AN ABSTRACT OF THE THESIS OF

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Wetlands in the Willamette Valley, Oregon.

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John Boone Kauffman

I examined hydrological and plant community changes associated with the implementation of a restoration management plan in two riparian meadows located within an agricultural landscape of the central Willamette Valley, Oregon. I established exclosure fencing (a form of passive restoration) in one agricultural field and established fencing and plugged a drainage ditch (active restoration) in a separate agricultural field. Permanent transects 15 m in length were established within two plant communities associated with hydrological regimes within these restored agricultural fields. Plant communities were classified as wetland meadow (inundated for more than 4 weeks/year) and mesic meadow (saturated within the upper 30 cm but not inundated) for at least 4 weeks/year. Four transects were randomly established within the wet meadow community and 6 transects were randomly situated within the mesic meadow community. Two shallow sub-surface piezometers were installed to a depth of 1 m at 5 m and 10 m along each of these 15 m transects. Additionally, two shallow sub-surface piezometers were established at the outer perimeter of the agriculturally excluded fields. Shallow

sub-surface and surface water table levels were measured at each piezometer after wetlands were inundated and continued until water table dropped below the piezometers (Dec. – June) for one pre-treatment and two post treatment years. The actively restored wet and mesic meadows demonstrated increased water table elevation and a decrease in water table fluctuation during both post treatment years. Increases in water table elevation were greatest in areas closest to active restoration but were significant up to 102 m. from restoration. Results indicate that filling drainage ditches induce hydrologic effects at great distances across floodplain soils.

Plant community composition (species response) was quantified in both restored sites as well as the adjacent agriculturally managed (untreated) sites one year before treatment and two post-treatment years. I sampled two plant community types: wet meadow and mesic meadow. I calculated species richness and the relative abundance of wetland indicator species, nuisance weeds, and native plants. Nuisance weeds increased and native plant abundance decreased in agriculturally managed mesic meadows. Wetland plant species abundance tended to increase in agricultural sites with light grazing, and decreased in areas that were plowed and re-seeded. Native plants increased and nuisance weeds decreased in the actively restored mesic meadow. The passively restored mesic meadow exhibited no change in native plant abundance and decreases in all other categories. In the actively restored wetland there were increases in plant species richness and nuisance weed abundance with a decrease in native plant abundance. Agriculturally excluded wetlands dominated by Reed canary grass (*Phalaris arundinacea*) exhibited no changes for the entire study period. Results suggest that for the first few years following agricultural exclusion, nuisance weed species do not increase, but active restoration may result in increases (due to disturbance). Additionally, results indicate restored agricultural landscapes dominated by introduced grasses demonstrate minimal short-term plant community change unless initiated by intense land management practices (e g., plowing, re-seeding, or removal of dominant plant communities).

Based upon results of this study, I conclude that restoration plans should repair damaged hydrological features before planting riparian plant species. Following this chronological sequence will minimize the potential destruction of planted communities by future shifts in water table elevation caused by hydrologic restoration. Furthermore, any active restoration that initiates a direct or indirect removal of the dominant plant community should be accompanied by aggressive plantings of desirable plant species and prolonged site maintenance.

# Hydrologic and Vegetation Responses Associated with Restoration of Wetlands in the Willamette Valley, Oregon.

By

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## 1. INTRODUCTION

# Hydrologic and Vegetation Responses associated with Restoration of Wetlands in the Willamette Valley, Oregon.

Riparian wetlands are critical areas for preserving water quality in the USA (Gilliam 1994). Riparian zones and wetlands have been shown to be effective as nutrient filters, at mediating the flow and temperature of adjacent streams, and as important wildlife/fisheries habitat (Stanley 1989, Kauffman 1996, Kauffman et al. 2000). In addition, riparian wetlands such as those that occur in the Willamette Valley of Oregon perform a number of ecosystem functions including their influence on aquatic and terrestrial biogeochemistry, biological diversity, and fluvial hydrology (Elmore and Beschta 1987, Gregory et al. 1991, NRC 1996). Wetlands in riparian zones have a significant influence on stream water quality. Riparian wetlands have been shown to retain sediment (Peterjohn 1984, Karr and Schlosser 1978), reduce nitrogen levels in surface and subsurface water through denitrification (Schnabel et al. 1994, Hanson 1994) and plant uptake (Lowrance 1984, Groffman 1992). Riparian wetlands have also been shown to possess higher concentrations of microbial biomass than adjacent agricultural areas (Horwath et al. 1998). Intact riparian wetlands in broad floodplains may also increase summer flows and decrease peak flows of streams through storage, attenuation, and base flow maintenance (Debano and Schmidt 1989, Schlosser 1981, McNatt 1980). Gilliam (1994) stated that the riparian wetland is the most important factor influencing nonpoint-source pollutants entering surface waters in many areas in the USA and the most important wetlands for surface water quality protection. Because of their multiple values riparian wetland restoration has become of great interest.

Wetlands are relatively rare and are estimated to only cover 6% of the land surface of the world (Maltby and Turner 1983, Mitsch and Cronk 1992). Prior to European settlement the amount of land covered by wetlands in the United States was estimated to have been between 60 and 75 million ha. Today, approximately 42 million ha of wetlands exist In the United States. This reflects a decrease of between 54 - 62% of the original land occupied by wetlands (Mitsch and Cronk 1992, Madsen 1986, Tiner 1984). Most wetland losses (about 87%) have been due to agricultural development (Tiner 1984). Agriculture has always been and continues to be the greatest contributor to wetland loss (Tiner 1984). A national wetland policy forum was convened in 1988 by the Conservation Foundation (CF) at the request of the U.S. Environmental Protection Agency (USEPA; National Wetlands Policy Forum 1988, Davis 1989). This meeting set significant goals for the remaining wetlands of the United States. The Forum recommended a policy: " to achieve no overall net loss of the nation's remaining wetlands base and to restore and create wetlands, where feasible, to increase the quantity and quality of the nation's wetland resource base"(National Wetlands Policy Forum, 1988). This shifted the activities of a great number of agencies such as the Department of the Interior, the USEPA, the U.S. Army Corp of Engineers, and the Department of Agriculture (Mitsch and Cronk 1992). It was not anticipated that there would be a complete halt to the draining of wetlands when economic or political reasons deemed other uses to be a more valuable use of the land. Therefore the "no net loss" concept equates to an increase in wetland restoration and creation (Mitsch and Cronk 1992). Although wetland restoration has become an important part of land management in the United States, little is known regarding what constitutes "success" in restored wetlands (Roberts 1993, Mitsch and Wilson 1996, Young 1996, Zedler 1996, Malakoff 1998).

Much research has been conducted regarding the possible modes for restoring wetlands. Kauffman et al. (1996) described ecological approaches to riparian restoration. In this paper the authors suggested that any good restoration plan would first initiate a process known as "passive restoration" (Figure 1.0). Passive restoration involves halting those land-use activities that may be responsible for degraded conditions in ecosystems or preventing their recovery. When the results from this passive restoration are inventoried and goals are not met then a more aggressive approach termed "active restoration" should be pursued.

For this study we examined combinations of passive restoration and active restoration on two mesic meadows in agricultural landscapes of the central Willamette Valley, Oregon (Figure 1.1). Both study sites had a long history of agricultural land use that included livestock grazing, haying, and ditching. These sites were riparian areas within agricultural landscapes containing mesic meadow plant communities dominated by *Alopecurus pratense* and *Lolium perene* and riparian wetland plant communities dominated by *Alopecurus aequalis* at the covered bridge site and *Phalarus arundinacea* at the EPA site. Our initial approach was to erect exclosure fencing and eliminate impact from agricultural activities in both research sites (passive restoration). In addition, at research site 1 we obtained wetland enhancement and water storage permits and increased the storage capacity of the existing wetland by removing soil adjacent to the wetland. Soil collected from the site was then used to fill a drainage ditch at the same wetland (active restoration).

The three goals of this study were:

Goal 1: Determine the influence that active hydrologic restoration has on shallow sub-surface hydrology of wetlands and adjacent mesic meadows.

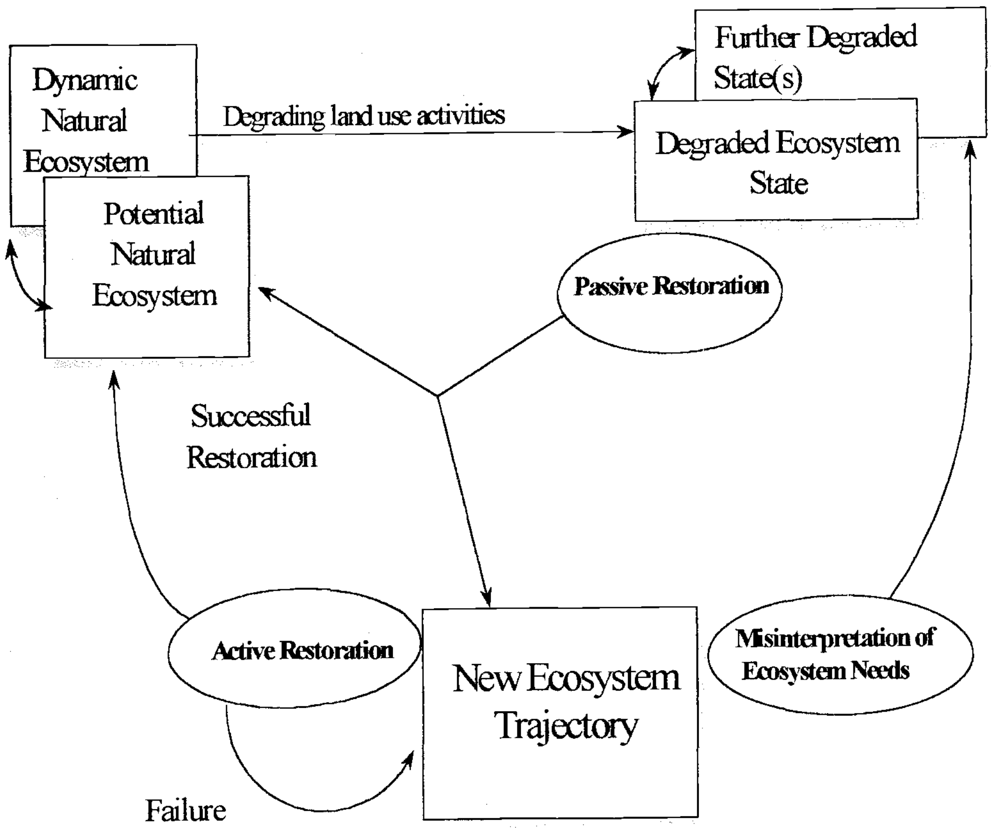
Goal 2: Determine if active hydrologic restoration of wetlands increased the occurrence of desirable plant species composition and diversity.

Goal 3: Determine if passive restoration of wetlands increased the occurrence of desirable plant species.

In chapter 2, I address the first goal of this study and evaluate the ground water dynamics associated with active restoration of ditched agricultural wetlands. My research design consisted of fifty-five shallow sub-surface wells installed within two restored agricultural landscapes. One area was passively restored (agricultural activities halted) and the other had a combination of passive and active restoration applied (agricultural activities halted and wetland storage increased and ditches filled). I sampled water table elevation at all wells in weekly intervals during the rainy season (Dec – May) and until wells were dry. Sampling was conducted during the rainy season before implementation of active restoration and for two seasons post implementation. My hypothesis was that active restoration of hydrologic features (i.e. filling of drainage ditches) in agricultural landscapes would contribute to a higher water table elevation and reduce the variation of the water table elevation in the site restored as well as in adjacent landscapes.

In chapter 3, I address the final two goals of this study and discuss the plant community dynamics associated with implementation of passive restoration and active restoration techniques. In order to demonstrate the differing consequences these restoration techniques exhibit, I sampled plant community composition before restoration activities and then continued sampling for the following two years after the restoration techniques had been implemented. My research design consisted of a series of transects situated within restored areas as well as a series of transects situated within fields still actively managed for agriculture. I hypothesized that restoration would have a strong influence on plant community composition in both the mesic meadows and wetland areas of this agricultural landscape.

In chapter 4, I discuss the conclusions and overall summary of the study.




(Figure 1.0) Hierarchy of methods for successful restoration implementation. Modified from Kauffman et al 1996.

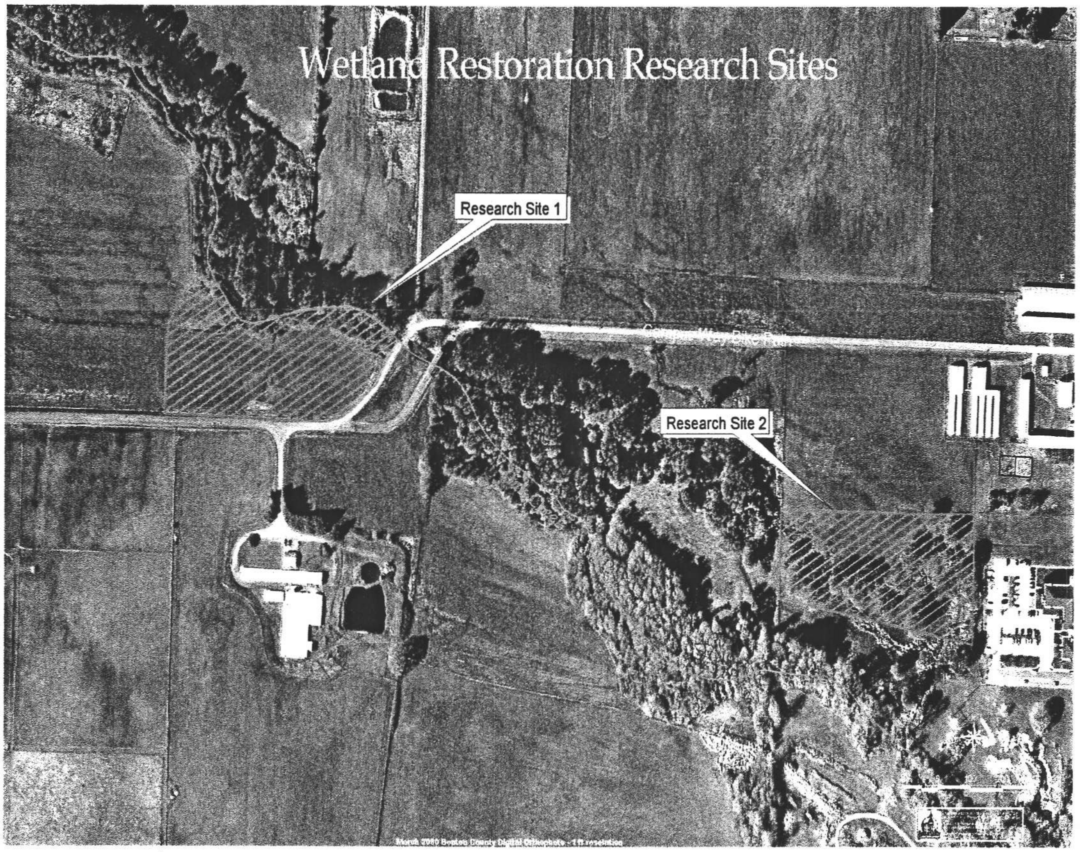



Figure 1.1. Riparian wetland research sites. The actively restored site (covered bridge) is denoted Research Site 1 and the agriculturally excluded site (EPA) is denoted Research Site 2. Control sites were agriculturally managed mesic meadow 1 (directly east of Research Site 1) and agriculturally managed mesic meadow 2 (directly north of Research Site 2).

# 2. HYDROLOGY CHANGES ASSOCIATED WITH RESTORATION OF WETLANDS IN THE WILLAMETTE VALLEY, OREGON.


## Abstract

I examined hydrological changes associated with the implementation of restoration management in two riparian meadows located in agricultural landscapes of the central Willamette Valley, Oregon. I established exclosure fencing (a form of passive restoration) in one agricultural field and established fencing and plugged a drainage ditch (active restoration) in a separate agricultural field. Permanent transects 15 m. in length were established within each particular hydrologic regime within these restored agricultural fields. Hydrologic regimes were classified as wetland meadow (inundated for more than 4 weeks/year) and mesic meadow (saturated within the upper 30 cm but not inundated) for at least 4 weeks/year. Four transects were randomly established within both wetland communities and 6 transects were randomly situated within both mesic meadows. Two shallow sub-surface piezometers were installed to a depth of 1 meter at the 5-meter and the 10-meter mark along each of these 15 m transects. Additionally, two shallow sub-surface piezometers were established at the outer perimeter of the agriculturally excluded fields. Shallow sub-surface and surface water table levels were measured at each piezometer after wetlands were inundated and continued until piezometers were dry (Dec. – June) for one pre-treatment and two post treatment years. Restored wet and mesic meadows demonstrated increased water table elevation and a decrease in water table fluctuation during both post treatment years. Increases in water table elevation were greatest in areas closest to active restoration but were significant up to 102 m. from restoration. Results indicate that filling drainage ditches induces hydrologic effects great distances across floodplain soils and the establishment of Reed canary grass may affect hydrologic character.

## Introduction

The riparian wetland represents a transitional zone between terrestrial and aquatic ecosystems. As the transitional zone between these ecosystems, riparian wetlands can have major effects on the quality and quantity of water in downstream systems (Loucks 1989, Gilliam 1994). Riparian wetlands have been shown to have a high capacity to filter and/or remove nitrogen in surface and subsurface flows from agricultural areas (Groffman 1992, Hanson 1994, Hubbard 1995, Jacobs 1985). Nitrate and other forms of N are reduced, sequestered, or transformed in riparian wetlands through several means. Riparian wetlands have a large capacity to denitrify incoming waters through denitrification (conversion of $NO_3^-$-N into gaseous states by facultative anaerobic microorganisms) and by microbial immobilization (Groffman 1992). Vegetative uptake of N is also a major factor contributing to N losses in the riparian wetlands (Lowrance 1984, Groffman 1992). Riparian wetlands also reduce or slow velocities of incoming water sources both above and below ground promoting suspended particulate to fall out of transport. Gilliam (1994) stated that the riparian wetland is the most important factor influencing nonpoint-source pollutants entering surface water in many areas in the USA and the most important wetlands for surface water quality protection.

Riparian wetlands have also been shown to be important wildlife and plant habitats (Guynup 1999). Kauffman et al. (2000) found that 70% of all wildlife species utilized riparian wetlands in Oregon. W.J. Mitsch (1986) reported that wetlands provide a haven for a wide variety of flora and fauna and offer a unique habitat for many rare and endangered species (Mitsch and Cronk, 1992). Kirkland (1999) found a significant correlation ($r^2$=0.80) between the number of federally listed mammals (Endangered Species List) in the United States and the loss of wetlands in those areas. Mitsch et al. (1998) showed an increase in plant diversity from 13 planted species to 65 total species within 3 years, representing a net increase of 500%. This same wetland study showed an increase of 17 bird species due to the creation of wetlands. Macro-invertebrates have also been shown to increase significantly with the addition of wetland habitats (Mitsch et al. 1998). Wildlife enhancement is typically a benefit of wetland restoration projects and is usually coupled with other specific goals. Mitsch (1992) described typical goals obtained from wetland restoration as: flood control, waste water treatment, storm water or non-point pollution control, ambient water quality improvement, wildlife enhancement, fisheries enhancement, replacement of similar habitat, or research.

The benefits of wetlands have not always been known or appreciated. Moreover, wetlands have historically been considered "waste places" without any true value. Because of this view the United States passed the Swamp Land Acts of 1849, 1850, and 1860. These acts promoted agricultural drainage as a beneficial management tool and have played a significant role in the loss of wetlands in the United States. The Swamp Lands Acts catalyzed the draining of agricultural land with incentives of free lands to state governments if these lands were drained and "reclaimed" for cultivation. This trend was further catalyzed in 1888 when John Klippart's book the "Principles and Practice of Land Drainage" described the 12 benefits of land drainage (Prince 1997):

- Removes stagnant water from the surface.
- Removes surplus water from the undersurface.
- Lengthens the growing season.
- Deepens the soil.
- Warms the undersoil.
- Equalizes the temperature of the soil during the growing season.
- Carries down soluble substances to the roots of plants.
- Prevents freezing out or heaving out.
- Prevents injury from drought.
- Improves the quality and quantity of crops.
- Increases the effects of manure.
- Prevents rust in wheat and rot in potatoes.

These 12 perceived benefits have prompted many agricultural managers to employ methods to drain riparian wetlands. Ditching (constructing ditches to facilitate rapid draining) of these wetlands has been a ubiquitous means for accomplishing these goals. Pavelis (1987) reported that 44.5 million ha. of rural wetlands had been drained in the United States by 1985. Of that 44.5 million ha., 28.3 million ha. were cropland and over 65% of that land had been drained using surface ditching (Hey and Philippi 1999).

Riparian wetlands in the Pacific Northwest are no exception. Many riparian wetlands in the Pacific Northwest have become impaired by land use activities (Baker et al. 1995). In Oregon over 38% of all wetlands have been destroyed or otherwise eliminated (Dahl 1990). The agricultural riparian wetlands of Oak Creek, which flows through the campus of Oregon State University (OSU) displays a similar trend. OSU lands have been managed for agriculture and livestock production since the late 1860s and the dairy and beef confined animal feeding operations have been in operation since the 1930s. Riparian wetlands have been drained and riparian areas have been cleared to establish pasture. Many studies have documented the ability of surface drainage ditches to drain wetlands. However few studies have focused on the hydrologic changes associated with plugging the ditches and "reclaiming" the wetland.

I examined hydrological changes associated with the implementation of restoration management in two agricultural riparian meadows located in the central Willamette Valley, Oregon. I established exclosure fencing (a form of passive restoration) in one agricultural field and established fencing and actively restored damaged hydrological features (filled ditches) in a separate agricultural field (Figure 2.0).

I then monitored hydrologic dynamics in both fields pre-treatment and two post-treatment years. The objectives were: (1) determine whether this restoration technique actually contributed to increased water table elevation (2) determine if this restoration technique reduced the water table variability of this drained wetland.

# Methods

## Study sites

The study sites are two riparian wetland meadows within an unconstrained reach of Oak Creek. Oak Creek flows from the Coast Range foothills into the central Willamette Valley at Corvallis, Oregon. Oak Creek is a third order stream of approximately 13.4 kilometers in length. Oak Creek is a tributary to the Mary's river, which is within the Willamette River Basin. The total drainage of Oak Creek is 33.2 square kilometers. Average annual precipitation for this area is 107.4 cm. (Yamaguchi 1993). The highest point in this watershed is about 549 m. with the steepest slopes concentrated in the forested headwater regions. Oak Creek flows from forested headwaters through an extensive agricultural area then flows through an urban portion of Corvallis, Oregon. These research sites are located within the "agricultural reach" of the Oregon State University campus between 35$^{th}$ and 53$^{rd}$ streets Corvallis, Oregon.

Site 1 (covered bridge) is approximately 2 ha in area and site 2 (EPA) is approximately 3 ha. Site 1 was located just upstream of the Covered Bridge on Campus way on the south side of Oak Creek. Two plant communities dominate the covered bridge site: (1) mesic prairies dominated by rye grass (*Lolium perenne*) and meadow foxtail (*Alopecurus pratensis*) as well as many exotic herbs; and (2) seasonally flooded wetlands dominated by little foxtail (*Alopecurus aequalis*). Before restoration the covered bridge site had a seasonally inundated open water wetland with approximately 775 $m^2$ of surface area. Restoration activities increased the surface area of this open water wetland to approximately 1332 $m^2$. The EPA site was a relict channel and floodplain ~ 400 m. down stream and on the north side of Oak Creek. Both sites are relatively flat and have roughly equal amounts of sunlight. Mesic meadows within the EPA site had plant communities similar to those of the covered bridge site but the flooded wetland plant community was completely different and dominated by Reed canary grass (*Phalaris arundinaceae*). Oregon white oak (*Quercus garryanna*) approximately 120 years old was dispersed on dry elevated microsites throughout the EPA site. A mix of riparian obligate trees Oregon ash (*Fraxinus latifolia*), Black cottonwood (*Populus balsamifera)*, Pacific willow (*Salix lasiandra*), and Red alder (*Alnus rubra*)) are along the creek and ecotonal to the covered bridge site. Adjacent to the treated sites are sites managed for grass/livestock production and are frequently plowed, fertilized, grazed, and hayed. Dairy, beef, and swine production facilities are located adjacent to these sites.

Soils in all sites are Waldo-Bashaw silty clay loams <2% slopes and are included in the local hydric soil list (Benton County soil survey). The surficial geologic units of this area is quaternary lower terrace deposits of semiconsolidated cobbles, gravel, sand, silt, clay, and organic material approximately 11 m thick above recent river alluvium (Buckley 1994).

## Restoration approaches

We wanted to examine ecosystem responses to restoration approaches that would be relatively inexpensive and easily implemented by landowners and managers. Our approaches to restoration follow those being implemented on many private lands with technical assistance provided by the Oregon Department of Fish and Wildlife (Steve Smith, ODFW, personal communication), the Natural Resource Conservation Service, and other agencies.

In 1999 we fenced the agricultural riparian meadows and halted all grazing, plowing, seeding, and chemical applications on the sites. In 2000, active restoration of site 1 was performed to repair hydrological damage caused by a drainage ditch and fill. We removed soil material from the east-end of site 1 and placed soil into the head-cut (erosion) and ditch at the north end of the site effectively removing the drainage ditch. We removed the soil in an area that facilitated the accumulation of surface water and also directed out flow to be discharged into the existing ditching system located on the outer perimeter of the site (Figure 2.0).

The active restoration activities at the covered bridge site were:

(1) The excavation of soils adjacent to the existing open water wetland.

(2) The filling of a drainage ditch with soil excavated from the wetland.

This resulted in increased storage capacity and open water surface area combined with a decrease of outflow from the open water wetland.

## Water Table Elevation

Water table elevation was determined during each winter for the pre-treatment and two years post-treatment at both sites. Permanent transects 15 meters in length were established within the major plant community types. Each research site had 10 transects established within the major plant community types. Each wetland community had 4 transects randomly established within their perimeter and each mesic meadow had 6 transects established in a grid fashion within their perimeter. Along each transect, two shallow sub-surface piezometers were placed at 5 meter intervals at the 5 and 10 meter mark for each transect. In addition two piezometers were placed at the edge of the fields nearest the outer perimeter. Piezometers were installed to a depth of 100 cm and capped with ventilated PVC covers. Piezometers casings were constructed from 2.2 cm diameter PVC pipe, drilled with 0.48 cm holes along the entire buried length. Water table elevation was measured with a wooden dowel that had two copper wires affixed along its length and was connected to an ohmmeter at the top. This dowel was lowered into the piezometer until contact with the water table initiated conductivity to the ohmmeter. Depth of contact with water was recorded. Water table elevation sampling began at the onset of winter rains after water was visibly standing in the field and was measured weekly thereafter until piezometers were dry. Moreover, the focus of our sampling strategy was to monitor the wetland during sustained saturated and/or inundated conditions while evapotranspiration rates were minimal and rains had already adequately saturated the wetlands. This was hypothesized to be the period when storage and outflow would control the system rather than precipitation and evapotranspiration.

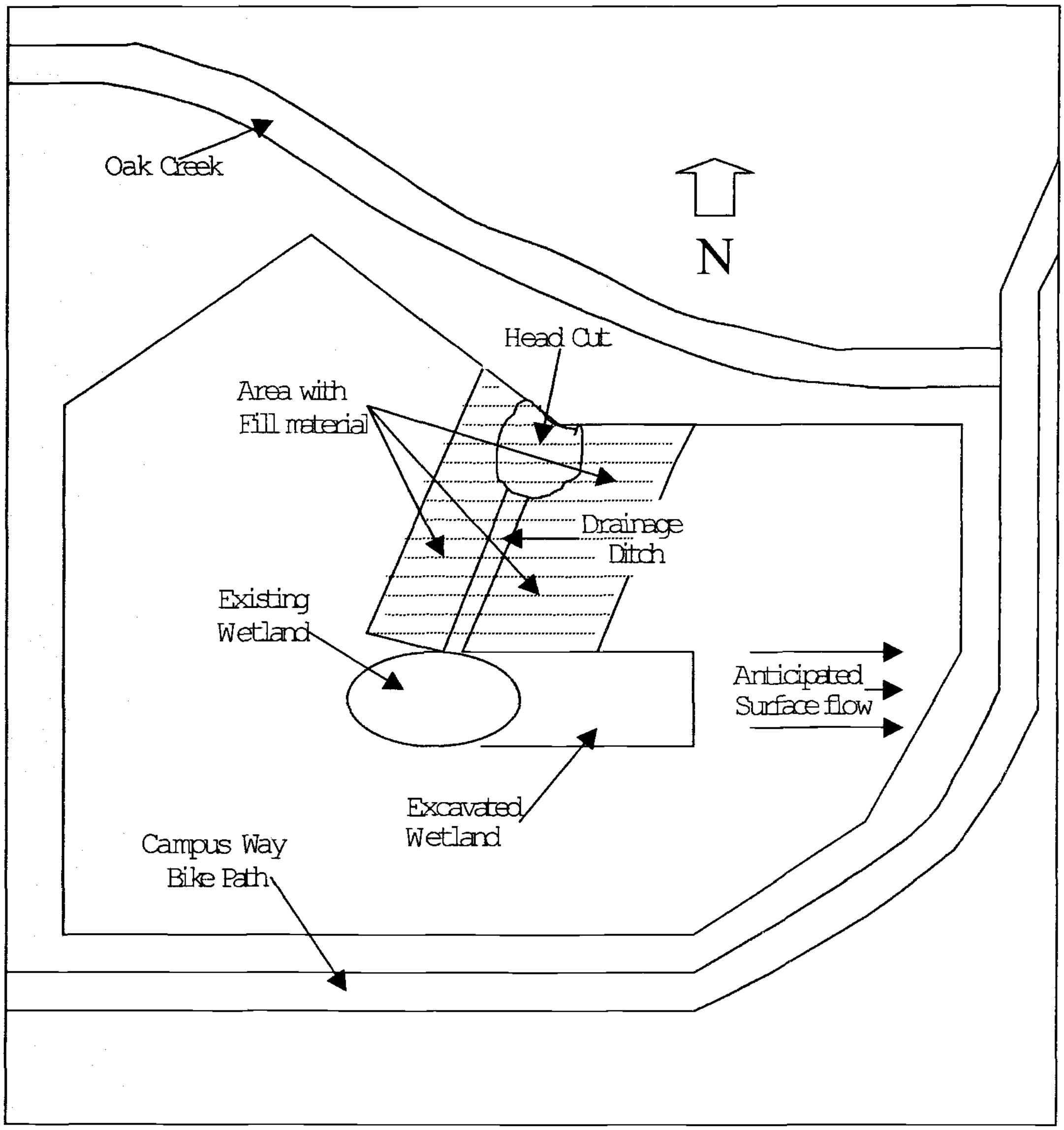



Figure 2.0. Diagram showing the Covered Bridge site with the extent of excavation and where excavated soils (fill dirt) were placed to fill head cut and ditch.

## Data Analysis

Water table elevation was graphed for all piezometers. Variation between piezometers was statistically evaluated using an F-test. Mean water table elevation was calculated for each research site using the mean of all piezometers that had similar variances (alpha=0.5). To test for differences through time the mean water table elevation for each research site was evaluated using a paired two-sample t-test for means (Ramsey and Schafer 1997). Year of data collection was evaluated as the independent variable and water table elevation was the dependent repeated variable. To test for differences in water table variability, mean water table elevation during sustained saturation and/or inundation conditions was evaluated using an F-test. This method was hypothesized to represent the degree of hydrologic stability accomplished by our restoration approach. Assuming that more flashy systems would have a larger variance (higher highs and lower lows) and an intact wetland would exhibit more moderate responses this would be a measure of the flashiness of that particular system. These metrics were graphed and then statistically analyzed.

Ideally, a study such as this would control as many variables as possible to minimize their influence on the conclusions. In situ research has many limitations because of the inability to control variables such as precipitation and temperature. This study was also limited because there were very differing precipitation patterns during the years of the study. Although annual precipitation values were similar during the study period (Table 2.1), water year average precipitation values were not. Water year average precipitation values for this area is 101.9 cm. During the 2000 water year near average precipitation (108.2 cm) occurred at the research sites. However, the following year (2001) had the lowest precipitation on record (58.4 cm). Water year 2001 was followed by a near average water year in 2002 with 115.6 cm of precipitation (Figure 2.1). These differences in precipitation values mandated statistical evaluation using the average water table elevation between the water years 2000 and 2002 rather than including water year 2001. Because precipitation values were so distinctly different only subjective conclusions can be made regarding hydrological changes associated with restoration management between years 2000 and 2001, and between years 2001 and 2002. However, these dramatic differences allow one to observe hydrological responses to extremely contrasting rainfall regimes.

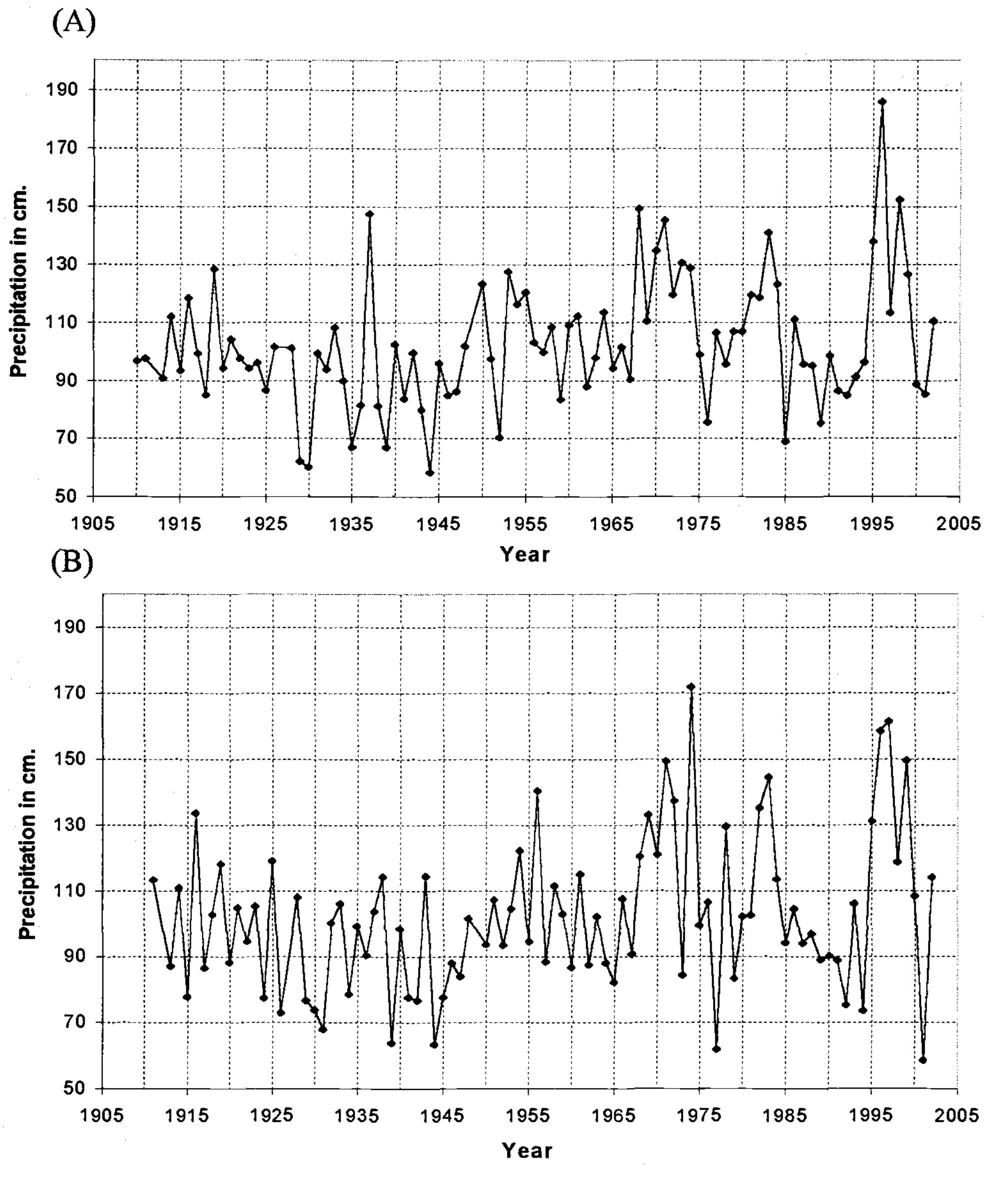



Figure 2.1. Annual precipitation (A) and water year precipitation (B) values for Corvallis, Oregon (1910 – 2002). Data are from the Hyslop Experimental Station, Oregon State University. Study years are 2000, 2001, and 2002.

## Results

Maximum water table elevation exhibited very different values between study years. The maximum water table elevation for the non-restored mesic meadow (EPA) pre-treatment year 2000 was - 251mm. During the following year the maximum water table elevation had dropped to - 438 mm. During the final study year the maximum water table elevation was - 142 mm (Figure 2.2). These trajectories exhibited by the water table elevation of this site clearly follow the observed pattern of precipitation over the study years. Pre-treatment year 2000 had a higher maximum than post treatment year 2001, and was followed by an additional increase and the highest maximum water table elevation during post treatment year 2002. This same trajectory was seen in the precipitation values from these years of 108.2, 58.4, and 115.6 cm respectively (Figure 2.1).

Patterns in water table elevation were dramatically different in the actively restored mesic meadow (covered bridge) when compared to the non-restored sites. Maximum water table elevation for the restored mesic meadow at the covered bridge during pre-treatment year 2000 was - 4 mm. For post treatment year 2001 the maximum water table elevation was + 7 mm and post treatment year 2002 had a maximum water table elevation of + 80 mm (Figure 2.4). This demonstrates the heightened capacity of water storage presumably induced by active restoration techniques performed at this site. Although there was significantly less rainfall during post treatment year 2001, there was more water stored in this site. Pre-treatment year 2000 exhibited no standing surface water while both post treatment years exhibited above ground standing water. Maximum water table elevation was higher despite lower precipitation values for the first post treatment year and when approximately equal precipitation values were experienced (2000 and 2002) this site demonstrated a significantly higher water table elevation when compared to pre-treatment.

Maximum water table elevation for the non-restored *Phalaris arundinacea* wet meadow (EPA) during pre-treatment year 2000 was + 184 mm. During post treatment year 2001 this same site had a maximum water table elevation of + 143 mm. Post treatment year 2002 exhibited a maximum water table elevation of + 254 mm. (Figure 2.3). This non-restored wet meadow exhibited a trajectory through the study years that closely emulated that of the precipitation values. This would suggest that no significant changes in water storage capacity had occurred at this site during the study period.

Maximum water table elevation for the restored *Alopecurus aequalis* wet meadow was 170 mm during the pre-treatment year 2000. This restored wet meadow exhibited a maximum water table elevation of + 190 mm and + 241 mm for post treatment years 2001 and 2002, respectively (Table 2.5). These data demonstrate a higher maximum water table elevation for post treatment year 2001 compared to pre-treatment year 2000 notwithstanding the discrepancies in precipitation between those years (58.4 and 108.2 cm, respectively). Maximum water table elevation for post treatment year 2002 was significantly higher than either previous study years despite of almost equal precipitation in year 2000 and 2002 (108.2 and 115.6 cm. respectively).

These measurements indicate that both wet meadows and mesic meadows located in close proximity to active hydrological restoration (filling ditches) demonstrate a positive hydrologic response. Both restored wet and mesic meadows exhibited an increase in maximum water table elevation for both post treatment years in spite of below average precipitation in post treatment year 2001 (Figure 2.1). Both control sites had water table elevation characteristics that closely emulated that of the precipitation regime.

Average water table elevation across each of the research sites during pre-treatment year 2000 exhibited a high degree of correlation to precipitation events. That is to say, when precipitation events occurred large fluctuations in water table elevation were recorded. Figures 2.6 through 2.9 illustrate that consonance between precipitation and water table elevation for all sites during pre-treatment year 2000. However, study sites that had hydrological features restored showed no such correlation for post treatment years (Figures 2.6 – 2.9). This would suggest that water table elevation response due to precipitation events was dampened after restoration activities had occurred. Evidence leads me to conclude that that restoration of hydrological features (filling drainage ditches) tends to promote storage of greater amounts of water for longer periods of time thus mediating the effects of any one precipitation event.

Precipitation and its effect on the water table elevation of these research sites were further evaluated through statistical analysis. Figures 2.6 – 2.9 display the average water table depth in consonance with precipitation events. This series of graphical figures displays the variation in water table depth as a consequence of precipitation arriving to the area. F-tests for differences in variation were performed in order to ascertain the level of variation that the restoration activities reduced at these sites. Under the assumption that flashy systems fluctuate rapidly as a result of precipitation events and wetlands with restored hydrology display mediation from that flashiness, the variance would then be a measure of the degree of flashiness. By testing variability with a simple F-test one can determine statistically the degree of variation and the possibility of a particular system exhibiting or not exhibiting similar characteristics in other study years. Using the null hypothesis that variances are equal, F-tests were performed on relevant data.

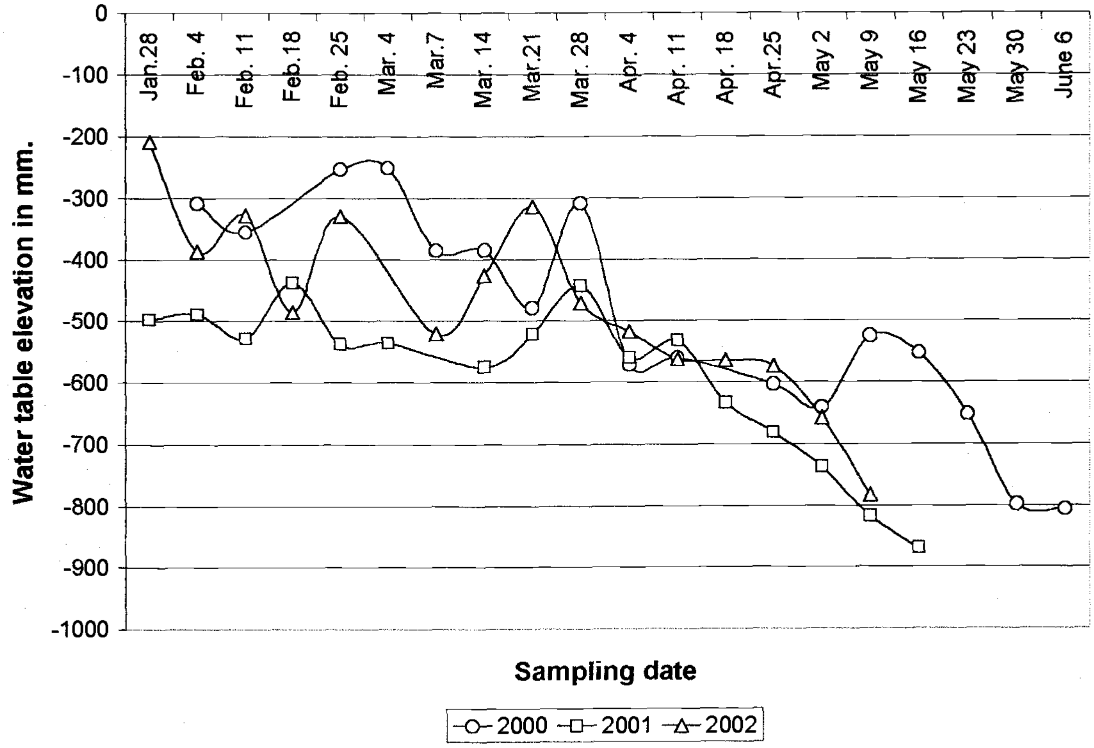



Figure 2.2. Water table elevation in the non-restored mesic meadow (EPA site). Year 2000 is pre-treatment and years 2001 and 2002 are post treatment.

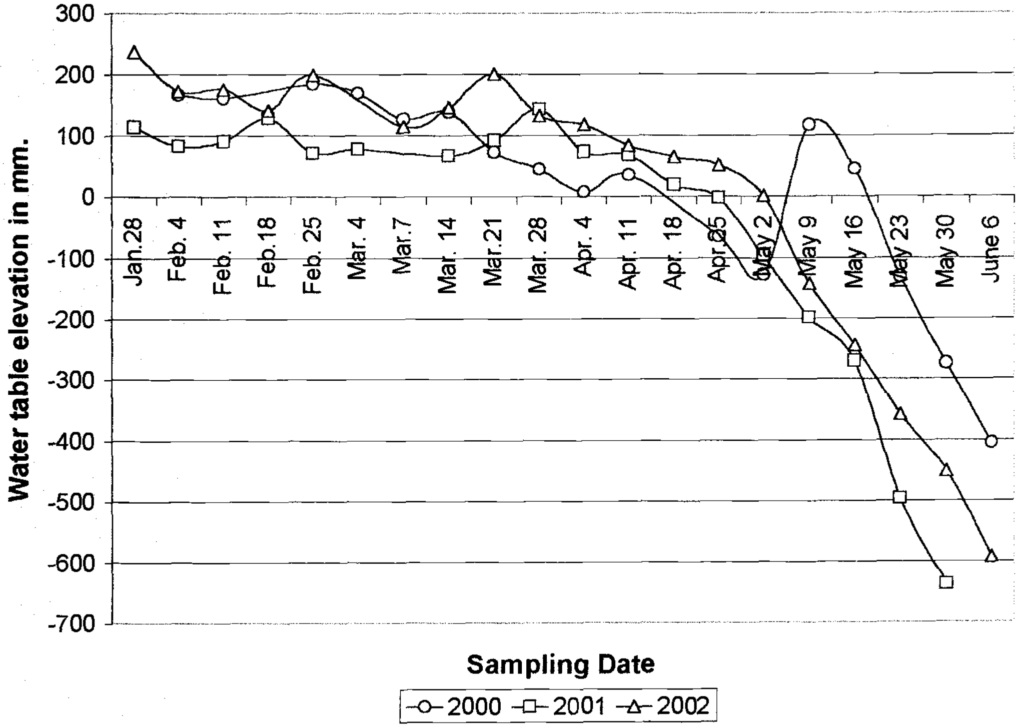



Figure 2.3. Water table elevation in non-restored wet meadow (EPA site). Year 2000 is pre-treatment and years 2001 and 2002 are post treatment.

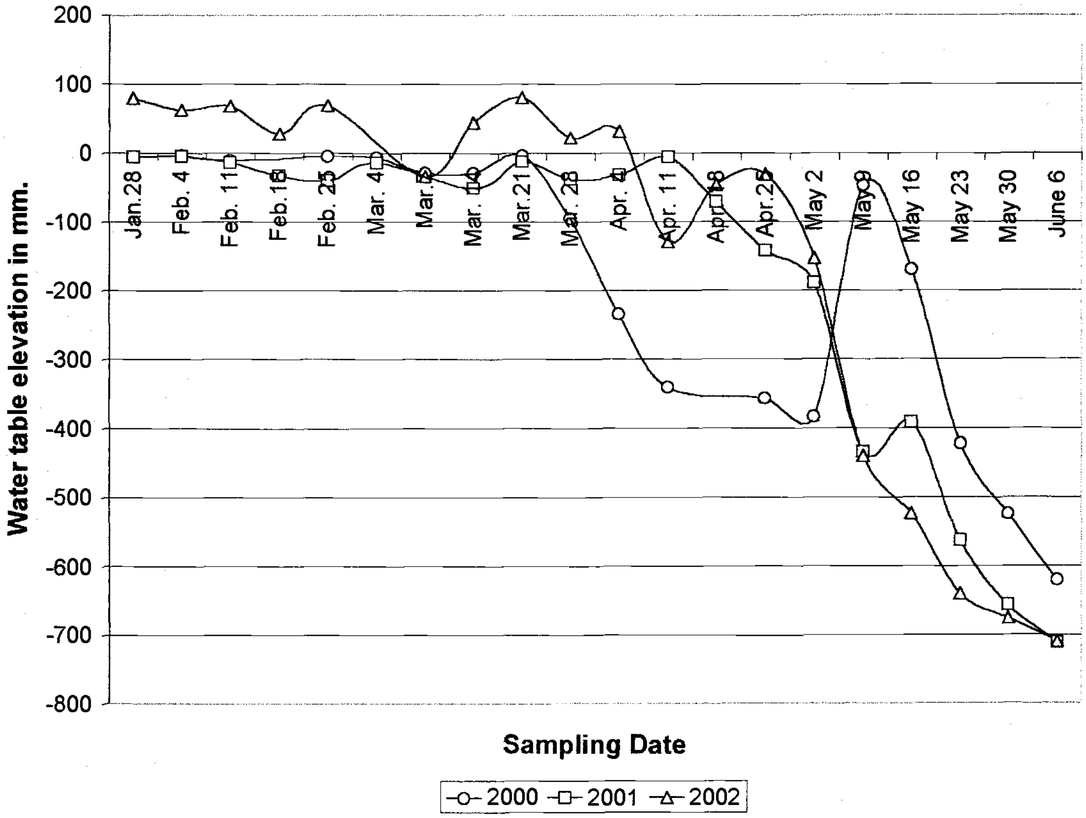



Figure 2.4. Water table elevation in the restored mesic meadow (Covered bridge site). Year 2000 is pre-treatment and years 2001 and 2002 are post treatment.

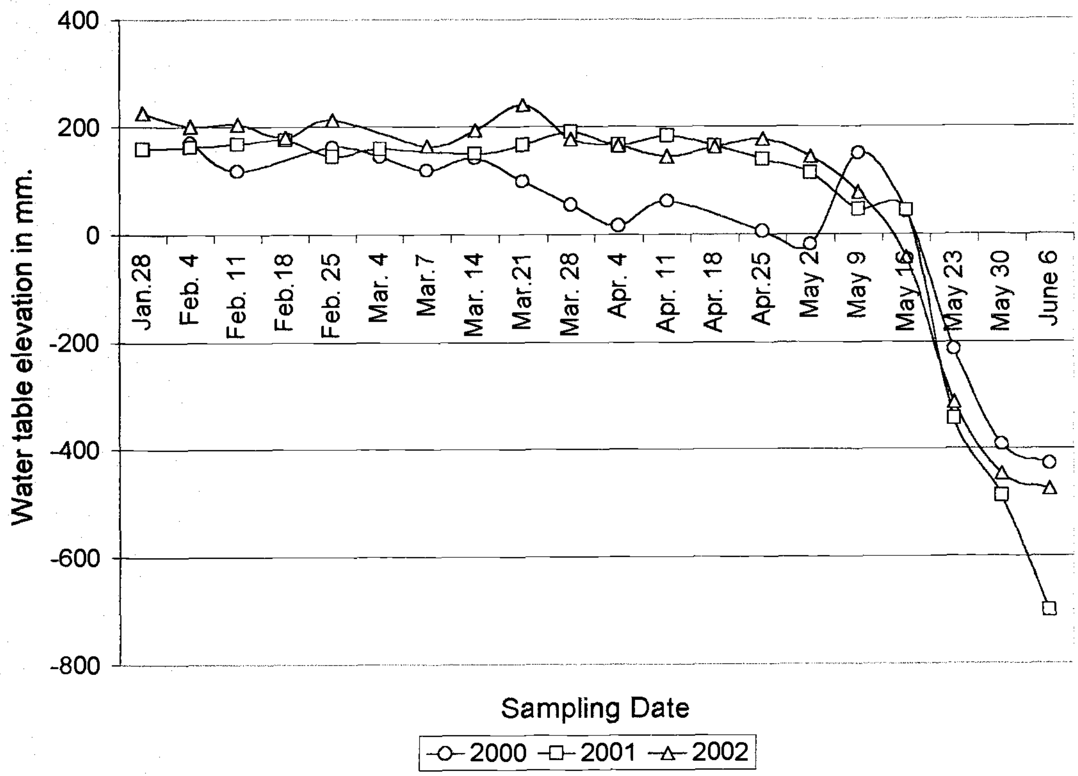



Figure 2.5. Water table elevation in the restored wet meadow (Covered bridge site). Year 2000 is pre-treatment and years 2001 and 2002 are post treatment.

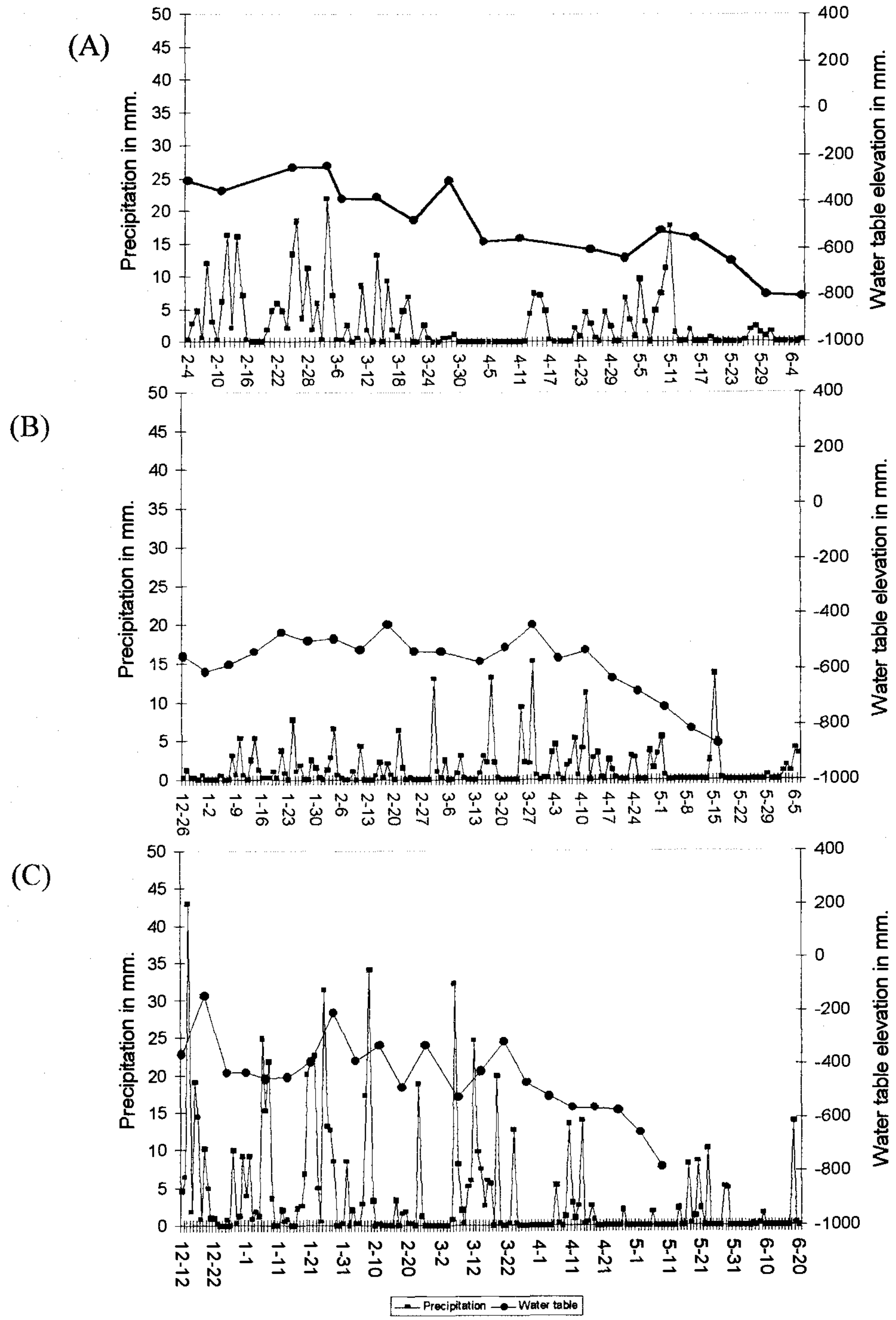



Figure 2.6. Water table elevation and precipitation events in the non-restored mesic meadow (EPA site). Precipitation values are from the Hyslop Experimental Station, Oregon State University. Year 2000 (A) is pre-treatment and years 2001 (B) and 2002 (C) are post treatment.

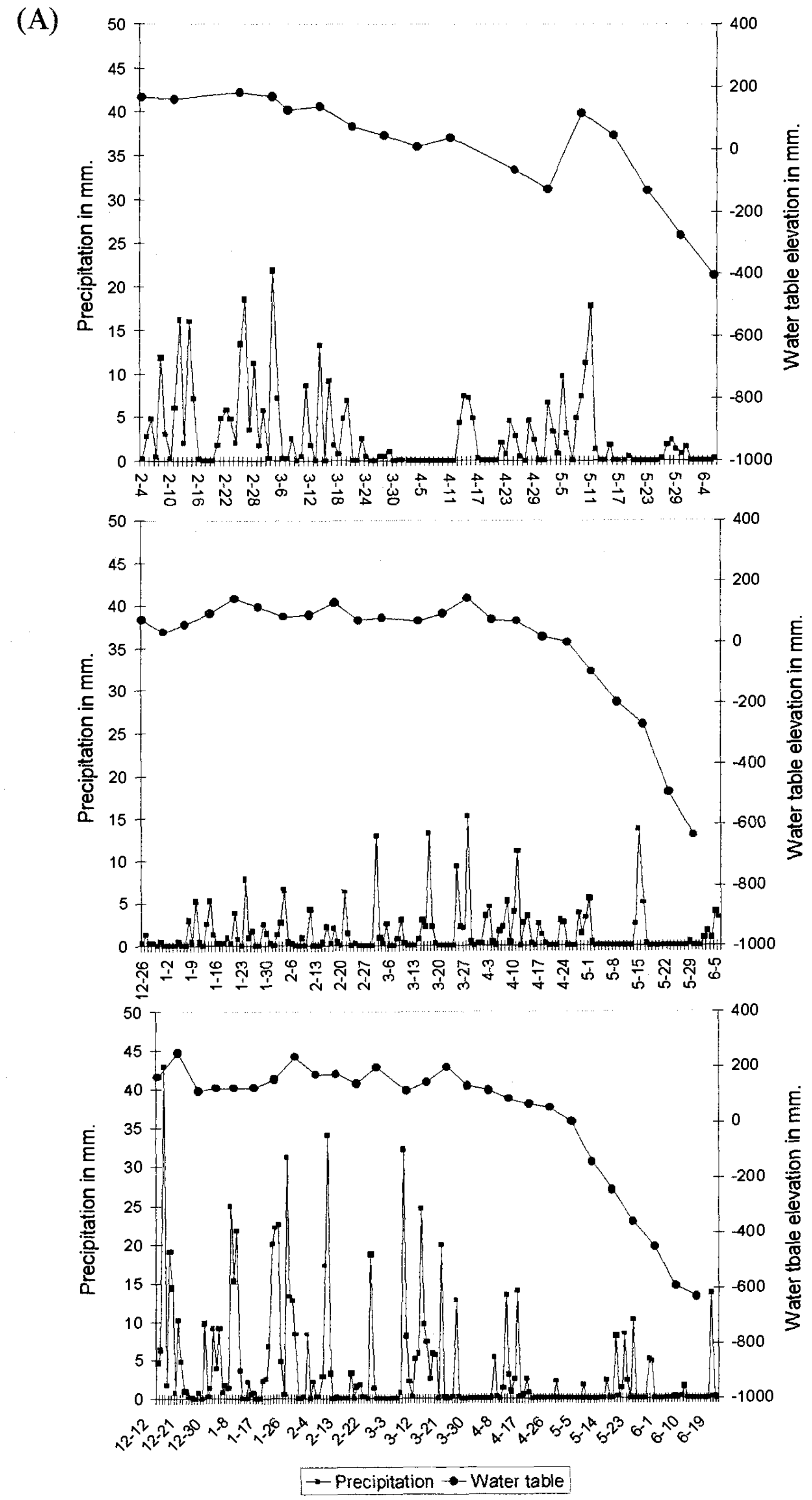



Figure 2.7. Water table elevation and precipitation events in the non-restored wet meadow (EPA site). Precipitation values are from the Hyslop Experimental Station, Oregon State University. Year 2000 (A) is pre-treatment and years 2001 (B) and 2002 (C) are post treatment.

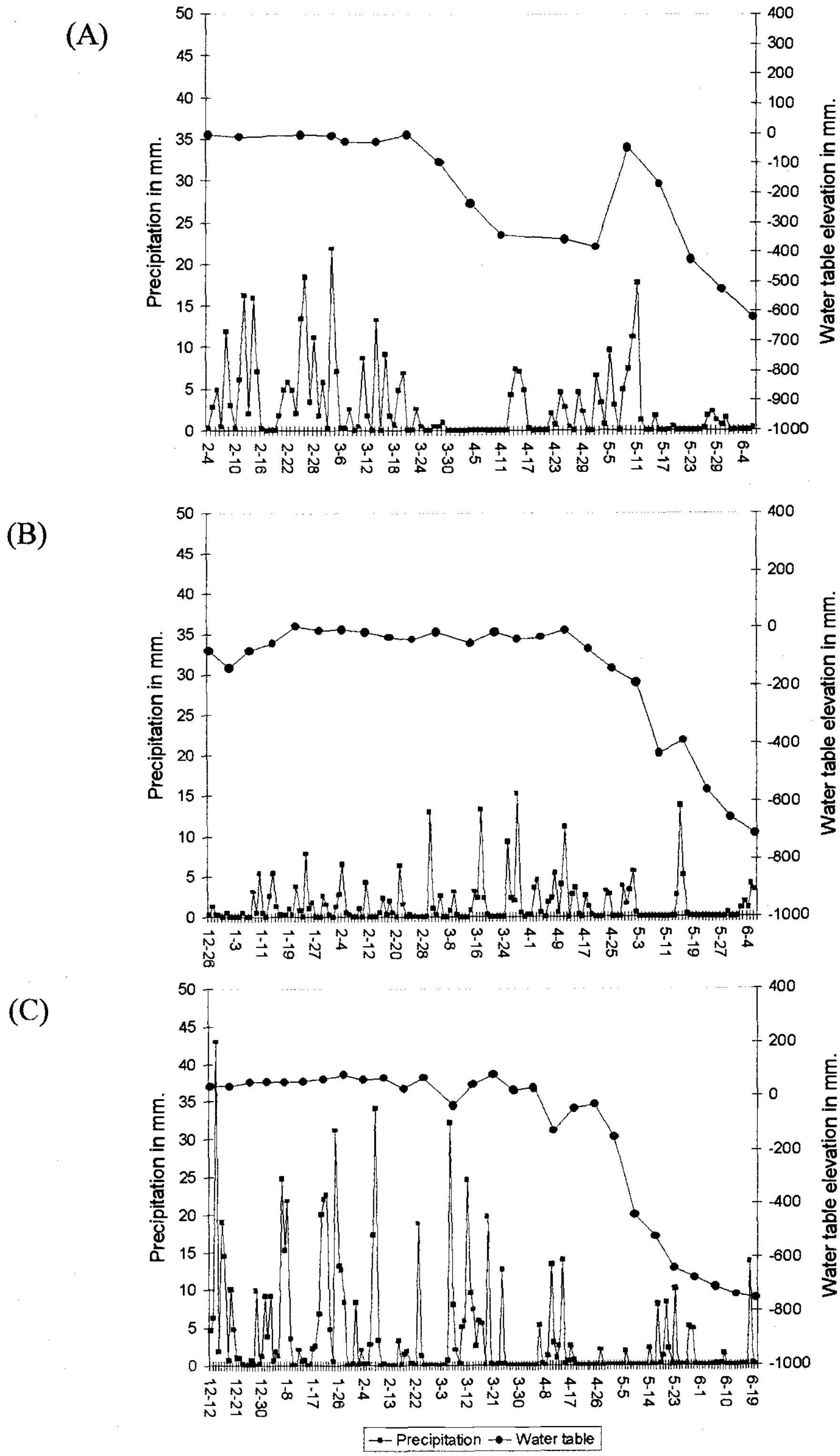



Figure 2.8. Water table elevation and precipitation events in the restored mesic meadow (Covered bridge site). Precipitation values are from the Hyslop Experimental Station, OSU. Year 2000 (A) is pre-treatment and years 2001 (B) and 2002 (C) are post treatment.

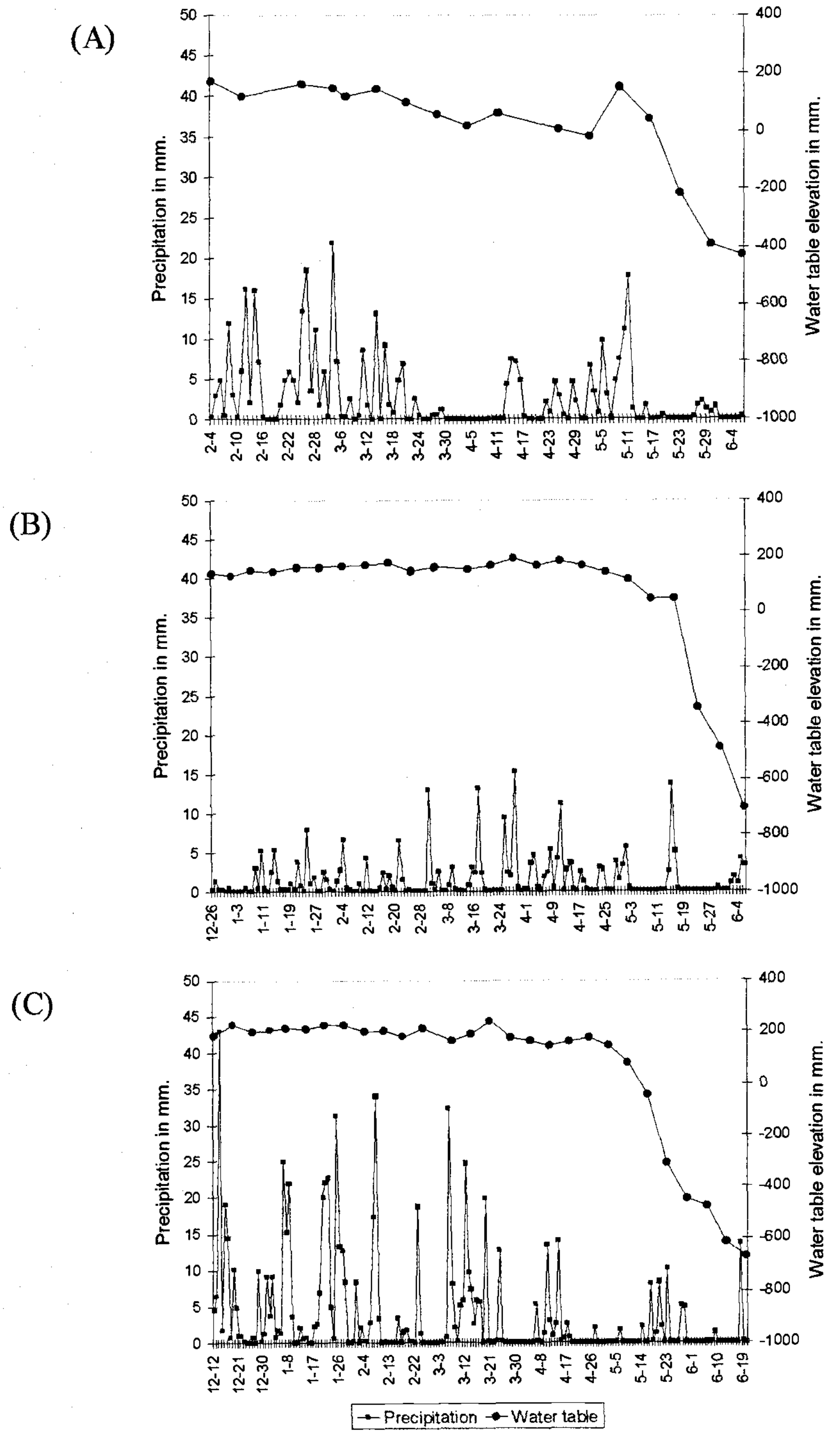



Figure 2.9. Water table elevation and precipitation events in the restored wet meadow (Covered bridge site). Precipitation values are from the Hyslop Experimental Station, OSU. Year 2000 (A) is pre-treatment and years 2001 (B) and 2002 (C) are post treatment.

Because data gathered at the first of the water year is most strongly affected by the timing and intensity of fall rains (i.e., when they arrive) and data gathered at the end of the rainy season is strongly affected by the temperature of that time of year (e.g. high temperatures affect plant growth and alter evaporative processes affecting surface and near surface hydrology), I incorporated only the data from the mid-portion of winter and early spring, the saturated period of the study year. Assuming this would be the period when water table elevation's dominant controlling factor was the ability to sequester water, hydrology data from this period would best represent the storage of, or inversely, the seepage from, that particular system.

Water table elevation in the non-restored mesic meadow displayed a variance found to be unequal between pre-treatment year 2000 and post treatment 2001 ($p=0.0002$), and post treatment years 2001 and 2002 ($p=0.005$). This difference is explained by the differing precipitation values between these study years (101.8 and 58.4, 58.4 and 115.6 cm. respectively). However, when pre-treatment year 2000 is compared with post treatment year 2002, F-tests show evidence that the two variances may be equal ($p=0.13$). This would indicate that under similar precipitation regimes hydrology at this site behaves similarly. This similarity would be expected in a control site (no active restoration) such as this.

F-tests for variation of water table elevation in the restored mesic meadow displayed much different values than that of the non-restored counter-part. F-tests for variance between pre-treatment year 2000 and post treatment year 2001 ($p<0.0001$) and between pre-treatment year 2000 and post treatment year 2002 ($p<0.0001$) showed strong evidence that variances are unequal. Additionally, F-tests performed for the variance between post treatment years 2001 and 2002 showed variances to be most likely equal ($p=0.18$). This would suggest that data collected at this site showed very different variances after restoration was implemented. These differing variances were of a converse nature to that of precipitation values and original hydrologic values suggesting that water retention was enhanced after restoration at this site. Restoration activities appear to have enhanced the ability of this site to sequester water despite lower precipitation values (2001) and inconsistent rainfall patterns between study years.

Water table elevation variances for the non-restored wet meadow (EPA site) displayed similar characteristics to the restored mesic meadow. F-tests for variance between pre-treatment year 2000 and post treatment year 2001 ($p<0.0001$) and between pre-treatment year 2000 and post treatment year 2002 ($p=0.001$) showed strong evidence that variances are unequal. However, variances between post treatment years 2001 and 2002 showed some evidence for similar variances ($p=0.10$). This site showed a similar trajectory to that of the restored mesic meadow suggesting that more water was stored and flashiness was reduced in this non-restored wet meadow. Although this area did not have hydrologic restraints actively restored it was excluded from agricultural management, which included grazing of the Reed canary grass (*Phalaris arundinacea*). Absence of grazing pressure may have induced thicker growth in the grass thus propagating a biotic blockage simulating the effects of a dam and facilitating conditions to slow the rate of outflow.

F-tests for the restored wet meadow (*Alopecurus aequalis*) suggested that hydrologic variances were similar to those of the restored mesic meadow and non-restored wet meadow. Whereas variances between pre-treatment year 2000 and post treatment year 2001 ($p<0.0001$) and between pre-treatment year 2000 and post treatment year 2002 ($p<0.0001$) showed strong evidence that variances are unequal. However, variances between post treatment years 2001 and 2002 showed strong evidence for similar variances ($p=0.15$). This would suggest that hydrologic variance was influenced by the restoration activities performed at this site. Data suggest that this restoration of this wet meadow promoted a enhanced capacity to store water and reduced the variability of hydrologic responses to precipitation events.

This series of statistical evaluations regarding the variability of hydrologic responses to precipitation events provides compelling evidence that in both actively restored study sites some mediation of hydrological fluctuations had occurred. It also suggests that this same mediation has occurred at the non-restored wet meadow. However, when this information is coupled with mean and maximum water table elevations from those sites different conclusions can be inferred. Although variation analysis (F-test) suggests that the non-restored wet meadow demonstrated a heightened ability to store water (reduced fluctuations), relative mean and maximum water table elevation data suggest otherwise.

Both restored study sites exhibited an increase in maximum and mean water table elevation during the first post treatment year despite a decrease in water year precipitation of 43.4 cm. Mean water table elevation during the study year 2000 for the restored mesic meadow was - 218 mm followed by a mean of - 156 mm in post treatment year 2001. This increase in water table elevation was inconsistent with the decrease (43.4cm) in precipitation over that same time period (Figure 2.1.) Mean water table elevation was higher at the restored mesic meadow in post treatment year 2002 than either of the previous study years which is consistent with the increase in water year precipitation throughout the study period (101.8, 58.4, and 115.6 cm, respectively). However, mean water table elevation was lower in the restored wet meadow during post treatment year 2002 than in post treatment year 2001 when water year precipitation had actually increased by 57.1 cm. Presumably, because surface water was present longer and persisted during more of the growing season, filamentous algae growth in the open water wetland was observed to be stimulated. This created a significant build up of filamentous algae on the soil surface (1.3 - 3.2 cm) and may explain the discrepancy from predicted results. These data suggest that restoring hydrological features induce a heightened capacity for adjacent wet and mesic meadows to sequester and store water.

Both mesic and wet meadows where restoration of hydrologic features was not performed exhibited hydrologic conditions that were more sensitive to precipitation events and closely paralleled trends of the precipitation regime. Mean water table elevation was at a moderate level pre-treatment year 2000 followed by a very low elevation post treatment year 2001 and the highest elevation in post treatment year 2002 (Figure 2.10). This presumably reflects water table dynamics of a system that had experienced no significant change in storage capacity over the study years.

Because mean water table elevation change can be skewed by a few exceptionally high water table elevation measurements these data can deceive one into concluding that water table elevation had increased throughout the meadow examined. Perhaps a more revealing way to look at the water table response of the restored meadows is by determining the mean water table change as a function of distance from the actual active restoration. This would reveal a more detailed look at water table dynamics from a smaller scale perspective and would represent the actual change in water table elevation throughout the site without making a broad conclusion that water table elevation had the same increase over the entire distance of the research site.

Figure 2.12 displays the average water table elevation change as a function of distance from active restoration over the entire research site. This information was gathered by measuring the distance of the wells from the edge of the actively restored wetland without regard to the wells being located in either wet or mesic parts of the meadow. Furthermore water table measurements from the actual area where active restoration occurred were not included in this analysis. This figure clearly demonstrates the increase in mean water table elevation displayed in the restored wet and mesic meadow as a function of the distance from the actively restored wetland.

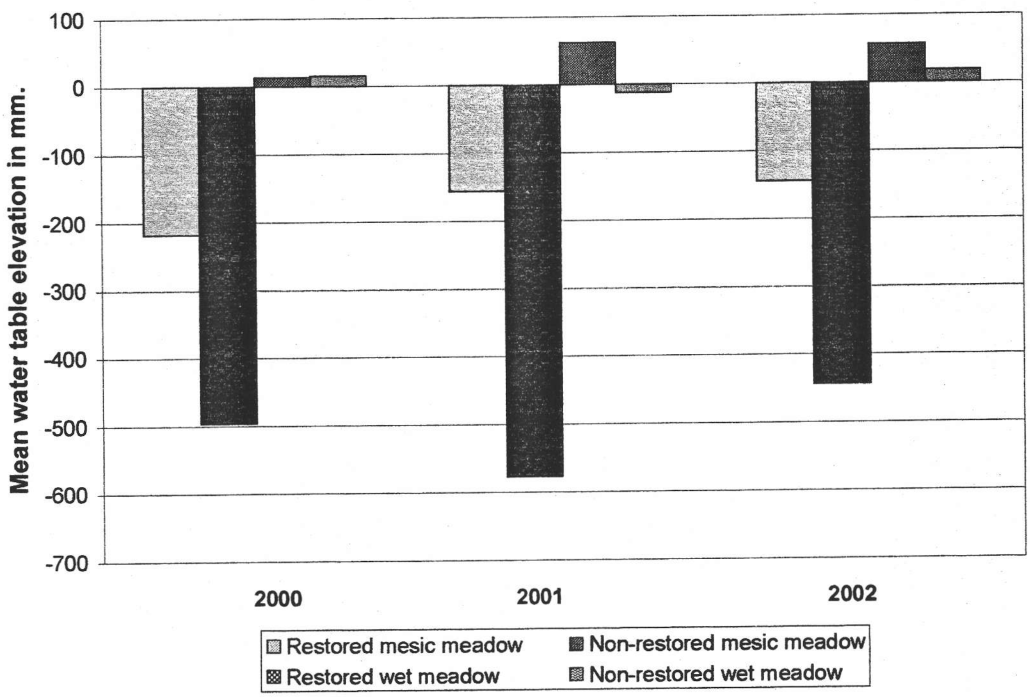



Figure 2.10. Mean water table elevation for each of the study sites. Pre-treatment year is 2000, post treatment years are 2001 and 2002.

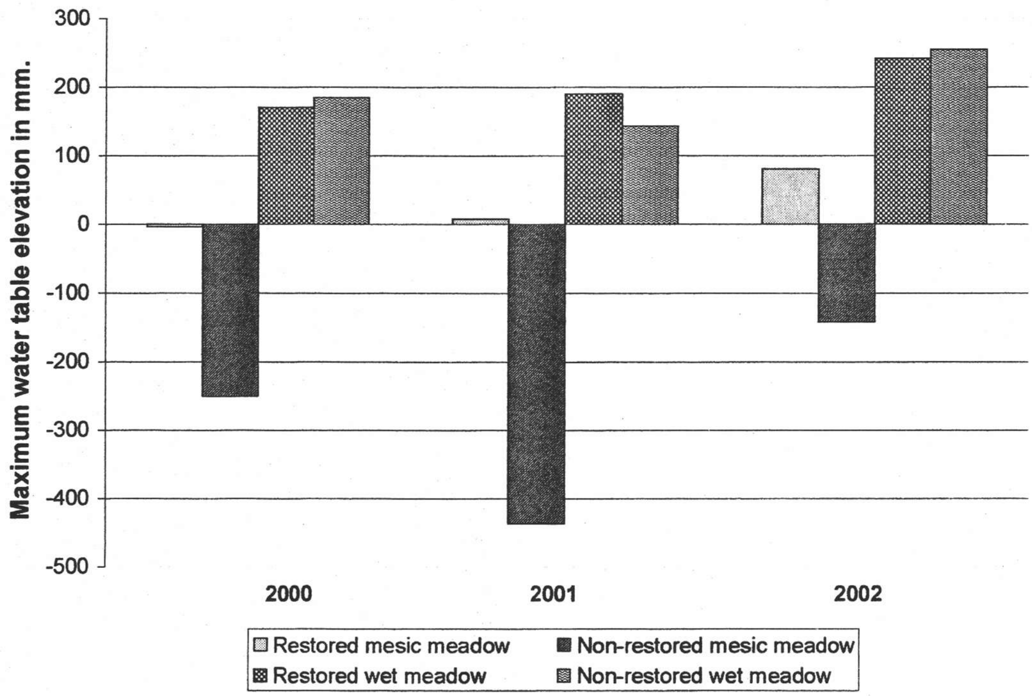



Figure 2.11. Maximum water table elevation for each of the study sites. Pre-treatment year is 2000, post treatment years are 2001 and 2002.

Between pre-treatment year and post treatment year 2001 mean water table elevation change interpolated by a linear regression was found to be almost 80 mm directly adjacent to the actively restored area and increased elevation was still recognizable at approximately 91 m from this area. This is a dramatic increase because water year precipitation during this same time period was much lower in the post treatment year as compared to the pre-treatment year (58.4 and 108.2 cm, respectively). This type of restoration appears to not only increase the water storage ability of the immediate area restored but extends this capacity to adjacent areas as well.

Between post treatment year 2001 and post treatment year 2002 mean water table elevation directly adjacent to the restored site was determined to have increased by approximately 20 mm and continued to increase for the entire distance of the meadow. At the furthest wells from the actively restored site (102.4 m) a mean increase of more than 50 mm was interpolated using the linear regression model (Figure 3.12). This would suggest that on years with average precipitation values water table elevation increases may occur many tens of meters from actively restored wetland areas. This also demonstrates the wetlands increased capacity to capture and store water when actively restored.

Between pre-treatment year 2000 and post treatment year 2002, water table elevation increased over 100 mm directly adjacent to the restored area and increased approximately 43 mm at the furthest edge of the meadow (102.4 m). These data show compelling evidence that when hydrologic features are repaired or restored, water table elevation will be increased at the point of restoration as well as areas at some distance from the actual site of restoration. In this study the farthest distance from the actual restoration was 102.4 m and the water table elevation demonstrated increases up to that point and likely beyond. One may conclude that care must be taken not to induce changes in water table elevation in areas adjacent to restoration work. In some cases increased water table elevation may be detrimental to existing plant communities.

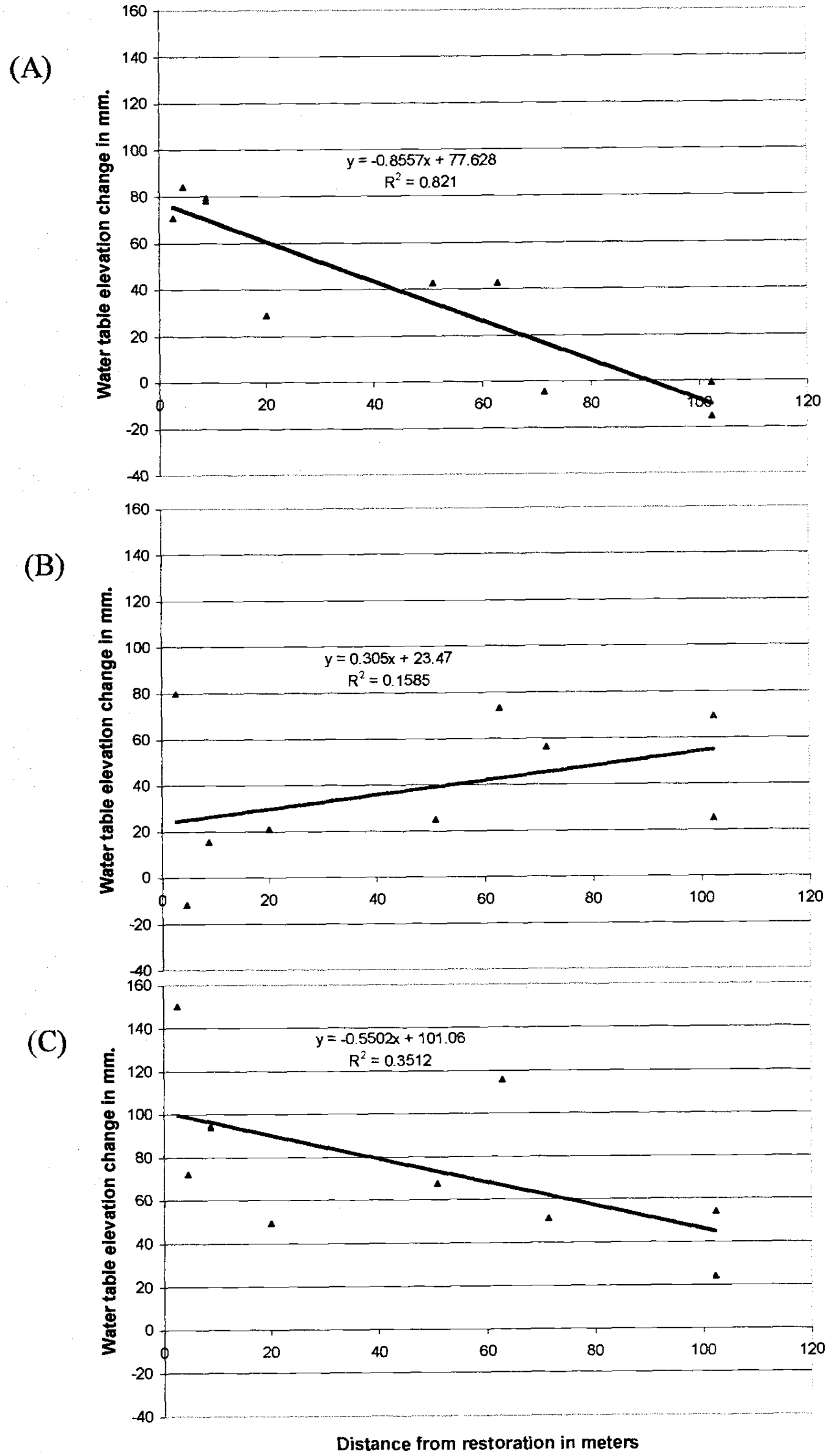



Figure 2.12. Mean water table elevation change as a function of distance from actively restored wetland. Elevation change is between years (A) 2000 - 2001, (B) 2001 - 2002, and (C) 2000 - 2002.

## Discussion and Conclusions

Water Table Elevation Change Associated with Wetland Restoration. Restoration of damaged hydrological features (filling drainage ditches) had profound influences on water table elevation at the site of restoration as well as areas adjacent to the restoration. Increased water table elevation and moderation of storage variability typified these influences.

In this experiment, restored sites showed an increase in maximum and mean water table elevation between pre-treatment year 2000 and post treatment year 2001. Maximum water table elevation showed a 75% increase in the restored mesic meadow and a 27% increase in the restored wet meadow (*Alopecurus*) during this time period (Table 2.11). Mean water table elevation showed a similar trajectory with 28% and 80% increases for the restored mesic and wet meadows, respectively (Table 2.10). Furthermore, the increases in water table elevation could be detected up to 91.4 m from the area that was actively restored impacting hydrology for the entire length of the meadow. Importantly, these increases were seen during a year when only 54% of the previous year's precipitation had occurred. One can hypothesize that this increase in mean and maximum water table elevation could affect stream flow dynamics of adjacent streams especially during very dry years such as the year 2001. It also demonstrates the potentially profound implications that building drainage ditches could have on farmlands. Ditching presumably affects water table dynamics across great distances in broad floodplains. However, this also explicates the need for careful planning when implementing wetland restoration. Projects such as this have the potential to change water table dynamics at great distances from their intended location.

Water table elevation variability also demonstrated significant changes after actively restoring hydrological features. Restored meadows showed a propensity to moderate fluctuations in water table elevation when compared to the non-restored mesic meadow. Restored areas tended to demonstrate a non-fluctuating water table elevation both during and between precipitation events for the post treatment years. This suggests that when drainage ditches are filled (plugged), water table dynamics become more static in nature. This consistent inundation could profoundly affect vegetation communities by allowing hydrophytic plants to establish and causing xeric plants to be eliminated. An additional point of interest is that the non-restored wet meadow (*Phalaris*) showed less variability for both post treatment years. This may be due to biomass accumulation in the Reed canary grass community as grazing pressure was removed the accumulated vegetation appeared to slow the flow of water from the site. Although it appears that Reed canary grass decreased the water table variability over the study period water table elevation decreased. This suggests that lack of water table variability can be short lived and have little consequence to the water table elevation. However assuming that this lack of water table elevation variability allows soils to become more anoxic (because of less frequent intervals of soil/air contact) many implications could be construed. Potentially Reed canary grass in wet meadows could slow velocities of exiting water to an extent that conditions were not suitable for xerically adapted plant species and therefore destroy entire plant assemblages. Increased water table elevation has been shown to have detrimental effects on adult trees through reduced growth, decreased root mass, and increased mortality (Denslow and Battaglia 2002).

This study showed the significant role that restored flood plain meadows can play in capturing and storing water. Moreover, it demonstrates the dramatic effect that a simple drainage ditch can have on surface and sub-surface water table elevation. These effects are most dramatic in the areas closest to the ditch but are also evident at significant distances. Evidence such as this suggests that manipulations of hydrologic features and establishment of exotic species may influence hydrologic characteristics for great distances and potentially cause unplanned results.

# 3. PLANT SPECIES SUCCESSION IN TWO RESTORED RIPARIAN WETLAND MEADOWS IN THE WILLAMETTE VALLEY, OREGON.


## Abstract

I examined plant community changes associated with the implementation of ecological restoration in two riparian meadows located in an agricultural landscape of the central Willamette Valley, Oregon. I established exclosure fencing (a form of passive restoration) in one agricultural field (EPA site) and established fencing and actively restored damaged hydrological features (filled ditches) in a separate agricultural field (covered bridge). I monitored plant community dynamics (species response) in both restored sites as well as the adjacent agriculturally managed (untreated) sites one year before treatment and two post-treatment years. I sampled two plant community types: wetland meadow and mesic meadow. From composition data, I calculated species richness and the relative abundance of wetland indicator species, nuisance weeds, and native plants. During this study, agriculturally managed mesic meadows displayed increased nuisance weeds but exhibited a decrease in native plant abundance. Wetland plant species abundance tended to increase in agricultural sites with light grazing and decreased in areas that were plowed and re-seeded. Actively restored mesic meadows exhibited an increase in native plants and a decrease in nuisance weeds. The passively restored mesic meadow exhibited an increase in native plant abundance and decreases in all other categories. The actively restored wetland displayed increases in plant species richness and nuisance weed abundance and a decrease in native plant abundance. Agriculturally excluded wetlands dominated by Reed canary grass exhibited no changes for the entire study period. Results suggest that agricultural exclusion does not increase nuisance weed species but active restoration may result in increases (due to disturbance). Additionally, results indicate that in agricultural landscapes dominated by introduced grasses, short-term plant community changes are minimal unless initiated by intense land management practices (e g., plowing, re-seeding, or removal of vegetative material).

## Introduction

Riparian wetlands represent a transitional zone between terrestrial and aquatic ecosystems. As the transitional zone between these ecosystems, riparian wetlands can have major effects on the quality of water in downstream systems (Loucks 1989, Gilliam 1994). Riparian wetlands have been shown to have a high capacity to filter and/or reduce nitrogen in surface and subsurface flows from agricultural areas (Groffman 1992, Hanson 1994, Hubbard 1995, Jacobs 1985). Nitrate and other forms of N are reduced, sequestered, or transformed in riparian wetlands through several means. Riparian wetlands have a large capacity to denitrify incoming waters through denitrification (conversion of $NO_3^-$-N into gaseous states by facultative anaerobic microorganisms) and by microbial immobilization (Groffman 1992). Vegetative uptake of N is also a major factor contributing to N losses in the riparian wetlands (Lowrance 1984, Groffman 1992). Riparian wetlands also serve as "hydraulic braking points" that is, they reduce or slow velocities of incoming water sources both above and below ground promoting suspended particulate to fall out of transport. Gilliam (1994) stated that the riparian wetland is the most important factor influencing nonpoint-source pollutants entering surface water in many areas in the USA and the most important wetlands for surface water quality protection.

Riparian wetlands have also been shown to be important wildlife and plant habitats (Guynup 1999). Kauffman et al. (2000) reported that 70% of all wildlife species utilized riparian wetlands in Oregon and Washington. Riparian wetlands provide a haven for a wide variety of flora and fauna and offer a unique habitat for many rare and endangered species (Mitsch and Cronk, 1992). Kirkland (1999) found a significant correlation ($r^2$=0.80) between the number of federally listed mammals (Endangered Species List) in the United States and the loss of wetlands in those areas. Mitsch et al. (1998) reported that after wetland restoration, plant diversity increased from 13 plant species to 65 total species within 3 years, representing a net increase of 500%. This same wetland study showed an increase of 17 bird species due to the creation of wetlands. Macro-invertebrates have also been shown to increase significantly with the restoration of wetland habitats (Mitsch et al 1998). Wildlife enhancement is typically a benefit of wetland restoration projects and is usually coupled with other specific goals. Mitsch (1992) described typical goals obtained from wetland restoration as: flood control, waste water treatment, storm water or non-point pollution control, ambient water quality improvement, wildlife enhancement, fisheries enhancement, replacement of similar habitat, or research.

Many riparian wetlands in the Pacific Northwest have become impaired by land use activities (Baker et al. 1995). In Oregon over 38% of all wetlands have been destroyed or otherwise eliminated (Dahl 1990). The agricultural riparian wetlands of Oak Creek, which flows through the campus of Oregon State University (OSU), are no exception. OSU lands have been managed for agriculture and livestock production since the late 1860s and the dairy and beef confined animal feeding operations have been in operation since the 1930s. Riparian wetlands have been drained and riparian areas have been converted to croplands or pastures dominated by exotic species. Along this reach, Oak Creek is incised and linkages between high flows and floodplains have been altered. Nevertheless, a number of relict channels and degraded wetlands exist with a high potential for restoration.

Many agricultural managers can and have employed methods to restore and/or enhance these riparian wetlands. Many use management regimes of passive restoration either for restoration purposes or to incorporate these riparian areas as "vegetation buffer strips" (Hubbard 1995, Jacobs 1985, Young 1980). Buffer strips are known to increase the quality of waters exiting agricultural areas and entering waterways (Young 1980). Many studies have documented the ability of small-vegetated buffer strips to increase surface and sub-surface water quality (Groffman 1992, Hanson 1994, Hubbard 1995, Jacobs 1985). Other studies have focused on biotic changes associated with restoration (Sprenger et al. 2002, Stevens et al. 2002, Mushet et al. 2002). However few studies exist that quantify the herbaceous plant community changes associated with restoration management of this type. I examined plant community changes associated with the implementation of restoration in two agricultural riparian meadows located in the central Willamette Valley, Oregon. I established exclosure fencing (a form of passive restoration) in one agricultural field and established fencing and actively restored damaged hydrological features (filled ditches) in a separate agricultural field (Figure 2.0). I then monitored plant community dynamics in both restored fields as well as the adjacent agriculturally managed fields (control) pre-treatment and two post-treatment years. The objectives were:

(1) Determine how these restoration techniques affect plant species composition particularly the native flora.

(2) Determine if these restoration techniques reduce the occurrence of noxious plants.

(3) Compare the efficacy of these two restoration management regimes.

## Methods

### Study sites

The study sites were two riparian wetland meadows associated with an unconstrained reach of Oak Creek. Oak Creek flows from the Coast Range foothills into the central Willamette Valley at Corvallis, Oregon. Oak Creek is a third order stream of approximately 13.4 kilometers in length. Oak Creek is a tributary to the Mary's river that is within the Willamette River Basin. The total drainage of Oak Creek is 33.2 square kilometers. Average annual precipitation for this area is 107.4 cm (Yamaguchi 1993). The highest point in this watershed is about 548.6 m with the steepest slopes concentrated in the forested headwater regions. Oak Creek flows from forested headwaters through an extensive agriculture area then flows through an urban portion of Corvallis, Oregon. These research sites are located within the "agricultural reach" on the Oregon State University campus between 35$^{th}$ and 53$^{rd}$ streets, Corvallis, Oregon. Site 1 (covered bridge) is approximately 2 ha in area and site 2 (EPA) is approximately 3 ha. Site 1 was located just upstream of the Covered Bridge on Campus way on the south side of Oak Creek. Two plant communities dominate this site: (1) mesic prairies dominated by rye grass (*Lolium perenne*) and meadow foxtail (*Alopecurus pratensis*) as well as many exotic herbs; and (2) seasonally flooded wetlands dominated by little foxtail (*Alopecurus aequalis*). Before restoration the covered bridge site had a seasonally inundated open water wetland with approximately 775 $m^2$ of surface area. Restoration activities increased the surface area of this open water wetland to approximately 1332 $m^2$. The EPA site was a relict channel and floodplain approximately 400 m. down stream and on the north side of Oak Creek. Both sites are relatively flat and have roughly equal amounts of sunlight. Mesic meadows within the EPA site had plant communities similar to those of the covered bridge site but the flooded wetland plant community was completely different and dominated by reed canarygrass (*Phalaris arundinaceae*). Oregon white oak (*Quercus garryanna*) approximately 120 years old was dispersed on dry elevated microsites throughout the EPA site. A mix of riparian obligate trees Oregon ash (*Fraxinus latifolia*), Black cottonwood (*Populus balsamifera)*, Pacific willow (*Salix lasiandra.*), and Red alder (*Alnus rubra*) are along the creek and ecotonal to the covered bridge site. Adjacent to the treated sites are sites managed for grass/livestock production and are frequently plowed, fertilized, grazed, and hayed. Dairy, beef, and swine production facilities are located adjacent to these sites.

Based upon observations of relic sites with similar hydrology and geomorphology we hypothesize that the native dominance (site potential) of the mesic prairies would be that of tufted hairgrass (*Deschampsia cespitosa*), rushes (*Juncus spp.*), and a rich suite of native forbs. Flooded wetlands would be dominated by a mix of sedges (*Carex spp*), rushes ( *Juncus spp.*), and bulrushes (*Scirpus spp.*).

Soils in all sites are Waldo-Bashaw silty clay loams <2% slopes and are included in the local hydric soil list (benton county soil survey). The surficial geologic units of this area is quaternary lower terrace deposits of semiconsolidated cobbles, gravel, sand, silt, clay, and organic material approximately 11m thick above recent river alluvium (Buckley 1994).

## Restoration approaches

We wanted to examine ecosystem responses to restoration approaches that would be relatively inexpensive and easily be implemented by landowners and managers. Our approaches to restoration follow those being implemented on many private lands with technical assistance provided by the Oregon Department of Fish and Wildlife (Steve Smith, ODFW, personal communication), and the Natural Resource Conservation Service.

In 1999 we fenced the agricultural riparian meadows and halted all grazing, plowing, seeding, and chemical applications on the sites. In 2000, active restoration of site 1 was performed to repair hydrological damage caused by a drainage ditch and fill. We removed soil material from the east-end of site 1 and placed soil into the head-cut (erosion) and ditch at the north end of the site effectively removing the drainage ditch. We removed the soil in an area that facilitated the accumulation of surface water and also directed out flow to be discharged into the existing ditching system located on the outer perimeter of the site (Figure 2.2).

The active restoration activities at the covered bridge site were:

(1) The excavation of soils adjacent to the existing open water wetland.

(2) The filling of a drainage ditch with soil excavated from the wetland.

This resulted in increased storage capacity and open water surface area combined with a decrease of outflow from the open water wetland.

## Plant species composition

Plant community composition was determined each summer for one-year prior to treatment and two years after treatment. Permanent transects 15 m in length were established within the two plant communities at each site. Each wetland community had four transects randomly situated within their borders and I established six transects within the mesic meadows which were larger in area. In addition to the six transects that were established in each restored mesic meadow, six transects were placed outside of the treated-restored areas to serve as controls (for the mesic meadows). Vegetation composition was determined from the establishment of 15 nested microplots at one meter intervals along each transect. The nested microplots consisted of three plots - 50X50, 25X50, and 25X25 cm in area.

Relative abundance, frequency, and species diversity was measured at each microplot along each transect. Frequency scores were given to each plant species according to juxtaposition within the microplot. Scores of four were given to species located within the 25X25-cm microplot, two for within the 25X50-cm plot and from the larger 50X 50-cm plot a score of one was recorded. This scoring was based upon the area of the sampled microplot because a species is more likely to occur in a larger plot than in a smaller one. This method was developed in order to ascertain relative abundance of existing common species while preserving the ability to detect new or invading plant species despite of initial low frequency. Nomenclature followed Hitchcock and Cronquist (1973) and native or introduced status followed Hitchcock and Cronquist (1973), Pojar and Mackinnon (1994), or Habeck (1961) where applicable. Wetland indicator status followed the National List of Vascular Plant Species that Occur in Wetlands, region 9 (USFW 1996).

## Data analysis

Relative abundance of each species was calculated for each transect. Frequency scores for each plant species were divided by the sum of all scores in order to obtain a relative abundance. These averages were summed across transects for each research metric and mean and standard error were calculated.

To test for plant community differences through time the mean relative abundance for each plant species was combined into respective metric categories and evaluated using a paired two-sample t-test for means (Ramsey and Schafer 1997). Year of data collection was evaluated as the independent variable and plant community composition was the dependent repeated variable.

Plant community composition measures were summarized using species richness and the relative abundance of the following metrics: wetland indicator status, native plants, and noxious weeds.

Species richness was calculated using the total number of plant species identified within each microplot and averaging those values over the transect length. This method allows for the allocation of plant species with minimal abundance to be incorporated into the data.

Wetland indicator status of plant species was determined by utilizing the PNW region of the National List of Vascular Plant Species that Occur in Wetlands (USFWS 1996). This list categorizes specific plant species according to their association to wetlands. Categories were; obligate wetland (OBL) occur almost always (99%) in wetlands, facultative wetland (FACW) usually occurs (67% - 99%) in wetlands, facultative (FAC) likely to occur (34% - 66%) in wetlands, facultative upland (FACU) usually occurs in non-wetland but occasionally occurs (1% - 33%) in wetlands, and obligate upland (UPL) occurs almost exclusively in non-wetlands (99%). All of these categories are specific to certain regions of the United States. Occasionally a plant species is not found in any wetland habitat in the U.S. and that plant species is not included in the National List and is denoted by NI. Wetland indicator (hydrophytic) plants are those listed in the obligate wetland (OBL), facultative wetland (FACW), and facultative (FAC) categories. Wetland indicator plant status was determined for each plant species. Wetland plant abundance was tabulated then divided by the sum of all plant species for each transect to determine relative abundance.

Native plant abundance was calculated as the relative abundance of plants determined to be of native origin. Native plant species abundance was calculated for each plant species and then divided by the sum of all plant species for each transect.

Nuisance weed abundance was defined as the relative abundance of plant species determined to decrease productivity of common agricultural crops or harbor common agricultural diseases (Royer and Dicinson 1999). These are also considered to have invasive propensities. Additionally Reed canary grass (*Phalaris arundinacea*) is listed as a nuisance weed because of it's known invasive and domineering properties (Morrison and Molofsky 1998; Green and Galatowitsch 2001). Nuisance weed characterization was determined for each plant species. Those plant species deemed nuisance were summed and then divided by the sum of all plant species for each transect to determine relative abundance.

Table 3.0. Plants identified in all research sites. Wetland indicator status is given for each plant species when available (OBL= obligate wetland; FACW= facultative wetland; FAC= facultative; FACU= facultative upland; UPL= upland; NI= not included in list (National List of Vascular Plant Species that Occur in Wetlands (USFW 1996). Native origin or introduced origin is based on information in Flora of the Pacific Northwest[1] (Hitchcock and Cronquist, 1973), Plants of the Pacific Northwest Coast[2] (Pojar and Mackinnon,1994) and (Habeck[3] 1961) (superscript denotes which source was used). Nuisance category consists of those plants deemed to decrease productivity or harbor common agricultural disease in "Weeds of the Northern U.S. and Canada" (Royer and Dickinson, 1999). Form is G= grass; S= sedge; R= rush; H= herb; T= tree

| Plant species | Wetland status | Native or Introduced | Nuisance Y/N | Form |
|---|---|---|---|---|
| *Agrostis tenuis* Sibth. | FAC | I[2] | N | G |
| *Alopecurus aequalis* Sobel. | OBL | N[3] | N | G |
| *Alopercurus pratensis* L. | FACW | I[2] | N | G |
| *Bromus vulgaris* (Hook) Shear | UPL | N[3] | N | G |
| *Dactylis glomerata* L. | FACU | I[1] | N | G |
| *Echinochloa crusgalli* (L.) Beauv. | FACW | I[2] | Y | G |
| *Festuca arundinacea* Schreb. | FAC- | I[1] | N | G |
| *Festuca occidentalis* Hook | NI | N[3] | N | G |
| *Glyceria occidentalis*(Piper)Nels. | OBL | N[3] | N | G |
| *Hordeum geniculatum* All. | NI | N[2] | N | G |
| *Juncus bufonis* L. | FACW | N[2] | N | J |
| *Lolium multiflorum* Lam. | FACU | I[1] | N | G |
| *Lolium perene* L. | FACU | I[1] | N | G |
| *Phalaris arundinacea* L. | FACW | I[1] | Y | G |
| *Poa pratensis* L. | FAC | I[2] | N | G |

Table 3.0 (continued)

| Plant species | Wetland status | Native or Introduced | Nuisance Y/N | Form |
|---|---|---|---|---|
| *Allium amplectens* Torr. | NI | N[2] | N | H |
| *Amaranthus retroflexus* L. | FACU | I[1] | Y | H |
| *Antennaria neglecta* Greene | NI | ? | N | H |
| *Cerastium arvense* L. | FACU | N[1] | N | H |
| *Cerastium viscosum* L. | NI | I[1] | N | H |
| *Chenopodium album* L. | FAC | I[2] | Y | H |
| *Cirsium arvense* (L.) Scop. | FAC | I[1] | Y | H |
| *Cirsium brevistylum* Cronq. | NI | ? | N | H |
| *Convolvulus arvensis* L. | NI | I[1] | Y | H |
| *Crepis capillaris* L. Wallr. | FACU | I[2] | Y | H |
| *Daucus carota* L. | NI | I[2] | Y | H |
| *Dipsacus sylvestris* Huds. | NI | I[1] | Y | H |
| *Dowingia yina* Appleg. | OBL | N[1] | N | H |
| *Epilobian ciliatum* L. | FACW- | ? | N | H |
| *Galium aparine* L. | FACU | N[2] | N | H |
| *Galium triflorum* Michx. | FACU | N[2] | N | H |
| *Geranium dissectum* L. | NI | I[1] | N | H |
| *Geranium robertium* L. | NI | I[1] | N | H |
| *Gnapthalium palustre* Nutt. | FAC+ | N[1] | N | H |
| *Gratiola ebracteata* Benth. | OBL | ? | N | H |
| *Hypochaerus radicata* L. | FACU | I[1] | Y | H |
| *Lactuca muralis* (L.) Fresen. | NI | I[1] | Y | H |
| *Lactuca serriola* L. | FACU | I[1] | Y | H |

Table 3.0 (continued)

| Plant species | Wetland status | Native or Introduced | Nuisance Y/N | Form |
|---|---|---|---|---|
| *Leontodon nudicalis* L. Merat. | NI | I[1] | Y | H |
| *Medicago lupulina* L. | FAC | I[1] | Y | H |
| *Montia linearis* (Dougl.) Greene | NI | ? | N | H |
| *Parentucellia viscosa* (L.) Car. | FAC- | I[1] | N | H |
| *Plagiobothrys figuratus* (Piper) Johnston | FACW | ? | N | H |
| *Plantago major* L. | FACU+ | I[2] | Y | H |
| *Polygonum aviculare* L. | FACW- | I[1] | Y | H |
| *Polygonum punctatum* Ell. | OBL | ? | N | H |
| *Ranunculus aquatilis* L. | OBL | I[2] | N | H |
| *Ranunculus muricatus* L. | FACW | I[1] | N | H |
| *Rorrippa islandica* (Oed) Borbas | OBL | N[1] | N | H |
| *Rumex crispus* L. | FAC+ | I[2] | Y | H |
| *Sonchus asper* (L.) Hill | FACW- | I[2] | Y | H |
| *Stellaria longipes* Goldie | FACW- | I[1] | N | H |
| *Taraxacum officinale* Weber | FACU | I[1] | Y | H |
| *Tragopogon dubois* Scop. | NI | I[1] | Y | H |
| *Trifolium pratense* L. | FACU | I[1] | Y | H |
| *Trifolium repens* L. | FAC | I[1] | Y | H |
| *Veronica arvensis* L. | FACU | I[1] | N | H |
| *Vicia sativa* L. | UPL | I[1] | N | H |
| *Crataegus monogyna* Jacq. | FACU | I[1] | N | T |
| *Populus trichocarpa* T. & G. | FAC | N[1] | N | T |
| *Quercus garryanna* Dougl. | NI | N[1] | N | T |

Table 3.1. Relative abundance (mean % and standard error) of plant species (EPA site) in 180, 50X50 cm. plots along permanent transects. Measurements were taken over a 3-year period following field exclusion from agricultural management. Standard errors were calculated using transects as the unit of measure (n = 6).

| Plant species | 2000 Restored | | 2000 Ag. Managed | | 2001 Restored | | 2001 Ag. Managed | | 2002 Restored | | 2002 Ag. Managed | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Mean | SE | Mean | SE | Mean | SE | Mean | SE | Mean | SE | Mean | SE |
| *Agrostis tenuis* | | | | | **30.7** | **0.9** | 3.3 | 1.3 | **29.7** | **1.7** | 0.3 | 0.4 |
| *Alopecurus aequalis* | | | | | | | 0.3 | 0.3 | | | 0.8 | 0.1 |
| *Alopecurus pratensis* | **30.6** | **1.5** | 6.2 | 1.6 | **30.7** | **0.9** | 24.8 | 1.8 | **29.9** | **1.6** | 33.5 | 2.4 |
| *Bromus vulgaris* | | | | | **0.7** | **0.7** | 0.7 | 0.7 | | | | |
| *Dactylis glomerata* | **7.6** | **1.7** | | | **3.6** | **1.0** | 0.6 | 0.4 | **1.5** | **0.6** | 0.2 | 0.1 |
| *Festuca arundinacea* | **5.3** | **0.9** | | | | | | | **0.3** | **0.3** | | |
| *Festuca occidentalis* | **2.1** | **3.3** | | | | | | | | | | |
| *Hordeum geniculatum* | **1.3** | **0.3** | 0.8 | 0.5 | | | 0.5 | 0.3 | | | | |
| *Lolium perene* | **1.4** | **0.6** | 45.4 | 6.2 | | | 26.8 | 4.0 | **0.4** | **0.3** | 33.5 | 4.4 |
| *Poa pratensis* L. | **30.6** | **3.3** | 37.9 | 1.0 | **20.0** | **3.6** | 28.4 | 2.4 | **28.3** | **1.1** | 19.3 | 4.4 |
| *Allium amplectens* | **0.5** | **0.5** | | | **0.5** | **0.4** | | | **2.4** | **0.9** | | |
| *Cerastium viscosum* | **0.1** | **0.1** | | | **2.3** | **1.6** | | | | | | |
| *Cirsium arvense* | | | 1.0 | 0.6 | **0.5** | **0.3** | | | **0.4** | **0.4** | 0.3 | 0.1 |
| *Convolvulus arvensis* | | | 7.0 | 3.5 | | | 4.4 | 2.8 | | | 4.9 | 2.8 |
| *Crepis capillaris* | | | 1.0 | 0.6 | | | | | **0.4** | **0.3** | 0.5 | 0.4 |
| *Daucus carota* | **0.2** | **0.2** | 0.8 | 0.5 | **0.5** | **0.5** | 0.1 | 0.1 | **0.5** | **0.4** | | |
| *Galium aparine* | | | | | **1.0** | **0.5** | | | **2.2** | **1.0** | | |
| *Galium triflorum* | **0.1** | **0.1** | | | | | | | | | | |
| *Geranium dissectum* | | | | | **2.2** | **1.8** | | | **2.2** | **1.3** | 0.2 | 0.1 |
| *Geranium robertium* | **3.8** | **1.8** | | | **0.2** | **0.2** | 0.2 | 0.2 | | | | |

Table 3.1 (continued)

| Plant species | 2000 Restored | | 2000 Ag. Managed | | 2001 Restored | | 2001 Ag. Managed | | 2002 Restored | | 2002 Ag. Managed | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Mean | SE | Mean | SE | Mean | SE | Mean | SE | Mean | SE | Mean | SE |
| *Gratiola ebracteata* | | | | | | | 1.7 | 1.3 | | | | |
| *Hypochaerus radicata* | | | | | | | 2.6 | 0.9 | **0.1** | **0.1** | 4.1 | 2.0 |
| *Lactuca muralis* | | | | | **0.5** | **0.3** | | | | | | |
| *Lactuca serriola* | **11.5** | **2.9** | | | | | 4.6 | 1.7 | | | | |
| *Medicago lupulina* | **0.4** | **0.4** | | | **0.1** | **0.1** | 0.6 | 0.6 | | | | |
| *Rumex crispus* | **0.3** | **0.2** | | | | | 0.4 | 0.2 | | | 0.5 | 0.3 |
| *Taraxacum officinale* | **0.2** | **0.2** | 1.3 | 0.8 | **0.2** | **0.2** | | | | | | |
| *Trifolium repens* | **2.3** | **1.1** | | | | | 2.7 | 1.2 | | | 2.5 | 0.9 |
| *Vicia sativa* | **0.3** | **0.3** | 1.7 | 0.7 | **0.1** | **0.1** | | | **2.0** | **1.3** | | |
| *Crataegus monogyna* | | | | | **1.0** | **0.1** | | | | | | |
| *Quercus garryanna* | | | | | **0.3** | **0.3** | | | | | | |

Table 3.2. Relative abundance (mean % and standard error) of plant species (covered bridge site) in 180, 50X50 cm. plots along permanent transects. Measurements were taken over a 3-year period. Agricultural management was excluded from restored area in fall of 1999. Data from year 2000 represent plant frequency before active hydrological restoration and data from years 2001 and 2002 represent plant frequency following said restoration activity. Standard errors were calculated using transects as the unit of measure(n = 6).

| Plant species | 2000 | | | | 2001 | | | | 2002 | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Restored | | Ag. Managed | | Restored | | Ag. Managed | | Restored | | Ag. Managed | |
| | Mean | SE | Mean | SE | Mean | SE | Mean | SE | Mean | SE | Mean | SE |
| *Agrostis tenuis* | | | | | 17.4 | 4.8 | 7.7 | 3.2 | 19.5 | 9.2 | | |
| *Alopecurus aequalis* | 3.2 | 1.9 | 28.6 | 10.1 | 6.5 | 2.8 | 16.2 | 3.7 | 6.6 | 2.6 | 7.7 | 6.7 |
| *Alopecurus pratensis* | 28.0 | 6.6 | | | 30.3 | 9.1 | 11.4 | 1.8 | 25.3 | 8.0 | 0.8 | 0.9 |
| *Echinochloa crusgalli* | | | | | | | | | 0.9 | 1.0 | | |
| *Festuca arundinacea* | 3.1 | 0.7 | | | | | | | 0.9 | 1.0 | | |
| *Festuca occidentalis* | | | | | 0.1 | 0.7 | | | 0.8 | 0.8 | | |
| *Glyceria occidentalis* | | | | | 1.8 | 0.7 | | | | | | |
| *Hordeum geniculatum* | 0.1 | 0.0 | | | 0.2 | 0.1 | 0.1 | 0.1 | | | | |
| *Juncus bufonis* | | | | | 0.7 | 0.4 | | | 0.1 | 0.1 | | |
| *Lolium multiflorum* | | | | | 0.2 | 0.2 | | | | | | |
| *Lolium perene* | 22.8 | 4.2 | 52.5 | 6.0 | 2.7 | 1.6 | 23.6 | 1.2 | 0.8 | 0.9 | 66.9 | 3.6 |
| *Poa pratensis.* | 12.6 | 2.2 | 4.9 | 2.5 | 14.5 | 4.2 | 21.1 | 2.0 | | | | |
| *Allium amplectens* | 0.2 | 0.2 | | | | | | | 0.2 | 0.1 | | |
| *Cerastium arvense* | | | 0.3 | 0.2 | | | | | 0.6 | 0.4 | | |
| *Cerastium viscosum* | | | 0.9 | 0.7 | | | 0.5 | 0.5 | | | 0.1 | 0.1 |
| *Chenopodium album* | | | | | | | | | 0.4 | 0.3 | | |
| *Cirsium arvense* | | | 0.2 | 0.2 | | | 0.1 | 0.0 | 0.3 | 0.2 | | |
| *Crepis capillaris* | | | | | | | | | | | 0.2 | 0.2 |
| *Dowingia yina* | | | | | 4.4 | 3.9 | | | 3.8 | 2.9 | | |

Table 3.2 (continued)

| Plant species | 2000 Restored | | 2000 Ag. Managed | | 2001 Restored | | 2001 Ag. Managed | | 2002 Restored | | 2002 Ag. Managed | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Mean | SE | Mean | SE | Mean | SE | Mean | SE | Mean | SE | Mean | SE |
| *Epilobian ciliatum* | | | | | | | | | **1.4** | **0.9** | | |
| *Geranium dissectum* | | | | | **0.2** | **0.2** | | | **0.1** | **0.1** | | |
| *Geranium robertium* | **1.5** | **0.9** | | | | | | | | | | |
| *Gnapthalium palustre* | | | | | | | | | **3.5** | **2.2** | | |
| *Gratiola ebracteata* | | | | | **1.4** | **1.3** | | | | | | |
| *Hypochaerus radicata* | **4.6** | **2.2** | | | **3.3** | **2.1** | 1.3 | 0.7 | | | | |
| *Lactuca muralis* | **1.4** | **0.8** | | | **2.2** | **2.2** | 0.6 | 0.3 | **1.6** | **1.2** | | |
| *Lactuca serriola* | **3.1** | **2.3** | | | **1.9** | **1.8** | 0.1 | 0.0 | **1.2** | **0.5** | | |
| *Leontodon nudicalis* | | | | | | | | | **4.9** | **2.7** | | |
| *Medicago lupulina* | **4.6** | **2.0** | | | **0.8** | **0.7** | 0.4 | 0.3 | | | | |
| *Montia linearis* | **0.2** | **0.2** | | | | | | | | | | |
| *Parentucellia viscosa* | **0.1** | **0.0** | | | | | | | | | | |
| *Plantago major* | | | | | | | 1.5 | 1.5 | | | | |
| *Polygonum aviculare* | | | | | | | | | **0.8** | **0.8** | | |
| *Ranunculus muricatus* | | | | | **1.0** | **0.9** | | | | | | |
| *Rorripa islandica* | | | | | **6.9** | **6.4** | | | **3.3** | **2.8** | | |
| *Rumex crispus* | **0.4** | **0.2** | 0.2 | 0.2 | **0.8** | **0.6** | 1.2 | 0.5 | **1.6** | **0.8** | 5.3 | 2.0 |
| *Sonchus asper* | | | | | | | 1.4 | 1.3 | | | | |
| *Stellaria longipes* | **0.1** | **0.0** | | | | | | | | | | |
| *Taraxacum officianalle* | | | | | **0.8** | **0.8** | | | | | | |
| *Tragopogon dubois* | | | | | | | | | **0.1** | **0.1** | | |
| *Trifolium pratense* | **0.2** | **0.2** | | | | | | | | | | |

Table 3.2 (continued)

| Plant species | 2000 Restored | | 2000 Ag. Managed | | 2001 Restored | | 2001 Ag. Managed | | 2002 Restored | | 2002 Ag. Managed | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Mean | SE | Mean | SE | Mean | SE | Mean | SE | Mean | SE | Mean | SE |
| *Trifolium repens* | 7.9 | **3.1** | 8.0 | 1.0 | **1.0** | **0.9** | 13.5 | 1.8 | **0.4** | **0.3** | 4.1 | 1.7 |
| *Veronica arvensis* | | | | | | | | | **0.8** | **0.5** | | |
| *Vicia sativa* | | | | | **0.3** | **0.2** | | | | | | |
| *Populus trichocarpa* | | | | | **0.1** | **0.1** | | | | | | |

Table 3.3. Relative abundance (mean % and standard error) of plant species in wetland sites in 60, 50X50 cm. plots along permanent transects. Actively restored is covered bridge site (*Alopecurus*) and passively restored is EPA site (*Phalaris*). Measurements were taken over a 3-year period. Both sites were excluded from agricultural management in the fall of 1999. Data from year 2000 represent plant frequency before active hydrological restoration and data from years 2001 and 2002 represent plant frequency following said restoration activity. Standard errors were calculated using transects as the unit of measure (n = 4).

| Plant species | 2000 | | | | 2001 | | | | 2002 | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Actively Restored | | Passively Restored | | Actively Restored | | Passively Restored | | Actively Restored | | Passively Restored | |
| | Mean | SE | Mean | SE | Mean | SE | Mean | SE | Mean | SE | Mean | SE |
| *Alopecurus aequalis* | **90.5** | **12.4** | | | **60.8** | **10.9** | | | **18.3** | **9.9** | | |
| *Alopercurus pratensis* | | | | | **0.3** | **0.5** | | | **0.2** | **0.3** | | |
| *Echinochloa crusgalli* | | | | | | | | | **5.5** | **4.2** | | |
| *Glyceria occidentalis* | | | | | **7.5** | **4.4** | | | | | | |
| *Phalaris arundinacea* | | | 100.0 | 0.0 | | | 100.0 | 0.0 | | | 100.0 | 0.0 |
| *Amaranthus retroflexus* | | | | | | | | | **8.3** | **5.1** | | |
| *Antennaria neglecta* | **2.3** | **4.0** | | | | | | | | | | |
| *Cerastium arvense* | **1.5** | **2.5** | | | | | | | | | | |
| *Chenopodium album* | **3.6** | **3.6** | | | **0.3** | **0.4** | | | **3.9** | **2.3** | | |
| *Cirsium arvense* | | | | | **0.6** | **1.0** | | | **3.5** | **3.2** | | |
| *Dipsacus sylvestris* | | | | | | | | | **0.5** | **0.8** | | |
| *Dowingia yina* | | | | | **2.8** | **4.2** | | | | | | |
| *Gnapthalium palustre* | | | | | **0.6** | **1.1** | | | | | | |
| *Gratiola ebracteata* | | | | | **1.7** | **2.9** | | | | | | |
| *Leontodon nudicalis* | | | | | | | | | **0.7** | **0.7** | | |
| *Plagiobothrys figuratus* | | | | | **0.5** | **0.5** | | | | | | |
| *Plantago major* | | | | | | | | | **0.2** | **0.3** | | |

Table 3.1 (continued)

| Plant species | 2000 | | | | 2001 | | | | 2002 | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Actively Restored | | Passively Restored | | Actively Restored | | Passively Restored | | Actively Restored | | Passively Restored | |
| | Mean | SE | Mean | SE | Mean | SE | Mean | SE | Mean | SE | Mean | SE |
| *Polygonum aviculare* | | | **0.3** | **0.6** | | | | | **1.0** | **1.6** | | |
| *Polygonum punctatum* | | | **4.4** | **5.1** | | | | | **7.6** | **6.9** | | |
| *Ranunculus aquatilis* | | | **1.7** | **2.9** | | | | | | | | |
| *Rorrippa islandica* | **2.1** | **2.9** | **16.8** | **5.5** | | | | | **47.8** | **4.8** | | |
| *Rumex crispus* | | | **0.4** | **0.7** | | | | | **0.8** | **1.3** | | |
| *Trifolium repens* | | | | | | | | | **0.7** | **1.2** | | |

## Results

### Plant Species Richness

A total of 59 species were identified in all communities (Table 3.0). For the restored mesic meadow (covered bridge) year 2000 showed an average 12 species, with a maximum of 21 and minimum of 6 species/transect identified. During post treatment year 2001 an average of 10 species occurred on those same transects, with a maximum of 23 and a minimum of 3 species identified. During post treatment year 2002 an average of 11 species occurred, and a maximum of 17 and minimum of 4 species were identified for each transect (Figure 2.0). Changes in plant species richness through time were not significant for the actively restored (covered bridge) mesic meadow ($p = 0.25$, 0.32, and 0.31, respectively).

The agriculturally managed mesic meadow (control covered bridge site) exhibited a mean of 5 species per transect with a maximum of 6 and a minimum of 3 species per transect for the pre-treatment year 2000. During post treatment year 2001 a mean of 9 plant species and a maximum of 13 and minimum of 6 were recorded. Post treatment year 2002 displayed an average of 4 species with minimums and maximums of 6 and 3 respectively (Figure 3.0). Species richness significantly increased between the year 2000 and 2001 from a mean of 5 to a mean of 9 plant species per transect ($p=0.009$). Plant species richness significantly decreased between years 2001 and 2002 from a mean 9 species/transect to a mean 3.8 species/transect ($p=0.006$). Changes in plant species richness were not statistically significant when pre-treatment 2000 means were compared to year 2002 ($p=0.17$). The decrease in plant species richness between year 2001 and 2002 correspond to a plowing and re-seeding event done in that site between sampling dates. The data indicated that between seeding events this agriculturally managed mesic meadow exhibited an increase in plant species richness but quickly decreased in response to plowing and re-seeding.

The agricultural excluded mesic meadow (passively restored EPA site) exhibited a mean of 10 species per transect and a maximum of 11 and minimum of 8 species per transect for the pre-treatment year 2000. During post treatment year 2001 a mean of 8 species per transect and a maximum of 11 and minimum of 5 species were recorded. For post treatment year 2002 a mean of 8 and maximum of 12 and minimum of 3 species were identified in these transects (Figure 3.0). There is reasonable evidence (p=0.04) that species richness decreased from a mean of 10 to a mean of 8 between the years 2000 and 2001. Changes in plant species richness between years 2001 and 2002 were found to be non-significant (p=0.20). However, when mean species richness from pre-treatment year 2000 and 2002 were compared statistical evidence showed that a significant decrease in plant species richness had occurred (p=0.03). There is reasonable evidence that species richness decreased from 10 to 8 during the first year post treatment and by post treatment year 2002 species richness was still lower than that of the pre treatment year (10 and 8, respectively). This evidence suggests that species richness decreased for both post treatment years within this agriculturally excluded landscape. Interestingly, nuisance plant species demonstrated the greatest decreases in this time period and many were absent by the end of the study (e.g. *Lactuca serriola, Taraxacum officionale, Medicago lupulina*; Table 3.1).

Agriculturally managed mesic meadow (control EPA site) exhibited a mean of 5 species and a maximum of 6 and minimum of 2 species/transect for the pre-treatment year. During post treatment year 2001 a mean of 8 and maximum of 13 and minimum of 3 species per transect were recorded. For post treatment year 2002 a mean of 6 and a maximum of 10 and minimum of 3 species per transect were recorded (Figure 3.0). Species richness changes were found to be significant between years 2000 and 2001 (p=0.02). This represents a mean increase in species richness from 5 to 8 species per transect. There is also strong evidence that between years 2001 and 2002 species richness decreased from 8 to 6 (p=0.01). Although species richness changes were significant between years 2000 and 2001 as well as 2001 and 2002 changes were not significant between years 2000 and 2002. These results show an increase in plant species richness during the first year post treatment followed by a decrease the second post treatment year resulting in an overall non-significant change for the study period.

Species richness in actively restored wetland transects (covered bridge) for pre-treatment year 2000 had a mean species richness of 3 and a maximum of 5 and minimum of 1 plant species per transect. Post treatment year 2001 showed a mean of 7 and a maximum of 9 and minimum of 5 species per transect. Post treatment year 2002 exhibited a mean of 9 species and a maximum of 11 and minimum of 8 species (Figure 3.0). There is reasonable evidence that species richness increased from a mean 4 species per transect to a mean of 7 ($p=0.05$) between pre-treatment year 2000 and post treatment year 2001. Additionally, species richness increased between years 2001 and 2002 ($p=0.07$). Species richness was evaluated for the entire study period (2000-2002) and increases displayed strong statistical evidence ($p=0.002$). This suggests that by the first year post treatment a considerable number of species had invaded this actively restored wetland and this increase continued through the second post treatment year with a small increase in plant species.

Agriculturally excluded wetlands (control EPA) exhibited no changes during the study period. The dominant and only plant occurring in this seasonally inundated wetland was Reed canary grass (*Phalaris arundinacea*). This dominant invasive species appeared to competitively exclude all other vegetation. This site contained only one species and no changes were recorded during all study years (Figure 3.0).

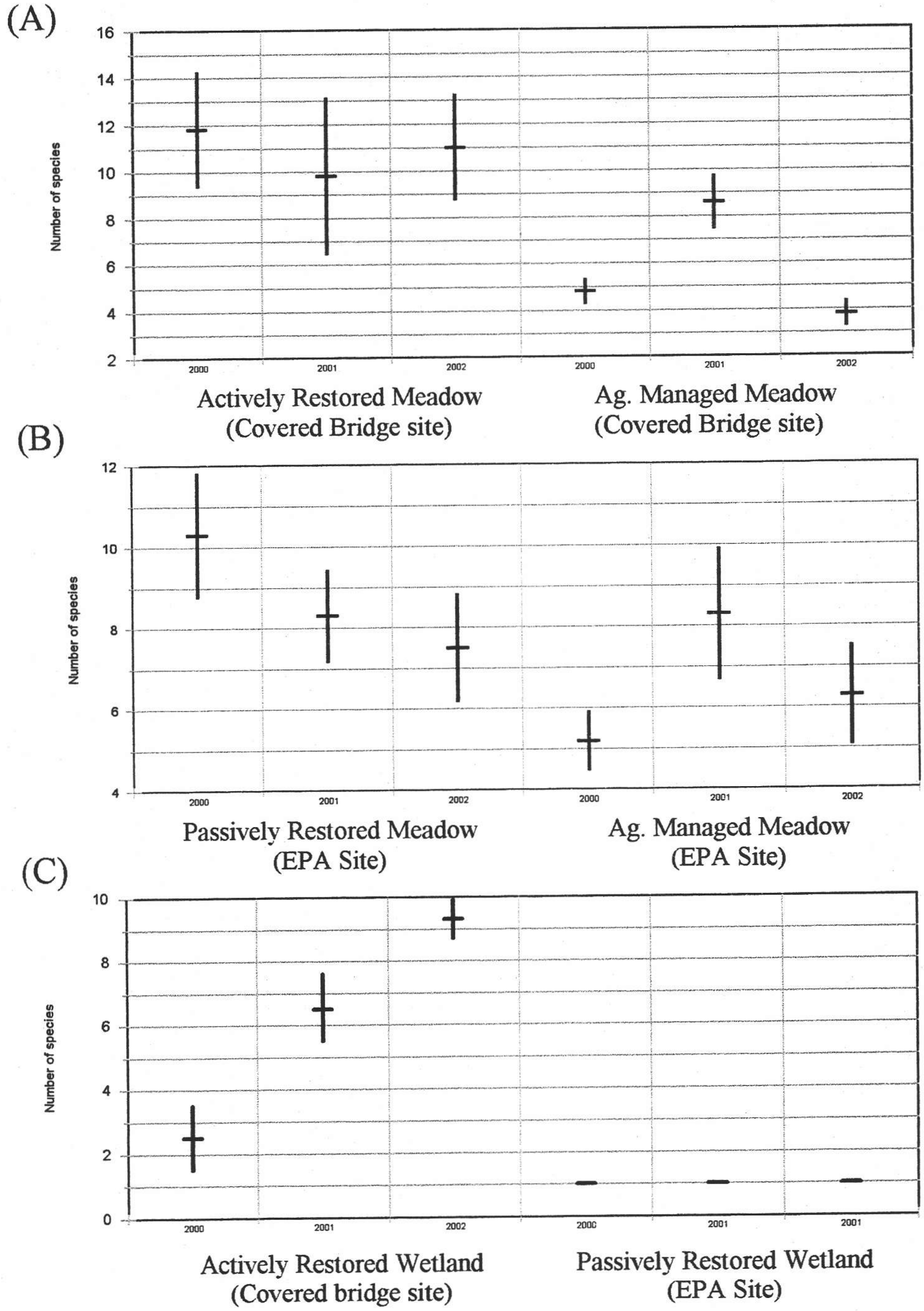



Figure 3.0. Species richness (number of plant species present) in all research sites. Sites are; (A) Mesic meadow covered bridge research site. (B) Mesic meadow EPA research site. (C) Wetland meadows in both research sites. Horizontal bar indicates mean value for transects and vertical bar indicates standard error. Year 2000 is pre-treatment and years 2001 and 2002 are post treatment.

## Native Plant Abundance

Only 15 of 59 plant species were identified to be of native origin in all communities combined. During the pre-treatment year 2000, the actively restored mesic meadow (covered bridge) had native plants comprising 4% (mean) of the relative species abundance with a range of 0 - 10% per transect. During the first post treatment year (2001), mean native plant species relative abundance in the actively restored mesic meadow (covered bridge) was 22% with a range of 0 - 100% per transect. The second post treatment year (2002) exhibited a mean of 20% with a range of 0 - 60% per transect in the actively restored mesic meadow (covered bridge; figure 3.1). All transects within this actively restored mesic meadow exhibited a propensity for some microplots to consist of all non-native plant species while only post treatment year 2001 exhibited at least one microplot with 100% native species. Mean native plant relative abundance in the actively restored mesic meadow (covered bridge) exhibited an increase from 4% to 22% between pre-treatment year 2000 and post treatment year 2001. However, this change was not statistically significant (p=0.15). Native plant relative abundance change was non-significant between year 2001 and 2002 (p=0.40). Native plant relative abundance change for the entire study period (2000-2002) was somewhat statistically significant (p=0.08). This reflects that native plant relative abundance may have increased (4%-20%) during the study period (2000 – 2002).

The agriculturally managed mesic meadow (control, covered bridge site) had a mean 29% with a range of 6 - 58% native species abundance for pre-treatment year 2000. The first year post treatment (2001) exhibited an average 16% with a range of 8 - 33% native species abundance. The second post treatment year (2002) exhibited an average 9% with a range of 0 - 44% native species abundance (Figure 3.1). Native plant abundance decreased from a mean 29% to a mean of 16% between pre-treatment year (2000) and the first post treatment year (2001) which was found to be statistically significant (p=0.07). Native plant abundance decreased from an average of 29% in pre–treatment year 2000 to an average of 9% by the second post treatment year (2002), which was statistically significant (p=0.02). Changes in native plant abundance were marginally significant between years 2001 and 2002 (p=0.06). This information suggests that native plant abundance changes demonstrated a trajectory of decreasing values during the entire study period. This may indicate that agricultural management at this site is a deterrent to the establishment of native plant species.

The agricultural excluded mesic meadow (EPA site) exhibited a mean of 3% native spp. in the community composition with a range of 0 - 4% for pre-treatment year 2000. The first post treatment year (2001) displayed a mean 2% with a range of 0 - 5% native plant abundance. The second post treatment year (2002) exhibited a mean 4% with a range of 0-11% native plant species abundance (Figure 3.1). Native plant abundance changes between 2000 to 2001, 2001 to 2002, and 2000 to 2002 were non-significant (p=0.31, 0.31, 0.23, respectively).

There were practically no native species in the agriculturally managed mesic meadow (control, EPA site). This site exhibited a mean 0.3% with a range of 0 - 2% native plant abundance for pre-treatment year (2000). The first post treatment year (2001) had a mean of 2% with a range of 0 - 4% native plant abundance. The second post treatment year (2002) had a mean 0.2% with a range of 0 - 0.9% native plant abundance (Figure 3.1). Mean native plant species abundance increased between pre-treatment year (2000) and the first post treatment year (2001; p=0.09). However, the results indicate that mean native plant abundance decreased from 1.45% to 0.23% between the first post treatment year (2001) and the second post treatment year (2002; p=0.04). Because the initial increase in native plant abundance (2000-2001) was followed by a decrease (2001-2002) the change over the entire study period (2000-2002) amounted to a small decrease in mean native plant abundance (p=0.07). These changes were very small and are most likely biologically insignificant.

Native species abundance in actively restored wetland (*Alopecurus*) transects (covered bridge) for pre-treatment year 2000 had a mean of 92% with a range of 77- 100% native plant abundance. The first post treatment year (2001) displayed an average of 92% with a range of 90- 94%. The second post treatment year (2002) had a mean 66% with a range of 49- 83% native plant abundance (Figure 3.1). Statistical evidence shows that no significant changes in mean native plant species abundance between pre-treatment year (2000) and the first post treatment year (2001) occurred (p=0.47). However, between the first post treatment year (2001) and the second post treatment year (2002) a decrease from 93% to 66% was found to be statistically significant (p=0.04). Statistical analysis also shows significant differences between means over the entire study period (2000-2002; p=0.01). Native plant species abundance did not significantly change over the first year of this study; however, by the second post treatment year (2002) a significant decrease was exhibited.

The *Phalaris* dominated agriculturally excluded wetlands (EPA) exhibited no changes during the study period. Reed canary grass was the dominant and only plant occurring in this seasonally inundated wetland. This species appeared to have excluded all other vegetation during all study years (Figure 3.1).

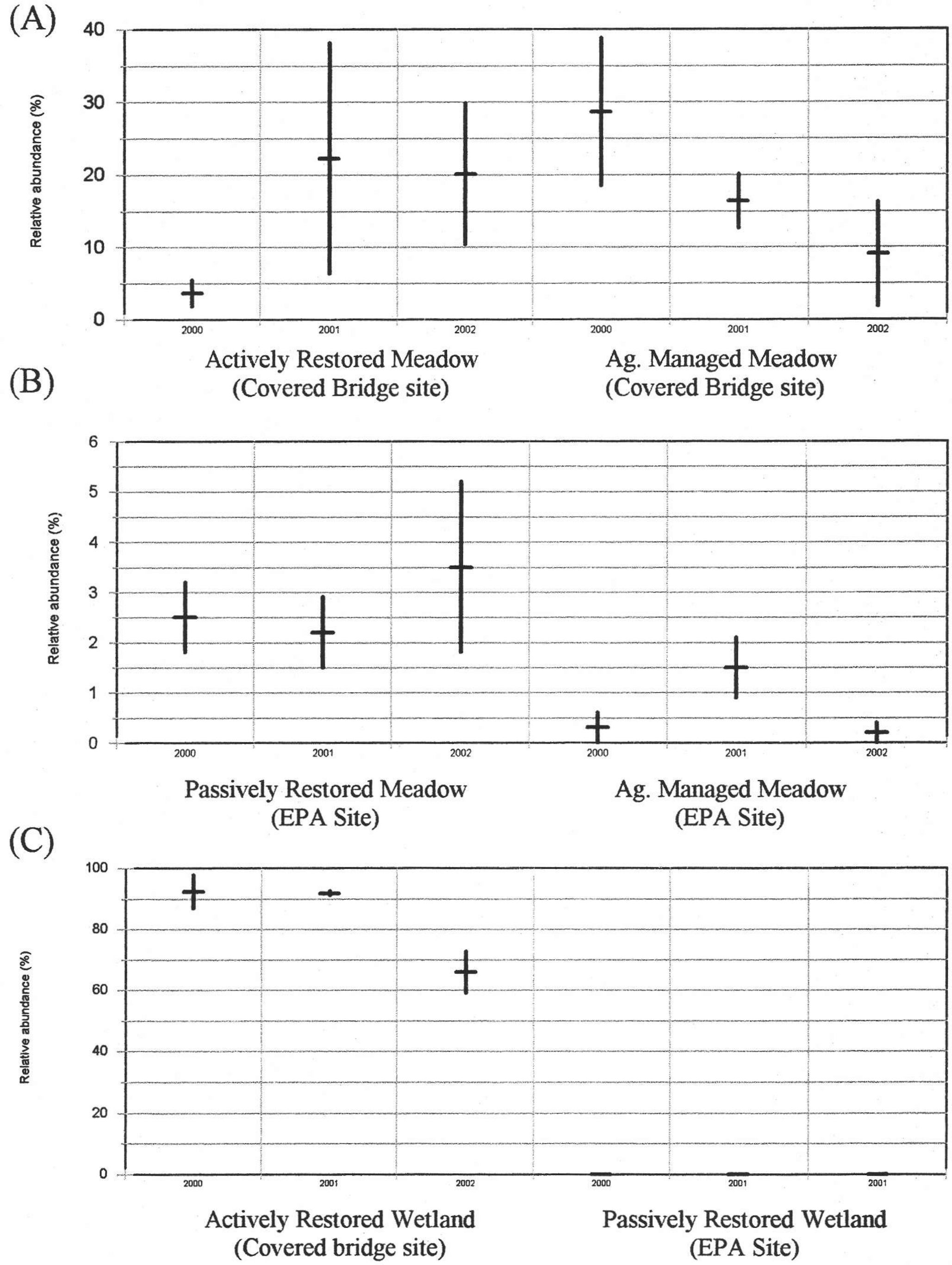



Figure 3.1. Relative abundance of native plant species in all research sites. Research sites are; (A) Mesic meadow, covered bridge research site. (B) Mesic meadow, EPA research site. (C) Wetland meadows, in both research sites. Horizontal bar indicates mean value for transects and vertical bar indicates standard error. Year 2000 is pre-treatment and years 2001 and 2002 are post treatment.

## Wetland Indicator Abundance

Prior to treatment (2000) there was a wetland species relative abundance of 55% with a range of 16 - 71% in the actively restored mesic meadow (covered bridge site). The first post treatment year (2001) exhibited a mean of 69% with a range of 38 -100% wetland species abundance. During the second post treatment year (2002) wetland species abundance was 54% with a range of 34 - 71% (Figure 3.2). Statistical analysis determined that the changes in wetland plant abundance between the pre-treatment year (2000) and the first post treatment year (2001) and between post treatment years (2001 to 2002) were significant ($p= 0.1$ and 0.03, respectively). However, wetland indicator abundance changes between the pre-treatment year (2000) and the second post treatment year (2002) were not statistically significant ($p=0.5$).

Agriculturally managed mesic meadow (control, covered bridge) exhibited a mean 42% with a range of 21 - 65% wetland plant abundance during pre-treatment year 2000. The first post treatment year (2001) displayed a mean 64% with a range of 56-77% wetland plant abundance. The second post treatment year 2002 had a mean of 22% with a range of 9 - 54% wetland plant abundance (Figure 3.2). Statistically significant changes were observed between pre-treatment year 2000 and post treatment year 2001 ($p=0.004$). Wetland plant species abundance decreased significantly from a mean of 42% to a mean of 22 % between post treatment years 2001 and 2002 ($p<0.0001$). Overall change during this study (2000-2002) displayed a decrease from 42% to a mean of 22% wetland plant abundance ($p=0.008$).

The agricultural excluded mesic meadow (EPA site) displayed an average 64% with a range of 40 - 81% wetland species abundance for pre-treatment year 2000. The first post treatment year (2001) had a mean 56% with a range of 48 - 63% wetland plant species abundance. The second post treatment year (2002) displayed a mean 58% with a range of 49 - 66% wetland plant species abundance (Figure

3.2). Changes in wetland plant species abundance were found to be statistically significant between 2000 and 2001 as well as 2001 and 2002 ($p=0.08$,and 0.1, respectively). However, because wetland plant abundance increased the first year and then decreased the second year, changes over the entire study period were not significant ($p=0.2$).

Agriculturally managed mesic meadow (control, EPA site) exhibited an average 47% with a range of 38 - 52% wetland plant abundance for pre-treatment year 2000. The first post treatment year (2001) exhibited an average 56% with a range of 46 - 64% wetland plant abundance. The second post treatment year (2002) had a mean 56% with a range of 41 - 68% wetland plant species abundance (Figure 3.2). There is strong evidence ($p=0.03$) that mean wetland indicator plant abundance increased form 47% to 56% during the first year after treatment. However, this trajectory did not continue between years 2001 and 2002 when mean wetland plant abundance was static ($p=0.5$). This represents an increase from 47-56% in wetland plant species over the study period (2000-2002; $p=0.02$).

Wetland plant species abundance in the actively restored wetland (*Alopecurus*) transects (covered bridge site) exhibited an average 96% and a range of 85 - 100% for pre-treatment year 2000. During the first post treatment year (2001) actively restored wetland transects displayed a mean of 99% with a range of 98 - 100% wetland plant species abundance. The second post treatment year (2002) exhibited a mean of 98% with a range of 92 - 100% wetland plant species abundance (figure 3.2). This represents a statistically stable abundance of wetland plant species ($p=0.25$, 0.27 and 0.23, respectively) as no significant changes occurred during the entire study period.

Agriculturally excluded wetlands (*Phalaris,* EPA site) exhibited no changes during the study period. *Phalaris arundinacea* the dominate and only plant species occurring in this wetland is a wetland obligate species (Figure 3.2).

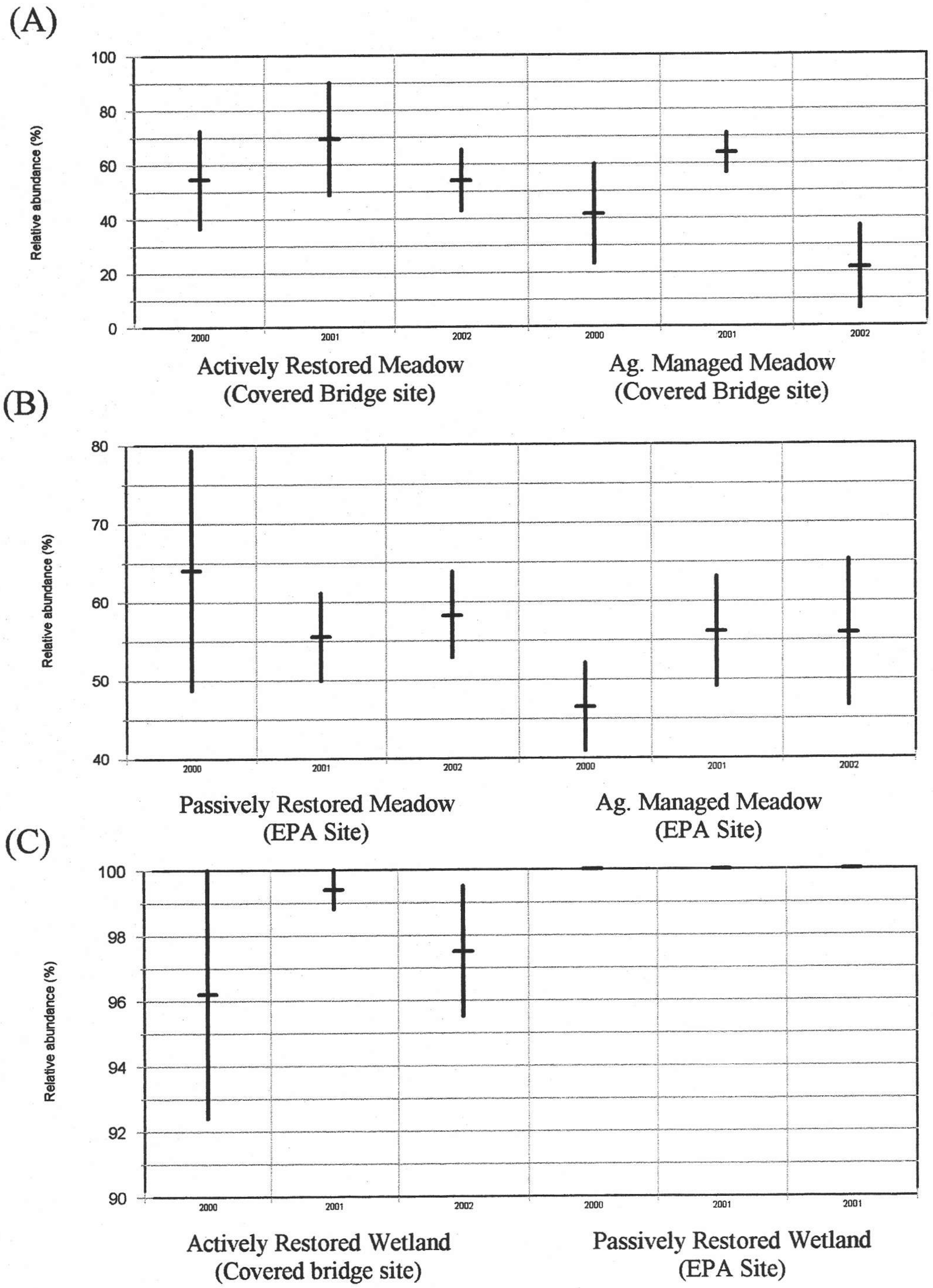



Figure 3.2. Relative abundance of wetland indicator plants in all research sites. Research sites are; (A) Mesic meadow, covered bridge research site. (B) Mesic meadow, EPA research site. (C) Wetland meadows in both research sites. Horizontal bar indicates mean value for transects and vertical bar indicates standard error. Year 2000 is pre-treatment and years 2001 and 2002 are post treatment.

## Nuisance Weed Abundance

Nuisance weeds are a concern for most agricultural land managers. Nuisance weeds are defined as those plants that reduce agricultural crop yields or harbor common agricultural pathogens (Table 3.0).

During pre-treatment year 2000 transects within the actively restored mesic meadow (covered bridge) exhibited a mean 22% relative abundance of nuisance weeds. The first post treatment year (2001) displayed a mean 10% nuisance weed relative abundance. During the second post treatment year (2002) there was a mean 15% nuisance weed abundance (Table 3.3). This decrease (22% to 10%) in nuisance weed abundance between 2000 and 2001 was statistically significant ($p=0.05$). However, this decrease was followed by an increase (10% to 15%) the following year (2001-2002). Although an increase was recorded between the first and second post treatment years overall changes showed a decrease from 22 –15% over the study period ($p=0.09$). Active restoration at this site resulted in the reduction of nuisance weeds.

The agriculturally managed mesic meadow (control, covered bridge) displayed a mean 10% weed abundance for pre-treatment year 2000. The first post treatment year (2001) exhibited a mean 17% nuisance weed abundance. During the second post treatment year (2002) a mean 11% nuisance weed abundance was recorded (Table 3.3). Statistical evidence shows that nuisance weed abundance increased from a mean of 10% to a mean of 17% between years 2000 and 2001 ($p=0.03$). However, the following study year exhibited a decrease of 6% in mean nuisance weed abundance ($p=0.07$). This decrease in nuisance weed abundance occurred during the year in which this agriculturally managed field was plowed and re-seeded. This evidence suggests that between years when agricultural managed fields are not plowed and re-seeded nuisance weeds abundance increase. However, nuisance weeds were reduced in years that aggressive plowing and re-seeding occurred.

During the pre-treatment year (2000) a mean 15% relative abundance of nuisance weeds were recorded in the agriculturally excluded mesic meadow (passive restored, EPA). During the first post treatment year (2001) the mean nuisance weed relative abundance was reduced to 2%. The second post treatment year (2002) exhibited a mean of 1% (Table 3.3). Evidence shows a strongly significant decrease (15 - 2%) in noxious weed abundance between years 2000 and 2001 (p=0.004). That reduction was static through the end of the study period (2000-2002). These data are reflected in the statistical evidence that noxious weed abundance decreased by a factor of 10.8 during the entire study period (p=0.004). Passive restoration decreased the abundance of nuisance weeds at this site and maintained this reduction throughout the period of this study.

The untreated pasture (control, EPA site) did not follow the same pattern as the covered bridge site. Here nuisance weed relative abundance had a mean 10% during pre-treatment year 2000. During the first post treatment year (2001) nuisance weed relative abundance was 14%. This was followed by a mean 13% nuisance weed relative abundance in 2002 (Table 3.3). During the study the agriculturally managed meadow exhibited a trajectory of increasing nuisance weed abundance. These increases were found to be statistically significant (p=0.06 and 0.10, respectively). This suggests that agricultural management in this meadow increased the abundance of nuisance weeds during this study. This is consistent with the agriculturally managed control site (covered bridge) during years that plowing and re-seeding did not occur.

The actively restored wetland (*Alopecurus,* covered bridge site) had a nuisance weed relative abundance of 4% during the pre-treatment year (2000). The first post treatment year (2001) was similar with a nuisance weed relative abundance of 5%. The second post treatment year (2002) showed a dramatic increase with 26% nuisance weed relative abundance (Table 3.3). Following the intense hydrological change associated with wetland restoration (2000 – 2002) nuisance plant abundance increased from 4% to 26%, which represented an increase of over 700% (p=0.009). Speculation leads one to believe that actively restoring this wetland created conditions (early seral stage) that allowed easy establishment of new species. In this case these species were predominantly nuisance weeds. I suggest that because establishment took two years to exhibit this result early intervention by a restoration practitioner could easily facilitate establishment of native plant communities while excluding some weeds.

Agriculturally excluded wetlands (*Phalaris*, control EPA site) exhibited no changes during the study period. *Phalaris arundinacea* the dominant and only plant species occurring in this wetland is considered a nuisance weed (Figure 3.3).

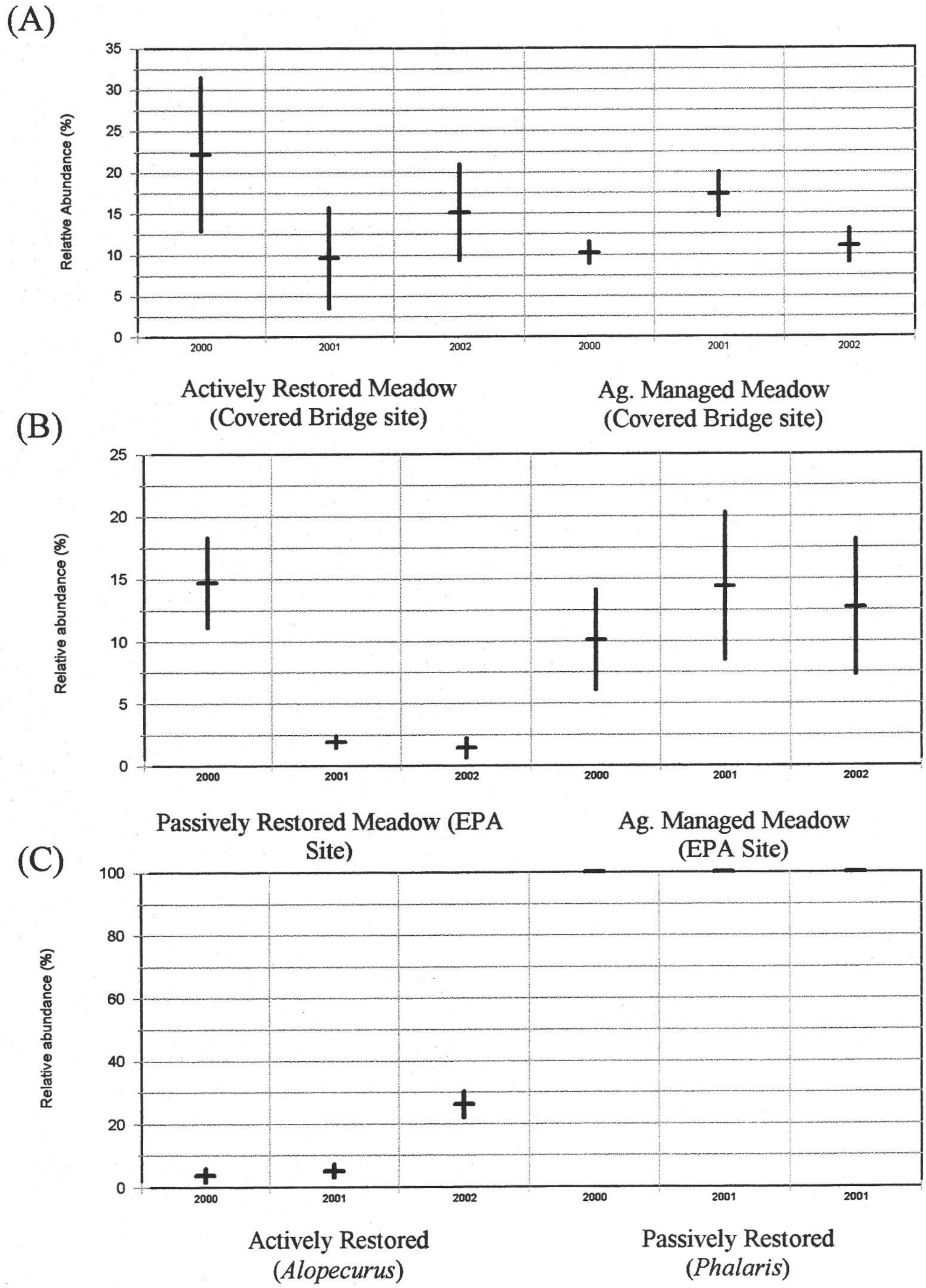



Figure 2.6. Relative abundance of nuisance plants in all research sites. Research sites are; (A) Mesic meadow, covered bridge research site. (B) Mesic meadow, EPA research site. (C) Wetland meadows, in both research sites. Horizontal bar indicates mean value for transects and vertical bar indicates standard error. Year 2000 is pre-treatment and years 2001 and 2002 are post treatment.

## Discussion and Conclusion

### Plant Species Composition

There were many changes in plant species composition associated with the differing management practices. Plant species richness tended to increase in the agriculturally managed fields. However, agricultural management that included plowing and re-seeding (control, covered bridge site) demonstrated complete replacement of plant communities and consequent reduction of species richness. Conversely, between plowing and re-seeding events, these same fields demonstrated an increase in species richness that included increases in nuisance weeds and a decrease in native plant abundance. Thus, any increase in species richness in theses agriculturally managed fields was attributed to the introduction and proliferation of nuisance weeds. Increases in nuisance weeds were likely promoted by the disturbance that is associated with agricultural management. Disturbance has been shown to provide open patches and opportunities for colonization of new species (Baldwin and Mendelssohn 1998). Likewise, the agriculturally managed meadow predominately used for grazing and haying (control, EPA site) demonstrated increased species richness accompanied by increased nuisance weed abundance and decreased native plant abundance. This would be an expected result as grazing has been shown to reduce biomass production of riparian plant species (Clary 1999) which may provide open gaps or re-generative niches for the introduction of invasive or nuisance plant species (White 1979, Grubb 1977). I conclude that in both cases agricultural management has contributed to an increase in nuisance weeds and a decrease in native plants.

Conversely, both the actively restored and the passively restored mesic meadows demonstrated decreasing species richness accompanied by increased native plants and decreased nuisance weed abundance. This decrease in species richness is consistent with findings of other restoration studies (e.g., Leck and Leck 1998, Odland 1997) Although there was a decrease in plant species present it was nuisance or unwanted plants that decreased. Although restoration is inherently at risk of introducing invasive species (Smith and Kadlec 1983), relative abundance of nuisance weeds decreased in both passively and actively restored mesic meadows during this study.

The actively restored wetland (*Alopecurus*, covered bridge site) demonstrated an increase in species richness largely related to an increase in nuisance weeds and accompanied by a decrease in native plants. This may have been contributed to the hydrologic disturbance created by filling the drainage ditch. Because changing water levels have been known to alter species diversity (Keddy 2000) and establishment of invasive species is often associated with disturbance (White 1979), this may be an expected result of filling drainage ditches. Because this wetland was inundated for much longer periods of time the dominant plant community was eradicated leaving essentially bare ground. This early seral stage allowed other species to establish that may not have otherwise. Due to the close proximity of nuisance weed species one may expect their proliferation into this newly created habitat. Other research studies found that disturbances such as this provide regenerative niches for species that would otherwise be excluded (Grubb 1977).

Wetland indicator abundance tended to remain static in both the passively restored and actively restored mesic meadows. Both agriculturally managed control sites exhibited a significant change in wetland indicator abundance over the study period. The agriculturally managed meadow at the covered bridge site showed a decrease in wetland plant species due to plowing and re-seeding, which was an expected result. However, the agriculturally managed mesic meadow at the EPA site demonstrated an increase in wetland indicator species. This was an unexpected result that may be explained by the different grazing management used at the site during this study. Prior to this study this meadow was grazed by up to 100 sheep in intense short interval rotations. During each rotation grass was grazed to a very short stature then allowed to grow back. However, during the study years a different grazing strategy was used. Grazing pressure was less intense during study years because fewer sheep (10-20) were rotated more frequently, leaving taller grass.

Agricultural excluded wetland (*Phalaris,* EPA site) showed no changes in plant community composition over the entire study period. This suggests that once Reed canary grass has established in an area it is persistent and dominant. Many studies have implicated Reed canary grass as a threat to native plant diversity (Green and Gallovitch 2002) and suggested it may inhibit the germination and growth of other plant species (Bosy and Reader 1995).

Evidence from this study shows that restoration done either passively or actively can increase the abundance and occurrence of native plant species and reduce the abundance of nuisance weed species. However, when active restoration dictates the physical or indirect removal of dominant plant species it may initiate a trajectory of community development not previously seen at the site or adjacent reference sections (Combroux et. al. 2002). Because active wetland restoration can create conditions favorable for the establishment of unwanted species, species in the vicinity of the restoration could and probably do contribute to the future plant community of that site.

Evidence such as this would suggest the need for an active re-vegetation management regime after active wetland restoration has been implemented. This re-vegetation strategy should be initiated for at least the first several years post treatment if a desirable plant community is to be obtained. Moreover, if surrounding vegetation is of an unwanted nature then more desirable vegetation must be brought to the site probably within one-year post treatment to negate the possibility of an unwanted plant community establishing. If the area to be restored has an established community of Reed canary grass it would probably require extensive manipulation to remove and constant maintenance for several years to allow other plant species to establish. Additionally, agricultural areas that are to be abandoned should be done so without disturbance to the dominant grass species.

Cropping, plowing, cutting, or grazing grass to short heights may facilitate the occurrence of nuisance and/or invasive plant species.

# 4. SUMMARY

The importance of wetlands as plant and wildlife habitat is well known. Additionally, research has shown that the occurrence of wetlands helps to promote clean water by reducing nutrient and sediment loads of waters, as such, wetlands lessen the impact of excessive nutrients on downstream aquatic ecosystems (Saunders and Kalff 2001). Because non-point source pollution contributes over 65% of the nutrient load to surface waters of the United States (Olson 1992), and intact wetlands function to reduce this nutrient load, wetland restoration and enhancement are important management tools. Nutrient management increasingly includes forested riparian buffers and the construction or enhancement of wetlands (Franklin et al. 2000). This study examined the efficacy of methods of implementing restoration management in riparian wetland meadows of the Willamette Valley, Oregon.

Active restoration of damaged hydrological features (filling ditches) initiated an immediate increase in water table elevation and consequent reduction in the variation of the water table elevation (Chapter 2). Water tables were shallower (closer to surface) and less variable through time after restoration activities were implemented. Other studies have shown that changes such as these have a positive effect on wetland functions; longer hydraulic residence time and increased submersed areas increase the capacity of wetlands to remove nutrients received from tributaries (Krieger 2003). Water table elevation was also measured at distances away from restoration activities in the adjacent floodplain.
Increases in water table elevation were measurable at distances greater than 100 m from the actual site of restoration. This suggests implementation of restoration contributed to an expansion in wetland size. This increased size will likely have a positive affect on wetland function; larger wetlands have been shown to reduce a greater proportion of pollutants than smaller ones (Krieger 2003). This study suggests that filling drainage ditches is a useful restoration tool and can

contribute to increased water table elevation and sustained inundation in wetlands.

During this study, research sites experienced very different annual precipitation regimes. Although there was over 43 cm less rainfall in the first post treatment year, water table elevation was higher in the actively restored wetland and adjacent meadow. Other studies have shown annual and seasonal variations in precipitation influences hydrology of wetlands, including depth to water table. Moreover, other studies have found that during periods of low precipitation (drought), average monthly water table elevation is lower in flood plains (Moorhead 2003, Mann and Wetzel 2000). This suggests that active restoration of hydrology features at this site contributed to moderation of water table elevations from year to year. Presumably, this moderating effect would contribute to persistent water table elevations through time and would enhance its reliability as wildlife habitat (e g. amphibians, waterfowl).

Plant community dynamics were also characterized in this study (Chapter 3). Implementation of restoration management regimes of active restoration and passive restoration were evaluated. Passive restoration (livestock exclusion) demonstrated a desirable trajectory towards the potential natural plant community. Native plant abundance increased and unwanted plant species decreased in passively restored meadows. However, passively restored wetlands dominated by Reed canary grass showed no changes over the entire study period. This invasive, introduced grass species appears to competitively exclude other species from establishment. Actively restored meadows demonstrated a trajectory of increasing native plants and decreasing nuisance weed species except where restoration activities excessively disturbed existing plant communities. Areas that had existing plant communities removed physically or by hydrologic changes demonstrated increased nuisance weeds and decreased native plant abundance. Perhaps, the direct or indirect removal of plant assemblages created an early succession stage advantageous to the establishment of new species. Because restoration activities occurred within agricultural landscapes composed of mostly introduced plant species this may be an expected result, because seed availability was influenced by exotics. Restoration such as this may have different consequences in ecosystems with more intact native plant populations. Additionally, because of the relatively short time span of this study (3 years), long term outcomes may be different.

Agriculturally managed control sites demonstrated increasing species richness that was contributed to increased nuisance weed species. In general, agricultural management, including grazing and haying, promoted increased nuisance weeds and decreased native plant abundance.

Collectively, this study demonstrates the profound influence repairing hydrological features can have on hydrologic character of wetlands and adjacent floodplains. Filling drainage ditches can contribute to the persistence of wetlands through years of low precipitation. Because evidence from this study is most likely applicable to similar landscapes, one may expect that abandonment of agricultural fields immediately after plowing, grazing, or cropping may create conditions that promote invasive and unwanted plant establishment. However, abandonment of intact agricultural meadow plant communities appears to initiate a trajectory towards the potential natural community.

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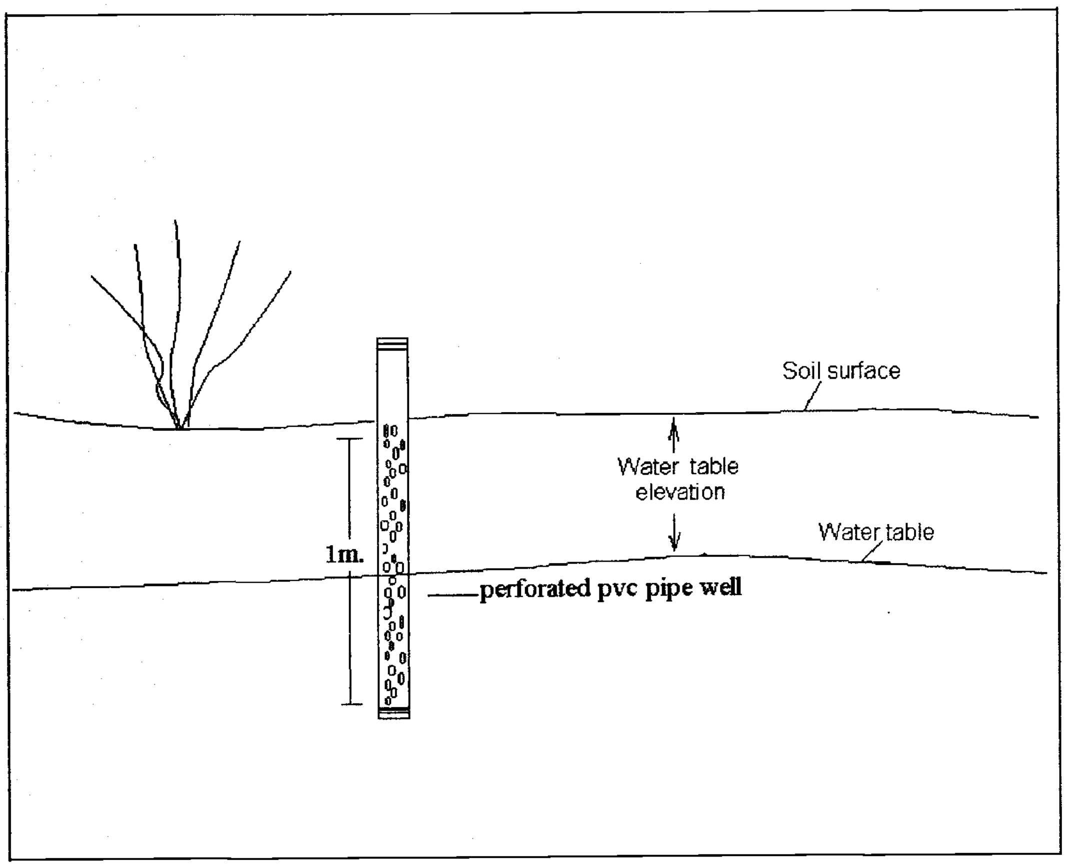
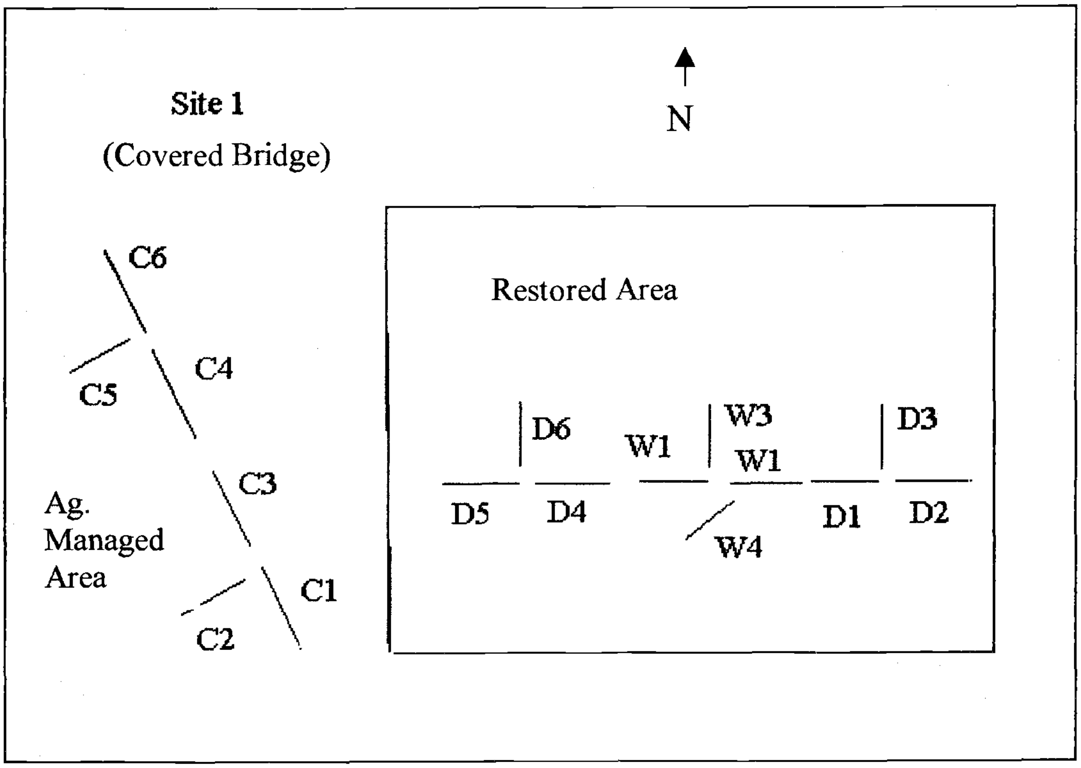
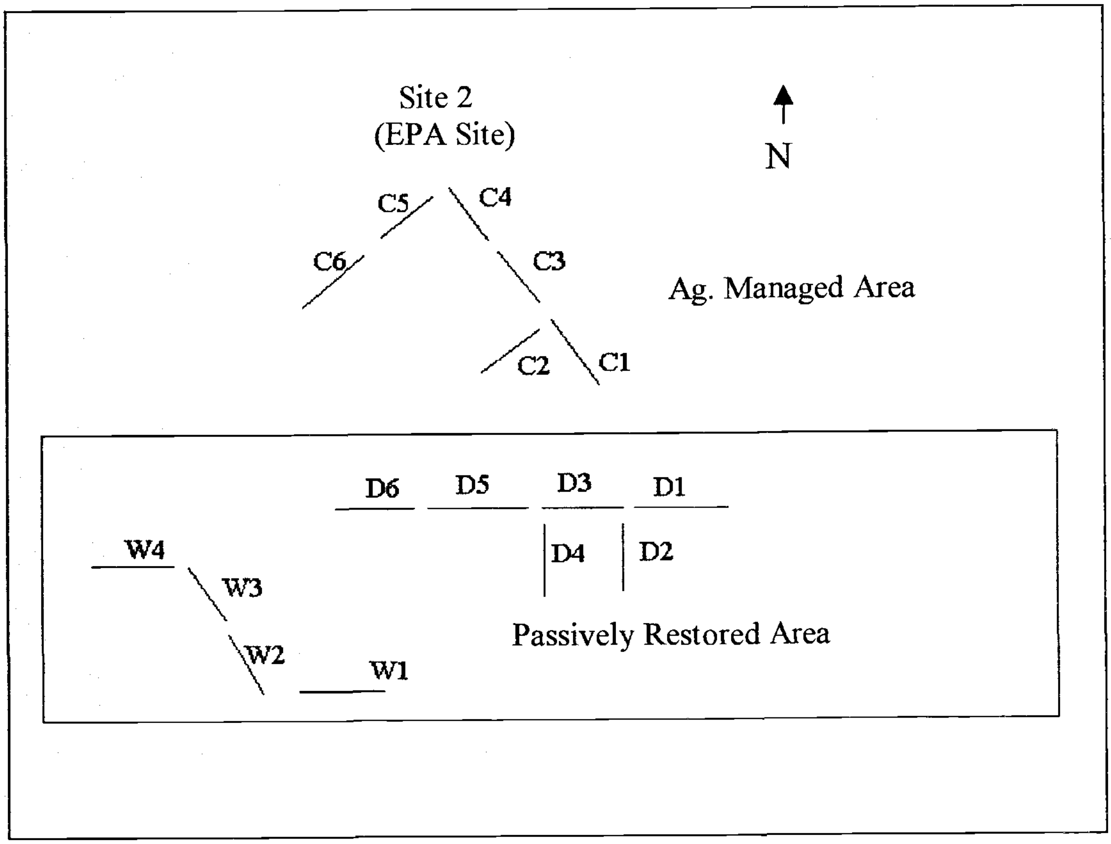
APPENDICES




Appendix 2.0. Diagram of well installation and water table elevation.




Appendix 2.1. Diagram of transect placement at the Covered bridge site. Map is approximated and not to scale. This map should be used as a general diagram only.




Appendix 2.2. Diagram of transect placement at the EPA site. Map is approximated and not to scale. This map should be used as a general diagram only.

Appendix 2.3. Statistical analysis of variance (f-test: two sample for variances) output tables for all study sites.

| | | | | |
|---|---|---|---|---|
| | | 2000 - 2001 | 2000 - 2002 | 2001 - 2002 |
| Non-restored Mesic Meadow | F | 8.07 | 1.84 | 4.37 |
| (EPA Site) | p- value | <0.001 | 0.13 | 0.004 |
| | Year | 2000 | 2001 | 2002 |
| | Mean | -496 | -578 | -444 |
| | Maximum | -251 | -438 | -142 |
| | | 2000 - 2001 | 2000 - 2002 | 2001 - 2002 |
| Restored Mesic Meadow | F | 19.4 | 31.7 | 1.6 |
| (Covered Bridge Site) | p- value | <0.001 | <0.001 | 0.18 |
| | Year | 2000 | 2001 | 2002 |
| | Mean | -218 | -156 | -144 |
| | Maximum | -4 | 7 | 80 |
| | | 2000 - 2001 | 2000 - 2002 | 2001 - 2002 |
| Non-restored Wet Meadow | F | 11.4 | 5.71 | 2.0 |
| (EPA Site) | p- value | <0.001 | 0.001 | 0.1 |
| | Year | 2000 | 2001 | 2002 |
| | Mean | 15.5 | -12.1 | 18.2 |
| | Maximum | 184 | 143 | 254 |
| | | 2000 - 2001 | 2000 - 2002 | 2001 - 2002 |
| Restored Wet Meadow | F | 40.5 | 22.7 | 1.78 |
| (Covered Bridge Site) | p- value | <0.001 | <0.001 | 0.15 |
| | Year | 2000 | 2001 | 2002 |
| | Mean | 13.6 | 63.1 | 56.4 |
| | Maximum | 170 | 190 | 241 |

Appendix 3.0. Statistical summary for all study sites. Arrow indicates statistically significant change (up or down) and N.C. indicates no statistically significant change. Specie richness is number of species all others are % relative abundance.

| | Passively Restored Mesic Meadow (EPA Site) | Ag. Managed Mesic Meadow (EPA Site) | Passively Restored Wetland Meadow (EPA Site) |
|---|---|---|---|
| | Significance | Significance | Significance |
| Species Richness | ↓ | N.C. | N.C. |
| Native Plants % | N.C. | N.C. | N.C. |
| Wetland Indicator % | N.C. | ↑ | N.C. |
| Nuisance weeds % | ↓ | ↑ | N.C. |
| | Actual Value | Actual Value | Actual Value |
| Species Richness | 10 - 8 - 7 | 5 - 8 - 6 | 1 - 1 - 1 |
| Native Plants % | 3 - 2 - 4 | 0.25 - 1.45 - 0.23 | 0 - 0 - 0 |
| Wetland Indicator % | 64 - 56 - 58 | 47 - 56 - 56 | 100 - 100 - 100 |
| Nuisance weeds % | 15 - 2 - 1 | 10 - 14 - 13 | 100 - 100 - 100 |
| | | | |
| | Actively Restored Mesic Meadow (Covered Bridge) | Ag. Managed Mesic Meadow (Covered Bridge) | Actively Restored Wetland Meadow (Covered Bridge) |
| | Significance | Significance | Significance |
| Species Richness | N.C. | N.C. | ↑ |
| Native Plants % | ↑ | ↓ | ↓ |
| Wetland Indicator % | N.C. | ↓ | N.C. |
| Nuisance weeds % | ↓ | ↑ | ↑ |
| | Actual Value | Actual Value | Actual Value |
| Species Richness | 12 - 10 - 11 | 5 - 9 - 4 | 3 - 7 - 9 |
| Native Plants % | 4 - 22 - 20 | 29 - 16 - 9 | 92 - 92 - 66 |
| Wetland Indicator % | 55 - 69 - 54 | 42 - 64 - 22 | 96 - 99 - 98 |
| Nuisance weeds % | 22 - 10 - 15 | 10 - 17 - 11 | 4 - 5 - 26 |

Appendix 3.1. Statistical output tables of t-test: paired two-sample for means for all plant community parameters at all study sites.

| Actively Restored Wetland (Covered Bridge Site) | | 2000 - 2001 | 2000 - 2002 | 2001 - 2002 |
|---|---|---|---|---|
| Species Richness | t-statistic | -2.190 | -7.131 | -1.997 |
| | p - value | 0.058 | 0.002 | 0.069 |
| Native Plants | t-statistic | 0.087 | 2.61 | 4.241 |
| | p - value | 0.468 | 0.04 | 0.012 |
| Wetland Indicator | t-statistic | -0.782 | -0.695 | 0.856 |
| | p - value | 0.246 | 0.269 | 0.227 |
| Nuisance weeds | t-statistic | -0.508 | -4.558 | -8.989 |
| | p - value | 0.323 | 0.01 | 0.001 |
| | Year | 2000 | 2001 | 2002 |
| Species Richness | Mean | 2.5 | 6.5 | 9.25 |
| Native Plants | Mean | 92.275 | 91.775 | 66.025 |
| Wetland Indicator | Mean | 96.225 | 99.375 | 97.525 |
| Nuisance weeds | Mean | 3.575 | 5.025 | 25.95 |

| Actively Restored Mesic Meadow (Covered Bridge Site) | | 2000 - 2001 | 2000 - 2002 | 2001 - 2002 |
|---|---|---|---|---|
| Species Richness | t-statistic | 0.719 | 0.469 | -0.521 |
| | p - value | 0.252 | 0.33 | 0.313 |
| Native Plants | t-statistic | -1.138 | 0.083 | 0.263 |
| | p - value | 0.153 | -1.625 | 0.402 |
| Wetland Indicator | t-statistic | -2.0 | 0.1 | 2.3 |
| | p - value | 0.1 | 0.5 | 0.03 |
| Nuisance weeds | t-statistic | 2.047 | 1.539 | -0.905 |
| | p - value | 0.048 | 0.092 | 0.204 |
| | Year | 2000 | 2001 | 2002 |
| Species Richness | Mean | 11.833 | 9.833 | 11 |
| Native Plants | Mean | 3.617 | 22.233 | 20.134 |
| Wetland Indicator | Mean | 54.6 | 69.4 | 54.1 |
| Nuisance weeds | Mean | 22.183 | 9.6 | 15.117 |

| Ag. Managed Mesic Meadow (Covered Bridge Site) | | 2000 - 2001 | 2000 - 2002 | 2001 - 2002 |
|---|---|---|---|---|
| Species Richness | t-statistic | -3.369 | 1.074 | 3.788 |
| | p - value | 0.01 | 0.166 | 0.006 |
| Native Plants | t-statistic | 1.713 | 2.818 | 1.904 |
| | p - value | -0.074 | 0.018 | 0.057 |
| Wetland Indicator | t-statistic | -4.17 | 3.6 | 10.4 |
| | p - value | 0.004 | 0.008 | <0.001 |
| Nuisance weeds | t-statistic | -2.382 | -0.572 | 1.744 |
| | p - value | 0.031 | 0.296 | 0.071 |
| | Year | 2000 | 2001 | 2002 |
| Species Richness | Mean | 4.833 | 8.667 | 3.833 |
| Native Plants | Mean | 28.617 | 16.416 | 9.0 |
| Wetland Indicator | Mean | 41.6 | 63.9 | 21.7 |
| Nuisance weeds | Mean | 10.233 | 17.3 | 10.967 |

Appendix 3.1 (continued)

| Passively Restored Wetland (EPA Site) | | 2000 - 2001 | 2000 - 2002 | 2001 - 2002 |
|---|---|---|---|---|
| Species Richness | t-statistic | N/A | N/A | N/A |
| | p - value | N/A | N/A | N/A |
| Native Plants | t-statistic | N/A | N/A | N/A |
| | p - value | N/A | N/A | N/A |
| Wetland Indicator | t-statistic | N/A | N/A | N/A |
| | p - value | N/A | N/A | N/A |
| Nuisance weeds | t-statistic | N/A | N/A | N/A |
| | p - value | N/A | N/A | N/A |
| | | 2000 | 2001 | 2002 |
| Species Richness | Mean | 1 | 1 | 1 |
| Native Plants | Mean | 0 | 0 | 0 |
| Wetland Indicator | Mean | 100.0 | 100.0 | 100.0 |
| Nuisance weeds | Mean | 100.0 | 100.0 | 100.0 |

| Passively Restored Mesic Meadow (EPA Site) | | 2000 - 2001 | 2000 - 2002 | 2001 - 2002 |
|---|---|---|---|---|
| Species Richness | t-statistic | 2.148 | 2.318 | 0.881 |
| | p - value | 0.042 | 0.034 | 0.209 |
| Native Plants | t-statistic | 0.535 | -0.516 | -0.802 |
| | p - value | 0.308 | 0.314 | 0.229 |
| Wetland Indicator | t-statistic | 1.7 | 1.1 | -1.5 |
| | p - value | 0.08 | 0.2 | 0.1 |
| Nuisance weeds | t-statistic | 4.307 | 4.278 | 0.674 |
| | p - value | 0.004 | 0.004 | 0.265 |
| | | 2000 | 2001 | 2002 |
| Species Richness | Mean | 10.333 | 8.333 | 7.4 |
| Native Plants | Mean | 2.533 | 2.167 | 3.483 |
| Wetland Indicator | Mean | 64.0 | 55.5 | 58.2 |
| Nuisance weeds | Mean | 14.65 | 1.883 | 1.367 |

| Ag. Managed Mesic Meadow (EPA Site) | | 2000 - 2001 | 2000 - 2002 | 2001 - 2002 |
|---|---|---|---|---|
| Species Richness | t-statistic | -2.939 | -1.234 | 3.162 |
| | p - value | 0.016 | 0.136 | 0.013 |
| Native Plants | t-statistic | -1.579 | 0.068 | 2.261 |
| | p - value | 0.088 | 0.474 | 0.037 |
| Wetland Indicator | t-statistic | -2.4 | -2.8 | 0.1 |
| | p - value | 0.03 | 0.02 | 0.5 |
| Nuisance weeds | t-statistic | -1.93 | -1.516 | 0.784 |
| | p - value | 0.056 | 0.095 | 0.234 |
| | | 2000 | 2001 | 2002 |
| Species Richness | Mean | 5.167 | 8.333 | 6.333 |
| Native Plants | Mean | 0.25 | 1.45 | 0.233 |
| Wetland Indicator | Mean | 46.5 | 56.1 | 55.9 |
| Nuisance weeds | Mean | 10.083 | 14.383 | 12.733 |