AN ABSTRACT OF THE THESIS OF

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Abstract approved:

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Bruce D. Dugger

Abstract

The Wetland Reserve Program (WRP) is one conservation tool that mitigates national wetland loss and a primary goal for the program includes optimizing wildlife habitat by restoring wetland functions and values. Few studies have evaluated the WRP, which limits our understanding of its impact on wildlife populations. I assessed the biological value of seasonal wetlands on WRP easements in the Willamette Valley (WV) and Lower Columbia River Valley (LCRV) of western Oregon and southwest Washington by comparing plant communities and waterfowl food production (i.e. seed biomass) to publicly managed wetlands that served as reference conditions. Estimating waterfowl food resources is used to evaluate habitat quality, restoration, management, and developing habitat specific conservation plans for wintering waterfowl. The most common technique to estimate the portion of seed in wetland soils is to collect and process soil cores, yet there is variability in the depth of cores collected because it is unknown how deep dabbling ducks can extract food from the substrate. Furthermore, recent food abundance studies have not partitioned food estimates by location within wetlands (i.e. above or below ground), which has implications for understanding food availability for ducks. I sampled 23 wetlands for each wetland type in fall 2008 and 2009. I determined the proportion of seeds above versus below ground and the vertical distribution of seed biomass within the soil profile by partitioning samples in 26 wetlands in 2008. There was significantly more seed biomass above ground (72%, 362 ± 50.8 kg/ha) than below (28%, 141 ± 18.5 kg/ha). Seed biomass also varied by soil depth layer with greater biomass in 0-2 cm than both 2-5 and 5-10 cm layers. The majority of seed biomass within the soil was in was in the top 5 cm of the soil profile (75%). I detected 113 plant genera/species and the total mean percent cover of all plant species was 49% native, 38% introduced, 13% bare ground, and 3% unknown. Plant communities differed between study regions but not by wetland type. Twenty-nine species differed by region with more annuals being indicative of the WV and perennial species in the LCRV. Overall, the mean seed biomass estimate was 505 ± 59 kg/ha. Seed biomass was similar between wetland type and study region; however, WRP wetlands in the WV produced more seed (560 ± 114 kg/ha) than those in the LCRV (188 ± 43 kg/ha). Lower seed production in the LCRV WRP sites was attributed to a dominance of perennial species, predominantly *Phalaris arundinacea* that produced low seed yields relative to annual plants. My results indicate most seed was near the soil surface with only 36 kg/ha in the 5-10 cm layer, providing evidence that most seed is available to foraging dabbling ducks. Reference sites and WV WRP seasonal wetlands produced seed biomass that was similar to managed wetlands in other major waterfowl wintering areas in the United States. Therefore, WRP is capable of mitigating wetland loss and contributing to the regional goals of the North American Waterfowl Management Plan. However, WRP and

reference seasonal wetlands are only providing 10% and 3.5% of dabbling ducks energetic demands in the WV and LCRV respectively due to a lack of habitat. Depending on whether the current landscape is meeting current dabbling duck energetic demands, this suggests either a need for more habitat, which is most feasible through restoration on private lands, or protecting other existing wetland habitats.

Assessing Biological Values of Wetland Reserve Program Wetlands for Wintering Waterfowl

by

Graham Raymond Evans-Peters

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I understand that my thesis will become part of the permanent collection of Oregon State University libraries. My signature below authorizes release of my thesis to any reader upon request.

Graham Raymond Evans-Peters, Author

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CONTRIBUTION OF AUTHORS

Dr. Bruce Dugger assisted with all aspects of this study, therefore his input and recommendations are tremendously appreciated. Dr. Mark Petrie assisted in bioenergetic model simulations and interpretation of the data.

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Chapter 1

GENERAL INTRODUCTION

More than half of the wetlands in the contiguous US have been lost because of conversion to agriculture or urban uses (Dahl 2006). Historically there were 243,000 ha of wetlands in Oregon's Willamette Valley (WV; Taft and Haig 2003). Since European settlement, approximately 99% of wetland prairie and 58% of emergent wetlands have been converted to agriculture and urban uses (Titus et al. 1996), and losses are likely to continue as the human population grows from two million people in 1990 to a projected four million people in 2050 (Baker et al. 2004). Additionally, the spread of invasive flora and fauna such as reed-canary grass (*Phalarus arundicacea*), purple loosestrife (*Lythrum salicaria*), nutria (*Myocastor coypus*), carp (*Cyprinus carpio*), and bullfrogs (*Rana catesbeiana*) continue to threaten and degrade remaining wetland habitats and restoration efforts (Roth et al. 2004).

Wetland losses impact wetland-dependent species including many species of waterfowl. Wetlands of the WV and Lower Columbia River Valley (LCRV) provide critical habitat for greater than 750,000 dabbling ducks during mid-winter (USFWS unpublished data) and a significant portion of the Pacific Flyway's 5.7 million wintering dabbling ducks use the WV and LCRV regions during migration (Collins and Trost 2009). The principle function of winter habitat for waterfowl is to provide food to meet daily energy demands (Cain 1988) and fuel important life history events such as molt, pairing, thermoregulation, and subsequent reproductive effort (Heitmeyer and

Fredrickson 1981, Krapu 1981, Heitmeyer 1988, Raveling and Heitmeyer 1989, Smith and Sheeley 1993, Heitmeyer 2006). Wetland loss has disproportionately impacted dabbling ducks like mallard (*Anas platyrhynchos),* northern pintail (*Anas acuta*) and green-winged teal (*Anas crecca*) because wetland conversion to agriculture (e.g., grass seed production) in western Oregon and Washington has provided food for geese and grazing ducks like American wigeon (*Anas americana*).

In response to increased appreciation of wetland functions and values, the federal government has implemented several policies to protect and restore wetlands (Rewa 2000, 2005, Dahl 2006). The Wetland Reserve Program (WRP), established by the 1990 Farm Bill, is a volunteer-based federal program designed to provide subsidies and technical assistance to private landowners for wetland restoration activities. The WRP has restored more than 769,000 hectares of wetlands in the United States including 328 projects totaling 30,000 hectares in Oregon and Washington. The WRP is continuing to grow as the 2008 farm bill increased the national acreage cap to over 1.2 million hectares, a 34% increase (NRCS 2010). However, despite the importance of the WRP to wetland restoration and its potential importance to increasing waterfowl habitat in winter, there have been few post-restoration evaluations of the biological functioning of wetlands created using the WRP program. The plant community is relatively simple to measure, provides a good indicator of biological value (Robach et al. 1996), and reflects other properties that define wetlands like hydrology and soils. If the WRP is meeting its primary goal of maximizing wetland functions and values (NRCS 2010) then it should be apparent in the plant community.

One goal of the WRP is to optimize wildlife habitat, especially for migratory birds (NRCS 2010), which corresponds with goals of the North American Wetlands Conservation Act and North American Waterfowl Management Plan (USFWS 2010). My thesis focuses on assessing the biological value of wetlands created using the WRP, in particular, assessing their value to waterfowl during the non breeding season. Food availability is one key factor limiting waterfowl populations during migration and winter (Miller 1986, Conroy et al. 1989, Reinecke et al. 1989), therefore estimating food production is an applicable metric for estimating biological function of WRP wetlands for wintering waterfowl.

Waterfowl conservation planners use bioenergetic models to estimate the amount of foraging habitat needed to meet regional waterfowl population goals. Estimates of habitat-specific area and food abundance are required to populate these models. Initially, estimates of food abundance were obtained from wetlands in the Midwest (Fredrickson and Taylor 1982) and extrapolated to other regions (Naylor 2002). However, in an effort to strengthen the biological foundation of regional bioenergetic models (NAWMP Plan Committee 1998, NAWMP Steering Committee 2007), recent research has quantified food abundance within various habitat types in several major waterfowl wintering areas (Stafford et al. 2006, Brasher 2007, Greer et al. 2007, Dugger et al. 2008, Kross et al. 2008, Olmstead 2010). That work indicates there is considerable variation in food abundance among regions, and the unique physical properties and distinct plant communities of the WV and LCRV warrant local sampling.

In Chapter 2, I compare food production and plant community composition between WRP wetlands and reference wetlands located on National Wildlife Refuges and

State Management Areas. Few, if any, unaltered wetlands exist in my study area that could serve as historic reference conditions, so I have chosen the relatively high quality wetlands on managed public lands as a benchmark to evaluate success (Brinson and Rheinhardt 1996). Many seasonal wetlands on public lands are actively managed to maintain early successional plant communities that provide the most forage for wintering waterfowl, mitigating wetland losses, while others are managed with a more passive approach. Therefore, if seasonal wetland habitats on WRP easements are actively managed for early seral plant communities their biological value should increase and be similar to actively managed reference sites. I categorized each wetland as unmanaged, passively managed, or actively managed to determine if increased management actions produce different plant communities and higher seed production.

Chapter 3 addresses a methodological issue associated with research and sampling designed to estimate food abundance for wintering waterfowl. Estimates of food abundance and subsequent conservation planning are predicated on food resources being available to waterfowl. Seeds can occur in several places within a wetland including on inflorescences, floating on the surface of the water, sitting on the soil or buried in the sediment. It is reasonable to assume that all seed above ground can be available to foraging ducks, but is not reasonable to make that assumption for seed located in the soil. The magnitude of error associated with making the latter assumption depends on the proportion of seeds in the soil relative to total biomass estimates. Although several recent studies have estimated food production in seasonal wetlands (Brasher et al. 2007, Greer et al. 2007, Johnson 2007, Dugger et al. 2008, Kross et al. 2008, Straub 2008, McWilliams 2010), the distribution of food within wetlands is seldom reported. I partitioned biomass estimates by location as aboveground, on the soil, or within the soil, as well as seed distribution by depth layer in the soil profile to determine the potential error associated with assuming that all seed biomass in the soil column is available as food to ducks. Additionally, I tested the possibility of using regression models to determine if seed biomass in upper soil layers could predict seed biomass at lower depths.

Chapter 4 synthesizes how information from chapters 2 and 3 is applicable to regional conservation planning for wintering waterfowl and estimating WRP wetlands relative contribution towards meeting energetic demands of dabbling ducks. I will quantify the contribution of WRP wetlands to meeting waterfowl conservation goals by estimating waterfowl energy needs being provided by WRP wetlands in western Oregon and Washington. Assessing management practices applied in the region and comparing different levels of intensity will identify if management is achieving goals of providing quality foraging habitat for migratory birds.

STUDY AREA

The study was conducted in WRP and publicly managed seasonal wetlands within the WV and LCRV of western Oregon and southwest Washington (Figure 1). The WV includes all or portions of Clackamas, Washington, Yamhill, Polk, Marion, Linn, Benton, and Lane Counties. The LCRV includes portions of Clatsop, Columbia, and Multnomah Counties in Oregon and Washington’s Skamania, Clark, Cowlitz, Wahkiakum, and Pacific Counties from the mouth of the Columbia River to Bonneville Dam, 235 km upstream. The WV is a 9,100 km^2 area of alluvial plain situated between the Coast Range on the west and Cascade Mountains to the east. The WV varies from approximately 20 to 60 km wide and is 290 km long, spanning from Portland in the north to Eugene in the south (Benner 1997). Elevation ranges from 3 m above sea level near the confluence of the Columbia River in the north to approximately 137 m at the south end of the valley floor (Hulse et al. 2002). The prominent hydrologic feature of the WV is the northerly flowing Willamette River, its associated floodplain, and its 13 major tributaries that drain 29,000 km^2 (Gregory et. al 1998, Roth et al. 2004). The climate of the region is cool Mediterranean with warm dry summers and cool wet winters. Average annual rainfall is 100-125 cm, 75% of that falling between October and March.

Over 96% of lands in the WV are privately owned (ODFW 2006), which increases the importance of conservation programs that protect, enhance, or restore wetland habitats on private land. Currently, remaining wetlands include small urban remnant wetlands, a growing number of conservation easements, state and federal wildlife refuges, and hundreds of small scattered privately owned agricultural wetlands (Taft and Haig 2006). Of the 9,100 km^2 in the WV only 72 km^2 are habitats that contain seasonal wetlands managed by Oregon Department of Fish and Wildlife and U.S. Fish and Wildlife Service (USFWS) including Fern Ridge Wildlife Management Area (2,130 ha) and the Willamette Valley National Wildlife Refuge Complex (NWR; 5,070 ha). Washington Department of Fish and Wildlife, ODFW and USFWS manage 77 km^2 containing seasonal wetlands in the LCRV, including Sauvie Island Wildlife Area (4,671 ha), Shilapoo Wildlife Area (959 ha) and Ridgefield NWR (2,084 ha). The size of wetlands that I sampled ranged from 1.4 ha to 33.6 ha ($x = 8.8 \pm 1.06$ ha), being slightly larger on average on public sites (11.0 ± 1.75 ha) than WRP easements (6.6 ± 1.07 ha). WRP easements sampled were between 21 and 316 ha in size ($x = 91 \pm 15.6$ ha) and contained 1 to 5 seasonal wetlands.

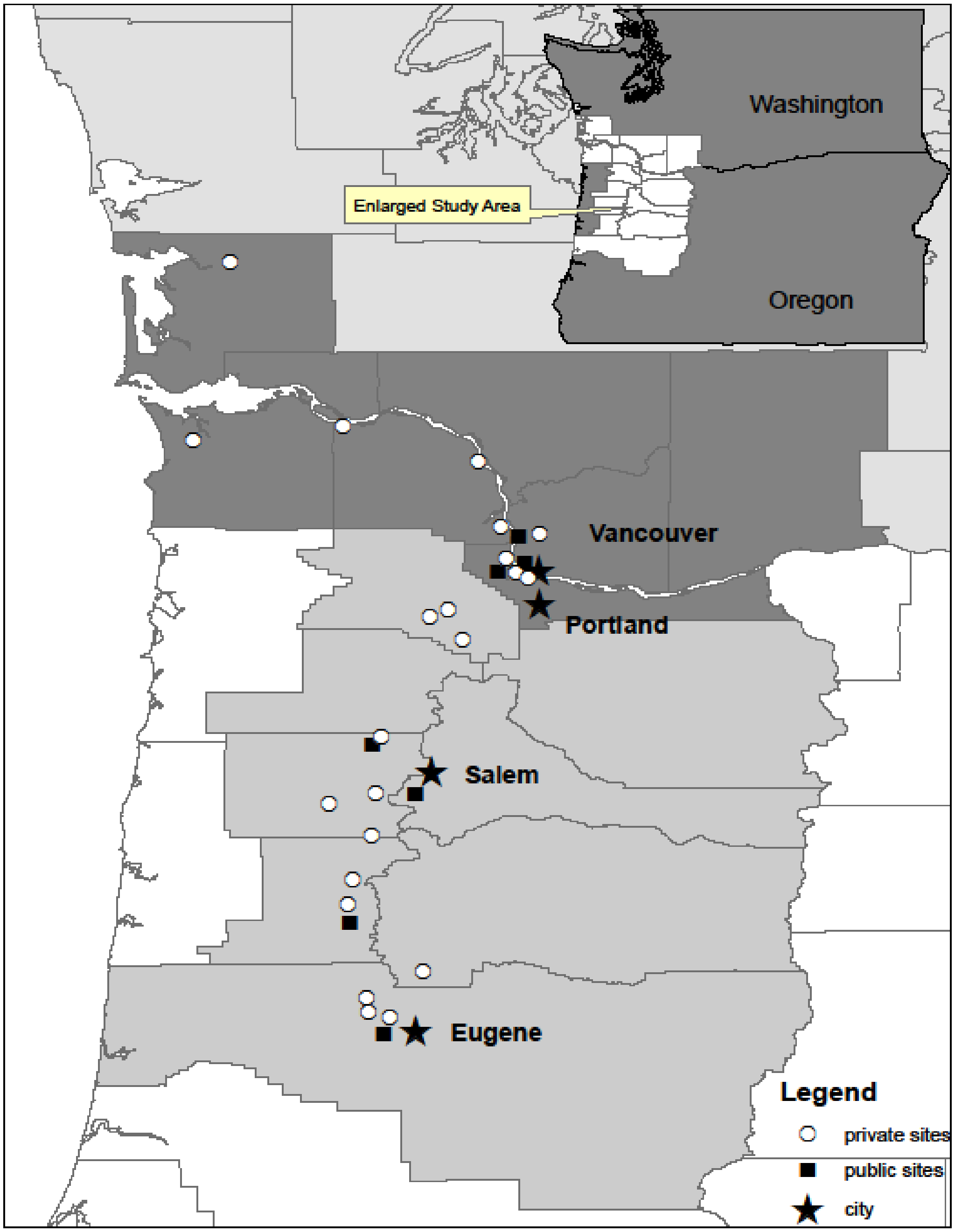

Figure 1.1 Location of WRP easements (white circles) and reference sites (black squares) in the Willamette Valley and Lower Columbia River Valley of western Oregon and southwest Washington where I sampled seasonal wetlands in fall 2008 and 2009.

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Chapter 2

ASSESSING PLANT COMMUNITY CHARACTERISTICS AND FOOD ABUNDANCE FOR WINTERING WATERFOWL IN WETLANDS CREATED BY THE WETLAND RESERVE PROGRAM

INTRODUCTION

Since 1780 over 50% of wetlands in the United States have been lost, primarily because of conversion to agriculture and development (Dahl 2006, Brinson et al. 2002). Regionally, losses can be more severe. For example, less than 1% of wet-prairie habitat remains in Oregon's Willamette Valley (WV) and freshwater tidal marshes in the Lower Columbia River Valley (LCRV) in Oregon and Washington have declined by an estimated 64% since European settlement in the Pacific Northwest (Hulse et al. 2002, Johnson et al. 2003). Additionally, 75% of remaining wetlands nationally and 99% of all lands in the WV are privately owned (NRC 1995, ODFW 2006), which increases vulnerability to loss or degradation.

Conservation easement programs directed at restoring wetlands on private land are one way to reverse the trend of wetland loss. Among the most influential of these programs is the Wetland Reserve Program (WRP) administered by the Natural Resource Conservation Service (NRCS). The primary goal of the WRP is to maximize wetland functions and values while optimizing wildlife habitat on private lands. This occurs by providing technical and financial support to protect, restore, and enhance wetlands via easements on lands retired from agricultural production (Frazier and Galat 2009). The WRP has enrolled more than 769,000 hectares of conservation easements in the U.S. including 328 projects totaling 30,000 hectares in Oregon and Washington (NRCS 2010).

The WRP is considered a success (Rewa 2000, Frazier and Galat 2009); however, few restoration projects have been monitored to evaluate biological function, which has limited our ability to understand how the WRP has impacted wildlife populations (King et al. 2006). Studies have inventoried land cover changes, wetland types restored, plant species present, and wildlife benefits (King et al. 2006, Frazier and Galat 2009) and two studies compared waterfowl abundance and plant community indices among WRP wetlands subjected to different management intensities (Kaminski et al. 2006, Fleming 2010). However, only one study has compared WRP wetlands to some desired state, or reference wetland (Brinson and Rheinhardt 1996, Anderson and Dugger 1998). In that work conducted in southern Illinois, Hicks (2003) found that hydrology, vegetation and waterbird abundance did not differ between WRP and reference wetlands. Additional work covering a broader range of biological metrics in different geographic areas is needed.

The goals of the WRP parallel the North America Wetlands Conservation Act and North American Waterfowl Management Plan (NAWMP), which support long term conservation of waterfowl and their associated wetland habitats (USFWS 2010). Consequently, one way to measure WRP restoration success would be to select metrics that reflect wetland value for waterfowl. Conservation planning for waterfowl during winter assumes food is limited, thus measuring food production for dabbling ducks (e.g. seeds) and the wetland plant communities that produce food are appropriate metrics to evaluate the value of WRP wetlands to waterfowl. Estimating food abundance in WRP and reference wetlands will provide a quantitative measure of regional wetlands contribution towards meeting regional waterfowl population objectives. Appraisal of

WRP and reference plant communities will allow comparison to some desired state and should reflect hydrology and management regimes.

In this chapter I evaluate the success of WRP wetland restoration by comparing plant community composition and food production between seasonally flooded wetlands created by the WRP and reference wetlands. Because previous work has shown that actively managed wetlands produce more food than non-managed or passively managed wetlands (Brasher et al. 2007, Kross et al. 2008, Fleming 2010, Olmstead 2010), I also compared how management intensity influenced the comparison. I used non-parametric community analyses to compare wetland types with the intent of providing a finer resolution to potential differences between plant communities by incorporating abundance estimates of all species. In addition to estimating waterfowl food resource production as an assessment of WRP wetlands, the seed biomass metric allows for a carrying capacity analysis of regional wetlands for wintering dabbling ducks.

METHODS

Sampling Design

I defined my WRP study population as all non-tidal, seasonal, fresh water wetlands located on privately owned WRP easements within my study area. There were 58 WRP easements in the WV and 10 easements in the LCRV that could have met these criteria. However, the habitat composition of each WRP easement could not be confirmed without a site visit. Thus, in 2008 I randomly selected 14 easements to visit that met the following criteria: 1) the site was ≥ 20 hectares; 2) the site had been enrolled for ≥ 3 years, and 3) the site's landowners were willing to grant access. If a selected easement did not contain a seasonal wetland, seasonal wetlands were undergoing active restoration, or permission was denied to sample the site by the landowner, I selected another property. After selecting 14 sites that met my criteria, I randomly selected one seasonal wetland on each WRP easement to sample (n = 14) and interviewed the landowner to obtain background information regarding management of that wetland. In 2009, I selected another 10 wetlands from the population of easements not sampled in 2008.

I used seasonal wetlands that occurred on state Wildlife Management Areas (WMA) and federal National Wildlife Refuges (NWR) as benchmarks to evaluate WRP restoration success (Brinson and Rheinhardt 1996; hereafter referred to as reference wetlands). Publicly owned lands included four NWRs and three WMAs, with four in the WV and three in the LCRV. I met with managers at each site to develop a complete list of wetlands that could be sampled. Wetlands experiencing active disturbance during the sampling season such as disking, mowing, or prolonged inundation were omitted from those eligible to be sampled, therefore the population of wetlands changed slightly between years. In 2008, I randomly selected two wetlands from each public site (n = 14). In 2009, I randomly selected one wetland from each public site that had not been sampled in 2008 (n = 7). Then I selected one wetland from the total population in the WV and two from the LCRV population to obtain a balanced sample size between regions (n = 10).

Based on information from public area managers, NRCS district conservationists, and landowners, I categorized each wetland as unmanaged, passively managed, or actively managed. I defined wetlands as unmanaged if no restoration, enhancement, or water control infrastructure existed. Passively managed wetlands were defined as those that had water control but received no active management in at least three years; whereas, wetlands that had been managed more recently were categorized as actively managed (Brasher et al. 2007, Kross et al. 2008, Fleming 2010). A wetland was defined as receiving management disturbance in a given year if more than half the wetland area was treated with a management practice such as mowing, herbicide application, disking, crop rotations, and/or a drawdown to suppress undesired species and promote early successional habitat (Kross et al. 2008).

My sampling scheme was designed to assure representative sampling for each wetland. Prior to sampling, I used physical cues to walk the circumference of each wetland and used a handheld Global Positioning System (GPS) to record my path. I imported the GPS data into a Geographic Information System ([GIS] ESRI ArcGIS version 9.0, 2004) and overlaid points on 0.5 m or 1.0 m resolution digital aerial photographs. Finally, I created a polygon of the wetland, partitioned it into a grid of approximately 30-40 cells, and randomly generated one point within each cell for sampling. When a polygon contained more than 30 cells, I used a random number generator to select 30 cells to sample in 2008 and 20 cells to sample in 2009. I uploaded sampling points in a GPS unit to locate each sample location within a wetland.

<u>Sampling Procedures</u>

I sampled one wetland per day from 25 August 2008 to 1 October 2008 and 28 August 2009 to 24 September 2009. Sampling occurred after seeds had matured, but prior to the initiation of fall flooding. Dabbling ducks foraged on fruits, seeds, or both from plants occurring in seasonal wetlands and are referred to synonymously as seeds. I began sampling in the southern WV and proceeded north alternating between WRP and reference wetlands to control for local variation in plant phenology. I characterized the plant community at each sampling location by estimating percent cover of all species and bare ground present within a 1 m^2 sampling frame. I clipped all inflorescences within a 0.0625 m^2 frame (25 x 25 cm) placed in the middle of the original 1 m^2 frame (Greer et al. 2007). Next, I trimmed all stems to ground level and used a gasoline-powered vacuum to collect all seeds on the substrate (Penny et al. 2006) within an 11 cm diameter sampling frame placed in the middle of the inflorescence frame. Finally, I collected a 10 cm diameter x 5 cm deep soil core within the 11 cm vacuum frame. Core samples were subsequently frozen until processing occurred to prevent seed deterioration.

Sample Processing

In the lab, I processed 20 of 30 seed biomass samples per wetland in 2008 and all samples in 2009. I processed seeds from each sample gear (inflorescence, vacuum, soil core) separately. Inflorescence and vacuum samples were air dried for 2-6 weeks before processing began (Laubhan and Fredrickson 1992). I threshed all seeds from chaff, residual vegetation, and debris using a series of graduated sieves (mesh sizes 2 mm, 1 mm, 500 μm, 355 μm, and 250 μm), manual gravity, and air separation techniques (Harmond et al. 1968), and hand removed seeds with forceps for above ground sampling techniques. Core samples were thawed, rinsed through screens, dried, and processed as stated above (Greer et al. 2007). All seeds were dried at 60ºC to constant mass and weighed to the nearest 0.0001 g.

Statistical Analysis

Plant Community

I calculated alpha diversity (number of plant species per wetland), Simpson's index (the likelihood that two randomly chosen individuals will be different species; Simpson

1949), beta diversity (amount of compositional variation among wetlands), and gamma diversity (total number of species in a group of wetlands; Whittaker 1972) by wetland type, region, and management intensity to generally characterize the plant community. I then prepared the plant data for additional analyses by calculating mean percent cover estimates of all plant species for each wetland.

The community covariates I used to identify relationships with plant species composition and sample units (wetlands) included wetland type (WRP vs. reference), study region, management intensity, mean wetland seed biomass, and bare ground abundance. I included management intensity as an explanatory variable because it has been shown to increase seed production (Naylor 2002, Kross et al. 2008) and I desired an estimate of its effect on the wetland plant community. I included seed biomass estimates because they are one measure of wetland quality from a wildlife perspective. Bare ground was measured as percent cover and served as a quantitative measure for disturbance level or successional stage of the plant community in a given wetland. I hypothesized that plant community composition would differ between WRP and reference wetlands due to the wider range of management activities among WRP easements and food abundance would be lower in WRP wetlands compared to reference wetlands because wetlands may not be as intensively managed.

I used multi-response permutation procedures (MRPP; Mielke 1984) (PC-ORD v. 6; McCune and Mefford 2009) to test the null hypothesis of no difference in wetland plant community composition between wetland type (WRP [n = 23], reference wetlands [n = 23]) and study region (WV [n = 26], LCRV [n = 20]). Unlike parametric ANOVA, MRPP only tests for group differences on a single variable. Consequently, I tested how plant communities compared between wetland types within each study region and how management intensity influenced plant community composition by conducting an MRPP analysis separately for each wetland type.

The initial species matrix consisted of 46 sample units (individual wetlands) by mean abundances for 113 plant species. All wetlands were within the population of interest and were retained for analyses. Data in the species matrix were square root transformed to reduce skewness and improve homogeneity of variance of species abundances. A Euclidean distance measure was used because absolute differences in plant communities and the relationship with food abundance were primary interests. No relativizations of row or column totals were required since the objective was to compare absolute abundance patterns of plant species between groups of wetlands.

I conducted indicator species analyses (ISA; Dufrêne and Legendre 1997) to identify individual wetland plant species that were characteristic of a group when MRPP tests indicated a difference in plant communities between groups. ISA compared indicator values for each species, exclusiveness and faithfulness to a group, to random indicator values calculated from 4,999 Monte Carlo simulations. P-values for indicator values were calculated as the proportion of randomized trials with an indicator value equal to or exceeding the observed indicator value. I considered p-values less than 0.1 biologically significant because of the relatively small sample size.

An ordination of sample units (individual wetlands) in species space was performed using non-metric multidimensional scaling (NMS; Mather 1976) in PC-ORD (v. 6; McCune and Mefford 2009) to examine correlations between wetlands, species abundance and community covariates. Species abundance data were $\log_{10}(x+1)$

transformed to further reduce skewness and subdue influence of dominant species. Seed biomass estimates were also $\log_{10}(x+1)$ transformed to satisfy the constant variance assumption. Euclidean distance was used for consistency and interest in absolute species abundances in relation to environmental variables. I used a random starting configuration in the slow and thorough autopilot option, which initiates 250 runs with real data and 250 Monte Carlo randomizations.

Seed Biomass

I standardized the seed biomass value for each sampling gear to kg/ha and summed values across sampling gear to generate a single biomass value for each sample location; I then conducted the analysis of seed biomass in two steps. First, I used a mixed effects model (PROC MIXED, SAS version 9.2; SAS Institute 2004) to test for differences in seed biomass by wetland type, study region, their interaction, and management intensity (Littell et al. 1996). After confirming normality by inspecting box and residual plots, raw data was used for subsequent analyses (Ramsey and Schafer 2002). I treated wetland type and region as fixed effects and management intensity as a random effect because it was not considered prior to wetland selection and I was initially interested in controlling for this variable while I looked for a wetland type effect. I conducted pairwise multiple comparisons using a Tukey-Kramer procedure and least squares means, which accounted for an unbalanced design, to test for differences between pairs of estimates. I report arithmetic means when displaying differences between groups. All means are reported ± SE.

In the second step I investigated the influence of management intensity on seed production. There were no reference wetlands that were unmanaged; therefore I limited my comparison to passively versus actively managed wetlands. Active management of seasonal wetlands on publicly owned sites like NWRs is known to increase seed yield (Kross et al. 2008); my question was does active management of WRP wetlands produce increased seed yields similar to reference wetlands? I fitted a general linear model with seed biomass as the dependent variable and management intensity, wetland type, and their interaction as the explanatory variables (PROC GLM). I hypothesized that if there was a wider array of management techniques used and a more inconsistent application of management actions on WRP wetlands then active management of WRP sites would result in lower seed production than reference sites (i.e., the interaction term would be significant).

RESULTS

Plant Community

I sampled 46 of the 48 wetlands. One reference wetland was not sampled because the unit was flooded before I could collect samples and one WRP easement was not sampled because of unforeseen restoration actions on the only seasonal wetland on the property. I detected 113 plant genera/species, 106 in WRP wetlands and 76 in reference wetlands. Mean species richness per wetland unit was 18 ± 1.0 and beta diversity was 5.3. Alpha, Beta, and Gamma diversity was higher in WRP than reference wetlands (Table 2.1). Unmanaged wetlands had considerably lower estimates for all diversity measures and gamma diversity was nearly half of managed wetlands (Table 2.1). Native species accounted for 52% of all species observed, followed by introduced species (43%), and unknown (4%). Unknown plants were those only identified to genus level and that genus contained both native and introduced species. Total mean percent cover of all plant species was 49% native, 38% introduced, 13% bare ground, and 3% unknown (Figure 2.1). The five most common native species ($\bar{x}$% cover) were creeping spike-rush (*Eleocharis palustris,* 14.1 ± 2.82), American water-plantain (*Alisma plantago-aquatica,* 6.4 ± 1.61), waterpepper (*Polygonum hydropiperoides,* 4.9 ± 1.49), false loosestrife (*Ludwigia palustris*, 3.8 ± 1.01), and knotgrass (*Paspalum distichum*, 2.4 ± 1.24). The five most common introduced species ($\bar{x}$% cover) were reed canary-grass (*Phalaris* arundinacea, 24.3 ± 4.09), barnyard grass (*Echinochloa crus-galli*, 3.7 ± 1.35), pennyroyal (*Mentha pulegium*, 2.4 ± 0.70), spatula-leaf loosestrife (*Lythrum portula*, 1.7 ± 0.68), and curlytop knotweed (*Polygonum lapathifolium,* 1.35 ± 0.57; Appendix II). Perennial plant species were more common in the LCRV than the WV (Figure 2.2).

Plant communities differed between study regions (MRPP; A = 0.074, $P < 0.001$) but not between WRP and reference wetlands (MRPP; A = 0.003, $P = 0.21$). Bootstrap resampling failed to reject the null hypothesis of no difference in plant communities between wetland types (98.8% of 1000 runs). Twenty-nine species differed by region, with 21 indicative of the WV and eight of the LCRV. Plant communities did not differ between passively and actively managed wetlands on reference sites (A = 0.007, $P = 0.212$); however, plant communities did differ by management intensity on WRP sites (A = 0.111, $P = 0.002$). Fourteen species were significant indicators of management intensity for WRP wetlands (Table 2.2). Annual species were exclusive to more intense management, whereas perennial species found in later seral stages were indicative of unmanaged wetlands.

The relationships between species abundances, sample units and community covariates were best represented in a 3-dimensional solution (NMS; final stress = 14.5, 81% of overall variance explained by 3 axes). The ordination ran 85 iterations to converge on a stable solution (final instability = 0.0000) and Monte Carlo simulations confirmed a similar final stress could not have been obtained by chance ($p = 0.008$). Axis 2 and 3 represented the largest proportion of variance with 34% respectively, followed by axis 1 (14%). Axis 1 was best represented by region (-0.639), supporting the MRPP analysis that identified a regional difference in plant communities (Table 2.3).

A NMS plot suggested separation between wetlands grouped by management intensity and axis 2 was best explained by a gradient associated with management intensity (Figure 2.3; no mgmt $r = -.299$, mgmt $r = .331$). The gradient in species composition reflected this with early successional annual species exhibiting positive correlation with axis 2 and later successional perennial species having negative correlation (Table 2.3). Axis 3 appeared to be positively associated with abundance of bare ground and seed biomass; therefore axis 2 and 3 may be interpreted as a gradient of early succession to later seral stages. The gradient of axis 3 was strongly related to biomass estimates ($r = .552$) and species with high biomass estimates also had strong correlations with axis 3 (Table 2.3). There was no indication that plant communities varied between wetland types.

Seed Biomass

I processed 20 biomass samples per wetland, except for seven wetlands in 2008 for which I only collected ten samples. Therefore, my sample for seed biomass estimates included 39 wetlands with 20 samples per wetland and seven wetlands with ten (n = 850). The mean seed biomass estimate was 505 ± 59 kg/ha, but the range was considerable (61 to 1,566 kg/ha; Appendix III). Species that contributed the largest percentage to biomass estimates ($\bar{x}$% biomass) were American water-plantain (28.8 ± 0.62), creeping spike-rush (10.2 ± 0.57), barnyard grass (9.8 ± 0.98), ovate spike-rush (*Eleocharis ovata*, 7.0 ± 0.88), and spatula-leaf loosestrife (5.7 ± 0.89; Figure 2.4; Appendix IV).

Seed biomass estimates differed among wetland types by region ($F_{1,40} = 9.40$, $P = 0.01$). There was suggestive evidence that WV WRP wetlands (560 ± 114 kg/ha) yielded higher seed biomass estimates than WV reference wetlands (473 ± 112 kg/ha; $P > 0.05$). The trend was opposite in the LCRV; evidence suggested reference wetlands (619 ± 132 kg/ha) produced higher seed biomass than WRP wetlands (188 ± 43 kg/ha; $P = 0.03$). Seed biomass differed between management intensities ($F_{3,35} = 8.04$, $P = 0.01$), but the relationship between seed biomass and management intensity did not differ by wetland type ($F_{3,35} = 0.18$, $P = 0.68$). Mean seed biomass estimates for passively managed wetlands was 408 ± 62 kg/ha and 762 ± 116 kg/ha for active.

Table 2.1. Species diversity indices for plant communities in seasonal wetlands located in the Willamette Valley and Lower Columbia River Valley summarized by wetland type (WRP, reference), study region (Willamette Valley, Lower Columbia River Valley), and management intensity category (active, passive, unmanaged). Alpha, Simpson's Index, Beta, and Gamma diversity are defined in the text.

		Species Diversity Measure			
Plant Community	*n*	Alpha (± SE)	Simpson's Index	Beta	Gamma
Wetland Type					
Reference	23	16.7 ± 1.2	0.86	4.56	76
WRP	23	19.3 ± 1.6	0.74	5.49	106
Region					
WV	26	21.2 ± 1.2	0.90	4.76	101
LCRV	20	13.8 ± 1.4	0.82	4.78	66
Management Intensity					
Unmanaged	7	13.6 ± 1.9	0.62	2.38	46
Passive	23	19.2 ± 1.6	0.74	4.16	99
Active	16	18.1 ± 1.6	0.73	2.98	72
Total	46	18.0 ± 1.0	0.87	5.28	113

Table 2.2. Results of indicator species analysis that identified plant species associated with differences in plant community composition between wetlands in different study regions and among wetlands with different management intensities. Results are for seasonal wetlands in the Willamette Valley (WV) and Lower Columbia River Valley (LCRV) of western Oregon and southwest Washington. Management intensity results include wetland reserve program wetlands only. I considered $\alpha \leq 0.10$ significant. Annual species (A) were indicative of actively managed wetlands and Willamette Valley wetlands, while perennial species (P) were associated with passively managed, unmanaged, and Lower Columbia River Valley wetlands.

Community Covariate	Group	Species	Life History	Indicator Value	*P*
Management Intensity[a]	Active	*Echinochloa crus-galli*	A	46.5	0.029
	Active	bare ground	N/A	48.7	0.088
	Active	*Eleocharis ovata*	A	50.1	0.014
	Active	*Gnaphalium palustre*	A	43.9	0.045
	Active	*Myosotis laxa*	Both	43.8	0.074
	Active	*Epilobium densiflorum*	A	63.6	0.023
	Active	*Rorippa curvisiliqua*	A	51.7	0.067
	Active	*Polygonum lapathifolum*	A	46.4	0.017
	Active	*Juncus bufonius*	A	56.0	0.022
	Passive	*Eleocharis palustris*	P	43.4	0.075
	Passive	*Typha latifolia*	P	44.6	0.090
	Unmanaged	*Phalaris arundinacea*	P	50.2	0.007
	Unmanaged	*Carex densa*	P	27.2	0.013
	Unmanaged	*Juncus effusus*	P	47.5	0.002
	Unmanaged	*Carex obnupta*	P	24.4	0.040
	Unmanaged	*Scirpus microcarpus*	P	28.1	0.011
	Unmanaged	*Ranunculus repens*	P	38.8	0.003
Region	WV	bare ground	N/A	57.6	0.009
	WV	*Lythrum portula*	A	36.9	0.021
	WV	*Ludwigia palustris*	P	56.3	0.014
	WV	*Mentha pulegium*	P	66.4	0.001
	WV	*Echinochloa crus-galli*	A	48.9	0.033
	WV	*Eleocharis palustris*	P	61.1	0.007
	WV	*Myosotis laxa*	Both	38.5	0.003
	WV	*Madia sativa*	A	46.2	0.001
	WV	*Agrostis spp.*	N/A	43.8	0.096
	WV	*Gnaphalium palustre*	A	53.0	0.005
	WV	*Veronica americana*	P	26.9	0.024
	WV	*Gratiola ebracteata*	A	25.9	0.039
	WV	*Alopecurus geniculatus*	P	28.8	0.026
	WV	*Beckmannia syzigachne*	A	53.8	0.001
	WV	*Carex unilateralis*	P	34.6	0.006
	WV	*Juncus tenuis*	P	23.1	0.044

Table 2.2 (Continued)

Community Covariate	Group	Species	Life History	Indicator Value	P
Region	WV	*Epilobium densiflorum*	A	51.0	0.001
	WV	*Typha latifolia*	P	32.7	0.049
	WV	*Veronica scutellata*	P	34.6	0.004
	WV	*Rorippa curvisiliqua*	A	48.4	0.001
	WV	*Alopecurus aequalis*	P	30.8	0.015
	WV	*Downingia elegans*	A	30.8	0.012
	LCRV	*Phalaris arundinacea*	P	72.2	0.001
	LCRV	*Juncus effusus*	P	36.5	0.003
	LCRV	*Polygonum amphibium*	P	53.2	0.001
	LCRV	*Scirpus* spp.	P	25.0	0.013
	LCRV	*Sagittaria latifolia*	P	38.4	0.001
	LCRV	*Carex obnupta*	P	15.0	0.083
	LCRV	*Ranunculus repens*	P	20.0	0.028
	LCRV	*Equisetum fluviatile*	P	25.0	0.010

[a] Wetlands were categorized as unmanaged if no restoration, enhancement, or water control infrastructure existed. Passively managed wetlands were defined as those that had received no active management in at least three years; whereas, wetlands that had been managed more recently were categorized as actively managed.

Table 2.3. Results from the three dimensional Non-Metric Scaling ordination that related individual plant species' Pearson correlations, ordination axes, and community covariates. Indicator species of management intensity and their life histories agree with community variable correlations among ordination axes. Plant species are significant indicators of active management, passive management, or no management ($p < 0.10$).

Genus/ Species	Management Intensity	Axis 1[a] (region)	Axis 2[b] (mgmt)	Axis 3[c] (biomass)	Life History
Eleocharis ovata	active	-0.335	-0.063	0.338	A
Echinochloa crus-galli	active	-0.032	0.140	0.668	A
Myosotis laxa	Active	-0.421	0.242	-0.065	Both
Gnaphalium palustre	Active	-0.445	0.238	-0.374	A
Epilobium densiflorum	Active	-0.169	0.538	-0.222	A
Juncus bufonius	Active	-0.182	0.187	-0.378	A
Rorippa curvisiliqua	Active	-0.142	0.119	0.536	A
Polygonum lapathifolium	active	-0.360	0.409	0.314	A
Eleocharis palustris	Passive	-0.296	-0.156	0.729	P
Typha latifolia	Passive	-0.143	-0.088	0.340	P
Phalaris arundinacea	unmanaged	-0.266	-0.092	-0.780	P
Juncus effusus	unmanaged	-0.170	-0.377	-0.372	P
Ranunculus repens	unmanaged	-0.243	-0.259	-0.175	P

[a] Negative correlation with axis one is indicative of the Willamette Valley and positive correlation is indicative of the Lower Columbia River Valley.
[b] Negative correlation with axis two is associated with no management and positive correlation is associated with active management.
[c] Negative correlation with axis three is associated with low seed biomass yields for the particular plant species and positive correlation is associated with high seed yields.

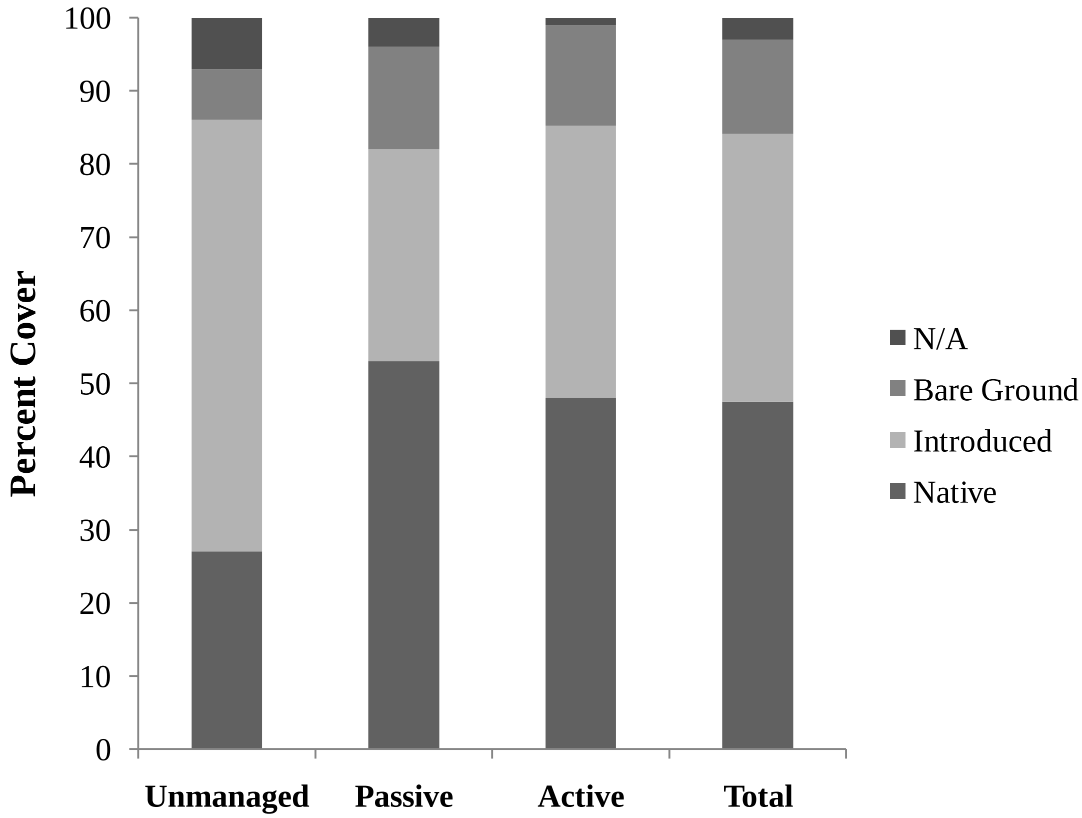

Figure 2.1. Mean percent cover of native species, introduced species, bare ground, and unknown taxa in seasonal wetlands in the Willamette Valley and Lower Columbia River Valley, autumn 2008 and 2009. Data shown for all wetlands combined and for wetlands grouped by management intensity. Unknown species are plants that were only identified to genus and it contained both native and introduced species.

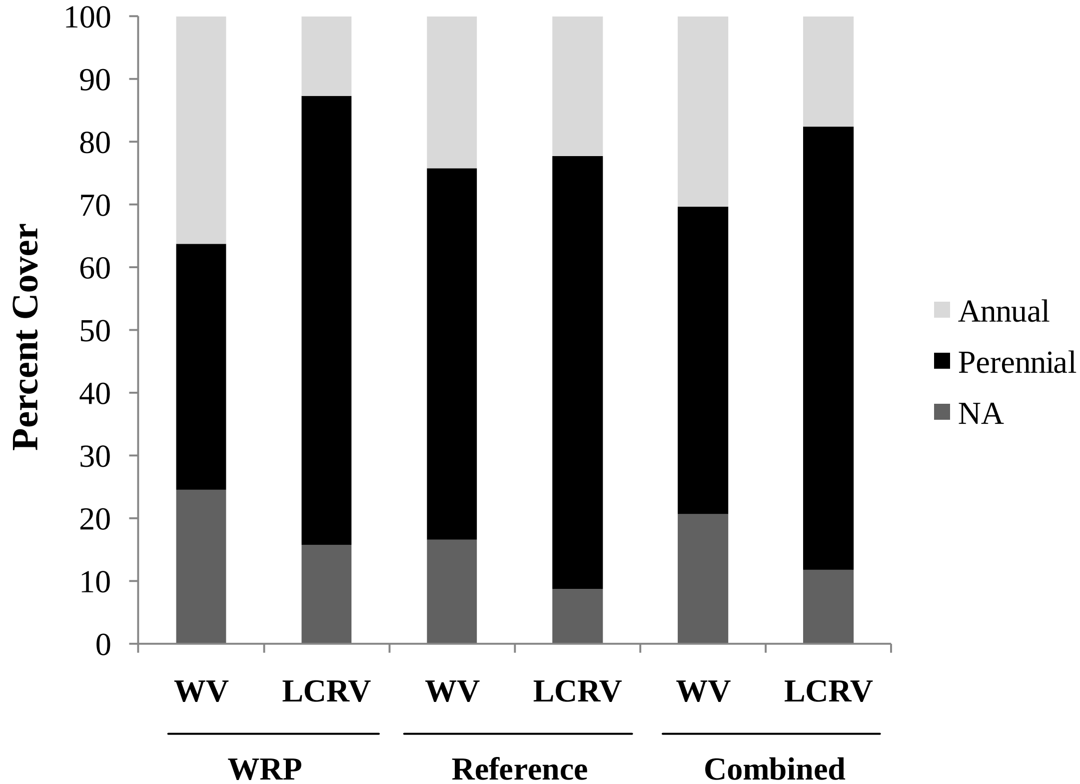

Figure 2.2. Mean percent cover of annual species, perennial species, and unknown taxa in seasonal wetlands sampled in the Willamette Valley (WV) and Lower Columbia River Valley (LCRV), autumn 2008 and 2009. Data is shown for all wetlands, wetland reserve program wetlands (WRP), and reference wetlands (REF) by study region. Unknown species are plants that are both annual or perennial, biennial or only identified to genus and it contained both native and introduced species.

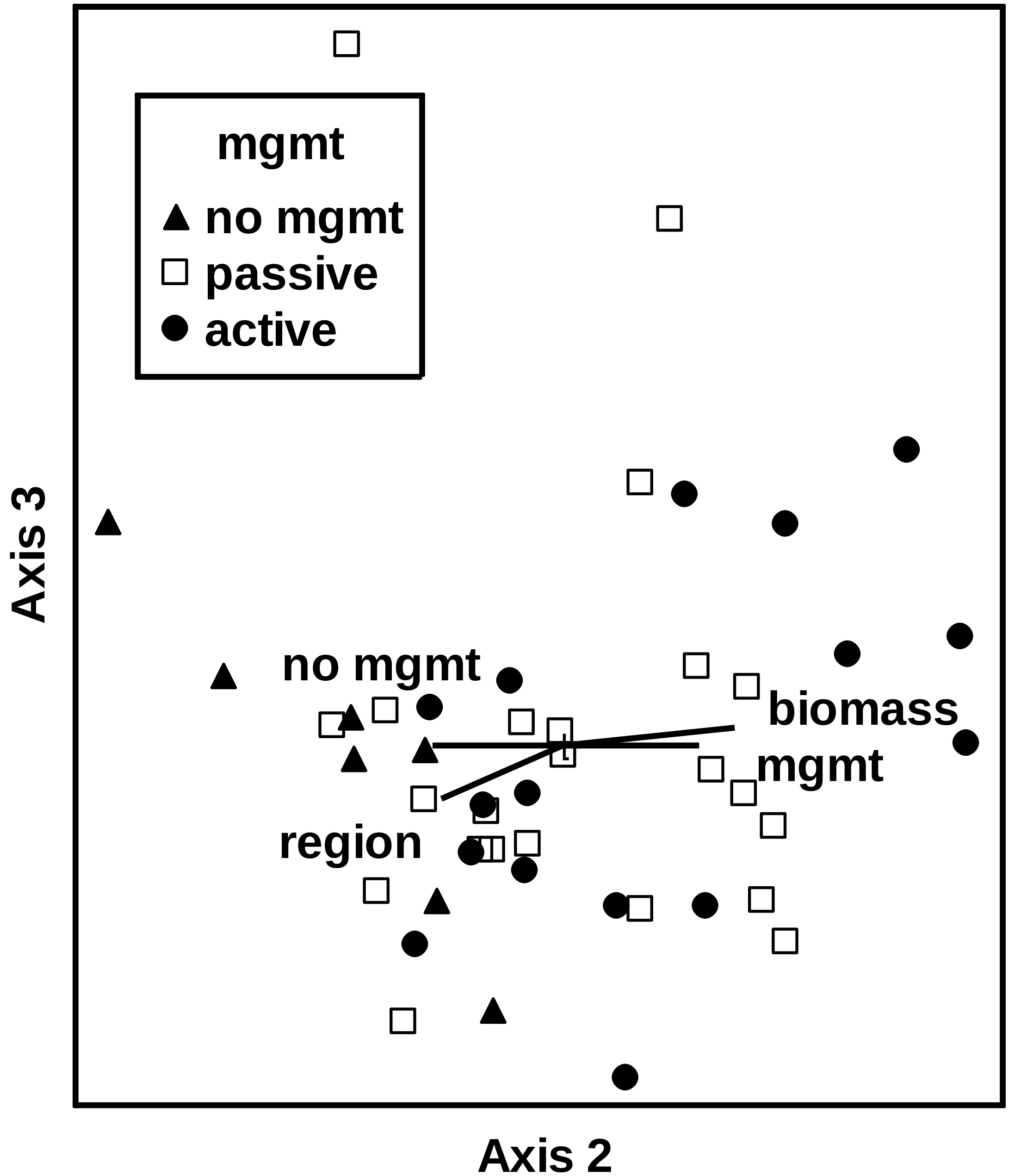

Figure 2.3. Non-Metric Scaling ordination of the plant community that occurred in 46 seasonal wetlands sampled in the Willamette Valley and Lower Columbia River Valley, autumn 2008 and 2009. Data points are wetlands labeled by management intensity in species space, where the distance among points is a measure of dissimilarity in plant community composition. Community covariates that are related to individual ordination axes are shown ($r^2 < 0.2$ for all covariates).

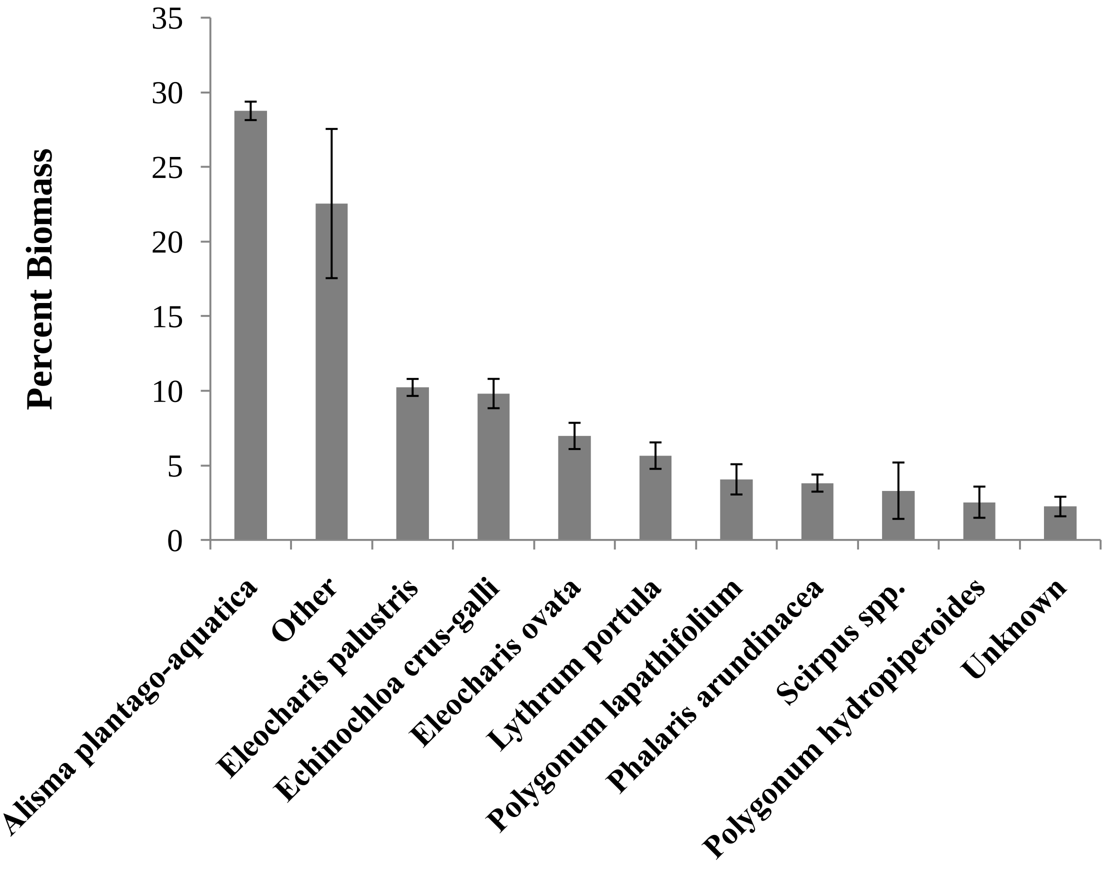

Figure 2.4. Plant species that made the largest contribution to seed biomass (mean ± SE) in seasonal wetlands (n = 46) on Wetland Reserve Program easements and reference sites in the Willamette Valley and Lower Columbia River Valley, fall 2008 and 2009. Species with < 2% biomass were grouped as other (n = 68).

DISCUSSION

Plant Community

I observed fewer plant species (113 taxa) compared to previous studies of wetlands in western Oregon (189 taxa, Gwin and Kentula 1990; 365 taxa, Magee et al. 1999). This difference is due, in part, to the fact that I sampled the plant community relatively late in the growing season; consequently, some early flowering annual species were probably not detected. Additionally, previous studies sampled a greater diversity of wetland habitats. However, the dominant plants in my study were similar with previous work. For example, my sample contained 18 of the 31 most common species reported by Magee and Kentula (2005). The species missing from my study were those indicative of other wetland types including forested wetlands (e.g. red alder [*Alnus rubra*], Oregon ash [*Fraxinus latifolia*] and black cottonwood [*Populus trichocarpa*]), permanent wetlands (e.g. duckweed [*Lemna minor*], large-leaf pondweed [*Potamogeton amplifolius*], and pond water-starwort [*Callitriche stagnalis*]), and uplands (e.g. Himalayan blackberry [*Rubus discolor*], common tansy [*Tanacetum vulgare*], and ivy [*Hedera helix*]).

Introduced plant species are common in seasonal wetlands in western Oregon and Washington. Forty three percent of all species were introduced, yet introduced species constituted only 37% of total percent cover estimates. In a study of mitigation wetlands, greater than 50% of species were introduced (Magee et al. 1999). Reed canary-grass was the most common introduced species in all studies. There were 34% introduced species in managed wetlands; considerably less than unmanaged wetlands (59%; Figure 2.1). Despite the abundance of reed canary-grass, the next three most abundant plants were native species with high seed production; creeping spike-rush, American water plantain, and waterpepper. I recommend management be used more frequently to control introduced species.

My study is the first to use a holistic community analysis to compare plant composition between Wetland Reserve Program wetlands and reference wetlands. Similar to my results, Wetland Reserve Program wetlands in the Cache River Watershed in southern Illinois did not differ among reference wetlands using plant community indices such as percentage of plant growth forms (i.e. forbs, grasses, sedges and rushes, vines, woody vegetation) as metrics (Hicks 2003). However, plant community indices only explain specific components of a community and may be correlated, whereas incorporating the entire plant community composition in statistical tests provides a finer resolution to understanding differences in wetland plant communities.

Plant communities in Wetland Reserve Program wetlands differed by management intensity. Annual species like barnyard grass and curlytop knotweed were indicative of actively managed wetlands and perennial species like reed canary-grass and common rush (*Juncus effusus)* were characteristic of passive or unmanaged wetlands. Annual plants produce more seed and the proportion of plants with each life history may act as a surrogate for assessing biomass production. Abundance of bare ground was positively correlated with biomass estimates indicating early successional plant communities produced higher seed yields. Management practices such as mowing, disking, or herbicide application promoted early successional species, which resulted in greater waterfowl forage production. Recent studies reported higher diversity and abundance of wetland dependent birds and greater waterfowl forage quality on actively managed versus passive or unmanaged wetlands Kaminski et al. (2006), Fleming (2010), and Olmstead (2010). The variability of metrics used in Wetland Reserve Program wetland studies confounds comparisons, but all studies indicate increased biological value for waterfowl in actively managed wetlands.

Seed Production

Seasonal wetlands in western Oregon and Washington are meeting their goal of providing biological value for migrating and wintering waterfowl by producing similar seed yields as other major wintering regions. My overall seed biomass estimate of 505 ± 59 kg/ha is similar to estimates for other regions in North America that traditionally support large numbers of wintering waterfowl including the Mississippi Alluvial Valley (496 ± 62 kg/ha; Kross et al. 2008, 528 kg/ha; Olmstead 2010), California's Central Valley (586 kg/ha; Naylor 2002), and the Upper Midwest (520 and 377 kg/ha respectively for active and passively managed seasonal wetlands; Brasher et al. 2007). Comparisons among studies are comparable as all focused on seasonally flooded managed wetlands.

Wetlands created using the Wetland Reserve Program in western Oregon and southwest Washington are producing waterfowl food resources comparable to wetlands on publicly managed lands. Dominant seed producing plants were similar in Wetland Reserve Program and reference wetlands with American water-plaintain, common spike-rush, barnyard grass, and ovate spike-rush contributing 61% and 52% respectively to total biomass estimates in each wetland type. However, a greater number of wetland plants contributed to seed biomass estimates in Wetland Reserve Program wetlands (Appendix IV). Plant community data revealed a similar pattern with Wetland Reserve Program wetlands being more diverse than reference wetlands, largely due to the presence of facultative and upland species. Thirty-eight species occurred exclusively in Wetland Reserve Program wetlands with 53% of those being facultative or obligate upland species.

Seed production did not differ between wetland type or study region, but Willamette Valley WRP wetlands produced greater seed biomass (560 ± 114 kg/ha) than Lower Columbia River Valley WRP wetlands (188 ± 43 kg/ha). Lower seed production in Wetland Reserve Program wetlands in the Lower Columbia River Valley is likely due to the dominance of perennial plant species that typically produce lower seed yields than annuals. There were 22% more perennial species in the Lower Columbia River Valley and the difference was greatest between Wetland Reserve Program wetlands in the two regions (33%, Figure 2.2). Reed canary-grass is a cryptogenic perennial species that forms monocultures, outcompeting native plants (Kentula et al. 2004), and was indicative of unmanaged wetlands. Despite being the most abundant species with 24% mean percent cover it only contributed 4% to seed biomass estimates. Furthermore, all indicator species in the Lower Columbia River Valley were perennial (8 of 8); whereas, 9 of 18 indicator species for the Willamette Valley were exclusively annuals.

The management applied by private landowners or restoration activities appears to be reasonably effective at increasing the biological value of WRP wetlands for waterfowl. Therefore, periodic management is as necessary on Wetland Reserve Program wetlands as public wetlands to achieve similar biological benefits. Plant species associated with active management tended to be early successional annuals with high seed biomass yields. For example, barnyard grass, ovate spike-rush, and curlytop knotweed are all annual species and were the 3rd, 4th, and 6th largest contributors to biomass estimates, respectively. Species associated with unmanaged wetlands were later successional perennial species with low biomass yields. Most WRP wetlands that were categorized as actively managed (4 of 5) was due to restoration actions taking place within the previous three years. The WRP is a relatively new program and management is often the responsibility of landowner. Management intensity results indicate that if early successional plant communities are not maintained on WRP sites, their biological value for wintering waterfowl will degrade over time.

MANAGEMENT IMPLICATIONS

Habitat restoration on private lands is the most feasible way to significantly increase wetland acreage in the United States; therefore it is imperative to maximize the biological value of these habitats. State and federal refuges are relatively established, making future acquisition less likely on public lands. Wetland Reserve Program seasonal wetlands are a viable means of increasing wetland acreage and waterfowl food resources in the region as WRP wetlands in my study did not differ from reference wetlands and produced relatively high seed biomass. From a conservation planning perspective, regional wetlands appear to be providing sufficient food resources, yet there is a paucity of habitat with only 96 km^2 of the 9100 km^2 of land in the Willamette Valley being Wetland Reserve Program or publicly managed seasonal wetlands.

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Chapter 3

ABOVE AND BELOW GROUND DISTRIBUTION OF WETLAND PLANT SEEDS IN SEASONAL WETLANDS

INTRODUCTION

Seed production by wetland plants is one wetland characteristic that can be used to assess habitat quality and evaluate habitat restoration and management (Laubhan and Fredrickson 1992, Gray et al. 1999). Seed biomass estimates are also important for developing habitat specific conservation plans for wintering waterfowl (NAWMP Plan Committee 1998, NAWMP Steering Committee 2007). In an effort to improve regional conservation planning for non-breeding waterfowl, a number of recent studies have estimated seed biomass to appraise wetland quality (e.g., Naylor 2002, Taylor and Smith 2005, Brasher et al. 2007, Greer et al. 2007, Kross et al. 2008).

Within a wetland, seeds can occur in four fundamental locations; on the plant, on top of the ground, in the water column, and in the soil. Allocation of seed biomass in wetland environments (i.e., above ground vs. below ground) is rarely reported in forage studies, yet it has implications for defining availability to foraging waterfowl (Nolet et al. 2001, Reinecke and Hartke 2005). While it is reasonable to assume that all seed on the plant, in the water, and on top of the ground can be available to foraging waterfowl, it is not reasonable to assume that all seed occurring in the soil is available. The size of the possible error associated with making this latter assumption depends on what percent of seed biomass occurs in the soil, its distribution by depth within the soil profile, and ultimately on the ability of ducks to feed in wetland soils. Quantifying the distribution of seed resources by location in various habitat types would provide insight regarding availability to ducks.

The most commonly used technique for estimating seed biomass in wetland soils is to collect and process a sample of soil cores (Reinecke and Hartke 2005, Kross et al. 2008). While the technique is common, there is variability among studies in the depth of soil core that is collected and processed (Taylor and Smith 2005, Greer et al. 2007, Kross et al. 2008), which can make direct comparison of estimates difficult. From a practical standpoint, processing soil cores is time consuming and expensive (Peters and Dugger pers. experience) and processing time is directly correlated with the volume of soil collected in each core. One study determined that collecting cores of 5 cm depth was adequate for sampling benthic chironomids when estimating availability of invertebrate prey to shorebirds (Sherfy et al. 2000). No such study has been conducted for wetland plant seeds.

I conducted a study to determine the distribution of seeds in seasonal wetland environments (on the plant, on top of the ground, and in the soil) and at different depths within the soil during late summer and early fall. I also conducted an analysis to determine if seed biomass in upper layers of the soil profile can be used to predict seed biomass at deeper depths with the goal of developing a sample processing and analysis technique that would expedite processing time in future studies of food abundance.

METHODS

I estimated seed biomass production by collecting a series of samples from 46 seasonal wetlands (26 in 2008 and 20 in 2009) located within the Willamette Valley (WV) in Oregon and the Lower Columbia River Valley (LCRV) in Oregon and Washington (Figure 1 in Chapter 1). Half of the wetlands were located on either Federal National Wildlife Refuges or State Wildlife Management Areas and the other half were located on privately owned Wetland Reserve Program easements (Chapter 2). I used a Geographic Information System ([GIS] ESRI ArcGIS version 9.0, 2004) to overlay a grid of 30-40 cells over each wetland, randomly generated one point within each cell for sampling, and used a random number generator to select 20 cells to sample.

I collected three sub-samples at each sampling location that allowed partitioning of seed biomass estimates to on the plant (inflorescence), on the substrate, and within the soil. At each sample location, I clipped all inflorescences within a 25 x 25 cm frame and cut remaining vegetation to ground level. Then, I used a gasoline powered vacuum to collect all seeds on the substrate inside an 11 cm diameter sampling frame (Penny et al. 2006) placed within the sampling frame used to clip inflorescences. Lastly, I extracted a 10 cm diameter soil core within the space that was vacuumed. Soil core depth differed between years, being 10 cm in 2008 and 5 cm deep in 2009.

In 2008, I collected deeper cores so I could partition each core into discrete depth layers to determine the distribution of seed biomass within the soil profile and test for correlations between biomass at different soil depths. The soil core sampler was equipped with a graduated plunger and stopper that allowed me to partition soil cores into depth layers (Figure 3.1). I partitioned the first 10 core samples from all wetlands (n = 26) into three depth layers (0-2, 2-5, and 5-10 cm). For 19 of the 26 wetlands, I partitioned samples 11-20 into two depth layers (0-5 and 5-10 cm). All core subsamples were placed in separate plastic freezer bags that were labeled and frozen until processing began. I sampled each wetland between 25 August and 1 October in fall 2008 or 2009 after most plants were mature but prior to the initiation of fall flooding.

In the lab, I processed inflorescence and vacuum subsamples by threshing all seeds from chaff, residual vegetation, and debris using a series of graduated sieves (mesh sizes 2 mm, 1 mm, 500 µm, 355 µm and 250 µm), manual gravity and air separation techniques (Harmond et al. 1968), and hand removed seeds with forceps. Inflorescence and vacuum samples were air dried for 2-6 weeks before processing began (Laubhan and Fredrickson 1992). Core samples were thawed, rinsed through a series of graduated sieves (sizes include 2 mm, 500 µm, and 250 µm), dried at 60ºC for 24 hours, and processed to extract seeds (Greer et al. 2007). All seeds were dried at 60ºC to constant mass and weighed to the nearest 0.0001 g.

Statistical Analysis

The sample unit for most analyses was the wetland ($n = 46$); I standardized all biomass sub-sample estimates to kg/ha, calculated a mean biomass value for each location within the wetland (inflorescence, top of soil, in soil) and depth layer within the soil, and used that value to summarize data and conduct statistical tests. For the analysis that compared above ground to below ground biomass, all cores were 5 cm deep (for 2008 I combined the 0-2 and 2-5 depth layers). I summed inflorescence and vacuum wetland sub-sample estimates because I assumed all seed biomass on the plant and on top of the soil (above ground) was available to foraging ducks; whereas, the availability of seeds in the soil is unknown. Actively managed, passively managed, and unmanaged wetlands have been shown to yield different amounts of seed (Brasher et al. 2007, Kross et al. 2008) and the distribution of those seeds also most likely varies among plant communities with different management intensities. I used PROC MIXED in SAS version 9.2 (SAS Institute 2004) to test for differences in seed biomass by location (i.e. above ground, below ground), study region (WV, LCRV), their interaction, and management intensity. I treated location and study region as fixed effects and management intensity as a random effect. Prior to conducting the test, I inspected box and residual plots and confirmed that data met distributional assumptions (Ramsey and Schafer 2002).

Using the data from the 10 cm soil cores in 2008, I determined the distribution of seeds within the soil profile by calculating the mean seed biomass estimate in each wetland for each depth layer. I then tested if mean seed biomass differed by depth layer, study region, their interaction, and management intensity using SAS Version 9.2 (PROC MIXED), first on the subset of samples partitioned into three depth layers; then I combined the 0-2 cm and 2-5 cm layers for those samples and tested for a difference between 0-5 and 5-10 using all samples. I conducted pairwise multiple comparisons using a Tukey-Kramer procedure to further examine differences in seed biomass among depth layers in three layered cores when the overall model was significant.

Finally, I used the 10 cm soil core data in 2008 to determine if seed biomass in upper soil layers could predict seed biomass at lower depths. I conducted regression analyses using 0-2 cm or 0-5 cm samples as the independent variable and 5-10 cm biomass as the dependent variable (PROC GLM). The sample unit for this analyses was each partitioned soil core; n = 260 for three layered cores and n = 450 for two layered cores. I $\log_{10}(x + 0.0001)$ transformed all biomass values to normalize data for regression analyses (Ramsey and Schafer 2002). I calculated Cook's distances, a measure of overall influence on estimated regression coefficients, to investigate the influence of potential outliers. All values were < 0.1 which indicated that no remedial measures were required (Ramsay and Schafer 2002, R Development Core Team 2006). I used the R-squared value from each regression to assess predictive ability of biomass deeper in the soil. All means are reported ± SE.

RESULTS

The mean seed biomass estimate for all wetlands combined was 505 ± 59 kg/ha, which was distributed as 55% on inflorescences (277 ± 41.2 kg/ha), 28% in the soil (141 ± 18.5 kg/ha), and 17% on the soil surface (86 ± 17.9 kg/ha). More seed was located above ground; however, the strength of the relationship differed between regions ($F = 7.75$, $P = 0.007$). The relationship was stronger in the WV with 79% of seed biomass above ground ($P = 0.009$) compared to 61% ($P = 0.046$) in the LCRV. Mean below ground seed biomass did not differ by study region ($P = 0.715$; Figure 3.2).

Seed biomass varied by depth layer in both three layered cores ($F_{5,72} = 6.27$, $P = 0.003$) and two layered cores ($F_{3,48} = 25.44$, $P < 0.001$; Figure 3.3). Pairwise comparisons indicated mean seed biomass was higher in the 0-2 cm layer compared to either the 2-5 cm ($t = 4.18$, $P = 0.001$) or 5-10 cm layers ($t = 3.29$, $P = 0.006$, but biomass was similar between the 2-5 cm and 5-10 cm layers ($t = 0.65$, $P = 0.52$). Most seed biomass (75%) was located in the upper 5 cm of the soil and only 25% was located from 5-10 cm (Figure 3.4). Mean seed biomass in three layered core samples was 143.2 ± 18.6 kg/ha and average percent seed biomass was distributed as 52% in the 0-2 cm depth layer, 27% in 2-5 cm and 21% in 5-10 cm (Table 3.1). Results were similar when I compared 0-5 cm vs. 5-10 cm; 75% of seed biomass was located in the top 5 cm. Lastly, seed biomass estimates in upper soil layers were generally poor predictors of biomass below; regression models only explained between six and seven percent of variation in the data (Figure 3.5).

Table 3.1. The distribution of wetland seed biomass among depth layers in the soil from seasonal wetlands (*n* = 26) in the Willamette Valley and Lower Columbia River Valley, Oregon and Washington during 2008. The 0-5 cm layer is the sum of the 0-2 cm and 2-5 cm values. Wetland seed biomass means ($\bar{x}$), standard errors (SE), coefficients of variation (CV), and 95% confidence intervals (C.I.) are in kg/ha.

Depth Layer	**$\bar{x}$ biomass**	**SE**	**CV (%)**	**95% C.I.**
0-2 cm	74.4	12.5	17	49.9-98.9
2-5 cm	38.7	5.9	15	27.1-50.3
0-5 cm	109.5	12.4	11	85.3-133.7
5-10 cm	35.8	7.0	20	22.1-49.5
Total Biomass	145.3	16.6	11	112.8-177.8

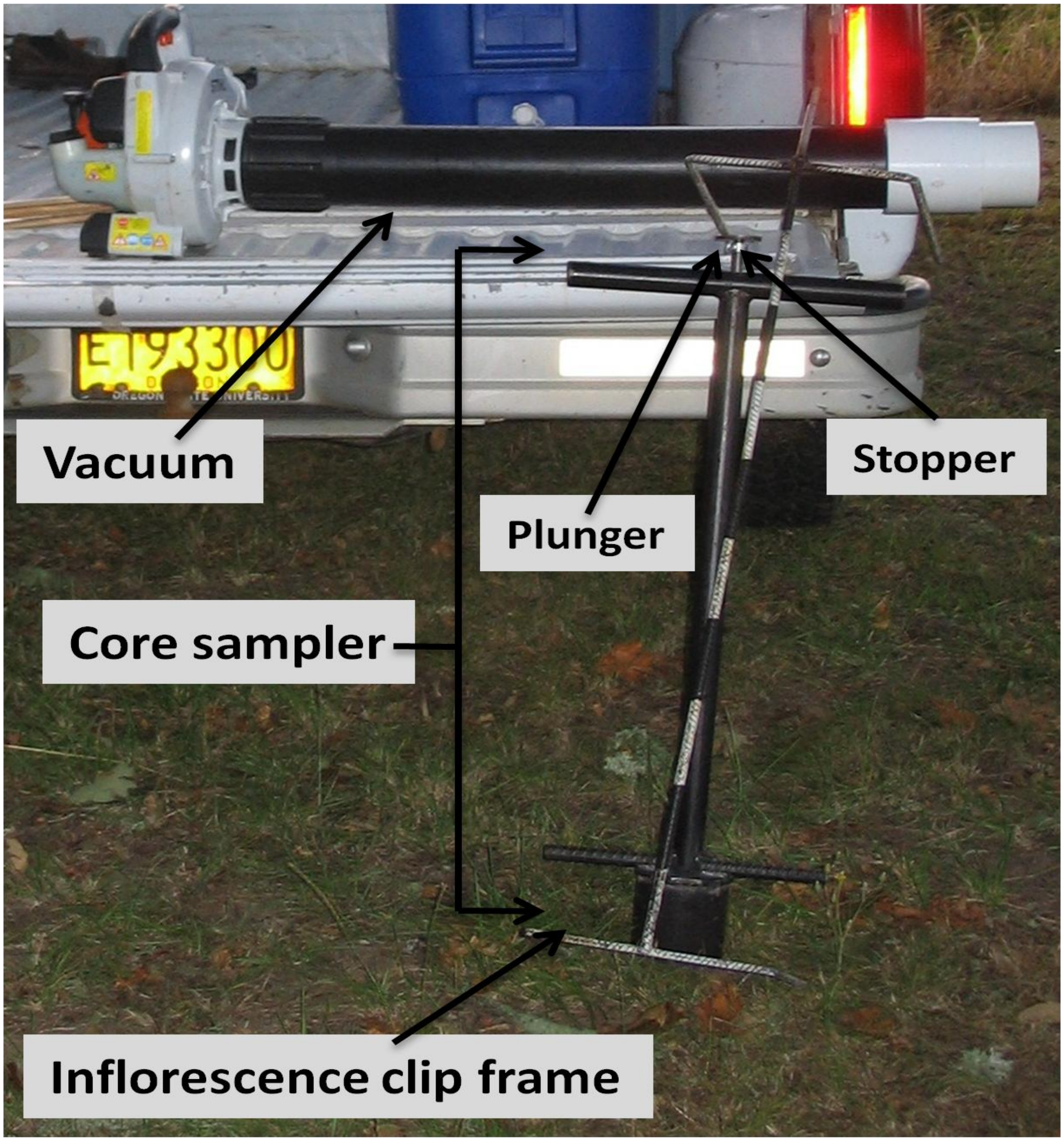

Figure 3.1. Photograph of sampling gear used to estimate seed abundance in seasonal wetlands. The core sampler is equipped with a 10 cm plunger that has an adjustable stopper on the shaft marked at 2 cm and 5 cm for removal of discrete core depth layers.

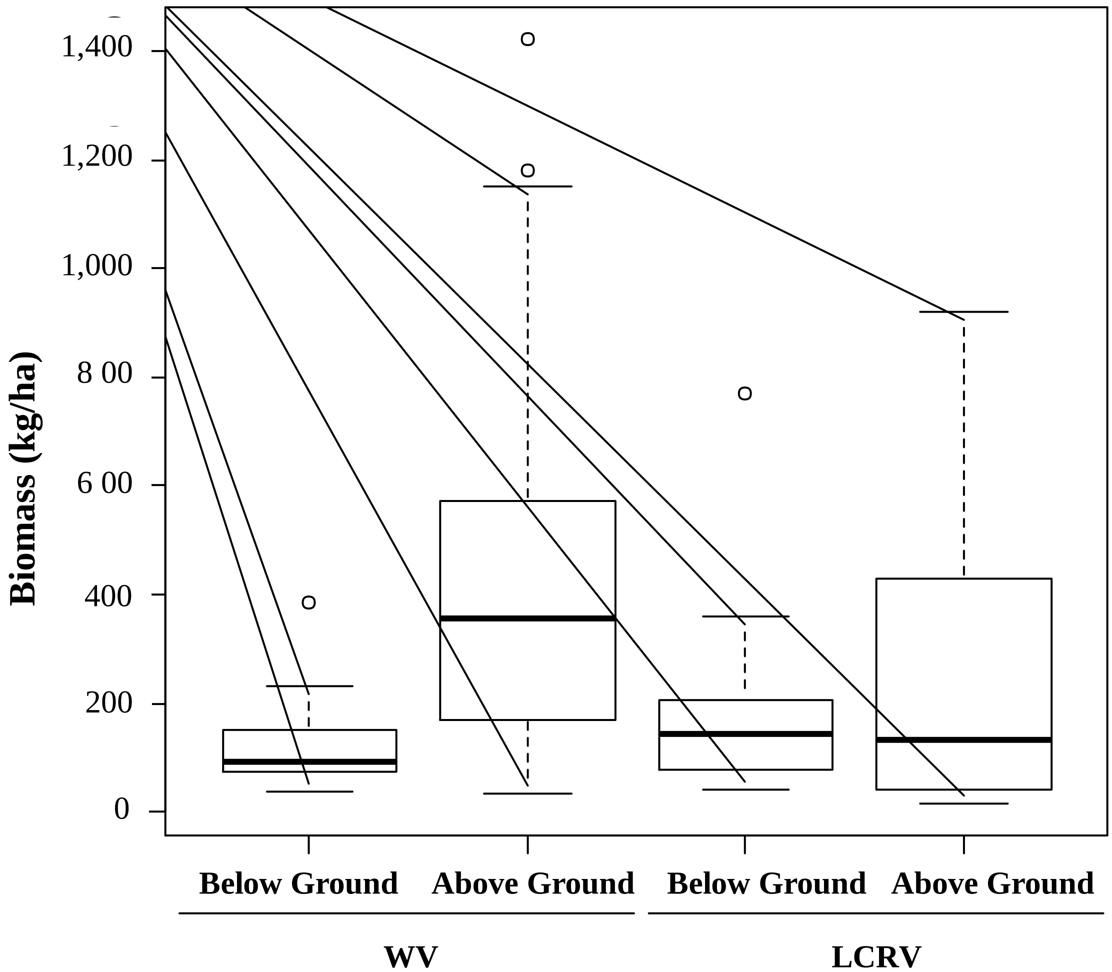

Figure 3.2. Box and whisker plot of above ground and belowground seed biomass in 46 wetlands sampled in the Willamette Valley and Lower Columbia River Valley, 2008 and 2009. More seed biomass was located above ground (72%; 362 ± 50.8) than below ground (28%; 141 ± 18.5 kg/ha).

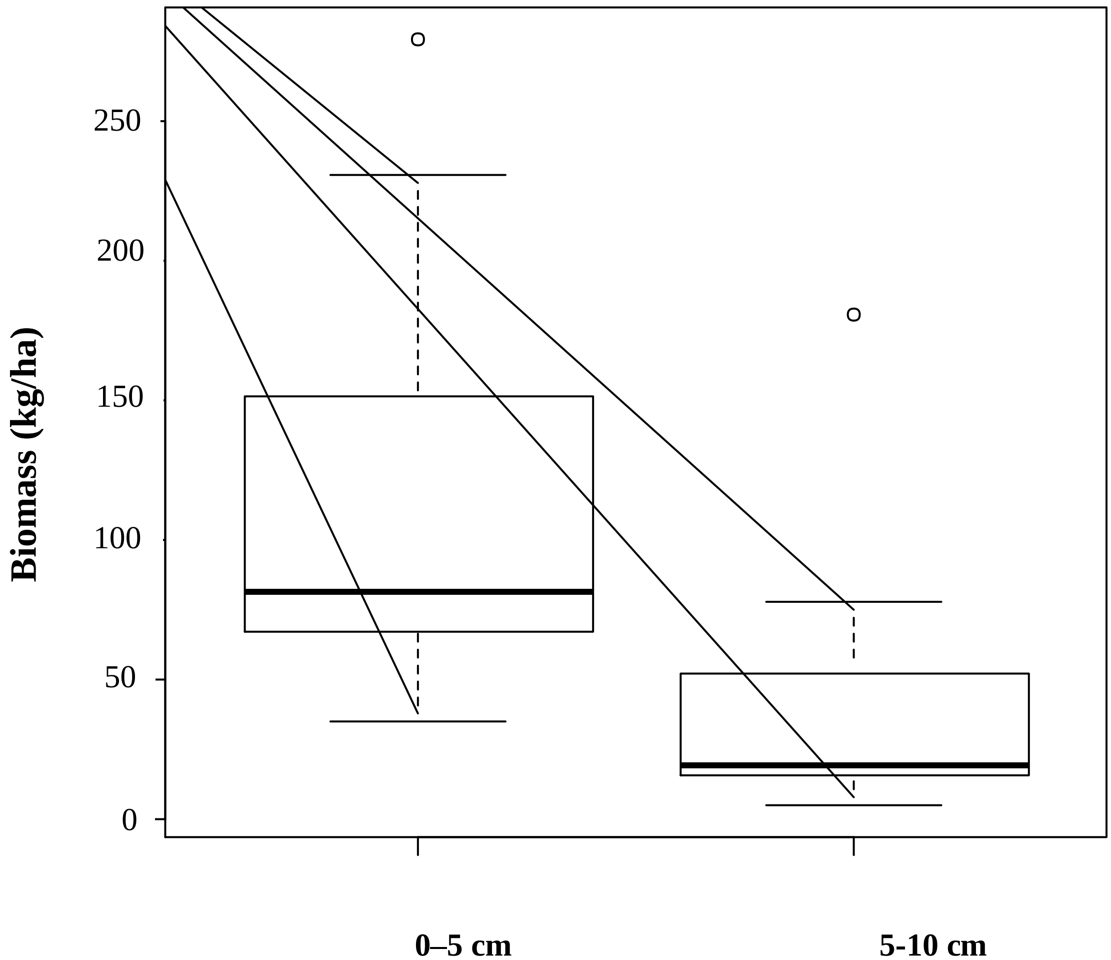

Figure 3.3. Box and whisker plot showing the distribution of seed biomass between 0-5 and 5-10 cm depth categories in the soil of 26 wetlands sampled in the Willamette Valley and Lower Columbia River Valley, 2008. Mean seed biomass was 109.5 ± 12.4 kg/ha in the top 5 cm and 35.8 ± 7.0 kg/ha in the 5-10 cm layer.

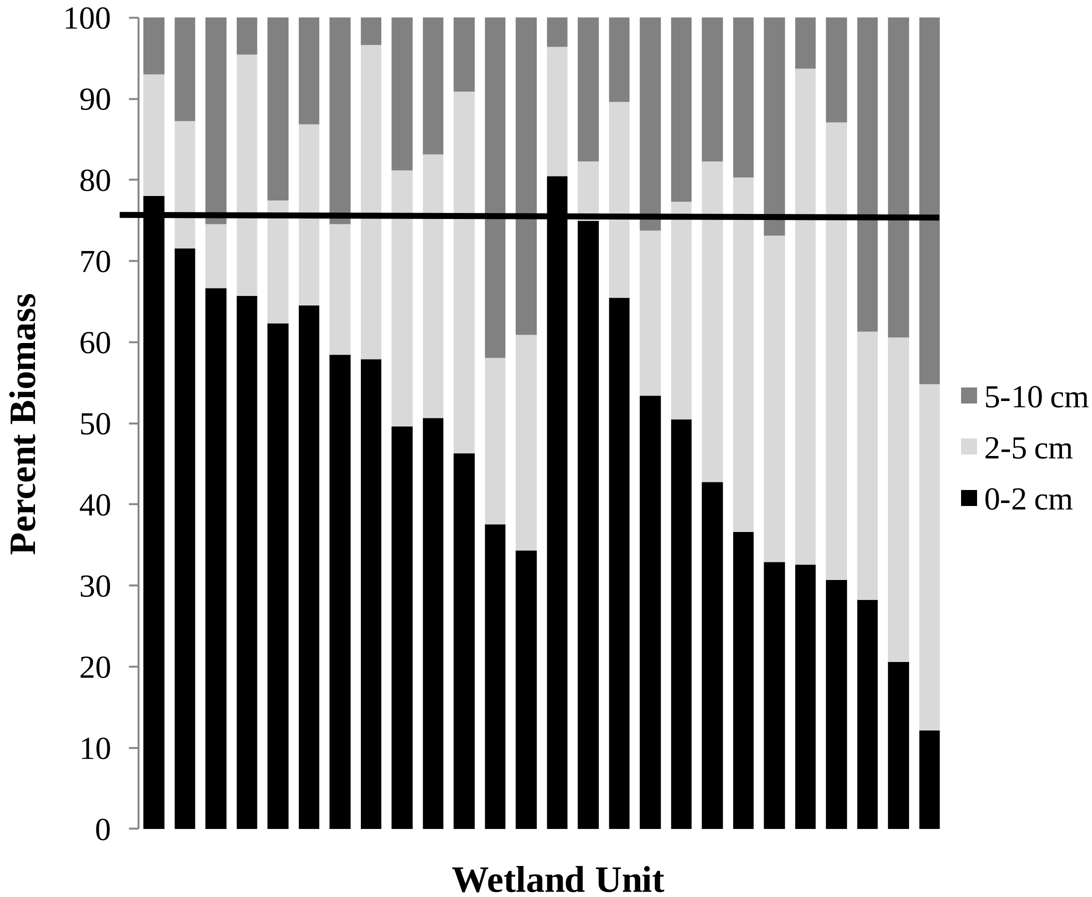

Figure 3.4. Mean percent seed biomass distribution in three layered core samples for all wetlands (n = 26) in the Willamette Valley and Lower Columbia River Valley, 2008. The black line indicates the mean percent biomass in the top 5 cm (75%).

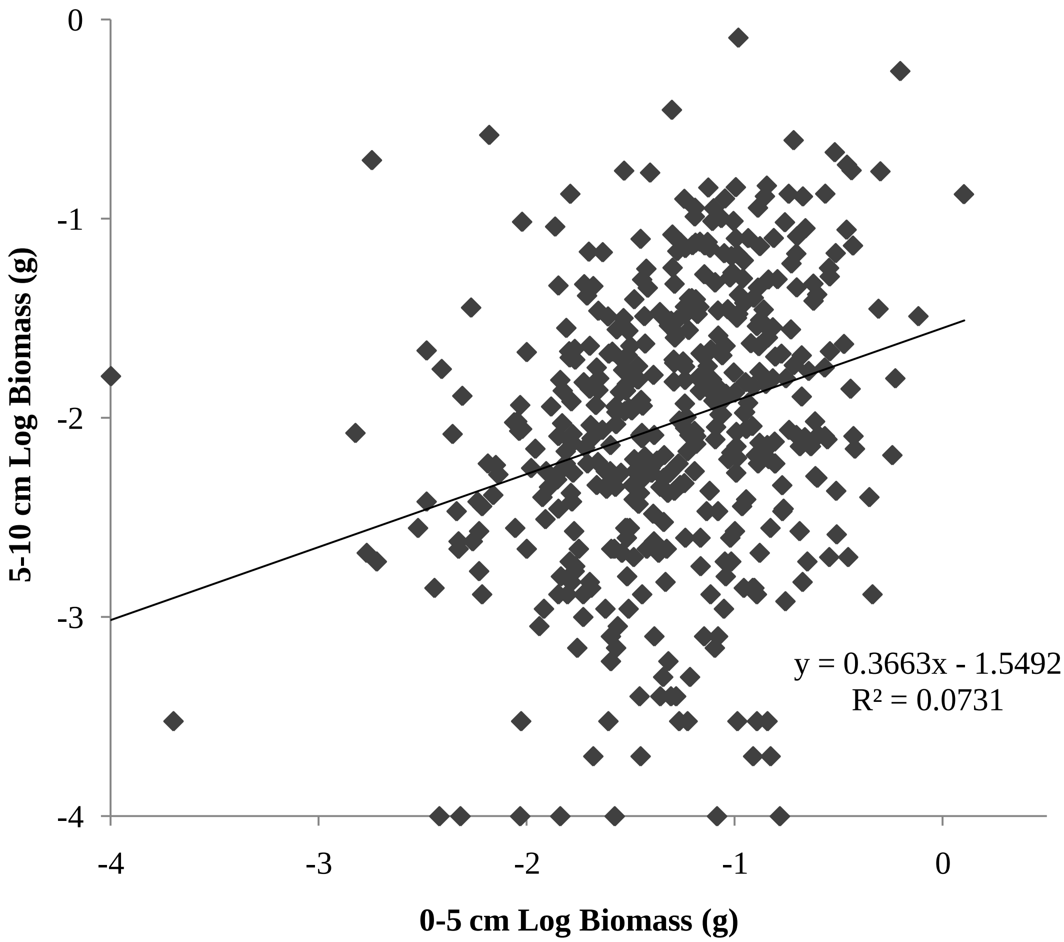

Figure 3.5. Linear regression model showing the association between seed biomass (log transformed data) in in the 0-5 cm and 5-10 cm depth layers of the soil from 26 seasonal wetlands (*n* = samples 450) in the Willamette Valley and Lower Columbia River Valley, autumn 2008.

DISCUSSION

This is the first study to partition seed biomass as above and below ground.. Most wetland seed was located above ground (72%); readily available to foraging ducks, yet my results indicate the proportion of seed above ground could vary among regions. Wetlands in the Willamette Valley had higher seed yields resulting in significantly more above ground seed biomass. The proportion of above to below ground biomass may vary annually due to disturbance, succession, climate and hydrology. However, the difference in above ground seed biomass between study regions appeared to be most related to seed yield of the plant communities present. Below ground seed biomass did not vary between regions despite different above ground estimates. However, 28% of seed biomass was located below the soil surface where the availability for dabbling ducks is unknown.

My results agree with similar studies that found greater seed biomass or number of viable seeds in the top layers of the soil profile. A study in the Mississippi Alluvial Valley found 70% of seed biomass in the 0-2.5 cm layer, followed by 20% in the 2.6-5 cm layer, and 10% in 5.1-10 cm (Olmstead 2010). Leck and Graveline (1979) found the proportion of seedlings from tidal freshwater marshes which germinated in 0-2, 4-6, and 8-10 cm layers was a 3:2:1 ratio respectively. Compared with grasslands (5:1 ratio in 0-2.5 and 2.5-5 cm layers; Major and Pyott 1966) and coniferous forests (5:1 ratio in 0-5 and 5-10 cm layers, Kellman 1970) freshwater tidal marshes have a more gradual decrease of seeds within the soil profile. My results indicate seasonal freshwater wetlands may exhibit a more pronounced pattern of a gradual decrease in seed biomass at increasing depths with a 2:1:1 ratio of seed biomass in 0-2, 2-5, and 5-10 cm layers.

Perhaps this gradual decreasing pattern is attributable to waterfowl consuming large portions of seed in the top two cm. The relatively large amount of seed biomass at deeper depths compared to ecosystems may be because those seeds are unavailable to foraging ducks. Within the soil profile, 75% of seed biomass was in the top 5 cm, yet regions or habitats with lower seed yield may have a smaller proportion of seeds near the surface.

Several factors may directly affect the vertical distribution and movement of seeds in wetland soils including physical and chemical properties of the soil, hydrology, and disturbance (Van Der Valk and Davis 1978, Leck 1989). All wetlands I sampled were seasonally flooded (fall-spring) with relatively similar depth, duration, and timing of inundation and hydric soils in the WV and LCRVs are similar. These wetland soil types have high clay content and shrink-swell properties that may concentrate seeds close to the surface or in cracks (NRCS 2010). A study examining the impact of plowing disturbance on soil seed banks in temporary marshes found more seeds in upper soil layers (0-6 cm) of fallow marshes, whereas more seeds were found in deeper layers (6-15 cm) of plowed pools (DeVictor et al. 2007). Variability and subsequent poor correlation between seed biomass in the upper soil depths (0-5 cm) versus the bottom (5-10 cm) may be due to management practices such as plowing, heterogeneous waterfowl foraging behavior within wetlands, climatic effects (i.e. wind and erosion), and differences in hydrology.

Core sub-sample estimates were not correlated between depth layers, which may have been attributed to a combination of factors affecting vertical distribution not measured in this study. The variability in the thickness of the top layer of the soil profile (i.e. organic horizon) largely depends on the plant community, hydrology, and disturbance factors (Collins and Kuehl 2001). Perennial plant species have more residual vegetation and some, such as reed canary-grass, have rhizomes that often results in a larger organic horizon (Peters pers. experience). Undecomposed organic debris and decomposed organic material lack the density of mineral particles found in deeper soil layers, which presumably facilitates leaching or vertical movement of seeds (Leck 1989). Wetlands with lentic hydrology may have less seed near the soil surface due to erosion and wetlands surrounded by trees or shrubs may reduce the erosive forces of wind. Finally, disturbance regimes such as disking, mowing, herbicide application or any management practice that alters vegetation and subsequent seed production will affect the vertical distribution of seed biomass.

Waterfowl conservation planners need to develop a better understanding of how ducks forage within wetland soils to place my results in better biological context. Maximum foraging depth in the soil likely varies among species due to differences in feeding behavior, culmen length, head musculature, and epidermal bill structure (Goodman and Fisher 1962). Most dabbling ducks have only a small central area of the bill tip occupied by the nail because this foraging guild does not require the forceful use of the bill for feeding, suggesting their bills are not adapted for foraging deep within the soil. Mallards (*Anas platyrhynchos*) are relatively large dabbling ducks that have an average bill length of 41.7 mm for males and 38.7 mm for females (Nudds and Kaminski 1984), suggesting maximum foraging depths of approximately 50 mm in the substrate, and foraging depths are likely much shallower for smaller species like green-winged teal. Mallards have been observed submerging their heads in the substrate to extract tubers in soft sediment (Drilling et al. 2002). Therefore, they may be able to forage slightly deeper than maximum bill length depending on sediment type. However, soils in our wetlands were very firm, even after being flooded (G. Peters pers. observation); consequently foraging below 5 cm seems unlikely.

Conservation planners account for unavailable food resources in waterfowl foraging habitats by subtracting a Giving Up Density from estimates of seed abundance (GUD; Reinecke et al. 1989, Nolet et al. 2006, Greer et al. 2009). A GUD of 50 kg/ha is typically applied to agricultural habitats (Reinecke et atl. 1989, Greer et al. 2009) and 34 kg/ha is commonly applied for moist soil plant seeds in seasonal wetlands (Naylor 2002). My results provide some measure of the range associated with uncertainty regarding food availability in seasonal wetlands. If the 36 kg/ha of seed biomass below 5 cm in the soil in my study were unavailable then the GUD of 34 kg/ha adjustment currently being used for waterfowl conservation planning is a reasonable estimate of what might be physically unavailable to a foraging duck. I would be overestimating seed availability to ducks by 105 kg/ha (141-36 kg/ha; 21%) if food resources in the soil were completely unavailable to ducks and 38kg/ha (7.5%) if ducks can forage down to 2 cm in the soil. Finally, my estimate of seed biomass physically available to a foraging duck is underestimated by 36 kg/ha if waterfowl can forage to a depth of 10 cm within the soil. My calculations are likely a liberal estimate of availability because I only account for physical availability of food to ducks who may also cease foraging in an area if food density falls below a threshold where it is no longer profitable to continue foraging (Sutherland and Parker 1985, Sutherland et al. 2002). Given waterfowl vary considerably in the size and shape of their feeding structures, an accurate estimate of food availability likely varies by species.

Previous food abundance studies that produced estimates from 10 cm deep cores are most likely overestimating food availability for dabbling ducks. Based on my results and a similar study in the Mississippi Alluvial Valley (Olmstead 2010), food abundance is overestimated by six to ten percent. Estimating seed biomass distribution within the soil profile is an important initial step regarding physical availability of waterfowl food resources, however conservation planners could accurately quantify available forage if waterfowl foraging depth were investigated. To accurately evaluate habitat specific foraging carrying capacity I recommend an experimental evaluation of the depth waterfowl can extract seed resources from various substrates.

MANAGEMENT IMPLICATIONS

Most seed biomass was in the top 5 cm of the soil core. Results from a single study cannot lead to a broad general conclusion; however, I recommend that future studies consider my results when designing their protocol by partitioning 10 cm soil cores into two depth categories, processing all samples for 0-5 cm, and processing a much smaller subsample of cores from 5-10 depth layer. The deep cores could be used to test site-specific findings against our results and provide a mechanism for adjusting biomass estimates of future work indicates that ducks can extract seeds at depths below 5 cm.

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Chapter 4

SYNTHESIS: INTEGRATING RESULTS INTO REGIONAL CONSERVATION PLANNING

INTRODUCTION

In 1986, the United States and Canada developed the North American Waterfowl Management Plan (NAWMP) with the goal of restoring waterfowl populations to 1970s levels through habitat protection, enhancement, and restoration (Williams et al. 1999). NAWMP is international in scope, but functions at a regional level through 14 Joint Ventures (JV; USFWS 2010), which are positioned at geographic locations important for waterfowl. The Pacific Coast Joint Venture (PCJV) implements NAWMP conservation strategies on the Pacific Coast from Alaska to northern California, including Oregon and Washington. Food availability is a key factor limiting waterfowl populations during migration and winter (Miller 1986, Conroy et al. 1989, Reinecke et al. 1989, CVJV 2006), therefore providing adequate foraging habitat is a key objective to limit the possibility that survival outside of the breeding season limits population size.

A primary goal of the Wetland Reserve Program (WRP) for middle and southern latitudes of the US is to provide habitat for migratory birds during the non-breeding season. Seasonal wetlands are a common habitat being protected and restored on WRP easements in the Willamette Valley (WV) and Lower Columbia River Valley (LCRV) and they often produce abundant food resources (Fredrickson and Taylor 1982, Brasher et al. 2007, Kross et al. 2008). There are 2,448 hectares enrolled in WRP in the WV and 2,098 hectares in the LCRV. Results from chapter 2 indicate that the WRP is contributing to waterfowl conservation goals of the NAWMP by producing seed biomass that is similar to managed wetlands on public lands. Determining the relative importance of WRP habitat compared to other habitats in the region requires considering how the foods provided on WRP wetlands contribute to meeting the energy needs of waterfowl that winter in the WV and LCRV.

Bioenergetic models are a useful tool for directly linking waterfowl energy needs to waterfowl population objectives and habitat objectives (NAWMP Steering Committee 2007), and they are able to evaluate the regional contribution of conservation programs like the WPR towards meeting waterfowl energy demands (CVJV 2006, Dugger et al. 2008). Information required to run a bioenergetic model includes an estimate of daily energy requirements for individual birds, waterfowl population objectives, habitat acreage, and habitat foraging values (CVJV 2006). I used my seed biomass estimates and an estimated true metabolizable energy value for moist-soil plant seeds (Kaminski et al. 2003) to estimate energy supplies. The results of chapters 2 and 3 in this thesis have provided data that I can use to quantify the impact of the WRP for ducks wintering the Willamette Valley and LCRV. In this chapter, I estimated bioenergetic model input parameters and ran model simulations to assess how seasonal wetlands, and specifically seasonal wetlands on WRP sites, contribute towards meeting wintering waterfowl energetic needs in my study regions.

METHODS

I obtained daily energy requirements for individual birds using a waterfowl specific metabolic equation based on average body mass (Miller and Eadie 2006) for the dabbling duck species composition from mid-winter surveys. Dabbling ducks are a group of waterfowl that generally forage in shallow water, either by skimming the surface or tipping their heads and necks underwater. They mainly eat seeds and other plant matter during the non-breeding season therefore I only included dabbling ducks in model simulations (Baldassarre and Bolen 2006). Monthly dabbling duck population objectives were established for each study region by stepping down a mid-winter objective from the NAWMP and adjusting temporally based on migration chronology (Fleskes et al. 2000). The mid-winter objective was 394,317 for the WV and 384,168 for the LCRV. I estimated the amount of seasonal wetland habitat that occurred on WRP easements in the WV and LCRV by calculating the proportion of the easements I sampled in Chapters 2 and 3 that were seasonal wetland and multiplying that proportion by the total acreage of WRP easements in the WV and LCRV. I estimated the acreage of reference wetlands by totaling all wetlands in my population for both 2008 and 2009 (chapter 2) and averaged those years. I used mean seed biomass estimates for WRP and reference sites by study region (Chapter 2) as the measure of total food abundance including a value of 560 kg/ha for WRP sites in the Willamette Valley, and 188 kg/ha for WRP sites in the Lower Columbia River Valley, and 505 kg/ha for reference wetlands in both regions. Based on data from Chapter 3, I considered all seed that occurred above 5 cm deep in the soil as available to ducks as food. This included 93% of the total estimated biomass for each habitat. Lastly I subtracted 34 kg/ha from each estimate to account for a giving up density (Naylor 2002).

I used the bioenergetic model TRUEMET (CVJV 2006) to estimate current contributions of WRP and reference wetlands toward meeting migrating and wintering dabbling duck energetic demands in the WV and LCRV. Additionally, I simulated the scenario of all WRP and reference seasonal wetland acreage being actively managed (i.e., 762 kg/ha) versus all wetlands being unmanaged to assess the importance of seasonal wetland management on regional dabbling duck demands. I assumed WV and LCRV non breeding dabbling duck populations are currently at NAWMP goal and all estimated food resources were available. Dabbling ducks, in order of population size, included in the model were mallard, northern pintail, American wigeon, green-winged teal, northern shoveler (*Anas clypeata*), and gadwall (*Anas strepera)*. Understanding the nutritional value (i.e., metabolizable energy) of food produced is an important component to bioenergetic modeling, yet that value is only known for 20 species of wetland plant seeds (Dugger et al. 2007). Therefore, I used a general estimate of 2.5 kcal/g true metabolizable energy for all wetland plant species contributing to seed biomass estimates (Kaminski et al. 2003).

RESULTS

The percentage of seasonal wetlands on WRP easements in my study regions was 32 ± 3.22. Consequently, I estimated 783 hectares of seasonal wetlands on WRP easements in the WV and 234 hectares on easements in the LCRV. There was an average of 411 and 391 hectares of seasonal wetlands on reference sites in the WV and LCRV. Considering both wetland types, seasonal wetlands in the study regions constitute a small fraction of the historic pre-European settlement acres. Under current conditions, seasonal wetlands on WRP easements in the WV are providing 6.5% of dabbling duck food requirements, while seasonal wetlands on reference sites provide an additional 3.4%, totaling 10 percent of total energy demands. Seasonal wetlands on WRP easements in the LCRV are currently providing 0.64%, with reference wetlands in the LCRV providing another 2.9%, totaling 3.5% of energetic demands on 625 hectares. If all WRP and reference seasonal wetlands in both study regions were actively managed they would provide 11%

of energetic demands versus 2% if all wetlands were unmanaged. I estimated that 12,141 hectares of seasonal wetland habitat is needed to meet 100% of dabbling duck needs in the WV and 19,020 hectares would be needed to meet 100% of dabbling duck needs in the LCRV.

DISCUSSION

Bioenergetic model simulations suggest that dabbling duck energetic requirements are mostly being supplied in other habitat types such as agriculture, and riverine, estuarine, and lacustrine wetland systems. A large diversity of wetland habitats exist in the region, therefore management objectives should focus on a variety of habitat types. For example, only 10 of the 27 easements in the LCRV possessed my defined population of non-tidal, freshwater, seasonal wetlands located on privately owned WRP lands, yet they contained tidal freshwater and estuarine wetlands. A food density estimate does not exist for these habitats and they most likely produce less food resources than freshwater seasonal wetland due to the dominance of perennial plant species not associated with high seed production (PCJV 2010). Conservation planners may need to consider protecting a portion of those habitats in perpetuity to protect against increasing development in the region.

The contribution of managed seasonal wetlands towards meeting regional dabbling duck energetic demand may be less in areas where wintering waterfowl densities are lower, an abundance of unmanaged seasonal wetlands or other habitat types exist, or few managed wetlands exist. The goal in California's Central Valley is to have managed wetlands provide 50% of the regions energy demands and it is assumed agriculture meets rest. The Central Valley has 83,186 hectares of managed seasonal wetlands. Approximately two thirds of those wetlands are privately owned and landowners are often actively engaged in management as they are managed for waterfowl hunting (CVJV 2006). This reliance on managed seasonal wetlands may not be possible in western Oregon and southwest Washington.

Despite the relatively large amount of acreage enrolled in WRP recently, solely acquiring additional managed seasonal wetlands through the WRP to achieve incremental gains in carrying capacity in the region does not appear to be the most practical approach from a conservation planning perspective. Although, the cumulative efforts of other wetland restoration programs like the North American Wetlands Conservation Act, Ducks Unlimited, US Fish and Wildlife Service's Partners for Fish and Wildlife Program, Oregon Watershed Enhancement Board along with WRP may achieve significant wetland habitat gains, which should equate to waterfowl populations exceeding NAWMP goals. Despite a smaller reliance on managed wetlands in the WV and LCRV habitat restoration on private lands is the most feasible way to significantly increase wetland acreage. State and federal refuges are relatively established, making future acquisition less likely on public lands. WRP seasonal wetlands are contributing nearly twice as much to energetic demands of WV wintering dabbling ducks as publicly managed lands in the WV largely due to differences in the amount of habitat versus forage produced. WRP is also protecting or restoring a considerable amount of tidal wetlands and wet-prairie that is also providing food resources for ducks. WRP seasonal wetlands are not exclusively contributing to meeting dabbling duck energetic demands, yet these managed habitats serve a critical role for wintering dabbling ducks because they provide consistent foraging habitat, an abundance of natural foods which contain essential nutrients that agriculture foods do not (Fredrickson and Taylor 1982).

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Chapter 5

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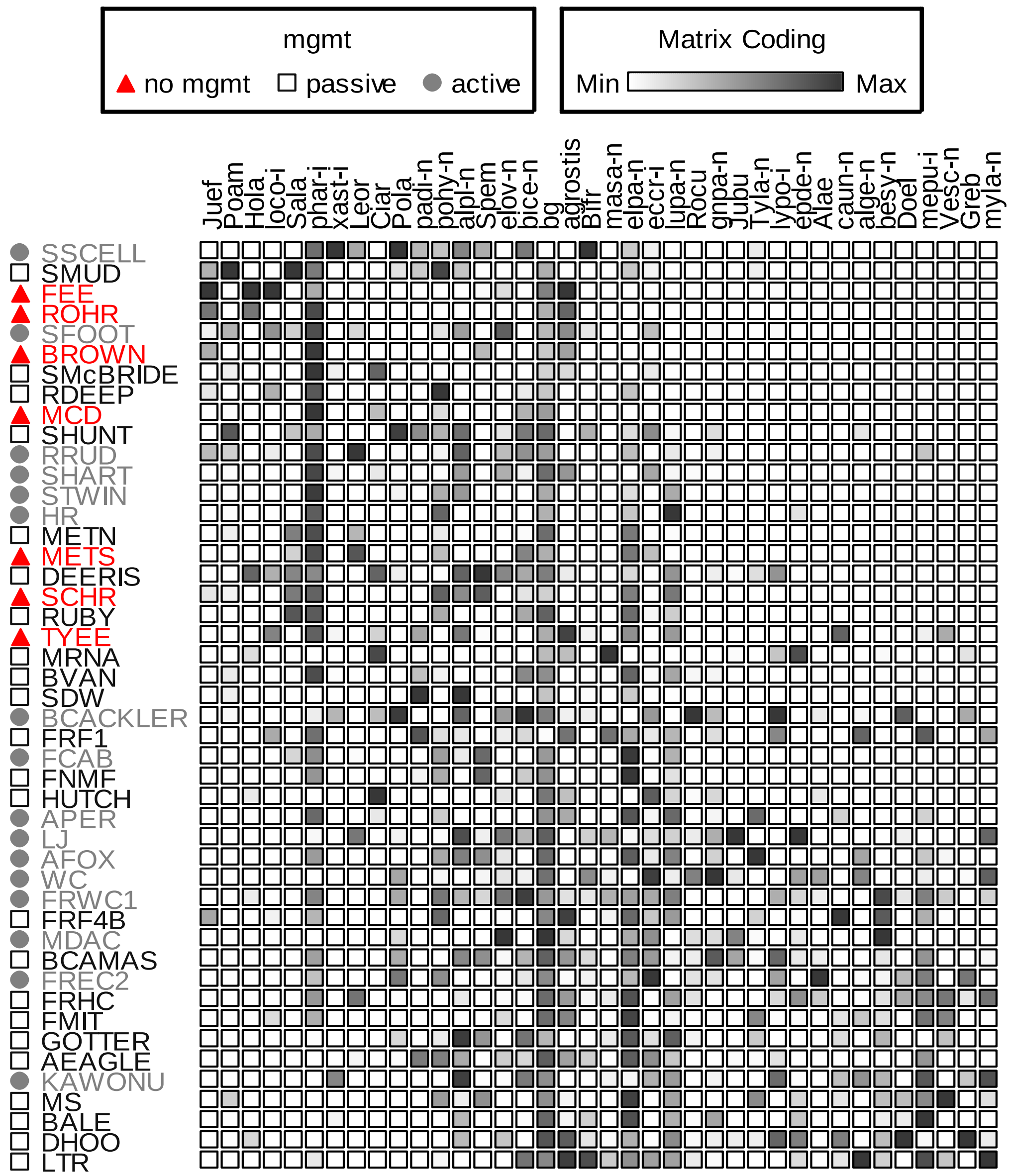
APPENDICES

Appendix I

An ordered main matrix produced from Non-Metric Scaling analysis showing similar plant communities when wetlands are grouped by management intensity. Species absent in eight wetlands or less were removed for graphical display (75 species) and cells in the matrix represent relative abundances of species in a wetland. Perennial and upland species were more abundant in unmanaged wetlands whereas annuals were more abundant in actively managed wetlands.



Appendix II

Mean percent cover (± SE) of plant genera/species in seasonal wetlands on Wetland Reserve Program easements (n = 23) and reference wetlands located on publicly managed lands (n = 23) in the Willamette Valley and Lower Columbia River Valley, fall 2008 and 2009.

Genus/species	Common name	Reference	WRP
Phalaris arundinacea	Reed canary-grass	26.03 ± 5.71	22.54 ± 5.97
Eleocharis palustris	Creeping spike-rush	17.53 ± 4.67	10.62 ± 3.09
Bare ground		10.87 ± 1.50	14.61 ± 2.32
Polygonum hydropiperoides	Waterpepper	7.31 ± 2.65	2.45 ± 1.22
Alisma plantago-aquatica	Water-plantain	6.18 ± 2.05	6.66 ± 2.52
Paspalum distichum	Knotgrass	4.58 ± 2.42	0.20 ± 0.20
Echinochloa crus-galli	Barnyard grass	3.59 ± 1.96	3.79 ± 1.89
Mentha pulegium	Pennyroyal	2.85 ± 1.11	1.91 ± 0.88
Ludwigia palustris	False loosestrife	2.63 ± 0.83	4.89 ± 1.83
Polygonum lapathifolium	Curlytop knotweed	2.41 ± 1.11	0.29 ± 0.15
Agrostis spp.	Bentgrass	1.87 ± 0.75	3.58 ± 1.20
Lythrum portula	Spatulaleaf loosestrife	1.69 ± 1.13	1.76 ± 0.80
Eleocharis ovata	Ovate spike-rush	1.44 ± 0.65	1.87 ± 1.14
Polygonum amphibium	Water smartweed	1.39 ± 0.86	0.07 ± 0.04
Bidens frondosa	Common beggarticks	1.30 ± 1.02	1.06 ± 0.73
Bidens cernua	Nodding beggarticks	1.25 ± 0.46	0.90 ± 0.26
Typha latifolia	Cattail	1.09 ± 0.64	0.25 ± 0.17
Xanthium strumarium	Common cocklebur	1.01 ± 0.89	0.25 ± 0.24
Scirpus tabernaemontani/acutus	Soft/hardstem bulrush	0.88 ± 0.47	0.00
Sagittaria latifolia	Wapato, Arrowhead	0.60 ± 0.49	0.86 ± 0.40
Alopecurus aequalis	Short-awned foxtail	0.58 ± 0.56	0.18 ± 0.12
Sparganium emersum	Simple-stem bur-reed	0.46 ± 0.22	0.74 ± 0.32
Juncus effusus	Common rush	0.37 ± 0.19	1.56 ± 1.10
Gnaphalium palustre	W. marsh cudweed	0.37 ± 0.15	1.55 ± 0.96
Beckmannia syzigachne	Slough grass	0.37 ± 0.24	0.44 ± 0.23
Leersia oryzoides	Rice cut-grass	0.33 ± 0.25	0.46 ± 0.23
Alopecurus geniculatus	Water foxtail	0.32 ± 0.20	0.56 ± 0.36
Lotus corniculatus	Bird's-foot trefoil	0.29 ± 0.14	0.60 ± 0.41
Downingia elegans	Downingia	0.25 ± 0.19	0.42 ± 0.31
Madia sativa	Coast tarweed	0.25 ± 0.19	0.54 ± 0.40
Veronica scutellata	Skullcap speedwell	0.23 ± 0.18	0.98 ± 0.60
Juncus tenuis	Slender rush	0.21 ± 0.19	0.17 ± 0.10
Rorippa curvisiliqua	Western yellowcress	0.18 ± 0.16	0.13 ± 0.07
Alopecurus pratensis	Meadow foxtail	0.17 ± 0.17	0.63 ± 0.55
Gratiola ebracteata	Bractless hedgehyssop	0.17 ± 0.12	0.29 ± 0.23
Eleocharis acicularis	Needle spike-rush	0.15 ± 0.11	0.27 ± 0.17
Carex unilateralis	One-sided sedge	0.13 ± 0.11	0.17 ± 0.09

Appendix II (Continued)

Genus/species	Common name	Reference	WRP
Zizania spp.	Wild rice	0.13 ± 0.13	0.00
Polygonum spp.	Smartweed	0.11 ± 0.11	0.01 ± 0.01
Rumex crispus	Curly dock	0.11 ± 0.07	0.03 ± 0.02
Myosotis laxa	Forget-me-not	0.07 ± 0.06	0.79 ± 0.32
Cyperus strigosus	False nutsedge	0.06 ± 0.04	0.00
Trifolium spp.	Clover	0.06 ± 0.04	0.55 ± 0.50
Epilobium densiflorum	Dense spike-primrose	0.06 ± 0.04	0.81 ± 0.34
Juncus spp.	Rush	0.06 ± 0.06	0.00
Juncus acuminatus	Taper-tipped rush	0.06 ± 0.04	0.08 ± 0.05
Cuscuta spp.	Dodder	0.05 ± 0.04	0.00
Epilobium ciliatum	Watson's willow-herb	0.04 ± 0.04	0.09 ± 0.07
Cirsium vulgare	Bull thistle	0.04 ± 0.04	0.06 ± 0.04
Festuca spp.	Fescue	0.03 ± 0.03	0.39 ± 0.24
Equisetum arvense	Field horsetail	0.03 ± 0.03	0.00
Malva neglecta	Common mallow	0.03 ± 0.03	0.00
Juncus articulatus	Jointed rush	0.02 ± 0.02	0.01 ± 0.01
Carex feta	Green-sheathed sedge	0.02 ± 0.02	0.01 ± 0.01
Plantago major	Broadleaf plaintain	0.02 ± 0.02	0.08 ± 0.07
Cirsium arvense	Canada thistle	0.02 ± 0.01	0.05 ± 0.02
Holcus lanatus	Velvet grass	0.02 ± 0.01	1.22 ± 0.69
Carex densa	Dense sedge	0.01 ± 0.01	0.26 ± 0.18
Juncus oxymeris	Pointed rush	0.01 ± 0.01	0.00
Triteleia hyacinthina	Hyacinth brodiaea	0.01 ± 0.01	0 .00
Anthemis cotula	Dog fennel	0.01 ± 0.01	1.23 ± 1.07
Veronica americana	American speedwell	0.01 ± 0.01	0.10 ± 0.06
Equisetum fluviatile	Water horsetail	0.01 ± 0.01	0.03 ± 0.02
Taraxacum officinale	Dandelion	0.01 ± 0.01	0 .00
Vicia spp.	Vetch	0.00	0.03 ± 0.02
Briza minor	Little quacking grass	0.00	0.03 ± 0.03
Daucus carota	Queen Anne's lace	0.00	0.41 ± 0.40
Glyceria spp.	Managrass	0.00	0.08 ± 0.06
Juncus bufonius	Toad rush	0.00	0.79 ± 0.52
Lotus spp.	Trefoil	0.00	0.02 ± 0.02
Sonchus spp.	Sowthistle	0.00	0.37 ± 0.37
Deschampsia cespitosa	Tufted hairgrass	0.00	0.14 ± 0.10
Grindelia integrifolia	Gumweed	0.00	0.25 ± 0.19
Kickxia elatine	Fluvellin	0.00	1.05 ± 0.95
Anagallis arvensis	Scarlet pimpernel	0.00	0.02 ± 0.02
Avena fatua	Wild oat	0.00	0.02 ± 0.02
Carex aperta	Columbia sedge	0.00	0.60 ± 0.48
Carex obnupta	Slough sedge	0.00	0.05 ± 0.03
Centaurium erythraea	European centuary	0.00	0.10 ± 0.09
Cerastium glomeratum	Sticky chickweed	0.00	0.04 ± 0.04
Cicuta douglasii	W. water-hemlock	0.00	0.12 ± 0.12

Appendix II (Continued)

Genus/species	Common name	Reference	WRP
Cynodon dactylon	Bermudagrass	0.00	0.08 ± 0.08
Digitaria sanguinalis	Large crabgrass	0.00	0.62 ± 0.62
Epilobium brachycarpum	Tall annual willowherb	0.00	0.04 ± 0.04
Galium parisiense	Wall bedstraw	0.00	0.16 ± 0.16
Galium trifidium	Small bedstraw	0.00	0.01 ± 0.01
Hordeum brachyantherum	Meadow barley	0.00	0.11 ± 0.11
Hypochaeris radicata	False dandelion	0.00	0.23 ± 0.15
Lactuca serriola	Prickly lettuce	0.00	0.13 ± 0.09
Lolium multiflorum	Italian ryegrass	0.00	0.03 ± 0.03
Lotus purshianus	Spanish clover	0.00	0.40 ± 0.40
Lythrum hyssopifolium	Hyssop loosestrife	0.00	0.07 ± 0.05
Mentha spp.	Mint	0.00	0.08 ± 0.08
Navarretia Squarrosa	Skunkbush	0.00	0.02 ± 0.02
Nuphar lutea spp. *polysepala*	Yellow pond-lily	0.00	0.01 ± 0.01
Panicum miliaceum	Broomcorn millet	0.00	0.02 ± 0.02
Panicum spp.	Panicgrass	0.00	0.04 ± 0.03
Plagiobothrys figuratus	Fragrant popcornflower	0.00	0.07 ± 0.05
Poaceae spp.	Unknown grass	0.00	0.20 ± 0.20
Polygonum aviculare	Prostrate knotweed	0.00	0.04 ± 0.04
Ranunculus repens	Creeping buttercup	0.00	0.47 ± 0.38
Ranunculus sceleratus	Celery-leaf buttercup	0.00	0.01 ± 0.01
Rumex acetosella	Sour dock	0.00	0.03 ± 0.03
Rumex salicifolius	Willow dock	0.00	0.04 ± 0.04
Scirpus microcarpus	Small-fruited bulrush	0.00	0.33 ± 0.24
Trifolium vesiculosum	Arrowleaf clover	0.00	0.96 ± 0.96
Verbascum blattaria	Moth mullein	0.00	0.01 ± 0.01
Salix spp.	Willow	0.00	0.00
Convulvulus arvensis	Field bindweed	0.00	0.00
Geum macrophyllum	Largeleaf avens	0.00	0.00
Linaria vulgarus	Yellow toadflax	0.00	0.00
Rosa nutkana	Nootka rose	0.00	0.00

Appendix III

Information on seed production (mean [SE] dry biomass in kg/ha) and management intensity ranking (MI) for each wetland sampled in the Willamette Valley and Lower Columbia River Valley on Wetland Reserve Program easements and reference sites during fall 2008 and 2009. Wetland units within WRP easements were not named. Wetlands were categorized as unmanaged (U) if no restoration, enhancement, or water control infrastructure existed. Passively managed wetlands (P) were defined as those that had received no active management in at least three years; whereas, wetlands that had been managed more recently were categorized as actively managed (A).

Wetland Type	Region	Site	Unit	x	SE	MI
Reference	Willamette Valley	Fern Ridge WMA	Field 1	241	44	P
			Field 4B	188	48	P
			W. Coyote 1	588	111	A
			E. Coyote 2	1566	252	A
		Finley NWR	Mitigation Pond	218	36	P
			S. Cabell	727	163	A
			McFadden	141	27	P
		Ankeny NWR	Peregrine	301	39	A
			Foxtail	451	63	A
			Eagle	372	71	P
		Basket Slough NWR	Aleutian	349	54	P
			Vancouver	112	28	P
			Cackler	908	122	A
	Lower Columbia River Valley	Sauvie Island WMA	Hunt	1200	310	P
			Deadwillow	857	229	P
			Mudhen S.	586	96	P
			Footbridge 2	1355	384	A
		Shilapoo WMA	Twin Ponds	189	92	A
			Hart	221	94	A
			S. McBride	222	41	P
			S. Cell	593	92	A
		Ridgefield NWR	Ruddy	707	201	A
			Deep	263	48	P
Wetland Reserve Program	Willamette Valley	Kawanu Acres		1275	378	A
		Long Tom Ranch		99	31	P
		Fern Ridge Hunt Club		216	24	P
		MDAC Farm		933	210	A
		Tyee		685	157	U
		Mary’s River		917	126	P
		Winter Creek		648	90	A
		Blue Camas		542	182	P

Appendix III (Continued)

Wetland Type	Region	Site	Unit	x	SE	MI
Wetland	Willamette	Gotter N.		748	93	P
Reserve	Valley	Bridgeport Farm		244	51	P
Program		Mud Slough		464	53	P
		Hutch		617	114	P
		Lovejoy			273	A
				1540		
	Lower	Ruby		209	85	P
	Columbia	Metro North		74	22	P
	River	Metro South		113	33	U
	Valley	Multnomah		61	18	U
		Channel Dairy				
		Hogan Ranch		199	37	A
		Deer Island		515	128	P
		Brown		134	17	U
		Fee		272	99	U
		Schriber		279	88	U
		Rohr		103	21	U

Appendix IV

Mean percent biomass (±SE) of plant genera/species in seasonal wetlands on Wetland Reserve Program (WRP) easements (*n* = 23) and reference wetlands located on publicly managed lands (*n* = 23) in the Willamette Valley and Lower Columbia River Valley, fall 2008 and 2009.

Genus/species	Common name	Reference	WRP
Alisma plantago-aquatica	American water-plantain	22.24 ± 1.58	36.11 ± 1.79
Eleocharis palustris	Creeping spike-rush	13.25 ± 1.49	6.84 ± 1.51
Echinochloa crus-galli	Barnyard grass	9.67 ± 3.26	9.99 ± 2.15
Eleocharis ovata	Ovate spike-rush	6.36 ± 2.05	7.67 ± 2.86
Lythrum portula	Spatulaleaf loosestrife	8.23 ± 2.20	2.77 ± 1.97
Polygonum lapathifolium	Curlytop knotweed	7.38 ± 2.04	0.34 ± 2.38
Phalaris arundinacea	Reed canary-grass	4.59 ± 1.55	2.96 ± 1.61
Scirpus tabernaemontani/acutus	Soft/Hardstem bulrush	6.21 ± 3.79	0.05 ± 2.31
Polygonum hydropiperoides	Waterpepper	4.36 ± 2.24	0.49 ± 2.01
Unknown spp.	Unknown	1.28 ± 1.70	3.34 ± 1.71
Bidens cernua	Nodding beggarticks	2.53 ± 2.21	0.44 ± 1.69
Cyperus strigosus	False nutsedge	2.60 ± 3.72	0.27 ± 3.01
Anthemis cotula	Dog fennel	0.15 ± 4.12	2.69 ± 4.18
Ludwigia palustris	False loosestrife	1.27 ± 1.82	1.38 ± 2.01
Downingia elegans	Downingia	0.71 ± 1.95	1.90 ± 2.26
Beckmannia syzigachne	Slough grass	0.89 ± 3.07	1.63 ± 1.93
Sparganium emersum	Simple-stem bur-reed	0.36 ± 2.51	1.94 ± 3.86
Trifolium vesiculosum	Arrowleaf clover	0.00	1.93 ± 4.35
Polygonum spp.	Smartweed	0.00	1.88 ± 4.34
Epilobium spp.	Willow-herb	0.70 ± 1.30	0.95 ± 1.96
Polygonum amphibium	Water smartweed	0.00± 4.35	1.60 ± 3.51
Mentha pulegium	Pennyroyal	0.79 ± 2.25	0.71 ± 2.27
Holcus lanatus	Velvet grass	0.82 ± 2.93	0.57 ± 2.55
Eleocharis acicularis	Needle spike-rush	0.37 ± 4.35	1.01 ± 3.31
Juncus spp.	Rush	0.97 ± 4.06	0.32 ± 2.22
Myosotis laxa	Forget-me-not	0.11 ± 2.61	1.21 ± 2.41
Paspalum distichum	Knotgrass	1.02 ± 2.11	0.13 ± 2.95
Sagittaria latifolia	Wapato, Arrowhead	0.88 ± 2.98	0.27 ± 2.60
Juncus effusus	Common rush	0.28 ± 2.67	0.59 ± 2.75
Galium parisiense	Wall bedstraw	0.00	0.89 ± 4.35
Veronica americana	American speedwell	0.02 ± 2.54	0.84 ± 3.64
Epilobium densiflorum	Dense spike-primrose	0.01 ± 3.57	0.77 ± 2.75
Lotus corniculatus	Bird's-foot trefoil	0.42 ± 3.08	0.28 ± 4.27
Carex spp.	Sedge	0.21 ± 2.51	0.47 ± 4.10
Alopecurus aequalis	Short-awned foxtail	0.27 ± 2.58	0.22 ± 3.79
Lotus purshianus	Spanish clover	0.00	0.51 ± 4.35
Kickxia elatine	Fluvellin	0.00	0.46 ± 4.03

Appendix IV (Continued)

Genus/species	Common name	Reference	WRP
Madia sativa	Coast tarweed	0.01 ± 3.02	0.40± 4.02
Leersia oryzoides	Rice cut-grass	0.16 ± 2.74	0.22 ± 3.55
Scirpus microcarpus	Small-fruited bulrush	0.00 ± 4.35	0.38 ± 3.99
Gnaphalium palustre	Western marsh cudweed	0.09 ± 2.63	0.27 ± 2.49
Juncus bufonius	Toad rush	0.01 ± 2.90	0.34 ± 2.70
Alopecurus pratensis	Meadow foxtail	0.07 ± 1.81	0.23 ± 2.18
Hordeum brachyantherum	Meadow barley	0.00	0.26 ± 4.35
Veronica scutellata	Skullcap speedwell	0.13 ± 4.27	0.11 ± 3.21
Rorippa curvisiliqua	Western yellowcress	0.16 ± 3.14	0.02 ± 3.07
Rumex spp.	Dock	0.01 ± 4.35	0.18 ± 3.54
Bidens frondosa	Common beggarticks	0.09 ± 2.54	0.05 ± 2.30
Daucus carota	Queen Anne's lace	0.00	0.15 ± 4.06
Gratiola ebracteata	Bractless hedgehyssop	0.07 ± 2.54	0.07 ± 3.81
Carex unilateralis	One-sided sedge	0.07 ± 2.95	0.08 ± 2.91
Sonchus spp.	Sowthistle	0.01 ± 3.58	0.13 ± 4.35
Panicum spp.	Panicgrass	0.00 ± 4.21	0.13 ± 3.73
Lolium multiflorum	Italian ryegrass	0.00	0.11 ± 3.86
Juncus articulatus	Jointed rush	0.05 ± 4.35	0.05 ± 4.35
Lythrum hyssopifolium	Hyssop loosestrife	0.00 ± 4.35	0.09 ± 4.33
Centuarium spp.	Centuary	0.06 ± 4.35	0.01 ± 3.62
Plantago major	Broadleaf plaintain	0.00	0.09 ± 4.35
Carex densa	Dense sedge	0.00	0.04 ± 3.02
Zizania spp.	Wild rice	0.01 ± 4.35	0.02 ± 4.35
Cerastium glomeratum	Sticky chickweed	0.00	0.03 ± 4.35
Epilobium ciliatum	Watson's willow-herb	0.00	0.03 ± 3.17
Trifolium spp.	Clover	0.00	0.03 ± 4.35
Clarkia amoena	Farewell to spring	0.00	0.02 ± 4.35
Juncus acuminatus	Taper-tipped rush	0.01 ± 3.01	0.01 ± 4.12
Ranunculus spp.	Buttercup	0.02 ± 4.35	0.00
Navarretia squarrosa	Skunkbush	0.00	0.01 ± 4.16
Digitaria sanguinalis	Large crabgrass	0.00	0.01 ± 4.35
Juncus tenuis	Slender rush	0.00 ± 4.35	0.00 ± 4.35
Anagallis arvensis	Scarlet pimpernel	0.00	0.00 ± 4.35
Vicia spp.	Vetch	0.00	0.00 ± 4.35
Panicum miliaceum	Broomcorn millet	0.00	0.00 ± 4.35
Cynodon dactylon	Bermudagrass	0.00	0.00 ± 4.35
Carex aperta	Columbia sedge	0.00 ± 4.35	0.00
Taraxacum officinale	Dandelion	0.00	0.00 ± 4.35
Rumex crispus	Curly dock	0.00	0.00 ± 4.35