

AN ABSTRACT OF THE THESIS OF

Satomi Inahara for the degree of Master of Science in Bioresource Engineering presented on July 29, 2002.

Title: Assessing Cumulative Influences of Watershed-Scale Landuses on Reed Canarygrass (*Phalaris arundinacea* L.) Abundance.

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Abstract approved:



John P. Bolte

The West Eugene Wetlands, Eugene, Oregon, which provide habitat for number of endemic and endangered plant species, are currently threatened by a Reed canarygrass (*Phalaris arundinacea* L.) invasion. This study addresses the hypothesis that *Phalaris* spread can be explained in part by using watershed-scale landuse patterns as surrogates of the water quality and hydrologic regime in the associated watersheds. A multiple regression model was constructed to test the effects of watershed-scale landuses on *Phalaris* abundance. The spatially referenced landuse attributes were estimated using accumulated inverse-squared flow path distances between each wetland and cells in a grid-based representation of landuse/cover data in a Geographic Information System. The locally measured physical and biological characteristics of wetlands were also incorporated into the model to adjust for site differences. Bias in the estimated parametric coefficients were found to be negligible using 1,000 iterations of a bootstrap re-sampling technique. Bias-corrected-and-accelerated intervals also supported the significance of these estimates.

The inferential model, consisting of the percentage open water cover in the wetlands, an indicator of highly disturbed site and watershed-scale urban and forest landuses, explained 67 percent variation of *Phalaris* abundance. There was convincing evidence that the cumulative watershed-scale landuse patterns affected *Phalaris* abundance (p -value < 0.001 ; extra-sum-of-squares F-test). In particular, urbanization was strongly associated with greater *Phalaris* abundance (p -value = 0.00001, two-sided t-test). The effects of watershed-scale landuse appeared to be persistent in the inside of the wetland edges. Implications for wetland management were discussed. Alternative restoration strategy addressing plant diversity was also proposed.

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Assessing Cumulative Influences of Watershed-Scale Landuses on Reed Canarygrass
(*Phalaris arundinacea* L.) Abundance

by
Satomi Inahara

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To my grandparents

Assessing Cumulative Influences of Watershed-Scale Landuses on Reed
Canarygrass (*Phalaris arundinacea* L.) Abundance

CHAPTER 1: INTRODUCTION

Protecting wetland native plant communities from invasive species has been a common problem throughout North America (Galatowitsch et al. 1999). Reed canarygrass (*Phalaris arundinacea* L.) is typical of invasive species that extirpate resident plant species and create monotypic stands (Apfelbaum and Sams 1987), potentially causing homogeneous vegetation in native wetlands (Lesica 1997; Wetzel and van der Valk 1998). *Phalaris* stands are highly persistent and difficult to eradicate, thereby seriously hindering wetland restoration. Although various strategies for controlling *Phalaris* have been attempted (for example, chemical control, mowing, burning, shading, flooding, etc.), currently none of these strategies are accepted as effective and practical without impacting native plant communities.

While surrounding landuse development has been related to the invasion of non-native species through changes in hydrology and chemical inputs, *Phalaris* management outside the wetland boundary has received little attention. Particularly, when considering restoration opportunities where wetland conservation and the pressures of urbanization need to be balanced, it is important to understand the relationships between *Phalaris* abundance in wetlands and surrounding landuse patterns.

To date, few studies have employed spatially referenced data to investigate the influence of surrounding landuses on wetland flora (e.g., Magee et al. 1999; Galatowitsch et al. 2000). None have specifically focused on *Phalaris* abundance. If surrounding landuses affect *Phalaris* invasion, wetland restoration solely based upon on-site management in wetlands may be inadequate. The investment in time and money necessary to eradicate *Phalaris* on-site and mitigate its impact on native species would then be wasted. Furthermore, much of the past research that quantitatively evaluated the landuse attributes focused on variation of landuse intensity within certain distances around the wetlands (e.g., Magee et al. 1999; Galatowitsch et al. 2000). The drawback of this

approach is that it ignores the impact of hydrology-related inputs from upstream drainage networks at the catchment scale, thereby failing to capture the potentially important influence of landuses outside of the delineated study area (Osborne and Wiley 1988). Therefore, there is a need for a watershed perspective that covers entire drainage areas. This study addresses the hypothesis that *Phalaris* spread can be explained in part by using landuse patterns as surrogates of the water quality and hydrologic regime in the associated watershed.

The goal of this study was to construct a statistical model describing the relationship between *Phalaris* abundance and spatially explicit landuse patterns at watershed-scales. The spatially referenced variables were quantified in a Geographic Information System (GIS) using remotely sensed wetland and landuse data. A multiple regression approach was employed to integrate these spatial variables with locally measured physical and biological characteristics of wetlands. This model was used to explore the role of the landuse patterns in predicting *Phalaris* abundance, and to examine possible mechanisms for its invasion and dominance.

CHAPTER 2: LITERATURE REVIEW

2.1 CHARACTERISTICS OF AN IDEAL INVADER IN A VARIABLE ENVIRONMENT

Some studies have attributed *Phalaris* spread to its vegetative vigor and plant architecture (Figiel et al. 1995; Lesica 1997; Wetzel and van der Valk 1998). The species can extract nutrients and moisture at high rates to form a tall, dense and large canopy that shades out neighboring species (Grime et al. 1988). While in its dormant period, *Phalaris*'s accumulated litter suppresses growth of coexisting natives (Lesica 1997; Wetzel and van der Valk 1998). In addition, recent studies have shown that the morphological plasticity and genetic variability of *Phalaris* may be of great importance in its early invasion process (Morrison and Molofsky 1999). Plasticity in growth behavior may allow *Phalaris* stands to readily respond to changes in physical and biological conditions (Morrison and Molofsky 1998). Genetic variability within a *Phalaris* population may also provide a wide range of strategies to deal with a variable environment, thereby increasing chances for the population to survive as a whole.

Within clones randomly selected from a population in Vermont, Morrison and Molofsky (1998) tested morphological plasticity (shoot to root biomass ratio and tiller formation) of *Phalaris* in response to density of neighboring plant communities. They found that *Phalaris* allocated more biomass to the roots in cases where the plots were more densely occupied by other species, suggesting that this sequestering of biomass provides *Phalaris* a competitive ability over neighboring species in succeeding growing seasons. They further demonstrated that plasticity in growth behavior occurred in *Phalaris* clones common in the genotype.

Growth behavior may also differ among genotypes. *Phalaris* is likely a hybrid of native populations and cultivars (Merigliano and Lesica 1998), and therefore high genetic variability can be expected (Comes 1971). Using isozyme electrophoresis, Morrison and Molofsky (1998) determined that 10 out of 16 clones from a pasture in Vermont were genetically distinctive. While overall relationships between growth behavior (biomass allocation and tiller formation) and the density of neighboring plants were the same for all

genotypes, the sensitivities of these behaviors to density of neighboring vegetation significantly differed among different genotypes (Morrison and Molofsky 1998; Morrison and Molofsky 1999). Some genotypes did not respond well to changes in neighboring vegetation, but others opportunistically invaded into new habitats where interspecific competition was low.

In other studies, biomass allocation to shoots and tiller formation were positively associated with changes in water depths and nitrate availability (Conchou and Fustec 1988; Rice and Pinkerton 1993). It was suggested that *Phalaris* allocated more biomass to shoots to tolerate standing water and to out-compete other species for sun light. In these studies, it is uncertain if the growth plasticity of *Phalaris* is attributable to the difference in genotypes or to the species trait regardless of genotype, because genetic variability was not controlled in both experiments. In field observations, Kentula and Magee (2001) and Owen (1999) found that *Phalaris* inhabited in a wider range of water levels than for native species, suggesting a competitive ability over native species in wetlands with variable hydrology.

Thus, growth plasticity and genetic variability can give *Phalaris* multiple strategies with which to deal with variable physical and biological conditions, increasing its likelihood of survival over native plant communities. *Phalaris* is an ideal invader because its opportunistic growth in variable environments allows the species to successfully invade new habitats.

2.2 ENVIRONMENTAL FACTORS PROMOTING *PHALARIS* INVASION

Davis et al. (2000) generalized a mechanism of plant invasion based on a theory of resource competition between alien and resident species. They asserted that resource (e.g., nutrients, light, and space) enrichment and a reduction of resource uptake by resident species could increase invasion potential of alien species. Plant communities become most vulnerable to the invasion when both resource enrichment and reduction of interspecific competition take place simultaneously. In a field experiment in the United Kingdom, Burke and Grime (1996) demonstrated that space availability encouraged non-native species invasion and that soil fertility magnified this invasion. This result appears to

conform to temperate North American wetlands. Galatowitsch et al. (1999) reviewed the potential causal factors of the spread of four invasive taxa, including *Phalaris*, and concluded that nutrient enrichment and availability of bare ground exposed by human disturbances play important roles in the spread of these taxa. They further listed altered hydrology and increased salinity as environmental factors enhancing the non-native species invasion. These altered physical conditions are presumably unsuitable for native species and may decrease resource competition among species, thereby increasing resource availability for non-native species and promoting the invasion (Morrison and Molofsky 1998).

Accordingly, *Phalaris* is commonly found in ditches, waterways, and roadsides, which are associated with high disturbance (Comes 1971). Such physically disturbed sites are ideal habitats for *Phalaris* because establishment space is often available, or competitive ability of resident species may be low. In addition, it has been shown that in situations of nutrient enrichment *Phalaris* grows aggressively (Figiel et al. 1995; Wetzel and van der Valk 1998; Green and Galatowitsch 2001) and will invade even under shade (Maurer and Zedler 2002). For example, in a study in the Pacific Northwest, a *Phalaris* monoculture has been reported in a nitrogen-enriched wetland (Cooke and Azous 1993).

Thus, successful *Phalaris* invasion appears to be associated with nutrient enrichment and physical disturbances that result in open areas. Even if resource release is ephemeral, *Phalaris* invasion may be promoted because of its ability to capture resources whenever they are available. In addition, physical conditions such as altered hydrology and degraded water quality that potentially reduce competitive ability of native species may also facilitate *Phalaris* invasion. All these abiotic conditions likely occur in disturbed landscapes, where physical conditions such as timing and magnitude of inflows and nutrient fluxes can vary due to altered watershed hydrology and various pollution sources (Ehrenfeld and Schneider 1990; Shaffer and Kentula 1997). Given species characteristics that capitalize on situations of nutrient enrichment and opportunistic occupation of available space, wetlands in human impacted landscapes provide ideal habitats for *Phalaris* spread.

2.3 EFFECTS OF LANDSCAPE MODIFICATIONS ON WETLAND VEGETATION THROUGH DEGRADED WATER QUALITY AND HYDROLOGY

Wetlands are often the interface between water bodies and terrestrial systems. They often depend upon inflows from uplands and, thus, can be sensitive to the amount, timing, and water quality of inflows that are potentially influenced by surrounding landuse activities (Mitsch and Gosselink 1993; Poiani and Bedford 1995). In previous studies, landscape modifications have been consistently linked to alterations of water quality and hydrology within wetlands. For example, agricultural or urban lands were found to be responsible for elevated nutrient concentrations in wetlands (Ehrenfeld and Schneider 1990; Detenbeck et al. 1996; Crosbie and Chow-Fraser 1999). Absence of forested lands and increased extent of agricultural lands in watersheds were also positively correlated with suspended solids in wetland surface waters (Ludwa 1994; Crosbie and Chow-Fraser 1999). Moreover, Crosbie and Chow-Fraser (1999) attributed the agricultural lands and urban lands to high concentrations of herbicide and synthetic hydrocarbons found in wetlands.

Wetland hydrology may also be subject to landscape changes. Some studies showed that urban wetlands exhibited wider ranges of water table fluctuations in a year, compared with those of intact wetlands (Ehrenfeld and Schneider 1990; Cooke and Azous 1993; Owen 1999). During storm events, Shaffer and Kentula (1997) found that hydrologic alteration in urban wetlands could be pronounced, and was exhibited as a flashy hydrograph with a rapid rise and recession.

Altered wetland hydrology may in turn intensify the impacts of chemical inputs on wetland water quality. Owen (1999) found, over a 140 year-period of watershed development, that overland flow in a wetland increased while groundwater inputs and overbank flows from a stream channel decreased with increasing development. She also found that overland flow contained high sodium and trace elements whereas ground water was calcium and magnesium rich, suggesting that a change in wetland water chemistry probably also occurred.

Furthermore, the altered status of wetland water quality and hydrology may have an influence on the physiology of individual plant species and eventually change the community structure. Many studies have reported vegetation changes in human-impacted

landscapes in relation to the alterations of wetland water quality and hydrology. For example, degraded wetland water quality was correlated with a dominance of non-native species over native herbaceous species in cedar-dominated wetlands in New Jersey (Ehrenfeld and Schneider 1990) and a reduction of species richness of submerged vegetation in marshes in Ontario (Crosbie and Chow-Fraser 1999). In a study of species transitions over 13 years in western Ireland, Van Groenendael et al. (1993) reported that a sharp increase in orthophosphate concentration in wetlands may have caused a loss of rare species and a reduction in species richness. The species change was a shift from oligotrophic to eutrophic taxa. At a calcareous fen in western Massachusetts, there were significant impacts of high salt concentrations on species richness, evenness, and total plant cover (Richburg et al. 2001). Furthermore, higher sedimentation rates appeared to increase abundance of sediment-tolerant species including *Phalaris* in wetlands in central Pennsylvania (Wardrop and Brooks 1998). In southern Wisconsin, a dramatic hydrologic alteration was associated with an increase in introduced species (*Phalaris*, *Typha angustifolia*, and *Typha X. glauca*) abundance at the expense of native sedge species (Owen 1999). Finally, in Puget Sound wetlands in Washington, species richness was negatively correlated with high annual water level fluctuations during the early growing season (Cooke and Azous 1993). They also found that the most abundant species of all the herbaceous taxa was *Phalaris*.

Other studies focused on plant species composition as an indicator of wetland degradation, due to landscape modifications. In wet meadows in Minnesota, Galatowitsch et al. (2000) found that the abundance of native graminoid and herbaceous perennials was negatively correlated with increases in the area of adjacent agricultural and urban lands. They also found that some introduced species, including *Phalaris*, appeared to have replaced the native plant community in stormwater-impacted wetlands. Similarly, in urban wetlands in Oregon, introduced species abundance was significantly greater in the wetlands near agriculture and commercial lands, as compared to those surrounded by undeveloped lands (Magee et al. 1999). The most commonly found introduced species in this study was *Phalaris*.

Thus, landscape modifications and associated alterations of wetland water quality and hydrology can be important forcing functions of changes in wetland vegetation. The

floristic responses appear to be the form of increased non-native species abundance, reduced native species abundance or lowered species richness, and *Phalaris* invasion.

2.4 ASSESSMENT OF LANDSCAPE MODIFICATIONS ON WETLAND VEGETATION

Past studies measured stressor gradients of spatially configured landuse types to assess the impacts of landscape modifications on wetland vegetation. A watershed-scale approach was generally not considered. Some studies exclusively quantified the landuse stressors but constrained their assessments to certain distances around wetland boundaries (Wardrop and Brooks 1998; Galatowitsch et al. 2000). Ehrenfeld and Schneider (1990) qualitatively described the variation in landuse stressors based upon the wetlands' proximity to roads, housings, and storm runoff pipes. Only Crosbie and Chow-Fraser (1999) applied a watershed-scale analysis, enumerating the landuse attributes as percentage cover of different landuse types within a watershed.

While the spatial scale studies that compared multiple sites successfully correlated landuse impacts with wetland vegetation, a gradient analysis constrained to a single wetland was found to be inconclusive. Richburg et al. (2001) explored the relationship between salt inputs from a highway and the distribution of a salt-loving invasive species (*Phragmites australis*) by constructing isocline maps of salt concentrations at various distances from the highway. Although they found changes in the community structure along the salt gradients, the relationship between *Phragmites* abundance and salt concentrations were not established.

Effects of landscape modifications may also be detected as a temporal trend in species composition within a single wetland based on historical vegetation inventories or maps (Prach 1993; Van Groenendael et al. 1993; Owen 1999). Although these studies demonstrated species transitions over various periods of time (from 13 to 140 years), it is difficult to relate the species transition to landuse impacts. To ascertain the effects of landuse impacts, these studies further explored the histories of landuse development or performed gradient analyses within the wetlands. Owen (1999) reconstructed the past hydrologic condition of a wetland, using the Soil Conservation Service rainfall-runoff

model and historic driller's logs. She also examined gradients of water levels and some water quality parameters within the wetland. Prach (1993) also quantified gradients of soil moisture and nitrogen content, using indicator plant species along the distance from a fish-pond that was considered to be source of human disturbance. Both studies reasonably interrelated the temporal dynamics of wetland vegetation with landscape modifications. However, as in most of the aforementioned studies, the assessment of landuse impacts was constrained either to the site-scale or to adjacent landuses.

In a coastal region of Ireland, Van Groenendael et al. (1993) employed both temporal and spatial scale analyses. They identified species transitions over time and compared the pattern of transition in inland bogs and lakeshore marshes that presumably differ in the magnitude of landuse impacts. The inland bogs were assumed to lack a permanent hydrologic connection with their uplands, and therefore were less sensitive to landuse development than the lakeshore communities were. Historical numbers of sheep and fertilizer use were used to relate the nutrient enrichment in the wetlands to the landscape modifications. Accordingly, the determination of landuse impacts was largely qualitative.

For both spatial and temporal scale analyses, factors key to identifying a correlation between landscape modifications and changes in wetland vegetation may be the sensitivity of the floristic response and the variability of the landuse stressor. In all studies reviewed, the landuse impacts became apparent when the floristic responses were defined in terms of abundances of native or non-native species assemblages, eutrophic taxa, sediment tolerant taxa, or submersed plants. Each response variable is a particular set of plant species, which is likely to be sensitive to altered physical conditions in wetlands (Keddy et al. 1993). For example, while Richburg et al. (2001) did not find the evidence of *Phragmites* spread for 13 years in relation to the salt inputs, Van Groenendael et al. (1993) demonstrated the species transition for the same time interval in which agricultural practice have been progressively becoming intensive. Van Groenendael et al. (1993) grouped multiple plant species as 'rare species', 'eutrophic taxa', and 'oligotrophic taxa'. This grouping allowed detection of the response to nutrient enrichment. On the other hand, the study of Richburg et al. (2001) was inconclusive possibly due to the lag-time of the *Phragmites* response or the presence of other competitive species. The time since *Phragmites* established on the site may not be long enough for *Phragmites* to spread in

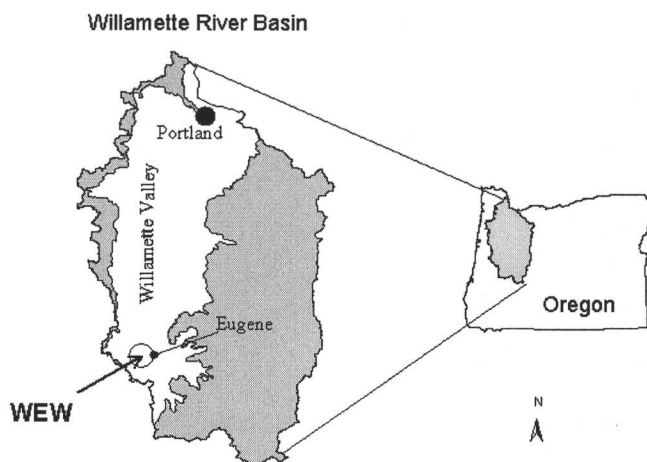
response to the salt gradients. The presence of other invasive species, which were not evaluated in their report, may have suppressed the *Phragmites* invasion. Moreover, the scale of salt gradients may be too fine, i.e., spatially constrained to a single wetland, to detect the *Phragmites* response. Considering that Richburg et al. (2001) reported effects of salt inputs on the community structure, the lack of sensitivity in *Phragmites* response may be the case.

In summary, while a temporal scale analysis demonstrates the temporal dynamics of wetland vegetation, determining the effect of landuse stressors is often either limited to the site-scale or is qualitative. In addition, it may be difficult to elucidate the effects of landscape modifications unless a control site is used to prove that the observed species transitions differ from natural succession. In contrast, a spatial scale analysis can be quantitative and can consider the influence of entire watershed. When the assessment is aimed at comparing the impacts among multiple sites, a spatial scale analysis is the approach of choice.

CHAPTER 3: SITE DESCRIPTION

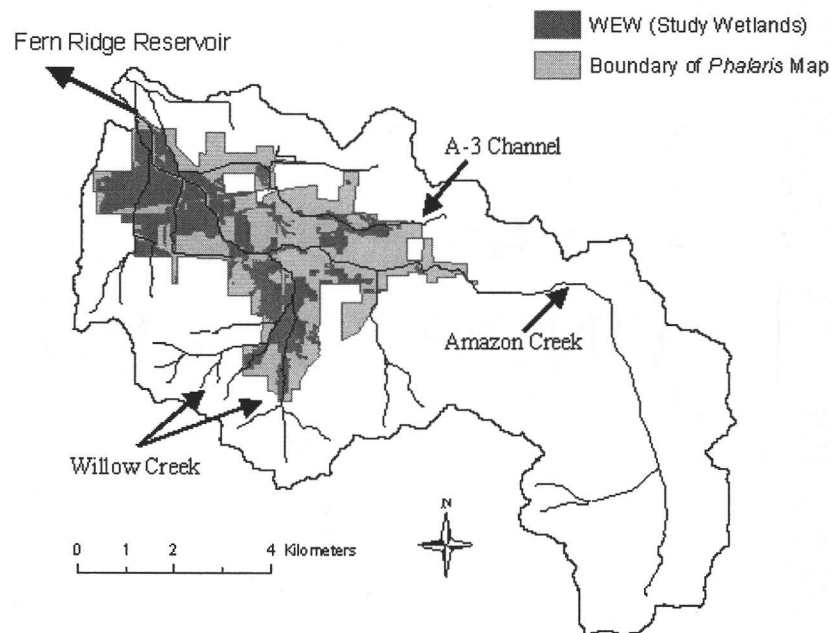
The wetlands studied were the West Eugene Wetlands (WEW), located west of downtown Eugene in the Willamette Valley, Oregon, U.S.A. (Figure 1). The city of Eugene has a population of 130,000, and is the second largest city in Oregon. The Willamette Valley is an area of low relief that is one of the three ecoregions of the Willamette River basin (Omernik and Gallant 1986). The Valley extends from south of Eugene north to the junction of the Willamette River with the Columbia River near Portland. It is bounded by the Cascade Mountains on the east and the Coast Range on the west. The general climate of the Valley is a temperate marine climate characterized by dry summers and mild wet winters. The mean annual precipitation was 122 cm based on the monthly means from the years 1952 through 2000 (at the Eugene Weather Service Operated Airport Station, Latitude 44°07'W, Longitude 123°13'E, Elevation 110 m) (Oregon Climate Service <http://www.ocs.orst.edu>). The wet season, from November through January, accounts for 49 percent of the mean annual precipitation, as opposed to only 2.7 percent occurring in July and August.

Figure 1: Location of Willamette Valley and West Eugene Wetlands (WEW).



The Upper Amazon Creek watershed, which encompasses an area of roughly 73 km², defines the drainage boundary of this study (Figure 2). Amazon Creek flows generally northwards from its origin in the south end of Eugene to the Fern Ridge Reservoir. Residential and industrial areas of Eugene occupy the eastern portions of the watershed. Agricultural lands occur in relatively low gradient landscapes, and forest fringes the upstream reaches.

Figure 2: Map showing Upper Amazon Creek watershed, WEW (study wetlands), and the three main tributaries.



Dam construction on the upper Willamette River has disconnected hydrological connections between the Upper Amazon Creek and the Willamette River (LCOG 1991). Intensive agricultural use and urban infrastructure development since the 1930s have also modified the local drainage through stream channelization, construction of ditches, and diversions. Of the three major tributaries of the Upper Amazon Creek watershed (Amazon

Creek, A-3 Channel, and Willow Creek, Figure 2), only the upper portion of Willow Creek remains in a relatively natural condition.

WEW are the jurisdictional wetlands initially designated in the West Eugene Wetlands Plan (WEWP). WEWP is Oregon's first wetland conservation plan. It was developed to ease the conflict between landuse development and wetland protection, while meeting federal and state wetland law and accommodating increasing local awareness of environmental protection (LCOG 1992). WEWP was adopted by the Eugene City Council and the Lane County Board of Commissioners in 1992 (LCOG 1992) and later was approved by Oregon Division of State Lands (ODSL) and the U. S. Army of Corps of Engineers in 1994 (ODSL 1994; US ACE and US EPA 1994). WEWP identifies the wetlands for protection or development based upon an evaluation of a wetland's functions including flood control, runoff treatment, and maintenance of biodiversity. To compensate for the loss of the wetlands due to development proposed under the WEWP, disturbed wetlands are being restored (COE PWE 1999a). The restoration efforts are primarily excavations of fill materials, construction of water control structures, and seeding and planting of native plants. Monitoring of water levels and vegetation establishment is being done to assess restoration success. WEWP has been amended to include more wetlands. The latest wetland inventory map was completed as the WEW Conservation Plan Inventory (WEW CPI) in 2000 (LCOG 2000). More detailed information on WEWP and its associated restoration projects is available at <http://www.ci.eugene.or.us/wewetlands/default.htm>.

WEW was chosen for this study because detailed site descriptions and geographic information system (GIS) data were available. Descriptions of landuse activities in the vicinity of the wetlands and a spatially explicit map of *Phalaris* distribution across WEW were crucial for this study. The study wetlands were 522 ha of palustrine wetlands (Cowardin et al. 1979), selected from WEW CPI within the boundary of the *Phalaris* map (Figure 2), which was constructed from an extensive *Phalaris* survey conducted by Haney (2000). The study wetlands, including recently restored wetlands from former agricultural fields, consist of wet prairie (51 percent), forested wetlands (10 percent), open water or marsh (8 percent), and small portions of scrub or shrub-dominant wetlands (1 percent) (LCOG 2000). Twenty-nine percent of the wetlands remain as cultivated or abandoned agricultural fields. The status of the remainder (0.3 percent) is unknown. Since the study

wetlands include 87 percent of the designated wetlands in the WEW CPI, the study wetlands represent nearly a census of WEW.

Historically, WEW were seasonally inundated by the Willamette River to the depth of three feet (Savonen 1988). Wet prairies occupied the large flat areas, and shrub-dominant and emergent wetlands fringed small streams and sloughs. Forested wetlands were also found along the larger stream channels. Prior to European settlement, the Kalapuya Indians periodically burned the wet prairies, and it is thought that the fire disturbance maintained the wet prairies as grasslands, inhibiting succession to forested wetlands. However, since the 1930s conversion to agricultural uses, grazing, and drainage have heavily impacted WEW (LCOG 1999). The remnant wetlands are bisected by paved roads and are surrounded by agricultural fields or residential areas.

Although the depth of seasonal inundation was dramatically reduced due to the drainage improvement (Savonen 1988), interactions between soils and hydrology and their seasonal patterns are still important characteristics of WEW. Concentrated rainfalls during winters dominate the WEW water budget, followed by surface and shallow sub-surface runoff, stream water advection and overbank flooding from adjacent stream channels (LCOG 1991). Major groundwater sources appear to be localized in shallow subsurface storage because thick layers of poorly drained soils of the Dayton and Natroy series, which occupy 85 percent of WEW soils, confine the regional aquifer in the older alluvium (LCOG 1991; USDA NRCS National Survey Center 1995). The shallow subsurface storage exists as a Quaternary alluvial aquifer or perched water table forming when these soil layers intercept infiltrating rainfall (William 1987; SRI 1989). Ground water inputs may become a part of the WEW water balance when concentrated precipitation recharges the shallow ground water storage in winters. Thus, because of the soil properties and the seasonally variable rainfall, WEW experiences wet and dry cycle each year. The water table rises above the soil layers and creates standing water in the wetlands in the winter. In contrast, the wetland soils dry and shrink in the summer, creating cracks in the soil surface.

Because of the seasonality of soil and water conditions, WEW can provide habitats for diverse plant communities. A number of endemic plant species that only occur in the Willamette Valley wet prairies have been found in WEW (COE <http://www.ci.eugene.or.us/wewetlands/plants.htm>; LCOG 1992). WEW and their associated uplands include over 190 native plant species, including tufted hairgrass

(*Deschampsia cespitosa*), Douglas' spiraea (*Spiraea douglasii*), Oregon ash (*Fraxinus latifolia*), sedges and rushes (*Carex sp.* and *Juncus sp.*). At least eight state- or federal-listed plants and animal species occur in WEW, including endangered, threatened, and proposed species, and species of concern, as well as scarce remnants of the wet prairie communities.

Wet prairies once extended over one third of the Willamette Valley. Currently their range has been reduced to only 0.3 percent of the historic distribution (Guard 1995). The wet prairies of WEW comprise a large portion (272 ha after the inclusion of restored wetlands) of the remnant wet prairies in the Valley. However, *Phalaris* infestation has been conspicuous in some of the wet prairies in WEW, and its eradication has been a concern in the restoration efforts (COE PWE 1999b; Haney 2000). *Phalaris* found in the Northwest is probably a mixture of a native population indigenous to North America and cultivars introduced from Europe in the 1800s (Merigliano and Lesica 1998). The date of the first occurrence of *Phalaris* in WEW is not known. The earliest collection of a *Phalaris* specimen from Eugene area is dated 1975 in the Oregon State University Herbarium. However, *Phalaris* stands might have colonized earlier because it is likely such a common weedy species had not been collected (Halse 2001). *Phalaris* stands were not reported in studies of historical vegetation of WEW prior to the 1930s (Savonen 1989; LCOG 1991). The dominant plant species or communities were tufted hairgrass (*Deschampsia caespitosa*) or sedge-meadow barley (*Carex unilateralis*-*Hordeum brachyantherum*) communities, suggesting that the monotypic stands of *Phalaris* appeared sometime after the 1930s when large farming and drainage development became intensive.

CHAPTER 4: METHODS

4.1 OVERVIEW OF THE METHODS

This section overviews the methodologies used to construct a statistical model relating *Phalaris* abundance to watershed-scale landuses. Testing the hypothesis addressed in this study is also briefly described. General flow charts of the database construction and model formulation are shown in Figure 3. The specific processes and strategies are described in detail in the following sections.

Existing GIS databases were initially obtained to quantify the response variable and its explanatory variables. The computation of a spatially referenced landuse dataset and hydrology-related attributes was accomplished with a resolution of 10 m by 10 m grid. Historical survey reports were also used to construct the database of the on-site characteristics of the wetlands. Various statistical strategies were employed to examine general relationships between *Phalaris* abundance and explanatory variables. The model was formulated in a multiple regression equation and validated by a re-sampling technique and case-influence statistics. This model was used for testing the effects of watershed-scale landuses on *Phalaris* abundance. Furthermore, *Phalaris* abundance was measured at the interior of the wetlands (internal wetlands). The internal wetlands were generated by buffering inside from the perimeters of wetland units. The model was then calibrated on these new sets of response variables to investigate the change in the role of watershed-scale landuse variables within the internal wetlands (edge effect experiment).

Figure 3: Flow chart of the database construction and model formulation. The squares represent GIS layers. The GIS layers in gray-color are the existing databases obtained for this study. The descriptions of GIS layer abbreviations are listed in Table 1.

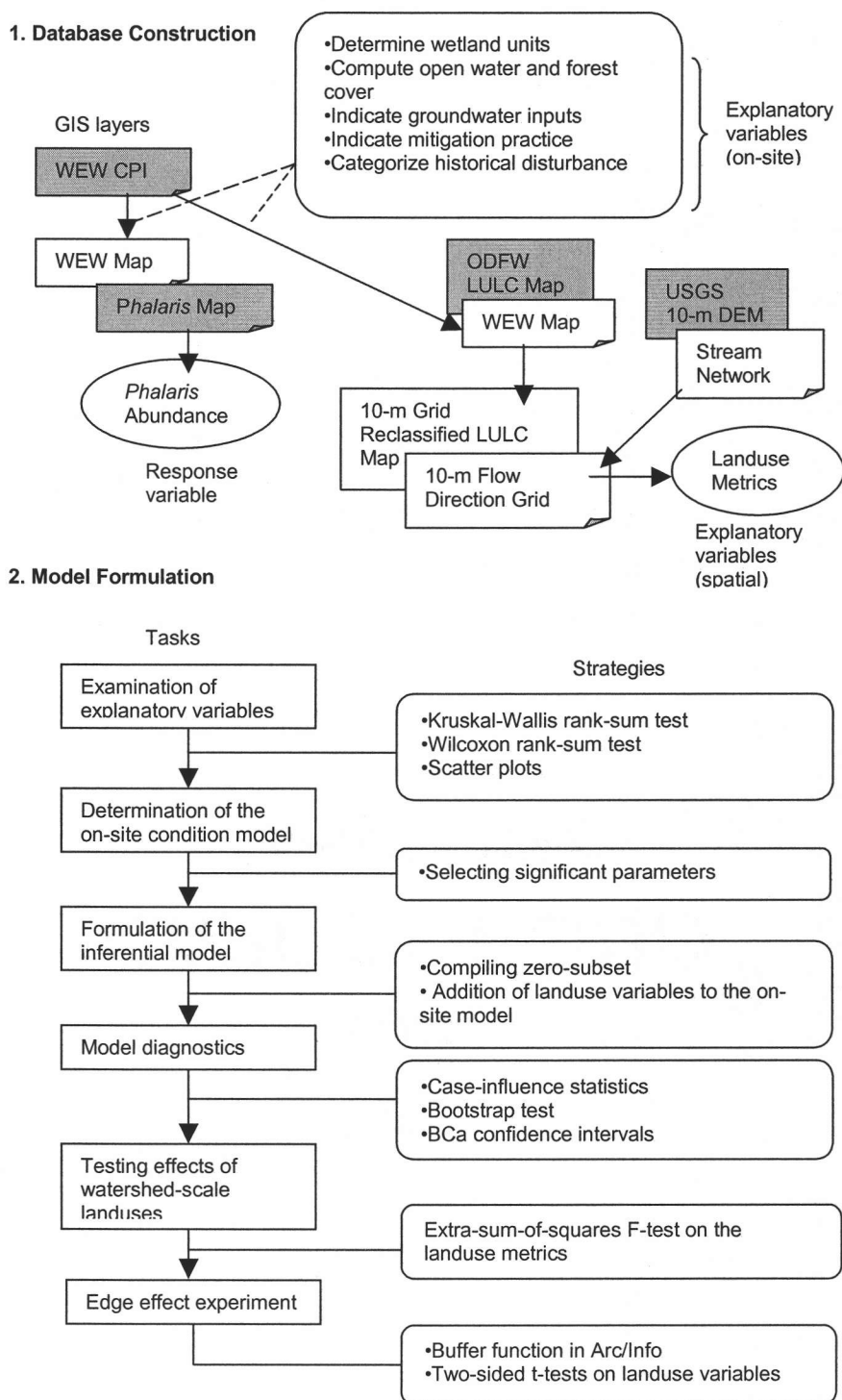


Table 1: List of abbreviations for the GIS layers described in Figure 3.

Abbreviation	Description
WEW CPI	West Eugene Wetlands Conservation Plan Inventory generated by Lane Council of Governments
WEW map	West Eugene Wetlands map modified for this study
ODFW LULC map	Oregon Department of Fish and Wildlife Land use/land cover map
USGS 10m-DEM	U.S. Geological Survey 10m-digital elevation model

4.2 EXISTING GIS DATABASES

Existing GIS databases used for this study were the WEW Conservation Plan Inventory, *Phalaris* distribution map, Willamette Valley Land use/Land Cover map, and a 10m-digital elevation model (Table 2).

Table 2: Summary of GIS databases obtained for this study.

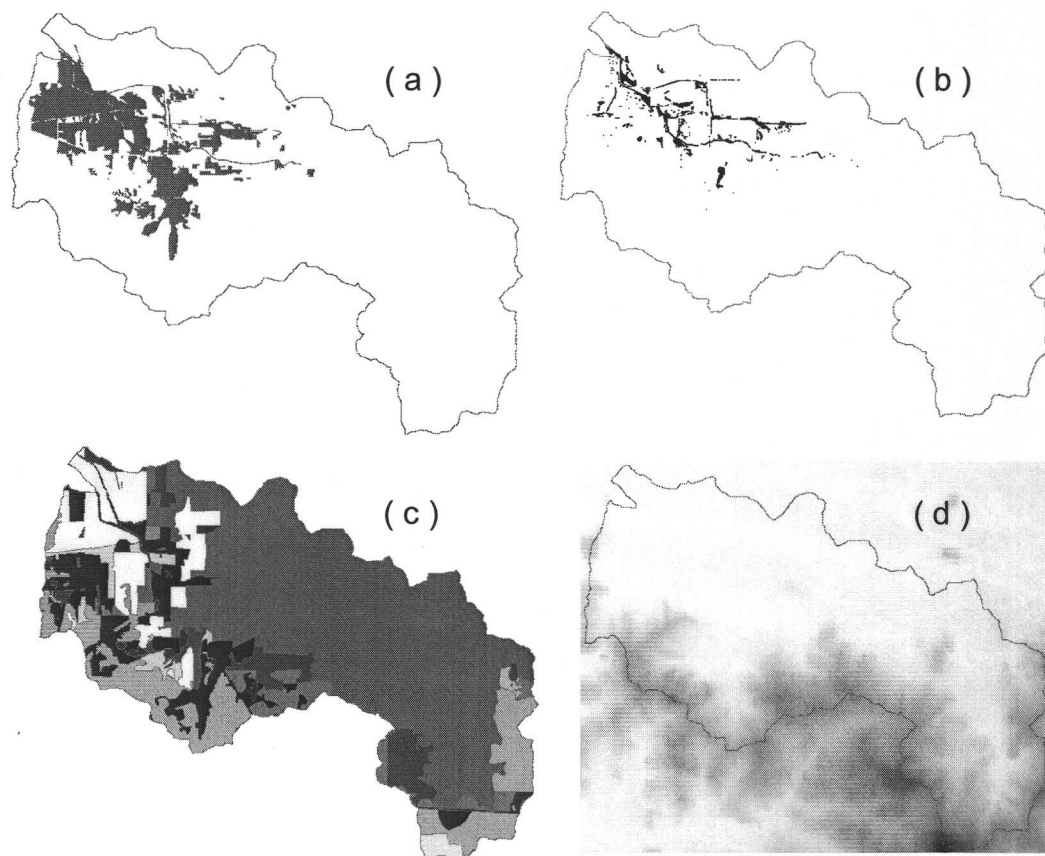
Title	Source	Year of completion	Scale	Citation
WEW Conservation Plan Inventory (WEW CPI)	Lane Council of Governments	2000	1/4,800	LCOG 2000
<i>Phalaris</i> distribution map (<i>Phalaris</i> map)	City of Eugene	2000	1/2,400	Haney 2000
Willamette Valley land use/land cover map (ODFW LULC map)	Oregon Department of Fish and Wildlife	1998	1/24,000	Klock et al. 1998
10m digital elevation model (USGS 10m-DEM)	U. S. Geological Survey	N/A	1/25,000	http://edc.usgs.gov/geodata

Visual representations of these GIS layers are shown in Figure 4. The WEW CPI and *Phalaris* distribution maps are vector coverages obtained to compute *Phalaris* abundance. The WEW CPI delineates the wetlands designated as WEW, and its attributes contain management types (protect, restore, and develop) and types of wetlands (e.g., wet prairie, forested, open water) (LCOG 2000). The wetland delineation in the WEW CPI was primarily based upon the field surveys using the "Federal Manual for Identifying and Delineating Jurisdictional Wetlands" (SRI 1989 and Guard 1994). The *Phalaris* map depicts a distribution of monotypic stands of *Phalaris* across the study wetlands based on 1999 aerial photographs taken at the scale of 1: 2,400 meters (Haney 2000). Field verifications by Haney were conducted between April and June 2000.

The ODFW LULC (vector format) was used to quantify the watershed-scale landuse attributes. The ODFW LULC characterizes the land cover surface of the entire Willamette Valley (Klock et al. 1998). Dominant vegetation types were identified for approximately 43,300 polygons (minimum 1200 m²) that constitute the valley. About 90 percent identification was made in the field and the rest were based upon color aerial photographs taken in 1993. Field verifications using a stratified random sampling resulted in an overall accuracy of 72 percent for the study area.

The USGS 10m-DEM is a grid of elevation data sampled at 10m-intervals in a raster format. It corresponds to the USGS 1: 25,000 scale topographic quadrangle map (<http://edc.usgs.gov/geodata>).

Figure 4: Visual representations of existing four GIS databases obtained for this study. (a) WEW CPI, (b) *Phalaris* map, (c) ODFW LULC, and (d) USGS 10m-DEM. The gray lines, which are not a part of each database, represent the boundaries of Upper Amazon Creek watershed.

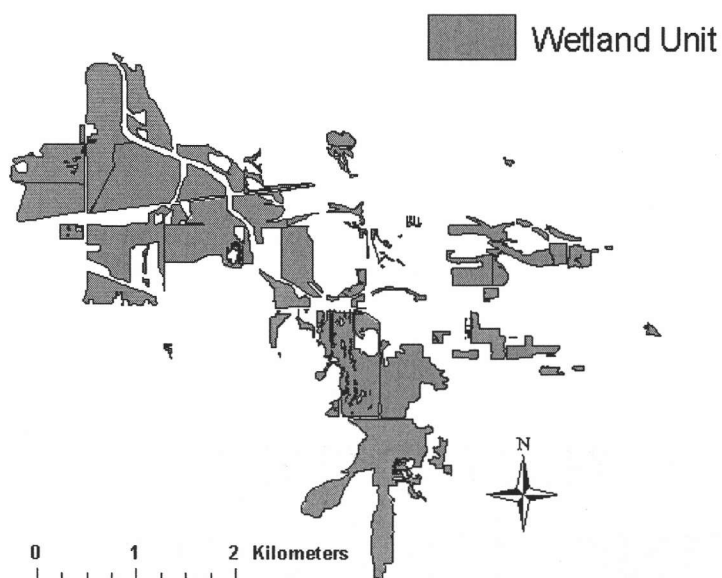


4.3 DETERMINATION OF A SAMPLING UNIT

To determine sampling units (wetland units) that were independent each other, the assessment units developed in the two original wetland inventories (SRI 1989; Guard 1994) were used. Each assessment unit in the original inventories, by definition, has no direct hydrologic connection with other units (SRI 1989), and is generally bisected by roads or channelized sections of streams. Considering the dispersal ability of *Phalaris* via rhizome, this separation among wetlands was important to improve independency of the response variable in the model. One wetland unit number was assigned to each original

assessment unit. However, the assessment units that were isolated by roads or had considerably different management statuses were further divided. The original assessment unit numbers for these wetlands were ignored but a new wetland unit number was assigned to each divided wetland. In addition, non-jurisdictional wetlands that are not regulated by ODSL and US ACE and the filled wetlands that had buildings on them were considered as non-wetlands, and therefore were excluded. Maximum buffering would remove some small wetlands in the edge effect experiment; thus, to be consistent in the number of observations, these wetlands were also excluded from this study. Applying these constraints resulted in the identification of 46 distinct wetland polygons (wetland units) in the WEW CPI. The WEW CPI was then modified to generate a wetland GIS layer for this study (the WEW map) by aggregating the wetland units in ArcInfo; the wetlands common in a wetland unit number were merged together to form a single wetland polygon (Figure 5). The total area of studied wetlands is 522 hectares. The 46 wetland units range in size from 0.2 to 65 hectares.

Figure 5: Map of study wetlands indicating the 46 wetland units.



4.4 CONSTRUCTION OF THE MODEL

The model represents how *Phalaris* abundance in wetlands can be related to watershed-scale landuse attributes, after adjusting for differences in on-site characteristics among wetlands. A multiple regression approach was employed to determine this relationship. The explanatory variables employed consist of two types: on-site conditions and watershed-scale landuses. The generic formulation of the multiple regression model takes the form:

$$\text{Phalaris abundance} = f(\text{On-site Conditions, Landuse})$$

where

Phalaris abundance = percentage of *Phalaris* cover in each wetland unit

On-site Conditions = effects of on-site condition variables

Landuse = effects of watershed-scale landuse variables.

The on-site condition variables represent the physical and biological characteristics within the wetlands. The landuse variables are spatially explicit attributes that represent the effects of landuses distributed across the watersheds. Due to the limited number of observations, a variable selection was conducted for the effects of on-site condition to reduce the number of variables. Using the selected on-site condition variables, the best on-site model was initially determined. Uncorrelated combinations of watershed-scale landuse variables were then added to the on-site model. Finally, the inferential model used for testing the hypothesis was formulated.

4.4.1 Computation of *Phalaris* abundance

To examine *Phalaris* cover in each wetland unit, the *Phalaris* map and the WEW map were superimposed in ArcView. *Phalaris* stands existed outside of the boundaries of the delineated wetland units described above. Because *Phalaris* is a facultative wetland

species that tolerates dry conditions, *Phalaris* can grow outside of jurisdictional wetlands. In this study, *Phalaris* stands outside of the jurisdictional wetlands were ignored. The areas of overlapped polygons between the *Phalaris* map and the WEW map were then summed to quantify *Phalaris* cover for each wetland unit. The proportion of the *Phalaris* cover to the area of wetland unit was computed as *Phalaris* abundance.

Phalaris abundance varied greatly across the 46 wetland units, ranging from zero to 34 percent with a mean value of 5.2 percent (Figure 6). *Phalaris* stands existed in 33 wetland units. Some of the largest stands of *Phalaris* were found in Units 21 and 23 (approximately 33 percent for both) and Units 16 and 18 (24 percent and 23 percent, respectively). Including these units, five units recorded greater than 20 percent abundance, four units ranged from 10 percent to 20 percent, and twenty-four units ranged from greater than zero to 20 percent of *Phalaris* abundance. Thirteen units contained no *Phalaris*. The spatial distribution of *Phalaris* abundance in the study wetlands is shown in Figure 7.

Figure 6: Distribution of *Phalaris* abundance across 46 wetland units.
----- indicates mean *Phalaris* abundance (%).

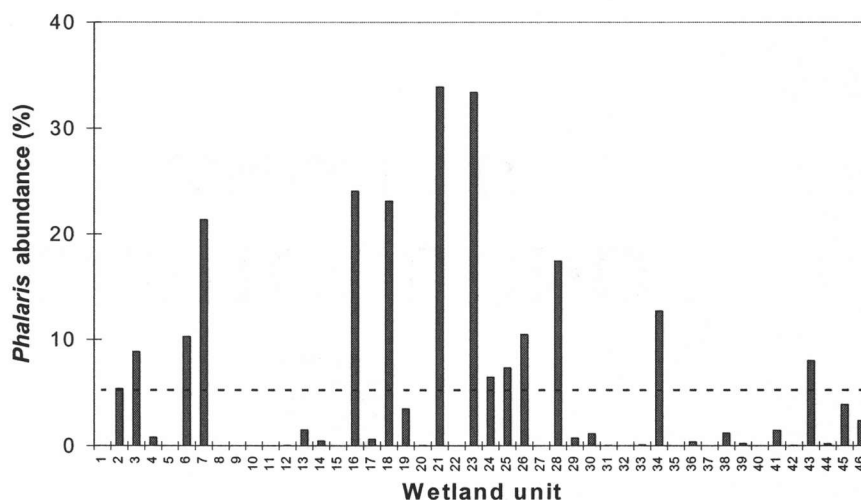
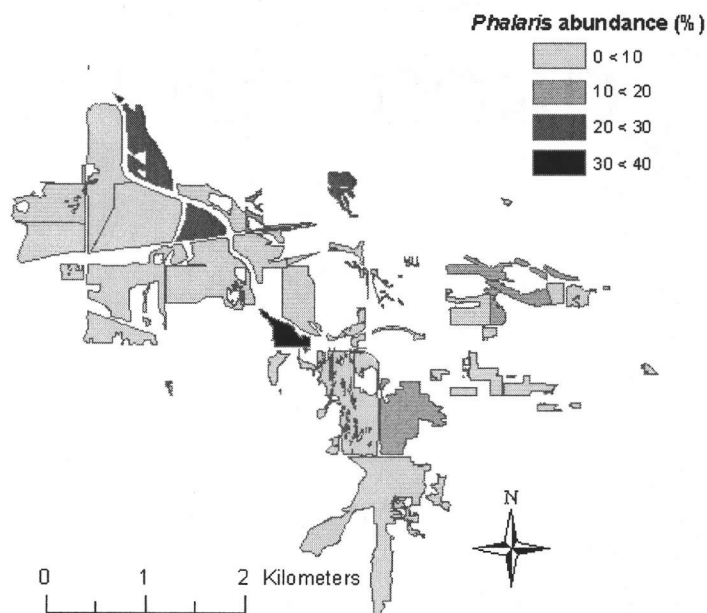


Figure 7: Spatial distribution of *Phalaris* abundance across 46 wetland units.

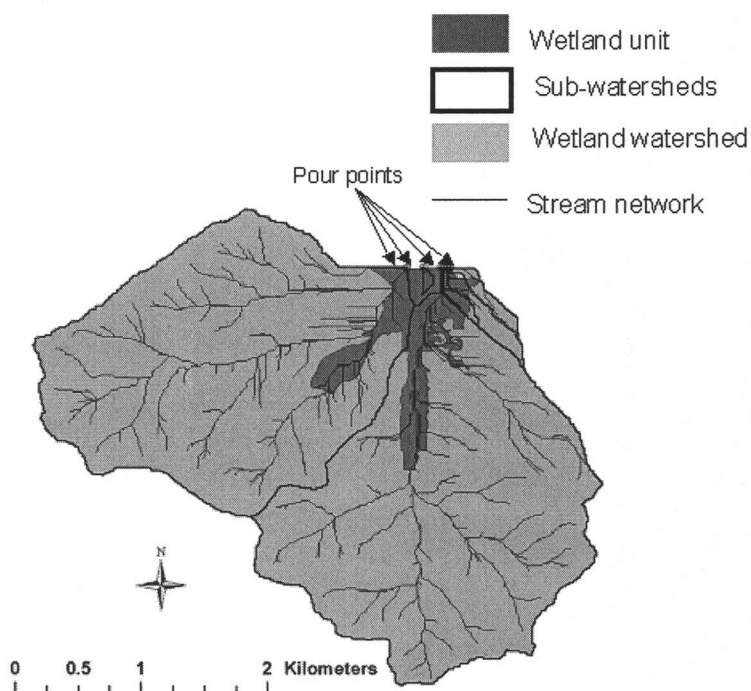


4.4.2 Determination of a wetland watershed

To determine a watershed for each wetland unit (wetland watershed), a stream network and 10m-flow direction grid were initially generated for the entire Upper Amazon Creek watershed using the USGS 10m-DEM. The upslope area of greater than 1,000 cells (10 hectares) appeared to define the actual drainage system of the Upper Amazon Creek watershed. However, many of the wetland units were artificially bisected and closely distributed to each other, so that an area delineated by a single pour point on the major tributary did not fully encompass the watershed for the wetland. Therefore, a lower minimum drainage area defining the streams (100 grid cells) was used to generate a finer stream network than the actual stream network; the area of 100 grid cells (10 m x 10 m x 100 cells = 1 hectare) at minimum drains to the generated stream network. Thus, more stream reaches were generated, which allowed selecting multiple pour points at the most downstream edge of a wetland boundary (Figure 8). Using a set of pour points, multiple sub-watersheds were then delineated for each wetland unit in Arc/Info. The sub-

watersheds for each wetland unit were then combined, representing a wetland watershed for the respective wetland unit. This process was repeated for all 46 wetland units to determine 46 wetland watersheds.

Figure 8: Determination of a wetland watershed.



4.4.3 Characterization of landuse/land cover (LULC)

The spatial extent of the ODFW LULC was initially reduced to the Upper Amazon Creek watershed. This reduced ODFW LULC consisted of 15 landuse classes (Table 3). No wetlands were classified in this map; wetland sites were classified primarily as unmanaged pasture, perennial grass, brush field, or bottomland pasture mosaic. These false classifications occurred because the ODFW LULC was based upon current vegetation type only, while the WEW map included soil samples and plant identifications.

Furthermore, many of the original wetlands in the study area have been converted to agricultural fields, confusing the vegetative identifications by ODFW.

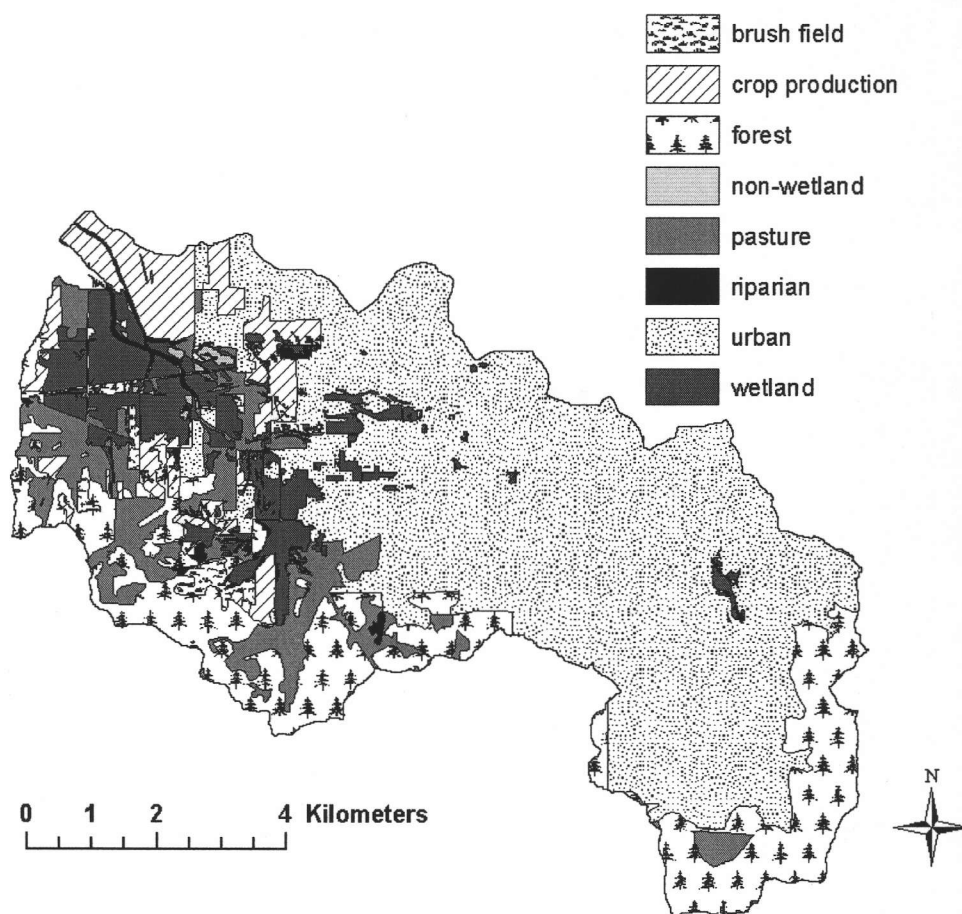
Table 3: Descriptions and distribution of landuse classes for the original ODFW LULC existed in the Upper Amazon Creek watershed (the original ODFW LULC) and the aggregation of landuse classes used in this study (LULC map).

Original ODFW LULC Classifications	LULC map Classifications	Percent of Each Landuse Across 46 Wetland Watershed		
		Mean	Range	St. Dev.
Annual grass agricultural	Crop production	12.7	0 - 98.1	24.5
Perennial grasses agricultural				
Orchards, vineyards, berries, Christmas trees, nursery stock				
Unmanaged pasture agricultural	Pasture	18.7	0 - 93.2	22.1
Urban	Urban	35.9	0 - 98.3	39.4
Parks agricultural				
Doug fir-Oak (<50% Oak)/Urban Interface				
Oak-Doug fir (>50% Oak)				
Doug fir-Oak (<50% Oak)	Forest	11.7	0 - 48.8	16.5
Oak-Madrone/Doug fir				
Maple-Alder				
Unclassified forest				
Black hawthorn, hedgerows, & brush fields				
Reed canarygrass	Riparian	0.36	0 - 1.98	0.58
Ash-cottonwood-bottomland pasture mosaic				
Wetland Inventory				
Wetland (Combined with ODFW LULC)	Wetland	18	1.7 - 85.5	17.5
Non-wetland (Combined to ODFW LULC)	Non-wetland			

To produce a LULC map for this study, which combines the ODFW LULC and the wetlands, the original ODFW LULC reduced for the Upper Amazon Creek watershed and the WEW map were overlain in ArcInfo. The resulting map had the combined attribute data from both input layers in the overlapped polygons. In these overlapped polygons, the land-use attributes from the original ODFW LULC were removed and replaced with the wetland attributes from the WEW map. Compared to the grid size used in this study (100 m²), the resolution of the ODFW LULC map (1200 m²) was considered to be coarse. To minimize the uncertainty of LULC map, the original ODFW LULC classes were simplified by aggregating the classes that had similar surface characteristics and sources of potential pollution (Zhu et al. 2000). This simplification of the landuse classes is particularly important to avoid errors in the predictors of a multiple regression model. Predictor errors that are not explained by the model give a biased estimate, and therefore should be minimized (Weisberg 1985). In addition, the generality of the model can benefit by the simplified classification system (Osborne and Wiley 1988).

Thus, fifteen ODFW LULC classes were simplified into six classes (Table 3). It should be noted that there are potential differences in nutrient loadings in the three crop types (annual and perennial grass production and orchards). However, an uncertainty in the predictors due to the separation of these classes should be avoided by aggregating them into a single class. Figure 9 shows the reclassified LULC map used in this study. This LULC map was converted into a raster format of 10m-resolution. Since surface characteristics and management types of the non-wetland class were unknown, this class was excluded from further analyses. Wetland class was not also considered because the wetlands themselves are the response variable.

Figure 9: The reclassified LULC map used in this study.



4.4.4 Selection of on-site condition variables

The on-site conditions considered in this study are 1) percentage cover of forested area, 2) percentage cover of open water, 3) groundwater input, 4) mitigation practice, and 5) anthropogenic physical disturbances associated with historical management of the wetlands (Table 4).

Table 4: Summary of on-site condition variables considered in this study.

Variable	Type	Values	Sources
Percentage forested cover	Numerical	Percentage	WEW map
Percentage open water cover	Numerical	Percentage	WEW map
Groundwater input	Indicator	High/Medium	SRI (1989) and Guard (1994)
Mitigation practice	Indicator	Mitigated/Non-mitigated	COE PWE (1999a)
Physical disturbance	Categorical	0: non-active, 1: plowed, 2: highly disturbed, 3: grazed/mowed	SRI (1989), Guard (1994), LCOG (1991), and COE PWE (1999b)

The percentage of forest and open water cover in the wetlands were considered because *Phalaris* growth can be suppressed by shading under the forest canopies and by reduced soil conditions in prolonged standing water (Brix and Sorrell 1996; Maurer and Zedler 2002). The forested cover and open water cover were determined by summing the areas of each attribute for each wetland unit in the WEW map. The percentages were quantified as the proportions of the area of each attribute to the wetland area.

The groundwater input variable is the estimated 'effectiveness of ground water discharge' that ranks the significance of a net groundwater discharge into a wetland (Adamus et al. 1991). The 'effectiveness of groundwater discharge' was included in the original wetland inventories (SRI 1989; Guard 1994). This variable was chosen because the unique chemical composition of groundwater can influence wetland vegetation (Mitsch and Gosselink 1993). The groundwater input variable was assigned as an indicator variable (1 for HIGH and 0 for Medium). Low effectiveness of ground water discharge was not found among the 46 wetland units.

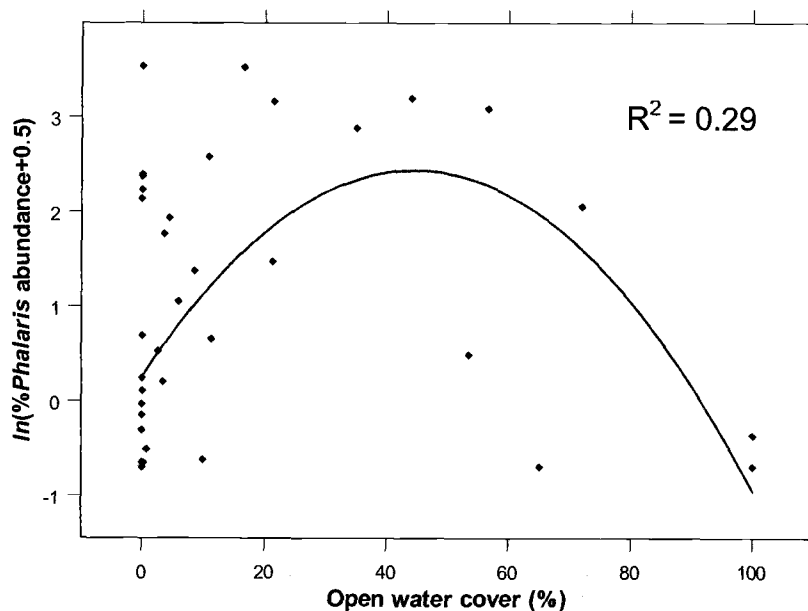
To represent the current management status of the wetlands, an indicator variable for mitigation practice was used. In study wetlands, formerly disturbed wetlands have been restored as mitigation wetlands to compensate for the loss of the other wetlands by

development (COE PWE 1999a). Because the details of management status for privately owned mitigation sites were not available, the variable was assigned as 1 if the wetland unit experienced a known mitigation practice, and 0 otherwise.

The variables of physical disturbance associated with historical management were constructed from three existing survey reports: the WEW special study area technical report prepared by LCOG (1991), the original wetland inventories by SRI (1989) and Guard (1994), and the WEW mitigation bank technical appendix by City of Eugene Public Works Engineering (1999b). Study wetlands were found to be categorized in four disturbance types: plowed, grazed/mowed, non-active fields, and highly disturbed. Plowed sites have been periodically plowed for rye grass production. Grazed/mowed sites had been grazed or mowed. The sites in the plowed and grazed/mowed categories were determined in 1991 (LCOG 1991). Some of the plowed sites may still be in use. Non-active fields were not actively used, but were known to have in the past been agricultural fields (LCOG 1991). A relatively undisturbed site was included in the category of non-active fields because only one wetland unit (Willow Creek Natural Area) represents the undisturbed site. The rest of the sites were classified as highly disturbed, and were wetlands used as artificial ponds, waste treatment facilities, and tree farms.

All on-site condition variables underwent a preliminary selection. For the numerical variables (the percentages of open water and forest covers), scatter plots were used to visually assess their general relationship with *Phalaris* abundance. The percentage of open water cover had the best fit with a 2nd order-polynomial line on log-transformed *Phalaris* abundance ($R^2 = 0.29$) (Figure 10). The scatter plots showed a weak unimodal trend of *Phalaris* abundance along the gradient of increasing percentage of open water cover. This trend suggests that the on-site model may need a squared term of this variable. The effect of open water on *Phalaris* abundance is different depending upon the percentage of open water cover. *Phalaris* abundance increases only up to a certain point of open water cover and decreases again in the wetlands where open water occupies since *Phalaris* cannot tolerate reduced conditions in standing water.

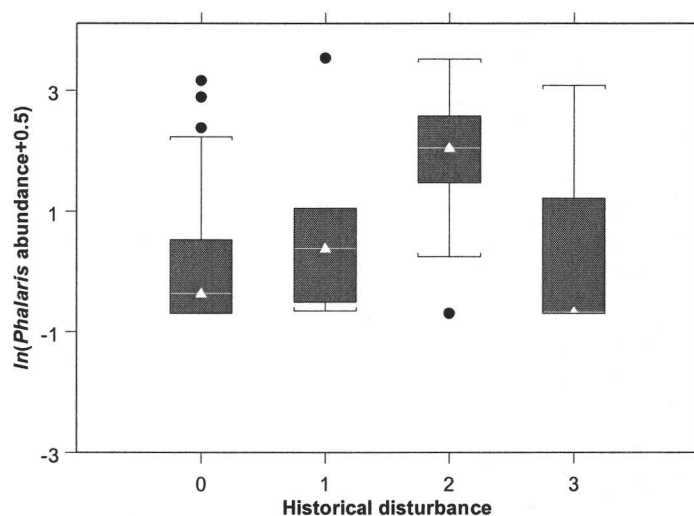
Figure 10: Relationship between log-transformed *Phalaris* abundance and percentage of open water cover in wetlands.



Because no relationship between *Phalaris* abundance and the percentage of forested area was found, this variable was excluded from the on-site model. Neither the groundwater input nor mitigation practice variables showed a significant effect on mean *Phalaris* abundance (two-sided p -value = 0.476 and 0.103, respectively from the Wilcoxon rank-sum tests). Therefore, these variables were excluded from the on-site model.

Of all physical disturbance classes, the median *Phalaris* abundance appeared to be the highest for highly disturbed sites, followed by plowed and non-active field sites (Figure 11). There was a relatively wide range in *Phalaris* abundance for grazed/mowed sites. The distribution of *Phalaris* abundance for non-active fields and grazed/mowed sites were skewed to low abundance. There were also some extreme observations in non-active fields, plowed sites, and highly disturbed sites.

Figure 11: Box plot of log-transformed *Phalaris* abundance across four types of historical disturbances. 0 = non-active fields, 1 = plowed sites, 2 = highly disturbed sites, and 3 = grazed/mowed sites. Triangles represent median *Phalaris* abundance. Black bars indicate 1.5 inter quartile ranges (IQR). Black points are the extreme values that are greater than 1.5 IQR.



The Kruskal-Wallis rank-sum test indicated that there was a significant difference between median *Phalaris* abundance across the four types of physical disturbances (two-sided p -value = 0.0181). To select the most significant disturbance class affecting *Phalaris* abundance, each class was sequentially compared to the other combined classes, using the Wilcoxon rank-sum test. These tests revealed a significant difference in median *Phalaris* abundance for highly disturbed sites and non-active field sites separately (two-sided p -value = 0.0038 and 0.0181, respectively). Therefore, only the highly disturbed class, which was the most significant variable, was selected. A further Kruskal-Wallis rank-sum test ignoring the highly disturbed sites (Ramsey and Schafer 1997) indicated equality of median *Phalaris* abundance for the other disturbance classes (two-sided p -value = 0.312).

From this preliminary variable selection for the on-site condition model, the percentage of open water cover and highly disturbed indicator were selected as the most important explanatory on-site variables.

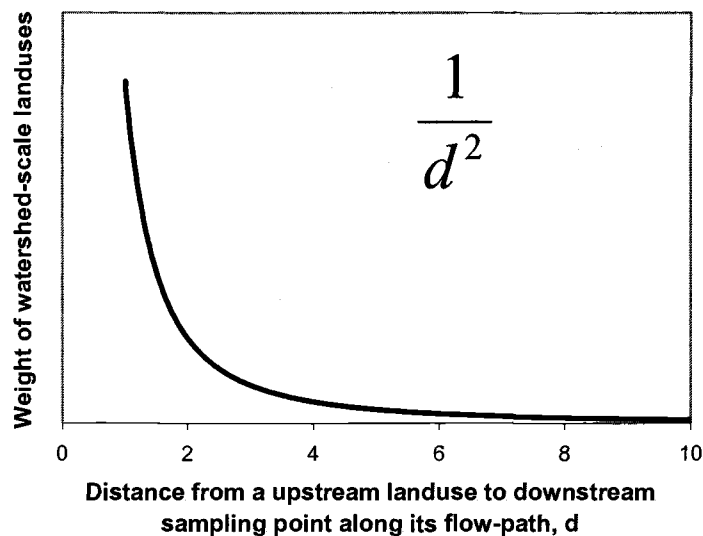
Because *Phalaris* distribution data across the entire Upper Amazon Creek watershed was not available, dispersal potential of *Phalaris* along the stream channels was

not considered. In addition, some wetlands were partially or entirely inundated in a 1965 flood and could have received more seed and propagules than other inundated wetlands. However, it is difficult to assess the dispersal potential of *Phalaris* for each wetland unit and its route at the watershed level, due to the greatly disturbed landscape and the close proximity of the wetlands to each other. Therefore, all wetland units are assumed to be equally susceptible to *Phalaris* seed or propagule sources.

4.4.5 Quantification of watershed-scale landuse variables

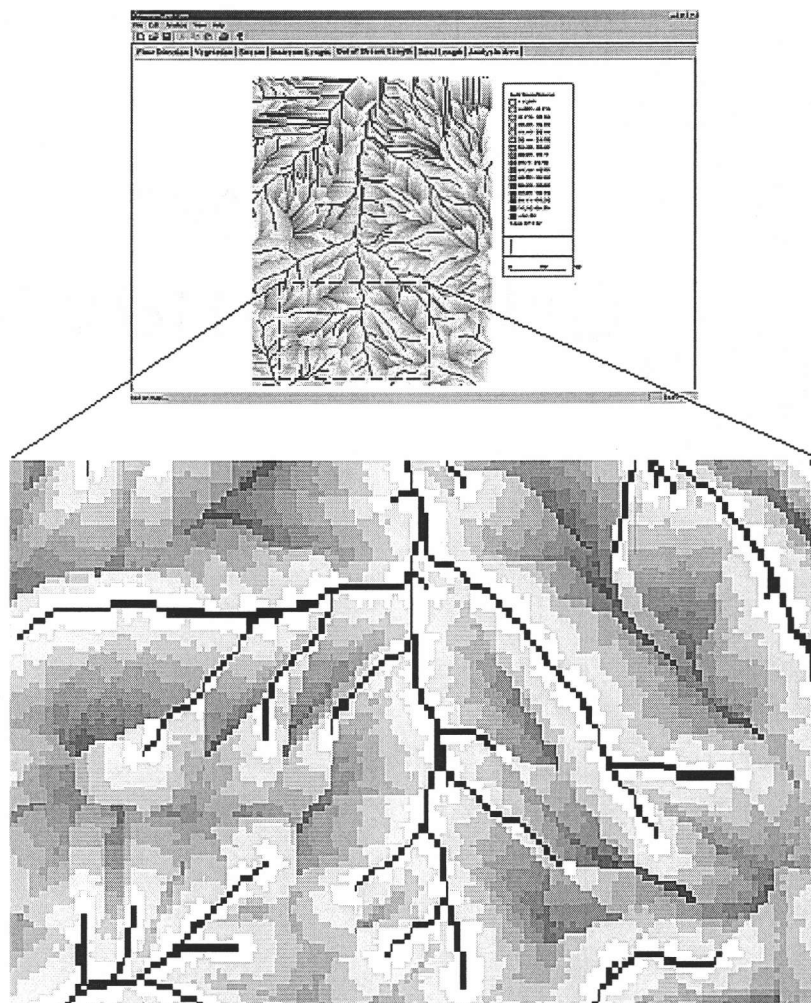
The landuse variables in this study were quantified using the inverse flow distance method developed by Kehmeier (2001). The inverse flow method estimates the effects of watershed-scale landuses as a function of accumulated flow path distances between a downstream sampling point and cells in a grid-based representation of LULC data. This study assumes that spatially distributed landuses are surrogates for water quality and hydrology that may influence *Phalaris* abundance in wetlands. Water derived from a specific site in the watershed moves along its flow path, where it is exposed to various transformational processes such as nutrient assimilation by organisms and adsorption onto river-bottom sediments. The movement of water may be slow or fast depending upon the land-surface characteristics. As water carrying the influence of upstream landuse travels to a wetland, the influence decreases. Therefore, the closer landuse has the greater influence on its respective wetland but the entire watershed also has an effect. To enumerate this weighted influence of landuse as a function of flow path distance, the distance for each cell is formulated in an inverse-squared form; thus effects of cells near sampling points are more heavily weighted than those further away as represented by the flow path distance (Figure 12).

Figure 12: Conceptual relationship between weight of watershed-scale landuses and a flow path distance for each cell using the inverse flow distance method.



Although surface and shallow sub-surface processes consist as in-stream and out-of-stream components, the weighted distance was computed for only out-of-stream processes (Figure 13). Out-of-stream distance is defined as the distance from a cell to an entry point into streams along its flow path (Kehmeier 2001). In-stream distance is the distance between a cell's entry point in the stream and the sampled point in its respective wetland. In this study, in-stream process was ignored because the sampling unit was a wetland polygon instead of a point. In-stream distance would include the reaches within a wetland unit, thereby inaccurately representing the influence of watershed-scale landuses on the entire wetland unit. Thus, the importance of the delivery process of overland and sub-surface flows was emphasized in this study.

Figure 13: Visual representation of weighted out-of-stream influences of each cell for a part of Willow Creek unit. Lighter colors represent greater out-of-stream influence. The dark line represents a stream network.



To quantify the landuse variables, a flow path from each cell to the pour point was identified on the 10m-flow direction grid, and its length along the flow path to the pour point or entering point into streams was computed. The length was formulated in an inverse-squared form for each cell. Furthermore, these weights were aggregated for each landuse type for each sub-watershed. The total weights for each sub-watershed were then computed for each wetland watershed to create a landuse metric for this study. The

landuse metric for each wetland watershed represents the cumulative influence of landuses across the watershed weighted by flow path distance from the landuse to the point of effect. Thus, the landuse metrics capture the spatial variability of landuse patterns for the entire watershed with the greatest weighting given to landuses near the wetlands. The landuse metric was computed for each of the 6 landuse types. Mathematical formulation of a landuse metric for each wetland watershed is:

$$LU_j = \sum_{k=1}^{WS_k} \sum_{i=1}^{C_j} (d_{i,j,k})^{-2}$$

where

LU_j = landuse metric for landuse type j

j = index of landuse class

i = index of cell of the landuse class j

k = index of a set of sub-watersheds for each wetland unit

C_j = number of the cells of landuse class j

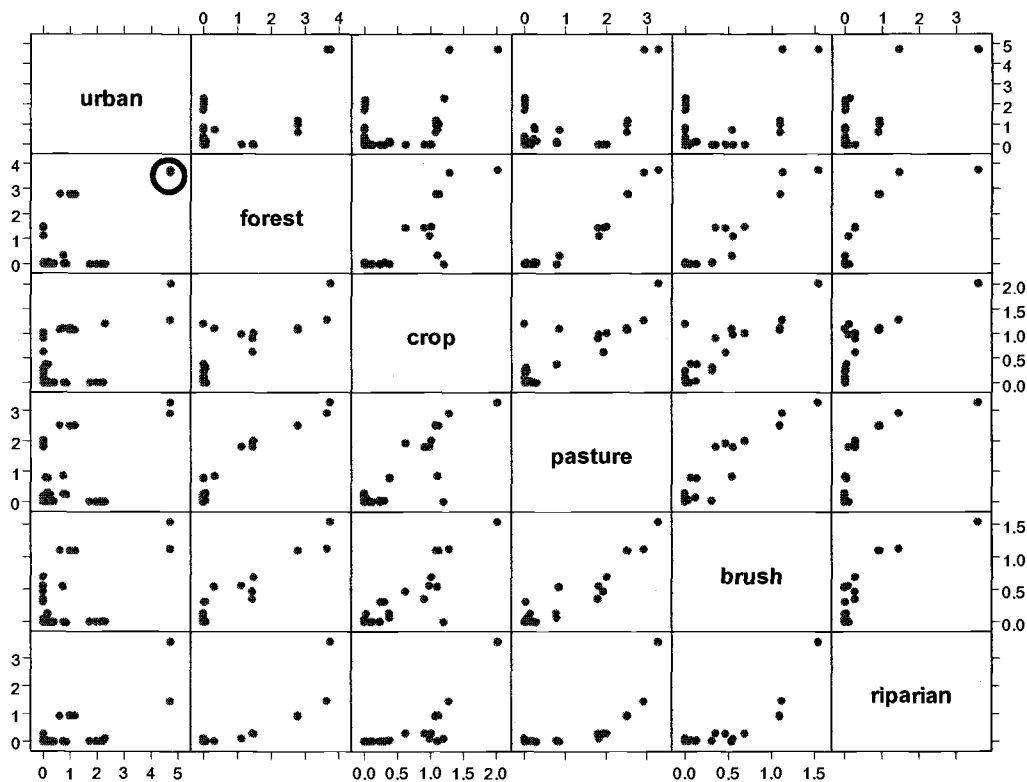
WS = number of the sub-watersheds of landuse class j

d = flow path distance,

when the landuse of $C_{i,k}$ = the landuse of LU_j .

To select independent combinations of landuse metrics, a correlation matrix showing all landuse metrics was visually assessed. Figure 14 shows the general relationships among the 6 landuse metrics. Urban appears to be the only landuse that is uncorrelated with any other landuse metrics. Therefore, combinations of landuse metrics for the inferential model were: urban and forest, urban and crop, urban and pasture, urban and brush field, and urban and riparian. There were several extreme observations that may be influential on the model.

Figure 14: Scatter matrix plots for landuse metrics. The values are log transformed after adding 1. An example of extreme observation is circled.



4.4.6 Model formulation

The large number of no *Phalaris* abundance (13 observations) was a concern. Zero as a response variable would create uncertainty because of the potential bias caused by adding a small value before the log-transformation. Parametric coefficients of a multiple regression model are sensitive to the magnitude of the value added to the response variables. To avoid this uncertainty, the observations that have no *Phalaris* abundance were compiled to a single observation as a zero-subset. To justify this, the range of each explanatory variable was compared between the zero-subset and the entire observations (Table 5).

Table 5: Ranges of explanatory variables of 46-samples versus zero-subset.

	Crop production	Pasture	Forest	Brush field	Riparian	Urban	Open water	Number of highly disturbed sites
46- samples	0 - 6.56	0 - 25.5	0 - 41.1	0 - 3.68	0 - 3.60	0 - 110	0 - 100	11
13-no <i>Phalaris</i> samples	0 - 0.257	0 - 0.261	0 - 0.0705	0 - 0.0438	0 - 0	0 - 0.485	0 - 100	1

The ranges of landuse variables of zero sub-set were considerably smaller than those of the entire observations. However, the percentage of open water ranged from zero to 100 percent for both of the zero-subset and the entire sample. Among the 13-no *Phalaris* samples, the percentage of open water cover generally either constrained in very small values (0 to 0.05 percent) or had 100 percent cover. Only one sample had an intermediate value of open water cover (65 percent). In addition, the no-*Phalaris* sample that had 100 percent open water cover had 1 for the indicator variable of highly disturbed sites. The no-*Phalaris* samples that had 65 percent open water cover and an indicator of highly disturbed site were considered to be anomalies with respect to the ranges of explanatory variables. Therefore, these two departures were treated as independent observations instead of grouping them into the zero-subset. For the other zero response sites, except for these two independent zero observations, a single observation was artificially created by averaging their numerical explanatory variables and taking common indicator variables. As a result, 36 observations in total (33 observations that contain non-zero responses, two that have zero responses, and one zero-subset representing 11 observations that also have zero responses) were used for constructing an inferential model. It should be noted that this simplification of some observations into a single subset would create an unexplained variation in the model, impairing the statistical power. However, this method was applied because the incorporation of a large amount of bias into the model was considered even more detrimental.

The selected on-site condition variables were regressed on 36 observations of *Phalaris* abundance. The best on-site model was determined as:

$$\text{Phalaris abundance} = f(\text{openwater}, \text{openwater}^2, \text{hd})$$

where

openwater = percentage of open water cover in the wetlands

hd = indicator variable for highly disturbed sites.

A summary of regression coefficients and statistics on the on-site model are presented in Table 6.

Table 6: Regression summary for the on-site condition model. Estimated standard error of *Phalaris* abundance is 1.188 on 32 degrees of freedom. $R^2 = 0.38$

Model Parameters	Parametric Coefficient	Standard Error	p-Value
open	0.071	0.0232	0.0044
open ²	-0.0009	0.0003	0.0019
hd	0.5767	0.218	0.0126

To determine the best inferential model, the independent combinations of landuse variables were sequentially added to the on-site model. Finally, among all combinations of landuse variables, the best inferential model combined two landuse variables, urban and forest. None of the other landuse variables showed a significant relationship with *Phalaris* abundance.

The inferential model takes the form:

$$Phalaris \text{ abundance} = f(\text{openwater}, \text{openwater}^2, \text{hd}, \text{urban}, \text{forest})$$

where

openwater = percentage of open water cover in the wetlands

hd = indicator variable for highly disturbed sites

urban = cumulated landuse metric for urban lands

forest = cumulated landuse metric for forested lands.

The regression coefficients and statistics on the inferential model are summarized in Table 7.

Table 7: Results of parametric regressions of *Phalaris* abundance, estimated using the inferential model. Estimated standard error of *Phalaris* abundance is 0.8895 on 30 degrees of freedom. $R^2 = 0.67$.

Model Parameters	Parametric Coefficient	Standard Error	p-Value
open	0.0513	0.0179	0.0075
open ²	-0.0007	0.0002	0.0016
hd	0.7461	0.1666	0.0001
urban	0.7406	0.1465	0.00001
forest	-0.3443	0.1567	0.0359

For all the multiple regression models constructed in this study, prior examination satisfied normality and constant variance requirements. No spatial auto-correlation was apparent. To stabilize the variances, *Phalaris* abundance and landuse variables were natural log transformed after adding either 0.5 or 1 to each observation. All statistical tests

were conducted at $\alpha = 0.05$ using S-Plus 4.5 Student Edition (Brooks/Cole Publishing Company 1999).

4.5 MODEL DIAGNOSTICS

Case influence statistics (leverages, student residuals, and Cook's distances) detected some potentially influential observations on the inferential model (Figure 15, 16, and 17). Leverages measure the distance between the explanatory variable values of an observation and the average of the explanatory variable values in the entire observations (Figure 15). The observation with a high leverage also has a small residual; thus, it may determine the slope of the estimated regression line. Student residuals evaluate the degree of departure of an observation in its response variable value. It is not unusual for five percent of the entire observations to fall outside of the range between -2 and $+2$ of student residuals (Ramsey and Schafer 1997). Using leverages and student residuals, the nature of the explanatory and response variable values for each extreme observation was examined. The wetland units 8 and 44 scored high leverages because these two observations had extreme values in open water (100 percent), indicating that these observations are not truly unusual (Figure 15). Next, the wetland unit 3 had a relatively high student residual (Figure 16), because the unit 3 has relatively high *Phalaris* abundance (8.85 percent) with low values of open water and landuse metrics and an indicator of highly disturbed site. The high student residual of the unit 3 was considered to be normal because it constituted only 2.8 percent of the entire observations.

Cook's distances measure the degree of anomaly of an observation in its explanatory and response variable values simultaneously (Ramsey and Schafer 1997). Therefore, the Cook's distance statistic was used to examine the effects of deletion of a suspected observation on the inferential model. The wetland units 8, 19, 21, and 44 showed high Cook's distances, suggesting that these observations were potentially influential on the model (Figure 17). The units 19 and 21 had extremely high urban landuse metrics because their wetland watersheds contain the main channel of Amazon Creek. The units 8 and 44 were detected due to their high leverages as mentioned in the above. Each of the extreme observations was sequentially excluded in order of greater

Cook's distance, and the inferential model was fitted at each time. The changes in the parametric coefficients and their significances were examined. The unit 8 was not excluded because the exclusion of the unit 44 eliminated the potential influence of the unit 8.

Figure 15: Influential cases in their explanatory variables. The number of the wetland unit and its value are indicated in parenthesis.

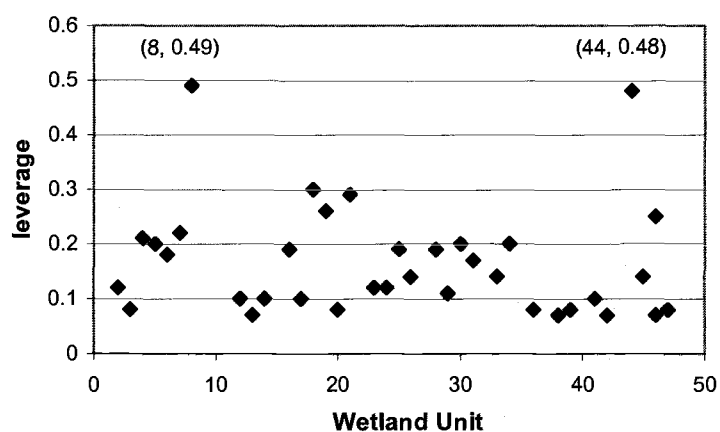


Figure 16: Influential cases in their response variables. The number of the wetland unit and its value are indicated in parenthesis.

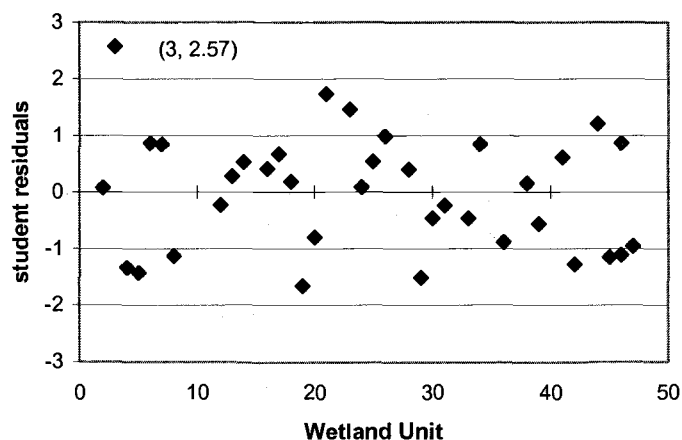
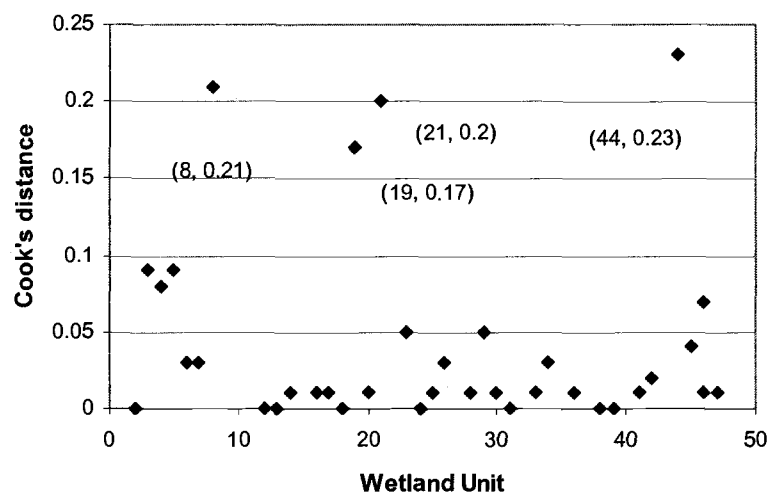


Figure 17: Overall influence. The number of the wetland unit and its value are indicated in parenthesis.



The parametric coefficients and their statistical significance changed little whether or not these extreme observations were omitted (Table 8). Rather, the overall model performance improved from $R^2=0.67$ to 0.70 . Thus, none of the observations detected in the case influence statistics was found to be truly outlier and influential on the inference of the relationship between *Phalaris* abundance and the landuse variables.

Table 8: Parametric coefficients and statistical significance p -values for the inferential model, compared between the inferential model and models excluded the observations that scored high Cook's distances.

Model Parameter	No exclusion		Without 45		Without 45 & 22		Without 45, 22, & 20	
	Parametric Coefficient	p -value	Parametric Coefficient	p -value	Parametric Coefficient	p -value	Parametric Coefficient	p -value
open	0.0513	0.0075	0.0589	0.004	0.0658	0.0013	0.0641	0.0016
open ²	-0.0007	0.0016	-0.0008	0.0011	-0.0009	0.0003	-0.0009	0.0004
hydro	0.7461	0.0001	0.7897	0.0001	0.7942	0.00001	0.7884	0.00001
urban	0.7406	0.00001	0.7293	0.00001	0.6267	0.0002	0.7058	0.0002
forest	-0.3443	0.0359	-0.3391	0.0375	-0.3686	0.02	-0.3403	0.0326

Furthermore, to assess the robustness of the parameter estimates of the inferential model, a bootstrap test was performed. The bootstrap involves the random re-sampling of observations (the same size as the observed data) from the observed data with replacement, thereby imitating the process of sampling observations from a larger population (Efron and Tibshirani 1993). The bootstrap therefore requires no assumption of normality. Instead, it estimates distributions for any statistic based on a population distribution generated by the bootstrap re-sampling. In this study, this re-sampling process was iterated for 1,000 times to compute bootstrap estimates of the mean and its bias from the estimated mean by the inferential model. The bias-corrected-and-accelerated (BCa) percentiles were constructed to evaluate the significance of the estimated parametric coefficients. The regression estimates and BCa intervals from the inferential model are presented in Table 9. The ratio of estimated bias to its respective standard error is less than 0.25, suggesting the bias is negligible (Efron and Tibshirani 1993). The ninety percent BCa intervals did not include zero, indicating that a significant relationship exists between all of the explanatory variables and *Phalaris* abundance.

Table 9: Comparisons of parametric and bootstrap regressions of *Phalaris* abundance. Parametric coefficients are estimated using the inferential model. Bootstrap coefficients are estimated from the bootstrap samples. Bias is the difference between a parametric coefficient and its respective bootstrap coefficient.

Model Parameters	Parametric Coefficient	Bootstrap Coefficient	Bias	BCa Lower 90% Confidence Interval	BCa Upper 90% Confidence Interval
open	0.0513	0.0520	0.00063	0.0142	0.0849
open ²	-0.0007	-0.0007	-0.000008	-0.0011	-0.0002
hd	0.7461	0.7692	0.0231	0.3855	1.0320
urban	0.7406	0.7243	-0.0163	0.4471	1.0710
forest	-0.3443	-0.3356	0.0087	-0.6503	-0.0657

4.6 EDGE EFFECT EXPERIMENT

The role of watershed-scale landuses in internal wetlands was examined. The perimeters of wetland units were buffered towards the inside of the wetlands to generate polygons for internal wetlands. Buffering mimics the removal of the edge effect from *Phalaris* abundance. Edge effects include filtering functions of the wetlands and physical disturbances occurring along the perimeters of wetlands. As water passes through the wetlands, pollutants are removed, decreasing the influence of landuses on *Phalaris* abundance. Physical disturbances can include grazing or stamping by cattle, mowing, ditching, or public access. Therefore, *Phalaris* abundance could be different after removing these edge effects.

Buffer areas in the wetlands were defined using the BUFFER function within ArcInfo with the width of 4 to 16m at 4-meter intervals (ESRI 1994). The minimum width was selected by the resolution of *Phalaris* map. The maximum was constrained by the number of samples; the greater widths eliminate more wetlands. *Phalaris* abundance was then computed for each buffer size and regressed using the inferential model. Therefore, for this edge effect experiment, four regression models were additionally constructed. The computation of landuse metrics were not repeated for all buffer sizes since moving the pour points within the wetland units has a minor effect on the cumulative flow path distances of watershed-scale landuses. The significance of landuse variables was then compared at the different buffer sizes.

CHAPTER 5: RESULTS

5.1 EFFECTS OF WATERSHED-SCALE LANDUSES ON *PHALARIS* ABUNDANCE

Convincing evidence was found that the urban landuse is positively related to *Phalaris* abundance (p -value = 0.00001, two-sided t-test) (Table 7). There was also moderate evidence that the forest landuse was negatively associated with *Phalaris* abundance (p -value = 0.0359, two-sided t-test) (Table 7). Both significant relationships were adjusted for on-site conditions. Added variable plots (Weisberg 1985) illustrated the individual influence of urban and forest landuses separately on *Phalaris* abundance but adjusted for the all other explanatory variables in the inferential model (Figure 18 and Figure 19 for urban and forest, respectively). The slope for each line is equivalent to its respective coefficient estimated by the model; thus, the trends represented by the lines are statistically significant. Figure 18 depicts a significant increasing trend of *Phalaris* abundance associated with an increase in urban lands. Figure 19 suggests a moderate negative association of forestlands on *Phalaris* abundance. There is convincing evidence of interactions between *Phalaris* abundance and the watershed-scale landuses of urban and forest together (p -value < 0.001; extra-sum-of-squares F-test). *Phalaris* abundance was better described by adding the watershed-scale landuse variables of urban and forest ($R^2 = 0.67$), when compared to the on-site model alone ($R^2 = 0.38$).

Figure 18: Influence of urban lands on *Phalaris* abundance after adjusting for the effects of on-site condition and forestland.

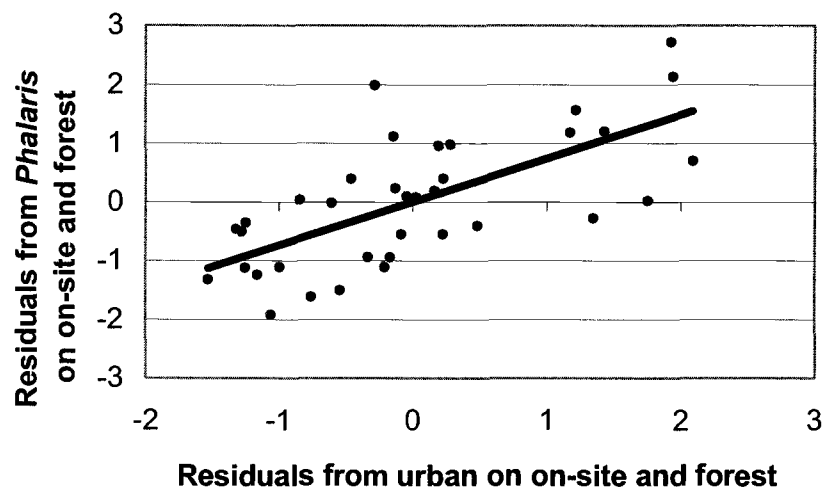
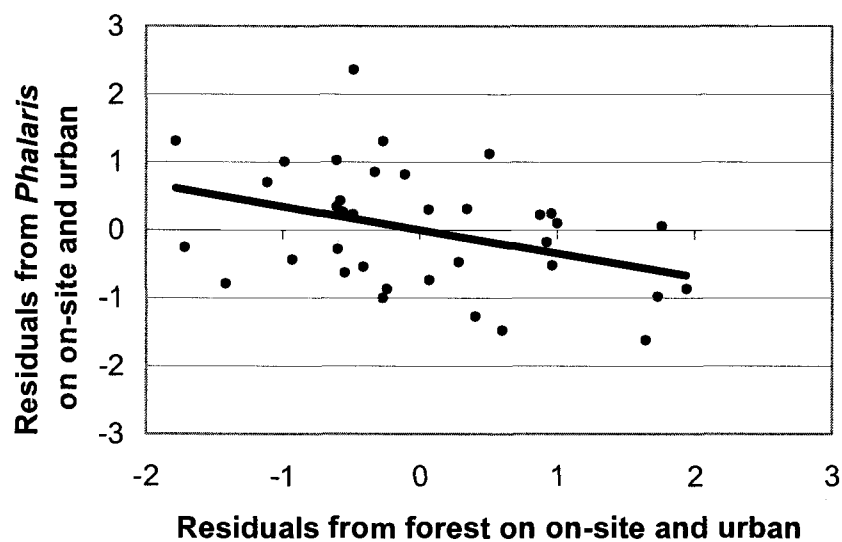


Figure 19: Influence of forestlands on *Phalaris* abundance after adjusting for the effects of on-site condition and urban land.



5.2 EDGE EFFECT EXPERIMENT

Over the full range from zero to 16 m of buffer width, the two landuse variables combining urban and forest remained significant (p -values < 0.01 ; extra-sum-of-squares F -tests). Since the forest landuse variable became insignificant for the buffer widths greater than 4 m, the edge effect was tested on the significance of urban landuse variable in the models that consist of only on-site condition and urban lands (the on-site & urban model). Figure 20 shows extra-sum-of-squares (ESS) of the urban landuse variable in the on-site & urban model, illustrating the change in significance of the urban variable as the buffer width increases. ESS was used for this analysis because the scale of p -values was so small making difficult to compare with the residual-sum-of-squares (RSS). The significance of the urban landuse variable decreases as the buffer width reaches 4 m and increases again from 4 m through 16 m of buffer width. Figure 21 plots RSS comparing the on-site model and the on-site & urban model. There is a general trend of increasing RSS for both models along the increasing buffer widths. While the RSS for the on-site model continues increasing for buffer widths greater than 4 m, RSS for on-site & urban model appears to level off for widths larger than 8 m.

Figure 20: Change in the significance of urban landuse variable along increasing buffer width.

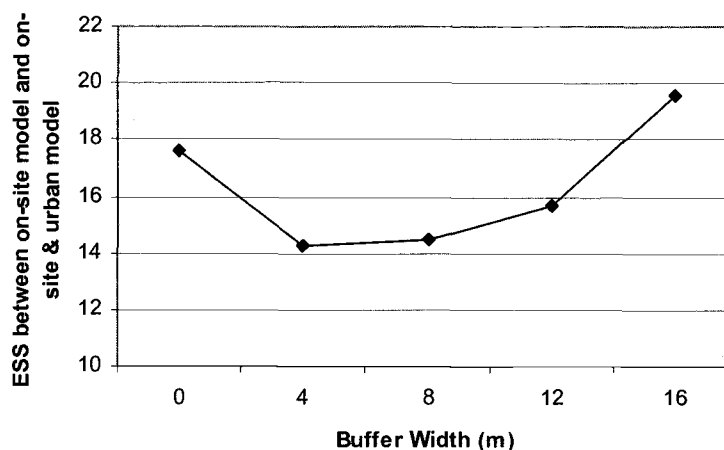
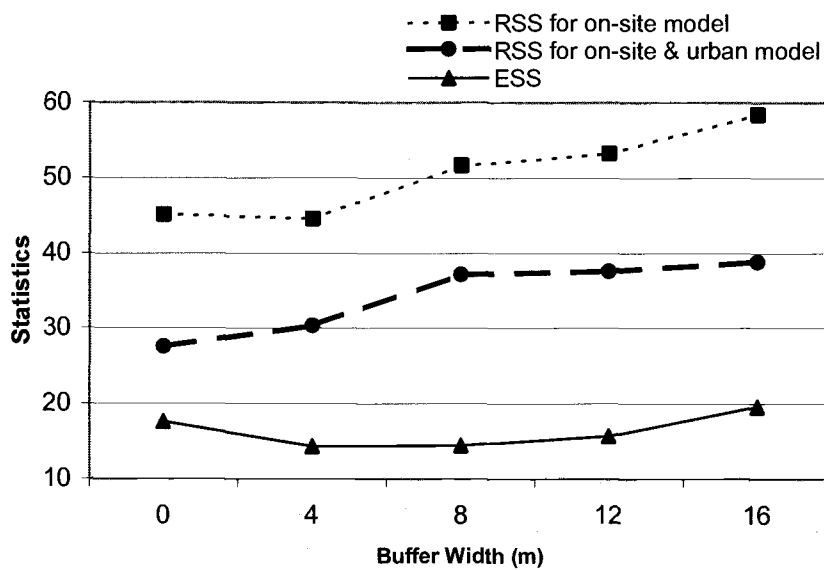


Figure 21: Changes in the unexplained variations of on-site model versus on-site & urban model and significance of urban landuse variable in the on-site & urban model along increasing buffer width.



CHAPTER 6: DISCUSSION

6.1 CUMULATIVE INFLUENCES OF LANDUSES ON *PHALARIS* ABUNDANCE

The result from multiple regression analyses indicated that there are cumulative influences of landuses on *Phalaris* abundance in the downstream wetlands. The cumulative influences represent the watershed-scale landuse influences on the downstream wetlands. The reason of cumulative is twofold. The watershed-scale influences are cumulative because they are made up of integrated landuse influences measured for each 10m by 10m scale landuse. This study also assumes that the landuse influences are cumulative propagating along the flow path from upstream to downstream. The inverse flow distance method was used because it captures the cumulative aspect of landuse influences along their flow paths and can be integrated across the entire watershed. Thus, this study suggests that the impacts of spatially configured landuses are cumulative across the watersheds and along the flow path, thereby increasing *Phalaris* abundance in the downstream wetlands.

Furthermore, an increase in the urban influence was found to be strongly associated with greater *Phalaris* abundance. Urban lands can be characterized by various point- and non-point source pollutions as well as decreased infiltration capacity due to impervious surfaces and improved drainage systems. Urban lands can contribute significant orthophosphate inputs into surface water (Osborne and Wiley 1988; Elosegui and Pozo 1994). In the study site, stream water quality in the urban dominant watersheds tends to be degraded when compared to that in naturally vegetated watersheds. The average of bimonthly water quality data in four consecutive years (from 1997 to 2000) obtained by COE showed that water quality appears to be more degraded in the Amazon Creek main channel than that in the Willow Creek (COE 2000). The metal concentrations, including lead, zinc, cadmium, silver, and gold, appeared to be much higher in the Amazon Creek main channel. On the contrary, the dissolved mineral content, including calcium, magnesium, hardness, total dissolved solids, and specific conductance, were higher in the Willow Creek samples. The watershed for the Amazon Creek main channel is heavily occupied by urban lands, whereas the Willow Creek watershed comprises permanent

vegetation dominated by brushy field, pasture, and forest. Accordingly, the regression analyses indicated that the influence of the pasture and brush fields on *Phalaris* abundance was insignificant. Unmanaged pasture and brush fields can be less important as nutrient sources and hydrologic determinants relative to urban lands, because their permanent vegetative cover intercepts overland flows and convert inorganic nutrients into their biomass (Baulac and Reckhow 1982). Thus, the significant influence of urban lands estimated by the model suggests that the urban development may have facilitated *Phalaris* spread through the alterations of chemical inputs and hydrology in surface waters.

The drainage improvement in the Upper Amazon Creek watershed may have favored the *Phalaris* spread. However, assuming that all landuse types experienced a common impact of drainage improvement, if the drier condition of Upper Amazon Creek watershed was the only cause of *Phalaris* invasion, all study wetlands would be equally susceptible to *Phalaris* invasion. Only the urban influence would not show an association with *Phalaris* abundance. Rather than the change in absolute wetness of the wetlands, the alteration of timing and magnitude can be more important (Owen 1999). Thus, *Phalaris* spread can be induced not only by the drier condition of this modern drainage system but also by greater hydrologic amplitude that likely occurs in the urban lands. In this study, this may also be supported by the significant association of urban lands and the insignificant one of the pastures or brush fields on *Phalaris* abundance.

While altered hydrology and nitrate enrichment have been related to *Phalaris* abundance in previous research, the role of other water quality attributes (e.g., salinity, heavy metals, synthetic organic compounds, and other nutrients) is unknown. However, if degraded water quality reduces the competitive ability of native species, *Phalaris* may invade into the habitat of the resident species. Thus, other chemical inputs than nutrients, which potentially originate in the urban lands, may have also induced the competitive reduction of native species, supporting the role of urban lands in *Phalaris* abundance found in this study.

Although the regression analyses indicated that forestland is negatively associated with *Phalaris* abundance, the interpretation of the role of forestlands is complicated. The negative association of the forest variable with *Phalaris* abundance is contradictory to the fact that the zero subset contained a small range of the forest variable. In fact, the significance of forest landuse is the smallest among all explanatory variables. In addition,

the marginal correlation between forest and urban variables may confuse the sign of the forest variable. Assuming the forest and urban variables are completely uncorrelated, the weak association of forest landuse with *Phalaris* abundance could be due to its small variation across the wetland watersheds.

It has been demonstrated that forestlands can reduce nutrient inputs and bank erosion into stream waters and stabilize hydrologic amplitudes (Baulac and Reckhow 1982; Ludwa 1994). Thus, if the forest variable proves to be truly significant, the result of this study may suggest that the forestlands alleviate the *Phalaris* spread.

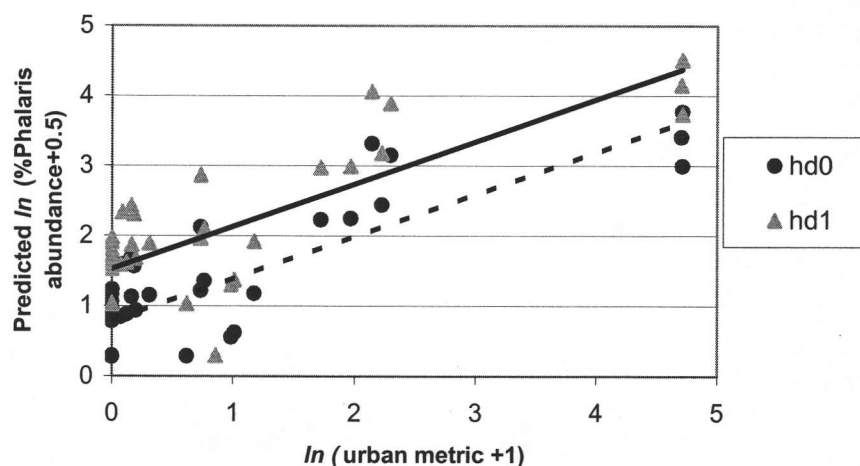
Even though there may be increased nutrient concentrations in wetlands caused by the crop production in the study areas, its impact was not detected. This was probably due to the aggregation of the three landuse classes (annual and perennial grass productions and orchards), which may be different in the influences on surface waters, into the single landuse class of the crop production. Irrigation practices in the orchard lands, plowing in annual grass production lands, as well as the exposed ground on fallow annual grasslands may cause sediment erosion and nutrient loadings. In contrast, the continual grass cover of perennial grasslands may stabilize hydrologic amplitudes and trap sediment and nutrients and may not produce the same level of contaminant loadings. This hypothesis is supported by the study of Kehmeier (2001), which demonstrated the favorable influence of perennial grasslands and the adverse effect of annual grasslands on native benthic fish biomass in the Willamette Valley. The benthic fish biomass may be impacted by the contaminant inputs associated with the orchard lands and annual grass production lands. In contrast to the study of Kehmeier, the differences in the effects of perennial and annual grasslands and orchards in this study may be embedded in the aggregated landuse class; thus, the relationship of each landuse class with *Phalaris* abundance could be masked.

6.2 EFFECTS OF ON-SITE DISTURBANCES ON *PHALARIS* ABUNDANCE

This study also found the significant relationships between some of the physical disturbance classes and *Phalaris* abundance. This suggests that the different physical disturbance classes associated with the historical wetland management have an impact on the current distribution of *Phalaris* abundance. In particular, the analysis shows that highly

disturbed sites are severely invaded by *Phalaris*. Figure 22 shows that *Phalaris* abundance is always greater in the highly disturbed sites than at any given influence of urban lands.

Figure 22: Effects of highly disturbed sites as a function of urban landuse variable. hd0 indicates no highly disturbed sites and is its estimated regression line. hd1 indicates highly disturbed sites and — is its estimated regression line.



The effect of severe hydrologic or physical disturbances on the competitive ability of natives may have resulted in the greater *Phalaris* abundance in the highly disturbed sites. However, this does not imply that these highly disturbed sites are more vulnerable to the impact of urban lands. In this study, highly disturbed sites and the other sites have the identical slopes along the gradient of increasing urban landuse metric. Therefore, the model predicts that all sites are equally subject to the change in the impact of urban landuse.

Although it was not statistically significant, the plowed sites show relatively low *Phalaris* abundance. This low *Phalaris* abundance may be due to the soil excavation and weed control in the restoration activity, which was applied on one of the 6 plowed sites (COE PWE 1999b). In addition, periodical plowing in past agricultural activities may have adversely affected not only native species but also *Phalaris* establishment. If the latter is

the reason for low *Phalaris* abundance in the plowed sites, *Phalaris* may take advantage of the competitive reduction of native species when the plowings are terminated. When this occurs, the linear trajectory of the estimated regression may potentially shift upward for the plowed sites, and *Phalaris* abundance would become much higher at any given impact of urban lands as the model inferred for the highly disturbed sites. Thus, the model suggests that the plowed sites may be susceptible to *Phalaris* spread in the future.

In the non-active field sites, *Phalaris* abundance was found to be significantly lower. The disturbance intensity in the non-active field sites may be insufficient to impair the competitive ability of resident species, and the native vegetation may have been recovering. One might suspect that Willow Creek natural area, one of the most intact wetland in the WEW, did not experience any artificial disturbances in the past. After excluding the Willow Creek unit from the non-active field sites, the overall trend of influences of all four historical disturbances on *Phalaris* abundance remained unchanged. The median *Phalaris* abundance in the non-active field sites remained lower than that of highly disturbed sites (Willcoxon rank-sum test, p -value < 0.0206).

In the grazed/mowed sites, *Phalaris* abundance was relatively variable. No clear relationship with *Phalaris* abundance was found. The historical management of these sites is unclear, and the sites could have been used in various ways such that the degree of disturbance impact on native vegetation and *Phalaris* invasion may not be detectable. However, when compared to other disturbance classes, *Phalaris* abundance in the grazed/mowed sites tends to be low.

6.3 IMPORTANCE OF WATERSHED-SCALE LANDUSES

The edge effect experiment evaluates a buffer function of the wetlands that ameliorates perturbations from the surrounding urban lands. This wetland function could occur greatly at the edge of the wetlands. Therefore, the influence of watershed-scale urban lands on *Phalaris* abundance may diminish in the internal wetlands while the influence of on-site conditions becomes progressively stronger. This decreasing effect of the urban landuse and increasing effect of on-site condition should be measurable, as

increasing significance of on-site condition variables over the urban landuse variable along the increasing buffer width.

Overall, RSSs for both models increase as the buffer width increases, indicating that both the on-site model and on-site & urban model become less able to explain *Phalaris* abundance than the models fitted for the zero-buffer width (Figure 21). This is an expected trend because the models were originally fitted for the particular set of response variable measured at the zero-buffer width. The regressions of the models on the new sets of response variable at different buffer widths likely result in an increase of unexplained variation. However, the significance of urban landuse variable showed an unexpected trend. Although the significance of urban landuse variable decreased once, it started increasing again from 4 m of buffer width (Figure 20). The RSS for the on-site & urban model slightly departs from that for the on-site model for buffer widths greater than 4 m (Figure 21). This may indicate that the performance of on-site & urban model is relatively stable after this point, whereas that of the on-site model continues to deteriorate.

The differences in direct and indirect disturbances may explain the result of edge effect experiment. The direct perturbation from the adjacent landuses includes non-native species invasion, physical disturbances by grazing, trampling, and mowing (Pearson and Leoschke 1992). Indirect impacts include altered hydrologic amplitude and degraded water quality (Keddy 1983), which include hydrology-related influence of watershed-scale landuses. By buffering a wetland, the direct disturbances from the adjacent urban lands may be removed, resulting in the decreased significance of the urban lands observed for the buffer widths from zero to 4 m. However, the indirect effects of urban lands may remain for the buffer widths greater than 4 m. Although it is impossible to separate the direct and indirect impacts in this study, the edge effect experiment suggests that the indirect effects including hydrology-related impacts accumulated over urban lands could be persistent, even in the internal wetlands.

CHAPTER 7: CONCLUSIONS AND IMPLICATIONS FOR WETLAND MANAGEMENT

The regression analysis indicated that watershed-scale landuses have the potential to influence *Phalaris* abundance in the study wetlands. The landuse variables were quantified in the way that represent the cumulative influences across the watersheds from upstream to downstream. The landuse variables are spatially explicit and express hydrologic connectivity across the watersheds. Thus, the model describes that the influences of spatially configured landuses on *Phalaris* abundance may occur through hydrology-related influences, such as the alterations of chemical inputs and amount and timing of inflows.

Furthermore, urbanization was strongly correlated with *Phalaris* abundance. *Phalaris* stands can spread and extirpate native species in the wetlands due in part to degraded water quality and altered hydrologic conditions that were presumably derived from the watershed-scale urban landuses. Competitive reduction of native species may have also occurred in the wetlands where experience altered abiotic conditions in surface waters and on-site physical disturbances, contributing to the *Phalaris* spread. The impact of urban lands became more predominant in the internal wetlands relative to the influence of on-site conditions, suggesting that the watershed-scale landuses surrogating for hydrology-related effects may be persistent.

Because of the limited number of observations and data availability, the model precluded potentially important environmental variables. It cannot be concluded that the explanatory variables excluded in the preliminary selection have no relationship with *Phalaris* abundance. These variables were excluded because they were found to be less important than the selected variables. Furthermore, an investigation of the role of flood disturbances and distribution of seed and propagule sources may be important to improve the accuracy of the model. It would also be interesting to consider the presence or absence of *Phalaris* stands because zero *Phalaris* abundance could be ecologically meaningful. Lastly, from the model, it is impossible to draw conclusions about the relative importance of water quality, hydrologic regime, and physical disturbances.

In light of relationship between *Phalaris* abundance and watershed-scale landuses, on-site management alone is insufficient. The impact of urban lands at watershed-scales must be addressed, otherwise it is unlikely that *Phalaris* will be controlled and the native plant community restored. More significantly, if *Phalaris* spread causes a floristic degradation in the wetlands, it is contradictory for a restoration objective to address both quality of wetland flora and wetland's filtering function reducing the impacts of urban runoff. Wetlands with strong urban influences likely experience *Phalaris* invasion, thereby potentially reducing the diversity of native species. Thus, there will be a tradeoff between the degradation of the native plant community and wetland treatment of urban runoff.

The provision of forestlands along streams and wetlands may suppress *Phalaris* invasion associated with the degraded water quality and hydrologic regime in urban areas. However, this study showed that the spatial configuration of landuses and hydrologic connectivity at watershed-scale were also important to control *Phalaris* in the wetlands. Therefore, an overall watershed plan focusing on the impacts of water quality and hydrology on the wetland physical conditions should be considered for *Phalaris* control. To ameliorate the watershed-scale landuse impacts on surface waters and ultimate wetland physical conditions, an increase in natural vegetation along the streams and wetlands, a reduction of paved surface and a removal of runoff pipes from the wetlands are recommended. Moreover, the restoration objectives in the WEW plan are not site-specific. The wetland designation for restored and protected are based upon the overall ratings of multiple wetland functions including both plant species diversity and urban runoff treatment. Considering the potential impacts of urban runoffs on native species diversity, this study recommends that the wetlands to be restored or protected for native plant diversity are located where urban influences are not expected. Use of this watershed-scale approach and awareness of cumulative impacts of degraded water quality and hydrology will be important for an effective *Phalaris* control and stable native plant diversity in the remnant wetlands.

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