersheds. In general, insect communities of wetlands characterized by summer drought and flashy hydrology are harsh and unpredictable environments are less diverse; most likely because fluctuating environments generally exhibit simpler food chains or, fewer linkages per species, than stable ecosystems.

Axis 2 of the DCA plots are not satisfactorily related to an environmental gradient. Summer dry, moderately and highly urbanized sites were scattered more widely on this axis than were wet sites. The four wet, non-urbanized sites by being closely clustered and showing little separation on either axes, indicate very similar invertebrate communities. Though, the moderately and highly urbanized, wet sites could also be distinguished form the dry sites, they displayed more separation on both axis 1 and 2 than the non-urbanized wet sites. The BBC24 community was usually distant on axis 1 from the other wet sites. This wetland exhibited high taxa richness, and contained many Odonata, Trichoptera and Chironomidae taxa which were not typical of other wetlands.

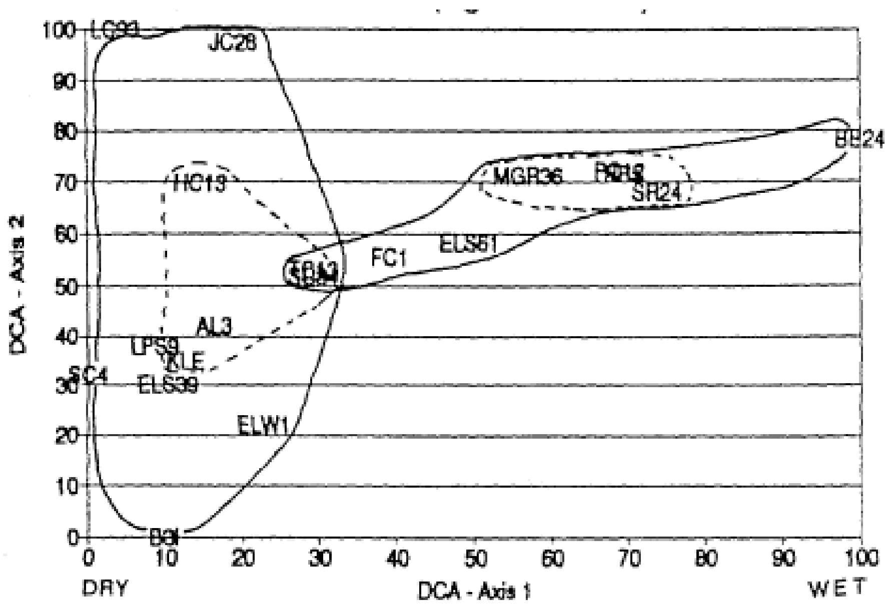

Figure 4-2. Terrestrial, aquatic and semi-aquatic taxa DCA analysis results.

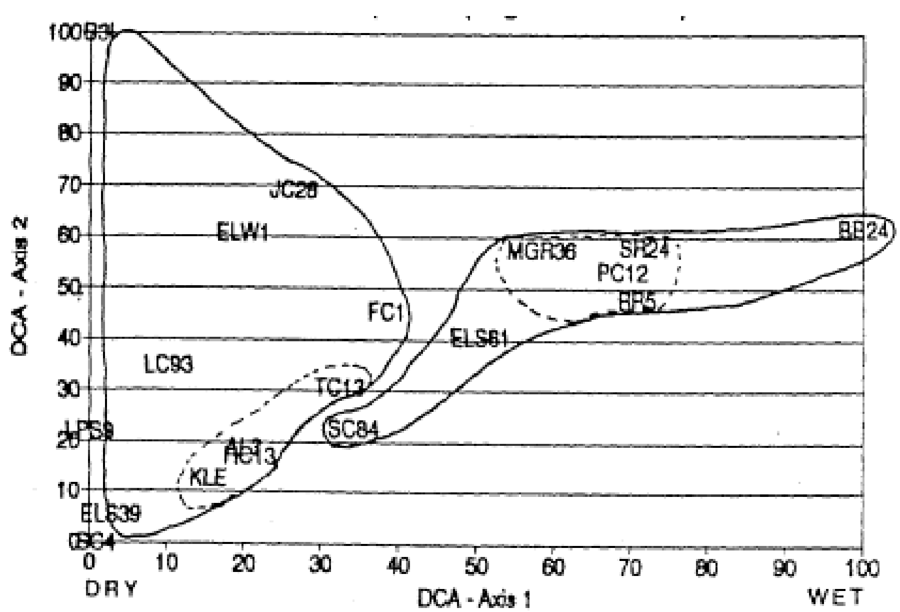

Figure 4-3. Aquatic and semi-aquatic taxa DCA analysis results.

On axis 2, communities of intermittent wetlands showed considerable separation from flooded wetlands indicating that seasonally flooded habitats were more variable in community structure than those with permanent standing water. As in the case for non-urbanized, wet sites, the four non-urbanized summer dry site insect communities clustered more closely together than the communities in moderately and highly urbanized site.

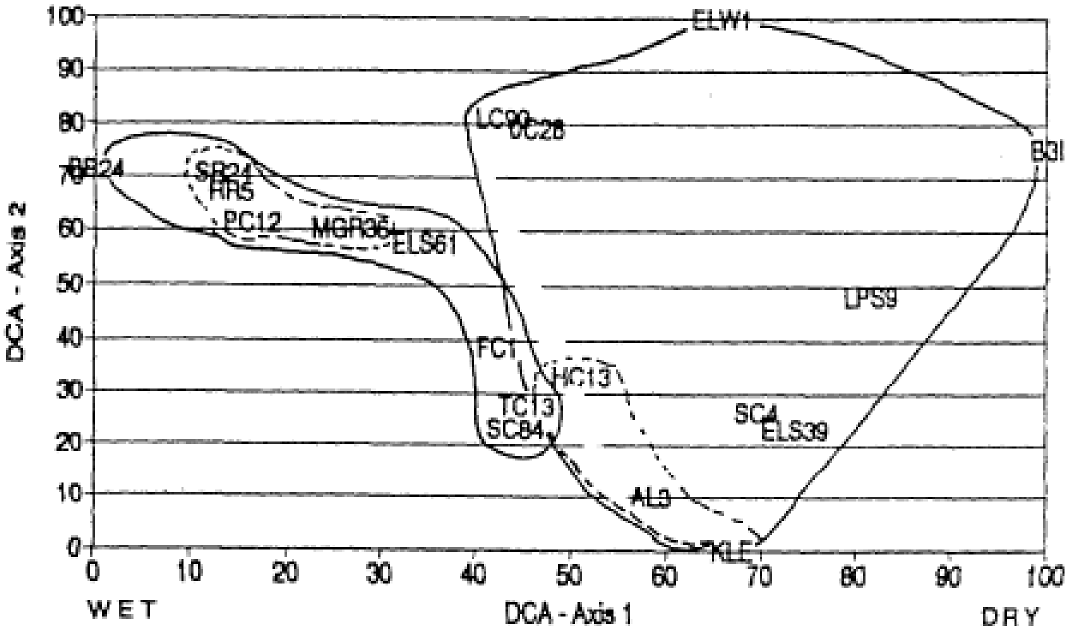

Figure 4-3. Midge taxa DCA analysis results.

CONCLUSION

Several studies have reported on invertebrates of lotic and lentic habitats. This paper is the most comprehensive to date on the distribution and abundance of emerging macroinvertebrates of palustrine wetlands in the Northwest. We feel confident that we have good descriptions of wetland emergent macroinvertebrates using the traps and conditions described from replicate captures among 19 wetlands during three years of survey between 1988 and 1995. Our descriptive statistical analysis of the high numbers of macroinvertebrates captured combined with estimates of variability among traps indicates that emergent trapping provides a good census of emerging aquatic insects in wetlands. Capture data further suggest that robust statistical comparisons of emergence data are possible by combining the three replicates at each site (Richter et. al. 1991). Nevertheless, increasing the number of replicates would be desirable and would provide additional power to our findings.

Our study is especially valuable in describing the chironomid midge communities. In North America, this group is represented by more species than all other orders of insects combined (McCafferty 1983). We identified 80 taxa in 1989 alone, including new species and extended the range extended extensions of several other taxa. Nearly half of the encountered taxa have not been previously reported in wetlands (Wrubeski 1987).

We identified 17 out of a total of 35 North American dipteran families associated with aquatic or semi-aquatic environments (McCafferty 1983) including several families not mentioned as found in marginal areas of shallow bodies of water including lakes, ponds, pools, marshes and bogs.

Non-dipteran aquatic and semi-aquatic insects identified within our survey were, for the most part, identified elsewhere in similar wetland ecosystems. The three taxa of dragonflies (Odonata) and *Callibaetis* (Ephemeroptera) are commonly found in Canadian marshes (Rosenberg and Danks 1987), whereas *Paraleptophlebia* is common in ephemeral streams of the Pacific Northwest. Plecoptera taxa found are also the ones typically inhabiting small perennial or temporary streams.

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APPENDIX 4-1. TABLES OF ARTHROPOD YIELD.

Table 4-2. Annual arthropod yield (per M2) from 19 wetlands in King County, Washington. Measured with three emergence traps (0.25 M2 area) at each wetland site run continuously for "insect year" 1988* (9/15/88 - 9/30/89, 1993 (4/10/93 - 4/9/94) & 1995 (1/1/95 - 10/30/95).

Wetland	Year	TOTAL TERR. ARTHROPODS	TOTAL AQUATIC AND SEMI-AQUATIC TAXA	Ephemeroptera	Odonata	Plecoptera	Trichoptera	Diptera	Brachycera	Nematocera	Non-Chironomid Nematocera	Chironomidae
	1989	41	4408	0	0	0	9	4399	2428	1971	724	1247
AL3	1993	9	2134	0	0	0	0	2134	184	1950	584	1366
	1995	31	1323	1	0	0	36	1286	225	1061	354	708
	1989	589	3037	0	0	0	1	3035	888	2147	634	1513
B3I	1993	222	2360	0	0	0	0	2360	1107	1253	616	637
	1995	277	735	0	0	0	0	735	168	567	460	108
	1989	56439	8858	3	24	1	132	8698	267	8431	203	8228
BBC24	1993	172	7515	0	0	0	0	7515	136	7379	145	7234
	1995	289	3020	1	0	0	15	3004	94	2910	93	2816
	1989	1972	7337	0	0	0	9	7328	1016	6312	3552	2760
ELS39	1993	122	4229	0	0	0	0	4229	404	3825	1970	1855
	1995	509	3204	0	0	0	5	3199	304	2895	2299	596
	1989	25935	20828	19	1	0	27	20781	5488	15293	3368	11925
ELS61	1993	597	10844	0	0	0	0	10844	3779	7065	2902	4163
	1995	452	1612	0	0	7	5	1600	133	1467	484	983
	1989	336	1238	0	0	0	0	1238	583	656	485	171
ELW1	1993	73	339	0	0	0	0	339	55	284	269	15
	1995	90	256	0	0	0	0	256	140	116	105	11
	1989	1531	4736	0	0	0	1	4734	1575	3160	461	2699
FC1	1993	115	6767	0	0	0	0	6767	73	6694	301	6393
	1995	113	2899	0	0	0	5	2894	15	2879	35	2844
	1989	308	8753	0	0	5	40	8708	2169	6538	3525	3013
HC13	1993	113	2272	0	0	0	0	2272	44	2228	270	1958
	1995	69	1522	0	0	1	21	1500	33	1467	435	1032
	1989	97	1134	105	0	32	3	994	69	925	169	756
JC28	1993	8	2900	0	0	0	0	2900	7	2893	53	2840
	1995	28	702	206	0	7	24	465	28	437	134	303
	1989	5076	9691	232	0	1576	61	7821	2925	4896	2580	2316
LCR93	1993	45	6234	0	0	0	0	6234	128	6106	193	5913
	1995	217	21501	0	0	0	0	21501	880	20621	1605	19016
	1989	2836	5126	0	0	0	3	5124	1160	3964	3313	651
LPS9	1993	153	1076	0	0	0	1	1075	432	643	504	139
	1995	62	2964	8	2	0	231	2723	86	2637	382	2255
	1989	8067	7365	7	0	0	35	7324	2884	4440	1072	3368
MGR36	1993	243	6699	0	0	0	1	6698	39	6659	226	6433
	1995	294	1606	0	0	3	23	1580	234	1346	944	402
	1989	74	8870	0	0	0	7	8863	2984	5879	1127	4752
NFIC12	1993	282	13047	0	0	0	0	13047	368	12679	1340	11340
	1995	169	2256	5	0	101	9	2141	952	1189	877	313
	1989	575	5892	11	0	0	36	5845	484	5361	440	4921
PC12	1993	119	5683	0	0	0	0	5683	288	5395	437	4958
	1995	149	3159	0	0	0	13	3146	902	2244	313	1931
	1989	945	8621	3	0	0	28	8591	884	7707	500	7207
RR5	1993	60	2413	0	0	0	1	2412	35	2377	132	2245
	1995	127	2067	0	0	42	5	2020	307	1713	1676	38
	1989	308	2952	0	0	5	12	2935	887	2048	1232	816
SC4	1993	104	2186	0	0	0	0	2186	626	1560	678	882
	1995	84	1696	0	0	0	12	1684	189	1495	367	1128
	1989	115	3692	3	0	0	0	3689	1377	2312	449	1863
SC84	1993	25	1106	0	0	0	0	1106	20	1086	181	904
	1995	45	642	0	0	0	1	641	6	635	77	558
	1989	2590	5598	24	0	0	27	5547	467	5080	815	4265
SR24	1993	37	2506	0	0	0	0	2506	12	2494	82	2412
	1995	29	662	0	0	0	1	661	76	585	96	489
	1989	85	4658	0	0	0	1	4656	1043	3614	407	3207
TC13	1993	36	2116	0	0	0	0	2116	9	2107	104	2003
	1995											

Table 4-3. Annual terrestrial arthropod yield (per M^2) from 19 wetlands in King County, Washington. Measured with three emergence traps (0.025 M^2 area) at each wetland, run continuously for "insect year", 1989* (9/15/88-9/31/89), 1993 (4/10/93-4/9/94) and 1995 (1/1/95-10/30/95).

			INSECTA														
Wet-land	Year	**Arach-nida**	Collem-bola	Thysanop-tera	Psocop-tera	Hemip-tera	Homop-tera	Non Aphididae	Aphid-idae	Neurop-tera	Coleop-tera	Lepidop-tera	Hymenop-tera	Other Hymenop-tera	Parasit-oid	Formic-idae	TOTAL TERR. ARTHRO-PODS
AL3	1989	7	3	0	11	0	3	0	1	0	11	2	5	0	5	0	41
	1993	3	0	0	0	0	0	0	0	0	0	1	5	0	5	0	9
	1995	1	4	1	8	0	0	0	0	0	1	1	13	0	13	0	31
B3I	1989	35	124	13	41	9	124	0	113	9	11	7	216	0	212	4	589
	1993	4	1	4	20	1	110	5	105	3	12	7	60	7	51	3	222
	1995	3	37	1	20	0	165	1	163	0	0	0	51	0	51	0	277
BBC24	1989	33	179	17	144	0	55823	0	55797	7	56	7	173	1	171	1	56439
	1993	5	20	0	3	0	3	1	1	1	88	3	49	0	49	0	172
	1995	14	68	1	17	0	3	2	1	0	119	8	59	0	59	0	289
ELS39	1989	67	1043	33	4	1	281	0	133	0	40	8	495	4	505	5	1972
	1993	5	13	1	7	1	36	9	27	0	1	1	56	0	55	1	122
	1995	11	144	0	77	0	140	8	132	0	40	3	94	2	92	0	509
ELS61	1989	19	223	7	68	3	25155	0	25213	4	143	5	309	12	325	0	25935
	1993	12	72	4	0	3	210	13	197	0	37	24	235	1	234	0	597
	1995	13	337	4	4	0	31	23	8	0	11	9	44	1	43	0	452
ELW1	1989	19	131	0	47	12	44	0	17	0	1	1	81	1	79	1	336
	1993	13	4	0	24	0	3	1	1	0	8	0	21	0	21	0	73
	1995	8	36	3	15	1	1	1	0	0	11	1	15	1	13	0	90
FC1	1989	73	768	41	0	0	457	3	316	1	15	0	175	0	172	3	1531
	1993	20	7	5	9	0	48	25	23	0	8	0	17	0	17	0	115
	1995	9	11	5	70	0	1	1	0	0	3	1	12	0	12	0	113
HC13	1989	31	23	1	27	11	35	0	13	1	13	1	165	1	161	3	308
	1993	9	3	4	4	0	68	0	68	0	16	0	9	0	9	0	113
	1995	5	3	1	39	1	1	0	1	0	3	0	16	0	16	0	69
JC28	1989	11	32	1	20	4	7	0	5	0	1	0	21	0	21	0	97
	1993	1	1	0	0	0	0	0	0	3	0	0	3	0	3	0	8
	1995	3	7	1	11	1	1	1	0	0	3	1	0	0	0	0	28
LCR93	1989	88	21	57	129	15	4219	0	4055	3	101	1	441	0	439	3	74
	1993	11	0	5	1	0	8	3	5	0	9	0	11	0	11	0	282
	1995	16	19	0	0	0	100	0	100	2	9	1	69	1	68	0	169
LPS9	1989	57	197	15	4	5	2140	1	2084	5	55	1	388	8	380	0	5076
	1993	20	15	0	0	1	18	5	13	0	5	0	94	5	89	0	45
	1995	4	13	2	3	6	0	0	0	0	6	0	28	0	28	0	217
MGR36	1989	49	85	36	41	5	7645	0	7607	5	51	1	147	0	145	1	2868
	1993	43	5	4	5	1	4	1	3	0	7	0	174	0	174	0	153
	1995	23	132	11	0	0	9	4	5	0	15	8	97	0	94	3	62
NFIC12	1989	7	8	5	11	0	11	0	9	0	3	3	27	3	24	0	8067
	1993	4	11	4	16	5	7	1	5	1	3	0	231	4	227	0	243
	1995	12	5	1	68	1	16	11	5	0	17	1	47	0	47	0	294
PC12	1989	25	213	13	7	0	169	0	25	0	39	6	103	0	134	1	575
	1993	32	13	8	0	0	11	4	7	0	16	1	37	0	36	1	119
	1995	3	11	1	3	0	55	55	0	0	7	24	47	3	44	0	149
RR5	1989	21	137	15	39	3	219	3	75	0	261	23	228	1	213	13	945
	1993	1	3	3	1	0	4	1	3	0	19	3	27	0	27	0	60
	1995	21	41	0	1	0	0	0	0	0	17	3	43	0	43	0	127
SC4	1989	64	53	7	1	13	19	1	3	0	40	3	108	4	97	13	308
	1993	1	28	3	3	1	15	14	1	1	7	15	30	1	29	0	104
	1995	8	12	0	35	0	23	23	0	0	0	0	7	0	7	0	84
SC84	1989	12	11	5	3	0	24	0	1	0	1	0	59	7	43	9	115
	1993	4	0	0	10	2	2	2	0	0	0	0	7	0	7	0	25
	1995	5	15	0	4	0	10	10	0	0	5	0	5	0	5	0	45
SR24	1989	3	31	15	3	0	2375	16	2207	0	9	0	155	1	152	1	2590
	1993	5	0	1	3	0	9	5	3	0	12	0	7	0	7	0	37
	1995	3	8	1	7	0	0	0	0	0	5	0	5	0	5	0	29
TC13	1989	11	17	8	3	0	1	0	1	0	7	0	39	0	39	0	85
	1993	9	3	7	1	0	3	1	1	0	4	0	9	0	9	0	36
	1995																

Table 4-4. Annual aquatic/semi-aquatic insect yield (per M^2) from 19 wetlands in King County, Washington. Measured with three emergence traps (0.025 M^2 area) at each wetland site, run continuously for "insect year" 1989* (9/15/88 - 9/31/89), 1993 (4/10/93 - 4/9/94).

		TAXON																					
Wetland	Year	Ephemeroptera	Odonata	Plecoptera	Trichoptera	Total Diptera/ Nematocera	Nematocera/ Chironomidae	Nematocera/ Non Chironomidae	Anisopodidae	Bibionidae	Cecidomyiidae	Ceratopogonidae	Chaoboridae	Culicidae	Dixidae	Misc. Nematocera	Mycetophilidae	Psychodidae	Scatopsidae	Sciaridae	Simuliidae	Tipulidae	Trichoceridae
AL3	1989	0	0	0	9	724			0	0	159	53	5	3	100	0	52	120	0	137	0	95	0
	1993	0	0	0	0	1950	584	1366	0	0	13	130	19	75	9	0	57	185	0	15	0	81	0
	1995	1	0	0	36	1061	354	708	0	0	5	25	72	13	19	0	4	21	0	9	0	185	0
B3I	1989	0	0	0	1	624			1	0	407	71	0	1	1	1	8	32	4	13	0	84	0
	1993	0	0	0	0	1253	616	637	1	0	246	1	0	0	5	0	76	16	0	21	0	249	0
	1995	0	0	0	0	567	460	108	0	0	301	3	0	0	0	0	9	5	0	19	0	122	0
BBC24	1989	3	24	1	132	203			0	0	11	121	11	1	33	15	0	1	0	9	0	0	0
	1993	0	0	0	0	7379	145	7234	0	0	3	25	35	39	28	0	0	8	0	5	0	3	0
	1995	1	0	0	15	2910	93	2816	0	0	5	15	33	8	7	0	4	9	0	11	0	1	0
ELS39	1989	0	0	0	9	3552			0	2	226	16	0	0	2	2	106	14	0	3152	0	32	0
	1993	0	0	0	0	3825	1970	1855	0	0	108	152	39	208	59	0	23	839	31	463	0	51	0
	1995	0	0	0	5	2895	2299	596	0	0	139	24	6	13	1	0	94	109	0	1906	0	7	0
ELS61	1989	19	1	0	27	3368			0	0	24	819	12	25	7	8	504	1449	1	336	0	183	0
	1993	0	0	0	0	7065	2902	4163	0	0	68	1886	0	37	20	0	218	315	0	153	0	205	0
	1995	0	0	7	5	1467	484	983	0	0	4	243	0	23	1	0	68	27	0	21	0	97	0
ELW1	1989	0	0	0	0	485			0	0	35	223	0	1	0	3	16	16	0	77	0	115	0
	1993	0	0	0	0	284	269	15	0	0	177	0	0	0	0	0	27	1	0	23	0	41	0
	1995	0	0	0	0	116	105	11	0	0	25	0	0	0	0	0	5	4	0	61	0	9	0
FC1	1989	0	0	0	1	461			1	0	32	48	0	53	116	11	40	120	1	5	0	33	0
	1993	0	0	0	0	6694	301	6393	0	0	1	29	0	33	181	0	0	52	0	3	0	1	0
	1995	0	0	0	5	2879	35	2844	0	0	3	3	0	0	18	0	0	3	0	8	0	1	0
HC13	1989	0	0	5	40	3520			3	0	104	116	0	1	7	8	39	2421	0	509	4	308	0
	1993	0	0	0	0	2228	270	1958	0	0	1	7	80	83	72	0	8	11	0	1	0	8	0
	1995	0	0	1	21	1467	435	1032	0	0	4	20	106	39	55	0	188	13	0	4	0	7	0
JC28	1989	105	0	32	3	169			0	0	1	65	0	0	7	11	3	55	0	11	0	16	1
	1993	0	0	0	0	2893	53	2840	0	0	8	0	0	0	29	0	8	5	0	1	0	1	0
	1995	206	0	7	24	437	134	303	0	0	11	19	0	1	15	0	16	15	0	40	0	19	0
LCR93	1989	232	0	1576	61	2579			31	0	241	764	0	0	191	229	177	367	0	129	152	297	0
	1993	4	0	0	9	6106	193	5913	0	0	11	21	0	1	48	0	31	0	0	3	25	53	0
	1995	0	0	0	0	20621	1605	19016	0	0	90	4	0	0	0	0	734	0	0	722	0	55	0
LPS9	1989	0	0	0	3	3313			4	0	1096	68	0	0	7	4	49	75	0	1528	0	483	0
	1993	0	0	0	1	643	504	139	0	0	130	9	0	0	1	0	82	1	0	266	0	15	0
	1995	8	2	0	231	2637	382	2255	0	0	7	289	0	25	22	0	4	13	0	21	0	0	0
MGR36	1989	7	0	0	35	1072			0	0	105	92	0	24	100	35	15	609	1	40	0	51	0
	1993	0	0	0	1	6659	226	6433	0	0	12	57	8	12	93	0	0	25	0	19	0	0	0
	1995	0	0	3	23	1346	944	402	0	0	57	44	0	0	0	0	41	1	0	497	0	303	0
NFIC12	1989	0	0	0	7	1127			0	0	352	72	4	15	1	0	21	145	0	489	0	27	0
	1993	0	0	0	0	12679	1340	11340	0	0	100	596	43	25	4	0	80	96	0	323	0	73	0
	1995	5	0	101	9	1189	877	313	0	0	138	67	0	0	20	0	67	61	0	448	4	72	0
PC12	1989	14	0	0	36	437			1	0	15	84	0	23	1	24	8	179	1	64	0	37	0
	1993	0	0	0	0	5395	437	4958	0	0	1	42	3	238	22	0	42	25	0	31	0	33	0
	1995	0	0	0	13	2244	313	1931	0	0	25	106	1	1	0	0	28	3	0	124	0	24	0
RR5	1989	3	0	0	28	500			0	0	11	241	7	0	0	16	16	101	0	75	0	33	0
	1993	0	0	0	1	2377	132	2245	0	0	3	114	3	3	0	0	3	0	0	4	0	3	0
	1995	0	0	42	5	1713	1676	38	0	0	91	3	0	161	0	0	55	15	0	1291	0	60	0
SC4	1989	0	0	5	12	1232			0	0	299	3	0	16	0	3	7	117	0	728	0	60	0
	1993	0	0	0	0	1560	678	882	0	0	122	37	0	120	15	0	57	131	0	120	0	75	0
	1995	0	0	0	12	1495	367	1128	0	0	4	5	133	71	112	0	1	28	0	9	0	4	0
SC84	1989	3	0	0	0	449			0	0	65	33	7	1	12	4	9	84	0	79	0	155	0
	1993	0	0	0	0	1086	181	904	0	0	9	0	80	18	61	0	2	10	0	2	0	0	0
	1995	0	0	0	1	635	77	558	0	0	11	6	11	6	3	0	22	10	0	7	0	0	0
SR24	1989	24	0	0	27	815			0	0	107	189	7	19	48	132	163	84	0	27	0	40	0
	1993	0	0	0	0	2494	82	2412	0	0	0	26	32	3	15	0	0	0	0	4	0	2	0
	1995	0	0	0	1	585	96	489	0	0	7	4	41	1	4	0	7	8	0	5	0	19	0
TC13	1989	0	0	0	1	407			0	0	88	84	1	0	5	3	11	73	1	89	0	51	0
	1993	0	0	0	0	2107	104	2003	0	0	7	39	17	8	24	0	0	5	0	4	0	0	0
	1995																						

Table 4-5. Annual aquatic/semi-aquatic insect yield (per M^2) from 19 wetlands in King County, Washington, as measured with 3 emergence traps (0.25 M^2 area) at each wetland site, run continuously between September 1988 and September 1989.

	AQUATIC AND SEMIAQUATIC INSECTS																		
	Non-Urbanized								Moderately Urbanized							Highly Urbanized			
	Perennial					Dry in Summer			Perrenial			Dry in Summer						Perennial	
TAXON	Mgr36	RR5	HC13	SR24	PC12	AL3	NFIC12	TC13	ELW1	BB24	ELS61	LCR93	LPS9	SC4	JC28	SC84	ELS39	B3I	FC1
Ephemeroptera	7	3	0	24	14	0	0	0	0	3	19	232	0	0	105	3	0	0	0
Callibaetis	0	3	0	24	13	0	0	0	0	3	19	0	0	0	0	3	0	0	0
Paraleptophlebia	7	0	0	0	1	0	0	0	0	0	0	232	0	0	105	0	0	0	0
Odonata	0	0	0	0	0	0	0	0	0	24	1	0	0	0	0	0	0	0	0
Ischnura cervula	0	0	0	0	0	0	0	0	0	20	1	0	0	0	0	0	0	0	0
Enallagma boreale	0	0	0	0	0	0	0	0	0	3	0	0	0	0	0	0	0	0	0
Coenagrion	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0
Plecoptera	0	0	5	0	0	0	0	0	0	1	0	1576	0	5	32	0	0	0	0
Capnia nr. oregona	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
Paraleuctra? vershina	0	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Malenka	0	0	0	0	0	0	0	0	0	0	0	83	0	0	0	0	0	0	0
Ostracerca dimicki	0	0	0	0	0	0	0	0	0	0	0	1	0	4	0	0	0	0	0
Podmosta delicatula	0	0	1	0	0	0	0	0	0	1	0	1325	0	1	1	0	0	0	0
Soyedina interrupta	0	0	0	0	0	0	0	0	0	0	0	137	0	0	31	0	0	0	0
Zapada cinctipes	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
Taenionema	0	0	0	0	0	0	0	0	0	0	0	27	0	0	0	0	0	0	0
Trichoptera	35	28	40	27	36	9	7	1	0	132	27	61	3	12	3	0	9	1	1
Unk. Trichoptera	0	0	0	0	6	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Hydroptila	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Oxyethira	0	1	0	0	0	0	0	0	0	112	0	0	0	0	0	0	0	0	0
Lepidostoma	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3	0	0	0	1
Lepidostoma cinereum	0	0	0	0	0	0	0	0	0	0	0	3	0	0	0	0	0	0	0
Clistoronia	16	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Clostoeca disjuncta	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	0	0	0	0
Glyphopsyche irrorata	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0
Halesochila taylori	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lenarchus rho	0	4	0	1	1	0	3	0	0	0	3	0	0	0	0	0	0	0	0
Lenarchus vastus	0	12	36	0	5	9	3	0	0	0	0	4	0	0	0	0	4	0	0

Table 4-6. Diversity and richness of aquatic/semi-aquatic arthropod and Chironomidae communities found at 19 wetland sites in King County, Washington.

Diversity and Richness: All Aquatic/Semi-Aquatic Taxa Except Brachycera

	Non-Urbanized							Moderately Urbanized										Highly Urbanized	
	Perennial				Dry in Summer			Perennial		Dry in Summer								Perennial	
Wetland	MGR36	RR5	HC13	SR24	PC12	AL3	NFIC12	TC13	ELW1	BBC24	ELS61	LCR93	LPS9	SC4	JC28	SC84	ELS39	B3I	FC1
Taxa richness	40	52	40	47	52	21	20	27	27	62	47	62	22	29	39	29	21	31	34
Annual adult yield per m2	4433	7732	6585	5082	5407	1852	5884	3614	655	8570	15340	6761	3969	2065	1064	2307	6332	2139	3159
Shannon Diversity Index (log 2	4.27	2.76	3.04	4.13	3.83	3.1	2.27	2.73	3.12	3.72	2.6	4.2	2.48	2.58	3.53	3.19	1.79	3.08	2.98
Pielou Evenness Index	0.803	0.48	0.57	0.74	0.67	0.71	0.53	0.57	0.66	0.624	0.467	0.71	0.56	0.53	0.67	0.66	0.41	0.62	0.59

Diversity and Richness: All Chironomidae Taxa

Wetland	MGR36	RR5	HC13	SR24	PC12	AL3	NFIC12	TC13	ELW1	BBC24	ELS61	LCR93	LPS9	SC4	JC28	SC84	ELS39	B3I	FC1
Taxa richness	28	38	23	34	35	11	8	16	19	42	29	32	11	16	26	19	9	19	22
Annual adult yield per m2	3368	7207	3013	4265	4921	1247	4752	3207	171	8228	11925	2316	651	816	756	1863	2760	1513	2699
Shannon Diversity Index (log 2	3.83	2.4	2.54	3.56	3.44	1.67	1.44	2.18	2.88	3.49	1.7	3.07	1.79	1.4	2.66	2.45	0.87	2.42	2.31
Pielou Evenness Index	0.798	0.46	0.56	0.7	0.67	0.48	0.48	0.54	0.68	0.648	0.35	0.62	0.52	0.35	0.57	0.58	0.28	0.57	0.52

versity and Richness: Chironomidae Taxa without Unidentified Females

Wetland	MGR36	RR5	HC13	SR24	PC12	AL3	NFIC12	TC13	ELW1	BBC24	ELS61	LCR93	LPS9	SC4	JC28	SC84	ELS39	B3I	FC1
Taxa richness	24	34	19	30	31	8	6	13	15	38	26	28	10	13	22	16	7	16	18
Annual adult yield per m2	1726	1989	1062	1905	2993	215	1175	1172	56	4099	3000	861	226	212	379	864	608	777	689
Shannon Diversity Index (log 2	3.71	3.88	3.06	3.61	2.99	1.77	1.2	2.2	3.4	3.85	2.48	3.64	2.58	1.53	2.64	1.94	0.44	2.65	2.67
Pielou Evenness Index	0.809	0.76	0.72	0.74	0.6	0.59	0.46	0.59	0.87	0.734	0.528	0.76	0.78	0.41	0.59	0.49	0.16	0.66	0.64

Table 4-7. Annual adult Chironomidae yield (per M^2) from 19 wetlands in King County, Washington, as measured with 3 emergence traps (0.25 M^2 area) at each wetland site, run continuously between September 1988 and September 1989.

	CHIRONOMIDAE TAXA																	Highly	
			Non-Urbanized								Moderately Urbanized							Urbanized	
	Perennial					Dry in Summer			Perennial			Dry in Summer						Perennial	
TAXON	MGR36	RR5	HC13	SR24	PC12	AL3	NFIC12	TC13	ELW1	BBC24	ELS61	LCR93	LPS9	SC4	JC28	SC84	ELS39	B3I	FC1
Diptera/Chironomidae																			
Boreochlus	145	0	8	0	0	0	0	0	0	0	0	0	0	3	0	0	0	0	0
Total Podonominae	145	0	8	0	0	0	0	0	0	0	0	0	0	3	0	0	0	0	0
Odontomesa	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4	0	0	7	0
Prodiamesa	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	169	0
Total Prodiamesinae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4	0	0	176	0
Brillia	15	0	0	0	1	0	0	0	9	5	5	9	0	0	7	0	0	73	1
Chaetocladius	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
Corynoneura	161	40	35	523	83	0	0	1	0	40	11	37	0	0	17	0	0	0	24
Cricotopus	0	0	0	65	24	0	0	0	1	25	0	0	0	0	0	0	0	63	0
Cricotopus bifurcatus	0	0	0	0	0	0	0	0	3	0	0	0	0	0	0	0	0	0	0
Doithrix	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	4
Limnophyes	103	531	261	36	51	124	849	167	7	12	799	289	67	160	152	63	574	12	133
Mesosmittia	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0
Metriocnemus	0	0	0	17	0	0	0	0	4	1	13	13	21	1	0	0	4	75	3
Nanocladius	0	43	0	0	0	0	0	0	0	8	0	39	0	0	1	0	0	0	0
Orthocladius	0	83	1	36	0	0	0	0	0	21	0	29	0	0	8	3	0	0	0
Parakiefferiella	0	0	0	0	0	0	0	0	1	0	0	3	0	3	0	0	0	0	0
Parametriocnemus	0	188	24	16	3	0	0	7	1	0	1439	29	7	1	11	0	0	1	0
Paraphaenocladius	17	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1	0
Poryophaenocladius	0	0	0	0	0	0	0	0	0	5	0	0	0	0	0	0	0	0	0
Psectrocladius	1	285	0	11	32	0	0	15	0	21	15	0	0	0	0	0	0	0	0
Pseudosmittia	0	29	273	0	0	0	1	385	0	0	71	4	9	7	5	425	8	1	4
Rheocricotopus	0	0	3	0	1	0	0	0	0	0	1	29	43	0	1	1	0	0	0
Smittia	0	0	111	0	0	12	121	0	0	0	9	19	55	17	0	5	4	0	0
Thienemanniella	39	0	0	0	17	0	0	8	0	44	0	7	0	0	0	1	0	0	0
Orthocladiinae m.	3	73	5	5	17	0	0	3	1	0	4	53	19	5	4	0	2	253	0
Orthocladiinae fm.	657	4612	1632	877	512	716	3240	1685	77	119	8429	1083	423	583	325	721	2148	727	1395
Total Orthocladiinae	1004	5884	2345	1587	741	852	4213	2271	105	303	10796	1644	643	779	532	1221	2740	1208	1564
Chironomus decorus gr.	111	64	87	19	13	52	188	229	0	29	16	1	0	8	1	281	4	0	267
Chironomus riparius	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
Cladopelma viridula	0	20	0	7	1	0	0	0	0	68	0	0	0	0	0	0	0	1	0
Demicryptochironomus nr	0	0	0	5	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Dicrotendipes	0	4	0	8	11	0	0	0	0	47	27	0	0	0	0	3	0	0	0
Endochironomus nigrican	0	5	0	56	24	0	0	0	0	19	71	3	0	0	0	1	0	0	0
Endochironomus subtend	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Glyptotendipes	0	1	0	79	5	0	0	0	0	104	0	0	0	0	0	0	0	0	0
Kiefferulus dux	0	0	48	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Microtendipes pedellus va	12	0	0	0	0	0	0	0	0	113	0	0	0	0	0	0	0	0	0
Microtendipes pedellus va	0	28	0	0	0	0	0	0	0	343	0	0	0	0	0	0	0	0	0
Parachironomus monochr	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Parachironomus cf. force	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Parachironomus sp. 1	0	44	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Parachironomus sp. 2	0	47	0	15	0	0	0	0	0	7	1	5	0	0	0	0	0	0	0
Paratendipes	0	0	0	0	0	0	0	0	0	0	0	8	0	0	21	0	0	0	0
Paratendipes albimanus	3	0	0	0	0	0	0	0	0	8	0	0	0	0	0	0	0	0	0
Phaenopsectra flavipes	3	7	0	0	17	0	0	0	0	5	0	0	0	0	1	3	0	0	3
Phaenopsectra punctipes	0	0	1	0	0	0	0	0	0	4	3	0	0	0	0	0	0	0	0
Polypedilum illinoense	0	4	0	0	0	0	0	0	11	13	19	0	0	0	0	0	0	0	0
Polypedilum ophioides	0	0	0	0	0	0	0	0	0	13	0	0	0	0	0	0	0	0	0
Polypedilum cf. simulans	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
Polypedilum gr. 1	105	109	95	128	851	16	15	7	4	864	128	28	1	4	13	19	0	0	55
Polypedilum gr. 2	0	71	0	65	61	0	0	1	0	419	72	0	3	0	0	0	0	0	0
Stictochironomus	0	48	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	3
Xestochironomus	0	0	0	0	0	0	0	0	0	5	0	0	0	0	0	0	0	0	0
Unk. Chironomini genus	0	0	0	0	0	0	0	0	0	0	0	4	1	0	0	0	0	0	0
Chironomini m.	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Chironomini fm.	253	400	232	540	680	184	336	347	23	1420	381	116	3	17	43	220	4	4	528
Total Chironomini	487	856	463	927	1668	253	539	584	39	3481	717	165	8	29	81	527	8	5	856

Table 4-7 Continued.

TAXON	MGR36	RR5	HC13	SR24	PC12	AL3	NFIC12	TC13	ELW1	BBC24	ELS61	LCR93	LPS9	SC4	JC28	SC84	ELS39	B3I	FC1
Chironomidae cont.																			
Ablabesmyia	57	33	0	335	35	0	0	0	0	275	68	1	0	0	0	0	0	0	11
Apsectrotanypus algens	129	3	0	0	0	0	0	0	0	80	0	15	0	0	117	0	0	0	0
Conchapelopia cf. currani	15	1	0	0	1	0	0	0	0	51	0	0	0	0	0	0	0	3	0
Conchapelopia dusena	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	112	8
Djalmabatista	0	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Hayesumyia serata	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
Labrundinia	0	0	0	0	0	0	0	0	0	75	0	0	0	0	0	0	0	0	0
Larsia	0	51	1	36	5	0	0	0	0	0	0	0	0	0	0	0	0	0	3
Meropelopia nr. american	0	0	0	0	0	0	0	0	7	0	0	0	0	0	0	0	0	0	0
Natarsia miripes	0	0	0	0	0	0	0	0	0	0	0	111	0	0	3	0	0	0	0
Procladius bellus	0	59	0	8	5	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Procladius nr. freemani	0	31	0	88	661	0	0	0	0	96	3	0	0	0	0	0	0	0	0
Procladius nr. sublettei	16	1	0	11	11	0	0	0	0	141	0	0	0	0	0	0	0	0	0
Procladius n. sp.?	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Paramerina smithae	161	7	0	12	3	0	0	0	0	21	4	19	0	0	3	0	0	0	0
Psectrotanypus dyari	77	1	11	137	656	5	0	345	4	96	28	4	0	0	0	12	0	0	111
Tanypus cf. parastellatus	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Zavrelimyia fastuosa	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0
Zavrelimyia sinuosa	0	0	0	0	0	0	0	0	0	0	61	0	0	0	0	0	0	0	0
Zavrelimyia thryptica	133	0	5	0	35	4	0	0	0	5	104	9	0	1	4	1	0	0	13
Macropelopiini m.	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
Macropelopiini fm.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4	0
Pentaneurini m.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Pentaneurini fm.	283	64	13	75	33	4	0	0	1	19	7	113	0	3	3	0	0	7	9
Total Tanypodinae	875	253	32	703	1449	13	0	345	13	859	275	272	0	4	131	13	0	127	156
Micropsectra gr. 1	384	0	75	19	257	0	0	1	1	67	12	21	0	0	0	4	12	0	44
Micropsectra gr. 2	16	19	17	1	84	0	0	3	0	149	16	43	0	0	1	41	0	0	0
Rheotanytarsus	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
Tanytarsus	17	52	0	161	20	0	0	0	0	800	0	29	0	0	3	0	0	0	0
Tanytarsini fm.	407	141	76	827	701	0	0	3	12	2552	108	141	0	1	4	51	0	0	77
Total Tanytarsini	824	212	168	1008	1063	1	0	7	13	3588	136	235	0	1	8	96	12	0	121

CHAPTER 5 AMPHIBIAN DISTRIBUTION, ABUNDANCE AND HABITAT USE

by Klaus O. Richter and Amanda L. Azous

INTRODUCTION

Amphibians are a diverse vertebrate class in forests, wetlands and undisturbed areas. Although their role in ecosystem dynamics has not been intensively studied, their potential abundance suggest significant roles in energy transfers and nutrient cycling. Burton and Likens (1975) in the Hubbard Brook Experimental Forest of New Hampshire found salamander (primarily Plethodon cinereus) numbers to regularly exceed 2,000 per hectare, with concomitant biomass at 1.65 kg per hectare, equaling that of small mammals and twice that of birds. Amphibians also reduce eutrophication of wetlands by their net export of nitrogen. At some wetlands the nitrogen in tadpoles is more than double that of residual pond nitrogen. Furthermore, amphibians in some wetlands (e.g., Rana pipiens, R. catesbeiana, and Ambystoma spp.) collectively export six to 12 times more nitrogen from the ponds than imported through spawning by breeding adults (Seale 1980). Finally, tadpoles also reduce the biomass of nitrogen-fixing blue-green algae and primary production by feeding on all forms of algae (Seale 1980, Beebee 1996).

Some King County wetlands are used by breeding western toads (Bufo boreas) and red-legged frogs (Rana aurora) that produce thousands of eggs and larvae and hundreds of metamorphs and juveniles. At these sites algae-grazing frog and toad tadpoles may significantly influence water nutrient and energy dynamics and provide food for larger aquatic invertebrates and fish. Although salamanders spawn fewer eggs than frogs and toads, their invertebrate-eating larvae also play important roles in aquatic composition and predator-prey relationships. Hundreds of metamorphs are pivotal in transferring biomass from wetland to adjacent terrestrial systems and become prey for reptiles, birds and mammals.

Along with our recognition of the increasing ecological importance of amphibians, studies have shown significant decreases of some populations and extinctions of others (Corn 1994). These declines, however, have been difficult to document because of inadequate information on the geographic distributions and abundances of populations.

The occurrence of Northwest amphibians noted on range maps (Leonard et al. 1993) and spot maps (Nussbaum et al. 1983) indicates a potential of 14 species in King County, 12 of which are associated with aquatic environments and 10 particularly with marshes, swamps, bogs and other wetlands. Recently, we (Richter and Azous 1995) sighted 10 species (e.g., seven lentic-breeding, one lotic-breeding and two terrestrial-breeding) during a two-year survey of 19 wetlands in the Puget Sound Basin. Furthermore, we reported that their distribution was unrelated to wetland characteristics of size, vegetation classes, presence of vertebrate predators and water permanence. Correspondingly, from our watershed land cover analysis we found that larger water level fluctuations resulting from higher impervious areas in highly urbanized watersheds accounted for decreasing species richness.

This paper describes the geographic distribution and relative abundance of amphibians within these 19 palustrine wetlands after an additional two years of surveys in 1993, and

1995. Its companion paper, Chapter 12, reports on the effects of watershed development and habitat conditions on amphibian populations within these wetlands.

METHODS

Information about the locations and physical, hydrologic, chemical and vegetative conditions found in the study wetlands is presented in Section 1 and Chapters 1, 2, and 3 of Section 2.

We determined the distribution of amphibians primarily by autumn pitfall trapping when amphibians are more active than during the summer, and during which time animals migrate to winter hibernacula,. Egg mass sightings, aquatic funnel trapping and fortuitous observations by knowledgeable biologists at the sites for other monitoring purposes augment our distribution data. Relative abundances of trapable species (no Pacific treefrogs) were determined from the results of 14-day autumn pitfall trapping surveys standardized for equal trap nights and for favorable climatic conditions such as temperatures above 4°C (Beebee 1996). Site selection and trap installation procedures are described in Richter and Azous (1995). Trapped amphibians were identified to species and released.

Spring egg surveys were used to determine amphibian breeding in wetlands. Briefly, these included February through April searches of shoreline to 1-m deep palustrine aquatic bed (PAB) and shoreline palustrine scrub-shrub (PSS), palustrine emergent (PEM) and palustrine forested (PFO) habitat types. Detailed survey descriptions are provided in Richter and Roughgarden (1995). We also captured some species in aquatic funnel traps (Richter 1995) within some wetlands to augment diversity data.

We determined wetland boundaries, wetland size, habitat types and land cover conditions within the wetland's watershed and within select distances of each wetland. This data was obtained from King County's Wetlands Inventory, King County Surface Water Management Division's GIS system , and the 1992 Landsat Thematic Mapper for the Puget Sound Region (King County 1990, Puget Sound Regional Council 1994). From Landsat images we identified and characterized ten cover types: 1) impervious surfaces, 2) freeway/parking/gravel areas, 3) cleared land, 4) grasslands/golf courses, 5) multi-family housing, 6) single family residential, 7) single family forest, 8) agriculture/pasture lands, 9) forests, and 10) open water. These were collapsed into favorable amphibian breeding, feeding, migration and hibernation habitat (cover types 7-10) and unfavorable types (cover types 1-6).

We identified habitat structure categories (e.g., aquatic bed, herbs, shrubs and trees) according to Cowardin et al. (1979) from aerial photos recorded on maps (King County 1987), and refined those designations with field surveys that sampled vegetation along transects that crossed the hydrologic gradients represented in the wetlands. Life history characteristics discussed in the text were taken from Nussbaum et al. (1983) and our own observations (Richter and Roughgarden 1995, Richter 1996a, Richter 1996b).

RESULTS

Ten amphibian species, representing all but one (spotted frog) of the regional amphibian fauna, were identified at the 19 wetlands studied. Eight amphibian species was the highest richness found (at SR24) and included the introduced bullfrog. Seven species, the greatest number of native species at a wetland, and representing 70% of the total potential native amphibian species, were identified at HC13, PC12 and SR24 east of Lake Sammamish. ELW1 had only one species captured, the bullfrog. Most wetlands exhibited five (50%) of the total potential native species. The most urbanized and isolated wetlands (B3I, FC1 and ELW1) had the lowest richness. Unexpected, however was the low richness at TC13 and RR5, relatively large wetlands in watersheds without extensive development. The proportional distribution of native amphibian richness within all wetlands is provided in Figure 5-1.

Sighted at 18 out of 19 wetland, the Pacific treefrog is likely the most broadly distributed amphibian (Table 5-1). Red-legged frogs, Northwestern salamanders, and long-toed salamanders were found in 16, 15 and 13 (84%, 79% and 68%respectively) of the wetlands surveyed. Of the two terrestrial-breeding buffer species Ensatina was found in 11 (58%) and Western red-backed salamanders in nine (47%) of wetlands.

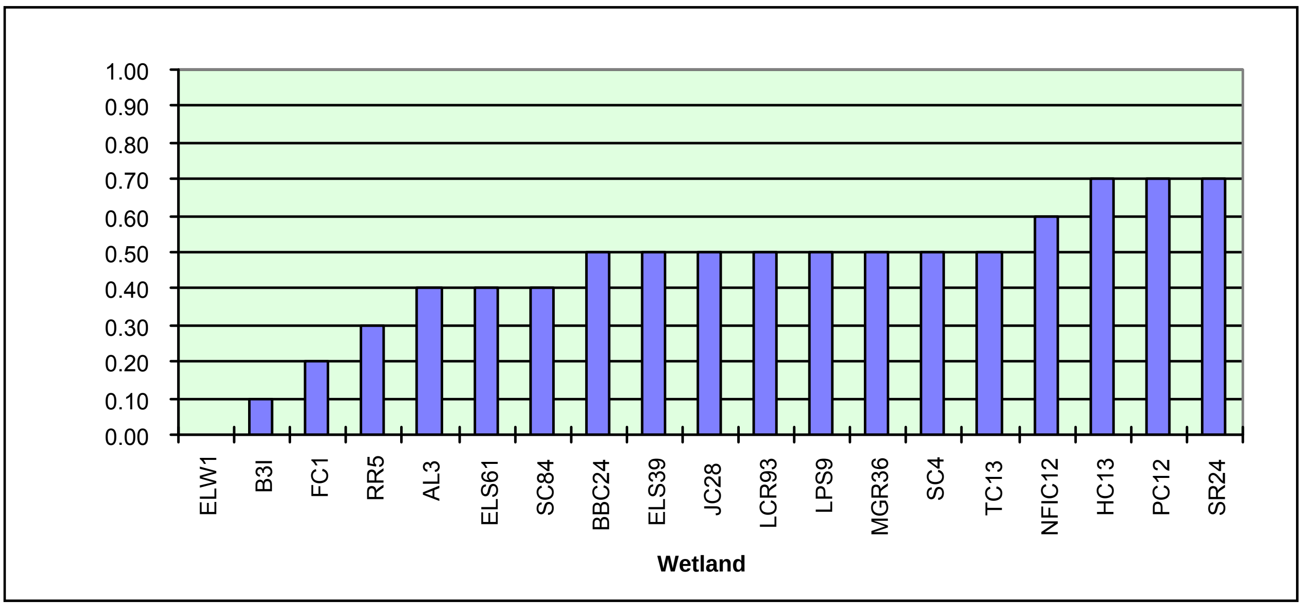

Figure 5-1. Proportional distribution of native amphibian richness within wetlands.

Table 5-1. Total amphibian fauna found in palustrine wetlands of the Puget Sound Basin.

Common Name	Scientific Name	Ames Lake 3	Bellevue 3I	Big Bear Creek 24	East Lake Sammamish 39	East Lake Sammamish 61	East Lake Washington 1	Forbes Creek 1	Harris Creek 13	Jenkin's Creek 28	Lower Cedar River 93	Lower Puget Sound 9	Middle Green River 24	North Fork Issaquah Creek 12	Patterson Creek 12	Raging River 5	Snoqualmie River 24	Soos Creek 4	Soos Creek 84	Tuck Creek 13	Percent of All Wetlands Species Was Present
Bullfrog	*Rana catesbeiana*			●		●	●	●	●					●			●			●	0.42
Ensatina	*Ensatina eschscholtzii*	●		●	●			●	●	●	●			●	●		●	●			0.58
Long-toed Salamander	*Ambystoma macrodactylu*	●	●		●	●		●	●		●	●	●		●		●		●	●	0.68
Northwestern Salamande	*Ambystoma gracile*			●	●	●			●	●	●	●	●	●	●	●	●	●	●	●	0.79
Pacific Giant Salamander	*Dicamptodon tenebrosus*														●		●				0.11
Pacific Treefrog	*Pseudacris regilla*	●	●	●	●	●		●	●	●	●	●	●	●	●	●	●	●	●	●	0.95
Red-legged Frog	*Rana aurora*	●		●	●	●			●	●	●	●	●	●	●	●	●	●	●	●	0.84
Roughskin Newt	*Taricha granulosa*			●		●				●											0.16
Western Red-backed Sa	*Plethodon vehiculum*	●		●					●	●	●	●	●					●	●		0.47
Western Toad	*Bufo boreas*								●						●	●	●				0.21

Spawn of the eggs of four species with large and readily identifiable eggs (Northwestern salamander, long-toed salamander, red-legged frog and Pacific treefrog) were identified at four wetlands, confirming breeding by these species at these sites. In contrast, eggs of the Western toad were not observed at any wetland, although metamorphs were sighted at BBC24 and RR5, corroborating that these wetlands are used by breeding toads. Though historically considered wide-spread (Nussbaum et al. 1983) roughskin newts were sighted in only three (16%) wetlands.

Lentic breeding species, as expected, were largely absent from wetlands with higher current velocities and channelized flows to which they are not well suited. High current velocity and water level fluctuations may thwart successful spawning, embryogenesis or larval survival of lentic breeding species. However, one lotic-breeding species, the Pacific giant salamander, was captured at PC12. Presumably, this animal was spawned in adjoining Patterson Creek, a cool, fast-running stream, similar to ones in which this species traditionally breeds.

We did not find spotted frogs, a native species. Historically never abundant in the Puget Sound Basin (McAllister and Leonard 1990, McAllister and Leonard 1991) spotted frogs were, nevertheless, expected at remote and undisturbed wetlands such as LCR93, MGR36, SR24 and RR5.

Bullfrogs were identified in several wetlands and in several drainages including Lake Sammamish (ELS61, NFIC12), Bear Creek (BBC24), Snoqualmie River (SR24), Tuck Creek (TC13), East Lake Washington (ELW1) and Harris Creek (HC13) drainages. Green frogs, another introduced species known to be in King County, were not seen within our wetlands. No native amphibians were captured in ELW1 although Pacific treefrogs were heard vocalizing. Although adult red-legged frogs were captured in pitfalls at AL3, neither spawn nor juveniles were observed during spring egg searches and summer site visits.

There were significant differences between the abundance of species captured within wetlands between 1988 and 1995. 1988 and 1989 were ranked similarly with average capture rates of 2.8 and 4.1 individuals per 100 trap nights respectively but differed significantly from 1993 and 1995 in which average capture rates were 0.8 and 1.5, respectively (Friedman test, $\chi 2 = 19.6$, $p = .0002$). Over the study period, the number of amphibian captures per 100 trap nights declined in 12 of the 19 wetlands. Six wetlands showed the highest capture rates in 1989 and then declined. Only one wetland, SC84, showed a slight 0.3 increase in capture rate between 1988 and 1995 (Figure 5-2).

Overall, the most abundant amphibian captured in pitfall traps was the red-legged frog, with particularly high capture rates in 1988 and 1989. Long-toed salamanders, Northwestern salamanders and Western red-backed salamanders were also numerically important. Capture rates of individual species in wetlands for each study year ranged from a high of 9.7, representing 29 Northwestern salamander captured in one night at one wetland, BBC24, in 1989 to the most frequent capture rate of 0.33, representing one individual of one species captured in a wetland for one year's trapping period. Captures of the same species in different years was unpredictable. The number of captures per 100 trap nights, summarized for each species across all wetlands in Figure 5-3, varied but statistical significance could not be evaluated due to the low number of captures. With the exception of Northwestern salamander, long-toed salamander, Ensatina and red-legged frog, species capture rates were 2 individuals or fewer most years. Appendix Table 5-1 gives the capture rates of individual species for each study year in individual wetlands.

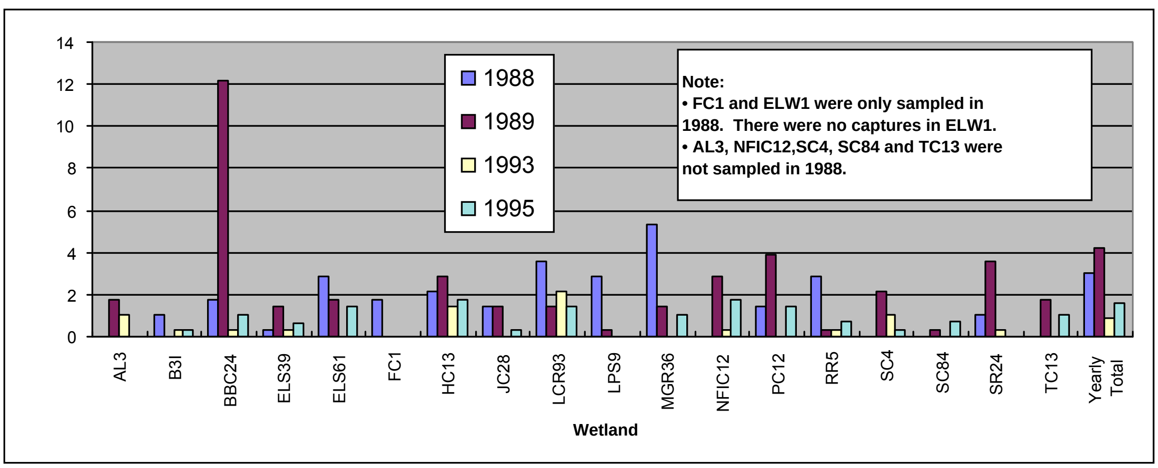

Figure 5-2. The number of total amphibian captures per 100 trap nights by wetland for each year of the study.

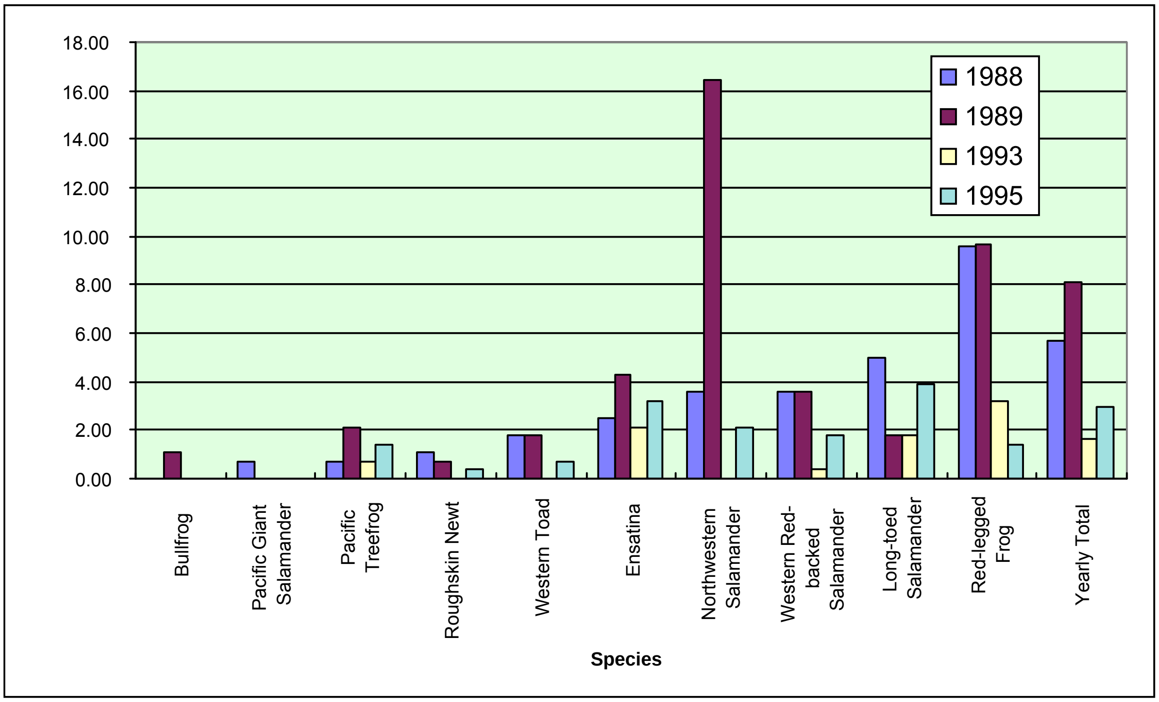

Figure 5-3. Number of captures per 100 trap nights for each species.

Land use in the watersheds of wetlands was related to amphibian richness. Wetlands with contributing watersheds in which more than 40 % of the land area was developed (usually housing with some commercial developments) were significantly more likely to have low amphibian richness of less than four species than wetlands within less urbanized watersheds, (Figure 5-4) ($\chi 2$, $P < 0.01$). Three wetlands with the highest native amphibian richness of more than 60% of all species observed, had very low watershed urbanization (less than 5%). Thirteen wetlands with medium amphibian richness of 40% to 60% of all species observed had urbanization ranging up to 90%. Three of the five wetlands with the highest urbanization had four or fewer species.

Since land use within the watershed wetland would directly affect hydrologic patterns in a wetland, we also evaluated whether minimum water levels, maximum water levels or the average range of fluctuation affected the richness of amphibian communities. Only average water level fluctuation (WLF) showed a statistically significant relationship with amphibian richness. When average WLF was 20 cm or more during the year, the number of amphibian species averaged three or fewer. Wetlands with lower WLFs (less than 20 cm) were significantly more likely to have a higher proportion of the potential amphibian richness, averaging five species (Mann Whitney, $p = 0.047$) (Figure 5-5).

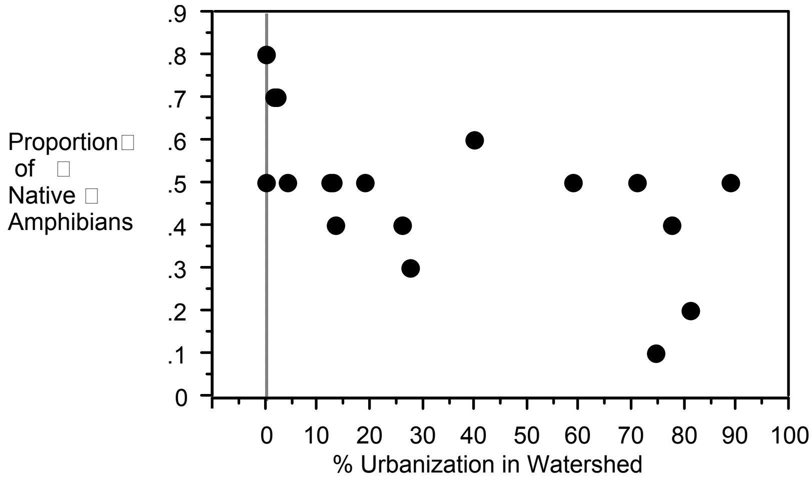

Figure 5-4. Relationship between the percent of native amphibian species present and percent of watershed urbanization.

Land use adjacent to a wetland was also found to be related to the richness of native amphibian populations. When land use within concentric areas of 10, 100, 500 and 1000 meters from the wetland were examined for statistically significant relations with amphibian richness we found that, within the distance encompassed by the 10 to 1000 M radii, amphibian richness was related to the percentage of favorable land available. Figure 5-6 shows the proportion of native species observed related to the percent of forest land within 10, 100, 500 and 1000 M of the wetland edge. In general, those wetlands which are adjacent to a high percentage of forest land were more likely to have richer populations of native amphibians. The significance of this relationship was weakest at 10 M ($R = 0.57$, $p = 0.01$) and strongest at 500 M ($R = 0.66$, $P = 0.004$). The graph shows that almost all wetlands had high proportions of forest land within 10 M and to a lesser extent at 100 M. But amphibian richness is highest in wetlands that retain at least 60% of adjacent area in forest land up to and exceeding 500 M. from the wetland and lowest in the wetlands that had a high proportion of forest land within 10 or 100 M but dropped significantly at 500 M and further from the wetland.

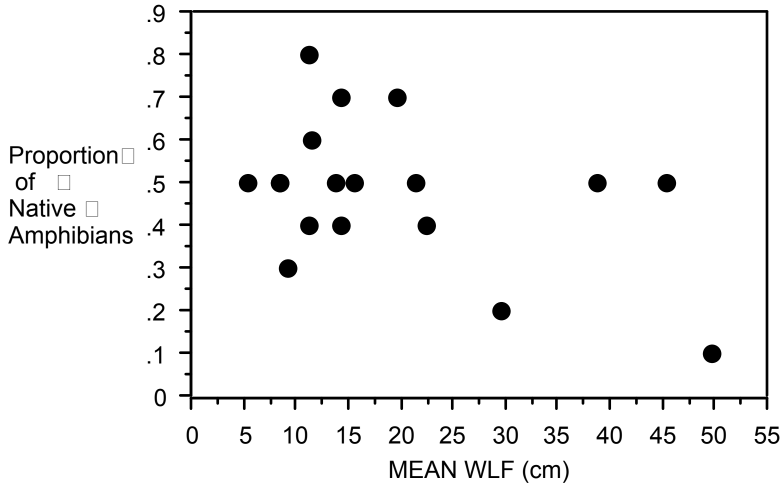

Figure 5-5. Relationship between the percent of possible amphibian species and average water level fluctuation.

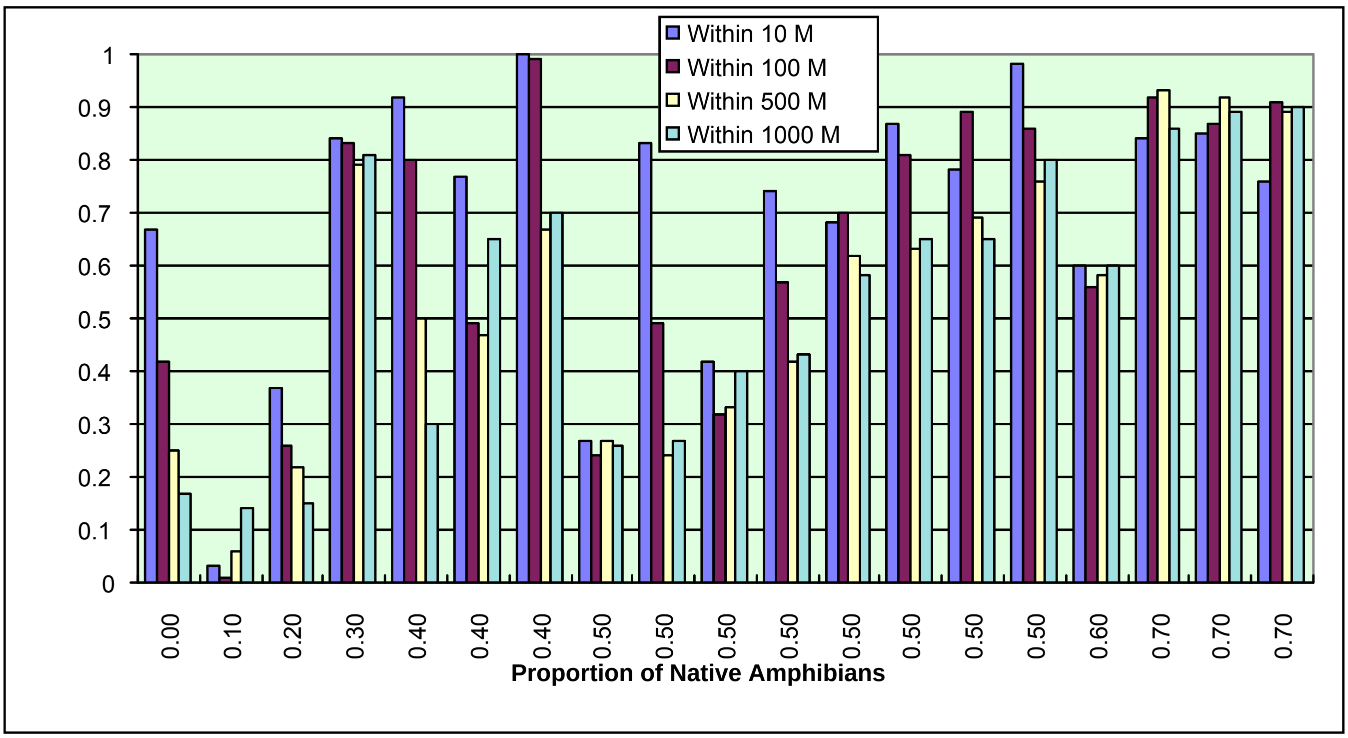

Figure 5-6. Plot of amphibian richness and the percent of favorable adjacent land.

DISCUSSION

Despite the low overall richness of amphibians within Puget Sound lowland palustrine wetlands when compared to the southeast (Gibbons and Semlitsch 1991) and central states (Clarke 1958, Clawson and Baskett 1982), the biomass of existing species may be high. The capture of 29 Northwestern salamander at one wetland on one night clearly shows the numerical importance of this salamander, and underscores the ecological significance of amphibians in general, moreover, it demonstrates the abundance of this species at a site which, when censused on other nights, would falsely suggest fewer individuals. Other species are also likely to be significantly more abundant than the capture data suggest.

This research supports our earlier analysis of capture and observation data collected from 1988 to 1991 (Richter and Azous 1995) in that no new species were identified in 1993 or 1995. Consequently, our recent studies also show no relationships between the number of amphibian species and wetland size or the number of Cowardin et al. (1979) habitat classes. These data also confirmed the relationship we found in earlier years between spawning and select vegetation classes, showing that amphibians spawn within the thin stemmed (non-cattail) emergent zone and with salamanders particularly selecting thin-stemmed emergent vegetation and tiny branches and root hairs of submerged vegetation on which to spawn (Richter and Roughgarden 1995).

Our study shows large differences in amphibian richness, using diverse survey techniques, and varying abundance (captures per 100 trap nights) between the survey years suggesting that multiple year studies are a prerequisite to the accurate identification of a wetlands' amphibian fauna. Explanations accounting for the dramatic differences could be weather related, as almost all the wetlands we studied responded with similar declines in richness and abundance over the study period. Pechmann et al. (1991) and Hairston (1987), in analyzing data from long term studies, show that for many amphibians, populations normally fluctuate dramatically over short periods but remain stable over longer periods of five to ten years. The extent to which distinct local populations, such as those found in our wetlands, vary asynchronously within a given year and for what reasons remain to be investigated.

We also found differences in amphibians identified at wetlands depending on survey technique, suggesting that multiple methods should be employed to accurately assess a wetland's amphibian population. For example, Pacific treefrogs were not captured in pitfalls anywhere, and large numbers of Northwestern salamanders that were breeding at SR24 were never captured in pitfalls. Similarly, we captured roughskin newts in funnel traps at ELS61 in early spring but never saw or captured them in pitfall traps. Also significant is that wetlands in which adults were captured in pitfalls were not observed to have spawn. Pitfalls on either side of drift fences totally encircling wetlands would be a good method of capturing most species and measuring abundances but was not feasible in a study of thisd many wetlands.

Our estimates of the number of captures per 100 trap nights appear similar to amphibian capture data elsewhere in the Northwest (McComb et al. 1993a, McComb et al. 1993b, Aubry and Hall 1991). However, differences in habitats used, timing of censuses and field techniques, including the possibility of counting recaptures in our study, do not allow direct statistical comparisons of our results with those of others.

The reduced richness of amphibians in wetlands with highly urbanized watersheds is likely due, in part, to differences in hydrologic patterns related to land use. Average WLF increases as the frequency of peak flood events increases. Such conditions may result in a frequently wet buffer affecting habitat for terrestrial breeders which prefer well drained soils that are not extremely wet, and tend to avoid soaked or flooded sites (Aubry and Hall 1991, Gilbert and Allwine 1991). Low numbers in wet riparian as opposed to dryer upland habitats have, for example, been documented with Ensatina (E.

eschscholtzii) in red alder (McComb et al. 1993a, McComb et al. 1993b), second-growth conifer (Gomez and Anthony 1996) and unmanaged Douglas-fir (Aubry and Hall 1991, Gilbert and Allwine 1991) stands. Aquatic and semi-aquatic breeders may be similarly affected by the increased frequency of flooding in that flooded habitats with high water level fluctuation may have less large downed woody material, litter and other organic material that provide food, cover and oviposition sites. Clearly, hydrology may account for the richness of the amphibian communities in the wetlands we studied, but may, in addition, be related to the proportion of adjacent area comprised of favorable habitat.

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APPENDIX TABLE 5-1. CAPTURE RATES OF AMPHIBIANS EACH YEAR IN INDIVIDUAL WETLANDS.

		Year			
Wetland	Data	Capture Rate 1988	Capture Rate 1989	Capture Rate 1993	Capture Rate 1995
AL3	AMGR				
	AMMA				
	ENES		0.667	0.333	
	BUBO				
	DITE				
	TAGR				
	RAAU		0.333		
	PSRE		0.333	0.333	
	PLVE		0.333	0.333	
	ENES		0.667	0.333	
B3I	AMGR				
	AMMA	1.000		0.333	0.333
	ENES				
	BUBO				
	DITE				
	TAGR				
	RAAU				
	PSRE				
	PLVE				
	ENES				
BBC24	AMGR	0.672	9.667		0.333
	AMMA	0.336	0.333		
	ENES		0.333	0.333	0.333
	BUBO				
	DITE				
	TAGR				0.333
	RAAU	0.672	1.000		
	PSRE				
	PLVE				
	ENES		0.333	0.333	0.333
ELS39	AMGR		0.333		
	AMMA				0.357
	ENES		0.667	0.333	0.357
	BUBO				
	DITE				
	TAGR				
	RAAU	0.333			
	PSRE		0.333		
	PLVE				
	ENES		0.667	0.333	0.357

Appendix Table 5-1 cont. Capture rates of amphibians each year in individual wetlands.

Wetland	Data	Capture Rate 1988	Capture Rate 1989	Capture Rate 1993	Capture Rate 1995
ELS61	AMGR	1.000	0.667		0.333
	AMMA	0.667	0.333		1.000
	ENES				
	BUBO				
	DITE				
	TAGR				
	RAAU	1.000	0.667		
	PSRE				
	PLVE				
	ENES				
ELW1	AMGR				
	AMMA				
	ENES				
	BUBO				
	DITE				
	TAGR				
	RAAU				
	PSRE				
	PLVE				
	ENES				
FC1	AMGR				
	AMMA	1.333			
	ENES	0.333			
	BUBO				
	DITE				
	TAGR				
	RAAU				
	PSRE				
	PLVE				
	ENES	0.333			
HC13	AMGR	0.333			
	AMMA			0.667	0.364
	ENES				0.364
	BUBO		0.333		
	DITE				
	TAGR				
	RAAU	1.357	1.667	0.667	0.727
	PSRE		0.333		
	PLVE	0.333	0.333		0.364
	ENES				0.364
JC28	AMGR	0.333			
	AMMA				
	ENES	0.333			
	BUBO				
	DITE				
	TAGR	0.333			
	RAAU		0.333		
	PSRE				
	PLVE	0.333	1.000		0.333
	ENES	0.333			

Appendix Table 5-1 cont. Capture rates of amphibians each year in individual wetlands.

Wetland	Data	Capture Rate 1988	Capture Rate 1989	Capture Rate 1993	Capture Rate 1995
LCR93	AMGR				
	AMMA		0.333		0.667
	ENES	0.333			0.333
	BUBO				
	DITE				
	TAGR				
	RAAU	2.333	0.667	2.000	0.357
	PSRE	0.333			
	PLVE	0.333	0.333		
	ENES	0.333			0.333
LPS9	AMGR	0.333			
	AMMA	1.000			
	ENES				
	BUBO				
	DITE				
	TAGR				
	RAAU	0.690	0.571		
	PSRE	0.333			
	PLVE	0.333			
	ENES				
MGR36	AMGR	0.333	0.667		0.333
	AMMA	0.333			
	ENES	1.000			
	BUBO				
	DITE				
	TAGR				
	RAAU	1.333			
	PSRE				
	PLVE	2.000	0.667		0.667
	ENES	1.000			
NFIC12	AMGR		1.345		
	AMMA				
	ENES		0.336	0.333	1.000
	BUBO				
	DITE				
	TAGR				
	RAAU		1.008		
	PSRE				0.333
	PLVE				0.333
	ENES		0.336	0.333	1.000
PC12	AMGR		0.667		
	AMMA				1.014
	ENES	0.333			
	BUBO		0.667		
	DITE	0.333			
	TAGR				
	RAAU	0.667	2.000		
	PSRE		0.333		0.338
	PLVE				
	ENES	0.333			

Appendix Table 5-1 cont. Capture rates of amphibians each year in individual wetlands.

Wetland	Data	Capture Rate 1988	Capture Rate 1989	Capture Rate 1993	Capture Rate 1995
RR5	AMGR	0.336	0.345		
	AMMA				
	ENES				
	BUBO	1.681			0.694
	DITE				
	TAGR				
	RAAU	0.672		0.333	
	PSRE				
	PLVE				
	ENES				
SC4	AMGR		1.000		0.333
	AMMA			0.333	
	ENES		0.667	0.333	
	BUBO				
	DITE				
	TAGR				
	RAAU				
	PSRE			0.333	
	PLVE		0.333		
	ENES		0.667	0.333	
SC84	AMGR				0.333
	AMMA				
	ENES				
	BUBO				
	DITE				
	TAGR				
	RAAU				0.392
	PSRE				
	PLVE		0.333		
	ENES				
SR24	AMGR				
	AMMA			0.333	
	ENES		1.000		
	BUBO		0.667		
	DITE	0.333			
	TAGR	0.667	0.667		
	RAAU		0.333		
	PSRE		0.667		
	PLVE				
	ENES		1.000		
TC13	AMGR		0.333		
	AMMA		0.333		
	ENES		0.333		0.667
	BUBO				
	DITE				
	TAGR				
	RAAU		0.667		
	PSRE				0.333
	PLVE				
	ENES		0.333		0.667

CHAPTER 6 BIRD DISTRIBUTION, ABUNDANCE AND HABITAT USE

by Klaus O. Richter and Amanda L. Azous

INTRODUCTION

Values and natural functions of wetlands gained growing recognition in the 1970s (Good et al. 1978, Greeson et al. 1979). Consequently, wetlands and are now considered sensitive habitats with diverse functions that are protected at federal, state and local levels. Of the many functions wetlands exhibit, their ability to provide resting, feeding and breeding habitat for a wide diversity of birds is among the most noticeable and appreciated. Abundant, often highly visible and unique avifauna are an important component of open space values, enriching quality of life. Despite these attributes, many hectares of marshes, swamps and other bird habitats are lost or impacted each year, in part due to our inadequate knowledge of how to protect the biologic function of wetlands.

Birds have been intensively studied in deciduous forests of east-central states (Blake and Karr 1984, Blake 1986), west coast coniferous forests (Artman 1990, Stofel 1993) and in other upland environments. Birds of coastal wetlands have also been widely studied (Craig and Beal 1992, Weller 1994). Fresh water wetland investigations, however, have been carried out by a relatively few biologists, who primarily documented the distribution and abundance of waterfowl and other marsh birds within pothole lakes and other wetlands in open landscapes of the Central Flyway (Weller and Spatcher 1965, Weller and Fredrickson 1974, Weller 1979). Although the importance of riparian corridors to avifauna, particularly passerines, woodpeckers and other non-game species has more recently been recognized (Brown and Dinsmore 1986, Knopf and Samson 1994), the avifauna of freshwater wetlands, specifically smaller palustrine wetlands distributed through forested landscapes, has not been well documented.

The purpose of this paper is to comprehensively describe palustrine wetland bird communities in the Lower Puget Sound Basin. The avifaunal literature is briefly reviewed to determine the uniqueness of palustrine wetland avifauna in a regional and landscape context. Then, we assess whether generalized landscape characteristics that account for bird distributions and abundances in upland ecosystems apply to predicting bird distributions within palustrine wetlands of the Northwest. We examined the diversity and proportional abundance of birds within the regional context of differing land use and the site-specific wetland attributes of size and vegetation structure, thereby building on the preliminary findings of Azous (1991) and Martin-Yanny (1992). The location, physical, chemical and vegetative description of the wetlands in this study are presented in Section 1 and Chapters 1, 2, and 3 of Section 2.

METHODS

The distribution and relative abundance of birds was determined based on surveys completed during the breeding period from late May to mid-June in 1988, 1989, 1991, 1992 and 1995. Birds were identified by non-territorial calls, territorial song, pecking and drumming, visual sightings and flyovers during 15-minute point counts (Johnston 1990, Verner 1985) at permanent census stations. Usually, four ornithologists surveyed each wetland totaling one hour per station. Surveys commenced one half-hour after sunup to approximately 9:00 am and stations were surveyed in alternating order to minimize time biases.

We calculated the gamma diversity, the collective species identified across all wetlands (a landscape metric) and alpha diversity, the species identified at a single wetland (a site metric) (Whittaker 1975) by summing the number of species. We calculated all diversity measures only including species observed two or more times. Because alpha diversity measures are insensitive to bird species composition, we calculated diversity indices for birds with specific breeding habitats, versatility ratings, residency traits, and urbanization affinities. This paper reports on some of the more general overall diversity metrics analyzed to date.

We estimated relative abundances for each species at a wetland using average detection values calculated by dividing the total number of a species sighted at a wetland (derived by combining 15-minute station totals into a 1-hour station total and then combining station totals) by the total number of 15-minute observation periods at a wetland. Using this detection value we standardized the data among wetlands with unequal sampling effort (e.g., more stations and hence more time at large wetlands).

We relied on Paulson (1992) to identify total species potentially occurring in palustrine wetlands habitats (Appendix Table 6-1) of the Puget Sound Basin. Species were classified as common residents, rare residents, or migrants according to abundance ratings provided in Hunn (1982). Habitat versatility ratings for bird species were obtained from Brown (1985) and represent the sum total of the number of plant communities and stand conditions used for breeding plus the number of plant communities and stand conditions used for feeding by a species.

Bird preferences for National Wetlands Inventory (NWI) wetland habitat classes (Cowardin et al. 1979) identified at each wetland were converted to habitat preferences identified in Paulson (1992) as follows: open water/unconsolidated bottom = ponds and lakes; emergent wetland, persistent = fresh [water] marsh; forested wetland, needle-leaved evergreen = wet coniferous forest; forested wetland, broad-leaved deciduous = riparian woodland; emergent wetland, nonresistant = wet lowland meadow; scrub-shrub =shrub thickets, and unconsolidated shore. Alpha and gamma diversities within the study wetlands were compared against the potential species richness documented in the Lower Puget Sound Basin that were known to occur in these respective habitats. Habitat land cover and fragmentation was determined by quantifying land cover within 1000 m using remote sensing methods and a geographic information system.

Statistical analysis of correlations and hypothesis testing utilized parametric statistics when assumptions of normality were met and non-parametric statistics when assumptions were violated. We chose $P < 0.05$, and $P > 0.05$ and ≤ 0.10 with $r \geq 0.4$ as significant and weakly significant, respectively. Nevertheless, significance should be interpreted cautiously because of the high variability of the data and concomitantly unacceptably wide confidence intervals for predictive level of significance. This is due to the low number of replicates (e.g., wetlands undergoing significant impacts) and discontinuities in habitat characteristics (e.g., unequal representation of all wetland size classes, etc.).

RESULTS

Regional Species Richness (Beta Diversity)

A total of 94 species were identified and sighted on at least two or more occasions among all the wetlands (Table 6-1). This total wetland diversity of 94 species represents only 59% of the 158 species that could be expected to use habitats found at wetlands in the Lower Puget Sound Basin (Paulson 1992) (Appendix Table 6-1). This diversity, however, is significantly higher than the 56 species found by Stofel (1993), the 23 species identified by Artman (1990) in rural upland second-growth forest, and the 48 species by Gavareski (1976) in urban park environments. All the species identified in these studies were identified at our surveyed wetlands, with the exception of great horned owl, Northern harrier, Northern rough-winged swallow, luzuli bunting, and turkey vulture as well as a few high elevation species such as gray jay, blue grouse, golden-crowned sparrow.

The relative diversity across the study wetlands ranged from 38% to 72% of all birds collectively identified across all wetlands (Figure 6-1). No more than 42% (67 species) of potential regional bird diversity (per Paulson) was present in any one wetland. This represented 71% of our collective wetland sightings and was observed at SR24, a large, open-water, vegetatively rich, and undisturbed wetland. In contrast, the lowest diversity of 37 species (23% of potential regional and 39% of our collective wetlands) was identified at NFIC12 a small, highly disturbed wetland situated between a large subdivision and a roadway. The next lowest richness of 38 (40% of collective) and 39 (41% of collective) species were identified at AL3 and ELS39, respectively, both small, intermittently flooded wetlands.

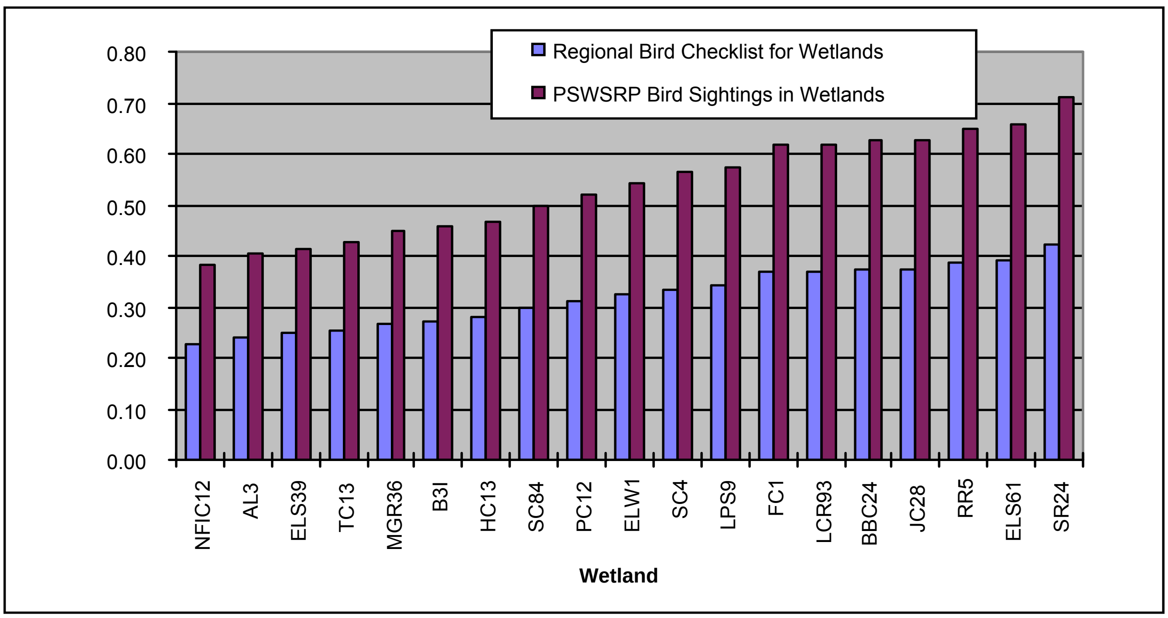

Figure 6-1. Percent of all species collectively found in wetlands.

Only three species, American robin, black-capped chickadee and song sparrow, and representing 3.2% of total diversity were shared between all 19 wetlands. Conversely, four species (4.3%), American coot, hooded merganser, savannah sparrow and spotted sandpiper were found in only one wetland. Interestingly, 47 species (50% of total) were found in 53% or more of the wetlands.

Migrants accounted for 37% of species (35). Common and rare residents respectively numbered 17 and 42 species and comprised 18% and 45% of sightings thus significantly enhancing the diversity of wetland avifauna. Many residents were species of adjacent terrestrial habitats using wetlands to drink, augment diet, and support their young.

During the study period the observations of thirteen species declined including two rare residents, orange-crowned warbler and red crossbill. Nine other rare resident species showed no change and six weren't observed in sufficient numbers to determine. Forty-nine percent of species showed no change in population and ten species increased. We did not have enough observations of 25 species to determine changes in population status.

The observations of birds known to avoid suburban and urban development both declined and increased depending on species. Three avoiders declined including orange-crowned warbler, varied thrush and willow flycatcher while two increased, black-throated gray warbler and Swainson's thrush. Seven species known to be adaptable to urbanization increased while nine declined.

Species Richness by Wetland (Alpha Diversity)

Species richness varied widely within wetlands over the study period (Figure 6-2). Species richness for all years was higher because different species were observed in different years. We saw the highest richness in 1989 in virtually all wetlands and the lowest in the last year of our research, 1995.

Table 6-1. Species and life history traits of birds sighted at study wetlands.

Bird Species	Percent of Wetlands 1989	Percent of Wetlands 1991	Percent of Wetlands 1995	Percent of Wetlands All Years	Status	Population	Adapt-ability	Versatility Rating
American Coot	0.05	0.06	0.05	0.05	resident	insufficient data	Adapter	10
American Goldfinch	0.79	0.50	0.68	0.84	resident	declining	Adapter	23
American Robin	1.00	1.00	1.00	1.00	resident	increasing	Adapter	37
Anna's Hummingbird	0.11	0.00	0.05	0.16	rare resident	insufficient data	Adapter	25
Bald Eagle	0.00	0.06	0.11	0.11	migrant	insufficient data	Adapter	19
Barn Swallow	0.26	0.22	0.42	0.53	resident	increasing	Adapter	18
Black-capped Chickadee	1.00	0.94	1.00	1.00	migrant	declining	Adapter	28
Belted Kingfisher	0.26	0.22	0.21	0.58	resident	no change	Adapter	Undetermined
Bewick's Wren	0.68	0.89	0.74	0.95	resident	declining	Adapter	22
Brown-headed Cow Bird	0.58	0.33	0.63	0.95	migrant	insufficient data	Adapter	9
Band-tailed Pigeon	0.05	0.06	0.05	0.16	migrant	increasing	Adapter	17
Bushtit	0.84	0.61	0.21	0.95	migrant	no change	Adapter	10
Canada Goose	0.11	0.06	0.11	0.16	resident	declining	Adapter	22
California Quail	0.05	0.00	0.16	0.21	rare resident	no change	Adapter	8
Chestnut-backed Chickadee	0.79	0.78	0.47	1.00	resident	increasing	Adapter	27
Cedar Waxwing	0.84	0.78	0.53	0.89	resident	insufficient data	Adapter	28
Cliff Swallow	0.05	0.11	0.11	0.16	migrant	insufficient data	Adapter	12
Common Yellow-throat	0.58	0.67	0.47	0.68	rare resident	no change	Adapter	9
Dark-eyed Junco	0.68	0.50	0.37	0.84	migrant	insufficient data	Adapter	Undetermined
Downy Woodpecker	0.47	0.56	0.63	0.89	resident	insufficient data	Adapter	21
Fox Sparrow	0.05	0.00	0.11	0.16	resident	insufficient data	Adapter	34
Gadwall	0.11	0.06	0.05	0.11	resident	insufficient data	Adapter	10
Great Blue Heron	0.42	0.28	0.21	0.63	resident	no change	Adapter	27
Golden-crowned kinglet	0.95	0.94	0.37	1.00	resident	no change	Adapter	14
Glaucous-winged Gull	0.16	0.06	0.05	0.16	migrant	declining	Adapter	26
Hammond's Flycatcher	0.26	0.33	0.05	0.47	migrant	no change	Adapter	26
Hairy Woodpecker	0.79	0.50	0.32	0.79	rare resident	insufficient data	Adapter	10
House Finch	0.58	0.22	0.32	0.68	resident	no change	Adapter	28
Hutton's Vireo	0.42	0.06	0.11	0.47	resident	no change	Adapter	27
Killdeer	0.21	0.00	0.11	0.32	resident	no change	Adapter	28
Mallard	0.42	0.28	0.42	0.58	resident	no change	Adapter	10
Marsh Wren	0.68	0.22	0.16	0.68	resident	no change	Adapter	8
Northern Flicker	0.37	0.39	0.37	0.63	migrant	declining	Adapter	27
Northern Oriole	0.11	0.00	0.11	0.21	resident	no change	Adapter	33
Pied-billed Grebe	0.26	0.06	0.11	0.26	resident	no change	Adapter	Undetermined
Pacific-slope Flycatcher	0.95	1.00	0.84	1.00	migrant	insufficient data	Adapter	10
Purple Finch	0.63	0.44	0.47	0.79	migrant	increasing	Adapter	24
Red-breasted Nuthatch	0.53	0.56	0.63	0.84	migrant	insufficient data	Adapter	Undetermined
Red Crossbill	0.32	0.67	0.16	0.79	rare resident	declining	Adapter	29
Red-eyed Vireo	0.11	0.00	0.11	0.16	resident	no change	Adapter	26
Rufous-sided Towee	0.89	0.89	0.89	1.00	migrant	no change	Adapter	37
Rufous Hummingbird	0.21	0.17	0.16	0.32	resident	insufficient data	Adapter	28
Ruby Crowned Kinglet	0.53	0.44	0.63	0.89	resident	no change	Adapter	31
Red-winged Blackbird	0.53	0.33	0.53	0.68	rare resident	insufficient data	Adapter	22
Savannah Sparrow	0.00	0.06	0.00	0.05	resident	increasing	Adapter	11
Song Sparrow	1.00	1.00	1.00	1.00	resident	no change	Adapter	24
Sharp-shinned Hawk	0.21	0.00	0.00	0.21	rare resident	no change	Adapter	15
Steller's Jay	0.58	0.61	0.68	0.84	rare resident	insufficient data	Adapter	33
Tree Swallow	0.58	0.39	0.42	0.84	rare resident	no change	Adapter	22
Violet-green Swallow	0.47	0.39	0.79	0.79	rare resident	insufficient data	Adapter	28
Virginia Rail	0.26	0.11	0.16	0.32	migrant	no change	Adapter	33
White-crowned Sparrow	0.32	0.22	0.05	0.32	migrant	no change	Adapter	29
Western Wood-pewee	0.32	0.17	0.32	0.47	migrant	declining	Adapter	30
Winter Wren	0.95	0.94	0.68	1.00	resident	increasing	Adapter	27

Table 6-1 continued. Species and life history traits of birds sighted at study wetlands.

Bird Species	Percent of Wetlands 1989	Percent of Wetlands 1991	Percent of Wetlands 1995	Percent of Wetlands All Years	Status	Population	Adapt-ability	Versatility Rating
Wood Duck	0.32	0.22	0.37	0.63	rare resident	no change	Adapter	25
Yellow Warbler	0.74	0.72	0.21	0.95	migrant	declining	Adapter	19
Yellow-rumped Warbler	0.26	0.11	0.21	0.47	rare resident	no change	Adapter	31
Black Headed Grosbeak	0.84	0.61	0.79	1.00	rare resident	no change	Avoider	34
Brewer's Blackbird	0.21	0.39	0.11	0.47	migrant	no change	Avoider	28
Brown Creeper	0.26	0.28	0.16	0.47	resident	no change	Avoider	32
Black-throated Gray Warbler	0.53	0.39	0.47	0.79	migrant	increasing	Avoider	24
Blue-winged Teal	0.00	0.00	0.11	0.11	resident	no change	Avoider	29
Caspian Tern	0.00	0.00	0.11	0.11	migrant	insufficient data	Avoider	Undetermined
Chipping Sparrow	0.11	0.06	0.11	0.26	migrant	no change	Avoider	36
Cooper's Hawk	0.11	0.00	0.16	0.26	migrant	no change	Avoider	8
Common Raven	0.00	0.00	0.11	0.11	rare resident	insufficient data	Avoider	32
Evening Grosbeak	0.21	0.06	0.21	0.32	rare resident	no change	Avoider	33
Green Heron	0.11	0.06	0.05	0.16	migrant	no change	Avoider	6
Hermit Thrush	0.84	0.33	0.21	0.84	resident	no change	Avoider	22
Hooded Merganser	0.05	0.00	0.05	0.05	migrant	insufficient data	Avoider	25
MacGillivary's Warbler	0.11	0.00	0.21	0.26	migrant	insufficient data	Avoider	Undetermined
Northern Pigmy Owl	0.05	0.06	0.05	0.16	migrant	no change	Avoider	20
Orange-crowned Warbler	0.74	0.44	0.37	0.84	rare resident	declining	Avoider	31
Olive-sided Flycatcher	0.16	0.22	0.11	0.32	resident	no change	Avoider	36
Pine Siskin	0.26	0.00	0.26	0.47	resident	no change	Avoider	27
Pileated Woodpecker	0.21	0.00	0.11	0.26	resident	no change	Avoider	32
Red-breasted Sapsucker	0.21	0.00	0.21	0.37	resident	no change	Avoider	24
Red-eyed Vireo	0.05	0.06	0.11	0.16	resident	no change	Avoider	26
Ruffed Grouse	0.05	0.11	0.05	0.16	resident	insufficient data	Avoider	29
Sora	0.00	0.06	0.11	0.16	migrant	no change	Avoider	28
Solitary Vireo	0.21	0.39	0.21	0.58	migrant	insufficient data	Avoider	10
Spotted Sandpiper	0.05	0.00	0.00	0.05	rare resident	no change	Avoider	4
Swainson's Thrush	0.95	1.00	0.95	1.00	resident	increasing	Avoider	32
Townsend's Warbler	0.68	0.06	0.37	0.79	migrant	no change	Avoider	26
Varied Thrush	0.21	0.00	0.00	0.21	migrant	declining	Avoider	29
Vaux's Swift	0.58	0.44	0.16	0.68	migrant	no change	Avoider	34
Warbling Vireo	0.68	0.17	0.26	0.79	resident	insufficient data	Avoider	10
Western Tanager	0.47	0.33	0.42	0.63	migrant	no change	Avoider	34
Willow Flycatcher	0.84	0.83	0.79	0.95	migrant	declining	Avoider	20
Wilson's Warbler	0.89	0.78	0.63	1.00	migrant	no change	Avoider	33
American Crow	0.84	0.94	0.89	0.95	resident	declining	Exploiter	32
European Starling	0.42	0.28	0.16	0.53	resident	no change	Exploiter	27
House Sparrow	0.21	0.22	0.05	0.42	resident	insufficient data	Exploiter	12
Rock Dove	0.11	0.11	0.00	0.11	resident	increasing	Exploiter	Undetermined

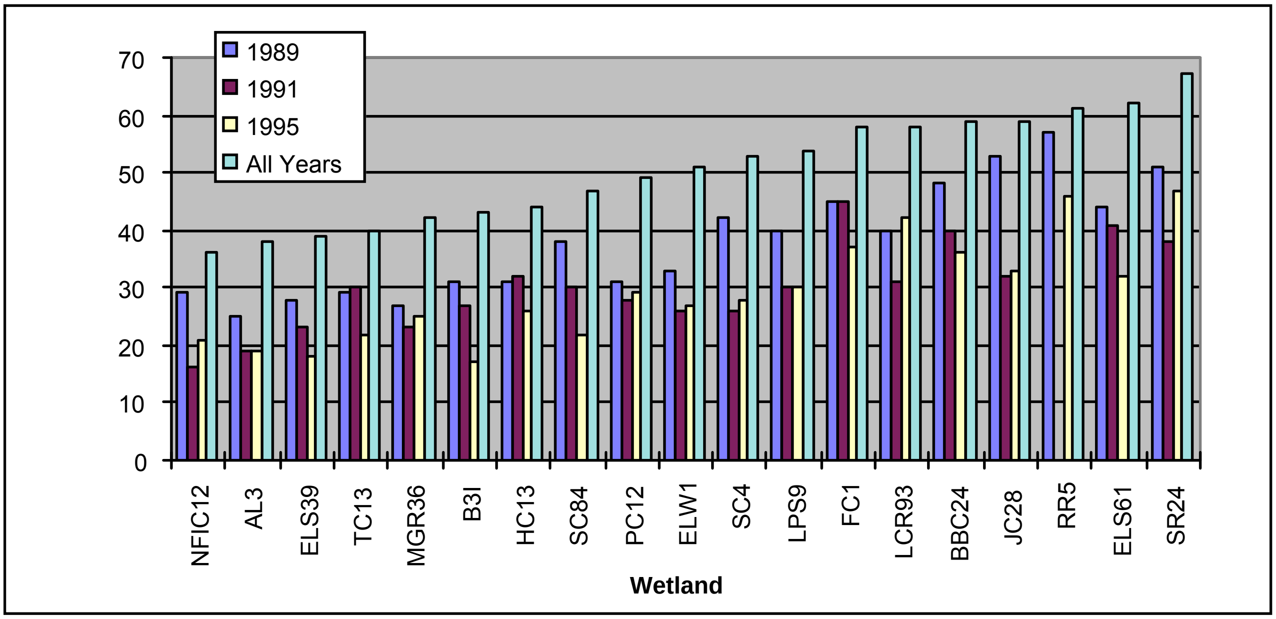

Figure 6-2. Total species diversity in wetlands for each study year.

We used total richness to measure species diversity and found it increased directly with wetland area (Fisher's r to z, R= 0.53, P= 0.018). Our study wetlands ranged from 0.6 to 12.6 ha with 13 wetlands less than four hectares. Among the six wetlands greater than four hectares, only one had less than 50 species present, whereas among the wetlands with less than four hectares, eight had richness of less than 50 (Figure 6-3).

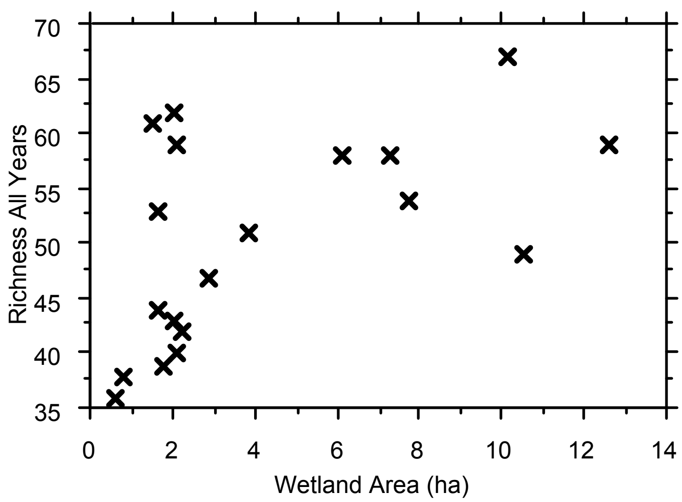

Figure 6-3. Relationship between bird species richness and wetland size.

Large wetland areas, while a major component of the most diverse bird communities we found, is not the only factor important as evidenced in that two of the smallest wetlands of less than two hectares had high richness of 61 and 62 species. Structural complexity was also found to be a contributing factor, as characterized by either the number of NWI vegetation (Fisher's r to z, R = 0.48, P = 0.04), or Paulson's habitat (Fisher's r to z, R = 0.6, P = 0.006) classes (Figure 6-4), though the statistical relationship was stronger with Paulson's habitat classifications. For example, three wetlands with only one NWI vegetation class had 55 bird species or more, representing the upper range of diversity, during the study period. The single NWI classifications used to describe the vegetation communities in those wetlands were equivalent to three of the bird habitat classifications probably better reflecting avian potential.

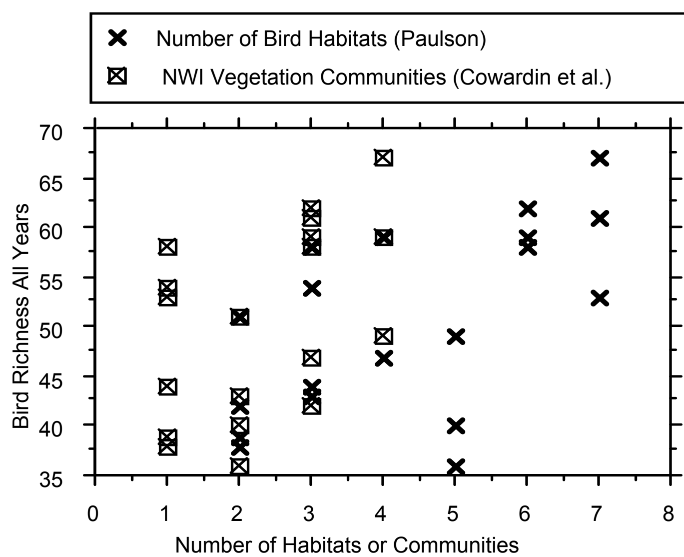

Figure 6-5. Relationship between bird species richness and vegetation community complexity.

Bird diversity in wetlands with adjacent lakes, for example FC1 and ELW1, and open water, such as SR28 and BBC24, was bolstered by waterfowl. Most frequent waterfowl encountered over the three years of complete surveys (e.g.,1989, 1991 and 1995) were mallard (99), Canada goose (10), pied-billed grebe (26), hooded merganser (9), and gadwall (7), with only occasional sightings of blue-winged teal (2), American widgeon (1), and lesser scaup (1).

Relative Abundance

Summary tables for species abundance determined by average detections are provided in Table 6-2, whereas detailed wetland-specific detections are provided in Appendix Tables 6-2. Found in each wetland and in decreasing order of abundance are song sparrow and American robin (both with at least one expected detection per visit), Swainsons thrush, red-wing blackbird and black-capped chickadees. Within selected other wetlands American crow, rufous-sided towhee and Pacific slope flycatchers, willow flycatcher winter wren and marsh wren were abundant.

DISCUSSION

Our bird diversities when compared with diversities observed in terrestrial habitats by others, indicate that wetlands are disproportionately used by birds and are probably the single most productive habitat for this vertebrate class in the Puget Sound Basin. Of all the species identified in Western Washington 82% are found in wetlands. Artman (1990) found only 23 species in 45-50 year old stands dominated by western hemlock but also containing Douglas-fir (*Pseudotsuga menziesii*), Pacific silver fir (*Abies amabilis*), and western red-cedar (*Thuja plicata*). Of the 48 species identified by Gavareski (1976) in 4-400 ha^2 diversely vegetated urban parks of Seattle only two (great horned owl and golden-crowned sparrow) were undetected at wetlands.

We also found significantly more species than identified by Milligan (1985) in a survey of wetlands of less than 4 ha^2 in urbanized areas of the Puget Sound Basin. From censuses in April, May and June of 1984, 60 species were found in combined wetland, and wetland and upland habitats, of 23 widely diverse sites characterized by varying density of development. Mulligan also found both total and average avifaunal diversity to be correlated to wetland habitat complexity measured by the number of NWI vegetation classes. Bird diversity was also found to correlate with the percentage of wetland buffered by shrubland or forest vegetation, although interestingly, there was only a minor predicted increase in diversity with increasing buffer width classes of 50, 100 and 200 feet from the wetland edge.

During the baseline surveys of wetlands for this study, Martin-Yanny (1992) listed 88 species. During subsequent surveys our study identified an additional six species, and presumably with continued surveys a few additional species may be expected at decreasing rates. Nevertheless it seems unlikely we would find the entire list of species identified by Paulson as potentially occurring in palustrine wetland habitats because of the limited geographic location of our wetlands within disturbed watersheds.

Paulson (1992) found that most resident species are maintaining their populations despite increasing urbanization. Our study results generally corroborate this finding though we did not have sufficient data to assess all species we observed. Declines were observed among some migrating species and some adapters.

Wetland area and habitat diversity were found to be critical factors in maintaining high biodiversity in wetland bird communities. When wetlands are assessed for function and value related to avian potential, methods based on bird preferences, such as the habitat classification by Paulson, would be more appropriate than the NWI classification system.

Table 6-2. Bird species abundance in order of increasing average detections.

Bird Species	1989	1991	1995	All Years	Detectability over all years, all wetland stations
Blue-winged Teal	0	0	2	2	0.0026
Savannah Sparrow	0	2	0	2	0.0026
Spotted Sandpiper	2	0	0	2	0.0026
Anna's Hummingbird	2	0	1	3	0.0039
Bald Eagle	0	1	2	3	0.0039
Northern Pigmy Owl	1	1	1	3	0.0039
California Quail	1	0	3	4	0.0052
Caspian Tern	0	0	4	4	0.0052
Northern Oriole	2	0	2	4	0.0052
Red-eyed Vireo	2	0	2	4	0.0052
Sharp-shinned Hawk	4	0	0	4	0.0052
Common Raven	0	0	5	5	0.0064
Rock Dove	2	3	0	5	0.0064
Ruffed Grouse	1	2	2	5	0.0064
Sora	0	2	3	5	0.0064
Chipping Sparrow	3	1	2	6	0.0077
Fox Sparrow	1	0	5	6	0.0077
Glaucous Winged Gull	3	1	2	6	0.0077
Gadwall	3	2	2	7	0.0090
Varied Thrush	7	0	0	7	0.0090
Band-tailed Pigeon	3	1	4	8	0.0103
MacGillivary's Warbler	2	0	6	8	0.0103
Red-breasted Sapsucker	4	0	4	8	0.0103
Red-eyed Vireo	2	1	5	8	0.0103
Cooper's Hawk	2	0	7	9	0.0116
Hooded Merganser	6	0	3	9	0.0116
Killdeer	6	0	3	9	0.0116
Canada Goose	2	2	6	10	0.0129
Green Heron	9	1	1	11	0.0142
House Sparrow	6	5	1	12	0.0155
Cliff Swallow	4	6	3	13	0.0168
Rufous Hummingbird	5	4	4	13	0.0168
Yellow-rumped Warbler	7	3	4	14	0.0180
American Coot	4	5	6	15	0.0193
Olive-sided Flycatcher	5	8	2	15	0.0193
Pileated Woodpecker	11	0	4	15	0.0193
Virginia Rail	7	3	5	15	0.0193
Evening Grosbeak	7	1	8	16	0.0206
Brewer's Blackbird	6	7	6	19	0.0245
Pine Siskin	7	0	12	19	0.0245
Belted Kingfisher	7	4	10	21	0.0271
Brown Creeper	9	7	5	21	0.0271
Hammond's Flycatcher	9	10	2	21	0.0271
Hutton's Vireo	19	1	2	22	0.0284
Solitary Vireo	5	13	4	22	0.0284
Wood Duck	9	4	9	22	0.0284
White-crowned Sparrow	14	9	1	24	0.0309

Table 6-2 continued. Bird species abundance in order of increasing average detections.

Bird Species	1989	1991	1995	All Years	Detectability over all years, all wetland stations
Pied-billed Grebe	8	2	16	26	0.0335
Western Wood-pewee	11	6	13	30	0.0387
Red Crossbill	9	23	4	36	0.0464
Vaux's Swift	17	13	8	38	0.0490
House Finch	23	8	8	39	0.0503
Great Blue Heron	17	7	17	41	0.0528
Northern Flicker	10	11	23	44	0.0567
Ruby Crowned Kinglet	19	9	20	48	0.0619
European Starling	32	11	9	52	0.0670
Townsend's Warbler	38	2	13	53	0.0683
Western Tanager	17	7	29	53	0.0683
Barn Swallow	12	11	31	54	0.0696
Downy Woodpecker	16	14	25	55	0.0709
Hairy Woodpecker	36	15	11	62	0.0799
Warbling Vireo	38	3	22	63	0.0812
Brown-headed Cow Bird	23	11	31	65	0.0838
Orange-crowned Warbler	38	23	11	72	0.0928
Black-throated Gray Warbler	25	13	44	82	0.1057
Dark-eyed Junco	40	17	25	82	0.1057
Purple Finch	24	22	38	84	0.1082
Red-breasted Nuthatch	15	29	40	84	0.1082
Violet-green Swallow	18	14	54	86	0.1108
Marsh Wren	55	19	23	97	0.1250
Bushtit	55	30	13	98	0.1263
Mallard	32	18	49	99	0.1276
Tree Swallow	43	27	31	101	0.1302
Hermit Thrush	84	11	8	103	0.1327
Golden-crowned kinglet	59	34	16	109	0.1405
Chestnut-backed Chickadee	41	37	38	116	0.1495
Steller's Jay	28	38	68	134	0.1727
Cedar Waxwing	57	41	42	140	0.1804
Yellow Warbler	67	50	26	143	0.1843
American Goldfinch	54	42	55	151	0.1946
Black Headed Grosbeak	56	37	64	157	0.2023
Bewick's Wren	48	42	68	158	0.2036
Common Yellow-throat	93	63	65	221	0.2848
Wilson's Warbler	115	71	77	263	0.3389
Winter Wren	109	85	114	308	0.3969
American Crow	73	106	134	313	0.4034
Rufous-sided Towee	99	94	140	333	0.4291
Willow Flycatcher	114	90	141	345	0.4446
Pacific-slope Flycatcher	127	147	145	419	0.5399
Black-capped Chickadee	152	138	170	460	0.5928
Red-winged Blackbird	280	147	165	592	0.7629
Swainson's Thrush	153	179	336	668	0.8608
American Robin	279	230	293	802	1.0335
Song Sparrow	454	389	395	1238	1.5954
Total Abundance	3426	2551	3337	9314	

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Appendix Table 6-1. List of bird species expected to be using palustrine wetlands in Western Washington.

		STATUS		BREEDING DISTRIBUTION							
SPECIES	Evolutionary Order	B	W	WA	PL	FM	WC	BF	RW	ST	WM
Common loon	1	x	x	o	SW						
Pied-billed grebe	2	x	x	x	R						
Horned grebe	3	x	x	o	SW						
Red-necked grebe	4	x	x	o	S						
Eared grebe	5	x	o	x	Sw						
Western grebe	6	o	x	x	SW						
Clark's grebe	7	o		o	S						
Double-crested cormorant	8	x	x	x	SW						
American bittern	9	x	o	x		SW					
Great blue heron	10	x	x	x		R					R
Great egret	11	o	o	o		S					
Black-crowned night-heron	12	o	o	x		SW					
Canada goose	13	x	x	x	R						R
Wood duck	14	x	o	x	R						
Green-winged teal	15	x	o	x	SW	SW					W
Mallard	16	x	x	x	R	R					W
Northern pintail	17	x	x	x	SW	SW					W
Blue-winged teal	18	x		x	S	S					
Cinnamon teal	19	o		x	S	S					
Northern shoveler	20	x	o	x	SW	SW					
Gadwall	21	x	x	x	SW	SW					W
American wigeon	22	x	o	x	SW	SW					W
Canvasback	23	x	x	x	SW	S					
Redhead	24	x	o	x	S	S					
Ring-necked duck	25	x	x	x	SW						
Barrow's goldeneye	26	x	x	x	S						
Bufflehead	27	x	x	o	SW						
Hooded merganser	28	x	x	x	SW						
Common merganser	29	x	x	x	W						
Ruddy duck	30	x	o	x	SW	S					
Bald eagle	31	x	x	x	R				R		
Northern harrier	32	x	x	x		SW					R
Sharp-shinned hawk	33	x	o	x			R	W	W	W	
Cooper's hawk	34	x	o	x			R	R	R		
Northern goshawk	35	x	x	x			R				
Red-tailed hawk	36	x	o	x			R	R	R		
American kestrel	37	x	x	x					R		
Peregrine falcon	38	o	o	x			r				
Spruce grouse	39	x	x	x			R*				
Blue grouse	40	x	x	x			R				
Ruffed grouse	41	x	x	x				R			
Sharp-tailed grouse	42	o	o	x					W		
Mountain quail	43	o	o	o						R	
Virginia rail	44	x	x	x		R					
Sora	45	x	o	x		SW					S
American coot	46	x	x	x	SW	S					
Sandhill crane	47	o	+	o		sM					sM
Killdeer	48	x	o	x	SW	SW					SW
Common snipe	49	x	o	x		SW					SW
Band-tailed pigeon	50	o	o	x			R	R			
Mourning dove	51	x	o	x				R	R		
Barn owl	52	o	o	x			r	R	R		
Western screech-owl	53	x	x	x			R	R	R		
Great horned owl	54	x	x	x			R	R	R		
Northern pygmy-owl	55	x	x	x			R				
Barred owl	56	x	x	x			R				
Long-eared owl	57	x	x	x					R		
Short-eared owl	58	x	x	x							R
Boreal owl	59	x	x	x			R*				
Northern saw-whet owl	60	x	x	x			R		W		

Appendix Table 6-1 cont'. List of bird species expected to be using palustrine wetlands in Western Washington.

		STATUS		BREEDING DISTRIBUTION							
SPECIES	Evolutionary Order	B	W	WA	PL	FM	WC	BF	RW	ST	WM
Common nighthawk	61	x		x			S		S		
Black swift	62	x		x			S				
Vaux's swift	63	x		x			S	S	S		
Black-chinned hummingbird	64	o		x				S	S		
Anna's hummingbird	65	o	o	o					R	R	
Calliope hummingbird	66	x		x				S	S		
Rufous hummingbird	67	x		x			S	S		S	
Belted kingfisher	68	x	x	x	SW						
Lewis' woodpecker	69	x	o	x					S		
Red-naped sapsucker	70	o		x				S	S		
Red-breasted sapsucker	71	x	o	x			R	R			
Downy woodpecker	72	x	x	x			r	R	R		
Hairy woodpecker	73	x	x	x			R	r			
Three-toed woodpecker	74	x	x	x			r*				
Black-backed woodpecker	75	x	x	x			R*				
Northern flicker	76	x	x	x			R	R	R		
Pileated woodpecker	77	x	x	x			R	R			
Olive-sided flycatcher	78	x		x			S	S			
Western wood-pewee	79	x		x			S	S	S		
Willow flycatcher	80	x		x					S	S	
Least flycatcher	81	x		o				S	S		
Hammond's flycatcher	82	x		x			S	S			
Pacific-slope flycatcher	83	x		x			S	S	S		
Ash-throated flycatcher	84	o		o					S		
Western kingbird	85	o		x					S		
Eastern kingbird	86	x		x					S		
Tree swallow	87	x		x	S				S		
Violet-green swallow	88	x		x	S		S				
Gray jay	89	x	x	x			R				
Steller's jay	90	x	x	x			R				
Black-billed magpie	91	x	x	x					R		
American crow	92	x	x	x			R	R			W
Common raven	93	x	x	x			R				
Black-capped chickadee	94	x	x	x				R	R	R	
Boreal chickadee	95	x	x	o			R				
Chestnut-backed chickadee	96	x	x	x			R				
Bushtit	97	o	o	x					R	R	
Red-breasted nuthatch	98	x	x	x			R	W	W		
White-breasted nuthatch	99	o	o	x				R			
Brown creeper	100	x	x	x			R	W	W		
Canyon wren	101	o	o	x					R		
Bewick's wren	102	o	o	x					R	R	
House wren	103	x		x				S	S	S	
Winter wren	104	x	x	x			R	R	W	M	
Marsh wren	105	x	o	x		R					w
Golden-crowned kinglet	106	x	x	x			R		W		
Townsend's solitaire	107	x	x	x			S*				
Veery	108	o		x				S	S		
Swainson's thrush	109	x		x			S	S			
Hermit thrush	110	x	o	x			S*				
American robin	111	x	x	x			R	R	S		W
Varied thrush	112	x	x	x			R	W	W		
Gray catbird	113	x		x				S	S	S	
American pipit	114	x	o	x		M					M
Bohemian waxwing	115	x	x	o			S		W		
Cedar waxwing	116	x	x	x			S	R	R		
Solitary vireo	117	x		x			S				
Hutton's vireo	118	o	o	o			R	R			
Warbling vireo	119	x		x			S	S		S	
Red-eyed vireo	120	x		x				S	S		
Orange-crowned warbler	121	x	+	x				S	S	Sw	

Appendix Table 6-1 cont'. List of bird species expected to be using palustrine wetlands in Western Washington.

SPECIES	Evolutionary Order	STATUS B	W	BREEDING DISTRIBUTION WA	PL	FM	WC	BF	RW	ST	WM
Nashville warbler	122	o		x					S	S	
Yellow warbler	123	x		x				S	S		
Yellow-rumped warbler	124	x	o	x			S*		Mw		
Black-throated gray warbler	125	o		x			S				
Townsend's warbler	126	x	o	x			S*				
Hermit warbler	127	o		o			S*				
American redstart	128	x		x				S	S		
Northern waterthrush	129	x		o					S		
MacGillivray's warbler	130	x		x				S	S	S	
Common yellowthroat	131	x		x		S					
Wilson's warbler	132	x		x			S	S		S	
Yellow-breasted chat	133	o		x					S	S	
Western tanager	134	x		x			S	S	S		
Black-headed grosbeak	135	x		x				S	S		
Lazuli bunting	136	x		x					S	S	
Rufous-sided towhee	137	x	o	x				R	R	R	
Savannah sparrow	138	x	o	x							S
Fox sparrow	139	x	x	x					S	SW	
Song sparrow	140	x	x	x		S			R	R	
Lincoln's sparrow	141	x	x	x		S*			M		
White-crowned sparrow	142	x	x	x			S		M	SR	
Dark-eyed junco	143	x	x	x			R	S	W	W	
Bobolink	144	o		o							S
Red-winged blackbird	145	x	o	x		SW					W
Yellow-headed blackbird	146	x	o	x		S					M
Brewer's blackbird	147	x	o	x					S		R
Brown-headed cowbird	148	x	o	x		S	S	S	S	S	
Northern oriole	149	o		x				S	S		
Pine grosbeak	150	x	x	x			R*				
Purple finch	151	x	x	x			R	R			
Cassin's finch	152	x	o	x			S*				
House finch	153	x	x	x					R	R	
Red crossbill	154	x	x	x			R				
White-winged crossbill	155	x	x	o			R*				
Pine siskin	156	x	x	x			R	W	W		
American goldfinch	157	o	o	x					R		
Evening Grosbeak	158	x	x	x			R	R			
Total:	158										

	PL	FM	WC	BF	RW	ST	WM
BREEDING SPECIES	66	38	84	65	69	30	19
NONBREEDING SPECIES	52	29	46	31	45	23	34
BREEDING HABITAT SPECIALISTS	34	13	31	4	5	2	5
NONBREEDING HABITAT SPECIALISTS	2	4	11	1	1	4	0

STATUS
B - breeding status

(also migratory status of nonbreeders)

S - summer
W - winter
M - migrant (spring and fall)
F - fall
W - wintering status
x - widespread
o - occurs in <33% of region
+ - occurs in <10% of region
c - coast only

DISTRIBUTION BY AREA
WA - Washington

x - widespread in area

o - occurs in <33% of area

Appendix Table 6-2. Detection rates for species within each wetland all years combined.

	Detection Rates																		
Species	AL3	B3I	BBC24	ELS39	ELS61	ELW1	FC1	HC13	JC28	LCR93	LPS9	MGR36	NFIC12	PC12	RR5	SC4	SC84	SR24	TC13
American Coot							1.25												
American Crow	0.67	1.50	0.33	0.92	0.83	1.00	1.67	0.67	3.08	0.58	3.58	1.92	0.75		1.67	1.92	3.75	0.33	0.92
American Goldfinch		1.08	0.33	0.25	0.33	0.08	0.50	0.25	2.75	0.42	4.42		0.17	0.08	0.42	0.42	0.25	0.83	
American Robin	0.67	2.42	5.75	0.75	3.33	2.92	2.42	1.75	6.00	5.17	5.92	2.58	1.17	2.25	3.00	8.42	5.25	5.50	1.58
Anna's Hummingbird			0.08		0.08													0.08	
Bald Eagle						0.08	0.17												
Barn Swallow		0.08	0.50		0.42	0.17	2.17		0.17		0.33				0.33	0.17		0.17	
Black-capped Chickadee	0.42	2.25	1.42	0.50	1.00	2.67	2.17	1.58	2.42	2.25	4.75	1.58	0.58	1.25	2.50	3.25	1.92	4.75	1.08
Belted Kingfisher			0.58		0.08	0.17	0.25	0.08	0.08	0.08		0.08			0.08	0.08		0.17	
Bewick's Wren	0.08	0.33		0.17	0.58	1.42	1.42	0.17	1.00	0.50	2.17	0.08	0.25	0.25	0.25	1.92	1.17	1.25	0.17
Brown-headed Cow Bird	0.17	0.42	0.33	0.17	0.42	0.08	0.25	0.08	0.50	0.42	0.50	0.08		0.33	0.50	0.08	0.17	0.83	0.08
Black Headed Grosbeak	0.17	0.42	1.50	0.17	1.42	0.67	0.83	0.33	0.58	1.25	1.33	0.17	0.08	0.50	0.58	0.67	0.17	2.00	0.25
Brewer's Blackbird		0.33		0.08	0.17		0.50			0.08	0.17	0.08				0.08		0.08	
Brown Creeper			0.33		0.08			0.08		0.08				0.17	0.42	0.17		0.25	0.17
Black-throated Gray Warbler	0.08		0.17		0.08			0.75	1.17	0.75	0.17	0.08	0.42	0.08	0.50	0.58	0.50	1.25	0.25
Band-tailed Pigeon								0.25								0.08		0.33	
Bushtit		0.50	0.08	0.17	0.33	0.17	0.50	0.17	0.33	0.42	2.33	0.33	0.25	0.25	0.25	1.00	0.08	0.75	0.25
Blue-winged Teal					0.08		0.08												
Canada Goose						0.42	0.33								0.08				
California Quail				0.08	0.08				0.08							0.08			
Caspian Tern						0.08	0.25												
Chestnut-backed Chickadee	0.17	0.17	0.67	0.17	0.33	0.17	0.25	0.17	1.00	0.58	0.42	0.25	0.42	0.25	1.00	0.92	0.42	2.08	0.25
Cedar Waxwing	0.08	1.00	1.00		0.50	0.42	1.25	0.50	0.92	0.67	0.17		0.17	1.25	1.08	0.67	0.33	1.50	0.17
Chipping Sparrow									0.17		0.08		0.08	0.08				0.08	
Cliff Swallow		0.08		0.08			0.92												
Cooper's Hawk	0.08				0.17				0.08	0.33								0.08	
Common Raven										0.17								0.25	
Common Yellow-throat		0.17	2.92		1.58	0.08	0.92	0.67	1.25	3.08	1.58	1.42		2.17	1.75			0.83	
Dark-eyed Junco	0.17		0.08	0.17	0.17			0.25	1.00	0.08	0.33	0.08	0.42	0.08	0.33	0.75	0.25	2.17	0.50
Downy Woodpecker	0.08	0.08	0.25		0.08	0.08	0.33	0.17	0.17	0.17	0.58	0.50		0.17	0.33	0.25	0.33	0.83	0.17
European Starling		0.92			0.17	0.33	1.42		0.50	0.17	0.50	0.08				0.17	0.08		
Evening Grosbeak	0.08								0.58	0.08	0.08				0.25			0.25	
Fox Sparrow		0.17	0.08							0.25									
Gadwall							0.50								0.08				
Great Blue Heron		0.17	0.17		0.25	0.50	1.17			0.08	0.17	0.08		0.17	0.17	0.08		0.42	
Golden-crowned kinglet	0.25	0.25	0.42	0.25	0.17	0.33	0.08	0.92	0.75	0.42	0.25	0.50	0.67	0.17	1.00	0.83	0.58	0.83	0.42
Green Heron		0.17					0.67					0.08							
Glaucous-winged Gull						0.08	0.33				0.08								
Hammond's Flycatcher	0.08				0.25			0.17			0.08	0.08		0.25	0.42		0.33	0.08	
Hairy Woodpecker		0.08	0.67		0.17	0.08		0.42	0.08	0.33		0.33	0.25	0.08	0.67	0.25	0.50	1.00	0.25
Hermit Thrush	0.25		0.50		0.42		0.08	0.67	0.25	0.92	0.08	0.50	0.42	0.67	0.58	0.42	0.92	1.58	0.33
House Finch	0.17	0.25	0.33	0.33	0.42	0.50	0.25		0.08		0.42				0.08	0.08	0.17	0.17	
Hooded Merganser															0.75				
House Sparrow		0.17		0.08	0.08	0.17	0.25		0.08					0.08	0.08				

Appendix Table 6-2 continued. Detection rates for species within each wetland all years combined.

			Detection Rates																
Species	AL3	B3I	BBC24	ELS39	ELS61	ELW1	FC1	HC13	JC28	LCR93	LPS9	MGR36	NFIC12	PC12	RR5	SC4	SC84	SR24	TC13
Hutton's Vireo			0.17		0.17		0.08						0.08		0.42	0.50	0.17	0.08	0.17
Killdeer		0.08				0.17			0.25		0.08					0.08		0.08	
Mallard		0.17	1.25	0.08	0.83	0.17	3.75				0.17	0.58		0.17	0.75			0.33	
Marsh Wren			0.58	0.08		0.58	4.25		0.25	0.42	0.50	0.17		0.17	0.17	0.08	0.08	0.75	
MacGillivary's Warbler									0.17	0.25	0.08				0.08				0.08
Northern Flicker		0.33	0.83		0.33		0.08		0.08	0.17	0.08	0.33			0.42	0.33	0.42	0.25	
Northern Oriole			0.08				0.08					0.08			0.08				
Northern Pigmy Owl	0.08									0.08							0.08		
Orange-crowned Warbler		0.17	0.83	0.58	0.58	0.17	0.25	0.25	0.17	0.17	0.17			0.42	0.08	0.25	0.42	1.00	0.50
Olive-sided Flycatcher	0.08		0.08			0.08			0.33								0.50		0.17
Pied-billed Grebe					0.08	0.42	1.50								0.08			0.08	
Pine Siskin			0.08	0.08					0.58	0.08	0.08		0.08			0.08	0.33	0.17	
Pileated Woodpecker			0.42						0.08					0.08	0.58			0.08	
Pacific-slope Flycatcher	1.42	0.42	2.75	0.25	0.58	0.33	0.50	1.83	3.08	2.50	1.17	2.58	1.83	1.50	2.25	1.92	0.83	7.42	1.75
Purple Finch	0.08	0.17		0.17	0.25	0.08			1.25	0.50	0.58		0.33	0.25	1.25	0.83	0.33	0.83	0.08
Red-breasted Nuthatch	0.08		0.58	0.08	0.25	0.08		0.17	0.92	0.67	0.08		0.08	0.33	0.83	0.17	1.00	1.17	0.50
Red-breasted Sapsucker			0.08		0.08					0.08		0.08		0.08	0.17	0.08			
Red Crossbill			0.17		0.08	0.17		0.42	0.08	0.08	0.25	0.08	0.17	0.08	0.33	0.33	0.33	0.25	0.17
Red-eyed Vireo									0.08						0.08			0.17	
Red-eyed Vireo								0.17		0.25								0.25	
Rock Dove		0.17					0.25												
Rufous-sided Towee	0.25	0.83	0.33	1.00	1.17	1.58	0.17	0.33	2.08	0.42	4.58	0.17	2.00	1.33	0.75	5.42	2.58	1.75	1.00
Rufous Hummingbird			0.42	0.08	0.17		0.08							0.08				0.25	
Ruffed Grouse										0.08				0.08				0.25	
Ruby Crowned Kinglet	0.17	0.08	0.25	0.17	0.17			0.42	0.25	0.17	0.17	0.25	0.42	0.08	0.33	0.08	0.25	0.67	0.08
Red-winged Blackbird		0.08	13.33		9.67	1.00	9.50	0.25	0.08	2.75	6.33	5.42			0.42			0.33	0.17
Savannah Sparrow									0.17										
Sora			0.17				0.17			0.08									
Song Sparrow	1.17	4.08	10.33	1.58	3.17	4.33	5.92	3.92	6.00	6.50	14.42	3.92	1.83	3.50	4.58	9.50	6.83	10.08	1.50
Solitary Vireo	0.08		0.17		0.17		0.08		0.33		0.08			0.17		0.33	0.08	0.17	0.17
Spotted Sandpiper									0.17										
Sharp-shinned Hawk				0.08		0.08				0.08	0.08								
Steller's Jay	0.08	0.08	2.17		0.25	0.17		0.17	1.08	0.58	0.17		0.25	0.75	1.00	0.67	0.75	2.50	0.50
Swainson's Thrush	1.42	0.67	2.08	0.42	1.00	0.33	0.75	2.75	5.50	8.58	2.08	3.33	2.17	2.67	4.17	4.17	3.33	7.17	3.08
Townsend's Warbler	0.33		0.33	0.17	0.08	0.08		0.17	0.83	0.17		0.08	0.25		0.67	0.25	0.42	0.50	0.08
Tree Swallow	0.08	0.25	0.67	0.25	0.25	0.50	3.17		0.67		0.25	0.08	0.08	0.08	1.25	0.17	0.08	0.58	
Varied Thrush				0.08	0.08		0.33											0.08	
Vaux's Swift	0.25		0.25		0.17	0.25	0.17	0.75		0.08	0.08		0.08		0.33	0.08		0.42	0.25
Violet-green Swallow			1.33	0.08	0.50	0.42	1.17	0.08	0.83	0.17	0.67		0.17	0.25	0.17	0.33	0.25	0.75	
Virginia Rail			0.50		0.08		0.33			0.08		0.17		0.08					
Warbling Vireo		0.08	0.67	0.08	0.17	0.08	0.25	0.08	0.08	1.50	0.58	0.08			0.58	0.17	0.25	0.58	
White-crowned Sparrow	0.08			0.42	0.50		0.08		0.67							0.25			
Western Tanager	0.17		0.75					0.33	0.42	0.50	0.08			0.25	0.25	0.17	0.08	1.25	0.17
Western Wood-pewee	0.25		0.25		0.42	0.17		0.08						0.33	0.50		0.08		0.42

Appendix Table 6-2 continued. Detection rates for species within each wetland all years combined.

	Detection Rates																		
Species	AL3	B3I	BBC24	ELS39	ELS61	ELW1	FC1	HC13	JC28	LCR93	LPS9	MGR36	NFIC12	PC12	RR5	SC4	SC84	SR24	TC13
Willow Flycatcher	0.17	0.42	2.75	0.75	1.92	0.58	1.50	1.50	1.08	4.25	3.33	1.75	0.17	0.75	1.83		0.58	4.08	1.33
Wilson's Warbler	0.58	0.17	1.67	0.33	0.17	0.17	0.75	1.83	1.08	4.50	0.08	1.50	0.75	0.67	3.08	0.67	0.25	2.75	0.92
Winter Wren	1.67	0.25	3.92	0.25	0.25	0.58	0.50	1.42	2.83	1.33	0.42	1.08	0.33	1.08	2.42	0.92	0.42	4.00	2.00
Wood Duck			0.33		0.17	0.17	0.08			0.25		0.08	0.08	0.08	0.08		0.17	0.25	0.08
Yellow Warbler	0.25	0.67	0.92	0.25	0.42	0.83	1.08	0.25	0.67	3.17	0.50		0.25	0.17	0.25	0.58	0.17	1.33	0.17
Yellow-rumped Warbler		0.08	0.08					0.33	0.08		0.08		0.08	0.08		0.25		0.08	

CHAPTER 7 SMALL MAMMAL DISTRIBUTION, ABUNDANCE AND HABITAT USE

by Klaus O. Richter and Amanda L. Azous

INTRODUCTION

Small mammals are an integral component of most ecosystems. In the Northwest the regional distribution of small mammals has been described by Ingles (1965) and Maser et al. (1981). Within unmanaged (e.g., old growth) Douglas-fir forests, small mammals were described in Aubry et al. (1991) by numerous biologists. Mammals in second growth forests under differing cutting practices and intensity of landscaping and development were described by (Stofel 1993). Local small mammal species and distributions in urban parks varying in size of approximately four to 400 ha within urbanizing areas were described by (Gavareski 1976).

The distribution and abundance of small mammals, similar to that of macroinvertebrates, amphibians and birds may be indicators of the environmental health of wetlands. They also exhibit the ability to shape wetlands through their influence on soil, water and plants. Several species such as Trowbridge's shrew, marsh shrew, shrew-mole, western red-backed vole and creeping vole (See Table 7-1 for scientific names) are endemic to the Pacific Northwest (Corn and Bury 1991) and could be expected at pristine wetlands. Our objective in this chapter is to present the relative distribution and abundance of small mammals across the wetlands we studied. We also examine wetland conditions such as size, hydrology and vegetation complexity to gain insight into habitat characteristics important for maintaining diversity and unique species.

METHODS

We used pitfall and Sherman trap captures during autumn (mid-October to mid-November) as indicators of small mammal distributions. We installed traps along two 250-meter transects on opposite sides of each wetland. A combination of 10 pitfalls and 25 Sherman traps at 10 meter intervals was used without drift fences. To minimize the ejection of pitfalls due to hydrostatic pressure, transects were located above winter high-water levels. Pitfalls locations and trap installation procedures are described elsewhere (Richter 1995). Pitfalls were operated for a total of 14, mostly consecutive, days and Shermans for total of six days (alternating between wetlands for three consecutive days). We closed and removed traps vandalized or disturbed by dogs, cats, raccoons and other mammals and continued trapping after several days, when predators were no longer expected at traps. At wetlands in which trap nights were less than attempted (because of ongoing disturbance), captures were adjusted by calculating rates on available traps which was assumed to have been the total number set less one half the number of traps unavailable (Sherman's closed with no captures or treadle stuck; pitfall disturbed by dogs or wildlife), and extrapolated to the full monitoring period (Nelson and Clark 1973) and specifically noted within our discussion. We also relocated traps that became permanently flooded during our study to higher ground where possible. We used wood stakes to mark the beginning and end of transacts and blue flagging to distinguish the trap sites.

All small mammals were identified to species. Deer mice and forest deer mice were distinguished from each other by tail lengths in which adults with tails exceeding 96 mm were identified as deer mice as opposed to forest deer mice with tails less than or equal to 96 mm. Additionally, we aged all deer mice species as adults and subadults (coarse-brown versus soft-gray pelage and weight), sexed and marked by cutting the "pencil-hairs" from the tip of tails, allowing us to determine recaptures and hence rough indexes of abundances for this taxa. The high mortality of shrews in pitfall traps also enabled us to use their capture data in population estimations since recapture rate was low.

We compared the number of National Wetlands Inventory (NWI) (Cowardin et al. 1979) vegetation associations with the diversity of mammal communities. We looked at wetland size and land use, including degree of urbanization and amount forest land within 1000 meters of the wetland. Quantification of these habitat and landscape characteristics are described in the amphibian and bird chapters (five and six) of this report.

RESULTS

We captured a total of 21 small mammal species, 19 of which are native within the wetlands censused (Table 7-1), excluding Norway rat and black rat. The range of species diversity among wetlands varied widely from a low of just one species in ELW1, Norway rat, to a high of 13 species (70% of observed native species) in LCR93 (Figure 7-1).

Table 7-1. Small mammals captured and observed in palustrine wetlands of the Puget Sound Basin.

Common Name	Scientific Name	Ames Lake 3	Bellevue 3I	Big Bear Creek24	East Lake Sammamish 39	East Lake Sammamish 61	East Lake Washington 1	Forbes Creek 1	Harris Creek13	Jenkin's Creek 28	Lower Cedar River 93	Lower Puget Sound 9	Middle Green Rivers 36	North Fork Issaquah Creek 12	Patterson Creek 12	Raging River 5	Snoqualmie River 24	Soos Creek 4	Soos Creek 84	Tuck Creek 13
Black Rat	*Rattus rattus*		●									●		●						
Bushy-tailed Woodrat	*Neotoma cinerea*											●								
Creeping vole	*Microtus oregoni*	●	●	●	●	●		●	●		●		●	●	●	●		●		●
Deer Mouse	*Peromyscus maniculatus*	●	●	●	●	●		●	●	●	●	●	●	●	●	●	●	●	●	●
Douglas Squirrel	*Tamiasciurus douglasii*								●				●							
Ermine	*Mustela erminea*								●		●		●							
Forest Deer mouse	*Peromyscus oreas*	●	●	●	●	●			●	●	●	●	●	●	●	●	●	●	●	●
Long-tailed Vole	*Microtus longicaudus*			●							●	●		●						
Marsh Shrew	*Sorex bendirei*			●					●		●		●			●			●	●
Masked Shrew	*Sorex cinereus*										●									
Montane Shrew	*Sorex monticolus*				●	●			●		●	●	●		●	●			●	
Northern Flying Squirrel	*Glaucomys sabrinus*			●																
Norway Rat	*Rattus norvegicus*						●													
Pacific Jumping Mouse	*Zapus trinotatus*		●	●												●				
Shrew-mole	*Neurotricus gibbsii*			●					●		●		●		●	●	●			●
Southern Red-backed Vole	*Clethryonomys gapperi*		●		●						●								●	
Townsend's Chipmunk	*Eutamias townsendii*			●	●				●		●								●	●
Townsend's Vole	*Microtus townsendii*				●	●						●			●			●		
Trowbridge's Shrew	*Sorex trowbridgei*	●	●	●	●	●			●	●	●	●	●	●	●	●	●	●	●	●
Vagrant Shrew	*Sorex vagrans*	●	●	●	●	●		●	●		●	●	●	●	●	●		●	●	●
Water Shrew	*Sorex palustris*																			●

Sites severely altered by urbanization, and harboring minimal populations of native species, include ELW1 and FC1. Surprisingly, B3I, a small wetland almost totally surrounded by urbanization and containing black rats had seven native mammal species. Several wetlands were visited by free ranging dogs (BBC24), unidentified animals (most likely dogs, opossum, raccoon (LPS9), and bear and cougar (RR5), whose activities disrupted our trapping program.

Small mammal richness ranged widely between study years, shown for native species in Figure 7-2. For example, LCR93, which had the highest number of species over the whole study, had at least ten native species the first year, 1988, yet only five native species were collected or observed in 1993 and 1995. At another wetland, HC13, we identified eight, nine and seven species respectively in 1988, 1989 and 1995, yet in 1993, only three species were captured.

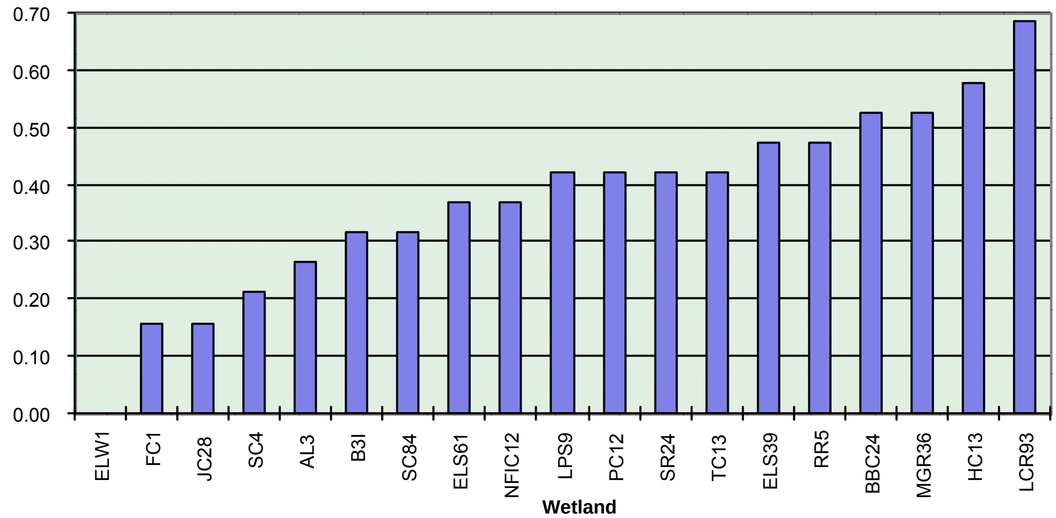

Figure 7-1. Proportional native species richness among 19 palustrine wetlands of the Puget Sound Region.

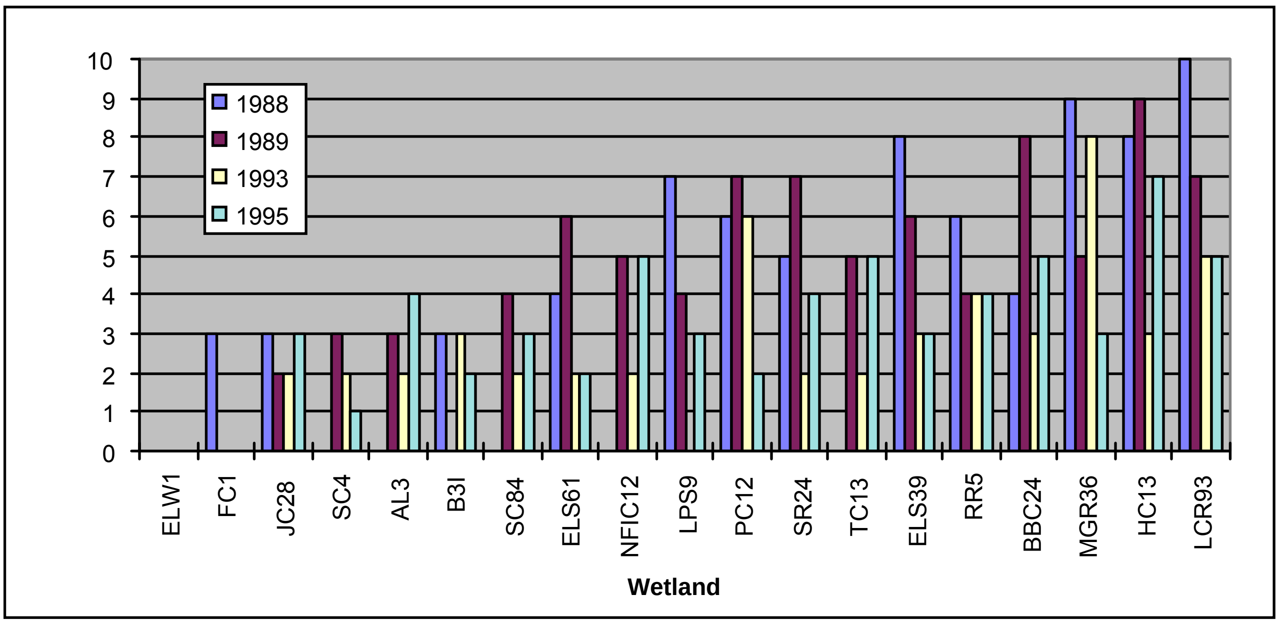

Figure 7-2. Native small mammal richness from 1988 to 1995 in study wetlands.

The abundance of deer mice and shrews varied widely between wetlands and between years. Table of capture rates of species by wetland and by year are available in Appendix Table 7-1. The deer mouse was by far the most abundant mammal captured in all years over all wetlands (Figure 7-3). The Montane shrew and forest deer mouse were the next most abundant and were captured in substantially fewer numbers than the deer mouse. The rarest capture was of the masked shrew, a fairly uncommon species in this area. The most unusual capture was that of the northern flying squirrel, traditionally an arboreal species and consequently unlikely to be captured in traps on the ground.

Wetland size, by itself, was not found to be significant to mammal richness or abundance (measured as number of captures per 100 trap nights). This result was expected for abundance, however, we expected mammal richness to be strongly related to wetland size, since intuitively, one would expect larger wetlands to have more niches and habitat opportunities. But wetland size was not by itself a major factor and neither were the number of NWI habitat classes. However, the total area of adjacent development was found to be weakly correlated with mammal richness ($R = 0.4$, $p = 0.09$). Though adjacent development was a factor, more critical to highly diverse mammal communities was the percent of forest land immediately adjacent to the wetland within 500 to 1000 meters ($R \geq 0.55$, $p \geq 0.02$) (Figure 7-4). Forest land included all deciduous and coniferous forest and also included lands with single family dwellings within forested parcels. We found that wetlands were more likely to have diverse mammal communities if a substantial part of the adjacent land was not cleared and was retained in forest land.

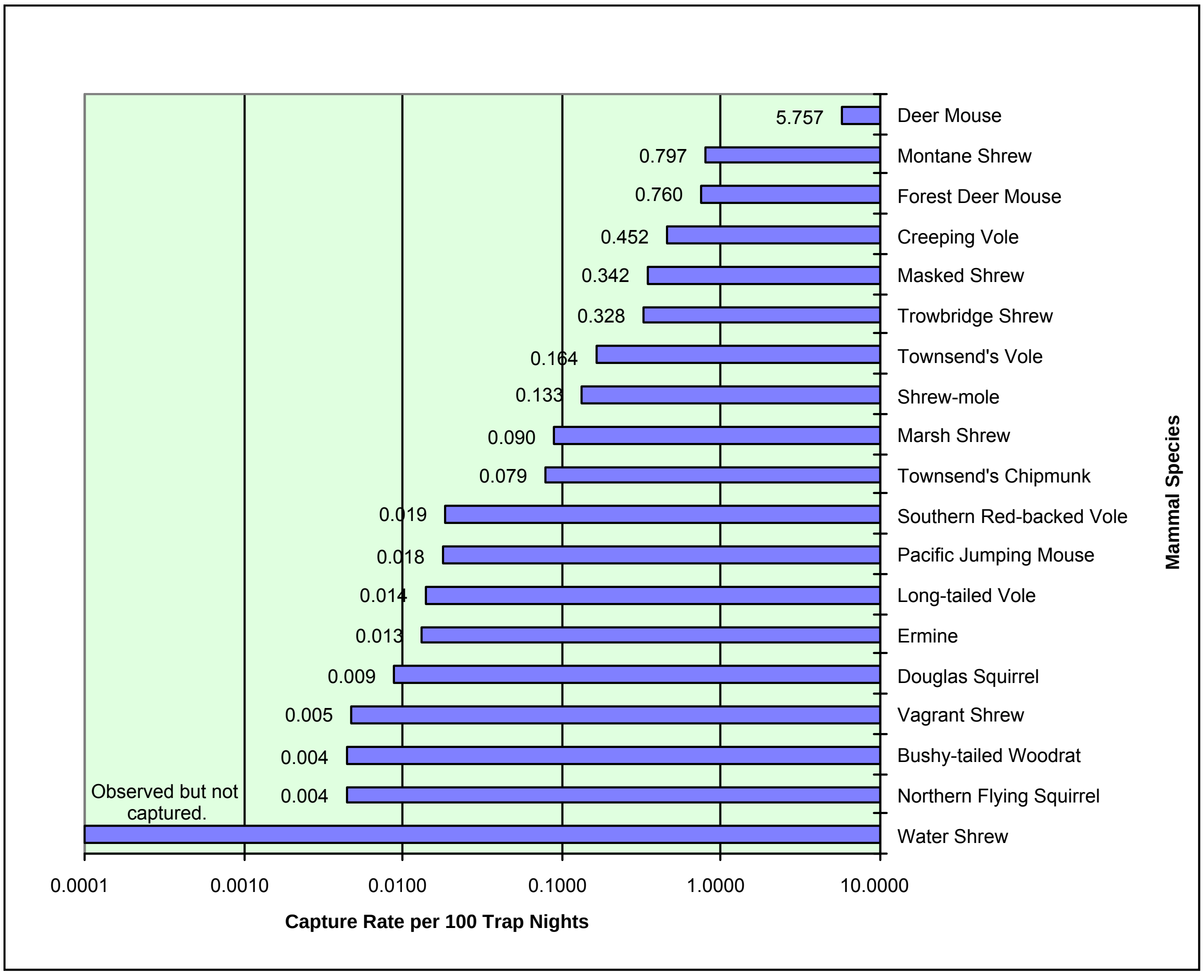

Figure 7-3. Capture rates of small mammals.

Large woody debris in the wetland buffer was also found to be a factor related to diversity. Small mammal richness was found to be associated with the combined factors of wetland size, adjacent land use and the relative quantity of large woody debris within the wetland buffer (Svendsen and Richter in prep.) (Figure 7-5).

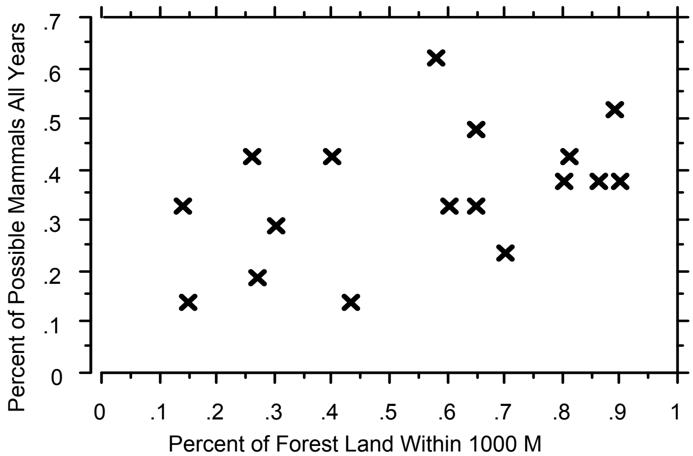

Figure 7-3. Relationship between small mammal diversity and forest land within 1000 meters of wetland.

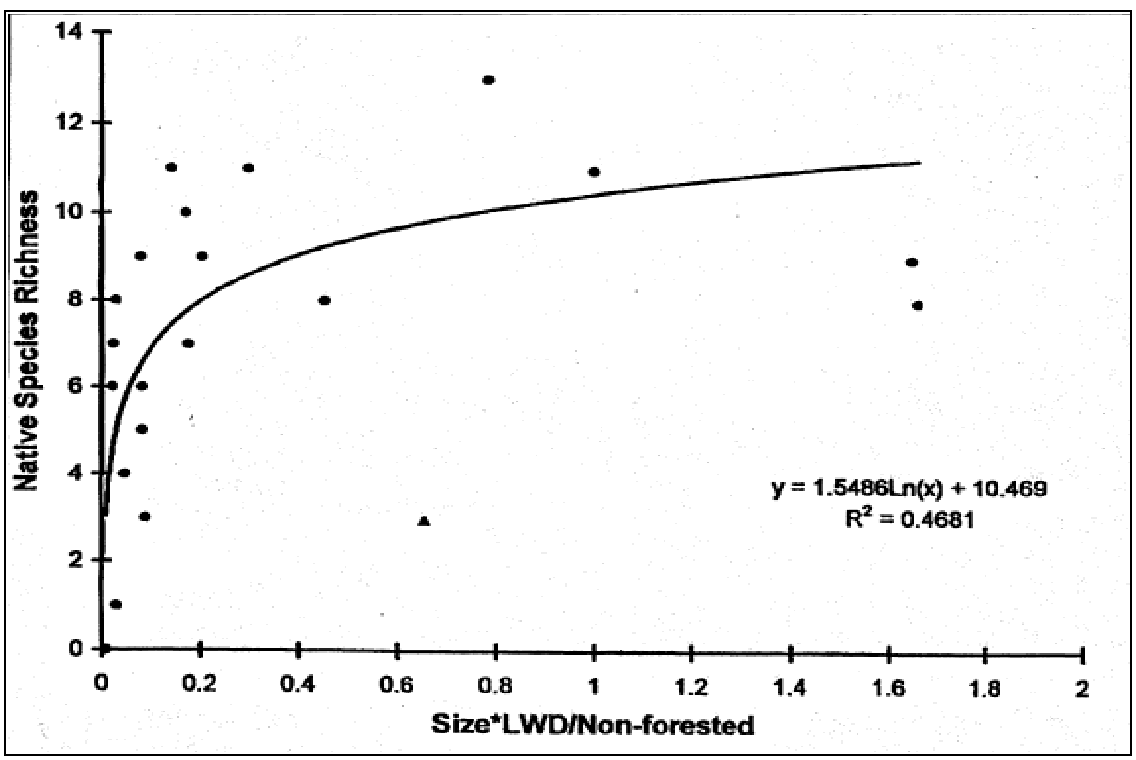

Figure 7-5. Relationship between small mammal species richness and habitat variables including wetland size, land use cover and large woody debris.

DISCUSSION

This study shows that the small mammal communities of wetlands are among the most diverse communities of mammals in the Puget Sound Basin. We captured 22 species (19 native), significantly more than in second-growth forests (Stofel 1993) and urban parks (Gavareski (1976). Because Northern flying squirrels, Douglas squirrels ermine, chipmunk and shrew moles are not well sampled by either pitfalls or Shermans, their true distribution and abundance remains unknown. We captured rats and mice (Muridae), which surprisingly Gavareski (1976) did not capture during her studies of urban parks. On the other hand, we did not capture or observe other non-native species of urban areas including Eastern-cottontail, (Sylvilagus floridanus), Fox squirrel (Sciurus niger) and Eastern gray squirrel (S. carolinensis).

Because of their numbers, deer mice most likely play important roles in trophic dynamics of palustrine wetlands. They appear to inhabit wetlands both in average years as well as in severe years, whereas other small mammals were not consistently captured.

Norway rats may be more damaging to native mammals than black rats in that wetlands with Norway rats appear to displace native species. This presumably happened in ELW1 and another wetland in a heavily urbanized landscape of Snohomish County studied by Svendsen and Richter (In press).

Perhaps one of the more significant findings is the importance of forest land and its consequent habitat component of large woody debris within the wetland buffer. Earlier statistical models that included the presence of vegetation structure (number of vegetation layers e.g., herb, shrub and tree cover), as well as the presence of development and its associated human and animal impacts (e.g., rats, cats and dogs) did not show the strong relationship that forest land and the presence large-woody debris exhibited. Consequently, this result suggests that a certain amount of development can occur and non-native mammals can be tolerated if enough forest land remains available for cover, food, shelter and microclimatic relief. Forest land can provide continuous production of large logs and tree stumps that provide habitat over time. These findings also point out the value of conserving and maintaining large woody debris in wetlands and wetland buffers to increase opportunities for small mammal habitat.

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Appendix Table 7-1. Pitfall capture rates of small mammals by wetland, species and year of capture.

Code	CLGA		EUTO			GLSA	MILO		
Species	Clethryonomys gapperi		Eutamias townsendii			Glaucomys grapperi	Microtus longicaudus		
Common Name	Southern Red-backed Vole		Townsend's Chipmunk			Northern Flying Squirrel	Long-tailed Vole		
Number of Captures per 100 Trap Nights									
SITE_ID	1988	1993	1988	1989	1995	1989	1988	1993	1995
AL3									
B3I	0.36								
BBC24						0.34		0.72	
ELS39	0.36		0.33	0.33	1.33				
ELS61									
ELW1									
FC1									
HC13				1.00					
JC28									
LCR93	0.33		2.33	0.67			0.36		
LPS9							0.33		
MGR36									
NFIC12									0.36
PC12									
RR5									
SC4									
SC84									
SR24		0.36		0.33					
TC13					0.33				

Code	MIOR				MITO				MUER	
Species	Microtus oregoni				Microtus townsendii				Mustela erminea	
Common Name	Creeping Vole				Townsend's Vole				Ermine	
Number of Captures per 100 Trap Nights										
SITE_ID	1988	1989	1993	1995	1988	1989	1993	1995	1988	1989
AL3				0.33						
B3I			0.36							
BBC24	1.37	1.01		2.15						
ELS39	5.95	1.00			7.24	2.07	0.36			
ELS61	1.36	0.33				0.33				
ELW1										
FC1	10.29									
HC13	4.02	0.67							0.33	
JC28										
LCR93	0.69			0.71						0.33
LPS9					1.05					
MGR36	6.90	0.69	0.36						0.33	
NFIC12		0.71		1.07						
PC12	1.71	1.33			1.69	1.33				
RR5	2.42									
SC4										
SC84		0.36				0.36				
SR24										
TC13		1.02								

Appendix Table 7-1. Pitfall capture rates of small mammals by wetland, species and year of capture.

Code	NECI	NEGI			PEMA			
Species	Neotoma cinerea	Neurotricus gibbsii			Peromyscus maniculatus			
Common Name	Bushy-tailed Woodrat	Shrew-mole			Deer Mouse			
Number of Captures per 100 Trap Nights								
SITE_ID	1989	1988	1989	1993	1988	1989	1993	1995
AL3						15.98	4.29	2.00
B3I					0.67		0.71	3.33
BBC24					3.70	14.48	0.72	13.01
ELS39					9.69	16.69	0.71	11.33
ELS61					8.00	12.38		2.02
ELW1								
FC1					8.67			
HC13			0.71		12.38	20.00	0.36	3.36
JC28					15.69	11.33	0.36	9.02
LCR93			0.36		32.40	7.33	2.86	8.67
LPS9	0.69				32.24	23.38		23.02
MGR36		0.36		0.36	7.79	14.74	4.64	9.05
NFIC12						9.64	0.36	2.33
PC12				1.07	3.69	11.05	5.00	9.02
RR5				3.21	8.42	10.11	1.79	4.05
SC4				0.36		14.76	0.36	23.00
SC84						73.50		16.33
SR24					1.67	19.02		2.00
TC13						0.67	1.43	3.38

Code	PEOR				RARA			SOBE		
Species	Peromyscus oreas				Rattus rattus			Sorax bendirei		
Common Name	Forest Deer Mouse				Black Rat			Marsh Shrew		
Number of Captures per 100 Trap Nights					1988	1989	1995	1988	1989	1993
SITE_ID	1988	1989	1993	1995						
AL3		1.67		3.33	1.33					
B3I				2.00					0.36	
BBC24	0.34	4.37		1.68						
ELS39				1.33						
ELS61	1.00	0.67		0.33						
ELW1										
FC1								0.71	0.36	
HC13	7.00	7.33		0.69						
JC28	6.33	1.00		0.67				0.71	1.07	
LCR93	5.36	2.67	0.36	0.67			1.67			
LPS9	1.33	0.33							0.36	1.43
MGR36	2.00	2.67	0.36	0.67						
NFIC12				0.67						
PC12		1.33	0.36	2.67				0.71		
RR5	1.68	1.34		0.34						
SC4		0.33								
SC84		6.02	0.36	1.67					0.36	
SR24	1.00	2.33		0.33					0.36	
TC13		0.33								

Appendix Table 7-1. Pitfall capture rates of small mammals by wetland, species and year of capture.

Code	SOCI	SOMO				SOTR			
Species	Sorex cinereus	Sorex monticolus				Sorex trowbridgei			
Common Name	Masked Shrew	Montane Shrew				Trowbridge Shrew			
	Number of Captures per 100 Trap Nights								
SITE_ID	1988	1988	1989	1993	1995	1988	1989	1993	1995
AL3								1.79	1.07
B3I								1.79	
BBC24						0.69	2.13	3.21	1.79
ELS39		1.38	1.38			0.36		0.36	
ELS61			2.79			1.05		0.71	
ELW1									
FC1									
HC13		0.36	1.79	0.36	1.79	0.71	0.36	0.71	1.07
JC28						0.67		1.07	0.36
LCR93	0.36	1.07		0.36		1.07	1.79	3.93	0.36
LPS9		0.69				2.45	2.76		0.36
MGR36		1.07		1.43		1.02	1.07	1.79	2.86
NFIC12			0.69				1.05	0.36	0.36
PC12		2.79	2.83	0.36		0.69	3.19	2.86	
RR5		0.34				3.21	1.05	2.14	
SC4							0.36		
SC84									0.71
SR24		0.36	0.71		0.36	0.71	3.21	2.14	1.43
TC13							0.36	0.71	1.79

Code	SOVA				TADO		ZATR	
Species	Sorax vagrans				Tamiasciurus douglasii		Zapus trinotatus	
Common Name	Vagrant Shrew				Douglas Squirrel		Pacific Jumping Mouse	
Number of Captures per 100 Trap Nights								
SITE_ID	1988	1989	1993	1995	1988	1995	1989	1995
AL3		2.12						
B3I	0.36							
BBC24		0.72					0.69	
ELS39	2.12	0.69						
ELS61		0.33	0.36					
ELW1								
FC1	0.33							
HC13	0.36	0.36		0.69		0.33		
JC28								
LCR93			0.36	0.33				
LPS9	1.40			0.71				
MGR36	0.69		1.43		0.33			
NFIC12		0.36						
PC12	1.07	1.76	2.14					
RR5		0.34	0.71					0.67
SC4								
SC84			0.36					
SR24	0.36	3.19						
TC13				1.07				

Section 3 Functional Aspects of Freshwater Wetlands in the Central Puget Sound Basin

CHAPTER 8 EFFECTS OF WATERSHED DEVELOPMENT ON HYDROLOGY

by Lorin E. Reinelt and Brian L. Taylor

INTRODUCTION

In urbanizing areas, the quantity (peak flow rate and volume) of stormwater can change significantly as a result of developments in a watershed. Increases in stormwater may result from new impervious surfaces, removal of forest cover, and installation of constructed drainage systems. Watershed development can also cause reduced recharge of groundwater and baseflow to streams, and less evapotranspiration.

Changes in hydrology, whether brought about intentionally or incidentally, have an influence on wetland systems. Wetlands will likely have a positive effect on downstream areas by dampening stormflows before discharging to streams and lakes. However, wetlands may also be adversely impacted by these same higher peak flows and volumes. For cases where wetlands are the primary receiving water for urban stormwater from new developments, it is hypothesized that the effects of watershed changes will be manifested through changes in the hydrology of wetlands.

Wetland hydrology is often described in terms of its hydroperiod, the pattern of fluctuating water levels resulting from the balance between inflows and outflows of water, landscape topography, and subsurface soil, geology, and groundwater conditions (Mitsch and Gosselink, 1993). Hydroperiod alterations are the most common effect of watershed development on wetland hydrology. This usually involves increases in the *magnitude*, *frequency* and *duration* of wetland water levels. In other words, increased stormwater flows tend to cause higher wetland water levels, on more occasions during the wet season, and for longer periods of time. These changes in wetland hydroperiod then result in impacts to plant and animal communities that were adapted to the pre-existing hydrologic conditions.

Puget Sound Wetlands and Stormwater Management Research Program

Palustrine wetland hydrology was studied as part of both components of the research program: (1) the study of the long-term effects of urban stormwater on wetlands, and (2) the study of the water-quality benefits to downstream receiving waters as urban stormwater flows through wetlands. This chapter presents results from the statistical analysis of 19 study wetlands from the long-term effects study, and from the water balance of two wetlands from the water-quality benefit study.

Research Objectives

The primary objective of this portion of the research program was to examine the effects of urban stormwater on wetland hydrology. However, there were also a variety of specific hydrologic questions addressed throughout the research which developed into the following specific objectives:

1. Identify the wetland and watershed hydrologic processes, and the factors governing these processes.
2. Determine how urban catchments behave differently from forested catchments.
3. Determine the percent contribution of wetland hydrologic inputs and outputs.
4. Relate wetland hydrologic conditions to wetland/watershed characteristics.
5. Characterize wetland hydroperiods and develop a set of dependent variables for analysis.

DATA COLLECTION AND ANALYSIS METHODS

As noted in Chapter 1, a conceptual model was used to show the relationship between factors influencing wetland and watershed hydrologic processes and the wetland hydroperiod (Figure 1-4). In the conceptual model, some of the key factors thought to influence wetland water level fluctuation included: (1) forested area, (2) impervious area, (3) wetland morphology, (4) outlet constriction, (5) wetland-to-watershed area ratio, and (6) watershed soils. Statistical analyses were carried out to determine which factors were most important.

Statistical Analysis of Development Impacts on Wetland Hydrology

A variety of graphical and statistical techniques were used in identifying relationships between the watershed or wetland characteristics and wetland hydroperiod (Taylor, 1993). Microsoft EXCEL was used in processing the data and SYSTAT was used for statistical analyses.

Graphical Analysis

The objective of the graphical analysis was to identify trends and threshold levels that could then be statistically tested to determine which statistical methods (parametric or nonparametric) were appropriate. Graphical analysis provided insights into which factors correlated to specific aspects of the hydroperiod; however, it failed to show the effects of multiple factors or varying importance simultaneously.

Normality Testing

In order to determine which statistical tests were appropriate for a given hypothesis, the normality of the data was assessed. The Kolmogorov-Smirnoff test was used to compare the maximum difference between two cumulative distributions. The Lilliefors test was used when the mean and variance of the distribution were unknown, in order to automatically standardize the variables and test whether the standardized distributions were normally distributed (Wilkinson, 1990). The Lilliefors test was used to assess the distribution of water level fluctuation measurements. The significance level used in testing normality was alpha equal to 0.05.

Threshold Testing

Threshold testing was done when a scatterplot suggested one or more threshold levels in the response of wetland water level fluctuations to a specific watershed or wetland characteristic. The data were grouped categorically based on thresholds suggested in the scatterplots. These groups were compared in a test of the null hypothesis that all groups were from equivalent distributions.

Because the water level fluctuation measurements were not normally distributed for all of the study sites, nonparametric tests were used: the Mann-Whitney test for two groups and the Kruskal-Wallis test for more than two groups. These two tests are analogous to the independent groups t-test for normally distributed data, but are based on data ranks rather than the data values (Zar, 1984; Wilkinson, 1990). The Kruskal-Wallis test will reject the null hypothesis if *any* of the groups are significantly different; nonparametric multiple comparisons were done to identify *which* groups were significantly different (Zar, 1984). The significance level used in evaluating thresholds was alpha equal to 0.05.

Multivariate Regression Models

Multivariate, least squares, linear regression models were calibrated to the study data to show how various wetland and watershed factors combine to effect wetland hydroperiod (Taylor, 1993). Models were developed by: (1) using step regression to identify factors important to the aspect of wetland hydroperiod being investigated, (2) determining the best way to quantify or express this factor, (3) evaluating model fit, and (4) examining the sensitivity to the predictor variables. The data for each wetland were weighted by sample size when appropriate; mean water level fluctuation was weighted by the total number of observations used in its calculation while the length of the dry period and seasonal water level fluctuations were weighted by the number of years used in their calculation.

The fit of the regression models was evaluated through various methods: the coefficient of determination (r^2) and the F-ratio, which compares the explanation provided by each predictor to the residual associated with each observation. The final step in the generation of the multiregression models was to examine the sensitivity of each predictor variable. The standardized coefficient of each predictor variable provides a way to compare the significance of the variables (Wilkinson, 1990). Additionally, variables were removed from the final model one at a time to determine their effect on the model r^2 and the standard error of the estimate.

Data Collection and Analysis for the Wetland Water Balance

In the detailed study of two wetlands (Bellevue 3I and Patterson Creek 12), a complete water balance was performed (Reinelt et al., 1993). This consisted of independent measurements of the following components: precipitation, evapotranspiration, surface inflow, surface outflow, groundwater exchange, and change in wetland storage. Precipitation was measured using an event recorder connected to a tipping-bucket gauge that recorded each 0.25 mm of rainfall. Continuous water flow measurements were taken at the inlet and outlet of the two wetlands using a variety of different techniques (Reinelt et al., 1990).

Shallow (1.2 to 4 m) and deep (6 to 18 m) piezometers were installed at both wetlands to aid in the estimation of groundwater flow using Darcy's Law (see Chapter 1). The hydraulic conductivity (K) of the underlying aquifer at both wetlands was determined using variable head pump and slug tests as described by Cedergren (1978) and Chapuis (1989). Piezometric head measurements were taken regularly to determine the hydraulic gradient (Surowiec, 1989). Control volumes were defined around each wetland to facilitate estimation of the horizontal and vertical components of groundwater flow.

Evapotranspiration was estimated from pan evaporation data from the Washington State University Extension Service Puyallup station representing the Puget Sound Lowlands region. Adjustments were made for differences between pan evaporation, open-water evaporation, and evapotranspiration by plants. Daily changes in wetland water depth (and corresponding storage volume) were estimated by correlating daily outflow data with regular gauge (water depth) readings. Storage volumes were determined for different water levels by multiplying the areal water coverage by water depth.

Identifying and describing seasonal differences in the hydrologic balance of the two wetlands was one objective of the study. Seasons were defined and analyzed by two classification methods. The first method included simply wet (October - March) and dry (April - September) seasons. The second method defined four seasons based on the climate of the Puget Sound region: wet (November-February), dry (June-September) and two transition (March-May; October) seasons. The division of data by season allowed for comparison of changes in the relative contributions of different inputs and outputs.

RESULTS AND DISCUSSION

Wetland Hydrology and Water Level Fluctuation

Three parameters were used to examine hydrologic conditions in the wetlands: water depth, water level fluctuation (WLF), and length of summer dry period. The minimum, maximum and range of water depths at the gauges are given in Table 8-1. Also given are the mean (according to equation 4 of Chapter 1) and maximum WLF, and days of summer drying in the wetland. Water depth and WLF varied widely for the 19 wetlands.

Table 8-1 Wetland watershed, outlet and hydrologic characteristics.

Wetland Name	Forest (%)	Imperv. Area (%)	Outlet Constr.	Range of Water Depth (m)	Mean WLF (m)	Max. WLF (m)	Mean Dry Period (days)	Calculated Mean WLF (m) Using Multiple Regression
AL3	73.9	3.4	1	0.00-0.62	0.07	0.31	101	0.21
MGR36	88.8	2.7	0	0.13-0.74	0.07	0.26	0	0.08
JC28	34.4	19.3	0	0.00-0.32	0.08	0.17	74	0.14
RR5	62.4	3.2	0	0.02-0.52	0.09	0.24	0	0.11
SC4	46.1	11.8	0	0.00-0.30	0.10	0.15	125	0.13
SR24	100.0	2.0	0	0.00-0.67	0.11	0.23	32	0.07
NFIC12	100.0	2.0	1	0.00-0.53	0.13	0.30	189	0.17
ELS61	0.0	3.9	0	0.05-0.84	0.14	0.33	0	0.19
PC12	75.2	3.9	1	0.20-1.19	0.14	0.84	0	0.20
BBC24	89.5	2.8	0	0.07-0.60	0.14	0.20	0	0.08
TC13	100.0	2.0	0	0.00-0.72	0.16	0.31	156	0.07
ELW1	0.0	19.9	0	0.00-0.66	0.22	0.44	19	0.19
HC13	76.6	3.6	1	0.09-1.56	0.24	0.41	0	0.20
SC84	20.1	15.9	0	0.00-1.08	0.26	0.53	62	0.16
FC1	14.7	30.8	0	0.11-1.01	0.28	0.62	0	0.38
LCR93	44.1	3.9	1	0.00-0.81	0.28	0.57	61	0.24
ELS39	0.0	28.0	1	0.00-1.61	0.46	1.29	151	0.51
B3I	0.0	54.9	1	0.63-2.37	0.57	1.54	0	0.51
LPS9	0.0	21.8	1	0.00-1.72	0.60	1.47	85	0.51

The largest range of water levels, as well as mean and maximum WLFs were found at B3I and LPS9, where the basins have among the highest percent of impervious area of any of the study sites and the wetland outlets are constricted (see B3I and LPS9 in Figure 8-1). Those wetlands with 90 percent or more forested cover and less than 3 percent impervious surfaces generally exhibited lower water ranges and low WLFs (see BBC24 and SR24 in Figure 8-1). As can be seen from Figure 8-1, these trends of low or high WLF are independent of whether the base level condition in the wetland is stable or fluctuating. Wetland JC28 was an exception to the normal relationship between high impervious area and high WLF; this was because the watershed soils are predominantly glacial outwash (highly permeable soils), thus reducing runoff volumes.

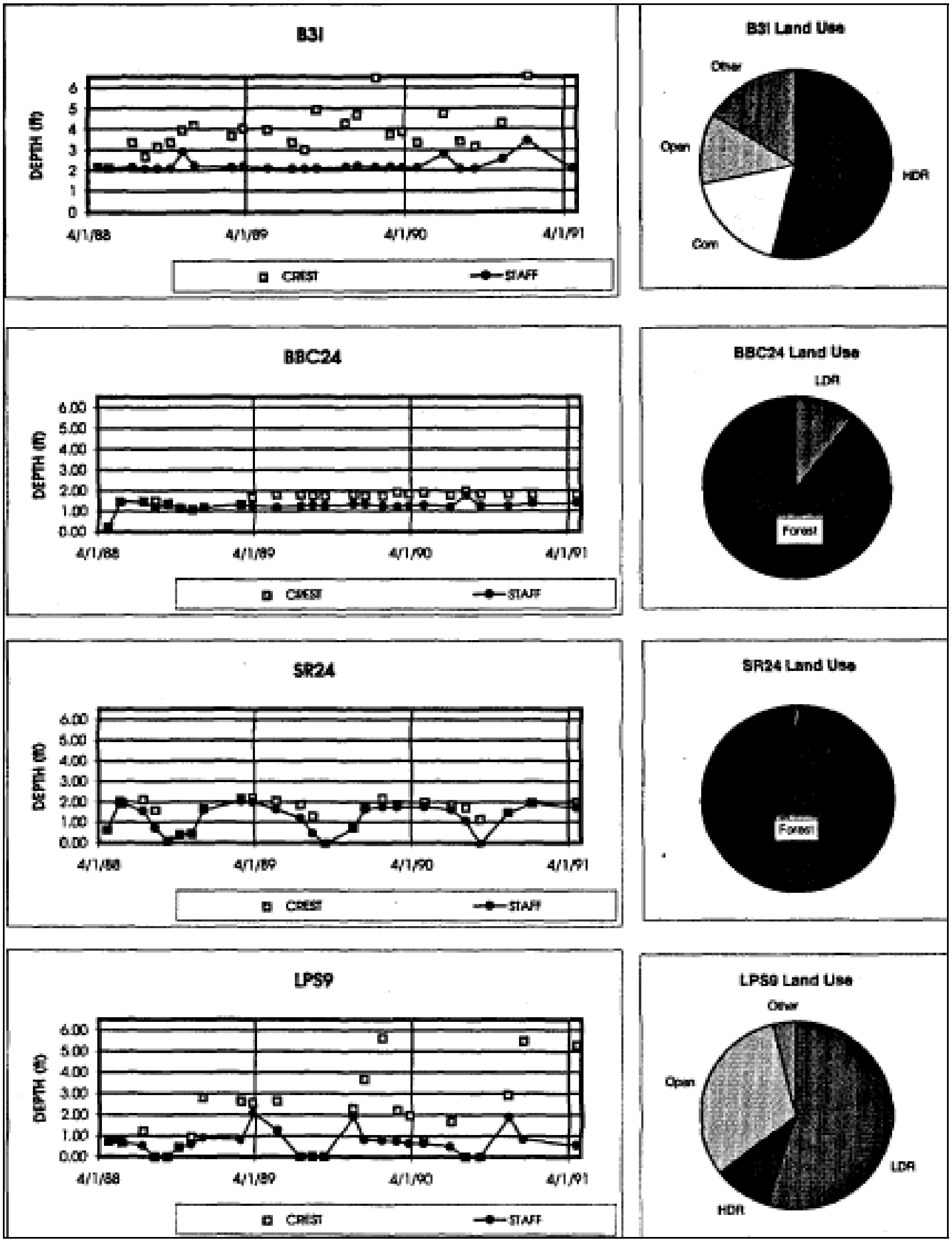

Figure 8-1. Wetland hydrographs (base and crest levels) and land use.

Threshold Level Analysis

Scatterplots of the event water level fluctuation data were plotted against the various wetland and watershed morphological parameters. Some of these plots showed apparent thresholds that signify a range of the hydrologic parameter where the event fluctuation data are similarly distributed. Within these ranges, characteristics such as the mean and variance of the data were approximately equal. Table 8-2 shows significant threshold levels (P < 0.05 for all thresholds) and characterizes the water level fluctuation data within each range.

Table 8-2 Parameters significant to wetland water level fluctuation.

Parameter	Range (a)	Mean WLF (m)	Std. Dev. (m)	n
Forested area	forest = 0%	0.384	0.338	97
	forest ≥ 14.7%	0.151	0.138	224
Total impervious area	2.0 ≤ TIA ≤ 3.5%	0.105	0.072	105
	3.5 < TIA ≤ 20%	0.176	0.151	143
	21.8 < TIA ≤ 54.9%	0.478	0.348	73
Outlet constriction	low to moderate	0.148	0.119	198
	high	0.34	0.33	123
Wetland-to-watershed area ratio	0.005 ≤ W/Ws ≤ 0.04	0.304	0.301	169
	0.05 < W/Ws ≤ 0.44	0.129	0.091	152
Watershed soils index	3.9 ≤ WSI ≤ 4.1	0.247	0.279	209
	4.2 < WSI ≤ 5.8	0.174	0.143	112

(a) The upper and lower bounds are the maximum and minimum values of the parameter within the range.

A key index relating urbanization to WLF was basin imperviousness. Two thresholds were identified in the relationship between event WLF and impervious area (Figure 8-2). The first threshold (3.5% impervious area) may represent the level of urbanization where scattered clearing of forests is added to by larger developments, and storm drainage systems that route runoff to the wetland are developed. Development within the first range was usually below 15% low density residential (LDR), whereas the second range begins around 24% LDR. Wetlands HC13 and LCR93 (in the second range) were exceptions to this tendency, because of the large proportion of their watersheds that were clear-cut. The second threshold (20% impervious area) may represent the point that changes in storm runoff caused by urbanization (e.g., flow volumes, flashiness) become dominant over the other factors that influence wetland hydroperiod.

The amount of forested area in a watershed was expected to be inversely related to event WLF. Forests store rainwater in the canopy, return water to the atmosphere through evapotranspiration, and typically have a highly permeable litter zone on the soil surface, all of which act to reduce storm runoff volumes and reduce the delivery rate to receiving waters. Furthermore, in an area such as the Puget Sound lowlands which are primarily forested until urbanization begins, forested coverage is an index of urban development. The expected relationship was observed (Figure 8-2). Sites with highly constricted outlets were expected to exhibit higher event WLF than those with less constricted outlets due to backwater effects. Figure 8-2 shows that this trend was observed, particularly in the maximum levels of event WLF.

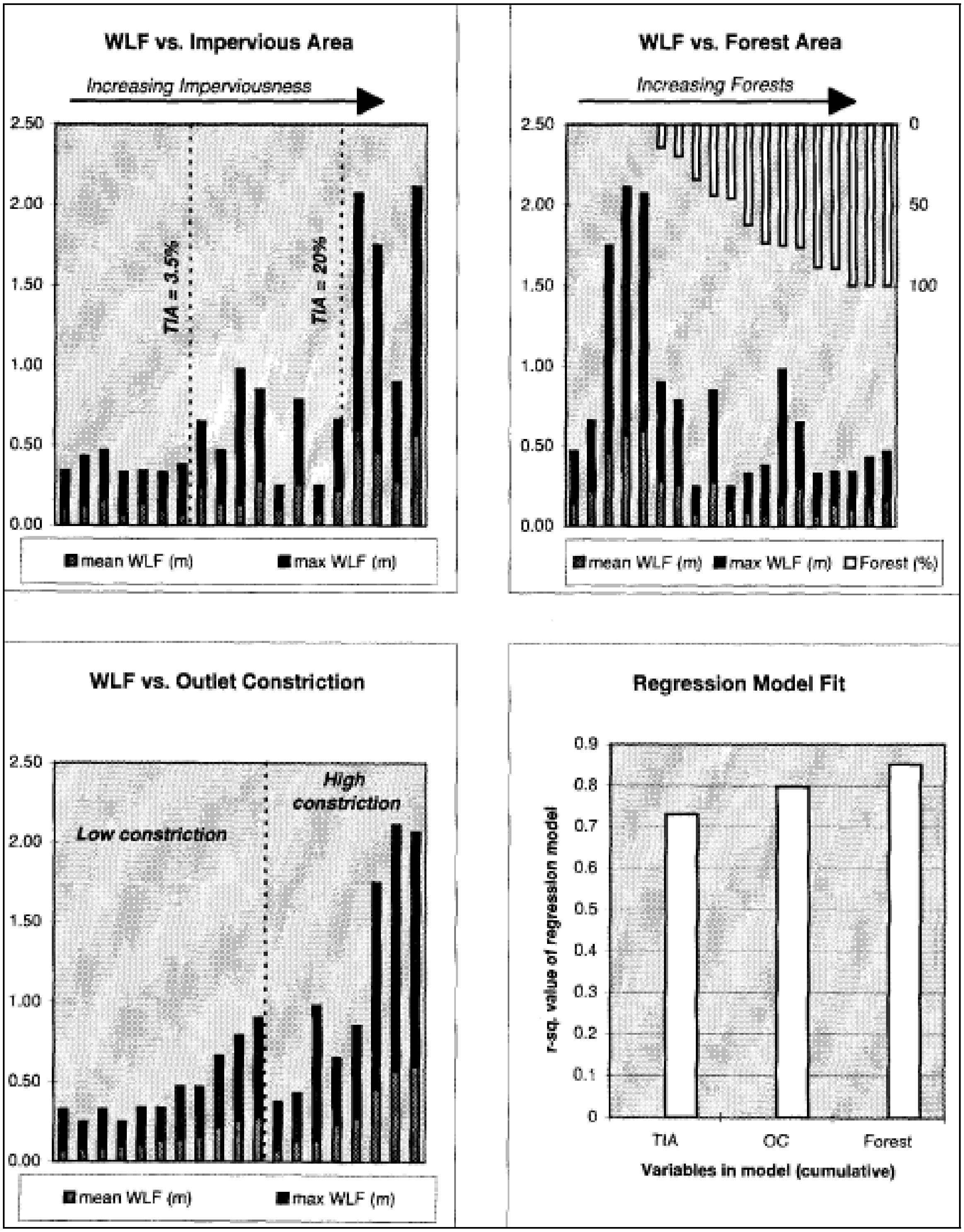

Figure 8-2. Relationships between water level fluctuations and imperviousness, forest area, and outlet constriction.

As shown in Table 8.2, there were two other variables that exhibited trends with wetland WLF: wetland-to-watershed area ratio and watershed soil index (WSI). The wetland-to-watershed ratio can be thought of as a "loading" term. The lower the ratio, the less area available to store storm runoff, resulting in higher event WLF. The threshold observed (ratio = 0.045) corresponds with the recommended ratio for stormwater detention ponds, which is five percent (KCSWM, 1990). The WSI was developed to quantify the soil drainage characteristics; since higher values indicate soils with high infiltration capacity, these values were expected and found to be associated with low event WLF.

Multiple Regression Analyses

Multiple regression analyses were done on the mean event WLF data from 1988 through 1991. The mean WLF data were weighted by the sample size, with the size of the weighted data set consisting of 321 observations. The best model fit was found using three variables: impervious area, outlet constriction, and forested area (see Figure 8-2). The following equation produced the best fit when using percent impervious and forested areas as continuous variables, and outlet constriction as a binary variable (0 or 1):

$$\text{Mean WLF (m)} = 0.145 + 0.0052*(\text{Impervious}) + 0.141*(\text{OC}) - 0.0011*(\text{Forest})$$
$$\text{where } R^2 = 0.790 \text{ and SE} = 0.08 \text{ m.}$$

The model fit explained 79% of the variation in mean event WLF between sites. Residual analysis showed no deviations from the model assumptions. All the parameter coefficients were of the sign (positive or negative) expected. This model was tested in later years using data from 1993 through 1995 and not confirmed (Chinn 1996), however there were some significant differences in the assumptions guiding the selection of data between the two analyses which likely account for the different results.

Dry Period

The length of the summer dry period for the study sites ranged from zero for the sites with stable base flow to nearly 200 days (Table 8-1). A variety of approaches were used to evaluate which factors are important in determining the permanence of a site and the length of the dry period for those sites that dry in the summer. Spearman rank correlations were used to investigate the relation between the mean length of the summer dry periods and morphologic parameters at sites that dry during the summer. Significant negative correlation was found between the length of the dry period and the area of the wetland. The significance of the wetland area is attributable to two factors of the hydrologic balance, evapotranspiration and groundwater exchange. Because the correlation is negative, however, it is assumed that groundwater discharge to the wetlands is driving the relationship.

Hydrologic Characteristics of Two Intensively Studied Wetlands

A summary of the natural and hydrologic characteristics during the study period (1988-90) for the B3I and PC12 wetlands is given in Table 8-3. The hydrologic reactions to storms exhibited by the two wetlands are typical of the respective watershed land uses. The reaction of B3I inlet flows to storms is fast and dramatic. Flows increase almost immediately because of the large impervious land area and piped storm drain system. Similarly, when storms end, the flow recedes quickly to near baseflow conditions. The PC12 inlet flow, on the other hand, reacts relatively slowly to storms, with the receding limb of the hydrograph extending much longer than at B3I. Significant inflows occurred at PC12 only from October to June; however, there was water in the wetland year round.

Table 8-3. Natural and hydrologic characteristics of two wetlands.

Variable (unit)	B3I Wetland	PC12 Wetland
Dominant land type	Urban	Forest
Watershed area (ha)	187	87
Wetland area (ha)	2	1.5
Wetland-to-watershed ratio	0.011	0.017
Total precipitation (mm)	1,813	1,934
Precip. volume (m^3) in drainage area	3.4×10^6	1.7×10^6
Mean daily inlet flow (m^3/s)	0.042	0.021
Maximum daily inlet flow (m^3/s)	0.75	0.22
Days with measurable flow during study	730	493
Total flow during study (m^3)	2.7×10^6	0.9×10^6
Wetland storage volume (m^3)[a]	400-5,000	600-7,000
Runoff/precipitation ratio	0.80	0.53

[a] Wetland storage volume varies depending on season and flow conditions.
(Note: Study period was two years for B3I and 20 months for PC12).

Nearly 80 % of the annual precipitation occurred between October and March. The maximum daily precipitation occurred on January 9, 1990 (approximately 80 mm at both sites). Pan evaporation data from the Puyallup station were used for ET estimates at the wetlands. The measured pan evaporation was greatest from May to August (exceeding 100 mm per month) and least from November to March. The maximum monthly and daily evaporation rates during the study were 160 mm (July 1989) and 16 mm (July 30, 1989), respectively.

Water storage volumes varied from 400 to 5,000 m^3 at B3I and from 600 to 7000 m^3 at PC12. Generally, changes in storage volume at B3I were short-term (on the order of hours) and directly related to storm events. Baseflow rates and water storage were comparable during the wet and dry seasons. At PC12, on the other hand, storage volumes changed during storm events and by season. Water volumes were greatest during large storms or groups of storms during the late wet season.

The results of the groundwater investigation indicate that both wetlands are discharge zones under most conditions meaning that groundwater discharges to the wetland and becomes surface water. Recharge wetlands, in contrast, replenish groundwater through infiltration of surface water. This was determined by the piezometric head measurements, and given the fact that groundwater flows from areas of high to low head. The head measurements in both wetlands generally increase with depth below the water table (as measured by the deep piezometer clusters) and distance from the wetland, indicating the groundwater flows both vertically and laterally to each wetland. Discharging wetlands have also been documented by other authors (Wilcox et al., 1986; Siegel and Glaser, 1987).

Wetland Hydrology by Season and Wetland

Table 8-4 summarizes the hydrologic inputs and outputs by season for the two wetlands. For both wetlands, surface water outflow accounted for greater than 99 % of the outputs during the study period. Thus, groundwater recharge and ET, the other potential sources of output, were insignificant on an annual basis. This is typical for wetlands that have a low wetland-to-watershed area ratio (1.1 and 1.7 % for B3I and PC12, respectively) and for wetlands that lie in a groundwater discharge area. For wetlands with low wetland-to-watershed ratios, inputs from the larger watershed (i.e., surface water flows) often dwarf the contributions from "in-wetland" components, such as groundwater and ET, because of the relatively small wetland area. Also, if groundwater exhibits mostly a discharge pattern as a result of topography and wetland location, then groundwater recharge is likely a minimal source of water output.

Table 8-4. Summary of hydrologic inputs and outputs by season (all values are in 1000 m^3; percent of total input or output in parentheses).

Wetland/ Season[a]	Precip- itation	Inputs Inflow	Ground- water[b]	Outputs Outflow	Evapo- ration	Error
B3I[c]						
Dry 88	2 (0.6)	289 (80.8)	66 (18.6)	319 (97.0)	10 (3.0)	28 (8.8)
Wet 88-89	12 (1.6)	639 (85.4)	99 (13.2)	762 (99.9)	1 (0.1)	-12 (-1.6)
Dry 89	6 (0.7)	668 (84.5)	116 (14.8)	627 (98.1)	12 (1.9)	150 (23.5)
Wet 89-90	14 (1.4)	863 (90.0)	82 (8.6)	989 (99.9)	0 (0.1)	-29 (-3.0)
Dry 90	2 (0.7)	239 (87.1)	33 (12.1)	231 (99.2)	2 (0.8)	40 (17.5)
Total	36 (1.2)	2,697 (86.1)	398 (12.7)	2,928 (99.2)	25 (0.8)	178 (6.0)
PC12[d]						
Wet 88-89	12 (2.1)	445 (79.5)	103 (18.4)	535 (99.9)	0 (0.1)	23 (4.4)
Dry 89	5 (3.9)	97 (72.4)	32 (23.8)	136 (93.6)	9 (6.4)	-9 (-6.4)
Wet 89-90	11 (2.5)	312 (74.1)	99 (23.4)	373 (99.9)	0 (0.1)	48 (13.0)
Dry 90	1 (2.5)	49 (82.3)	9 (15.2)	62 (97.7)	1 (2.3)	-4 (-6.4)
Total	29 (2.5)	904 (76.9)	243 (20.7)	1,105 (99.0)	11 (1.0)	58 (5.2)

[a] Dry season = April-September; wet season = October-March

[b] Positive groundwater values indicate groundwater discharge to wetlands.

[c] B3I study period: June 1988 - May 1990

[d] PC12 study period: October 1988 - May 1990

Surface water inflows accounted for 86 and 77% of the inputs for B3I and PC12, respectively, on an annual basis. Groundwater discharge to the wetlands accounted for most of the remaining input (13 and 21% for B3I and PC12, respectively). Direct precipitation inputs were quite small in the overall balance. During individual months or groups of months, however, groundwater and precipitation contributed substantially more to the wetland water inputs, particularly at PC12.

Differences also existed in the magnitudes of inputs and outputs for the wet and dry seasons. This was particularly true for precipitation, with 75 to 80% occurring during the wet season, and ET, with approximately 90% occurring during the dry season. At B3I, 60% of annual surface water flow occurred during the wet season, whereas at PC12 this component totaled approximately 80%. At PC12, ET accounted for greater than 50% of the output from July to September 1989 when baseflows were minimal. During the same period, ET at B3I was less than 5% of the output, because of the stable and relatively high baseflow. The direct groundwater input to B3I was fairly steady throughout the year. However, at PC12, nearly 83% of the groundwater contribution to the wetland occurred during the wet season.

Urbanization and Other Factors Affecting Wetland Hydrology

The dynamics of wetland hydrology are governed by factors that may change seasonally or slowly over time. Seasonal changes result from variation in climate (e.g., precipitation, solar radiation), plant growth and groundwater recharge. Longer-term changes result from human activities, including watershed development, groundwater withdrawal, wetland outlet modification or drainage activities. Although this study was not designed to investigate change over time, some general conclusions can be drawn from comparisons between urbanized and nonurbanized catchments.

The runoff-to-precipitation ratios were 0.80 and 0.53 for B3I and PC12 wetland watersheds, respectively. Thus, more water is captured in the nonurbanized catchment, resulting in less runoff to the wetland. Potential pathways for the difference in water reflected in these numbers are ET, regional groundwater recharge and withdrawal in the watershed itself. The ET in the forested nonurbanized catchment of PC12 is undoubtedly greater than in the developed urbanized catchment of B3I. Regional or deep groundwater recharge within the PC12 watershed is also likely greater than in the case of B3I, because of milder topography and less impervious surface. Finally, groundwater withdrawal to meet local water needs is likely more significant in the PC12 watershed.

Water level fluctuation is perhaps the best single indicator of wetland hydrology, because it integrates nearly all hydrologic factors. The mean WLFs were 0.15 and 0.49 m for the PC12 and B3I wetlands, respectively. The higher mean occasion WLF at B3I reflects the effect of many factors, including its urbanized catchment, piped storm drain system and constricted outlet. The maximum study period WLFs were quite similar. This apparent discrepancy occurred because of the evaporation and lowered water level in PC12 during the summer. In summary, both wetlands experienced similar long-term fluctuations; however, the urban wetland was exposed to much more frequent and greater WLFs.

Hydrologic Components Error Analysis

By measuring all components of the water balance shown in Equation 1 (Chapter 1), it was possible to determine error estimates for the seasonal balances. The seasonal errors (Table 8-4) ranged from -6.4 to +23.5% of the total hydrologic outputs. For the entire study period, the errors were 6.0 and 5.2% for B3I and PC12, respectively. This reduction reflects the cancellation effect of positive and negative errors when summed over a longer time period. Generally, the larger percentage errors occurred during the dry seasons, reflecting the increased importance of groundwater inputs and ET in the overall balance at that time.

The type and magnitude of the errors associated with hydrologic or water balances may be characterized in several ways. These include errors associated with the: (1) equipment (e.g., inaccurate calibration), (2) measurements (e.g., representativeness of measurement), (3) calculations (e.g., weak stage-discharge correlations, groundwater calculations), and (4) summation of balance components. It is important to note that these errors can improve or degrade the apparent accuracy of a water balance depending on the interaction between errors.

If precautions are taken to minimize the errors associated with the equipment, measurements and calculations, and if all components are included in a water balance, it is possible to reduce potential errors greatly. An assessment of the importance of the different components of a balance is a critical task in this process. Because of the above-noted errors, it is recommended that no components of a balance be estimated by difference. Using this technique simply masked the errors in the unknown or unmeasured component (usually ET, groundwater, or both).

CONCLUSIONS

The quantity of stormwater entering many palustrine wetlands in the Puget Sound region has changed as a result of rapid development in urbanizing areas. The purpose of this chapter has been to characterize the hydrology of wetlands affected by urban stormwater, in comparison to unaffected or forested systems. This information, then, may help to explain observed changes in wetland soils, plants and animals over time. Additionally, if observed effects of stormwater on wetlands can be documented, it may be possible to mitigate these effects through watershed controls and stormwater management efforts.

The hydrology of wetlands as measured by water level fluctuation was highly variable. Differences in water level fluctuation were attributed to level of watershed imperviousness, forested cover, and wetland outlet constriction. A multivariate model using these three parameters, calibrated to the study sites, was found to predict water level fluctuations relatively accurately. This model should be verified and tested further using similar data sets from all years of collection in future research efforts.

For the two study wetlands, surface water inflow and outflow were the dominant components in the water balance on an annual basis. It was concluded that this is typical for wetlands with low wetland-to-watershed ratios. The ET was insignificant in the overall water budget on an annual basis; however, it was the major source of water output from the PC12 wetland from July to September, when outflows were minimal. Both wetlands were identified as primarily groundwater discharge zones, with groundwater contributing significant inputs. Like ET, the influence of groundwater was greatest at PC12 during the summer months.

Differences were also identified in the hydrology of both wetlands because of the level of watershed urbanization. In the urbanized watershed, a greater proportion of the precipitation was realized as surface inflow to the wetland. Storm runoff was delivered more quickly and in greater short-term volumes to the urban wetland. The result of these conditions was greater and more rapid water level fluctuations in the urban wetland. This characteristic would probably be replicated in most wetlands where development occurs in the watershed.

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CHAPTER 9 THE EFFECTS OF WATERSHED DEVELOPMENT ON WATER QUALITY AND SOILS

by Richard R. Horner, Sarah Cooke, Lorin E. Reinelt, Kennneth A. Ludwa, Nancy Chin and Marion Valentine

INTRODUCTION

This chapter emphasizes water and soil quality in wetlands with significant urbanization in their watersheds. Like other chapters in this section, its purpose is to characterize particular elements of Puget Sound Basin freshwater wetlands having urbanized watersheds. The urbanized cases were divided into two major categories. The "treatment" group included the wetlands whose watersheds had a more than 10% rise in urbanization between 1989 and 1995. Conversely, the "control" category consisted of wetlands whose watersheds experienced a less than 10% increase in urban land cover between 1989 and 1995. The urban control wetlands were subdivided into two further classifications: (1) the most highly urbanized sites (H) had watersheds that were both ≥ 20% impervious and ≤ 7% forest by area; and (2) moderately urbanized wetlands (M) had watersheds that were 4-20% impervious and 7-40% forested by area. The nonurbanized category (N), with both < 4% impervious land cover and ≥ 40% forest, made up the balance. This latter category is emphasized in Chapter 2 but will be mentioned in this chapter at times for comparison. Table 1 of Section 1 gives characteristics of the individual watersheds.

This chapter first describes water quality conditions in urban control wetlands, and then discusses changes in these conditions in treatment wetlands. It then proceeds to cover soil characteristics in a similar way. Chapter 2 covers the methods with which the data were collected. Tables 1, 2, and 4 of Chapter 2 summarize water quality results for the urban control wetlands as well as the nonurbanized cases. Tables 5 and 6 in that chapter perform the same function for the soils data. These tables are not repeated in this chapter but are referenced here several times.

As has been stated elsewhere in this volume, the research program concentrated on palustrine wetlands of the general type most prevalent in the lower elevations of the central Puget Sound Basin. The results and conclusions presented here are probably applicable to similar wetlands to somewhat to the north and south of the study area but may not be representative of higher, drier, or more specialized systems, like true bogs and "poor" (low nutrition) fens.

THE EFFECTS OF WATERSHED DEVELOPMENT ON WATER QUALITY

This section first profiles the urban control wetlands, both moderately and highly urbanized, using the statistical summary data in Chapter 2, Tables 2-1, 2-2, and 2-4. Following the profiles is a more general summary of other applicable findings from the research.

Moderately Urbanized Wetlands

A water quality portrait of Puget Sound Basin lowland palustrine wetlands moderately affected by humans would show slightly acidic (median pH = 6.7) systems with DO often well below saturation, and in fact sometimes quite low (< 4 mg/L). Dissolved substances are fairly high relative to nonurbanized wetlands (median conductivity about three times as high) but somewhat variable. Suspended solids are only marginally higher than N wetlands but, like them, quite variable. Median total dissolved nitrogen concentrations (the sum of ammonia, nitrate, and nitrite) are more than 20 times as high as dissolved phosphorus, a ratio very similar to the nonurbanized wetlands but with higher magnitudes in both cases. Again, plant and algal growth is generally limited by P, rather than N. TP at the median level is more than twice as high in the M compared to the N wetlands (70 µg/L). The median fecal coliform concentration is close to the 50 CFU/100 mL criterion applied as a geometric mean to lakes and the highest class of streams by Washington state water quality criteria. This quantity is highly variable in all wetlands, most extremely so in those of the M class. More than half of the individual FC values for moderately urbanized flow-through wetlands exceeded the maximum 200 CFU/100 ml criterion applied to the lowest class streams (however, their geometric mean may not do so). As with N wetlands, both mean and median heavy metals concentrations in the moderately urbanized sites are in the low parts per billion range, with standard deviations just about identical to the means. The median lead concentration, however, is close to the chronic water quality criterion set for lakes and streams having hardness of 50 mg/L as $CaCO_3$.

In summary, the following general statements can be made to characterize the water quality of Puget Sound Basin lowland palustrine wetlands in a moderately urbanized state:

> These wetlands are highly likely (≥ 71% of cases observed) to have median conductivity > 100 µS/cm but median TSS in the range 2-5 mg/L, NH_3-N < 50 µg/L, and total Zn < 10 µg/L.
>
> Moderately urbanized wetlands are highly likely (71% of cases) to have median fecal coliforms < 50 CFU/100 mL, but also to have many individual measurements above 200.
>
> They are highly likely (100% of cases to have TP > 20 µg/L and likely (57% of cases) to have TP > 50 µg/L and NO_3+NO_2-N < 100 µg/L. The latter variable is highly likely (86% of case) to be < 500 µg/L.
>
> The pH and DO in these wetlands are unpredictable from consideration of urbanization status alone, being dependant on other factors.

Highly Urbanized Wetlands

Highly urbanized wetlands are harder to profile because of the small set of only two in the control group. Also, both of these wetlands are the flow-through type, not giving any picture of how morphology might affect the conclusions. What can be said is the following:

> There is some tendency for these wetlands to be the closest to neutral in pH among the three urbanization status categories. They tend to fall in the same region as the other classes in dissolved oxygen.

They most likely would have median conductivity > 100 µS/cm.

Like the other two classes, highly urbanized wetlands are very likely to have median NH_3-N < 50 µg/L.

Unlike the other two classes, they are very likely to have median NO_3+NO_2-N > 100 µg/L and TP > 50 µg/L.

These sites are likely, but somewhat less than the other categories, to have autotrophic growth limited by phosphorus.

They are likely to exceed the 50 CFU/100 mL level of fecal coliforms.

These wetlands have a higher tendency than the other categories to have Zn > 10 µg/L but in most instances still not to exceed the chronic criterion of 59 µg/L for relatively soft waters.

Other Findings

In a synoptic study of 43 urban and 27 nonurban wetlands during 1987, before routine sampling began, the program found that FC and enterococcus were significantly higher in urban wetlands (Horner et al. 1988). Although the mean counts for both types of bacteria were within water quality standards, bacteria substantially exceeded standards in wetlands in high density areas that showed evidence of human intrusion. The watersheds of most of the wetlands in which bacteria also exceeded standards were in watersheds characterized by low density residential development, while some of the watersheds of the remainder of the wetlands had high density residential development. None of the watersheds with bacteria in excess of standards were dominated by commercial development. Other than pH, FC count was the only water quality variable measured by the survey.

After four years of regular data accumulated, a major effort was undertaken to relate water quality conditions to watershed and wetland morphological circumstances. In this work it was found that certain water quality parameters varied in response to the changes in watershed wetland characteristics that can accompany urbanization (Ludwa 1994). The characteristics used as independent variables in this analysis included (1) percent forest cover, (2) percent total impervious area, (3) percent effective impervious area (the area actually linked to a storm drain system), (4) the ratio of wetland to watershed area, (5) the ratio of forest to wetland area, (6) wetland morphology (open water or flow-through), and (7) outlet contstriction. These measures may be expressed as either continuous (ranges) or categorical (binary or ternary) variables. Multivariate linear regressions were used to determine if there is an adequate relationship between these characteristics and water quality parameters. If there is such a relationship, the equations could be used to analyze probable changes in water quality following development.

Specific watershed land uses and wetland morphological values were significantly associated with most water quality values (Ludwa 1994). The dependent water quality variables exhibiting the best associations and most correctly predicted when verified with a portion of the data set held aside for verification were conductivity, pH, and TSS (Ludwa 1994). Pollutants that are often adsorbed to particulates, specifically TP, Zn, and FC, showed similar degradation across key levels of the independent variables. Conductivity, TSS, FC, and enterococcus degraded the most consistently between more highly developed watersheds and those that were moderately urbanized or rural (Taylor et al. 1995). Conductivity, TSS, Zn, DO, and FC varied by the greatest amounts (Ludwa 1994).

Based on program data from 1988-93, percent forest cover was the best predictor of water quality for Pacific Northwest palustrine wetlands, followed by percent total impervious area, forest-to-wetland areal ratio, and morphology (Ludwa 1994). All variables except NO_3+NO_2-N were higher in wetlands with no forest in their watersheds compared to those in watersheds with at least 14.7% forest cover (Taylor et al. 1995). Conductivity, TP, and FC rose significantly when the percentage of impervious surface exceeded the values of 3.5% and 20% (Taylor et al. 1995). Forest-to-wetland areal ratio strongly influenced conductivity, TSS, NO_3+NO_2-N, TP, SRP, and FC, where the ratio was less than 7.2 (Taylor et al. 1995). Conductivity, TSS, NO_3+NO_2-N, TP, SRP, and FC, had significantly higher means in relatively channelized wetlands, although it should be noted that these results may have been influenced by extraneous factors (Taylor et al. 1995). Outlet constriction and wetland- to-watershed areal ratios had inconsistent roles in influencing water quality (Taylor et al. 1995). It should be noted that because the breakpoint values are expressed as fixed ranges, and it is unknown if thresholds exist or what they are, it is entirely possible that other water quality constituents also vary significantly with characteristics of urbanization on a continuous basis (Taylor et al. 1995).

The analysis indicated that, for similar watersheds in the region, there is a definite degree of deforestation and development above which average wetland water quality will become degraded. However, if amounts of forest remain above some minimum level, water quality will comply with criteria. It should be noted that because extremes in water quality often have greater impacts on biological resources than average conditions, attention should also be given to the relationships of conditions with minimum and maximum values. Minimums and maximums were found to vary widely across urbanization and morphological levels, so that entire ranges of water quality variables shift significantly to degraded conditions.

Ludwa (1994) found that the strongest regression relationships were for mean, maximum, and minimum conductivity, TSS, and DO. In view of the correlation coefficients, urbanization was consistently related to all water quality values except NH_3-N and SRP. The strong regressions of TSS and conductivity with urbanization suggested that an increase in watershed imperviousness will facilitate the movement to wetlands of runoff containing inorganic particulate and dissolved matter. Total suspended solids and conductivity are directly and indirectly harmful to wetland biological communities. Wetland morphological factors had similar effects, although they were less consistent for outlet constriction.

Predictions were generally better for mean and maximum values, since these values exhibited more variability than minimum values from site to site. The choice of factors to be included in the regression equations was manipulated to improve predictive value, although the process was somewhat subjective. Wetland-to-watershed areal ratio was the most frequently used factor. Although little can be done to affect this ratio other than by changing wetlands physically or by diverting inflows, the importance of this ratio does not suggest that development or deforestatation are unimportant. To the contrary, where a wetland covers a smaller portion of a watershed, the effect of deforestation may be magnified. Effective impervious area, which expresses how much land is actually drained by a storm drainage system, had more predictive power than total impervious area. However, there was no consistent relation between outlet constriction and water quality. Categorical predictors seemed to have slightly better predictive value than continuous ones, and are also simpler to use.

The crucial values for water quality lie between 4 and 12% for total impervious area and 0 to 15% for forested area. Theory and observations in other regional ecosystems (e. g., Horner et al. in press) have demonstrated that there is likely to be a continuous and relatively rapid decline in various measures of ecosystem "quality" as forest begins to decrease in favor of impervious surfaces. As conversion progresses the decline is likely to slow in rate but to continue. Therefore, with a continuous pattern of variation normally prevailing, numerical values should not be regarded as thresholds but as points where degradation becomes evident and demonstrable, and where standards generally accepted as necessary to support biota probably will not be met, at least at times.

Ludwa recommended that total impervious area in Pacific Northwest watersheds with strong wetland protection goals be not more than 10% and that a forest cover of at least 15% be maintained. Whether more effective implementation of urban runoff best management practices would permit these thresholds to be shifted toward more urbanization is a matter only for conjecture. However, development and deforestation will ultimately have to be limited if high quality and well functioning wetland systems are to be preserved. Channelized sites usually had lower water quality, hence a shift to more channelized conditions intentionally or by inadvertant flooding resulting from increased urbanization should be avoided.

Treatment Wetlands

The treatment wetlands studied by the program were: Big Bear Creek 24 (BBC24), East Lake Sammamish 61 (ELS61), Jenkins Creek 28 (JC28), North Fork Issaquah Creek 12 (NFIC12), and Patterson Creek 12 (PC12). Their watersheds experienced increases of urbanization in the range of 10.2 to 10.5% in three of the five cases (JC28, ELS61, and PC12), 42.2% for BBC24, and 100% in the case of NFIC12. The most common change in land use was from forest to single family residential, a development pattern typical of the early stages of urbanization (Chin 1996). Table 9-1 shows land cover in 1995 and the changes since 1989.

This distribution of changes gave an opportunity to observe the relative effects with substantial compared to more limited watershed alterations. The timing of development in relation to the program's schedule also offered the chance to observe effects during the construction-phase, when soils are often bare for long intervals, versus the subsequent period when areas finished with construction are restabilized.

Table 9-1. Land cover in 1995 and the changes from 1989.

	Forest		Impervious Surface	
Wetland	1995(%)	Change(%)	1995(%)	Change(%)
JC28	19.8	-14.6	20.6	+0.6
ELS61	3.7	+1.2	10.6	+5.5
PC12	64.7	-10.5	6.8	+1.7
BBC24	47.4	-42.1	10.6	+7.2
NFIC12	0.0	-100.0	40.0	+38.0

In terms of the urbanization groupings used to classify the control wetlands, after the development that occurred through 1995 in the treatment watersheds, NFIC12 would be categorized as H and all of the others as M. Morphologically, JC28 is a flow-through type, and the remainder are all open water.

Observations for Individual Treatment Wetlands

Wetlands in urbanizing watersheds are especially vulnerable to erosion during the construction phase of development. Total suspended solids concentrations often increase greatly during such periods, but return to approximately pre-development levels as bare land is covered by structures and vegetation. During periods of construction, mean TSS values increase more dramatically than median values because of the influence of especially high concentrations. For instance, the ELS61 wetland recorded a median TSS concentration of 10.4 mg/L in 1989 and had a maximum concentration of 59 mg/L in August, as a result of construction site runoff. An increase in TSS at JC28 in 1989 was also linked to land disturbances. At both of these sites, TSS declined in the following year.

Elevated sediment in runoff from construction sites also corresponds to increases of concentrations of other pollutants, especially phosphorus and nitrogen), that are contained in soils (Novotny and Olem 1994). Subsequent to construction, application of fertilizers can further increase nutrient concentrations in runoff. In the JC28 wetland, land disturbances, including expansion of an adjacent golf course in 1989 marked the commencement of a regime of higher nutrient concentrations. Median NO_3+NO_2-N and SRP values increased by 63% and 96%, respectively, between 1988 and 1989, and continued to climb steadily from 1990 to 1995. The initial increases probably resulted from land disturbance, with the subsequent rises attributable to fertilizer runoff from the golf course. Mean NH_3-N also also rose sharply in 1989, with a maximum value of 619 μg/L, and median NH_3-N was higher in subsequent years. More than half of the NO_3+NO_2-N readings exceeded 500 μg/L.

At the ELS61 wetland, NH_3-N and NO_3+NO_2-N initially rose in 1989, but declined in 1993, although not to predevelopment levels. Concentrations of NH_3-N climbed again in 1993, while NO_3+NO_2-N greatly increased in 1995. Many NH_3-N and NO_3+NO_2-N concentrations exceeded 100 and 500 μg/L, respectively, during these years. Average SRP and TP concentrations were actually the highest in 1988, perhaps because of the operations at a small livestock farm next to the wetland. However, after declining from 1988 to 1990, SRP and TP concentrations were substantially higher in 1993 and 1995. One of the two highest chlorophyll a concentrations in the first two years of the program was recorded at ELS61 (Reinelt and Horner 1990).

NFIC12, the wetland that had the greatest amount of development in its watershed between 1989 and 1995, increasing from 0 to 100%, displayed different water quality patterns than ELS61 and JC28. Average values for TSS rose modestly from 1989 to 1995, with a maximum peak value of 16 mg/L in 1993. Average concentrations of NH_3-N and NO_3+NO_2-N did not appear to rise during this period, but NH_3-N and NO_3+NO_2-N did reach maximum concentrations 120 and 1400 μg/L, respectively, in 1993. Concentrations of SRP and TP, however, rose steadily, reaching median concentrations of 148 and 202 μg/L, respectively, in 1995. For all years, mean and median TP concentrations exceeded 50 μg/L.

Results were less conclusive for the other two treatment wetlands, PC12 and BBC24, demonstrating that there is not necessarily a link between development and water quality degradation, even for wetlands whose watersheds have undergone similar amounts of development. A possible explanation for the difference in results may be that the watersheds of PC12 and BBC24 remained approximately half forested, retarding transport of pollutants to the wetlands in runoff. The watersheds of JC28, ELS61, and NFIC12, on the other hand were only 0 to 19.8% forested by area. In addition, a large wet pond meeting current design standards was constructed to treat storm runoff from the development built adjacent to PC12. These observations are signs that concerted action to maintain forest cover and impose structural storm water management measures can avoid water quality degradation.

Increases in nutrient loadings can have serious consequences for normally nutrient-limited bogs and fens. In one of the bog-like wetlands covered by the program in a special study, East Lake Sammamish 34 (ELS34), also known as Queen's Bog, the *Sphagnum* mat was observed to be decomposing, probably because of stormwater inflow. Nitrogen input exponentially increases decomposition rates.

Profile of Treatment Wetlands

Table 9-2 shows statistics for the five treatment wetlands in the baseline period, when little or no urbanization had occurred (1988-1990), and then the later years (1993 and 1995), after most of the changes in land use were either well underway or complete. Very little change in pH was evident. DO exhibited some fluctuation in three wetlands, but only ELS61 registered a notable decline in the median level with time (≈ 2 mg/L).

Most direct comparisons for all of the other water quality variables and all wetlands indicated no change or reduction during the program, but there were some exceptions to that generality that bear examination. NH_3-N appeared to rise in ELS61 from predominantly < to > 50 µg/L values. NO_3+NO_2-N showed increases in all but NFIC12. Median concentrations still stayed mostly < 100 µg/L in PC12 and ELS61. The increase in BBC24 kept the median still below 500 µg/L. JC28 increased from an already relatively high median > 500 to > 1000 µg/L. In NFIC12 relatively high concentrations of SRP and TP increased further after development, reaching among the highest levels seen in the entire program. Increases in TP also occurred in JC28, but stayed in the 20-50 µg/L range, and in ELS61, where the median moved from the area of 50-100 to > 100 µg/L. Relatively small rises in fecal coliform statistics were registered in JC28 and ELS61, but medians remained below 50 CFU/100 mL. Although relatively high detection limits in the early years make comparisons more difficult for the metals, there was no sign that any of the three metals increased substantially anywhere or threatened a violation of the water quality criteria applied to other water bodies.

Wetlands in moderately and highly urbanized watersheds are generally profiled earlier in this chapter. Whether or not the treatment wetlands fit these profiles after going through development will now be examined. The four M wetlands all fit the profile for that category in the cases of conductivity, NH_3-N, zinc, and fecal coliforms. It should be noted that they almost always fit the same profile in the baseline years; thus, preexisting factors are most responsible for how these wetlands profile. ELS61 and JC28 failed to fit the profile for TP and NO_3+NO_2-N, respectively, being higher in both cases. In consequence, ELS61 did not appear to be generally phosphorus-limited in photosynthetic production, in contrast to the profile. A lack of fit occurred in only one other instance, TSS in BBC24, but the median was lower than the profile value. The only highly urbanized treatment wetland, NFIC12, exhibited fewer fits to the general H profile, but usually because it had lower values. This was the case for conductivity, NO_3+NO_2-N, and fecal coliforms. It did fit for NH_3-N, TP, and Zn, but actually fit in those cases before development too. This wetland also appears to tend toward nitrogen rather than phosphorus limitation, unlike the profile. Finally, it demonstrated no tendency toward more neutral pH, as the profile states. The humic acid-producing vegetation and peat prominent in this wetland apparently were not affected by the extensive urbanization, at least not yet.

Table 9-2. Water quality statistics for treatment wetlands in baseline and post-development years.

Site/ Years	Statistic	pH	DO (mg/L)	Cond. (µS/cm)	TSS (mg/L)	NH3-N (µg/L)	NO3+NO2-N (µg/L)	SRP (µg/L)	TP (µg/L)	FC (CFU/100ml)	Cu (µg/L)	Pb (µg/L)	Zn (µg/L)
JC28 88-90	Mean	6.59	6.9	99	< 5.68	< 72	710	17	44	< 237	< 5.0	< 5.5	< 28.9
	St. Dev.	0.24	1.4	28	> 9.30	> 159	414	29	45	> 578	> 0.0	> 1.1	> 37.2
	CV	4%	21%	28%	164%	220%	58%	174%	101%	244%	0%	19%	129%
	Median	6.67	7.1	94	2.9	13	653	4	29	20	< 5.0	< 5.0	< 20.0
	n	19.00	19.0	16	19.0	19	19	19	19	19	8	8	8
93-95	Mean	6.74	6.9	98	< 4.9	< 34	1002	< 27	84	83	< 0.7	< 1.3	< 8.7
	St. Dev.	0.20	1.4	9	> 3.8	> 37	448	> 37	90	102	> 0.3	> 1.0	> 8.5
	CV	3%	20%	10%	78%	109%	45%	136%	107%	123%	45%	75%	98%
	Median	6.77	7.0	95	3.6	20	1080	8	43	36	0.6	< 1.0	< 5.0
	n	6.00	14.0	14	13.0	14	14	12	14	14	6	14	14
PC12 88-90	Mean	6.72	7.0	68	< 3.0	< 75	< 456	11	52	< 63	< 5.0	< 5.0	< 15.2
	St. Dev.	0.32	3.4	11	> 2.6	> 76	> 551	10	45	> 146	> 0.0	> 0.0	> 5.4
	CV	5%	48%	15%	88%	101%	121%	89%	87%	233%	0%	0%	36%
	Median	6.62	7.5	71	2.4	35	108	7	40	8	< 5.0	< 5.0	16.0
	n	23.00	22.0	20	23.0	23	22	23	23	23	9	9	9
93-95	Mean	6.55	6.5	73	< 2.5	< 33	< 786	< 11	< 88	46	< 0.7	< 0.8	< 3.0
	St. Dev.	0.12	3.1	15	> 2.1	> 26	> 980	> 9	> 218	124	> 0.3	> 0.4	> 1.7
	CV	2%	48%	20%	87%	79%	125%	79%	248%	271%	40%	47%	57%
	Median	6.57	6.3	75	2.0	20	430	8	24	2	0.7	0.8	2.5
	n	8.00	16.0	16	16.0	15	13	15	15	16	8	16	16
ELS61 88-90	Mean	6.59	5.3	101	13.9	< 43	< 725	< 76	166	< 100	< 5.8	< 5.1	< 17.2
	St. Dev.	0.27	3.1	19	16.8	> 94	> 1086	> 85	125	> 188	> 1.9	> 0.3	> 6.6
	CV	4%	58%	19%	121%	218%	150%	112%	76%	188%	32%	6%	38%
	Median	6.61	4.9	103	8.0	25	109	58	149	20	5.0	5.0	20.0
	n	23.00	23.0	20	23.0	23	17	22	23	23	10	10	10
93-95	Mean	6.31	< 3.8	91	< 9.5	< 136	< 527	35	101	321	< 0.9	< 0.8	< 2.3
	St. Dev.	0.19	> 2.8	19	> 20.9	> 190	> 592	41	95	992	> 0.3	> 0.3	> 1.2
	CV	3%	73%	21%	219%	140%	112%	116%	94%	309%	30%	34%	52%
	Median	6.28	3.4	90	3.0	74	344	21	62	39	< 0.9	< 0.8	< 2.5
	n	8.00	13.0	16	16.0	15	9	16	16	16	8	16	16

Table 9-2 continued. Water quality statistics for treatment wetlands in baseline and post-development years.

Site/ Years	Statistic	pH	DO (mg/L)	Cond. (µS/cm)	TSS (mg/L)	NH3-N (µg/L)	NO3+NO2-N (µg/L)	SRP (µg/L)	TP (µg/L)	FC (CFU/100ml)	Cu (µg/L)	Pb (µg/L)	Zn (µg/L)
BBC24	Mean	6.76	6.1	83	< 2.0	< 44	210	5	23	< 186	< 5.0	< 5.0	< 15.1
88-90	St. Dev.	0.25	1.9	19	> 2.0	> 35	183	3	13	> 542	> 0.0	> 0.0	> 5.6
	CV	4%	31%	23%	97%	80%	87%	64%	55%	292%	0%	0%	37%
	Median	6.77	5.4	84	1.1	31	189	4	21	14	< 5.0	< 5.0	17.0
	n	22.00	23.0	20	23.0	23	23	23	23	23	10	10	17
93-95	Mean	6.84	6.7	90	< 3.5	< 34	< 396	< 6	< 27	< 411	< 0.5	< 0.8	< 1.8
	St. Dev.	0.22	2.0	38	> 6.6	> 20	> 347	> 3	> 12	> 1492	> 0.0	> 0.4	> 1.3
	CV	3%	29%	43%	187%	57%	88%	51%	44%	362%	4%	50%	72%
	Median	6.82	7.5	82	1.7	30	323	5	28	18	0.5	0.8	2.5
	n	8.00	16.0	16	16.0	16	15	13	15	16	8	16	15
NFIC	Mean	5.08	3.4	50	< 2.3	< 41	< 54	75	119	< 2	< 5.0	< 5.0	< 19.0
88-90	St. Dev.	0.69	1.0	46	> 2.3	> 29	> 45	104	94	> 0	> 0	> 0	> 7
	CV	14%	29%	90%	98%	71%	83%	140%	79%	21%	0%	0%	37%
	Median	4.84	3.5	37	1.0	39	34	53	80	2	< 5.0	< 5.0	22.0
	n	12.00	12.0	10	12.0	12	10	12	12	12	5	5	5
93-95	Mean	4.72	< 3.6	43	< 4.0	< 40	< 477	126	253	8	2.9	< 2.2	19.6
	St. Dev.	0.18	> 2.3	13	> 5.1	> 41	> 799	95	303	16	1.1	> 1.7	9.6
	CV	4%	63%	31%	129%	102%	167%	76%	120%	207%	37%	78%	49%
	Median	4.74	3.2	39	2.0	20	20	115	177	2	2.9	1.9	18.9
	n	4.00	9.0	10	10.0	10	3	10	10	10	4	10	10

THE EFFECTS OF WATERSHED DEVELOPMENT ON SOILS

This section first profiles the soils of the urban control wetlands, both moderately and highly urbanized, using the statistical summary data in Chapter 2, Tables 2-5 and 2-6. Following the profiles is a more general summary of other applicable findings from the research.

Urbanized Wetland Soil Profiles

A soils portrait of Puget Sound Basin lowland palustrine wetlands moderately affected by urbanization shows a somewhat acidic condition, more so (by about 1 pH unit) in open water than flow-through wetlands. The range of median values can be expected to be approximately 5.1-6.1. These soils will be aerobic in many instances, but their redox potentials not infrequently are below the levels where oxygen is depleted. TP is likely to be in the range 500-1000 mg/kg, and TKN up to 10 times as high. Median levels of soil organic content are approximately 15%. No general statement is possible concerning particle size distribution. Metals appear to be less variable than in nonurban sites, but still have coefficients of variation ranging from about 60 to 100%. It is most likely for Cu concentration to be in the vicinity of 15 mg/kg, for Pb and Zn to be very roughly twice as high, and for As to be about half as concentrated. Copper and lead lowest effect threshold freshwater sediment criteria would be violated in some samples.

Only two sites represent the highly urban control sites, which is a very small sample from which to construct a profile. This group appears to have much less acidic soils than the other two, with median pH of 6.5. Soils in this urbanization category are the most likely to be anaerobic, with median redox less than 100 mv. Nutrients are no higher than in the moderately urbanized `wetlands, and may even be a bit lower. Median organic content in the small sample suggests a level of about 20%. Again, PSD is a function of local factors. The available values show metals to be distinctly higher than in the soils of the other urbanization categories, about double the values given in the preceding paragraph. These concentrations would routinely exceed lowest effect thresholds for Cu and Pb, but not for Zn. Severe effect thresholds would still not be approached often.

Other Findings

Before regular sampling began, the research program conducted a synoptic survey of 73 wetlands, about 60% urban and the balance nonurban. Samples were analyzed in the laboratory for 31 of the wetlands. In the data from this study, significant differences appeared in soil Pb concentrations between urban and nonurban wetlands in the inlet and emergent zones (Horner et al. 1988). There were also significant differences at α = 0.10 between the concentrations of both Cd and Zn in the emergent zones of urban and nonurban wetlands.

Metals accumulations may be linked to soil toxicity, as estimated by the Microtox method. The Microtox test assesses the potential toxicity of an environmental sample by measuring the reduction of the light output of bioluminescent bacteria when exposed to the sample for a period of time. The method yields the effective concentration (EC), which indicates the reduction of light output after a certain length of exposure. The lower the EC value, the more toxic the sample is (Horner et al. 1988). In the synoptic study, urban open water zone soils had significantly lower ECs in both 5- and 15-minute tests and were, therefore, relatively more toxic. There was also a significant difference in emergent zone in the 15-minute test. However, there were no significant differences between the inlet and scrub-shrub zones of urban and nonurban wetlands.

Microtox analysis of wetland soils in 1993 failed to confirm the conclusion of Horner et al. (1988) that urban wetland soils were more toxic. It should be noted that only one 1993 sample from each wetland underwent Microtox analysis, and there was no attempt to compare the toxicity of various wetland zones, as in the synoptic study. Nevertheless, the 1993 results generally indicated that urban wetland soils were certainly no more toxic than those of rural wetlands. In fact, the three soils with the most toxic compounds came from less urbanized wetlands. The extraction and concentration of naturally occurring organic soil compounds in the laboratory, and not the presence of anthropogenic toxic substances, probably explained the results for these wetlands (Houck 1994). The results suggested that the soils of FC1, an urban wetland, and AL3, a rural wetland, possibly contained anthropogenic toxicants, because the results indicated toxicity in the absence of visible organic material. There were no evident accumulations of toxicants in the AL3 soil in 1993. The FC1 wetland, on the other hand, had the highest result for Cu (59 mg/kg), a Pb concentration (60 mg/kg) second only to the highly urban B3I, and a total petroleum hydrocarbon concentration (TPH) (840 mg/kg) more than three times greater than for any wetland except B3I, which exhibited an equally high value. That metals and TPH should be high at FC1 and B3I is not surprising in view of the intensity of commercial and transportation land uses in their watersheds.

Working with 1993 data, Valentine (1994) studied the efficacy of using regression relationships between widely distributed crustal metals (aluminum, Al, and lithium, Li) and toxic metals in relatively unimpacted wetlands to evaluate whether particular wetlands have enriched concentrations of toxic metals in their soils. The method was applied independently to the 1995 data. The regression analysis is based initially on relationships between crustal and heavy metals in relatively pristine wetlands that, it is assumed, have not received significant metal loadings of anthropogenic origin. These regressions must be developed for each region, since the natural background of metals varies with soils. If, in a given wetland, the concentration of a toxic metal is above a given confidence limit (95% in Valentine's study) of the linear regression, it is probable that there has been anthropogenic toxic metal pollution of the wetland's soil.

For the purpose of her 1994 study, Valentine divided the program wetlands into the same three groups outlined earlier in this chapter. She used the nomenclature Group 1 for nonurban (N) wetlands, Group 2 for moderately urbanized (M) ones, and Group 3 for highly urbanized (H) cases. In 1996, she employed only two groups, one less urban and the other more urban. Using the 1993 soil metals data, Valentine (1994) found that Li may be as good or better a reference metal than Al for As, Pb, and Zn. Nickel (Ni) bore a stronger relationship to Al, while Cu correlated equally well with both Al and Li.

Figures 9-1 and 9-2 illustrate the assessment tool using a nickel-aluminum pairing with 1993 and 1995 data, respectively. The regression line represents the best-fit line of the Group 1 (N) wetlands. The 95% confidence limits are the upper and lower bounds for one additional sample that is being assessed for Ni contamination. Sample contamination is gauged by considering the corresponding point's location on the graph. If the point lies on or above the upper 95% confidence limit, then the sample is judged to be enriched with the contaminating metal. Thus, in Figure 9-2, for example, seven samples above the line were judged to have Ni contamination of anthropogenic origin. The two relationships from the separate data sets exhibit a close correspondence.

Valentine (1994) found that the most urbanized wetlands had a higher rate of soil metals enrichment than moderately urbanized wetlands, considering both the Al and Li regressions. There were far fewer indications of metals enrichment in the moderately urbanized wetlands in 1993. The regressions of 1993 Pb with both Al and Li strongly agreed that most soil samples from each of the most urbanized wetlands were Pb enriched. The first set of regressions of As with both Al and Li generally agreed that soil samples from each of the most urbanized wetlands were As-enriched. Both Cu regressions using 1993 data indicated Cu enrichment in two of the most urbanized wetlands, which are also the wetlands listed as highly urbanized wetlands in Table 1 of Section 1. However the Li-Cu regression using 1995 data indicated enrichment in only two nonurban and one urban wetlands. The Al-Ni regression with 1993 data indicated Ni enrichment in three of the four most urban wetlands, although the Li-Ni relationship failed to show any enrichment in these sites. For 1995 data, no Ni enrichment appeared in the less urban wetlands, while there were five and six cases of enrichment according to the Al-Ni and Li-Ni regressions, respectively, in the more urbanized wetlands. The Li-Ni regression using 1995 data showed enrichment in all cases in which the Al-Ni regression also indicated enrichment. The first set of regressions for Zn showed a few cases of enrichment in the most urban group, although fewer in number and with less agreement between the regressions than for the other metals. The Li-Zn relationship in the 1995 data indicated Zn enrichment in ten of the more highly urbanized wetlands, in comparison to only one of the less urbanized wetlands.

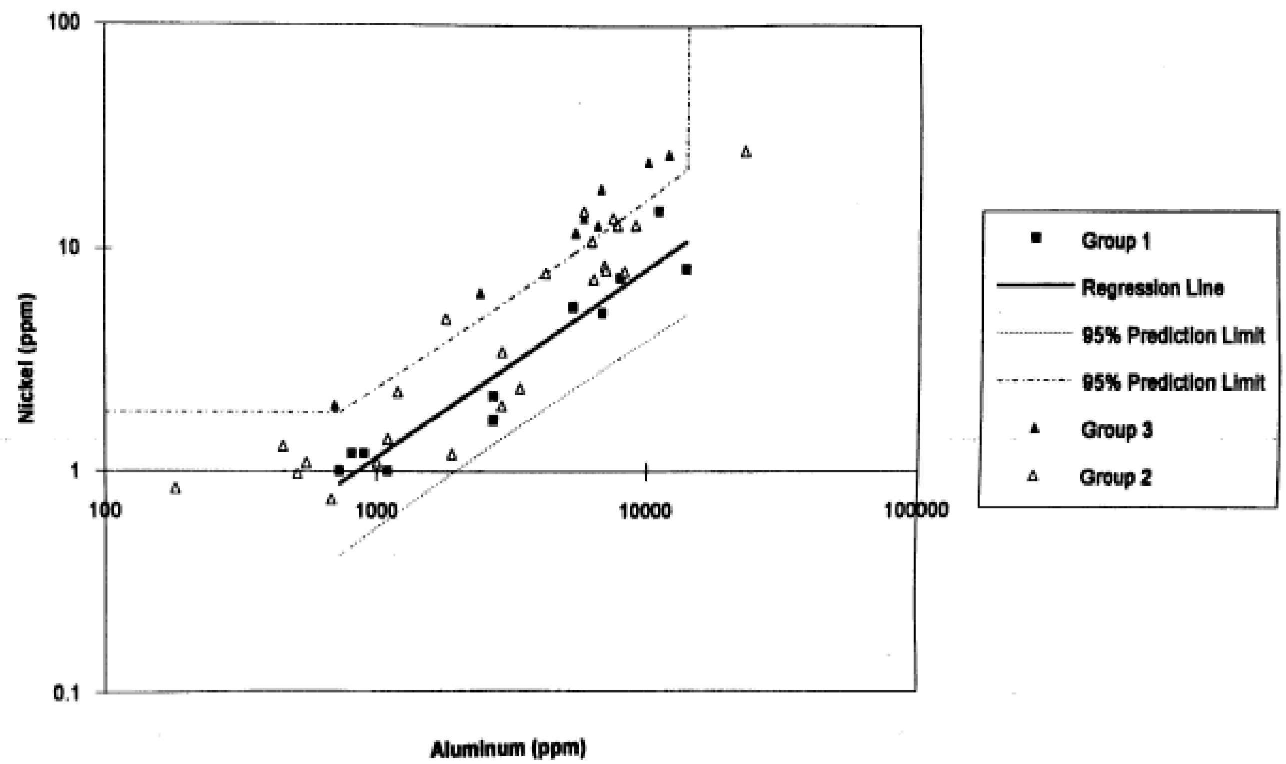

Figure 9-1. The Assessment Tool for NI Using Al as the Reference Element 1993 Data.

Although Valentine classified some of the wetlands in different groups than they would be in according to the GIS analysis, the results of her study agree well with observations based on the soil data statistics. Therefore, anthropogenic sources clearly impact the sediments of palustrine wetlands in the Puget Sound Basin. While wetlands can remove metals from the water column, the accumulation of metals could still harm wetland functions. Valentine noted that long-term effects of atmospheric emissions from past operations of the ASARCO smelter in Tacoma on wetland soils are unknown. It is possible that such distant sources could play a role in the enrichment of toxic metals wetlands. Rainfall removes such suspended metals from the atmosphere and provides the runoff which transports the metals to wetlands, where they accumulate in the sediments.

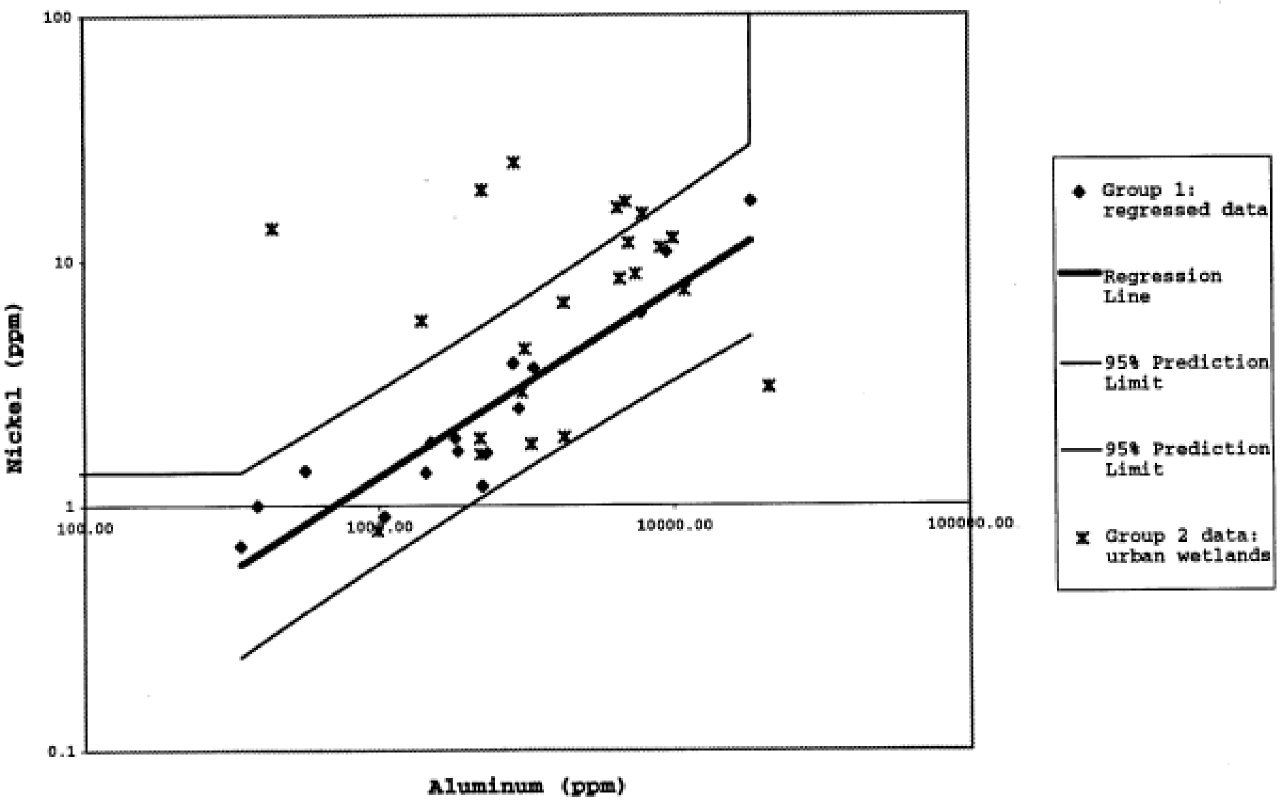

Figure 9-2. The Assessment Tool for NI Using Al as the Reference Element (1995 Data).

Treatment Wetlands

Table 9-3 shows soil statistics for the five treatment wetlands in the baseline period, when little or no urbanization had occurred (1988-1990), and then the later years (1993 and 1995), after most of the changes in land use were either well underway or complete.

It appears from program data that soil pH may have increased over the years at BBC24 and, especially, NFIC12, the wetlands whose watersheds underwent the greatest amounts of development. For NFIC12, rises in pH are entirely expected because (1) this wetland was a late successional peat bog that had the lowest soil pH readings of any of the wetlands, with no place to go except up; and (2) its watershed went from 0% to 100% urbanization between 1989 and 1995.

The treatment wetlands exhibited the particle size distributions in 1989 and 1995 shown in Table 9-4. BBC24 and ELS61 both registered transition from relatively sandy to silty soils, while clay stayed constant. NFIC12 exhibited a clay increase while sand was constant. The first result could be explained by sedimentation of finer particles over the pre-development substrate, but a 30% increase in clay is not easily explained.

Table 9-3. Soil statistics for treatment wetlands in baseline and post-development years.

Year	Stat.	pH	Redox. Potential (mV)	TP (mg/kg)	TN (mg/kg)	Volatile Solids (%)	Cu (mg/kg)	Pb (mg/kg)	Zn (mg/kg)	As (mg/kg)
JC28	Mean	5.89	12	964	6661	49.6	19.5	33.2	27.1	6.4
88-90	St. Dev.	0.11	240	637	4578	26.5	3.2	15.1	23.7	2.0
	CV	2%	2003%	66%	69%	53%	16%	46%	88%	31%
	Median	5.87	-20	578	6056	59.9	20.4	32.6	18.0	7.1
	n	7.00	7	7	8	9.0	7.0	7.0	7.0	7.0
93-95	Mean	5.63	545	153	1392	13.1	5.0	8.1	3.4	1.8
	St. Dev.	0.32	102	150	1241	1.2	2.6	2.3	2.4	1.6
	CV	6%	19%	98%	89%	9%	51%	28%	71%	89%
	Median	5.53	608	176	1980	12.8	4.3	8.5	2.7	1.2
	n	5.00	5	5	5	5.0	5.0	5.0	5.0	5.0
ELS61	Mean	5.70	162	755	4409	33.6	31.8	60.3	47.8	9.0
88-90	St. Dev.	0.34	187	716	3821	22.6	20.3	20.3	24.8	5.4
	CV	6%	115%	95%	87%	67%	64%	34%	52%	60%
	Median	5.84	70	1024	3443	29.5	31.0	32.4	56.6	9.2
	n	11.00	11	13	14	14.0	13.0	13.0	13.0	13.0
93-95	Mean	5.70	164	748	4382	33.8	31.6	60.2	47.6	9.0
	St. Dev.	0.71	298	973	5092	29.3	158.4	77.9	64.1	16.8
	CV	13%	182%	130%	116%	87%	502%	129%	135%	187%
	Median	5.67	547	69	537	8.1	12.7	9.5	14.3	2.8
	n	191.00	188	194	204	208.0	195.0	195.0	195.0	195.0
PC12	Mean	6.08	107	1285	8431	52.5	26.1	170.7	52.7	14.3
88-90	St. Dev.	0.29	291	1122	6813	23.2	8.7	197.8	31.5	7.2
	CV	5%	273%	87%	81%	44%	33%	116%	60%	50%
	Median	6.04	114	1089	6347	59.2	26.9	105.1	39.7	12.5
	n	7.00	7	6	7	6.0	6.0	6.0	6.0	6.0
93-95	Mean	6.15	351	111	911	6.9	3.7	9.1	7.5	1.7
	St. Dev.	0.14	99	115	866	1.5	0.9	6.0	3.2	0.5
	CV	2%	28%	104%	95%	21%	24%	65%	43%	32%
	Median	6.10	310	109	1060	7.0	3.6	10.3	6.6	1.8
	n	5.00	5	5	5	5.0	5.0	5.0	5.0	5.0
BBC24	Mean	5.90	-197	469	4171	22.7	19.1	44.9	39.0	11.0
88-90	St. Dev.	0.35	15	434	4802	17.6	12.4	25.2	16.5	5.5
	CV	6%	-8%	93%	115%	77%	65%	56%	42%	50%
	Median	5.73	-200	297	1692	15.3	17.9	51.6	48.0	10.7
	n	6.00	6	6	7	7.0	7.0	6.0	6.0	7.0
93-95	Mean	5.97	326	43	441	7.4	5.0	4.4	6.0	1.4
	St. Dev.	0.24	49	54	596	3.7	2.9	0.7	0.5	0.3
	CV	4%	15%	126%	135%	50%	58%	16%	9%	26%
	Median	5.97	326	43	441	7.4	5.0	4.4	6.0	1.4
	n	2.00	2	2	2	2.0	2.0	2.0	2.0	2.0

Table 9-3 continued. Soil statistics for treatment wetlands in baseline and post-development years.

Year	Stat.	pH	Redox. Potential (mV)	TP (mg/kg)	TN (mg/kg)	Volatile Solids (%)	Cu (mg/kg)	Pb (mg/kg)	Zn (mg/kg)	As (mg/kg)
NFIC12	Mean	3.97	144	2188	15984	76.0	19.5	48.6	10.4	9.2
88-90	St. Dev.	0.26	345	1138	7440	18.9	10.1	40.0	5.3	5.6
	CV	7%	240%	52%	47%	25%	52%	82%	51%	61%
	Median	3.94	140	2182	15698	81.0	16.3	44.4	10.9	9.9
	n	4.00	4	4	4	4.0	4.0	4.0	4.0	4.0
93-95	Mean	5.05	633	167	1950	12.5	2.5	10.3	3.0	2.2
	St. Dev.	1.05	2	67	113	0.7	0.3	0.8	2.2	0.3
	CV	21%	0%	40%	6%	6%	13%	8%	71%	13%
	Median	5.05	633	167	1950	12.5	2.5	10.3	3.0	2.2
	n	2.00	2	2	2	2.0	2.0	2.0	2.0	2.0

Table 9-4. Particle size distributions for treatment wetlands.

	%Sand/Silt/Clay	
Wetland	1989	1995
BBC24	61/29/10	45/47/8
ELS61	35/49/16	15/69/16
JC28	52/35/13	46/36/18
NFIC12	4/59/37	3/32/65
PC12	58/37/5	63/30/7

Otherwise, there was a strong trend for redox to rise but for nutrients, organic content, and metals all to fall from the pre-development to the post-development years. The reasons for these unexpected results can only be given speculation. What can be said is that, other than the pH increase at NFIC12, there is no obvious signal in the soils yet that negative changes may have accompanied recent urbanization.

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CHAPTER 10 THE HYDROLOGIC REQUIREMENTS OF COMMON PACIFIC NORTHWEST WETLAND PLANT SPECIES

by Sarah S. Cooke and Amanda L. Azous

INTRODUCTION

The vegetation and associated hydrologic regime of wetland study sites located in the Puget Sound Basin was evaluated between 1988 to 1995. Our study examined the role of hydrology in determining the vegetation composition of wetlands in the region. The observed hydrologic regime of common vegetation communities as defined by Cowardin et al. (1979) was evaluated and included forested, scrub-shrub, emergent, and aquatic bed type communities.

Additionally, the hydrologic requirements of individual species was examined in order to determine the optimal conditions and tolerances for some common wetland plants. Several hydrologic conditions present where plant species were growing, including water depth and water level fluctuation (WLF), were examined seasonally and annually. This paper presents an analysis of some of these vegetation and hydrology associations.

METHODS

The wetlands evaluated in this study are inland palustrine wetlands ranging in elevation from 50 m to 100 m above mean sea level and characterized by a mix of forested, scrub-shrub, emergent, and aquatic bed wetland vegetation classes. Twenty-six wetlands total were surveyed. In addition to the nineteen study wetlands surveyed at least three times between 1988 and 1995, the data set also includes seven other wetlands which were surveyed at least once during the years 1993, 1994 and 1995 as part of several related studies.

Sample plots were assigned a category based on the dominant structure of the vegetation community classified in the Cowardin system (Cowardin et al. 1979). The categories included aquatic bed (PAB), emergent (PEM), scrub-shrub (PSS), forested (PFO), upland, or some transition zone between them, for example, PEM/PSS. In some cases vegetation communities changed over time and were then re-categorized. Plant species presence was sampled in permanent plots established every 50 M installed along a gradient from the upland through the transition zones and, at intervals, crossing the different wetland vegetation communities.

The year was divided into four seasonal periods important to plant growth, early growing season, defined to be from March 1 through May 15, intermediate growing season which lasts from May 16 to August 31, senescence lasting from September 1 to November 15th and dormancy and decay, November 16 to February 28. The seasonal hydrologic regime was calculated for each vegetation sample station from 1988 to 1995. Species found in each sample station were associated with the seasonal hydrologic regime we observed at the station . This data was used to describe a hydrograph for many commonly found wetland plants showing typical conditions for mean and maximum water depths and water level fluctuation. This method presumes that plant species presence is associated with conditions favorable to their survival and that, with sufficient observations, ranges of hydrologic conditions successfully tolerated by species, could be determined.

Hydrologic measurements, including instantaneous water levels from staff gauges for measuring typical water level conditions and peak levels from crest gauges to measure depths from storm events occurring between gauge visits, were recorded at least eight times annually (measured every four to six weeks) while water was present in the wetlands (Reinelt and Horner 1990). Gauge measurements were averaged to obtain mean and maximum water depth for each season or for the year. Water level fluctuation (WLF) was computed as the difference between the peak level and the average of the current and previous instantaneous water levels for each four to six week monitoring period. Mean WLF was calculated by averaging all WLFs for a given season or year.

The elevation of each vegetation community was surveyed in order to tie the hydrologic conditions found at the gauge site with the vegetation communities observed at the sample stations. The elevation of all vegetation plots relative to the water depth measured at the wetland gauge was used to determine the likely water depths at the vegetation sample stations for a given sampling date (Figure 10-1).

The method assumed that the water depth measured at the staff gauge would accurately reflect the hydrologic conditions found throughout the wetland, after the data was corrected for elevation. This was not always true as hydrologic conditions in and around the plots sometimes varied from the conditions predicted through elevation change alone. Sometimes intervening topography or large woody debris would produce localized impoundments or dry hummocks unaccounted for in our methods which may affect the accuracy of our results. Whenever such plots were identified we made more accurate surveys or eliminated the plots from the analysis. In addition, our method did not determine whether soils were dry or saturated. We could only estimate saturation given the water depths found at the gauge.

$$D = G - E$$

where: D = water depth at the vegetation plot
G = gauge reading, depth above zero water level and
E = plot survey elevation above zero water level

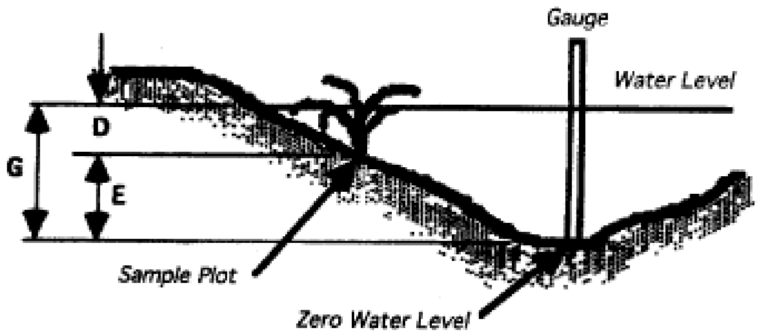


Figure 10-1 Relationship between water depth at vegetation sample plot and the depth at the water gauge (Taylor 1993).

Negative numbers in the hydrographs are interpreted as below the water surface, or inundated. Positive numbers represent the distance, in elevation, above the water surface. Negative numbers are interpreted as depth of inundation. Positive numbers indicate the plant or community being examined is dry during that period. Bar charts show the median and the central 80% of the range of observations for the condition being evaluated. The solid portion of the bar represents the central 50% of observations. We eliminated outliers from our analysis because we wanted to define the most likely range of wettest to driest conditions where particular wetland communities or species would be found.

RESULTS

Hydrologic Regimes By Community Type

The range of average conditions we calculated for instantaneous and maximum water depths found during the study period was in the PEM, PFO, PAB, and PSS communities are displayed in Figure 10-2. The solid bars in the figure show 50% of observations and, including the tails, represent 80% of observations. Forested communities were, as expected, the driest of the community types with a median of 62 cm above typical water levels, and ranged from about 12 cm inundation to about 210 cm above typical water levels. By contrast, the annual instantaneous median condition in the emergent zones was about 5 cm inundation and ranged from 96 cm of inundation to about 28 cm above typical water levels.

The biggest variation from wet to dry conditions through the year was observed in the PSS communities where the median condition for instantaneous water depths was 18 cm above water levels but decreased to 0 for maximum water depth conditions. This corresponds to field observations of different types of shrub communities. Willow dominated communities (*Salix lucida* var. *lasiandra*, *S. sitchensis*) including red stem dogwood (*Cornus sericea*) represent the wetter shrub communities, while Scouler willow (Salix scouleriana), salmonberry (*Rubus spectabilis*), and black twinberry (*Lonicera involucrata*) represent the drier shrub communities. While the central 50% of observations depict the scrub-shrub communities as saturated to dry under normal conditions, during storm events, represented by the maximum depth, conditions in the PSS zones were much wetter, ranging between about 40 cm of inundation to 25 cm above water levels.

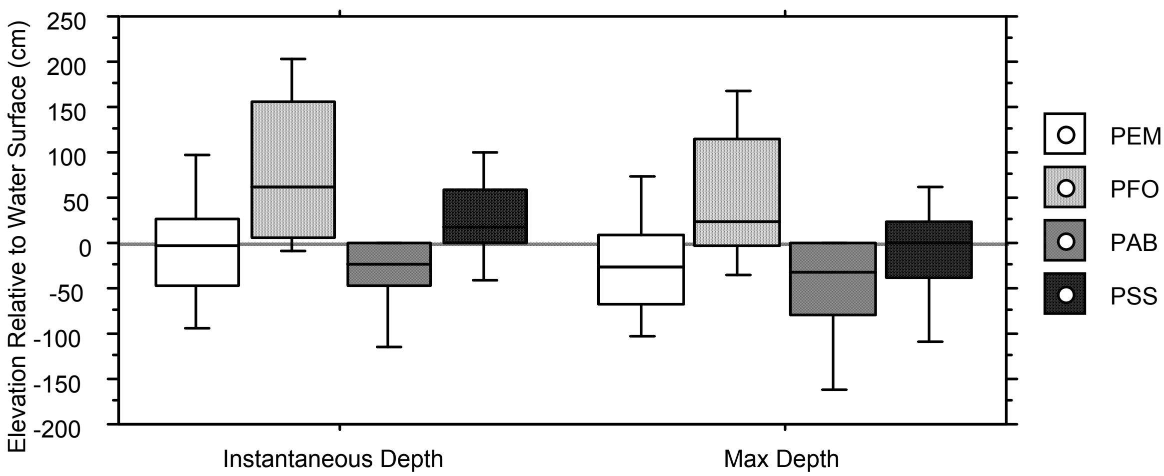


Figure 10-2. Annual mean instantaneous and maximum water depths (Max) associated with vegetation community types, 1988 through 1995.

In fact, annual water level fluctuation averaged 21 cm among all scrub-shrub zones as compared with about 12 cm in the aquatic bed communities and 14 cm in the emergent zones. Forested zones were usually at an elevation above surface inundation, so water level fluctuation was not a significant factor. Aquatic bed communities were observed to have very high water level fluctuations averaging 60 cm as compared with 11 cm and 18 cm, respectively, for emergent and scrub-shrub zones. Figure 10-3 shows the median and range of water level fluctuation calculated in each zone for all four seasons. Open water and scrub shrub zones showed the greatest variation in water level fluctuation between seasons while emergent zones were fairly stable. The median WLF for the aquatic bed zones was always less than 20 cm however the range of observations was quite a bit higher than seen in other community types, in all seasons, but particularly the early growing season of March through May.

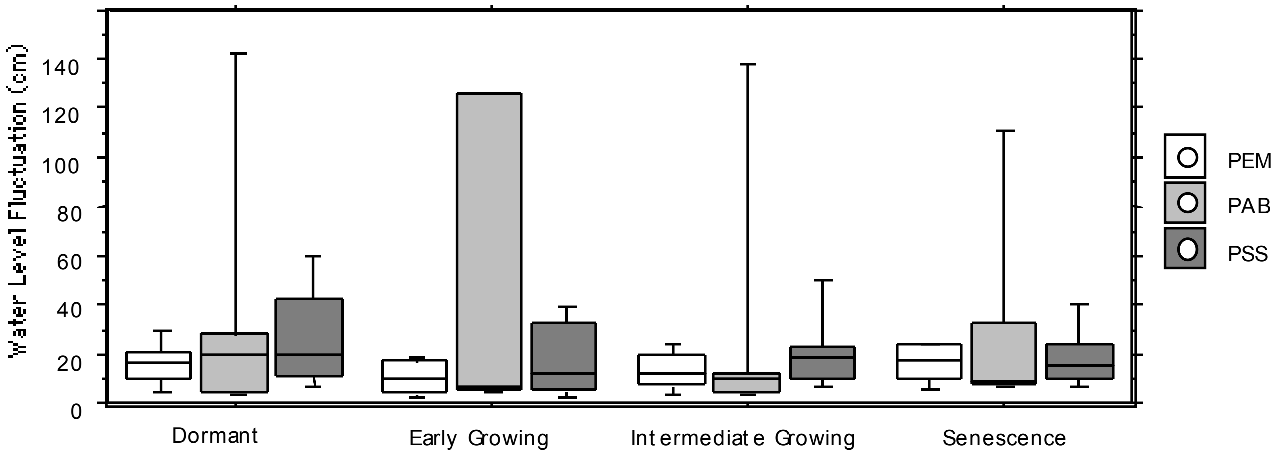


Figure 10-3. Water level fluctuation associated with vegetation community types in each season.

The range of instantaneous and maximum water levels for all seasons were plotted for each vegetation zone and are shown in Figures 10-4 through 10-7. The seasonal changes in each community are apparent in the box plots. The period of senescence, September through November is definitely drier in all zones, including the aquatic bed communities. Most aquatic bed zones had no surface water during this period except during storm events although many were observed to have saturated soils. Most emergent communities were inundated most of the year with the exception of during senescence. Forest wetland communities were relatively wetter during the early growing season than other seasons unlike the other community types which were mostly wettest during the dormant season.

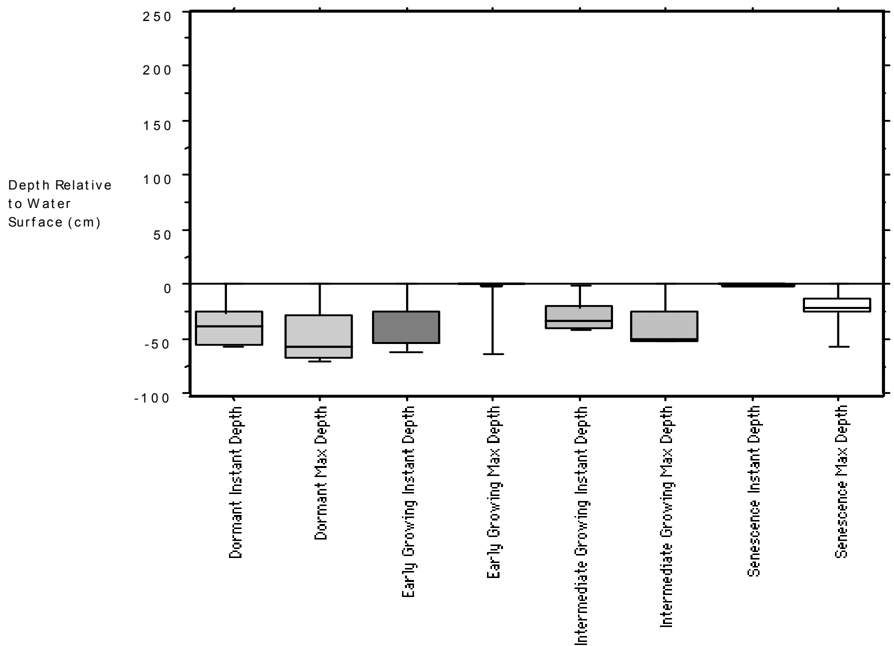


Figure 10-4. Seasonal hydrology associated with aquatic bed communities.

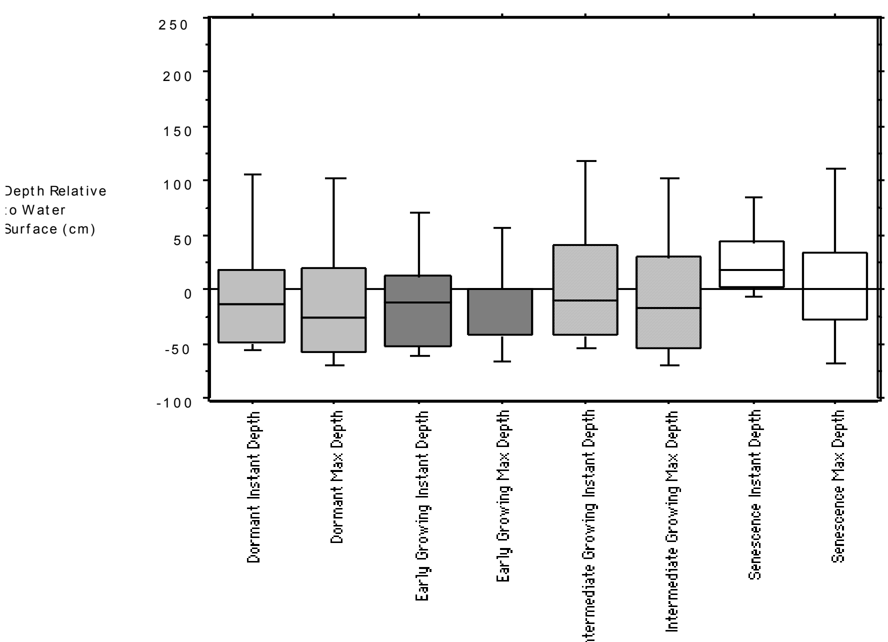


Figure 10-5. Seasonal hydrology associated with emergent communities.

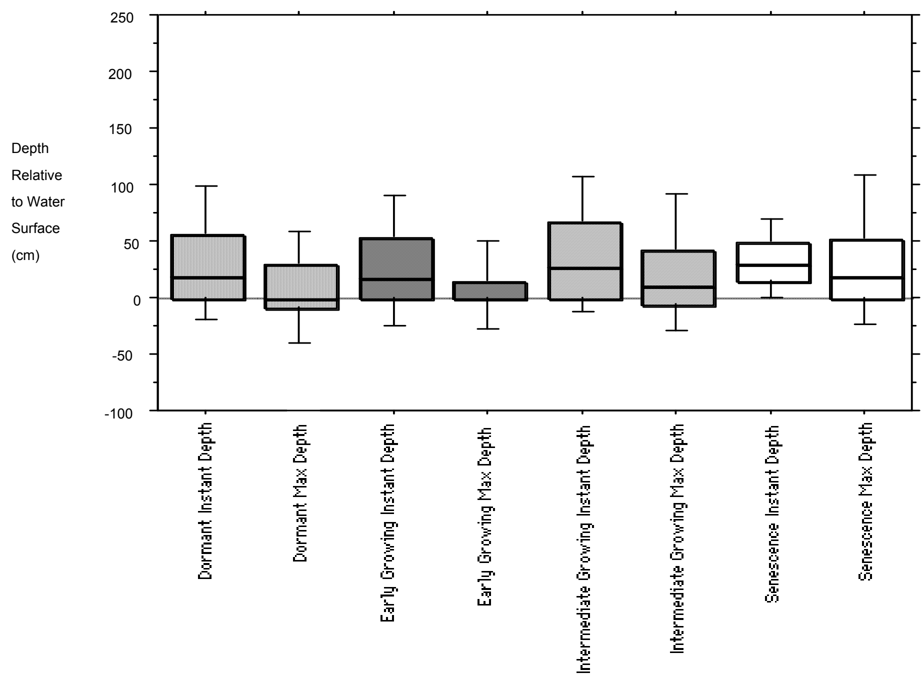


Figure 10-6. Seasonal hydrology associated with scrub-shrub wetland communities.

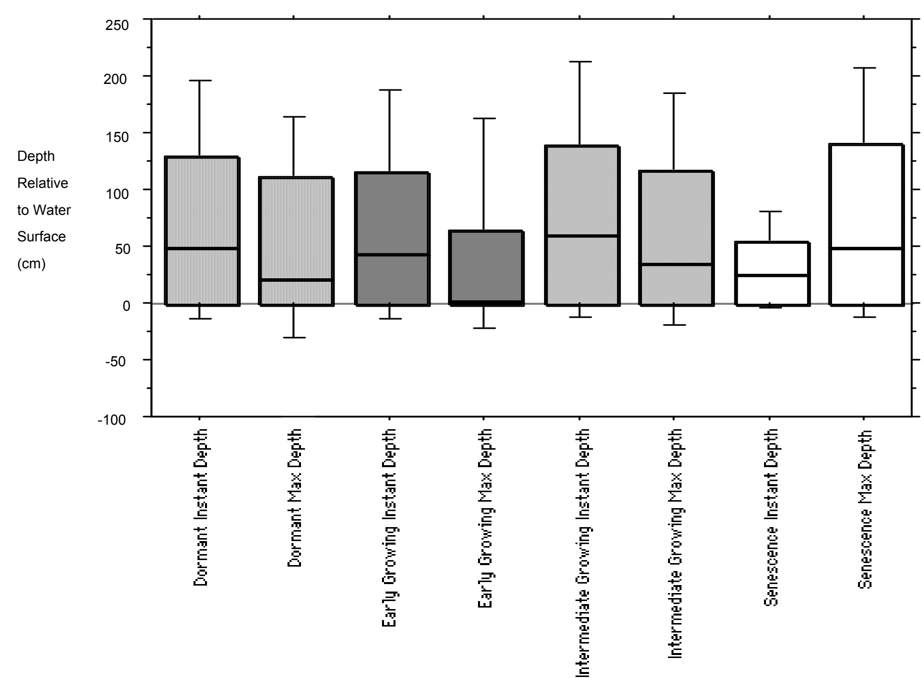


Figure 10-7. Seasonal hydrology associated with forested wetland communities.

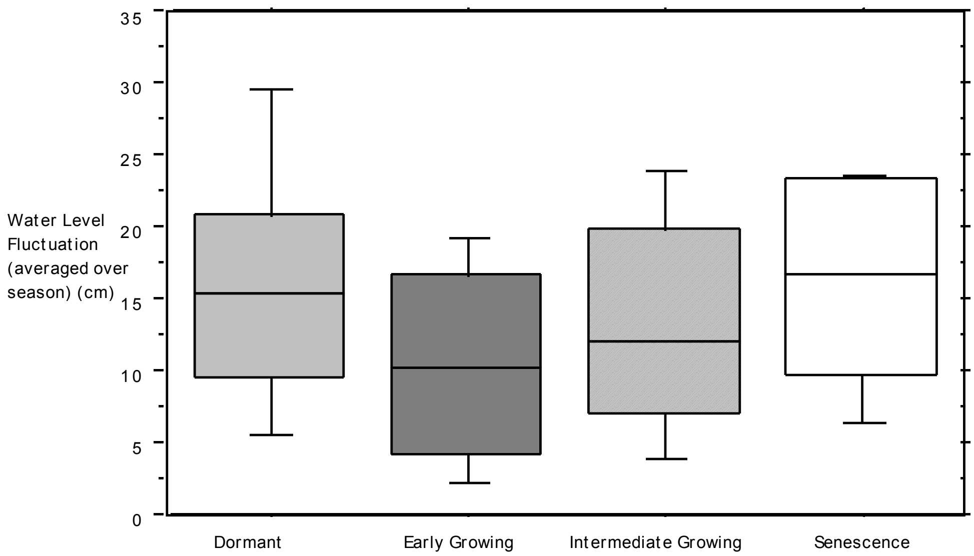


Figure 10-7. Seasonal WLF in the emergent zone.

Individual Plant Hydrologic Requirements

The hydrologic regime for some common wetland species was determined in the same manner as for the wetland zones. Black cottonwood (*Populus balsamifera* spp. *trichocarpa*) was mostly found in areas where there was little to no surface water during the active growing period but it was observed in stations which, on average, are inundated to 20 cm of water at some time during all seasons (Figure 10-8).

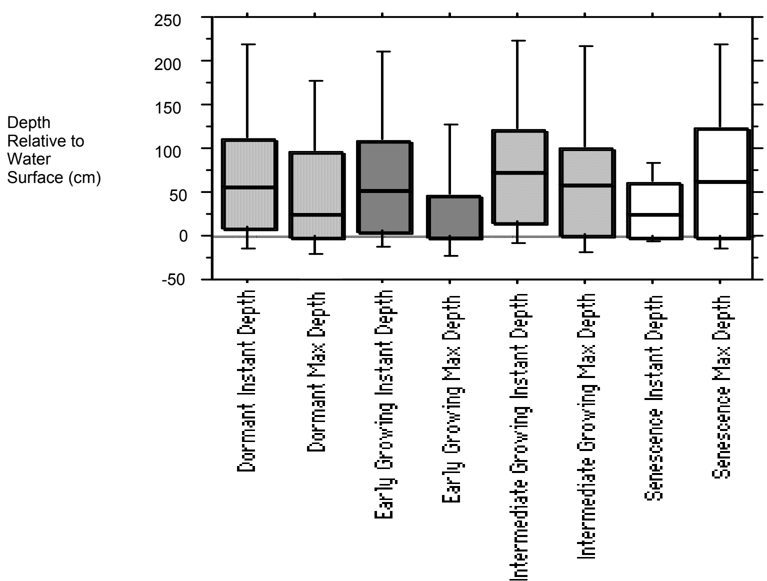


Figure 10-8. Hydrologic characteristics of instantaneous and maximum water depth (Max) of *Populus balsamifera*.

Hard hack spirea *(Spirea douglasii*) was found in a wide range of hydrologic conditions from typically dry through the year, to being frequently temporarily inundated, to complete inundation through both growing seasons (Figure 10-9). This adaptability is probably one reason why this species was among the most widely distributed in our study. In addition hard hack was found in wetlands with some of the highest water level fluctuations measured in our study, averaging as high as 57 cm in the dormant season and 35 cm in the early growing season.

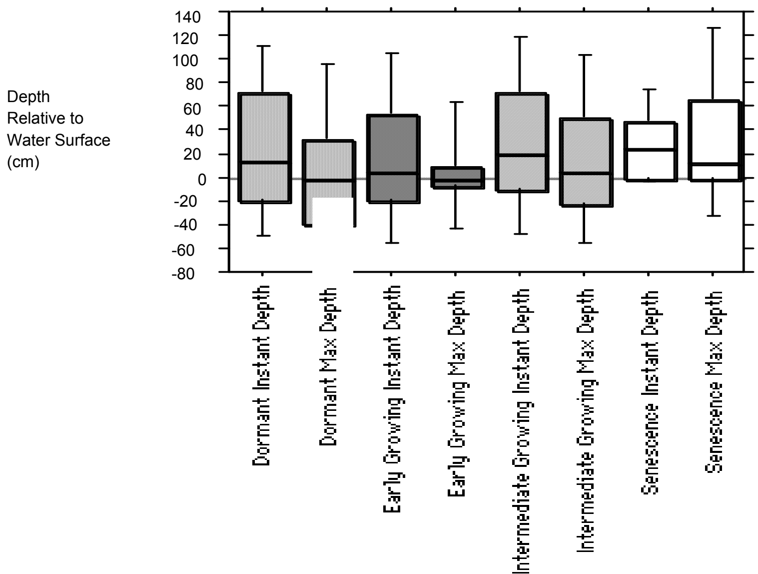


Figure 10-9 Hydrologic characteristics of instantaneous water depth, maximum water depth (Max) of *Spirea douglasii*.

Comparisons were made between plants that either fill the same niche, or are different species of the same genera but found in different habitats. Analysis of the hydrologic conditions where species were observed often showed seasonal hydrologic differences which may account for their distribution (Table 10-1). For example, two common wetland trees, red alder (*Alnus rubra)* and Oregon ash (*Fraxinus latifolia)* were both more prevalent on the drier sites we studied. Red alder differed, however, in that, it was found on many sites subjected to high average seasonal WLF (greater than 20 cm) during the early growing season, suggesting that it is frequently inundated for periods while Oregon ash was observed in areas with mostly stable water levels in the early growing season (Figure 10-10). Oregon ash was often observed in areas where the soil was organic rish and, though rarely inundated, soils were saturated for most of the growing season while red alder was mostly found growing in mineral soils that typically went dry in the summer.

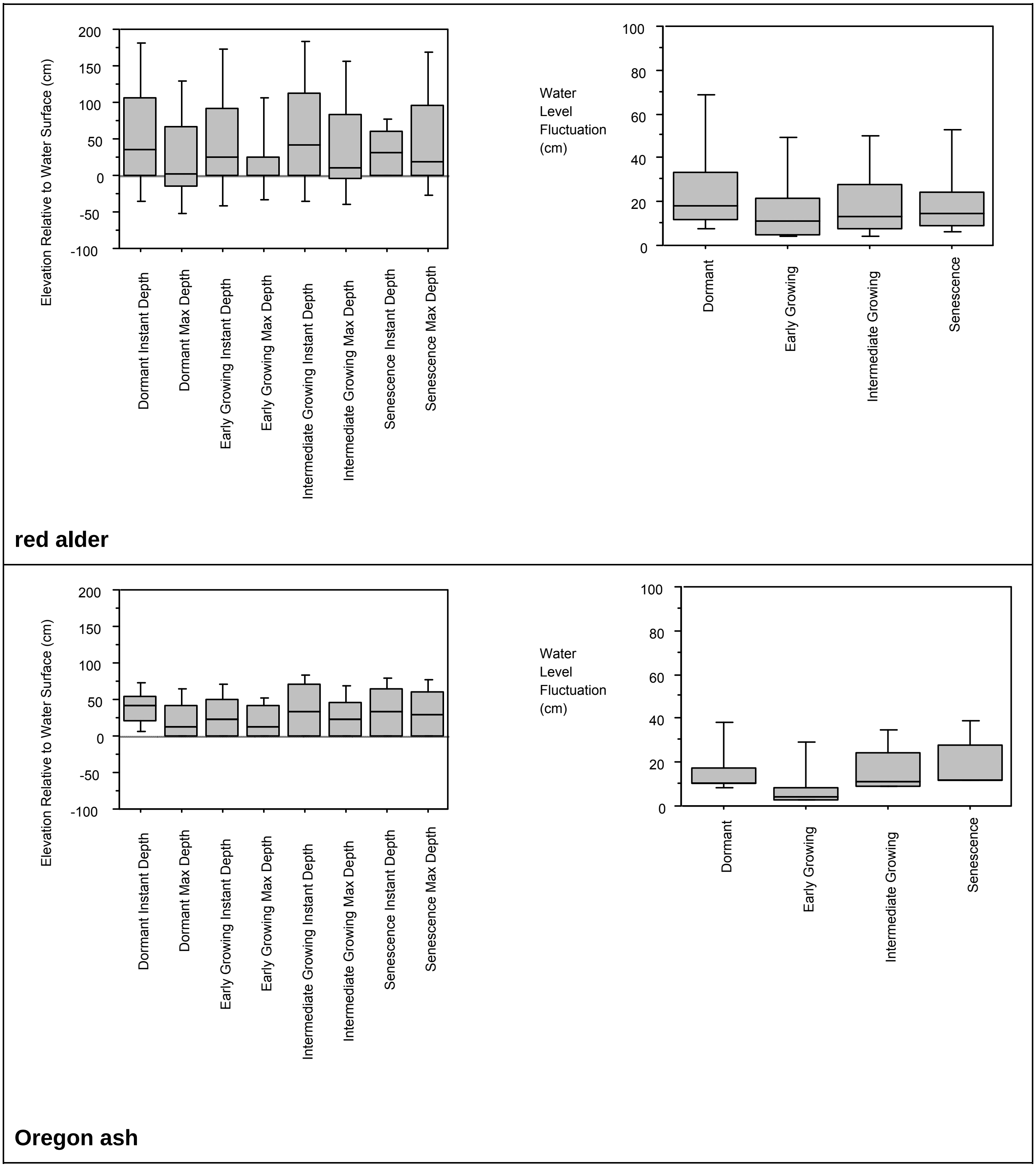


Figure 10-10. Seasonal hydrology and water level fluctuation of red alder and Oregon ash.

Scouler willow (*Salix scouleriana)* and Sitka willow *(Salix sitchensis*), two willows common in the Puget lowland, were both found in areas from inundated to dry in all seasons, but overall, the medians, means and most of the population data, indicate Sitka willow was growing closer to water than Scouler willow (Figure 10-11).

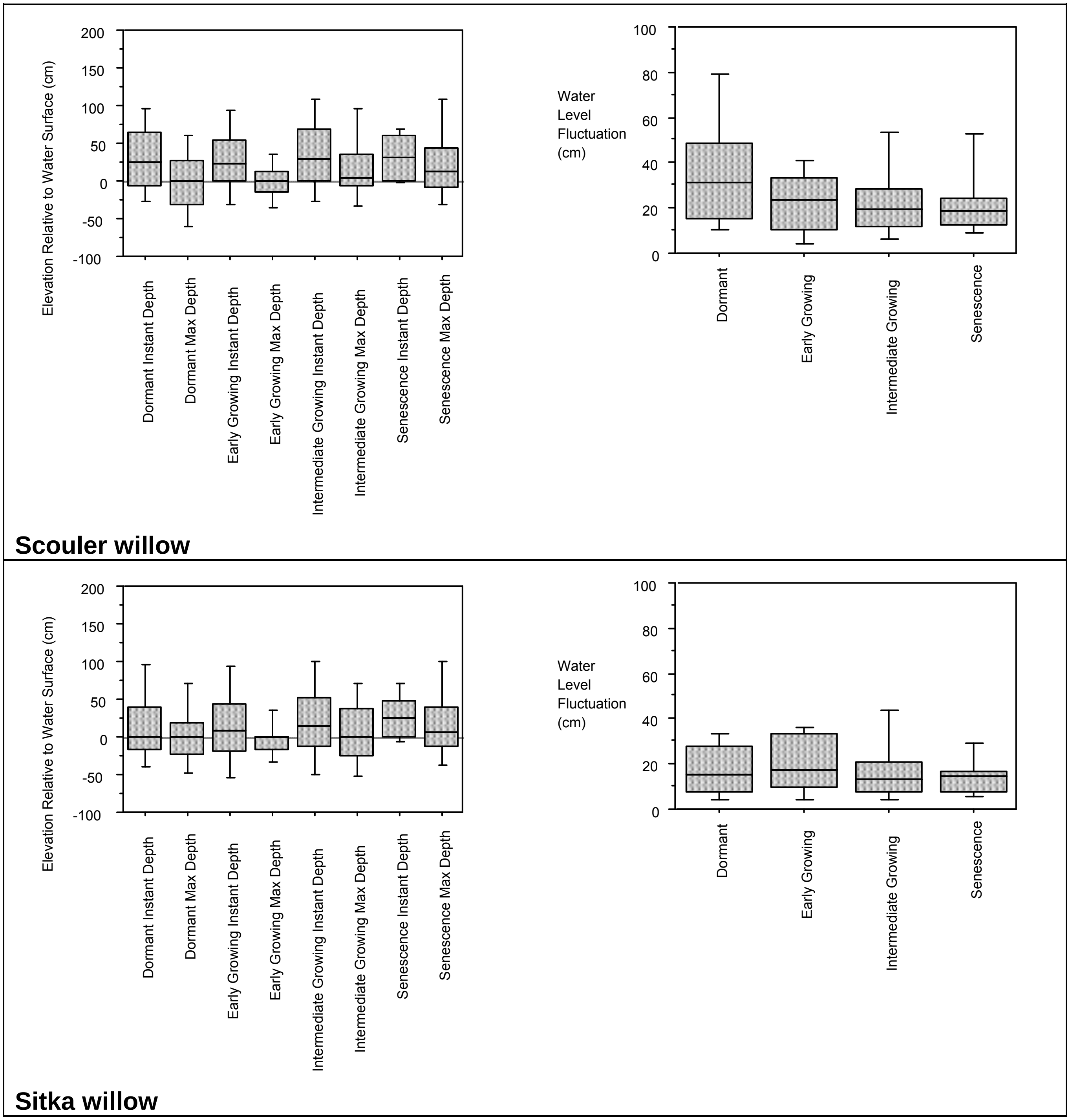


Figure 10-11. Seasonal hydrology and water level fluctuation of Scouler willow and Sitka willow.

Though both species were found growing in a similar range of conditions, tapertip rush (*Juncus acuminatus*) was found on more dry sites than soft rush (*Juncus effusus*) which was usually shallowly inundated during the early spring (Figure 10-12).

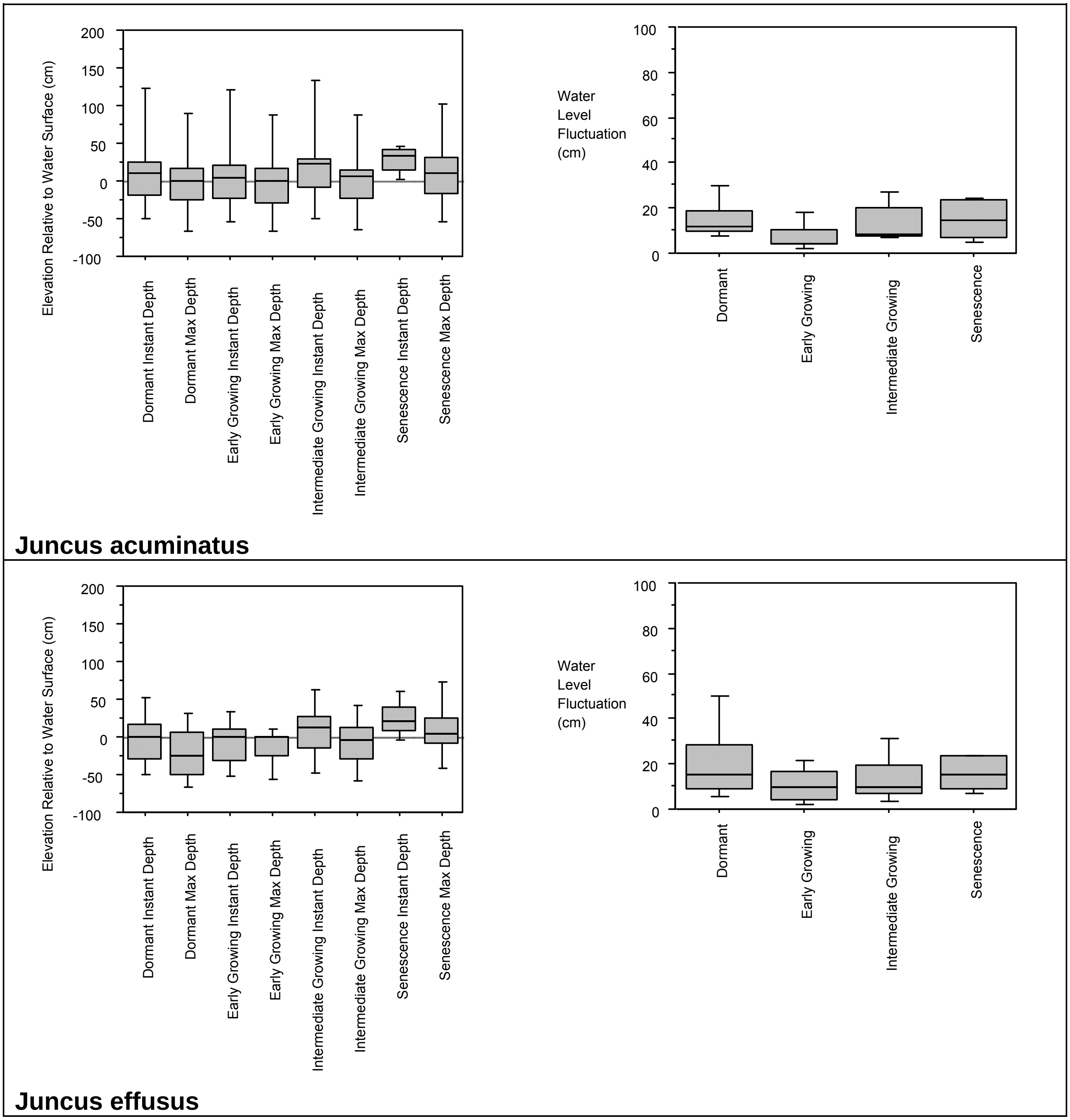


Figure 10-12. Seasonal hydrology and water level fluctuation of tapertip rush and soft rush.

Two sedges were evaluated, Slough sedge (*Carex obnupta)* which is very common in wetlands around the region, and inflated sedge (*Carex exsiccata*) (old name = *C. vesicaria*) which is found almost exclusively in relatively undisturbed wetlands (Figure 10-13). Slough sedge was found in drier areas above the water level during the early and intermediate growing seasons, while inflated sedge grew in saturated soils and areas of shallow inundation. Both species were found in areas inundated during the dormant season and both were found in conditions of high water level fluctuation.

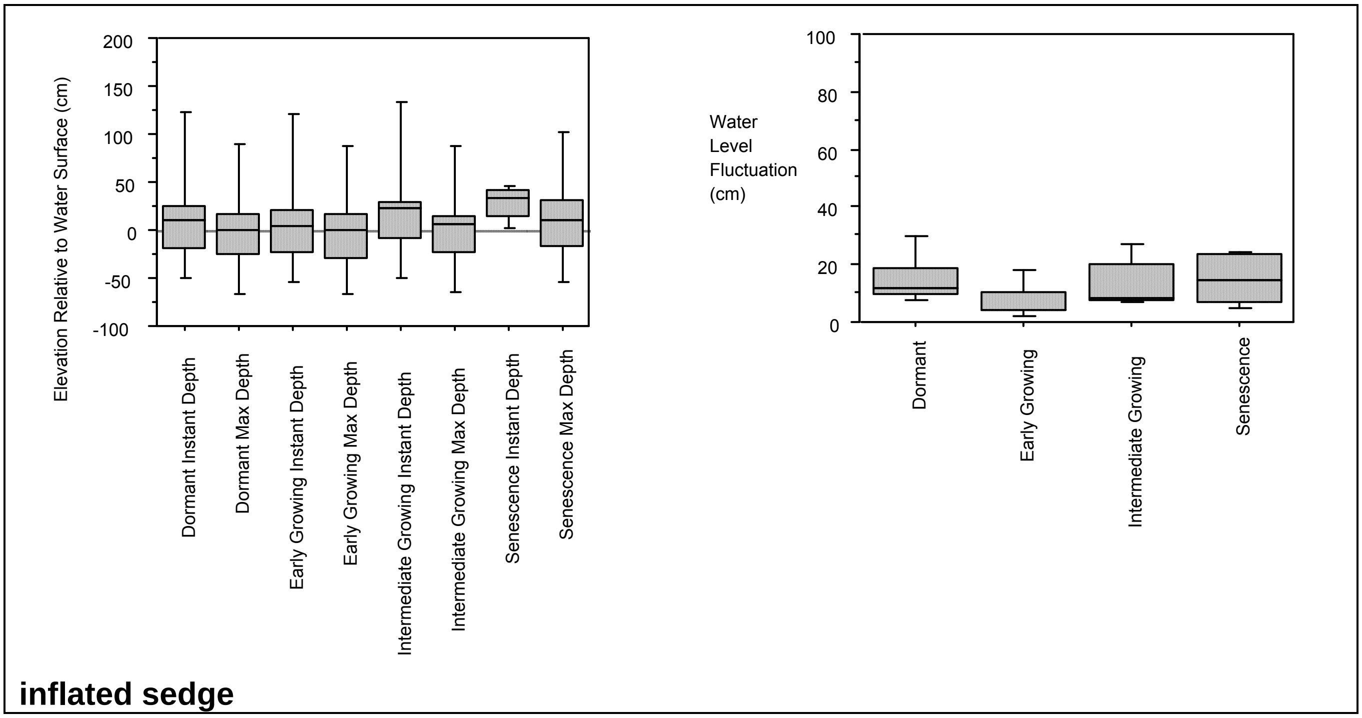
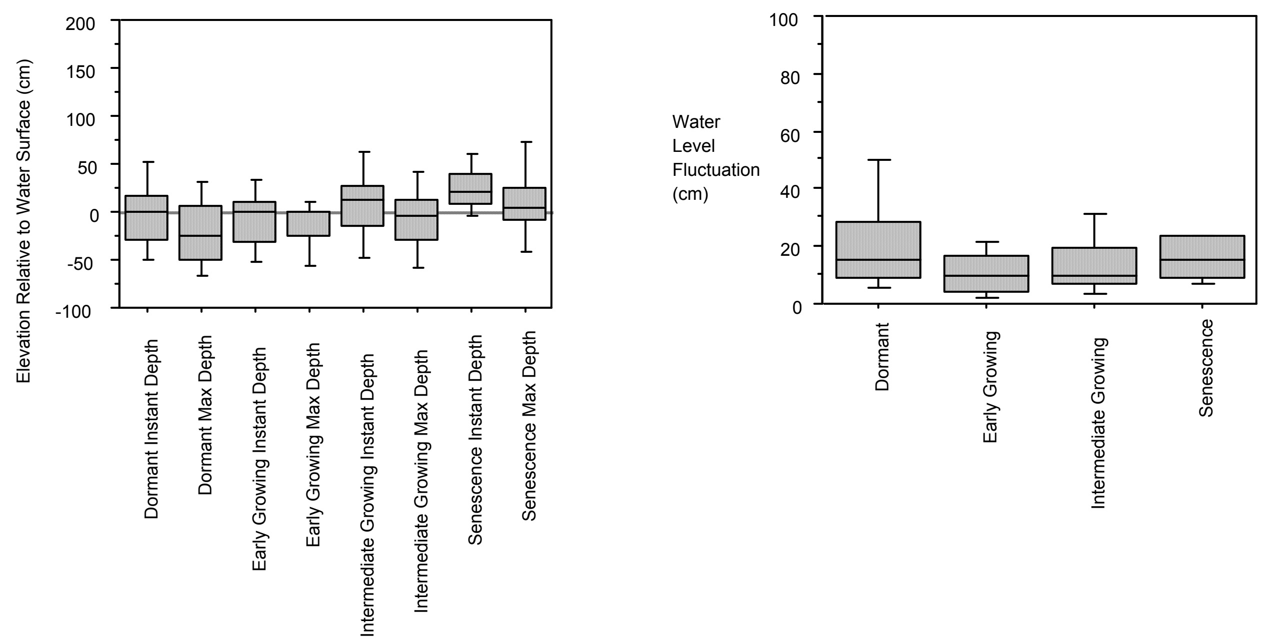


Figure 10-13. Seasonal hydrology and water level fluctuation of inflated sedge and slough sedge (figure continued next page).



slough sedge

Figure 10-13 (continued). Seasonal hydrology and water level fluctuation of inflated sedge and slough sedge.

Small fruited bulrush, *Scirpus microcarpus*, found in disturbed wetlands, and wooly sedge, *Scirpus atrocinctus* (old name = *S. cyperinus*), found in relatively undisturbed wetlands, were observed growing in similar hydrologic conditions (Figure 10-14). Wooly sedge, however, was found in slightly wetter conditions during the early growing season. In addition, small-fruited bulrush was found in wetlands with high WLF throughout the growing season where wooly sedge was not.

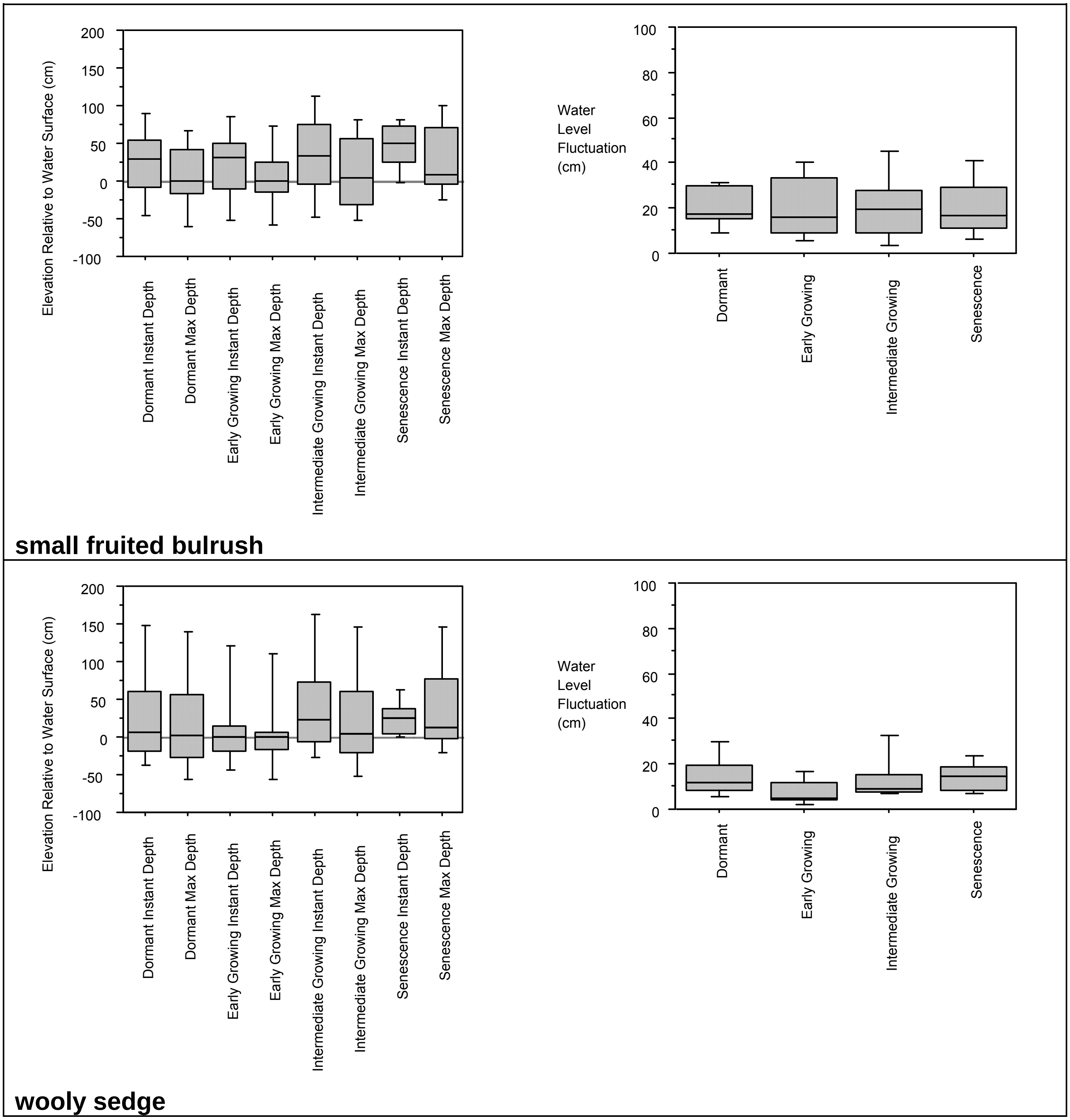


Figure 10-14. Seasonal hydrology and water level fluctuation of small fruited bulrush and wooly sedge.

Several invasive species including soft rush, reed canarygrass (*Phalaris arundinaceae)*, and cattail (*Typha latifolia)* were evaluated to see if there were hydrologic conditions common to aggressive species. Of the three, reed canarygrass grows in the driest areas, and cattail in the wettest (Figure 10-15). Reed canarygrass was found in many wetlands with very high seasonal WLF whereas cattail and soft rush were found in areas where there is low WLF except during the dormant period. The only consistent hydrologic condition shared between these species was their distribution, most commonly within the emergent zones of wetlands.

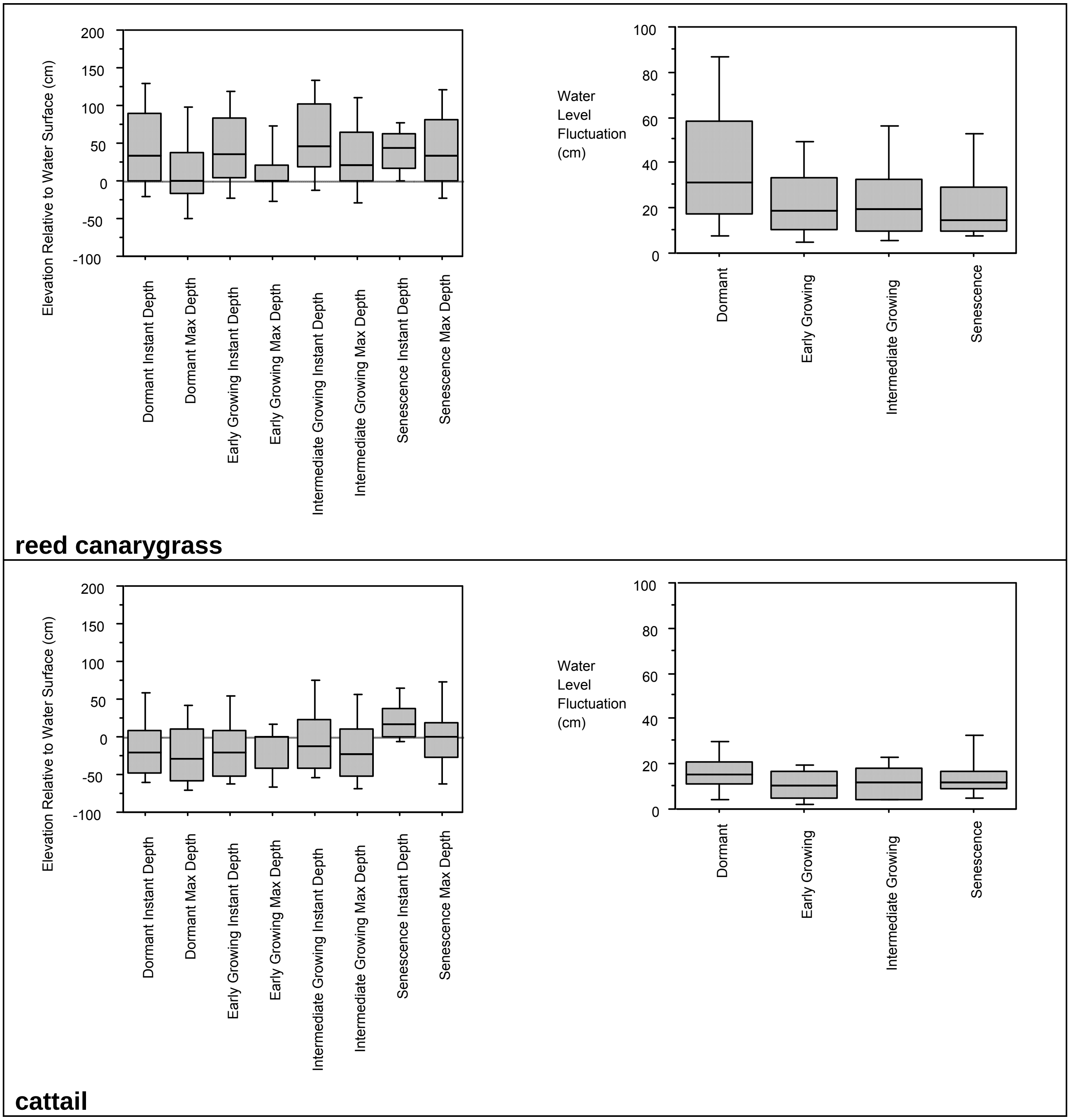


Figure 10-15. Seasonal hydrology and water level fluctuation of reed canarygrass and cattail.

Table 10-1 Comparison of different species and their hydrologic characteristics.

Measure Note: Negative numbers are depth under water. Positive numbers are distance from water surface.	Dormant Instant Depth	Dormant Max Depth	Dormant WLF	Early Growing Instant Depth	Early Growing Max Depth	Early Growing WLF	Inter-mediate Growing Instant Depth	Inter-mediate Growing Max Depth	Inter-mediate Growing WLF	Senes-cence Instant Depth	Senes-cence Max Depth	Senes-cence WLF
Species:	**Alnus rubra**											
Wettest Elevation From Water Surface (cm)	-97	-98	0	-98	-99	0	-97	-96	2	-10	-97	1
Driest Elevation From Water Surface (cm)	430	419	142	438	421	126	441	404	138	98	424	111
Median (cm)	35	2	18	25	0	11	41	11	13	30	20	15
Mean (cm)	58	32	30	48	21	21	60	41	21	33	51	21
Species:	**Fraxinus latifolia**											
Wettest Elevation From Water Surface (cm)	-55	-58	8	-62	0	2	-43	-56	9	-9	0	11
Driest Elevation From Water Surface (cm)	185	129	69	188	71	50	200	193	57	87	196	53
Median (cm)	42	13	10	23	12	4	34	23	11	33	29	12
Mean (cm)	40	22	18	31	20	9	43	31	18	36	37	20
Species	**Salix scouleriana**											
Wettest Elevation From Water Surface (cm)	-96	-98	4	-96	-100	0	-99	-99	0	-7	-94	5
Driest Elevation From Water Surface (cm)	286	261	139	237	194	116	233	185	95	82	235	84
Median (cm)	26	0	31	22	0	23	28	4	19	30	13	19
Mean (cm)	31	2	38	27	3	25	33	17	25	31	24	23
Species	**Salix sitchensis**											
Wettest Elevation From Water Surface (cm)	-97	-83	0	-98	-99	0	-97	-95	0	-9	-97	1
Driest Elevation From Water Surface (cm)	429	399	139	425	385	125	432	376	122	95	406	88
Median (cm)	0	0	15	8	0	17	15	0	13	25	6	15
Mean (cm)	18	8	22	17	3	23	24	10	19	28	20	16
Species:	**Juncus acuminatus**											
Wettest Elevation From Water Surface (cm)	-65	-82	5	-65	-79	2	-66	-79	6	-4	-82	5
Driest Elevation From Water Surface (cm)	169	156	135	166	158	125	182	153	122	88	182	88
Median (cm)	11	1	12	3	0	4	23	7	9	34	10	15
Mean (cm)	16	0	18	12	1	11	24	3	17	30	15	16
Species:	**Juncus effusus**											
Wettest Elevation From Water Surface (cm)	-68	-90	0	-64	-87	0	-60	-81	4	-9	-77	1
Driest Elevation From Water Surface (cm)	163	159	87	163	156	33	172	157	56	88	179	41
Median (cm)	0	-25	15	0	0	10	13	-3	10	21	5	15
Mean (cm)	1	-16	23	-5	-8	11	11	-4	14	26	10	15
Species:	**Carex exsiccata**											
Wettest Elevation From Water Surface (cm)	-23	-90	5	-31	-57	2	-16	-26	4	2	-21	17
Driest Elevation From Water Surface (cm)	106	102	77	75	65	27	117	103	32	87	111	27
Median (cm)	-11	-27	15	0	0	27	19	1	7	16	5	17
Mean (cm)	9	-16	31	0	-6	19	28	11	12	28	20	20

Table 10-1 (continued). Comparison of different species and their hydrologic characteristics.

Measure Note: Negative numbers are depth under water. Positive numbers are distance from water surface.	Dormant Instant Depth	Dormant Max Depth	Dormant WLF	Early Growing Instant Depth	Early Growing Max Depth	Early Growing WLF	Inter-mediate Growing Instant Depth	Inter-mediate Growing Max Depth	Inter-mediate Growing WLF	Senes-cence Instant Depth	Senes-cence Max Depth	Senes-cence WLF
Species:	**Carex obnupta**											
Wettest Elevation From Water Surface (cm)	-89	-95	4	-93	-94	0	-81	-92	2	0	-85	4
Driest Elevation From Water Surface (cm)	362	320	87	367	194	50	367	340	59	95	334	53
Median (cm)	26	0	17	20	0	11	31	10	19	25	20	19
Mean (cm)	43	19	26	40	11	16	51	31	22	35	39	22
Species	**Scirpus microcarpus**											
Wettest Elevation From Water Surface (cm)	-62	-93	8	-53	-70	5	-49	-68	4	-10	-70	4
Driest Elevation From Water Surface (cm)	175	162	69	173	162	50	188	162	57	90	173	53
Median (cm)	29	0	17	31	0	16	34	4	20	50	9	16
Mean (cm)	29	10	22	27	9	21	41	15	20	47	29	20
Species	Scirpus atrocinctus											
Wettest Elevation From Water Surface (cm)	-56	-90	5	-61	-78	2	-60	-81	6	-9	-77	5
Driest Elevation From Water Surface (cm)	187	173	77	181	176	34	202	182	37	88	199	41
Median (cm)	7	2	12	0	0	5	22	5	9	24	13	15
Mean (cm)	27	16	16	15	7	8	38	22	13	26	40	15
Species	**Phalaris arundinaceae**											
Wettest Elevation From Water Surface (cm)	-97	-93	0	-98	-99	0	-95	-88	0	-9	-97	1
Driest Elevation From Water Surface (cm)	430	419	142	438	421	126	441	395	138	99	424	111
Median (cm)	32	0	31	35	0	19	46	21	20	43	34	15
Mean (cm)	51	17	40	48	17	28	60	36	27	41	46	22
Species	**Typha latifolia**											
Wettest Elevation From Water Surface (cm)	-95	-98	0	-98	-98	0	-97	-96	4	-9	-97	1
Driest Elevation From Water Surface (cm)	308	283	50	259	185	40	157	146	57	86	175	41
Median (cm)	-20	-29	15	-21	0	10	-13	-23	12	17	0	12
Mean (cm)	-9	-17	17	-11	-13	11	-3	-14	13	22	3	15

DISCUSSION

Preliminary evaluations of the hydrologic characteristic of some common wetland species have shown that water level fluctuation and depth of inundation during the year, but especially the early growing season, can be key factors in the development of plant associations. The distribution of individual species or vegetation community are related to the hydrologic profile. If the hydrologic profile changes, such as through upstream controls, outlet controls or changing land use in the watershed, it is likely the plant community will shift towards the conditions produced by the new hydrograph. This can have both beneficial or negative consequences depending on the conditions created by management of the upstream watershed. For example, many common dominating species were found in a wider range of conditions of drought to inundation and water level fluctuation. In contrast, less common and less dominant species were almost always found in narrow ranges of hydrologic conditions.

Similarly, hydrologic profiles can help to determine the appropriate plantings for wetland design. The design and successful establishment of plant communities depends on whether individual species in those communities can flourish in the hydrologic regime. Planting designs should be developed based on the seasonal hydrologic conditions.

While our data is useful as a guideline, it is limited in its application. We did not measure plant responses or vigor with respect to hydrologic conditions. Ewing (1996), who measured and analyzed actual tree responses, observed that A. rubra was stressed by repeated cycles of inundation while Fraxinus shows no significant response to repeated inundation, provided the duration of the flooding was less than two to three days. Our methods did not measure such detailed impacts. We also did not accurately account for soil saturation and did not consider soil type which additionally effects species distribution.

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CHAPTER 11 EMERGENT MACROINVERTEBRATE COMMUNITIES IN RELATION WATERSHED DEVELOPMENT

by Klaus O. Richter, Kenneth A. Ludwa and Robert W. Wisseman

INTRODUCTION

Aquatic invertebrates play important roles in the food chain of fresh water wetlands. They are the pivotal link between the primary production and detrital trophic levels and higher level consumers including fish, amphibians and aquatic avifauna and mammals (Cummins and Merritt 1996). Moreover, aquatic macroinvertebrates have historically been used as biological indicators in riverine and lacustrine environments (Rosenberg and Resh 1996). Studies by scientists with the Puget Sound Wetlands and Stormwater Management Research Program (Ludwa 1994), (Azous 1991) and others (Murkin and Batt 1987), (Rosenberg and Danks 1987), (Wrubleski 1987), (Hicks 1995, Hicks 1996) have demonstrated the utility of macroinvertebrates as indicators of the health of palustrine environments, particularly for assessing the impacts of urbanization.

Macroinvertebrate communities are noted for their response to the four major wetland stresses identified by EPA's Environmental Monitoring and Assessment Program (EMAP): (1) altered hydroperiod, (2) excess sediment, (3) changes in nutrient cycling; and (4) contaminants (Liebowitz and Brown 1990). Unlike other wetland animal communities (amphibians, mammals, birds, fish) the larval forms of aquatic macroinvertebrates are completely confined to the water within a particular wetland, over entire growing seasons or years, until emergence. Therefore, the aquatic macroinvertebrate community is an excellent integrator of wetland impacts; it does not register impacts that may occur to other wetland animals that migrate outside the wetland for periods of time.

The goal of this study was to establish the impacts of watershed development and particularly, urban stormwater inputs on macroinvertebrate communities. Specific objectives included (1) developing a preliminary wetland macroinvertebrate community-based biotic index based on methods proven for streams, and (2) applying this index to examine the impacts of watershed urbanization on specific aspects of macroinvertebrate communities, over a range of watersheds with different levels of existing development, and within developing watersheds over time. The latter objective is based upon several hypotheses regarding the response of the aquatic macroinvertebrate community to anthropogenic changes to wetlands and their watersheds. These included (1) Changes in macroinvertebrate taxa richness and numbers of individual organisms will reflect changing land use, environmental pollution, direct habitat degradation, and general system health; (2) proportions of sensitive and tolerant taxa will change with increasing watershed urbanization and wetland habitat degradation; and (3) proportions of functional taxa groups will change with alterations to a wetland’s nutrient cycle.

Although aquatic macroinvertebrates include non-insect taxa, the sampling device used in this study collected only adult aquatic insects. Therefore, the terms macroinvertebrates and insects shall be used interchangeably in this paper.

METHODS

We periodically monitored emergent aquatic macroinvertebrates in nineteen palustrine wetlands in the Puget Sound Basin from 1988 to 1995. These wetlands were located in watersheds in various stages of urban and suburban development and have been described in earlier chapters.

Trapping protocols are extensively described in Chapter 4 and are briefly summarized here. We made an attempt to place traps in conditions as similar as possible between wetlands (open still water, fine sediment). Location of traps was particularly important because the presence or absence of certain vegetation or substrate types can substantially influence the character of the aquatic macroinvertebrate community. We deployed the traps in each wetland over the periods listed in Table 1. Field staff collected the trap contents and replaced the preservative on an approximately monthly basis from April to September during each monitoring period, with a season-end collection also made in October and/or November. We made no collections from December through March because of low invertebrate activity during this period. The traps provided a cumulative measure of insect emergence between each occasion that the traps are emptied.

Table 11-1. Approximate aquatic invertebrate emergence trap sampling periods for growing seasons 1989, 1993, and 1995.

	1989	1993	1995
Start collection	September 1, 1988*	April 10, 1993	January 1, 1995
End collection	September 31, 1989	April 9, 1994	October 30, 1995

* Monitoring at Fourteen sites were started in September 1988; five more sites were added in April 1989.

We identified and enumerated the macroinvertebrates collected in 1989 to the lowest level possible, in most cases genus or species. We identified insects collected in 1993 and 1995 only to family for Dipteran taxa, and to order for all other taxa. We made identifications to a consistent level within each taxonomic group for all samples.

Using the 1989 data set, we developed a multimetric biological index based on principles of the Benthic Index of Biotic Integrity (Fore et al. 1995). We proceeded by first testing metrics to determine whether they differentiated between the two best and two worst sites; we then confirmed these metrics by testing them over the whole range of nineteen sites (Ludwa 1994) (Fore et al. 1995). We tested and adapted existing lotic macroinvertebrate community metrics to the wetland insect community, and tested and added new metrics unique to palustrine communities.

Because the level of taxonomic effort was considerably coarser for the 1993 and 1995 collections, we found it necessary to develop and test a new set of metrics suitable for that level of information. We performed this step with the 1989 collections by elevating the taxonomic data to the same levels as the 1993 and 1995 collections. Again, we followed the same procedures described by (Ludwa 1994). Most of the coarser-level metrics were based on, (Ludwa 1994) original metrics for the 1989 collections.

We tested the overall index scores against land use and wetland morphology thresholds reported by (Taylor et al. 1995) and Ludwa (Ludwa 1994) using the Mann-Whitney test (Zar 1984), the nonparametric equivalent of the independent groups t-test. We also tested index scores against parameters for wetland hydrology and water quality, and separately against wetland morphology and watershed land use, using multiple regressions (Zar 1984). All statistical analyses were performed at a significance level of $p > 0.05$.

RESULTS

It is important to note that we designed and calculated the 1989 species/genus-level metrics using data spilt into distinct sampling periods: April-June, July-September, and October-November (Ludwa 1994). The data split into these periods, especially the two summer periods, responded more strongly to urbanization parameters than did the year-long data set. We designed and calculated the 1989 order/family-level metrics using the year-round data sets. Taxa richness values for the coarser-level data were too low for individual sampling periods to differentiate between sites. We assumed that the difference between the length of sampling periods between the three years (Table 1) did not significantly affect taxa richness values, but that it did affect total numbers of individuals collected. The metrics developed for the order/family-level data were taxa richness- and proportion-oriented; therefore we assumed that different sampling period lengths did not affect metric design or calculation.

The metrics recommended for further testing by (Ludwa 1994) for emergent collections with genus-species level taxonomy are listed in Table 2. Although taxa belonging to orders Ephemeroptera, Plecoptera, and Trichoptera are often the basis of stream biological metrics, we found a paucity of these taxa in the wetland insect collections (including order Odonata, these orders are referred to as EPOT). Therefore, although EPOT richness and abundance did yield two metrics, most of the metrics (numbers 7 through 22, including all new wetland- oriented metrics) related to order family Chironomidae of order Diptera (aquatic midges and true flies). Chironomids are a highly diverse family only sparsely detailed in ecological literature; although generally considered to be negative indicators for running waters, Chironomids are adapted to lentic environments, and therefore may be more appropriate indicators of their health.

Using an index composed of the metrics listed in Table 2, (Ludwa 1994) calculated index scores and compared them to direct and indirect measures of wetland stress. Ludwa (1994) emphasized that further verification of this index and its component metrics is necessary before it can be used as an independent measure of wetland ecological health. Conclusions drawn from (Ludwa 1994) analyses follow.

Table 11-2. Biotic index metrics recommended for use with wetlands, based on emergent macroinvertebrate collections with genus/species-level identification (Ludwa 1994).

Metrics Included in Final Wetland Biotic Index (Genus/Species-level Taxonomy)	
Adapted from stream metrics:	Unique Wetland Metrics:
1. Taxa richness	9. Percent individuals as Chironomini tribe
2. Scraper and/or piercer taxa presence	10. Chironomini tribe taxa richness
3. Shredder taxa presence	11. Percent individuals as Tanypodinae subfamily
4. Collector taxa richness	12. Tanypodinae subfamily taxa richness
5. EPOT[1] taxa richness	13. Presence *Thienemanniella*
6. Percent individuals as EPOT	14. Presence *Endochironomus nigricans*
7. Percent individuals as tanytarsini tribe	15. Presence *Parachironomus* spp. 2
8. Tanytarsini tribe richness	16. Presence *Polypedilum* gr.1 and 2
	17. Presence A*blabesmyia*
	18. Presence *Aspectrotanypus algens*
	19. Presence *Paramerina smithae*
	20. Presence *Psectrotanypus dyari*
	21. Presence *Zavrelimyia thryptica*
	22. Presence *Tanytarsus*

[1]EPOT = Ephemeroptera, Plecoptera, Odonata, and Trichoptera.

There appeared to be two primary periods of insect emergence, in the early summer and again in the late summer/early autumn; sampling periods in April-June and July-September were most appropriate for calculation of biotic index scores. Collections made in October-November did not appear to be as effective for purposes of bioassessment.

Biotic index scores responded significantly to land use and wetland morphology parameters. A multiple regression revealed that scores responded negatively to total watershed impervious area, wetland channelization, and incidence of dryness. The regression explained 67 percent of the variance in index scores. Threshold analyses also revealed that index scores were significantly higher with increasing watershed forest coverage and lower with increasing impervious area. Highly channelized sites had significantly lower scores, consistent with the observation of degraded water quality for most parameters in highly channelized sites.

A multiple regression indicated that water quality and hydrology parameters explained a significant amount of variation of the index scores (as high as 73 percent). Index scores responded negatively to hydrogen ion concentration (antilog pH), conductivity, suspended solids, water level fluctuation, and incidence of wetland dryness. Suspended solids, conductivity, and water level fluctuation were demonstrated by (Ludwa 1994), (Taylor et al. 1995), and (Chin 1996) to be the water quality and hydrology parameters in these sites most significantly degraded by increases in watershed impervious area and decreases in forest cover. This illustrates the interrelationship between a wetland's watershed, its physical and chemical parameters, and the health of its biological communities.

The order/family-level metrics developed with the 1989 data are listed in Table 3; Table 4 lists the resulting index scores calculated with these metrics for 1988, 1993, and 1995. Although the order/family-level metrics responded to indicators of urbanization, the overall index comprised of the metrics had much less power to discern between sites with different levels of urban impact. For example, the multiple regression of 1989 genus/species index scores versus total impervious area, wetland channelization, and incidence of dryness explained 67 percent of the index score variance. The same regression explained only 21 percent of the 1989 index score variance for the order/family data.

Table 11-3: Biotic index metrics recommended for use with wetlands, based on emergent macroinvertebrate collections with genus/species-level identification.

Metrics Included in Final Wetland Biotic Index (Order/Family-level Taxonomy)
• Family/Order Richness • Shredder Presence • Collector Richness • EPOT Order Richness • % Individuals as EPOT • % Individuals as Dixidae

After 1989, the next year in which land use data was available was 1995. The 1995 index scores were not significantly related to total impervious area or forested area, nor did the scores respond significantly in the multiple regression against total watershed impervious area, wetland channelization, and incidence of wetland dryness. Furthermore, the changes in index scores between 1989 and 1995 did not correspond to changes in land use. For example, NFIC12, which experienced an increase in impervious area from 2 percent to 40 percent, showed the highest percent increase in its index score, exactly opposite that which would be predicted (Figure 11-1).

Table 11-4. Order/Family macroinvertebrate index scores.

	Index Score		
	1989	1993	1995
AL3	16	10	20
B3I	12	8	6
BB24	26	10	16
ELS39	10	12	12
ELS61	18	10	18
ELW1	8	6	6
FC1	16	14	10
HC13	22	14	24
JC28	22	10	26
LCR93	28	16	6
LPS9	8	10	18
MGR36	20	12	16
NFIC12	10	10	24
PC12	18	10	10
RR5	10	6	18
SC4	16	10	12
SC84	14	14	12
SR24	18	10	14
TC13	10	12	10

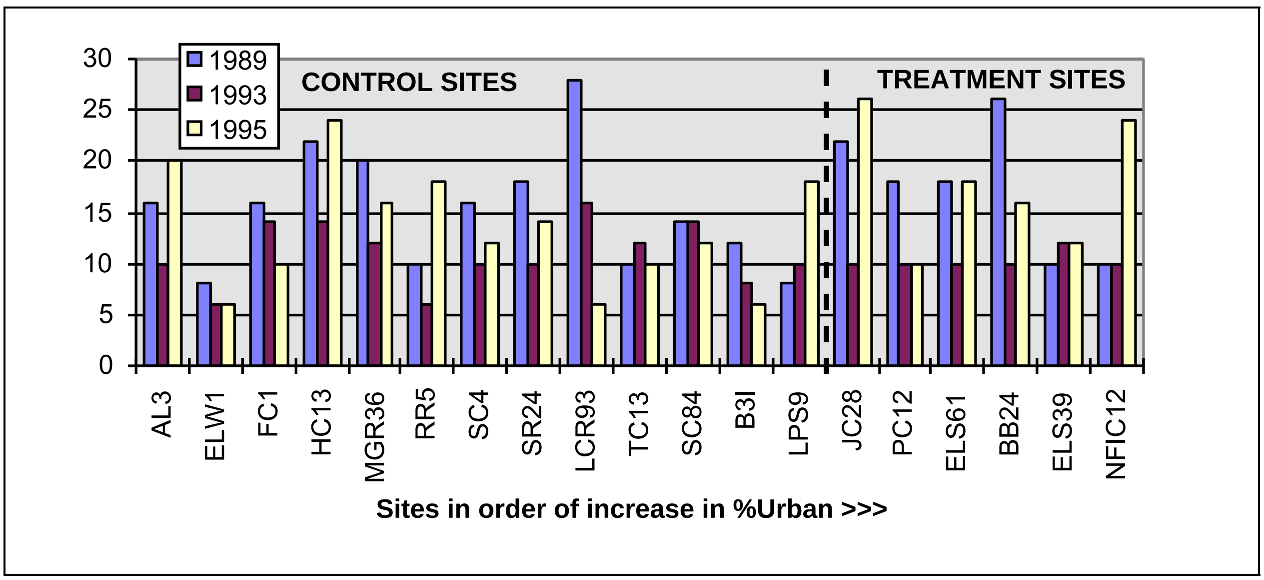

Figure 11-1. 1989, 1993, and 1995 Wetland macroinvertebrate index scores versus change in watershed urbanization.

In addition to relating index scores to changing watershed characteristics, we also examined changing taxa richness and abundance data to describe the impact of urbanization on emergent macroinvertebrates. Table 5 lists abundance and taxa richness values for each site in each year. Multiple regressions and threshold tests revealed no significant patterns in order/family taxa richness related to impervious area, between sites or years. In other wetland animal communities, taxa richness of sensitive species is often more responsive to wetland degradation than is overall taxa richness (e.g., Power at. al., 1989). The index developed for the species/genus-level data incorporates this concept by including sixteen metrics based on the presence of taxa that are assumed to be more sensitive to disturbance. The order/family data does not allow enough resolution to indicate sensitive taxa. Numbers of individuals decreased from 1989 to 1995 in 14 out of 19 sites, but, as discussed above, we assume that this is primarily a function of a longer sampling period in 1989.

Table 11-5. Insect abundance and order/family richness: 1988, 1993, and 1995.

	Abundance			Taxa Richness		
	1989	1993	1995	1989	1993	1995
AL3	4408	3619	1946	12	11	13
B3I	3027	2219	988	14	10	8
BB24	8857	14742	5815	14	10	13
ELS39	7337	6267	3773	12	12	12
ELS61	20828	13457	2808	16	10	12
ELW1	1239	503	157	10	7	7
FC1	4736	13332	5751	14	9	9
HC13	8748	4436	2934	15	11	13
JC28	1133	5778	1251	13	8	13
LCR93	9689	12148	40464	15	12	7
LPS9	5127	1006	5490	12	10	12
MGR36	7365	13276	1918	14	10	10
NFIC12	8869	24866	2015	12	11	13
PC12	5893	10701	4350	15	11	11
RR5	8621	4748	2150	12	10	11
SC4	2952	2794	2962	12	10	12
SC84	3692	2159	1254	13	9	11
SR24	5598	4982	1140	14	8	12
TC13	4657	4204	4657	13	9	13

SUMMARY

We recommend further development of macroinvertebrate community-based biological indices for assessment of wetland biological health. Our results suggest that this kind of index may be as useful as comparable indices established for running waters. Further testing of the metrics proposed by this study are necessary before the index may be used as an independent wetland assessment tool in the Puget Sound Ecoregion. Furthermore, refinement of insect tolerance and feeding group information may allow the index to be used as a diagnostic tool. Alternatively, in a set of proposed guidelines for assessing wetland health, Brooks and Hughes (1988) advocate a broad multi-taxa approach that not only includes invertebrates but plants and vertebrates as well.

We recommend genus and species-level taxonomic identification of macroinvertebrates for use of taxa richness values and calculation of biological indices. Coarser-level identifications do not appear to adequately discern insect functional groups, tolerance levels, and specific sensitive genera or species.

Results from the 1989 comparisons of insect data across wetlands with different levels of watershed development suggest that urbanization affects emergent macroinvertebrate communities by (1) decreasing overall taxa richness, (2) eliminating or reducing taxa belonging to scraper and shredder functional feeding groups (leaving a dominance of collector taxa), (3) reducing EPOT taxa richness and relative abundance, and (4) eliminating or reducing specific Dipteran taxa, particularly those belonging to the Chironomidae family.

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CHAPTER 12 BIRD COMMUNITIES IN RELATION TO WATERSHED DEVELOPMENT

by Klaus O. Richter and Amanda L. Azous

INTRODUCTION

Wetlands are recognized because of the disproportionate habitat value they provide for birds (Chapter 6 in this volume). Wetlands, however, are under increasing threat from watershed development in urbanizing areas. Landscape conversion from forests to residential housing and other developments remove or alter habitat immediately adjacent to wetlands and fragment habitat that remains. Moreover, wetlands themselves may be altered in their hydrology and water quality, directly influencing bird populations or indirectly affecting them by altering wetland vegetation. Collectively, these alterations may change breeding, nesting or feeding habitat and competitive interactions among and between species resulting in population shifts.

Striking bird population changes in terrestrial habitat within urbanizing landscapes have been documented. Blair (1996) in his review of researchers' findings of bird distributions along terrestrial gradients of urbanization, summarized that: (1) species composition changes in an area as it becomes urbanized; (2) almost always, the number of species decreases with increasing urbanization; and (3) all agree that bird density or abundance increases with urbanization. More specifically, urbanization is generally found to be correlated with increasing biomass and density and favoring dominance by a few urban ground gleaners where forest insectivores, canopy foliage gleaners or bark drillers used to forage (Beissinger and Osborne 1982).

Few studies, however, have investigated the impacts of watershed development on birds of wetlands. Birds of wetlands may directly be threatened by impacts to marshes, swamps and bogs and secondarily by habitat changes attributable to urbanization within the landscape. Foremost, wetland impacts include urban stormwater runoff that flood nest sites and disperses pollutants that may bio-acumulate in birds through aquatic food chains. Moreover, runoff may alter the areal extent of open water, existing hydrology, vegetation classes and other wetland characteristics influencing cover, nesting habitat and food distribution. Concomitantly, urbanization may influence wetland buffers and adjacent lands, which may also be reflected in changing bird distributions and abundances.

In this paper we describe the changing bird communities in wetlands across a gradient of increasing watershed development and within wetlands that have been altered during the duration of this study. We hypothesize that bird species diversity and abundance changes with increasing watershed development. Although total bird diversity may remain the same in wetlands, we predict that abundances of native species, especially urban-intolerant species, should decline and urban adapters and exploiters increase. Specifically, the proportion of species with low tolerances to habitat changes should be lower in wetlands affected by development than unaffected wetlands.

In part, these predicted changes are based on the fact that the distribution and abundance of birds are widely accepted as functions of vegetation structure and diversity which, in itself, is altered by development in watersheds. Therefore, we hypothesize that bird species richness, diversity, and relative abundance reflect the structural diversity of vegetation at wetlands, with those wetlands with greatest vegetation changes exhibiting the greatest avifaunal changes.

METHODS

Bird survey methods are described in the companion paper on bird distributions in the wetlands of the Puget Sound Basin (Chapter 6). In this chapter we compare the pre-development and post-development alpha diversities of birds for life history characteristics covering adaptability and residency. We also evaluate bird density as measured by the average number of detections per visit to a wetland. Initially, to examine adaptability, we characterized species as invasive and non-invasive by identifying invasive birds as alien species spreading naturally (without the direct assistance of people) in natural or seminatural wetlands, to produce a significant change in terms of composition, structure or ecosystem process, which was a definition applied to invasive

vegetation by Cronk and Fuller (1995). Subsequently we identified species as 1) urban exploiters, 2) urban avoiders and 3) suburban adaptable using the criteria specified by Blair (1996) and based on species sensitivity to human-induced changes in wetlands and watersheds. We also characterize birds by whether they were common residents, rare residents or seasonal migrants according to Hunn (1982).

Wetland vegetation, hydrology and surrounding land use were measured as described in Sections 1 and 2 of this report. In addition, we characterize wetlands according to watershed condition and their level of disturbance, or treatment, during the course of our study. These experimental categories included wetlands in rural areas which did not change during our study (Rural Controls), wetlands which began the study in an urbanized area (Urban Controls) and wetlands which had 10% or more of their watershed develop, regardless of previous condition, during the study period (Treatments). We also examined the availability of suitable habitats for birds adjacent to wetlands, including forests, with and without single family housing, open water and shorelines. Undeveloped meadow and shrub-land were also evaluated as additions to suitable habitats whereas unsuitable habitat always included developed or cleared land and agricultural lands.

Statistical analysis of correlations and hypothesis testing utilized parametric statistics when assumptions of normality were met and non-parametric statistics when assumptions were violated. We chose $p \leq 0.05$ and $p \leq 0.10$ as significant and weakly significant, respectively, for reporting results. Nevertheless, significance should be interpreted cautiously because of the variability in sampling populations of species and the low number of wetlands undergoing impacts that could be observed in changing bird sightings during the period of our study.

RESULTS

Total alpha diversity decreased significantly among all wetlands between 1989 and 1995 (Friedman test (F), $\chi2 = 18.3$, $p \leq 0.0001$). Total alpha diversity also decreased among all wetlands when analyzed by experimental category. Both wetlands in developed (urban controls) and undeveloped (rural controls) watersheds showed a significant decline in total diversity (F, $\chi2 = 5.6$, $p = 0.06$ and F, $\chi2 \geq 4.8$, $p = 0.09$, respectively), as did wetlands in watersheds with increased development (treatments) during the study (F, $\chi 2 = 9.0$, $p = 0.01$).

Total diversity in a single wetland ranged from 16 to 57 species over the study period and averaged 38 among all wetlands in 1989, the year of highest recorded richness. During that same year, we observed an average of 37 bird species in both the urban control and rural control wetlands and an average of 38 in the treatment wetlands. By the last year of our surveys, 1995, total diversity within wetlands with undeveloped uplands averaged 31. In the treatment wetlands and in the urban control wetlands, an average of 28 species were detected.

Average alpha diversity, similar to total diversity decreased significantly for all wetlands (F, $\chi 2 = 13$, $p = 0.0015$). However, average alpha diversity only decreased significantly among the wetlands with watersheds affected by urbanization whether past (urban controls) (F, $\chi 2 = 7.0$, $p = 0.03$) or during the study period (treatments) (F, $\chi 2 = 5.5$, $p = 0.06$). Average diversity for all wetlands in undeveloped watersheds at the end of our study (controls) remained unchanged (F, $\chi 2 = 3.1$, $p = 0.2$) (Figure 12-1).

The average number of birds detected at all 19 wetlands slightly increased, from 1989 to 1995 (F, $\chi 2 \geq 4.8$, $p = 0.09$), but simultaneously, we found average detections unchanged among all experimental categories, the urban controls (F, $\chi 2 \geq 2.0$, $p = 0.37$), the treatment wetlands (F, $\chi 2 \geq .33$, $p = 0.84$) and among the rural control wetlands (F, $\chi 2 \geq 3.2$, $p = 0.2$) (Figure 12-2). A complete list of detection rates for all species is available in Appendix Table 12-1.

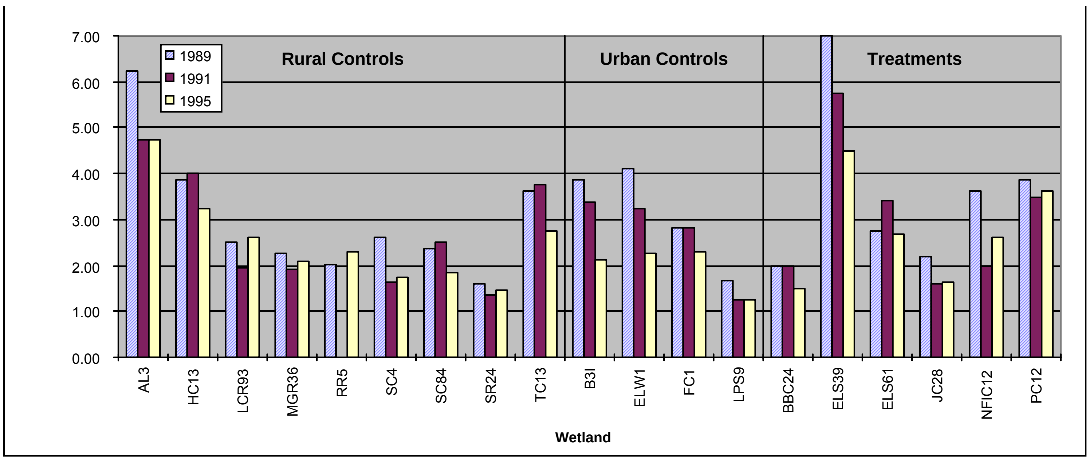

Figure 12-1. Average wetland alpha diversity over the study period by experimental category.

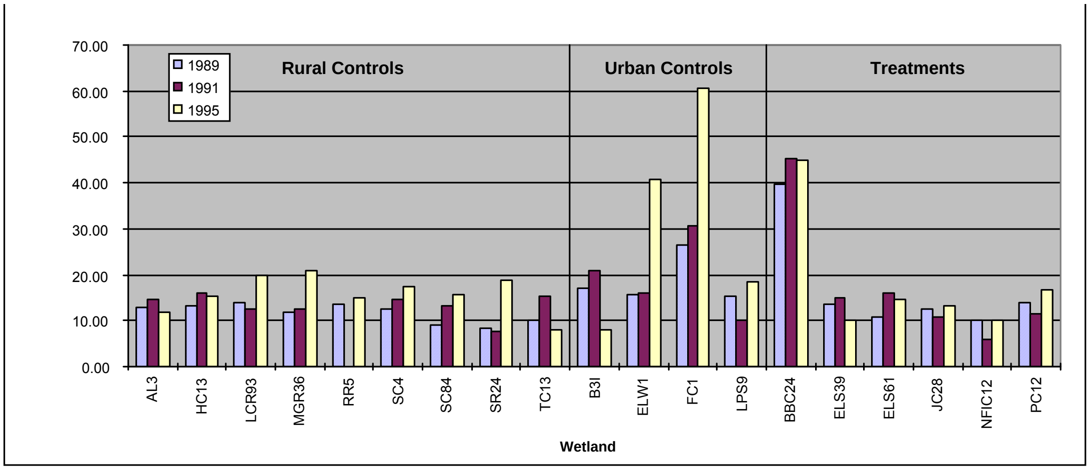

Figure 12-2. Average avian detection rate over the study period by wetland and experimental category.

We found that bird richness decreased and abundance remained the same in wetlands with developed or developing watersheds (urban control or treatment) but found richness unchanged in wetlands with rural, relatively pristine watersheds (rural controls).

Interestingly, although alpha bird diversity was statistically related to development in the watershed, we did not find diversity to be related to urbanization within 1000 meters of the wetlands. Although, increasing percentages of forest land within 1000 meters of the wetland did not add to diversity, the presence of forest land did affect the structure of bird communities from about 500 meters to 1000 meters (the maximum distance we studied). We found that species richness of birds known to avoid human development (avoiders) increased over the study period primarily in wetlands with high percentages of adjacent forest land within 500 meters (Mann-Whitney (MN), $p < 0.09$) whereas they decreased among the already urban wetlands and in those where land use changes decreased watershed habitat (Figure 12-3).

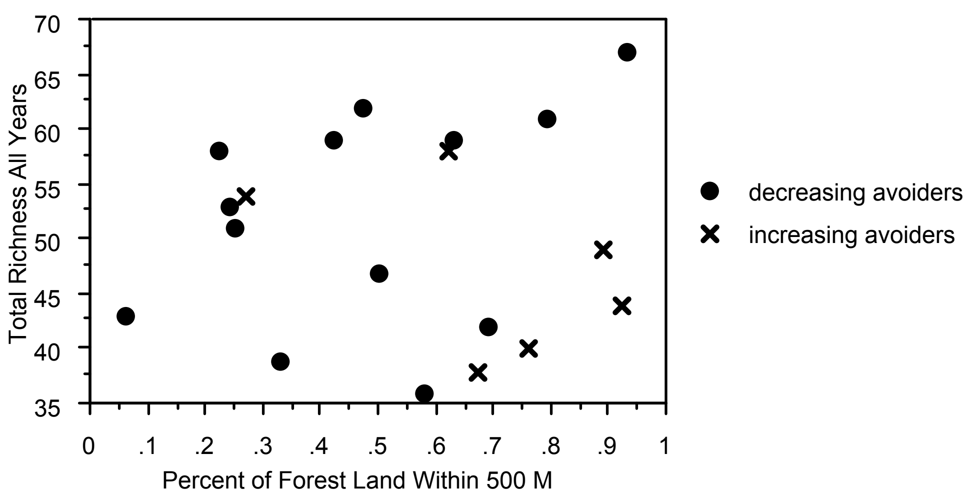

Figure 12-3. Species richness and whether the number of avoiders in the population increased or decreased related to the presence of forest land.

Detections of migrants declined during the study among all wetlands combined (F, $\chi2$ = 31.6 $p \leq 0.0001$) as did rare residents (F, $\chi2$ = 6.4, p = 0.04) while detections of residents remained the same. Migrants also declined within all experimental categories (F, $\chi2 \geq 7.1$ $p \leq 0.02$) but detections of rare residents did not show any significant change within the experimental groups. Detections of resident species did not change among the rural control and treatment wetlands but declined in the urban control wetlands (F, $\chi2$ = 5.1, p = 0.07).

Across all wetlands, the number of detections of species that avoid development and adaptive species declined during the study (F, $\chi2 \geq 10.1$, $p \geq 0.007$) while densities of invasive or exploitive species stayed the same. Detections of avoiding species declined among the already urban and treatment wetlands but not the rural control wetlands (F, $\chi2 \geq 9.1$, $p \leq 0.01$). The greatest declines of adaptive species occurred in treatment wetlands (F, $\chi2 \geq 7.5$, $p \leq 0.02$). While exploitive species detections were not significantly different between years in wetlands overall, among the rural control wetlands in non-urbanized areas, densities of exploitive species increased significantly (F, $\chi2$ = 5.6, p = 0.06) from 1989 to 1995. Density changes included increases in such invasive species as American crow, European starling and house sparrow.

Three wetlands, ELS39, ELS61 and NFIC12 exhibited dramatic vegetation changes during our study and also showed significant changes in bird species. At ELS39 species richness decreased from 28 to 23 and then to 18, from 1989, 1991 and 1995, respectively. Species disappearing included marsh wren, pine siskin and red-breasted nuthatch. Species increasing included, among others, urban habitat exploiters and adapters such as American crow, mallard, California quail, and rufous-sided towhee. At ELS61 species richness decreased from 44 to 32 species between 1989 and 1995 and at NFIC12 species decreased from 29 to 21. Within both wetlands sightings of American robin and black-capped chickadees increased.

DISCUSSION

Although our study intensively covers the wetlands of the lower Puget Sound region and represents a first comprehensive account of wetland bird diversity, we consider our work to date as a rough initial attempt to assess bird densities and population trends over the study period. Blair (1996) found that urbanization affects bird diversity in two distinct ways: moderate levels of development may both increase overall species diversity and decrease native bird diversity whereas increasingly severe development lowers total and native species diversity. Although moderate development increases diversity this increase seems attributable to the addition of widely distributed species at the expense of native species. Our findings agree with Blair in that, in general, we found average alpha diversity decreasing in wetlands in watersheds affected by urbanization but also in some wetlands not affected by urbanization. In addition, we found that abundance of birds (detection rate) increased among all the wetlands, yet remained unchanged in all experimental categories in undeveloped areas but decreased in those wetlands where development occurred or pre-existed. Moreover, detection of many native species that avoid urbanization decreased in all but rural wetlands in which development did not occur.

Decreasing diversity and increasing numbers in response to isolation were observed by Brown and Dinsmore (1986) who found that wetland size and isolation account for 75%

of the variation in species richness observed within prairie marshes. They also found that species richness was often greater in wetland complexes than in simple larger isolated marshes. Although, we found that the presence of forest within 0 to 500 meters was not correlated to avian richness or overall abundance, forests within the entire watershed did suggesting that wooded areas near but not adjacent to wetlands are important. We also found that wetlands with significant forest land remaining within 500 to 1000 meters, did account for increasing numbers of species that avoid urbanization, even though adaptable and exploitive species generally declined during the same period.

For the most part we found the wetland avifauna to be an extension of the upland avifauna. As expected, in wetlands of undisturbed landscapes (such as SR24 and RR5) species diversity is dominated by residents and migrants whereas wetlands in more urban areas (such as B3I and FC1) bird diversity is characterized by increasing numbers of non-native species including American crow, European starlings, house sparrows and some brown-headed cowbirds. We have seen European starlings displace cavity nesters including swallows and chickadees. Moreover, we have seen American Crows raid passerine nests. The shift of bird communities from predominantly native species in undisturbed areas to invasive species in highly developed areas is well documented in terrestrial environments (Blair 1996) and we saw similar shifts among some, but not all, wetlands within this study. Nevertheless, observations must be cautiously interpreted as recent literature suggests that determining bird diversity and abundance is extremely difficult (James et al. 1996, Thomas and Martin 1996), and furthermore, may be driven by immigration from few large regional source sites that produce surpluses (Brawn and Robinson 1996) rather than by more local conditions.

Based on these results, we predict that the distribution and abundance of species will change more dramatically as urbanization continues and becomes more severe. Specifically, we would expect decreasing diversity and abundances of migrants and residents and increasing nest predators including urban exploiters like the American crow and European starling as well as and nest parasites such as brown-headed cowbird. Other factors contributing to declines in birds that avoid urbanization are the density of predators like domestic cats and introduced rodents such as Norway rats and brown rats. We especially expect significant reductions in ground nesting species as increasing numbers of predators are introduced with human development.

Many wetlands in our study still exhibit a wide variety of vegetation structure and microhabitats that enable a rich diversity of birds to be found. However, with increasing urbanization and habitat fragmentation that separates wetlands from larger upland habitats and wetlands from each other, diversity of native species may be expected to decrease (as for example in urban areas, Milligan 1985). To avoid these effects, we recommend that forest land with complex structure be retained to the greatest extent possible in areas adjacent to wetlands. Dense stands of herbs and shrubs should also be retained to provide cover to birds and restrict the movement of avian predators. Access via roads, trails and footpaths that enable disturbance by humans and use by pets should be limited and edge habitat minimized as edge-related problems of thermo-regulation, predation and nest-parasitism increases along edges.

Our data supports the increasingly accepted view that total species richness is not an adequate measure of community condition under threat because the increasing diversity, attributable to urban exploiters and urban adaptable species, is in fact an indication of wetland functional deterioration. To maintain regional biodiversity, it is critical to differentiate between native species with distinct habitat preferences and invasive species and adaptable species associated with urbanization, and to maintain habitat for native, specialized species rather than the increasingly common adptable birds. Finally, wetlands must be viewed as dynamic ecosystems which must be managed for diversity over the entire landscape and not just as individual isolated habitats.

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Appendix Table 12-1. Abundance and detection rates of species over all wetlands.

	Abundance				Detection Rate			
Species	1989	1991	1995	All Years	1989	1991	1995	All Years
American Coot	4	22	9	35	0.014	0.087	0.034	0.045
American Crow	117	160	287	564	0.418	0.635	1.087	0.727
American Goldfinch	99	76	67	242	0.354	0.302	0.254	0.312
American Robin	294	239	322	855	1.050	0.948	1.220	1.102
Anna's Hummingbird	2		1	3	0.007	0.000	0.004	0.004
Bald Eagle		1	3	4	0.000	0.004	0.011	0.005
Barn Swallow	19	18	64	101	0.068	0.071	0.242	0.130
Black-capped Chickadee	213	194	245	652	0.761	0.770	0.928	0.840
Belted Kingfisher	7	4	10	21	0.025	0.016	0.038	0.027
Bewick's Wren	49	42	68	159	0.175	0.167	0.258	0.205
Brown-headed Cow Bird	23	16	39	78	0.082	0.063	0.148	0.101
Black Headed Grosbeak	57	38	64	159	0.204	0.151	0.242	0.205
Brewer's Blackbird	10	15	127	152	0.036	0.060	0.481	0.196
Brown Creeper	9	8	5	22	0.032	0.032	0.019	0.028
Black-throated Gray Warbler	25	13	44	82	0.089	0.052	0.167	0.106
Band-tailed Pigeon	4	2	4	10	0.014	0.008	0.015	0.013
Bushtit	126	88	141	355	0.450	0.349	0.534	0.457
Blue-winged Teal			2	2	0.000	0.000	0.008	0.003
Canada Goose	6	4	259	269	0.021	0.016	0.981	0.347
California Quail	1		3	4	0.004	0.000	0.011	0.005
Caspian Tern			13	13	0.000	0.000	0.049	0.017
Chestnut-backed Chickadee	63	77	74	214	0.225	0.306	0.280	0.276
Cedar Waxwing	111	74	110	295	0.396	0.294	0.417	0.380
Chipping Sparrow		1	2	3	0.000	0.004	0.008	0.004
Cliff Swallow	18	9	4	31	0.064	0.036	0.015	0.040
Cooper's Hawk	2		7	9	0.007	0.000	0.027	0.012
Common Raven			5	5	0.000	0.000	0.019	0.006
Common Yellow-throat	95	63	69	227	0.339	0.250	0.261	0.293
Dark-eyed Junco	40	17	32	89	0.143	0.067	0.121	0.115
Downy Woodpecker	16	14	28	58	0.057	0.056	0.106	0.075
European Starling	122	180	445	747	0.436	0.714	1.686	0.963
Evening Grosbeak	23	1	23	47	0.082	0.004	0.087	0.061
Fox Sparrow	1		5	6	0.004	0.000	0.019	0.008
Gadwall	5	4	4	13	0.018	0.016	0.015	0.017
Great Blue Heron	18	9	25	52	0.064	0.036	0.095	0.067
Golden-crowned kinglet	96	73	19	188	0.343	0.290	0.072	0.242
Green Heron	12	1	1	14	0.043	0.004	0.004	0.018
Glaucous Winged Gull	3	1	2	6	0.011	0.004	0.008	0.008
Hammond's Flycatcher	9	10	2	21	0.032	0.040	0.008	0.027
Hairy Woodpecker	40	17	13	70	0.143	0.067	0.049	0.090
Hermit Thrush	85	11	8	104	0.304	0.044	0.030	0.134
House Finch	23	8	16	47	0.082	0.032	0.061	0.061
Hooded Merganser	14		9	23	0.050	0.000	0.034	0.030
House Sparrow	9	5	2	16	0.032	0.020	0.008	0.021
Hutton's Vireo	21	1	3	25	0.075	0.004	0.011	0.032
Killdeer	6		4	10	0.021	0.000	0.015	0.013
Mallard	44	50	223	317	0.157	0.198	0.845	0.409
Marsh Wren	56	23	24	103	0.200	0.091	0.091	0.133
MacGillivary's Warbler	2		6	8	0.007	0.000	0.023	0.010
Northern Flicker	10	12	24	46	0.036	0.048	0.091	0.059
Northern Oriole	4		2	6	0.014	0.000	0.008	0.008

Appendix Table 12-1 continued. Abundance and detection rates of species over all wetlands.

	Abundance				Detection Rate			
Species	1989	1991	1995	All Years	1989	1991	1995	All Years
Northern Pigmy Owl		1	2	3	0.000	0.004	0.008	0.004
Orange-crowned Warbler	38	23	12	73	0.136	0.091	0.045	0.094
Olive-sided Flycatcher	5	8	2	15	0.018	0.032	0.008	0.019
Pied-billed Grebe	8	2	20	30	0.029	0.008	0.076	0.039
Pine Siskin	14		18	32	0.050	0.000	0.068	0.041
Pileated Woodpecker	13		4	17	0.046	0.000	0.015	0.022
Pacific-slope Flycatcher	127	147	145	419	0.454	0.583	0.549	0.540
Purple Finch	24	22	40	86	0.086	0.087	0.152	0.111
Red-breasted Nuthatch	15	29	42	86	0.054	0.115	0.159	0.111
Red-breasted Sapsucker	4		4	8	0.014	0.000	0.015	0.010
Red Crossbill	9	42	4	55	0.032	0.167	0.015	0.071
Red-eyed Vireo	2		9	11	0.007	0.000	0.034	0.014
Red-eyed Vireo	2	1	5	8	0.007	0.004	0.019	0.010
Rock Dove	5	4		9	0.018	0.016	0.000	0.012
Rufous-sided Towee	101	98	143	342	0.361	0.389	0.542	0.441
Rufous Hummingbird	6	5	4	15	0.021	0.020	0.015	0.019
Ruffed Grouse	1	2	2	5	0.004	0.008	0.008	0.006
Ruby Crowned Kinglet	21	10	20	51	0.075	0.040	0.076	0.066
Red-winged Blackbird	353	203	228	784	1.261	0.806	0.864	1.010
Savannah Sparrow		2		2	0.000	0.008	0.000	0.003
Sora		2	3	5	0.000	0.008	0.011	0.006
Song Sparrow	476	395	419	1290	1.700	1.567	1.587	1.662
Solitary Vireo	5	13	4	22	0.018	0.052	0.015	0.028
Spotted Sandpiper	3			3	0.011	0.000	0.000	0.004
Sharp-shinned Hawk	4			4	0.014	0.000	0.000	0.005
Steller's Jay	33	67	89	189	0.118	0.266	0.337	0.244
Swainson's Thrush	154	181	344	679	0.550	0.718	1.303	0.875
Townsend's Warbler	38	2	13	53	0.136	0.008	0.049	0.068
Tree Swallow	101	63	67	231	0.361	0.250	0.254	0.298
Varied Thrush	41			41	0.146	0.000	0.000	0.053
Vaux's Swift	18	13	8	39	0.064	0.052	0.030	0.050
Violet-green Swallow	56	68	151	275	0.200	0.270	0.572	0.354
Virginia Rail	9	3	6	18	0.032	0.012	0.023	0.023
Warbling Vireo	38	3	22	63	0.136	0.012	0.083	0.081
White-crowned Sparrow	14	9	1	24	0.050	0.036	0.004	0.031
Western Tanager	17	9	29	55	0.061	0.036	0.110	0.071
Western Wood-pewee	11	6	13	30	0.039	0.024	0.049	0.039
Willow Flycatcher	116	90	142	348	0.414	0.357	0.538	0.448
Wilson's Warbler	115	72	78	265	0.411	0.286	0.295	0.341
Winter Wren	109	85	115	309	0.389	0.337	0.436	0.398
Wood Duck	10	4	9	23	0.036	0.016	0.034	0.030
Yellow Warbler	67	50	26	143	0.239	0.198	0.098	0.184
Yellow-rumped Warbler	7	3	4	14	0.025	0.012	0.015	0.018
Totals	4203	3338	5215	12756	15.011	13.246	19.754	16.438

Section 4 Management of Freshwater Wetlands in the Central Puget Sound Basin

CHAPTER 13 MANAGING WETLAND HYDROPERIOD: ISSUES AND CONCERNS

by Amanda L. Azous, Lorin E. Reinelt and Jeff Burkey

INTRODUCTION

Land use changes and stormwater management practices usually alter hydrology within a watershed. A major finding of our study was that hydrologic changes were having more immediate and greater effects on the composition of vegetation and amphibian communities than other environmental conditions we monitored. Early study results showed wetland hydroperiod, which refers to the depth, duration, frequency and pattern of wetland inundation to be a key factor in determining biological responses.

Continuous recording gages were unavailable for the study, but we were able to monitor hydroperiod in the wetlands with instantaneous staff and crest stage gages. From these measurements a metric was developed called water level fluctuation (WLF) which showed statistically significant relationships with several measures of biological health (Azous 1991a). WLF is measured as the average difference between the maximum depth and average instantaneous or base depth in a time period (Taylor 1993, Taylor, Ludwa and Horner 1995).

Consistently we observed reduced numbers of plant and amphibian species when WLF was high in wetland areas (Azous 1991b, Cooke and Azous 1993, Richter and Azous 1995). As a result, substantial attention was given to understanding WLF and developing management guidelines for protecting wetland plants and animals.

A local jurisdiction, King County Surface Water Management (KCSWM) expressed an interest in developing wetland management guidelines that could be used in continuous flow event simulation computer models. In addition, only a few of the wetlands in the original 19 study wetlands showed extreme water level changes and we wanted to measure more plant and amphibian communities with high WLF conditions. We undertook a cooperative study to monitor the hydroperiods of six wetlands with continuous recording gages, and measure the plant and amphibian communities, in order to better understand the relationship between biological diversity, WLF, and the pattern of water depth, duration and frequency of inundation in wetlands.

This paper will discuss the methods and results of this study. The information has significant implications for evaluating the level of protection afforded wetlands from changing hydroperiod.

METHODS

Continuous recording gages were installed in six wetlands in late 1994 and early 1995. The gages were programmed to record water surface elevations at 15-minute increments. Two of the wetlands we monitored were in relatively undisturbed watersheds and were already experimental controls in our ongoing study. The remaining four were recommended by KCSWM field staff as wetlands known to experience large changes in water depth throughout the year.

Water levels in all six wetlands were monitored over one year, however due to unexpected seasonable differences in rainfall and some losses of data due to malfunctioning equipment, there was only a partial water year for all the wetlands. The hydroperiod data was used to calculate WLF and to calibrate the computer model Hydrologic Simulation Program- FORTRAN (HSPF), a continuous event model with the ability to simulate hydrologic processes in a watershed. The model is used to predict rainfall runoff from different watershed conditions and is more accurate when field measurements are used to adjust runoff from simulated rainfall events with the outflows and stages resulting from actual events.

Of the six wetlands, two control wetlands were not calibrated nor modeled. The complexity of the wetlands' hydraulics were beyond the scope of this project. The remaining four wetlands all had well defined outlets, hydraulics and bethymetry which allowed reasonably accurate stage-storage-discharge relationships to be developed. Based on the margin of errors in the spatial distribution of precipitation represented by nearby gages and the length of the field record, the accuracy of the model's simulated wetland water levels to recorded water levels was limited to plus or minus 0.5 ft. (15 cm).

Emergent (PEM), scrub-shrub (PSS) and forested (PFO) wetland zones were surveyed and evaluated for plant species richness and the presence and dominance of exotic invasive species using the protocols for vegetation field work documented in Cooke et al. (Cooke et al. 1989). Disturbed commodities were those sample stations found to be dominated (>60%) by a weedy species. Amphibians were sampled during the fall and spring breeding seasons using methods described in Richter and Azous (1995).

The condition of plant and amphibian communities were compared with the observed and predicted water depths, the duration of storm events and the frequency of storm events for the whole season and the early growing season (March 1 through May 15). . We analyzed the emergent, scrub-shrub and forested zones to determine if there were significant differences in community composition related to hydroperiod regimes .

The six special study wetlands were also added to the larger database of 19 wetlands and all the data analyzed for differences corresponding to WLF conditions. All sample stations that were inundated at least once during the year were included in the analysis of water level fluctuation. The data was analyzed using StatView (Abacus Concepts Inc. 1993) statistical applications program. The plant richness data were not normal; therefore the non-parametric Kruskal-Wallace (KW) and Mann-Whitney (MW) tests were used to compare the distributions among categories, depending on the number of variables in the category being compared. Both tests indicate whether the underlying distributions for different groups are the same. Both use ranked data and are resistant to outliers.

Much of the data was categorized to provide more statistical rigor given the small data set and the 0.5 ft. (15 cm.) margin of error. Categories were based on frequency distributions of the data and a very limited sensitivity analysis of statistically significant breaks in the data.

We measured frequency of storm events in a hydroperiod by defining an event as an excursion which we define as a water level increase above the monthly average depth of more than 0.5 ft. (15 cm.). Duration was defined as the time period of an excursion. In a stepwise regression, we looked at the statistical relationship between WLF, frequency and duration. Table 1 shows the categories used in the analysis.

Table 13-1. Category Definitions for Water Depth and Excursion Duration.

Frequency of Excursions	Water Depth*	Duration of Excursions
less than 6 per year	Greater than 2.0 ft. depth (>60 cm.)	less than 3 days
more than 6 per year	2 ft. to 0 ft. depth (-60 to 0 cm.)	3 to 6 days
	0 to 2.0 feet above water surface. (0 to +60 cm.)	more than 6 days

*Negative numbers are under water.

RESULTS

Plant richness in the sample stations ranged from three to 31 species in the POW zones, three to 22 in the PSS zones and 17 to 25 in the forested areas. Very few invasive weedy species were found and were dominant in only a few localized areas.

Frequency and Duration and Plant Richness

Plant richness was found to be significantly lower if water depths were usually deeper than 2 feet (60 cm.) (KW, $p < 0.0001$). To control for this, frequency and duration were evaluated separately for different water depths. The test for differences in duration and frequency showed that, in general, plant communities in areas subjected to more than six hydrologic excursions per year tended to have lower richness. In both the greater than 2.0 feet range and zero to 2.0 feet range the difference is statistically significant (MW, $p \leq 0.004$). It was not significant for the -2.0 to zero range (Figure 13-1).

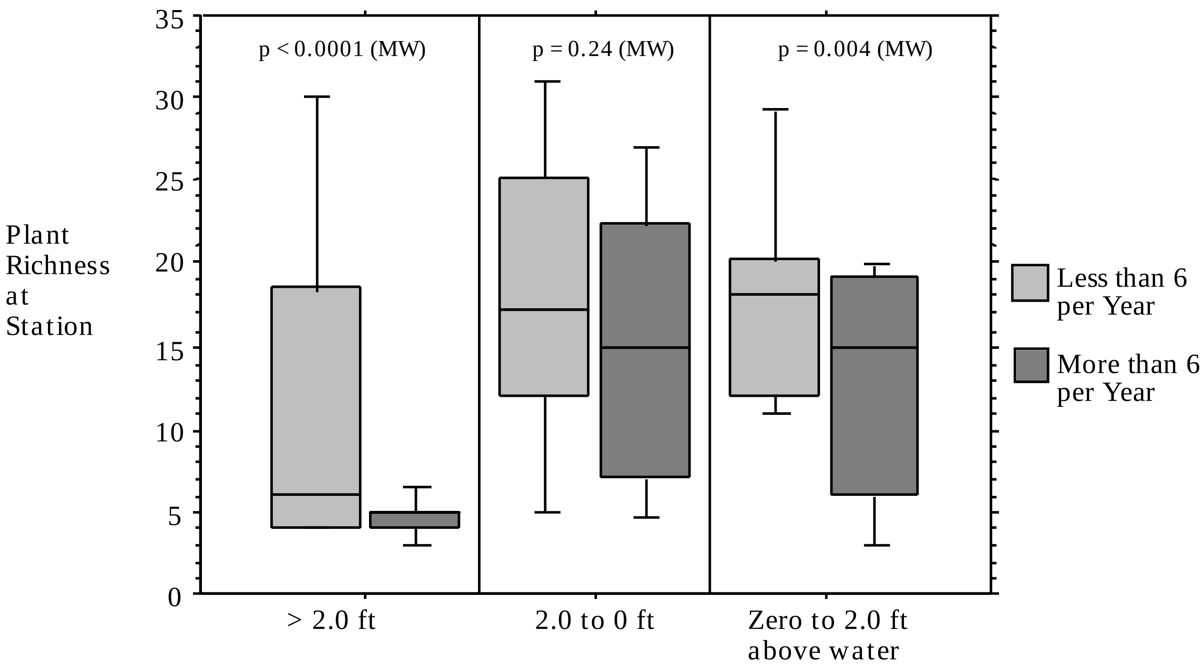

Figure 13-1. Plant richness, water depth and frequency of excursions.

The duration of excursions was compared to plant richness and water depth. Duration alone was a significant factor only in the deepest zones of -8.0 to -2.0 feet (KW, $p < 0.001$) (Figure 13-2). From -2.0 feet to 2.0 feet, increased duration did not significantly contribute to the variability of plant richness.

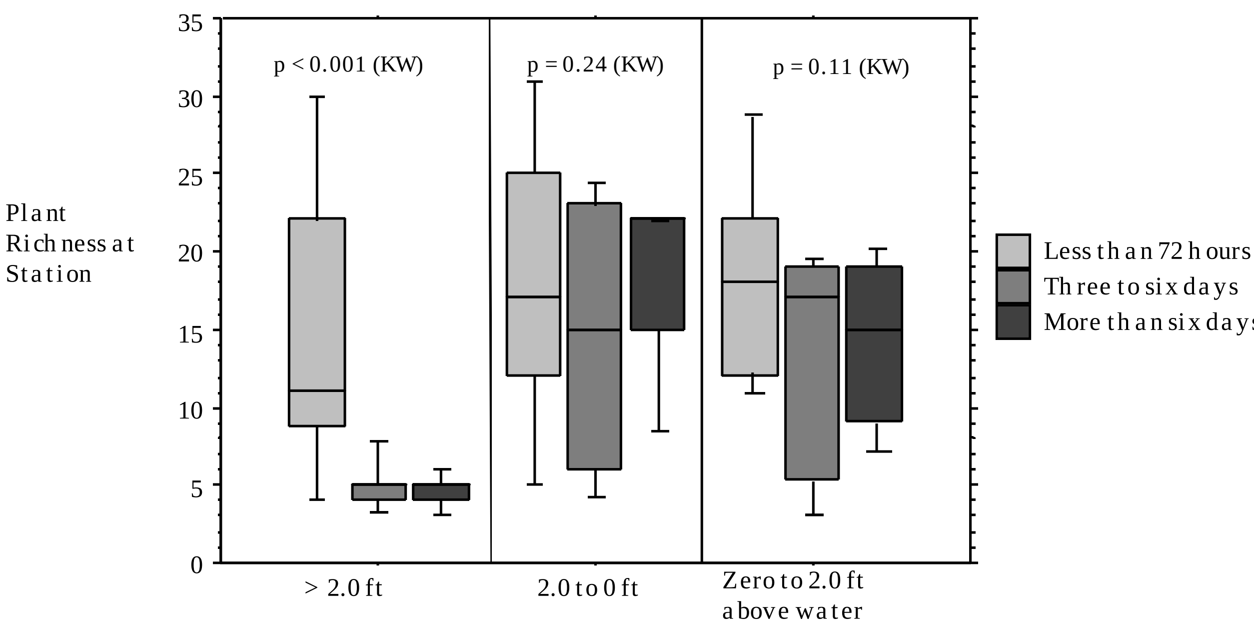

Figure 13-2. Plant richness, water depth and duration of excursions.

When the effects of excursion frequency and duration were combined, the relationship with plant richness was much stronger. Plant richness was found to decrease significantly with excursions longer than six days duration even with frequencies of less than six per year (KW, $p < 0.0001$). For excursion frequencies greater than six per year, richness dropped significantly when duration' exceeded three days per month (KW, $p < 0.0001$) (Figure 13-3)

These results were significant for both emergent and scrub-shrub zones and indicate that the average monthly duration of inundation can be significant to plant species richness, when the frequency of inundation is greater than six times per year on average or when the length of inundation exceeds three days per month. The frequency of excursions did not account for variability in species richness until excursion durations exceeded three days per month. There were an insufficient number of forested zones in the wetlands where frequency and duration were measured to adequately test for differences in the forested conditions and open water.

Water Level Fluctuation and Plant Richness

We looked at the relationship of water level fluctuation to plant richness in different zones of the wetlands. We examined all sample stations inundated at any time of the year and found richness was lower in wetlands with high WLF hydroperiods in the emergent and scrub-shrub zones but not the forested zones. There were not enough aquatic bed zones for adequate evaluation. Emergent zones subject to mean WLFs greater than 0.8 ft. (24 cm.) ranked significantly lower in the number of plant species present (MW, $U \geq 55$, $P \leq 0.003$) than emergent areas with mean WLF less than 0.8 ft. (24 cm.). This relationship was even more significant when richness was compared with water level fluctuation during the early growing season (Figure 13-4). Shrub-scrub zones also showed a significant difference in plant richness related to annual and early growing season water level fluctuation (MW, $U \geq 55$ $p < 0.0001$) (Figure 13-5). Forested zones showed no differences in richness accounted for by WLF.

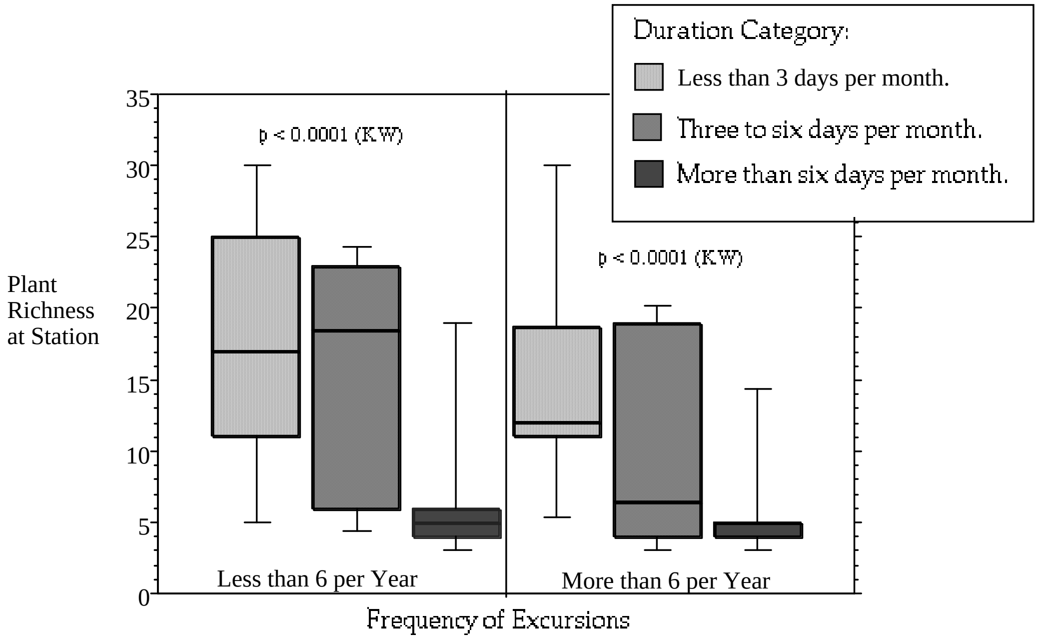

Figure 13-3. Plant richness, frequency and duration of excursions.

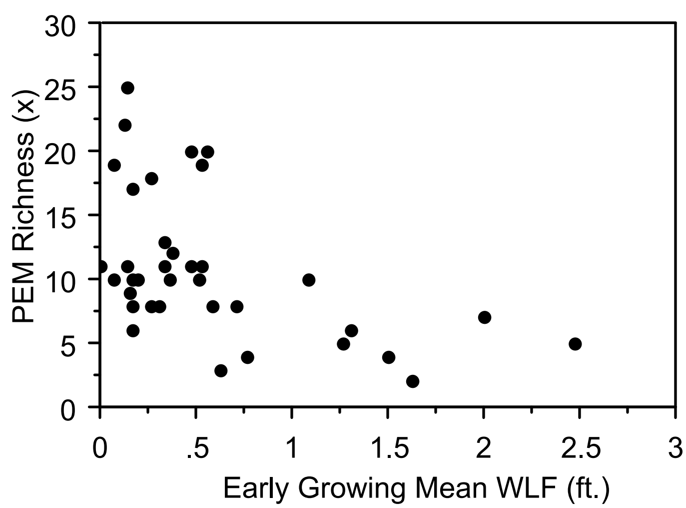

Figure 13-4. Plant rchness in the emergent zones in relation to mean WLF.

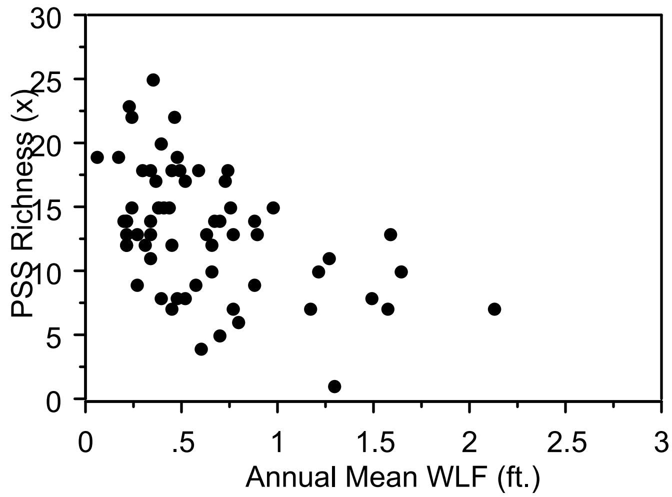

Figure 13-5. Plant richness in the scrub-shrub zones in relation to mean WLF.

Amphibian Results

Our study of amphibians left us with an incomplete picture. All of the wetlands in this study as well as the PSWSRP study had far fewer amphibian species in 1995 than collected in previously years. For example, seven species were collected in a rural wetland, BBC24, in 1989 and only three in 1995. Five species were collected in the urban surrounded wetland, LPS9, in 1989, compared with none in 1995. Eight were captured in SR24 in 1989 and again none were captured in 1995. Figure 13-6 shows amphibian richness for each wetland for both 1989 and 1995 trapping years. The lack of captures prevented analysis of frequency and duration effects for this study's wetlands.

Nevertheless, we were able to measure WLF relationships between amphibian communities over all years and all wetlands using the PSWSMRP wetlands database. The richness of amphibian communities was found to be lower in wetlands with WLF less than 0.8 ft. (24 cm). Wetlands with greater WLF were significantly more likely to have low amphibian richness with three or fewer different species present (*FE*, P = 0.046) as compared with four to eight.

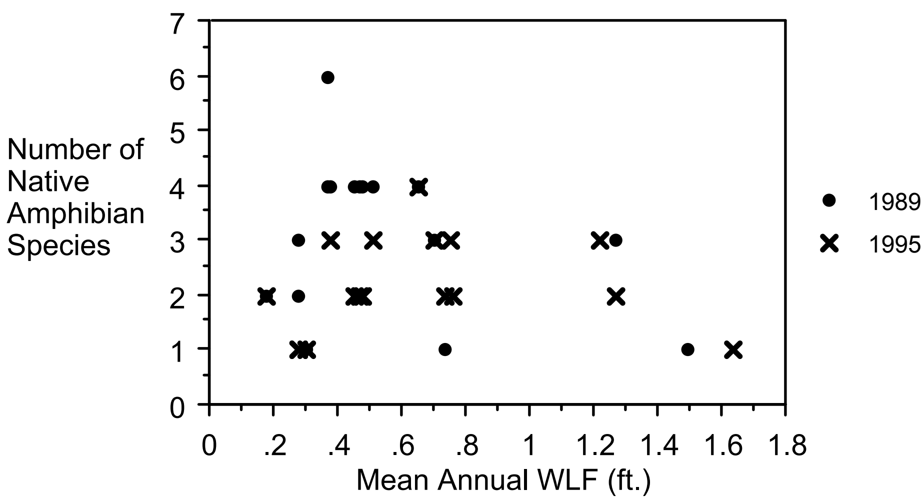

Figure 13-6. Amphibian richness as a function of mean WLF.

The reasons for the amphibian decline in 1995 are not understood. Amphibians sometimes breed in alternate years, hence in one year, populations could be much lower than the next. But we don't know if that phenomenon occurs across a population or just to particular individuals. The fact that low numbers were found in all wetlands suggests that it may be rainfall or climate related and 1995 was a drier spring than usual, but we are speculating.

WLF was found to be statistically related to excursion duration and frequency. Forty-one percent of the variation in WLF can be explained by the duration of events. Adding the effect of excursion frequency can explain as much as 53% of the variability in WLF ($p<0.0001$).

APPLICATION OF RESEARCH RESULTS

These results show that increasing the duration of storm events can be a significant factor in reducing wetland plant diversity. The frequency of storm peaks is also a factor and compounds the duration impact. Decreasing richness in the emergent and scrub-shrub zones and increasing frequency and duration are also associated with high mean water level fluctuation, annually, but particularly during the early spring growing season and amphibian breeding seasons.

Current stormwater protection measures primarily rely on stormwater detention for protecting wetlands. Detention acts to increase the duration of a storm event in order to reduce the peak depth. Water is captured, stored and released after the storm over a longer period of time. It was a management tool designed primarily for controlling floods and erosion in streams, however, it may operate counter to management goals as a tool for wetland protection.

The result of these findings has been to recommend for there to be limits on the durations of storm events as well as the frequency of excursions, when wetlands will be affected by changes in hydroperiod. The recommendations are that the frequency of water levels greater than 15 cm. (.5 ft.) above pre-development levels be limited to an annual average of six or less per year and that the durations of water levels greater than 15 cm. (.5 ft.) above or below pre-development levels be limited to less than three days per excursion.

The data set we analyzed was limited, as were time and funding and some questions remain about the potential for trading flood frequency and flood duration. For example, it might be possible to extend the durations of storm flows in wetlands if the frequency of those events is reduced. Similarly, it may also be possible to reduce durations in trade for allowing greater frequency. These areas of refinement remain largely unexplored.

Irrespective of any further results, it will be difficult for urbanizing jurisdictions to meet such standards in all areas. It is also not likely to happen if detention is the primary management tool. Achieving real resource protection of high value wetlands will require a more comprehensive approach.

Early in the research the PSWSRP learned that wetland management must be holistic, that wetlands are part of a system in a larger landscape and should be managed accordingly. This view has a number of implications for management:

- It is necessary to consider incidental effects on wetlands of activities in their watersheds, along with any engineering performed on the wetland itself for stormwater management purposes;
- Wetland response and management depend on a host of landscape factors, including retention of forest and other natural cover, maintenance of natural storage reservoirs and drainage corridors; the separation of human activities from wetlands; and public awareness.
- Wetland protection means finding root cause solutions e.g. source control practices that prevent or minimize quantities of runoff and release of pollutants, with downstream retention/detention for quantity control and treatment for pollutant capture regarded as secondary back-up measures where source controls alone can not ensure resource protection.
- Potential runoff infiltration opportunities should be explored and those that are found to be workable hydrogeologically and not threaten groundwater quality should be explored.

The experience of King County in its attempts to meet the PSWSRP recom-mendations is noteworthy and affords a view of some alternative approaches to detention.

The PSWSRP guidelines have been used in King County in both the basin and master drainage planning processes. Most of the applications have focused on minimizing water level fluctuation, as it was identified as the most direct effect on wetland functioning, vegetation communities, and habitat for breeding amphibians. Regulations governing factors that affect WLF have been targeted at new development on the urban side of the Urban Growth Boundary (UGB), where the most significant impacts are likely to occur. The general information on construction impacts generated by the Wetlands Research Program has also led to the application of seasonal clearing limits in the drainage areas of Class 1 wetlands.

Basin Planning

The basin planning process was developed by King County to address the significant and rapid land use changes occurring in the county that have an impact on water resources, including flooding, habitat, and water quality. The outcome of the basin planning process is a way of developing a comprehensive set of management recommendations that involve development regulations, capital improvement projects, education programs, improved maintenance practices, and monitoring.

The East Lake Sammamish Basin Plan (King County Surface Water Management Division (KCSWM) 1992) is an example where the results of the Wetlands Research Program were directly applied to management solutions. The East Lake Sammamish basin encompasses about 16 square miles east of Lake Sammamish. Since 1980, the basin has experienced rapid development, converting from low-density residential and forested land uses to higher density residential and some commercial uses. The diversity of the basin's more than 40 inventoried wetlands is as great as anywhere in King County, with nine wetlands ranked as unique and outstanding (Class 1 rating). As one of the prime resources in the basin, wetlands received significant attention for protection from the County and the citizenry.

Wetland Management Areas

Prior to adoption of the basin plan, wetland protection in King County was achieved primarily through the Sensitive Areas Ordinance (SAO). The wetland protection in the SAO provides for discrete buffer widths as a function of assigned rating (e. g., 100 feet for Class 1 wetlands). Although these buffers confer some protection to wetlands, they are inadequate to protect other functions influenced by the broader watershed and surrounding landscape. To address these issues, King County developed wetland management areas (WMA) focused on watershed-based controls to protect the nine Class 1 wetlands. The intent of these controls was to minimize the stormwater-related impacts on wetlands by minimizing impervious surfaces, retaining forests, clustering, and providing constructed infiltration systems, where feasible.

A major component of the wetland management strategy was the limitation of total impervious area in the catchment to eight percent, where allowed by zoning. From the Wetlands Research Program data, it was clear that there were significant increases in WLF between wetlands with watersheds less than 4 percent and those with watersheds greater than 12 percent impervious surface (Taylor 1993; Taylor, Ludwa, and Horner 1995). It was difficult to define this more precisely, because of the absence of impervious surfaces between 4 and 12 percent. Booth and Reinelt (1994) summarized several data sets showing loss of aquatic system function with impervious surface areas above about 10 percent, as measured by changes in channel morphology, fish and amphibian populations, habitat, and water chemistry. While the precise threshold will vary by watershed and the effectiveness of mitigation strategies, 8-10 percent impervious surface appears to be an appropriate threshold.

A requirement for 50 percent forest retention was also imposed in the catchments of some wetlands. This limitation is consistent with King County's reserve tract requirements associated with clustering and growth-reserve zoning. Taylor (1993) found a correlation between forest retention and reduced WLF, but no specific threshold was identified in this work. Clustering of development away from hydrologic source areas (landscape features transmitting water to wetlands during the wet season) was also recommended. An additional requirement in one wetland watershed was the use of constructed infiltration systems to reduce increases in stormwater volumes. This was feasible given the extensive glacial outwash soils in this watershed that were amenable to substantial infiltration. Finally, seasonal clearing limits for construction activities were imposed in eight of the nine watersheds. This limitation prevents clearing and grading during the wet season (October-April) when up to 88 percent of erosion occurs (KCSWM 1992).

King County has continued this approach of wetland management areas for protection of Class 1 wetlands in the Cedar River Basin Plan currently under development. Four Class 1 wetlands in the Cedar basin that are on the urban side of the UGB or that receive runoff from urban areas have been targeted.

Master Drainage Planning and Guidelines

King County uses the Master Drainage Planning (MDP) process for large or complex development sites to assess the potential impacts of development on aquatic resources (KCSWM 1993). The MDP process is required for Urban Plan Developments (UPD), for subdivisions with more than 100 single-family residences, and for projects which clear 500 acres or more within a subbasin. In addition, there are lower thresholds for development in the drainage areas of Class 1 wetlands, regionally significant resource streams, or over sole source aquifers. For Class 1 wetlands, an MDP is required if a project seeks to convert more than 10 percent of the wetland's total watershed area to impervious surface.

The updated guidelines for MDP monitoring and studies (KCSWM 1993), supported in part by results of the Wetlands Research Program, require monitoring for purposes of: (1) assessing wetland functions in storing and releasing stormwater, (2) determining baseline WLF in relation to vegetation and amphibian communities, and (3) establishing baseline conditions from which to measure potential post-development changes. Specific concerns potentially resulting from development are: (1) loss of live storage and infiltration functions of wetlands, (2) stability of outlet control conditions, (3) the effects of increases in flow rates and volumes, (4) changes in spring WLF and resultant habitat changes, and (5) changes in groundwater and interflow.

For purposes of assessing wetland impacts, the MDP guidelines require determination of the following: bathymetry (morphometry) of the wetland; outlet control description and measurement; stage-discharge volume relationships; surface area of open water, including ordinary high water levels; and the dead and live storage maximum elevation and volume. Specific monitoring requirements are: (1) monthly instantaneous and crest water levels to determine WLF in the permanent pool area of the wetland; (2) inflow and outflow rates of the wetland; and (3) the duration of summer drying, if applicable.

For the North Fork Issaquah Creek Wetland 7 Management Area and Grand Ridge MDP, the East Sammamish Community Plan limited development in the drainage area tributary to North Fork Issaquah Creek Wetland 7 (NFIC-7), a Class 1 wetland, to no more than eight percent impervious surfaces and 65 percent forest retention. This condition applies to all development proposals submitted prior to adoption of the Issaquah Basin Plan (KCSWM 1994) and for all developments not going through the MDP process. In the basin plan, impervious surfaces are limited to a maximum of eight percent for all new subdivisions, short subdivision, and UPDs.

The proposed Grand Ridge development in the North and East Fork Issaquah Creek basins involved two development options: rural estates at a density of one unit per 5 acres and an urban proposal consisting of 580 acres of urban development and 1400 acres of permanent open space. In a study of potential development scenarios carried out using the Wetlands Research Program guidelines and a model developed by Taylor (1993), it was possible to examine the development impacts on the water level fluctuation of wetland NFIC-7. Based on the results of that analysis, mitigations were proposed that focused on maintaining greater forested area and utilizing infiltration to reduce stormwater volumes.

CONCLUSION

Fundamentally managing stormwater to protect wetland ecosystems must operate holistically within context of the hydrologic cycle. That requires that we consider infiltration and evapotranspiration in addition to storage, when we think about strategies. Controls focused on minimizing impervious surfaces and maximizing forest retention are likely to be the most widely usable effective strategies; however, additional mitigations that reduce stormwater volumes through infiltration are highly recommended when hydrogeological conditions permit.

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CHAPTER 14 WETLANDS AND STORMWATER MANAGEMENT GUIDELINES

by Richard R. Horner, Amanda A. Azous, Klaus O. Richter, Sarah S. Cooke, Lorin E. Reinelt and Kern Ewing

If you are unfamiliar with these guidelines, read the description of the approach and organization that follows. If you are familiar, proceed directly to the appropriate guide sheet(s) for guidelines covering your issue(s) or objective(s):

Guide Sheet 1: Comprehensive Landscape Planning for Wetlands and Stormwater Management--page 202

Guide Sheet 2: Wetlands Protection Guidelines-- page 209

APPROACH AND ORGANIZATION OF THE MANAGEMENT GUIDELINES

Introduction

The Puget Sound Wetlands and Stormwater Management Research Program performed comprehensive research with the goal of deriving strategies that protect wetland resources in urban and urbanizing areas, while also benefiting the management of urban stormwater runoff that can affect those resources. The research primarily involved long-term comparisons of wetland ecosystem characteristics before and after their watersheds urbanized, and between a set of wetlands that became affected by urbanization (treatment sites) and a set whose watersheds did not change (control sites). This work was supplemented by shorter term and more intensive studies of pollutant transport and fate in wetlands, several laboratory experiments, and ongoing review of relevant work being performed elsewhere. These research efforts were aimed at defining the types of impacts that urbanization can cause and the degree to which they develop under different conditions, in order to identify means of avoiding or minimizing impacts that impair wetland structure and functioning. The program's scope embraced both situations where urban drainage incidentally affects wetlands in its path, as well as those in which direct stormwater management actions change wetlands' hydrology, water quality or both.

This document presents preliminary management guidelines for urban wetlands and their stormwater discharges based on the research results. The set of guidelines is the principal vehicle to implement the research findings in environmental planning and management practice.

Guidelines Scope and Underlying Principles

Note: For terms in **boldface** type see item 1 under Support Materials below.

1. These provisions currently have the status of guidelines rather than requirements. Application of these guidelines does not fulfill assessment and permitting requirements

that may be associated with a project. It is, in general, necessary to follow the stipulations of the State Environmental Policy Act and to contact such agencies as the local planning agency; the Washington Departments of Ecology, Fisheries, and Wildlife; the U. S. Environmental Protection Agency; and the U. S. Army Corps of Engineers.

2. Using the guidelines should be approached from a problem-solving viewpoint. The "problem" is regarded to be accomplishing one or more particular planning or management objectives involving a **wetland** potentially or presently affected by stormwater drainage from an urban or urbanizing area. The objectives can be broad, specific, or both. Broad objectives involve comprehensive planning and subsequent management of a drainage catchment or other **landscape unit** containing one or more wetlands. Specific objectives pertain to managing a wetland having particular attributes to be sustained. Of course, the prospect for success is greater with ability to manage the whole landscape influencing the wetland, rather than just the wetland itself.

3. The guidelines are framed from the standpoint that some change in the landscape has the potential to modify the physical and chemical **structure** of the wetland environment, which in turn could alter biological communities and the wetland's ecological **functions**. The general objective in this framework would be to avoid or minimize negative ecological change. This view is in contrast to one in which a wetland has at some time in the past experienced negative change, and consequent ecological degradation, and where the general objective would be to recover some or all of the lost structure and functioning through **enhancement** or **restoration** actions. Direct attention to this problem was outside the scope of the Puget Sound Wetlands and Stormwater Management Research Program. However, the guidelines do give information that applies to enhancement and restoration. For example, attempted restoration of a diverse amphibian community would not be successful if the water level fluctuation limits consistent with high amphibian species richness are not observed.

4. The guidelines can be applied with whatever information concerning the problem is available. Of course, the comprehensiveness and certainty of the outcome will vary with the amount and quality of information employed. The guidelines can be applied in an iterative fashion to improve management understanding as the information improves. Appendix A lists the information needed to perform basic analyses, followed by other information that can improve the understanding and analysis.

5. These guidelines emphasize avoiding structural, hydrologic, and water quality **modifications** of existing wetlands to the extent possible in the process of urbanization and the management of urban stormwater runoff.

6. In pursuit of this goal, the guidelines take a systematic approach to management problems that potentially involve both urban stormwater (quantity, quality, or both) and wetlands. The consideration of wetlands involves their areal extent, **values**, and functions. This approach emphasizes a comprehensive analysis of alternatives to solve the identified problem. The guidelines encourage conducting the analysis on a landscape scale and considering all of the possible stormwater management alternatives, which may or may not involve a wetland. They favor **source control best management practices** (BMPs) and **pre-treatment** of stormwater runoff prior to release to wetlands.

7. Furthermore, the guidelines take a holistic view of managing wetland resources in an urban setting. Thus, they recognize that urban wetlands have the potential to be affected structurally and functionally whether or not they are formally designated for stormwater management purposes. Even if an urban wetland is not structurally or hydrologically engineered for such purposes, it may experience altered hydrology (more or less water), reduced water quality, and a host of other impacts related to urban conditions. It is the objective of the guidelines to avoid or reduce the negative effects on wetland resources from both specific stormwater management actions and incidental urban impacts.

Support Material

1. The guidelines use certain terms that require definition to ensure that the intended meaning is conveyed to all users. Such terms are printed in **boldface** the first time that they appear in each guide sheet, and are defined in Appendix B.

2. The guideline provisions were drawn principally from the available results of the Puget Sound Wetlands and Stormwater Management Research Program, as set forth in Sections 2 and 3 of the program's summary publication, Wetlands and Urbanization, Implications for the Future (Horner et al. 1996). Where the results in this publication are the basis for a numerical provision, a separate reference is not given. Numerical provisions based on other sources are referenced. See Appendix C for references.

3. Appendix D presents a list of plant species native to wetlands in the Puget Sound Region. This appendix is intended for reference by guideline users who are not specialists in wetland botany. However, non-specialists should obtain expert advice when making decisions involving vegetation.

4. Appendix E compares the water chemistry characteristics of *Sphagnum* bog and fen wetlands (termed **priority peat wetlands** in these guidelines) with more common wetland communities. These bogs and fens appear to be the most sensitive among the Puget Sound lowland wetlands to alteration of water chemistry, and require special water quality management to avoid losses of their relatively rare communities.

GUIDE SHEET 1: COMPREHENSIVE LANDSCAPE PLANNING FOR WETLANDS AND STORMWATER MANAGEMENT

Wetlands in newly developing areas will receive urban effects even if not specifically "used" in stormwater management. Therefore, the task is proper overall management of the resources and protection of their general **functioning**, including their role in storm drainage systems. Stormwater management in newly developing areas is distinguished from management in already developed locations by the existence of many more feasible stormwater control options prior to development. The guidelines emphasize appropriate selection among the options to achieve optimum overall resource protection benefits, extending to downstream receiving waters and ground water aquifers, as well as to wetlands.

The comprehensive planning guidelines are based on two principles that are recognized to create the most effective environmental management: (1) the best management policies for the protection of wetlands and other natural resources are those that prevent or minimize the development of impacts at potential sources; and (2) the best management strategies are self-perpetuating, that is they do not require periodic infusions of capital and labor. To apply these principles in managing wetlands in a newly developing area, carry out the following steps.

Guide Sheet 1A: Comprehensive Planning Steps

1. Define the **landscape unit** subject to comprehensive planning. Refer to the definition of landscape unit in Appendix B for assistance in defining it.

2. Begin the development of a plan for the landscape unit with attention to the following general principles:

- Formulate the plan on the basis of clearly articulated community goals. Carefully identify conflicts and choices between retaining and protecting desired resources and community growth.

- Map and assess land suitability for urban uses. Include the following landscape features in the assessment: forested land, open unforested land, steep slopes, erosion-prone soils, foundation suitability, soil suitability for waste disposal, aquifers, aquifer recharge areas, wetlands, floodplains, surface waters, agricultural lands, and various categories of urban land use. When appropriate, the assessment can highlight outstanding local or regional resources that the community determines should be protected (e. g., a fish run, scenic area, recreational area, threatened species habitat, farmland). Mapping and assessment should recognize not only these resources but also additional areas needed for their sustenance.

3. Maximize natural water storage and infiltration opportunities within the landscape unit and outside of existing wetlands, especially:

- Promote the conservation of forest cover. Building on land that is already deforested affects basin hydrology to a lesser extent than converting forested land. Loss of forest cover reduces interception storage, detention in the organic forest floor layer, and water losses by evapotranspiration, resulting in large peak runoff increases and either their negative effects or the expense of countering them with structural solutions.

- Maintain natural storage reservoirs and drainage corridors, including depressions, areas of permeable soils, swales, and intermittent streams. Develop and implement policies and regulations to discourage the clearing, filling, and channelization of these features. Utilize them in drainage networks in preference to pipes, culverts, and engineered ditches.

- In evaluating infiltration opportunities refer to the stormwater management manual for the jurisdiction and pay particular attention to the selection criteria for avoiding groundwater contamination and poor soils and hydrogeological conditions that cause these facilities to fail. If necessary, locate developments with large amounts of impervious surfaces or a potential to produce relatively contaminated runoff away from groundwater recharge areas. Relatively dense developments on glacial outwash soils may require additional runoff treatment to protect groundwater quality.

4. Establish and maintain **buffers** surrounding wetlands and in riparian zones as required by local regulations or recommended by the Puget Sound Water Quality Authority's wetland guidelines. Also, maintain interconnections among wetlands and other natural habitats to allow for wildlife movements.

5. Take specific management measures to avoid general urban impacts on wetlands and other water bodies (e. g., littering, vegetation destruction, human and pet intrusion harmful to wildlife).

6. To support management of <u>runoff water quantity</u>, perform a hydrologic analysis of the contributing drainage catchment to define the type and extent of flooding and stream channel erosion problems associated with existing development, redevelopment, or new development that require control to protect the beneficial uses of receiving waters, including wetlands. This analysis should include assembly of existing flow data and hydrologic modeling as necessary to establish conditions limiting to attainment of beneficial uses. Modeling should be performed as directed by the stormwater management manual in effect in the jurisdiction.

7. In wetlands previously relatively unaffected by human activities, manage <u>stormwater quantity</u> to attempt to match the **pre-development hydroperiod** and **hydrodynamics**. In wetlands whose hydrology has been disturbed, consider ways of reducing hydrologic impacts. This provision involves not only management of high runoff volumes and rates of flow during the wet season, but also prevention of water supply depletion during the dry season. The latter guideline may require flow augmentation if urbanization reduces existing surface or groundwater inflows. Refer to Guide Sheet 2, Wetland Protection Guidelines, for detail on implementing these guidelines.

8. Assess alternatives for the control of <u>runoff water quantities</u> as follows:

a. Define the runoff quantity problem subject to management by analyzing the proposed land development action.

b. For existing development or **redevelopment**, assess possible alternative solutions that are applicable at the site of the problem occurrence, including:

- Protect health, safety, and property from flooding by removing habitation from the flood plain.

- Prevent stream channel erosion by stabilizing the eroding bed and/or bank area with **bioengineering** techniques, preferably, or by structurally reinforcing it, if this solution would be consistent with the protection of aquatic habitats and beneficial uses of the stream (refer to Chapter 173-201A of the Washington Administrative Code (WAC) for the definition of beneficial uses).

b. For new development or redevelopment, assess possible regulatory and incentive land use control alternatives, such as density controls, clearing limits, impervious surface limits, transfer of development rights, purchase of conservation areas, etc.

c. If the alternatives considered in Steps 8a or 8b cannot solve an existing or potential problem, perform an analysis of the contributing drainage catchment to assess possible alternative solutions that can be applied **on-site** or on a **regional** scale. The most appropriate solution or combination of alternatives should be selected with regard to the specific opportunities and constraints existing in the drainage catchment. For new development or redevelopment, on-site facilities that should be assessed include, in approximate order of preference:

- Infiltration basins or trenches;

- Retention/detention ponds;

- Below-ground vault or tank storage;

- Parking lot detention.

Regional facilities that should be assessed for solving problems associated with new development, redevelopment, or existing development include:

- Infiltration basins or trenches;

- Detention ponds;

- **Constructed wetlands**;

- Bypassing a portion of the flow to an acceptable receiving water body, with treatment as required to protect water quality and other special precautions as necessary to prevent downstream impacts.

d. Consider structurally or hydrologically engineering an existing wetland for water quantity control only if upland alternatives are inadequate to solve the existing or potential problem. To evaluate the possibility, refer to the Stormwater Wetland Assessment Criteria in Guide Sheet 1B.

9. Place strong emphasis on water resource protection during construction of new development. Establish effective erosion control programs to reduce the sediment loadings to receiving waters to the maximum extent possible. No preexisting wetland or other water body should ever be used for the sedimentation of solids in construction-phase runoff.

10. In wetlands previously relatively unaffected by human activities, manage stormwater quality to attempt to match pre-development water quality conditions. To support management of runoff water quality, perform an analysis of the contributing drainage catchment to define the type and extent of runoff water quality problems associated with existing development, redevelopment, or new development that require control to protect the beneficial uses of receiving waters, including wetlands. This analysis should incorporate the hydrologic assessment performed under step 6 and include identification of key water pollutants, which may include solids, oxygen-demanding substances, nutrients, metals, oils, trace organics, and bacteria, and evaluation of the potential effects of water pollutants throughout the drainage system.

11. Assess alternatives for the control of runoff water quality as follows:

a. Perform an analysis of the contributing drainage catchment to assess possible alternative solutions that can be applied on-site or on a regional scale. The most appropriate solution or combination of alternatives should be selected with regard to the specific opportunities and constraints existing in the drainage catchment. Consider both **source control BMPs** and **treatment BMPs** as alternative solutions before considering use of existing wetlands for quality improvement according to the following considerations:

- Implementation of source control BMPs prevent the generation or release of water pollutants at potential sources. These alternatives are generally both more effective and less expensive than treatment controls. They should be applied to the maximum extent possible to new development, redevelopment, and existing development.

- Treatment BMPs capture water pollutants after their release. This alternative often has limited application in existing developments because of space limitations, although it can be employed in new development and when redevelopment occurs in already developed areas. Following is a list of treatment BMPs that should be considered. Each has appropriate and inappropriate applications and advantages and disadvantages and must be carefully selected, designed, constructed, and operated according to the specifications of the stormwater management manual in use in the jurisdiction.

- Infiltration basins or trenches;
- Constructed wetlands;
- Wet or extended-detention ponds;
- Biofiltration facilities (vegetated swales or filter strips);
- Filters with sand, compost, or other media;
- Water quality vaults;
- Oil/water separators.

b. Consider structurally or hydrologically engineering an existing wetland for water quality control only if upland alternatives are inadequate to solve the existing or potential problem. Use of Waters of the State and Waters of the United States, including wetlands, for the treatment or conveyance of wastewater, including stormwater, is prohibited under state and federal law. Discussions with federal and state regulators during the research program led to development of a statement concerning the use of existing wetlands for improving stormwater quality (**polishing**), as follows. Such use is subject to analysis on a case-by-case basis and may be allowed only if the following conditions are met:

- If **restoration** or **enhancement** of a previously **degraded** wetland is required, and if the upgrading of other wetland functions can be accomplished along with benefiting runoff quality control, <u>and</u>
- If appropriate source control and treatment BMPs are applied in the contributing catchment on the basis of the analysis in Step 10a and any legally adopted water quality standards for wetlands are observed.

If these circumstances apply, refer to the Stormwater Wetland Assessment Criteria in Guide Sheet 1B to evaluate further.

12. Stimulate public awareness of and interest in wetlands and other water resources in order to establish protective attitudes in the community. This program should include:

- Education regarding the use of fertilizers and pesticides, automobile maintenance, the care of animals to prevent water pollution, and the importance of retaining buffers;
- Descriptive signboards adjacent to wetlands informing residents of the wetland type, its functions, the protective measures being taken, etc.

- If beavers are present in a wetland, educate residents about their ecological role and value and take steps to avoid human interference with beavers.

Guide Sheet 1B: Stormwater Wetland Assessment Criteria

This guide sheet gives criteria that disqualify a natural wetland from being structurally or hydrologically engineered for control of stormwater quantity, quality, or both. These criteria should be applied only after performing the alternatives analysis outlined in Guide Sheet 1A.

1. A wetland <u>should not</u> be structurally or hydrologically engineered for runoff quantity or quality control and should be given maximum protection from overall urban impacts (see Guide Sheet 2, Wetland Protection Guidelines) under any of the following circumstances:

- In its present state it is primarily an **estuarine** or **forested wetland** or a **priority peat system**.

- It is a rare or irreplaceable wetland type, as identified by the Washington Natural Heritage Program, the Puget Sound Water Quality Preservation Program, or local government.

- It provides **rare, threatened, or endangered species** habitat that could be impaired by the proposed action. Determining whether or not the conserved species will be affected by the proposed project requires a careful analysis of its requirements in relation to the anticipated habitat changes.

In general, the wetlands in these groups are classified in Categories I and II in the Puget Sound Water Quality Authority's draft wetland guidelines.

2. A wetland <u>can be considered</u> for structural or hydrological modification for runoff quantity or quality control if most of the following circumstances exist:

- It is classified in Category IV in the Puget Sound Water Quality Authority's draft wetland guidelines. In general, Category IV wetlands have monotypic vegetation of similar age and class, lack special habitat features, and are isolated from other aquatic systems.

- The wetland has been previously **disturbed** by human activity, as evidenced by agriculture, fill, ditching, and/or introduced or **invasive weedy plant species**.

- The wetland has been deprived of a significant amount of its water supply by draining or previous urbanization (e. g., by loss of groundwater supply), <u>and</u> stormwater runoff is sufficient to augment the water supply. A particular candidate is a wetland that has experienced an increased summer dry period, especially if the drought has been extended by more than two weeks.

- Construction for structural or hydrologic modification in order to provide runoff quantity or quality control will disturb relatively little of the wetland.

- The wetland can provide the required storage capacity for quantity or quality control through an outlet orifice modification to increase storage of water, rather than through raising the existing overflow. Orifice modification is likely to require less construction activity and consequent negative impacts.

- Under existing conditions the wetland's experiences a relatively high degree of water level fluctuation and a range of velocities (i. e., a wetland associated with substantially flowing water, rather than one in the headwaters or entirely isolated from flowing water).

- The wetland does not exhibit any of the following features:

  — Significant priority peat system or forested zones that will experience substantially altered hydroperiod as a result of the proposed action;

  — Regionally **unusual biological community types**;

  — Animal habitat features of relatively high value in the region (e. g., a protected, undisturbed area connected through undisturbed corridors to other valuable habitats, an important breeding site for protected species);

  — The presence of protected commercial or sport fish;

  — Configuration and topography that will require significant modification that may threaten fish stranding;

  — A relatively high degree of public interest as a result of, for example, offering valued local open space or educational, scientific, or recreational opportunities, unless the proposed action would enhance these opportunities;

- The wetland is threatened by potential impacts exclusive of stormwater management, and could receive greater protection if acquired for a stormwater management project rather than left in existing ownership.

- There is good evidence that the wetland actually can be restored or enhanced to perform other functions in addition to runoff quantity or quality control.

- There is good evidence that the wetland lends itself to the effective application of the Wetland Protection Guidelines in Guide Sheet 2.

- The wetland lies in the natural routing of the runoff. Local regulations often prohibit drainage diversion from one basin to another.

- The wetland allows runoff discharge at the natural location.

GUIDE SHEET 2: WETLAND PROTECTION GUIDELINES

This guide sheet provides information about likely changes to the ecological **structure** and **functioning** of **wetlands** that are incidentally subject to the effects of an urban or urbanizing watershed or are **modified** to supply runoff water quantity or quality control benefits. The guide sheet also recommends management actions that can avoid or minimize deleterious changes in these wetlands.

Guide Sheet 2A: General Wetland Protection Guidelines

1. Consult regulations issued under federal and state laws that govern the discharge of pollutants. Wetlands are classified as "Waters of the United States" and "Waters of the State" in Washington.

2. Maintain the wetland **buffer** required by local regulations or recommended by the Puget Sound Water Quality Authority's draft wetland guidelines.

3. Retain areas of native vegetation connecting the wetland and its buffer with nearby wetlands and other contiguous areas of native vegetation.

4. Avoid compaction of soil and introduction of exotic plant species during any work in a wetland.

5. Take specific site design and maintenance measures to avoid general urban impacts (e. g., littering and vegetation destruction). Examples are protecting existing buffer zones; discouraging access, especially by vehicles, by plantings outside the wetland; and encouragement of stewardship by a homeowners' association. Fences can be useful to restrict dogs and pedestrian access, but they also interfere with wildlife movements. Their use should be very carefully evaluated on the basis of the relative importance of intrusive impacts versus wildlife presence. Fences should generally not be installed when wildlife would be restricted and intrusion is relatively minor. They generally should be used when wildlife passage is not a major issue and the potential for intrusive impacts is high. When wildlife movements and intrusion are both issues, the circumstances will have to be weighed to make a decision about fencing.

6. If the wetland inlet will be modified for the stormwater management project, use a diffuse flow method, such as a spreader swale, to discharge water into the wetland in order to prevent flow channelization.

Guide Sheet 2B: Guidelines for Protection from Adverse Impacts of Modified Runoff Quantity Discharged to Wetlands

1. Protection of wetland plant and animal communities depends on controlling the wetland's **hydroperiod**, meaning the pattern of fluctuation of water depth and the frequency and duration of exceeding certain levels, including the length and onset of drying in the summer. A hydrologic assessment is useful to measure or estimate elements of the hydroperiod under existing **pre-development** and anticipated **post-development** conditions. This assessment should be performed with the aid of a qualified hydrologist. Post-development estimates of watershed hydrology and wetland hydroperiod must include the cumulative effect of all anticipated watershed and wetland modifications. Provisions in these guidelines pertain to the full anticipated build-out of the wetland's watershed.

This analysis hypothesizes a fluctuating water stage over time before development that could fluctuate more, both higher and lower after development; these greater fluctuations are termed **stage excursions**. The guidelines set limits on the frequency and duration of excursions, as well as on overall water level fluctuation, after development.

To determine existing hydroperiod use one of the following methods, listed in order of preference:

- Estimation by a continuous simulation computer model--The model should be calibrated with at least one year of data taken using a continuously recording level gage under existing conditions and should be run for the historical rainfall period. The resulting data can be used to express the magnitudes of depth fluctuation, as well as the frequencies and durations of surpassing given depths. [Note: Modeling that yields high quality information of the type needed for wetland hydroperiod analysis is a complex subject. Providing guidance on selecting and applying modeling options is beyond the scope of these guidelines but is being developed by King County Surface Water Management Division and other local jurisdictions. An alternative possibility to modeling depths, frequencies, and durations within the wetland is to model durations above given discharge levels entering the wetland over various time periods (e. g., seasonal, monthly, weekly). This option requires further development.]

- Measurement during a series of time intervals (no longer than one month in length) over a period of at least one year of the maximum water stage, using a crest stage gage, and instantaneous water stage, using a staff gage--The resulting data can be used to express water level fluctuation (WLF) during the interval as follows:

  Average base stage = (Instantaneous stage at beginning of interval + Instantaneous stage at end of interval)/2

  WLF = Crest stage - Average base stage

  Compute mean annual and mean monthly WLF as the arithmetic averages for each year and month for which data are available.

To forecast future hydroperiod use one of the following methods, listed in order of preference:

- Estimation by the continuous simulation computer model calibrated during pre-development analysis and run for the historical rainfall period--The resulting data can be used to express the magnitudes of depth fluctuation, as well as the frequencies and durations of surpassing given depths. [Note: Post-development modeling results should generally be compared with pre-development modeling results, rather than directly with field measurements, because different sets of assumptions underlie modeling and monitoring. Making pre- and post-development comparisons on the basis of common assumptions allows cancellation of errors inherent in the assumptions.]

- Estimation according to general relationships developed from the Puget Sound Wetlands and Stormwater Management Program Research Program, as follows (in part adapted from Chin 1996):

  — Mean annual WLF is very likely (100% of cases measured) to be < 20 cm (8 inches or 0.7 ft) if total impervious area (TIA) cover in the watershed is < 6% (roughly corresponding to no more than 15% of the watershed converted to urban land use).

  — Mean annual WLF is very likely (89% of cases measured) to be > 20 cm if TIA in the watershed is > 21% (roughly corresponding to more than 30% of the watershed converted to urban land use).

  — Mean annual WLF is somewhat likely (50% of cases measured) to be > 30 cm (1.0 ft) if TIA in the watershed is > 21% (roughly corresponding to more than 30% of the watershed converted to urban land use).

  — Mean annual WLF is likely (75% of cases measured) to be > 30 cm, and somewhat likely (50% of cases measured) to be 50 cm (20 inches or 1.6 ft) or higher, if TIA in the watershed is > 40% (roughly corresponding to more than 70% of the watershed converted to urban land use).

  — The frequency of stage excursions greater than 15 cm (6 inches or 0.5 ft) above or below pre-development levels is somewhat likely (54% of cases measured) to be more than six per year if the mean annual WLF increases to > 24 cm (9.5 inches or 0.8 ft).

  — The average duration of stage excursions greater than 15 cm above or below pre-development levels is likely (69% of cases measured) to be more than 72 hours if the mean annual WLF increases to > 20 cm.

2. The following hydroperiod limits characterize wetlands with relatively high vegetation species richness and apply to <u>all zones</u> within <u>all wetlands</u> over the <u>entire year</u>. If these limits are exceeded, then species richness is likely to decline. If the analysis described above forecasts exceedences, one or more of the management strategies listed in step 5 should be employed to attempt to stay within the limits.

- Mean annual WLF (and mean monthly WLF for every month of the year) does not exceed 20 cm. Vegetation species richness decrease is likely with: (1) a mean annual (and mean monthly) WLF increase of more than 5 cm (2 inches or 0.16 ft) if pre-development mean annual (and mean monthly) WLF is greater than 15 cm, or (2) a mean annual (and mean monthly) WLF increase to 20 cm or more if pre-development mean annual (and mean monthly) WLF is 15 cm or less.

- The frequency of stage excursions of 15 cm above or below pre-development stage does not exceed an annual average of six.

- The duration of stage excursions of 15 cm above or below pre-development stage does not exceed 72 hours per excursion.

- The total dry period (when pools dry down to the soil surface everywhere in the wetland) does not increase or decrease by more than two weeks in any year.

- Alterations to watershed and wetland hydrology that may cause perennial wetlands to become **vernal** are avoided.

3. The following hydroperiod limit characterizes **priority peat wetlands** (bogs and fens as more specifically defined by the Washington Department of Ecology) and applies to all zones over the entire year. If this limit is exceeded, then characteristic bog or fen wetland vegetation is likely to decline. If the analysis described above forecasts exceedence, one or more of the management strategies listed in step 5 should be employed to attempt to stay within the limit.

- The duration of stage excursions above the pre-development stage does not exceed 24 hours in any year.

- Note: To apply this guideline a continuous simulation computer model needs to be employed. The model should be calibrated with data taken under existing conditions at the wetland being analyzed and then used to forecast post-development duration of excursions.

4. The following hydroperiod limits characterize wetlands inhabited by breeding native amphibians and apply to breeding zones during the period 1 February through 31 May. If these limits are exceeded, then amphibian breeding success is likely to decline. If the analysis described above forecasts exceedences, one or more of the management strategies listed in step 5 should be employed to attempt to stay within the limits.

- The magnitude of stage excursions above or below the pre-development stage does not exceed 8 cm, and the total duration of these excursions does not exceed 24 hours in any 30 day period.

- Note: To apply this guideline a continuous simulation computer model needs to be employed. The model should be calibrated with data taken under existing conditions at the wetland being analyzed and then used to forecast post-development magnitude and duration of excursions.

5. If it is expected that the hydroperiod limits stated above could be exceeded, consider strategies such as:

- Reduction of the level of development;

- Increasing runoff infiltration [Note: Infiltration is prone to failure in many Puget Sound Basin locations with glacial till soils and generally requires **pretreatment** to avoid clogging. In other situations infiltrating urban runoff may contaminate groundwater. Consult the stormwater management manual adopted by the jurisdiction and carefully analyze infiltration according to its prescriptions.];

- Increasing runoff storage capacity; and

- Selective runoff bypass.

6. After development, monitor hydroperiod with a continuously recording level gauge or staff and crest stage gauges. If the applicable limits are exceeded, consider additional applications of the strategies in step 5 that may still be available. It is also recommended that goals be established to maintain key vegetation species, amphibians, or both, and that these species be monitored to determine if the goals are being met.

Guide Sheet 2C: Guidelines for Protection from Adverse Impacts of Modified Runoff Quality Discharged to Wetlands

1. Require effective erosion control at any construction sites in the wetland's drainage catchment.

2. Institute a program of **source control BMPs** to minimize the generation of pollutants that will enter storm runoff that drains to the wetland.

3. Provide a water quality control facility consisting of one or more **treatment BMPs** to treat all urban runoff entering the wetland and designed according to the following criteria:

- The facility should be designed to remove at least 80 percent of the total suspended solids in the runoff.

- If the catchment could generate a relatively large amount of oil (e. g., certain industrial sites, bases handling large vehicles, areas where oil may be spilled or improperly disposed), the facility should include an appropriate oil control device.

- If the wetland is a **priority peat wetland** (bogs and fens as more specifically defined by the Washington Department of Ecology), the facility should include a BMP with the most advanced ability to control nutrients (e. g., an infiltration device, a wet pond or constructed wetland with residence time in the pooled storage of at least two weeks). [Note: Infiltration is prone to failure in many Puget Sound Basin locations with glacial till soils and generally requires **pretreatment** to avoid clogging. In other situations infiltrating urban runoff may contaminate groundwater. Consult the stormwater management manual adopted by the jurisdiction and carefully analyze infiltration according to its prescriptions.] Refer to Appendix E for a comparison of water chemistry conditions in priority peat versus more typical wetlands.

Refer to the stormwater management manual to select and design the facility. Generally, the facility should be located outside and upstream of the wetland and its buffer.

4. Design and perform a water quality monitoring program for priority peat wetlands and for other wetlands subject to relatively high water pollutant loadings. The research results (Horner 1989) identified such wetlands as having contributing catchments exhibiting either of the following characteristics:

- More than 20 percent of the catchment area is committed to commercial, industrial, and/or multiple family residential land uses; or

- The combination of all urban land uses (including single family residential) exceeds 30 percent of the catchment area.

A recommended monitoring program, consistent with monitoring during the research program, is:

- Perform pre-development **baseline sampling** by collecting water quality grab samples in an open water pool of the wetland for at least one year, allocated through the year as follows: November 1-March 31--4 samples, April 1-May 31--1 sample, June 1-August 31--2 samples, and September 1-October 31--1 sample (if the wetland is dry during any period, reallocate the sample(s) scheduled then to another time). Analyze samples for pH; dissolved oxygen (DO); conductivity (Cond); total suspended solids (TSS); total phosphorus (TP); nitrate + nitrite-nitrogen (N); fecal coliforms (FC); and total copper (Cu), lead (Pb), and zinc (Zn). Find the median and range of each water quality variable.

- Considering the baseline results, set water quality goals to be maintained in the post-development period. Example goals are: (1) pH--no more than “x” percent (e. g., 10%) increase (relative to baseline) in annual median and maximum or decrease in annual minimum; (2) DO--no more than “x” percent decrease in annual median and minimum concentrations; (3) other variables --no more than “x” percent increase in annual median and maximum concentrations; (4) no increase in violations of the Washington Administrative Code (WAC) water quality criteria.

- Repeat the sampling on the same schedule for at least one year after all development is complete. Compare the results to the set goals.

If the water quality goals are not met, consider additional applications of the source and treatment controls described in steps 2 and 3. Continue monitoring until the goals are met at least two years in succession.

Note: Wetland water quality was found to be highly variable during the research, a fact that should be reflected in goals. Using the maximum (or minimum), as well as a measure of central tendency like the median, and allowing some change from pre-development levels are ways of incorporating an allowance for variability. Table 14-1 presents data from the wetlands studied during the research program to give an approximate idea of magnitudes and degree of variability to be expected. Nonurbanized watersheds (N) are those that have both < 15% urbanization and < 6% impervious cover. Highly urbanized watersheds (H) are those that have both lost all forest cover and have > 20% impervious cover. Moderately urbanized watersheds (M) are those that fit neither the N nor H category.

Table 14-1. Water quality ranges found in study wetlands.

	N			M			H		
Metric	Median	Mean	Std.Dev./n[a]	Median	Mean	Std.Dev./n[a]	Median	Mean	Dev./n[a]
pH[b]	6.4	6.4	0.5/162	6.7	6.5	0.8/132	6.9	6.7	0.6/52
DO (mg/L)	5.9	5.7	2.6/205	5.1	5.53.6/173	6.3	5.4	2.9/67	
Cond. (µS/cm)	46	73	64/190	160	142	73/161	132	151	86/61
TSS (µg/L)	2.0	4.6	8.5/204	2.8	9.2	22/175	4.0	9.2	15/66
TP (µg/L)	29	52	87/206	70	93	92/177	69	110	234/67
N (µg/L)	112	368	485/206	304	598	847/177	376	395	239/67
FC (no./100mL)	9.0	271	1000/206	46	2665	27342/173	61	969	4753/66
Cu (µg/L)	<5.0	<3.3	>2.7/93	<5.0	<3.7	>1.9/78	<5.0	<4.1	<2.5/29
Pb (µg/L)	1.0	<2.7	>2.8/136	3.0	<3.4	>2.7/122	5.0	<4.5	>4.0/44
Zn (µg/L)	5.0	8.4	8.3/136	8.0	9.8	7.2/122	20	20	17/44

[a] Std. Dev.--standard deviation; n--number of observations.
[b] Values do not apply to priority peat wetlands. The program did not specifically study these wetlands but measured pH in three wetlands with “bog-like” characteristics. The minimum value measured in these wetlands was 4.5, and the lowest median was 4.8; but pH can be approximately 1 unit lower in wetlands of this type. Refer to Appendix E for a comparison of water chemistry conditions in priority peat versus more typical wetlands.

Guide Sheet 2D: Guidelines for the Protection of Specific Biological Communities

1. For wetlands inhabited by breeding native amphibians:

    - Refer to step 4 of Guide Sheet 2B for hydroperiod limit.
    - Avoid decreasing the sizes of the open water and aquatic bed zones.
    - Avoid increasing the channelization of flow. Do not form channels where none exist, and take care that inflows to the wetland do not become more concentrated and do not enter at higher velocities than accustomed. If necessary, concentrated flows can be uniformly distributed with a flow-spreading device such as a shallow weir, stilling basin, or perforated pipe. Velocity dissipation can be accomplished with a stilling basin or rip-rap pad.
    - Limit the post-development flow velocity to < 5 cm/s (0.16 ft/second) in any location that had a velocity in the range 0-5 cm/s in the pre-development condition.
    - Avoid increasing the gradient of wetland side slopes.

2. For wetlands inhabited by forest bird species:

    Retain areas of coniferous forest in and around the wetland as habitat for forest species.

Retain shrub or woody debris as nesting sites for ground-nesting birds and downed logs and stumps for winter wren habitat.

Retain snags as habitat for cavity-nesting species, such as woodpeckers.

Retain shrubs in and around the wetland for protective cover. If cover is insufficient to protect against domestic pet predation, consider planting native bushes such as rose species in the buffer.

3. For wetlands inhabited by **wetland obligate** bird species:

- Retain **forested zones**, sedge and rush meadows, and deep open water zones, both without vegetation and with submerged and floating plants.

- Retain shrubs in and around the wetland for protective cover. If cover is insufficient to protect against domestic pet predation, consider planting native bushes such as rose species in the buffer.

- Avoid introducing **invasive weedy plant species**, such as purple loosestrife and reed canarygrass.

- Retain the buffer zone. If it has lost width or forest cover, consider re-establishing forested buffer area at least 30 meters (100 ft) wide.

- If human entry is desired, establish paths that permit people to observe the wetland with minimum disturbance to the birds.

4. For wetlands inhabited by <u>fish</u>:

- Protect fish habitats by avoiding water velocities above tolerated levels (selected with the aid of a qualified fishery biologist to protect fish in each life stage when they are present), siltation of spawning beds, etc. Habitat requirements vary substantially among fish species. If the wetland is associated with a larger water body, contact the Department of Fisheries and Wildlife to determine the species of concern and the acceptable ranges of habitat variables.

- If stranding of protected commercial or sport fish could result from a structural or hydrologic modification for runoff quantity or quality control, develop a strategy to avoid stranding that minimizes disturbance in the wetland (e. g., by making provisions for fish return to the stream as the wetland drains, or avoiding use of the facility for quantity or quality control during fish presence).

APPENDIX A: INFORMATION NEEDED TO APPLY GUIDELINES

The following information listed for each guide sheet is most essential for applying the Wetlands and Stormwater Management Guidelines. As a start, obtain the relevant soil survey; the National Wetland Inventory, topographic and land use maps, and the results of any local wetland inventory.

Guide Sheet 1

1. Boundary and area of the contributing watershed of the wetland or other landscape unit

2. A complete definition of goals for the wetland and landscape unit subject to planning and management

3. Existing management and monitoring plans

4. Existing and projected land use in the landscape unit in the categories commercial, industrial, multi-family residential, single-family residential, agricultural, various categories of undeveloped, and areas subject to active logging or construction (expressed as percentages of the total watershed area)

5. Drainage network throughout the landscape unit

6. Soil conditions, including soil types, infiltration rates, and positions of seasonal water table (seasonally) and restrictive layers

7. Groundwater recharge and discharge points

8. Wetland category (I - IV in draft Puget Sound Water Quality Authority wetland protection guidelines); designation as rare or irreplaceable. Refer to the Washington Natural Heritage Program data base. If the needed information is not available, a biological assessment will be necessary.

9. Watershed hydrologic assessment

10. Watershed water quality assessment

11. Wetland type and zones present, with special note of estuarine, priority peat system, forested, sensitive scrub-shrub zone, sensitive emergent zone and other sensitive or critical areas designated by state or local government (with dominant plant species)

12 . Rare, threatened, or endangered species inhabiting the wetland

13. History of wetland changes

14. Relationship of wetland to other water bodies in the landscape unit and the drainage network

15. Flow pattern through the wetland

16. Fish and wildlife inhabiting the wetland

17. Relationship of wetland to other wildlife habitats in the landscape unit and the corridors between them

Guide Sheet 2

1. Existing and potential stormwater pollution sources

2. Existing and projected landscape unit land use (see number 4 under Guide Sheet 1)

3. Existing and projected wetland hydroperiod characteristics

4. Wetland bathymetry

5. Inlet and outlet locations and hydraulics

6. Landscape unit soils, geologic and hydrogeologic conditions

7. Wetland type and zones present (see number 11 under Guide Sheet 1)

8. Presence of breeding populations of native amphibian species

9. Presence of forest and wetland obligate bird species

10. Presence of fish species

APPENDIX B: DEFINITIONS

Baseline sampling: Sampling performed to define an existing state before any modification occurs that could change the state.

Bioengineering: Restoration or reinforcement of slopes and stream banks with living plant materials.

Buffer: The area that surrounds a wetland and that reduces adverse impacts to it from adjacent development.

Constructed wetland: A wetland intentionally created from a non-wetland site for the sole purpose of wastewater or stormwater treatment. These wetlands are not normally considered Waters of the United States or Waters of the State.

Degraded (disturbed) wetland (community): A wetland (community) in which the vegetation, soils, and/or hydrology have been adversely altered, resulting in lost or reduced functions and values; generally, implies topographic isolation; hydrologic alterations such as hydroperiod alteration (increased or decreased quantity of water), diking, channelization, and/or outlet modification; soils alterations such as presence of fill, soil removal, and/or compaction; accumulation of toxicants in the biotic or abiotic components of the wetland; and/or low plant species richness with dominance by invasive weedy species.

Enhancement: Actions performed to improve the condition of an existing degraded wetland, so that functions it provides are of a higher quality.

Estuarine wetland: Generally, an eelgrass bed; salt marsh; or rocky, sandflat, or mudflat intertidal area where fresh and salt water mix. (Specifically, a tidal wetland with salinity greater than 0.5 parts per thousand, usually semi-enclosed by land but with partly obstructed or sporadic access to the open ocean).

Forested communities (wetlands): In general terms, communities (wetlands) characterized by woody vegetation that is greater than or equal to 6 meters in height; in these guidelines the term applies to such communities (wetlands) that represent a significant amount of tree cover consisting of species that offer wildlife habitat and other values and advance the performance of wetland functions overall.

Functions: The ecological (physical, chemical, and biological) processes or attributes of a wetland without regard for their importance to society (see also Values). Wetland functions include food chain support, provision of ecosystem diversity and fish and wildlife habitat, flood flow alteration, groundwater recharge and discharge, water quality improvement, and soil stabilization.

Hydrodynamics: The science involving the energy and forces acting on water and its resulting motion.

Hydroperiod: The seasonal occurrence of flooding and/or soil saturation; encompasses the depth, frequency, duration, and seasonal pattern of inundation.

Invasive weedy plant species: Opportunistic species of inferior biological value that tend to out-compete more desirable forms and become dominant; applied to non-native species in these guidelines.

Landscape unit: An area of land that has a specified boundary and is the locus of interrelated physical, chemical, and biological processes.

Modification, Modified (wetland): A wetland whose physical, hydrological, or water quality characteristics have been purposefully altered for a management purpose, such as by dredging, filling, forebay construction, and inlet or outlet control.

On-site: An action (here, for stormwater management purposes) taken within the property boundaries of the site to which the action applies.

Polishing: Advanced treatment of a waste stream that has already received one or more stages of treatment by other means.

Pre-development, post-development: Respectively, the situation before and after a specific stormwater management project (e. g., raising the outlet, building an outlet control structure) will be placed in the wetland or a land use change occurs in the landscape unit that will potentially affect the wetland.

Pre-treatment: An action taken to remove pollutants from runoff before it is discharged into another system for additional treatment.

Priority peat systems: Unique, irreplaceable fens that can exhibit water pH in a wide range from highly acidic to alkaline, including fens typified by *Sphagnum* species, *Rhododendron groenlandicum* (Labrador tea), *Drosera rotundifolia* (sundew), and *Vaccinium oxycoccos* (bog cranberry); marl fens; estuarine peat deposits; and other moss peat systems with relatively diverse, undisturbed flora and fauna. Bog is the common name for peat systems having the *Sphagnum* association described, but this term applies strictly only to systems that receive water income from precipitation exclusively.

Rare, threatened, or endangered species: Plant or animal species that are regional relatively uncommon, are nearing endangered status, or whose existence is in immediate jeopardy and is usually restricted to highly specific habitats. Threatened and endangered species are officially listed by federal and state authorities, whereas rare species are unofficial species of concern that fit the above definitions.

Redevelopment: Conversion of an existing development to another land use, or addition of a material improvement to an existing development.

Regional: An action (here, for stormwater management purposes) that involves more than one discrete property.

Restoration: Actions performed to reestablish wetland functional characteristics and processes that have been lost by alterations, activities, or catastrophic events in an area that no longer meets the definition of a wetland.

Source control best management practices (BMPs): Actions that are taken to prevent the development of a problem (e. g., increase in runoff quantity, release of pollutants) at the point of origin.

Stage excursion: A post-development departure, either higher or lower, from the water depth existing under a given set of conditions in the pre-development state.

Structure: The components of an ecosystem, both the abiotic (physical and chemical) and biotic (living).

Treatment best management practices (BMPs): Actions that remove pollutants from runoff through one or more physical, chemical, biological mechanisms.

Unusual biological community types: Assemblages of interacting organisms that are relatively uncommon regionally.

Values: Wetland processes or attributes that are valuable or beneficial to society (also see Functions). Wetland values include support of commercial and sport fish and wildlife species, protection of life and property from flooding, recreation, education, and aesthetic enhancement of human communities.

Vernal wetland: A wetland that has water above the soil surface for a period of time during and/or after the wettest season but always dries to or below the soil surface in warmer, drier weather.

Wetland obligate: A biological organism that absolutely requires a wetland habitat for at least some stage of its life cycle.

Wetlands: Lands transitional between terrestrial and aquatic systems that have a water table usually at or near the surface or a shallow covering of water, hydric soils, and a prevalence of hydrophytic vegetation.

APPENDIX C: REFERENCES

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APPENDIX D: NATIVE AND RECOMMENDED NONINVASIVE PLANT SPECIES FOR WETLANDS IN THE PUGET SOUND BASIN

Caution: Extracting plants from an existing wetland donor site can cause a significant negative effect on that site. It is recommended that plants be obtained from native plant nursery stocks whenever possible. Collections from existing wetlands should be limited in scale and undertaken with care to avoid disturbing the wetland outside of the actual point of collection. Plant selection is a complex task, involving matching plant requirements with environmental conditions. It should be performed by a qualified wetlands botanist. Refer to Restoring Wetlands in Washington by the Washington Department of Ecology for more information.

Plants preferred in Puget Sound Basin freshwater wetlands

Open water zone:
Potamogeton species (pondweeds)
Nymphaea odorata (pond lily)
Brasenia schreberi (watershield)
Nuphar luteum (yellow pond lily)
Polygonum hydropiper (smartweed)
Alisma plantago-aquatica (broadleaf water plantain)
Ludwigia palustris (water purslane)
Menyanthes trifoliata (bogbean)
Utricularia minor, *U. vulgaris* (bladderwort)

Emergent zone:
Carex obnupta, *C. utriculata, C. arcta, C. stipata, C. vesicaria C. aquatilis, C. comosa, C. lenticularis* (sedge)
Scirpus atricinctus (woolly bulrush)
Scirpus microcarpus (small-fruited bulrush)
Eleocharis palustris, E. ovata (spike rush)
Epilobium watsonii (Watson's willow herb)
Typha latifolia (common cattail) (Note: This native plant can be aggressive but has been found to offer certain wildlife habitat and water quality improvement benefits; use with care.)
Veronica americana, V. scutellata (American brookline, marsh speedwell)
Mentha arvensis (field mint)
Lycopus americanus, L. uniflora (bugleweed or horehound)
Angelica species (angelica)
Oenanthe sarmentosa (water parsley)
Heracleum lanatum (cow parsnip)
Glyceria grandis, G. elata (manna grass)
Juncus acuminatus (tapertip rush)

Juncus ensifolius (daggerleaf rush)
Juncus bufonius (toad rush)
Mimulus guttatus (common monkey flower)

Scrub-shrub zone:
Salix lucida, S. rigida, S. sitchensis, S. scouleriana, S. pedicellaris (willow)
Lysichiton americanus (skunk cabbage)
Athyrium filix-femina (lady fern)
Cornus sericea (redstem dogwood)
Rubus spectabilis (salmonberry)
Physocarpus capitatus (ninebark)
Ribes species (gooseberry)
Rhamnus purshiana (cascara)
Sambucus racemosa (red elderberry) (occurs in wetland-upland transition)
Loniceria involucrata (black twinberry)
Oemleria cerasiformis (Indian plum)
Stachys cooleyae (Stachy's horsemint)
Prunus emarginata (bitter cherry)

Forested zone:
Populus balsamifera, ssp. *trichocarpa* (black cottonwood)
Fraxinus latifolia (Oregon ash)
Thuja plicata (western red cedar)
Picea sitchensis (Sitka spruce)
Alnus rubra (red alder)
Tsuga heterophylla (hemlock)
Acer circinatum (vine maple)
Maianthemum dilatatum (wild lily-of-the-valley)
Ivzula parviflora (small-flower wood rush)
Torreyochloa pauciflora (weak alkaligrass)
Ribes species (currants)

Bog:
Sphagnum species (sphagnum mosses)
Rhododendron groenlandicum (Labrador tea)
Vaccinium oxycoccos (bog cranberry)
Kalmia microphylla, ssp. *occidentalis* (bog laurel)

Exotic plants that should not be introduced to existing, created, or constructed Puget Sound Basin freshwater wetlands

Hedera helix (English ivy)
Phalaris arundinacea (reed canarygrass)
Lythrum salicaria (purple loosestrife)
Iris pseudacorus (yellow iris)
Ilex aquifolia (holly)
Impatiens glandulifera (policeman’s helmet)
Lotus corniculatus (birdsfoot trefoil)

Lysimachia thyrsiflora (tufted loosestrife)
Myriophyllum species (water milfoil, parrot's feather)
Polygonum cuspidatum (Japanese knotweed)
Polygonum sachalinense (giant knotweed)
Rubus discolor (Himalayan blackberry)
Tanacetum vulgare (common tansy)

Native plants that should not be introduced to existing, created, or constructed Puget Sound Basin freshwater wetlands

Potentilla palustris (Pacific silverweed)
Solarum dulcimara (bittersweet nightshade)
Juncus effusus (soft rush)
Conium maculatum (poison hemlock)
Ranunculus repens (creeping buttercup)

APPENDIX E: COMPARISON OF WATER CHEMISTRY CHARACTERISTICS IN *SPHAGNUM* BOG AND FEN VERSUS MORE TYPICAL WETLANDS

Water Quality Variable	Typical Wetlands	*Sphagnum* Bogs and Fens
pH	6 - 7	3.5 - 4.5
Dissolved oxygen (mg/L)	4 - 8	Shallow surface layer oxygenated, anoxic below
Cations	Divalent Ca, Mg common	Divalent Ca, Mg uncommon; Univalent Na, K predominant
Anions	HCO_3^-, CO_3^{2-} predominant	Cl^-, SO_4^{2-} predominant; almost no HCO_3^-, CO_3^{2-} (organic acidsform buffering system)
Hardness	Moderate	Very low
Total phosphorus (µg/L)	50 - 500	5 - 50
Total Kjeldahl nitrogen (µg/L)	500 - 1000	~ 50

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