AN ABSTRACT OF THE DISSERTATION OF

Sarah M. Marshall for the degree of Doctor of Philosophy in Water Resources Engineering presented on September 16, 2011.
Title: The Effects of Land Use on Mineral Flat Wetland Hydrologic Processes in Lowland Agricultural Catchments

Abstract approved: ____
Desireé D. Tullos

Hydrologic processes within mineral flat wetlands, along with their connections to groundwater and downstream surface water in lowland agricultural catchments are poorly understood, particularly under different land uses. In the three field studies included in this thesis, we examined infiltration, wetland hydroperiod, groundwater recharge dynamics, surface runoff generation, and water quality in mineral flat wetlands using a combination of soil and hydrometric measurements, stable isotope tracers, and water chemistry analysis. Our overarching objectives were to examine, for mineral flat wetlands under native prairie, farmed grassland, and restored prairie land cover: 1) how different land management influences infiltration and wetland hydroperiod at the plot scale, 2) the effects of land use on seasonal groundwater-surface water dynamics at the field scale, and 3) seasonal variation in runoff sources and nutrient transport from native prairie and farmed wetlands at the small catchment scale.

At the plot scale, our results suggest that edaphic factors, particularly those related to soil structure, are strongly associated with wetland infiltration and overall hydroperiod across least-altered prairie, farmed, and restored prairie mineral flat wetlands. The hydroperiod metrics we examined were generally more sensitive to level of site disturbance than land use alone. At the field scale, our results indicate that, in spite of land use differences and slight variations in soil stratigraphy, many similarities exist in overall wetland hydroperiod, water sources and evaporation rates for mineral flat wetlands in the Willamette Valley lowlands. Isotopic evidence

suggests that the greatest degree of groundwater-surface water mixing occurs in the upper 0.5 m of the saturated soil profile across sites under all land uses. Finally, at the small catchment scale, farmed wetland runoff was isotopically similar to field surface water for most of the wet season, indicating that saturation excess was an important runoff generation process. Prairie wetland runoff was isotopically similar to upstream water throughout the winter, and briefly similar to shallow groundwater and surface water within the wetland in mid-spring. Throughout the wet season, elevated nitrate, sulfate, and chloride concentrations were observed in groundwater and surface water at the farm site, and deeper groundwater at the prairie site. Upstream-downstream runoff chemistry remained similar throughout the wet season at the prairie site. Farm site runoff chemistry reflected the dominant water source within the farm field throughout the wet season. Our findings suggest that, while surface water pathways dominate runoff from wetland flats under farm land use, large wetland flat fields have a high potential to absorb, store, and process nutrients and agrochemicals from on-site and nearby off-site chemical inputs.

Mineral flats that maintain wetland hydrology in spite of farm use represent a unique balance between agricultural production and preservation of some of the water storage and delay, and water quality-related ecosystem services once provided at a much larger scale in the Willamette Valley lowlands. We anticipate that results of this work will lead to better understanding of key site-scale edaphic and hydrologic factors to consider when prioritizing and managing sites for restoration, and how site disturbance under a variety of land uses may impact different hydrologic processes and components of the wetland hydroperiod. Additionally, our results provide a better understanding of how land use affects seasonal runoff generation processes in mineral flat wetlands, and the water quality implications of modifying groundwater and surface water connectivity between mineral flats and surrounding surface drainage networks.

The Effects of Land Use on Mineral Flat Wetland Hydrologic Processes in Lowland Agricultural Catchments

by

Sarah M. Marshall

A DISSERTATION

Submitted to

Oregon State University

in partial fulfillment of

the requirements for the

degree of

Doctor of Philosophy

Presented September 16, 2011

Commencement June 2012

Doctor of Philosophy dissertation of Sarah M. Marshall presented on September 16, 2011.

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

Sarah M. Marshall, Author

ACKNOWLEDGEMENTS

Countless people have enriched my academic experience over the last five years. From the very beginning, my advisor Desireé Tullos has believed in my abilities as a researcher, and wholeheartedly supported my academic pursuits. Desireé has a brilliant mind, high standards, and unshakeable ethics, and I am grateful to have her for a mentor, friend, and role model. She has also always acknowledged the importance of balancing hard work with a well-rounded life, and I am deeply appreciative of her encouraging my outdoor activities and community service work outside of graduate school.

I am proud to be part of the Water Resources Graduate Program at Oregon State University, and my academic home in the Department of Biological and Ecological Engineering. Dr. Mary Santelmann, Dr. John Bolte, Elena Maus, Susan Atkisson have been truly supportive throughout my graduate school experience, and I am fortunate to know them. I am grateful for three years of financial support from a National Science Foundation Graduate Research Fellowship, and to Mary Strickroth for administering my fellowship. A University Graduate Laurels Scholarship and John & Matha Wolfe Memorial Scholarship were also tremendously helpful in providing assistance during the final stretch of graduate school.

Throughout my academic career, many teachers, past and recent, have contributed to my knowledge and enthusiasm about wetlands, hydrology, soil, and aquatic ecosystems. Al James, my 7th grade science teacher; Betsy Halpern at South Eugene High School; Dr. Dennis Todd at the University of Oregon; Craig Cornu at the South Slough National Estuarine Research Reserve; Dr. Janet Hodder and Dr. Richard Emlet at the Oregon Institute for Marine Biology; James Cassidy, Dr. Jon (Jack) Istok, Dr. John Selker, Dr. Richard Cuenca, Dr. Jeff McDonnell, Dr. Stan Gregory at Oregon State University are a few of these wonderful people.

As I transitioned to the field portion of my study, I met and interacted with many wonderful people who give me hope about the future of wetlands in the Willamette Valley. Jarod Jebousek, Jock Beall, and Molly Monroe from USFWS;

Paul Gordon, Diane Steeck, and Trevor Taylor from the City of Eugene; Jason Nuckols from the Nature Conservancy of Oregon; and Wes Messinger from the US Army Corps of Engineers were all extremely helpful in locating and accessing research sites, and taught me a great deal about wetland prairie conservation and restoration. Chuck Stolsig and Tom Danelski taught me about farming in the Willamette Valley—both historic and current—and granted me access to wetlands on private land. Conversations with my committee members Dr. Mary E. Kentula and Dr. Jeff McDonnell, along with Dr. Jim Wigington, Dr. Brian Wood, Dr. Jack Istok, Dick Novitzki, and Carin Baitis were invaluable in informing my study design.

I was fortunate to have a cadre of wonderful volunteer family members and friends for companionship and assistance during two years of field work. My parents, Jim and Laura Marshall, were my most reliable and regular field companions, and spent countless hours helping me build, install, and monitor piezometers, conduct infiltration tests, and collect soil samples. My friend Alicia Omlid also helped me install a number of piezometers, and monitor them throughout the 2009-2010 field season. Friends Leah Wilson and Diana Sharps helped me with various field tasks, and I enjoyed hearing their fresh perspectives on wetlands and water in the landscape. Fellow lab mates Cara Walter and Matt Cox were extremely helpful in training me to use and troubleshoot our lab's field equipment—sometimes after normal work hours—and have provided much technical assistance over the years.

My friend and committee member Dr. Scott Bridgham was incredibly helpful in sharing his wealth of knowledge about wetlands, providing feedback on field and lab techniques related to wetland biogeochemistry, and loaning me lab space and field equipment. Kathy Motter, lab manager at the Institute for Water and Watersheds Collaboratory, provided invaluable guidance in water quality sampling and analysis, taught me how to safely and efficiently collect and process my samples, helped me troubleshoot numerous field-related challenges, and was absolutely wonderful to visit with. Caroline Patrick and Tina Garland were incredibly helpful with my isotope processing effort, and negotiating with temperamental lab equipment. Sam Henkels and Vincent Likewise assisted me in processing numerous soil samples in the lab, and I could not have completed this work without their help.

Finally, and most importantly, my partner Mark Kacmarcik has loved and supported me throughout my entire graduate school experience. I met Mark before coming to OSU, and our adventures backcountry skiing, backpacking, kayaking, and canoeing in the Pacific Northwest and beyond have sustained my happiness and sanity throughout the past five years. I am grateful to have such a wonderful companion to keep me in good spirits and health.

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CHAPTER 1. INTRODUCTION

Land use change in the United States during the last four centuries has led to the conversion, fragmentation, and loss of over half of pre-European settlement wetland area (Dahl and Allord 2004). Lowland prairie, savanna, forest, and emergent marsh wetland ecosystems have been especially impacted by drainage for agricultural land use (Dahl and Allord 2004). Mineral flat wetlands, a hydrogeomorphic class of wetland located on nearly flat floodplain terraces, interfluves, and large former lake bottoms (Brinson et al. 1995), were regularly converted to agricultural and silvicultural use due to their flat topography and seasonal drying cycles. Following centuries of fire suppression and land use conversion, least-disturbed mineral flat wetlands, including the wet pine flatwoods, pine savanna, and hardwoods of the Gulf Atlantic Coastal Plain and wetland prairie in the Willamette Valley, Oregon, are now some of the rarest ecosystems in their respective regions (PNW-ERC 2002; Rheinhardt et al. 2002; PNW-ERC 2002).

While land use change has converted many mineral flats to uplands, some flats maintain wetland hydrology under agricultural (Taft and Haig 2003) or silvicultural (Sun et al. 2001; Xu et al. 2002) land use. Several studies have examined the hydrology of flats managed for timber production (McCarthy et al. 1991; Sun et al. 1998; Sun et al. 2001; Xu et al. 2002) and in urban areas (Ehrenfeld et al. 2003; Stander and Ehrenfeld 2009), but we currently lack information on how farmed mineral flats compare with their least-altered counterparts in terms of hydrology and water quality (Rheinhardt et al. 2002). Farmed flats that maintain wetland hydrology, in spite of drainage ditches and other local hydrologic modifications, may a) store and delay wet-season surface runoff as seasonally farmed wetlands, or b) be targeted by state, federal, and private land stewards for restoration to native wetland plant communities (De Steven and Lowrance 2011). In light of efforts to conserve and restore wetlands at the catchment scale, and better understand the hydrologic and biogeochemical ecosystem services provided by wetlands under a variety of land uses, there is a need investigate the processes governing how mineral flat wetlands affect soil and groundwater recharge, and downstream water quality and quantity in lowland agricultural catchments.

In the three studies included in this dissertation, we examined the overall resiliency of wetland soil structure and hydrology to anthropogenic disturbance from farming and restoration activities. Beginning at the plot scale, and moving to the small catchment scale, we addressed key hydrologic processes influencing wetland hydroperiod, groundwater recharge, surface runoff generation, and water quality using a combination of soil and hydrometric measurements, stable isotope tracers, and water chemistry analysis. Our overarching objectives were to examine, for mineral flat wetlands under native prairie, farmed grassland, and restored prairie land cover: 1) how different land management influences infiltration and wetland hydroperiod at the plot scale, 2) the effects of land use on seasonal groundwater-surface water dynamics at the field scale, and 3) seasonal variation in runoff sources and nutrient transport from native prairie and farmed wetlands at the small catchment scale.

We anticipate that results of this work will lead to a better understanding of how land use affects key site-scale edaphic and hydrologic factors that determine how mineral flat wetlands absorb and store precipitation inputs during the wet season. Additionally, our results provide a better understanding of how land use affects seasonal runoff generation processes in mineral flat wetlands, and the water quality implications of modifying groundwater and surface water connectivity between mineral flats and surrounding surface drainage networks. Such knowledge is critical in evaluating ecosystem services provided by existing wetlands, prioritizing and managing sites for restoration, and predicting future catchment hydrologic responses to stressors such as land use and climate change.

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CHAPTER 2. EFFECTS OF LAND USE ON INFILTRATION AND HYDROPERIOD WITHIN GRASSLAND MINERAL FLAT WETLANDS IN LOWLAND AGRICULTURAL CATCHMENTS

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To be submitted for publication

Abstract

Mineral flat wetlands in an agricultural landscape may exist under a variety of land uses ranging from native prairie and forest to restored and farmed grassland. Our knowledge of ecological functions related to specific components of the mineral flat wetland hydroperiod is growing, but relationships between mineral flat hydrology and edaphic, plant cover, and microtopographic variables remain poorly understood, especially for grassland flats under different land uses. In this study, we compared least-altered prairie, farmed, and restored prairie mineral flat wetlands in Oregon's Willamette Valley with the objectives of 1) characterizing the hydroperiod and soil structure of mineral flat wetlands under different land uses, and 2) identifying plot-scale edaphic, plant cover, and microtopographic variables that are most strongly associated with infiltration rates and hydroperiod metrics across land use. Our results suggest that edaphic factors, particularly those related to soil structure, are strongly associated with wetland infiltration and overall hydroperiod across least-altered prairie, farmed, and restored prairie mineral flat wetlands. The hydroperiod metrics we examined were generally more sensitive to level of site disturbance than land use alone.

Introduction

Hydrology is considered to be the most important factor structuring wetland soils and biotic communities (NRC 1996; Mitsch and Gosselink 2000; Fennessy et al. 2007). Soil structure and hydraulic properties can, in turn, influence wetland hydrology, and are often impacted by anthropogenic disturbance associated with farming, restoration, and creation of wetlands (e.g. Galatowitsch 1996; Campbell et al. 2002; Bruland and Richardson 2005; Ullah and Faulkner 2006). While soil hydraulic properties are closely tied to wetland hydroperiod, groundwater recharge, plant community structure, and nutrient cycling (e.g. Richardson and Vepraskas 2001; Rheinhardt et al. 2002), relationships between soil variables and wetland hydroperiod across different land uses are poorly understood. In many common wetland assessment methodologies, soil hydraulic properties are neglected altogether (Fennessy et al. 2007).

With growing interest in restoring degraded wetlands in agricultural landscapes (e.g. De Steven and Lowrance 2001; Tweedy and Evans 2001; Zedler 2003), there is a need to better understand how key hydrologic processes, including infiltration, and wetland hydroperiod are affected by different types of land use. Of particular importance in de-coupling relationships between land use and wetland hydrology are seasonal, grassland mineral flat wetlands in lowland agricultural catchments. Mineral flats are hydrogeomorphically unique type of wetland that may maintain wetland hydrology during the wet season while being farmed during the dry season (e.g. Taft and Haig 2003; De Steven and Lowrance 2011). Flats have nearly level topography, receive the majority of their hydrologic inputs from precipitation, and often have a low-permeability soil or rock layer that restricts downward movement of water into the soil profile (Brinson et al. 1995; Rheinhardt et al. 2002). While mineral flats are easily drained by closely spaced surface ditches and subsurface tile drainage systems (e.g. McCarthy et al. 1991; Tweedy and Evans 2001), their water source and stratigraphy may make them less hydrologically vulnerable to regional hydraulic modifications, such as stream channel incision, reduced stream-floodplain connectivity, and aquifer drawdown.

Land use within mineral flats may influence several key aspects of wetland hydrology, including infiltration rates and hydroperiod. Surface infiltration is affected by land-use related factors such as saturated hydraulic conductivity, surface roughness, organic matter (both coarse litter at the surface and soil organic matter), soil surface compaction, and soil structure (e.g. Dingman 2002; Brady and Weil 2008). Soil structure is affected by physical (e.g. surface cracking in shrink-swell soil), biological (e.g. root decomposition and animal burrowing), and chemical processes (Dingman 2002; Brady and Weil 2008). Reduced infiltration can alter the wetland hydroperiod by increasing surface runoff generation, decreasing soil and groundwater recharge, and increasing the frequency of surface ponding (e.g. Mitsch and Gosselink 2000; Brady and Weil 2008).

Given the regulatory emphasis on the period of time when wetlands are saturated during the growing season (USACE 1987), many studies neglect to consider the entire hydrologic regime when evaluating the effects of land use on wetland hydrology. Metrics commonly employed in the fields of stream ecology and river management to describe the entire stream flow regime, including the magnitude, frequency, timing, duration, and rate of change in flow (e.g. Poff et al. 1997; Poff et al. 2006), can also be used to describe the wetland hydroperiod. Such metrics are especially useful in evaluating aquatic ecosystem response to anthropogenic perturbation (e.g. Richter et al. 1996). Recent research has demonstrated that land use-related modifications to the timing and duration of saturation can impact key biogeochemical processes, such as denitrification (Stander and Ehrenfeld 2008), in mineral flat wetlands. Additionally, hydroperiod components such as the timing, duration, and frequency of surface ponding and saturation within the root zone are tied to wetland flat plant community composition (Rheinhardt et al. 2002) and wildlife habitat (Taft and Haig 2003; Pearl et al. 2005).

While our knowledge of ecological functions related to specific components of the mineral flat wetland hydroperiod is growing (e.g. Rheinhardt et al. 2002; Stander and Ehrenfeld 2008), relationships between mineral flat hydrology and edaphic, plant cover, and microtopographic variables remain poorly understood, especially for grassland flats under different land uses. We thus undertook field studies of least-altered prairie, farmed, and restored mineral flat wetlands in Oregon's Willamette Valley with the objectives of 1) characterizing the hydroperiod and soil structure of mineral flat wetlands under different land uses, and 2) identifying plot-scale soil, plant cover, and microtopographic variables that are most strongly associated with infiltration rates and hydroperiod metrics. The results of our field studies address fundamental land management questions, including: 1) does land management affect how mineral flats absorb and store precipitation inputs at the beginning of the wet season, and 2) is land use, or site mechanical disturbance history most strongly associated with infiltration rates, hydroperiod metrics, and soil structure? We hypothesized that infiltration rates would increase from farm to restored to prairie sites, across all site groups. We also hypothesized that factors relating to the development of soil structure (e.g. soil organic matter) would result in higher infiltration rates, reduced frequency and height of surface ponding, and longer duration of saturation in the root zone across all land use types.

We anticipate that results of this work will lead to better understanding of key site-scale edaphic and hydrologic factors to consider when prioritizing and managing sites for restoration, and of how site disturbance under a variety of land uses may impact different hydrologic processes and components of the wetland hydroperiod.

Methods

Study Area

We selected three sites to represent current least-altered wetland prairie conditions in the southern Willamette Valley lowlands: North Prairie at Finley National Wildlife Refuge, Fisher Butte Prairie within the Coyote Creek catchment, and Willow Creek Prairie in the Amazon Creek catchment. Herein, these sites are referred to as Finley Prairie, Coyote Prairie, and Amazon Prairie, respectively. Prairie sites were selected from a group of representative least-altered sites used to develop the Willamette Valley Hydrogeomorphic (HGM) Guidebook (Adamus 2001), and exist on Dayton or Natroy hydric soils with no known history of plowing. Like most historic wetland prairie ecosystems in the Willamette Valley lowlands, the prairie sites used in this study were historically maintained by fire, and have recently been grazed, burned, or mowed to prevent the encroachment of shrubs and trees (Johannessen et al. 1971; Clark and Wilson 2001). Plant communities at the prairie sites include tufted hairgrass (*Deschampsia cespitosa),* along with a variety of sedges (e.g. *Carex unilateralis*), rushes, and forbs (Wilson 1998). In addition to the native plant community, a variety of non-native grasses, forbs, and shrubs (e.g. *Anthoxanthum* *odoratum, Mentha pulegium*, and *Rosa eglanteria*) also occupy the prairie sites (Wilson 1998).

For each least-altered site, we selected a pair of farmed and restored wetlands in the same small stream sub-basin (<200 km^2) with similar topography, soil, and hydrology (Table 1; Figure 1). Farmed wetlands were located within cultivated grass seed and hay fields, typical of many former Dayton and Natroy soil wetlands in the southern valley lowlands. Restored wetlands were former ryegrass fields that had been retired from cultivation and re-planted with *Deschampsia cespitosa* and other native wetland prairie species.

The Dayton soils covering the Finley-area sites and parts of the Coyote and Amazon sites are extensive in the Willamette Valley lowlands on nearly-flat prairie terraces (NRCS 2006). Currently, these soils support conservation and farming land uses without artificial drainage, and farming, rural residential, urban, and other land uses when artificially drained (e.g. using drainage tiles and surface ditches). These soils commonly have a perched water table at the beginning of the fall wet-up, and deeper groundwater near the soil surface between the months of November and April, with some ponding between December and April (NRCS 2006). Perching is due to an abrupt textural change between near-surface silt loam A/E horizons and the silty clay Bt horizons in the upper 40 cm of the soil profile (NRCS 2006). The presence of two separate water tables, with a perched water table occurring above 60 cm, has been documented in the Dayton soil farm-lands north of the Finley NWR (Austin 1994). Both the near-surface and perching soil horizons contain smectite clay minerals, resulting in surface crack formation as these soils transition from saturated winter conditions to dry summer conditions.

In the lowland portions of the Amazon Creek and Coyote Creek catchments, Natroy soils are intermingled with Dayton soil. Natroy soils have a silty clay loam layer atop a clay layer between 25 and 50 cm below the soil surface (James and Baitis 2003), which is deeper than the approximately 13 cm reported in the soil series description (NRCS 2006). The NRCS reports an apparent water table between

November and May for Natroy soils (NRCS 2006). Depth to the Bt horizon for Dayton soils in the West Eugene area is between 23 and 40 cm (James and Baitis 2003), which is similar to the approximately 40 cm depth reported for Dayton soils further north in the valley (Austin 1994, NRCS 2006).

For all study sites, mean annual temperatures are 3°C in January and 19°C in July (WRCC 2011), and average annual precipitation is between 102 and 127 cm (PNW-ERC 2002). Approximately 70- 80% of total precipitation, and 41 cm of mean annual groundwater recharge occurs between November and April (PNW-ERC 2002; Conlon et al. 2005). Evapotranspiration rates in the Willamette lowlands are highest between April and July, and estimated annual evapotranspiration from lowland areas where the water table is within 3 m of the soil surface is between approximately 40 and 50 cm (Lee and Risley 2002).

Methods

At the mesoscopic scale (1 m^2), we examined associations between land management, soil properties, plant cover, microtopography, and hydroperiod at each of the nine study sites. One 200 m x 200 m macroplot was randomly located at each site to allow comparison across sites of different shapes and sizes, reduce spatial autocorrelation, and increase sampling efficiency. For each site, we overlaid a 2 m x 2 m grid across each macroplot and the locations of six study plots were determined by randomly selecting six grid cells.

Hydrometric Measurements

Within each sample plot, we monitored water table elevations and conducted surface infiltration tests. One shallow well with a 2.54 cm slotted screen was installed at the center of each plot to monitor shallow water table levels throughout the 2009-2010 water year. We were unable to install wells at the Amazon Prairie and Finley Restored sites due to time constraints and permit restrictions related to field use by waterfowl. For the seven other sites, wells were designed, installed, and maintained according to technical standards established by the US Army Corps of Engineers Wetlands Regulatory Assistance Program (USACE 2005). Well holes were hand-augered to a depth immediately above or just within the less-permeable perching soil horizon (approximately 30-40 cm), and depth to the perching layer was recorded. Thickness of the perching layer was estimated from local soil survey information (NRCS 2006), along with deeper soil excavations within or adjacent to the sites (Austin 1995; James and Baitis 2003; Marshall and Tullos in prep). Wells were monitored weekly during the fall wet-up, on a biweekly schedule beginning in November, 2009, and then weekly at the end of the wet season in spring, 2010. Increased monitoring frequency during the fall and spring were intended to improve our accuracy in capturing dates of wet-up and drying in the upper 30 cm of the soil profile. Depth of any surface ponding relative to the ground surface was also recorded during field visits. Water levels were checked using a 3-m steel tape measure with millimeter gradations and a non-toxic, washable marking pen. Water table elevation data were used to calculate eight hydroperiod metrics describing the wetland hydrologic regime (Table 2).

Between August and September 2010, prior to the onset of fall rains and when the water table at each site was at least 1.5 m below the soil surface, we conducted infiltration tests at all nine study sites to measure how precipitation or ponded surface water entered the vadose zone under unsaturated and saturated conditions (Dingman 2002). Single-ring infiltration tests were conducted using the Beerkan method (Haverkamp et al. 1996; Lassabatere et al. 2006) with de-chlorinated local tap water. We fabricated the rings from schedule 80 PVC conduit pipe (9.72 cm ID, 11.43 cm OD). Nine rings were installed at each plot at 1 m grid spacing for a total of 54 tests per site, and water was added to rings in 100 ml increments.

Soil Sampling and Analysis

Four soil samples were collected from the top 10-15 cm of the soil profile prior to each infiltration test. Subsamples of each soil sample were reserved for water stable aggregate tests and oven-drying at 105°C for 24 hours to determine antecedent gravimetric water content. Following each test, a 5.4-cm soil corer was used to extract an undisturbed soil sample from the top 12 cm of each ring location to determine gravimetric water content at saturation, bulk density, and percent organic matter in the laboratory. Cores were placed in air-tight containers, weighed on the day of collection, and then oven-dried at 105°C for 24 hours to determine saturated water content and calculate bulk density. A composite sample from the nine cores taken from each sample plot was used for particle size distribution analysis using the hydrometer method (Day 1965) as modified by Gee and Bauder (1986). Soil texture was determined using the USDA classification system (Soil Survey Staff 1975). The sand fraction was determined by wet sieving with a 53-μm sieve and the clay fraction was estimated using the simplified clay fraction procedure detailed in Gee and Or (2002). Percent soil organic matter by dry soil weight was determined using loss by ignition in a muffle furnace at 450°C for eight hours (Nelson and Sommers 1996).

Vegetation and Microtopography

Vegetation and microtopography were documented during late summer, contemporaneous with infiltration tests and soil sampling. Each 2 m x 2 m plot was divided into four quadrants and aerial and basal cover was measured with a 1 m x 1 m free-standing sampling frame using the point-intercept method (Levy and Madden 1933; Goodall 1952). Metal rods hanging vertically from the sampling frame allowed us to sample a grid of fifty points in each quadrant. Aerial cover was characterized as live plants or standing litter, and grouped into five plant categories (grass, sedge, rush, woody plant, and forb). Bare ground and litter was recorded where there was no standing vegetation. We recorded plant and soil surface basal cover for bunch grass, non-bunching grass, forbs, sedges, rushes, bare ground, litter, moss, soil cracks >1mm, and animal burrows. To quantify microtopographic relief in plots, we also counted soil mounds using the point-intercept method (Levy and Madden 1933; Goodall 1952).

Statistical Analyses

Prior to running multivariate tests, we created a scatterplot matrix of all variables to be used in the statistical analyses, and calculated skewness and standard kurtosis values for each variable to test for normality (e.g. Ramsey and Shafer 2002). Variables with non-normal distributions (infiltration rates, coefficients of variation for water table elevation, relative plot elevation, % clay, % bare ground, % forb cover, and % grass cover) were then natural-log transformed. We identified outliers using box plots, and Grubbs' test with the 95% confidence interval for each mean, and several extreme outliers with median absolute deviation scores exceeding 3.5 were removed. Following data transformations (Table 2), we tested for associations between land use categories, hydrologic metrics, and soil explanatory variables using Analysis of Variance (ANOVA) tests with multiple comparisons. We also examined associations between hydrologic metrics, soil variables, and categories representing different levels of mechanical site disturbance. To compare soil properties with site disturbance history, we created a categorical variable representing different levels of mechanical disturbance in our ANOVA analysis. Disturbance levels were 1) never plowed, 2) within the past year, 3) within 1-5 years, and 4) more than 5 years prior. Next, we evaluated correlations between infiltration, wetland hydrologic metrics and explanatory site variables (Table 2).

Results

Characterizing hydrology and soil properties across land use

Infiltration rates and hydroperiod across land use

Across land use, we observed a decrease in mean infiltration rates moving from prairie (mean = 1.78 m/hr ± 0.029 sd) to farmed (mean 0.427 m/hr ± 0.020 sd) to restored (mean 0.347 m/hr m/hr ± 0.068 sd) sites (*n* = 18 for each land use group). Farm sites had the greatest range and variation (*CV*) in infiltration rates (1.95 m/hr; *CV* = 50.9%), followed by prairie sites (0.269 m/hr; *CV* = 20.4%) and restored sites (0.129 m/hr; *CV* = 19.0%). For the individual site groups, we observed the highest overall mean infiltration rates at the Coyote sites (3.51 m/hr ± 0.02sd at Coyote Prairie), and the lowest overall infiltration rates at the Finley sites (0.04 m/hr ± 0.02 sd at Finley Farm) (Table 3; Fig. 2). Finley and Coyote prairies had the highest maximum infiltration rates of all sites (6.08 m/hr and 5.69 m/hr, respectively; Table 3).

Timing of wet-up was significantly different between land use groups, becoming later in the wet season moving from farmed to prairie to restored sites ($p < 0.05$ at the 95% confidence level; Table 4). We observed a decrease in mean groundwater elevation and a longer duration of continuous saturation moving from farmed to prairie to restored sites, but did not observe a significant difference ($p > 0.05$ at the 95% confidence level) between land use groups. The end of continuous saturation, coefficient of variation in groundwater elevation, mean surface ponding depth, and frequency of surface ponding lacked a statistically significant association with land use ($p > 0.05$).

Comparisons between hydroperiod metrics and level of site disturbance revealed significant differences across disturbance groups for timing of wet-up ($p < 0.01$), frequency of surface ponding ($p < 0.01$), mean surface ponding depth ($p < 0.05$), mean groundwater elevation ($p < 0.01$), and coefficient of variation in groundwater elevation ($p < 0.05$; all at the 95% confidence level). Disturbance classes were not strongly associated with the end of continuous saturation or duration of continuous saturation ($p > 0.05$). Sites disturbed within the past year had the earliest wet-up, followed by sites that had never been plowed and then sites with disturbance between 1-5+ years prior. We observed a similar trend for frequency of surface ponding, mean surface ponding depth, and mean groundwater elevation, with sites most recently disturbed having a shallower water table and more frequent, deeper ponding. Sites that had never been plowed had more variable water table elevations during the wet season (mean = 2.43 ± 1.03 sd), followed by sites disturbed within the past year, and sites disturbed within the past 1-5 years.

Some of the trends we observed in the hydroperiod metrics with land use and site disturbance are likely the result of correlation between hydroperiod metrics. Of the eight metrics we compared (Table 2), timing of wet-up and coefficient of variation in groundwater elevation during the wet season were most strongly correlated with the greatest number of other measured hydroperiod metrics (Table 5; Fig. 3; Fig. 4). Sample plots with an earlier timing of wet-up had a longer period of continuous saturation, a longer overall period of saturation, more frequent surface ponding, deeper surface water, and a shallower and more variable water table throughout the wet season ($p<0.05$ for duration of continuous saturation and $p<0.01$ for all other variables; Table 5; Fig. 3). Plots with greater variability in groundwater elevations throughout the wet season tended to have a longer duration of continuous and overall saturation, more frequent and deep surface ponding, and a shallower water table (Table 5; Fig. 3).

Trends in soil characteristics across land use

Mean bulk density was significantly associated with land use across all sites ($p<0.01$ at the 95% confidence level), and increased moving from prairie (0.98 g/cm^3 ± 0.084 sd) to farm (1.06 g/cm^3 ± 0.13 sd) to restored (1.15 g/cm^3 ± 0.16 sd) sites. Across all sites, *SOM* was significantly different between land use groups ($p<0.05$ at the 95% confidence level). Mean *SOM* was highest for prairie sites (8.23% ± 1.8 sd), followed by farm sites (6.76% ± 2.4 sd), and restored sites (6.07% ± 2.4 sd).

Our ANOVA tests revealed a significant association between level of site disturbance and mean bulk density ($p<0.001$ at the 95% confidence level). Mean bulk density was lowest for prairie sites with no plowing history (0.98 g/cm^3 ± 0.084 sd), followed by sites with plowing in the past year (1.04 g/cm^3 ± 0.14 sd), sites with disturbance more than 5 years prior (1.09 g/cm^3 ± 0.099 sd), and sites with mechanical disturbance between 1-5 years prior (1.20 g/cm^3 ± 0.17 sd). *SOM* also differed between levels of disturbance ($p<0.01$ at the 95% confidence level), with prairie sites having the highest mean *SOM* (8.23% ± 1.82), followed by sites disturbed within the past year, sites disturbed within the past 1-5 years, and sites disturbed more than 5 years prior.

For the Finley-area sites, we observed increasing mean bulk density and decreasing *SOM* moving from prairie to restored to farmed site plots (Table 3). The Amazon-area sites showed a similar trend with increasing mean bulk density and decreasing *SOM* moving from prairie to farmed to restored plots (Table 3). Within the Coyote-area site group, farmed plots had the lowest mean bulk density and highest amount of *SOM* (Table 3). Infiltration rates were strongly associated with bulk density at the Amazon sites ($r = -0.87$, $p < 0.001$) and Finley sites ($r = -0.77$, $p<0.001$), and to a lesser extent at the Coyote sites ($r = -0.52$, $p<0.05$) (Fig. 2).

Relationships between wetland hydrologic metrics across land use

Infiltration

Infiltration rate was most strongly correlated with soil variables related to reduced land use intensity, increased soil structure, and drainage, including *SOM* ($r = 0.65$, $p<0.01$) and bulk density ($r = -0.59$, $p<0.01$; Table 6). We also observed a negative relationship between infiltration and percent silt near the soil surface. Of the hydroperiod metrics we examined, infiltration rates were positively associated ($p<0.05$) with the duration of continuous saturation during the previous wet season and negatively associated ($p<0.05$) with the mean depth of ponded water during the previous season (Table 5).

For the plant cover variables we examined, infiltration was positively correlated with forb cover ($r = 0.42$, $p<0.01$) and basal cover, and negatively correlated with bare ground (Table 6). We were unable to make any statistical comparisons between infiltration and microtopographic variables (% mound cover, rodent burrow cover, and soil crack cover) due to non-normal distribution and skewness of the data that could not be remedied by log10, inverse, or arcsine squareroot transformations.

Individual hydroperiod metrics

Plots with earlier wet-up tended to have a higher clay content in the soil ($r = -0.50$, $p<0.01$), and less sand (Table 6). Continuous saturation later into the growing season (end of continuous saturation) was associated with less silt ($r = -0.58$, $p<0.01$) and a higher sand content in the soil profile (Table 6). We found a near-zero correlation between plot elevation and timing of wet-up. Of the plant cover variables we observed, plots with a later timing of wet-up were associated higher forb and grass cover, and less litter (Table 6). Our analysis did not reveal any significant ($p<0.1$) associations between the date of last continuous saturation and any other measured variables.

Overall duration of saturation within the upper 30-cm of the soil profile was associated with a high clay content ($r = 0.70$, $p<0.01$), lower bulk density, higher organic matter, and less sand (Table 6). The duration of continuous saturation was negatively associated with decreasing silt content ($r = -0.61$, $p<0.01$), and positively associated with higher clay content to a lesser degree (Table 6). Our analysis revealed a negative correlation between longer duration of all saturation, bare ground, and grass cover (Table 6). We also observed a positive correlation between duration of all saturation and the amount of surface litter.

Frequency of surface ponding was moderately correlated with the amount of clay in the soil ($r = 0.42$, $p<0.01$), and showed a weak negative correlation the amount of sand (Table 6). Forb cover decreased with increasing frequency and depth of surface ponding ($r = -0.40$, -0.42, respectively, with $p<0.01$; Table 6). We did not observe any other significant associations with frequency of surface ponding or mean depth for the other variables analyzed.

Our analysis revealed a slight negative correlation between plots with a shallower seasonal water table and percent sand. Increased coefficient of variation in groundwater elevation between sampling dates was weakly correlated with the amount of clay in the soil. A higher average water table was associated with less forb cover (r

= -0.57, p<0.01; Table 6), while higher coefficient of variation in groundwater elevation was associated with increased surface litter (Table 6).

Discussion

Overall, our findings suggest that edaphic factors, particularly those related to soil structure, are strongly associated with wetland infiltration and overall hydroperiod across a variety of land uses in grassland mineral flat wetlands. Infiltration, timing of wet-up, timing of the end of continuous saturation, duration of saturation, and frequency of ponding were all related to different soil particle size fractions. This result suggests that site-specific soil texture (e.g. relying on detailed soil textural classification rather than NRCS soil mapping units and general soil descriptions) is an important factor to consider when planning for wetland restoration or evaluating the potential implications of different land management practices. Plots with a high silt content tended to have overall lower infiltration rates, whereas plots with a higher clay content tended to have higher infiltration rates and overall resiliency to farming and restoration-associated soil disturbance. High-clay plots also tended to have earlier timing of wet-up, a longer hydroperiod duration, and more frequent ponding across the different land uses.

The hydroperiod metrics we examined were generally more sensitive to level of site disturbance than land use/land cover alone. Timing of fall wet-up was the only hydroperiod metric that was significantly different across land use. Of the hydroperiod metrics we examined, the duration of continuous saturation and timing of the end of continuous saturation, which are commonly used in wetland delineations and other regulatory determinations, were not significantly associated with level of site disturbance or land use. Additionally, these metrics were not significantly correlated with any of the plant cover variables included in our analysis. Timing of wet-up, overall duration of saturation, and metrics describing shallow groundwater and surface ponding conditions were more strongly associated with factors relating to plant cover and community composition.

Of the soil properties we examined relating to soil structure (bulk density and organic matter), and all explanatory variables used in this study, bulk density and organic matter were most strongly associated with infiltration rate (Table 6). Bulk density and soil organic matter were highly correlated for our wetland plots, and both metrics are useful in quantifying overall soil condition across different land use and levels of disturbance in wetland environments (e.g. Galatowitsch 1996; Campbell et al. 2002; Bruland and Richardson 2005; Ullah and Faulkner 2006).

While we did see the highest and most variable infiltration rates at prairie sites within each site group, our overall results do not support the hypothesis that infiltration rates would increase moving from farmed to restored to prairie sites on account of level of site disturbance and land cover. The Finley area sites, which had a higher silt content than the other site groups, were the exception to this finding, as infiltration rates increased along the progression from most to least-disturbed soil. Given that the Finley Restored site has not experienced any site-wide mechanical soil disturbance in over ten years, we suspect that the Amazon and Coyote restored sites may have higher infiltration rates with time. Our results do support previous findings that wetland soil properties tend to be most heterogeneous at least-altered sites compared to more disturbed wetland sites of the same HGM class (e.g. Bruland and Richardson 2005), particularly for infiltration. Also, our findings suggest that the soil disturbance legacy of agricultural land use is not immediately remedied by wetland restoration activities (Bruland et al. 2003). The restored sites we examined in this study generally had low infiltration rates, higher bulk density, and lower *SOM* relative to farmed and prairie sites, suggesting a higher degree of site disturbance. We suspect that the level of management required in establishing high native plant species cover on restored sites—especially mitigation banks—is associated with greater mechanical disturbance and overall foot traffic throughout the year and disruption to soil structure.

Our findings that many of the farm plots, especially at Amazon and Coyote Farm sites, had infiltration rates, bulk density, and *SOM* values that were comparable to many of the less-disturbed sites, suggest that management activities on these sites are minimizing processes like soil compaction, surface crust formation, and water erosion. Compared to the other farm sites, Finley Farm had a much higher silt content, which was negatively correlated with infiltration rates. We suspect that the timing of most farming activities for grass seed and hayfield crops, which requires that the soil be dry enough for large farm equipment and grass be dry enough to avoid problems with mold in hay bales and seed, coincides with window of time where soils are least-susceptible to compaction and aggregate break-down (Brady and Weil 2002). Additionally, leaving crop stubble and litter on farm fields most likely increases overall soil structure with time (Brady and Weil 2002). It is important to note that we compared land uses in minimally drained catchments in this study, and that row crops, or more frequently tilled and thoroughly drained sites, such as flats in intensively farmed and drained areas along the Gulf Atlantic Coastal Plain (e.g. Ehrenfeld et al. 2003) or flats used to grow wheat and row crops in the Willamette Valley, may have very different hydrologic behavior.

Conclusions

In this study, we compared infiltration and wetland hydroperiod metrics with edaphic and plant cover variables to better understand how land use influences wetland hydrology at the plot scale, and which factors may confound traditional assumptions about land use, hydrology, and soil hydraulic properties in wetland environments. Our findings suggest that land use does not necessarily dictate the overall hydrologic behavior of mineral flat wetlands, particularly if land management practices minimize mechanical soil disturbance and compaction. If we are to manage for wetlands with high infiltration rates, and thus a greater potential to store and delay storm runoff, our results suggest that we must use practices that minimize soil compaction, increase soil organic matter content, and promote diverse plant cover.

Our research provides a first look into key site-scale edaphic and hydrologic factors to consider when prioritizing and managing mineral flat wetlands for restoration, assessing levels of site disturbance, and anticipating which types of site disturbance under a variety of land uses may impact different hydrologic processes and components of the mineral flat wetland hydroperiod. Future research, using a randomized sampling design across a larger number of sites and geographic regions, is necessary to test whether our findings truly represent the larger mineral flat wetland population within agricultural catchments across the United States.

Acknowledgements

We thank Laura and Jim Marshall, Alicia Omlid, Terrence Morris, and Diana Sharps for assistance with the installation and monitoring of water table wells, soil sample collection, and field infiltration tests. We are grateful to Scott Bridgham, Richard Cuenca, and Maria Dragila for feedback related to study design, field methods, and laboratory methods, along with field and lab equipment use. Thanks also to Jock Beall, Jarod Jebousek, and Molly Monroe from the US Fish and Wildlife Service, Diane Steeck, Trevor Taylor, and Paul Gordon from the City of Eugene, Wes Messinger from the US Army Corps of Engineers, Jason Nuckols from The Nature Conservancy of Oregon, Chuck Stolsig, and Tom Danelski for help with field logistics, site background information, permitting, and access to wetland study sites. This material is based upon work supported under a National Science Foundation Graduate Research Fellowship. Any opinions, findings, conclusions or recommendations expressed in this publication are those of the authors and do not necessarily reflect the views of the National Science Foundation.

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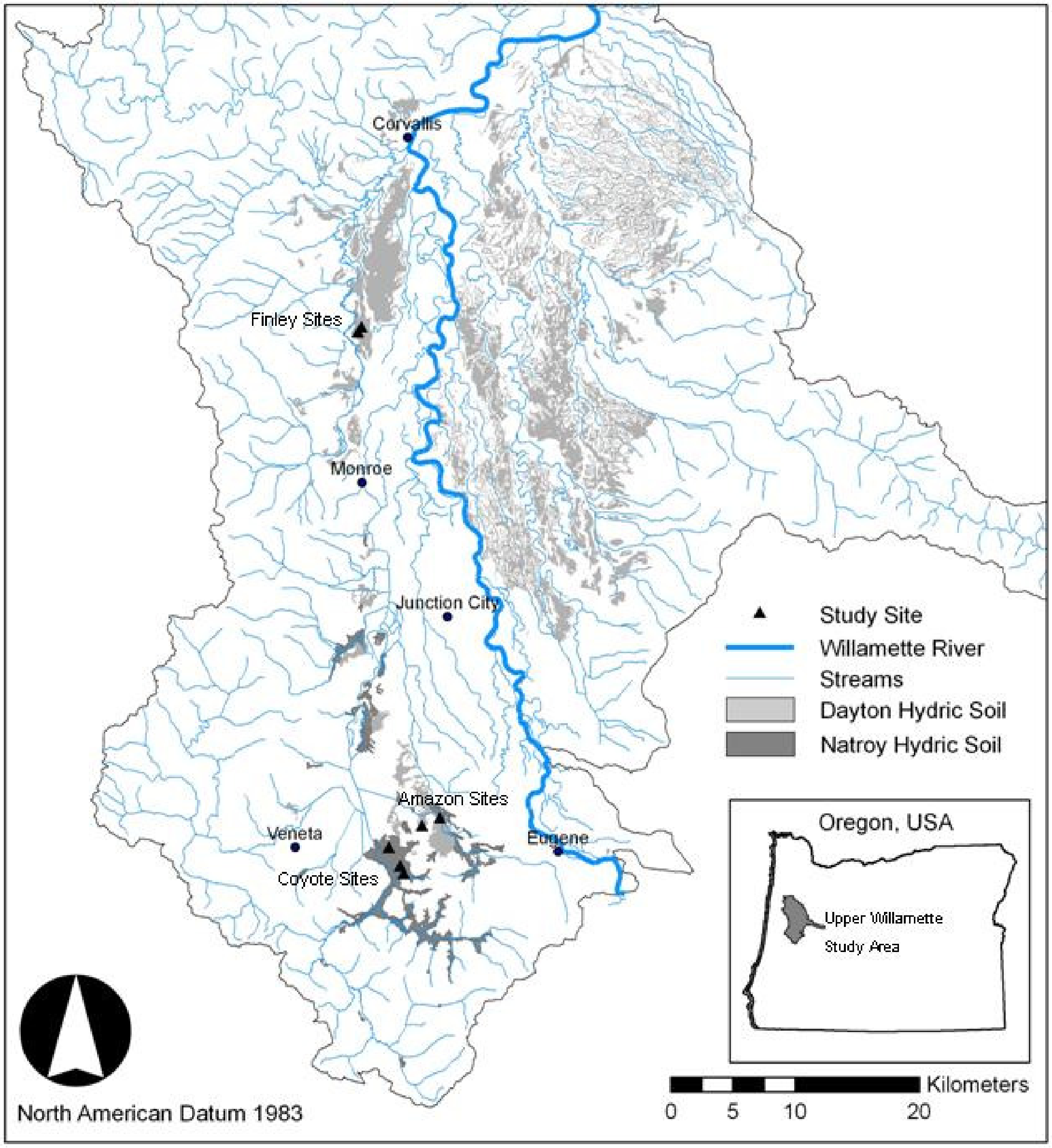

Fig. 1: Study area in the southern Willamette Valley lowlands, Oregon.

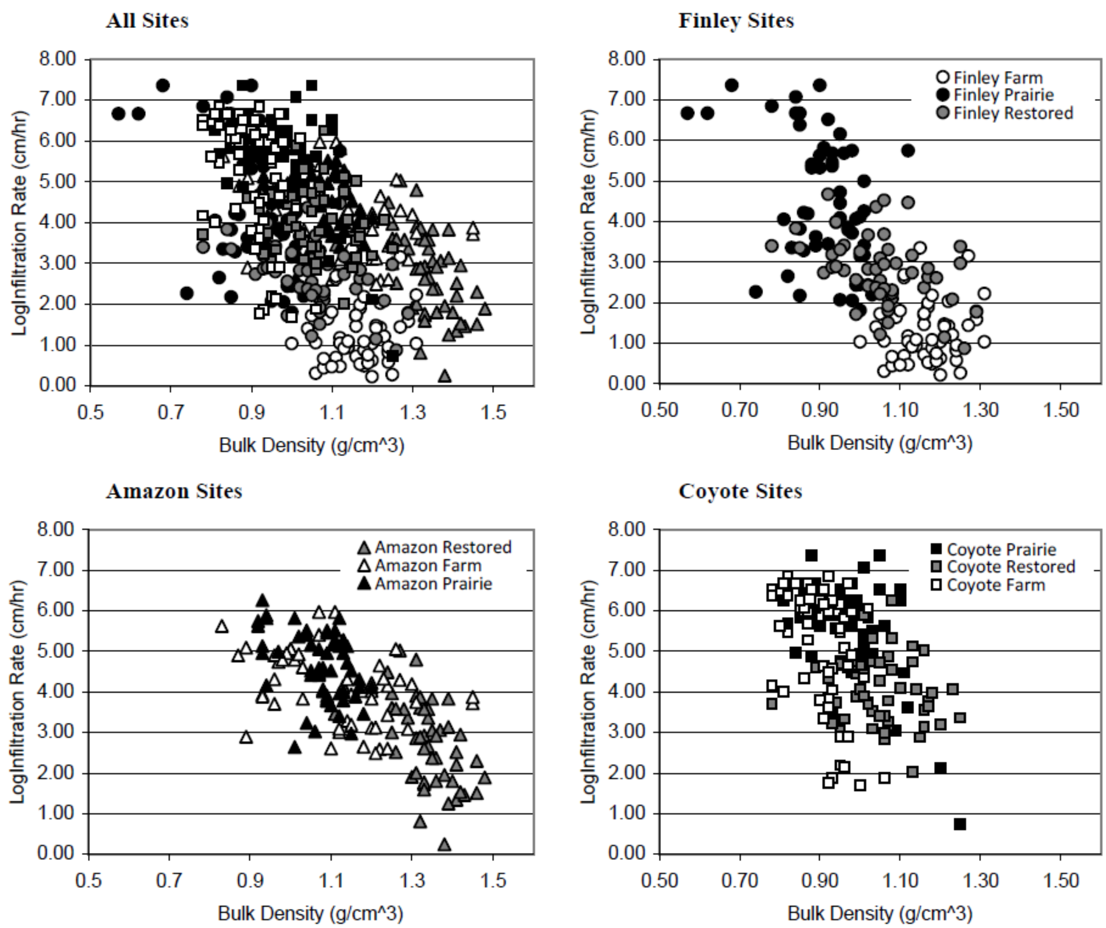

Figure 2. Infiltration rates and soil bulk density values for all sites; together, and by site groups.

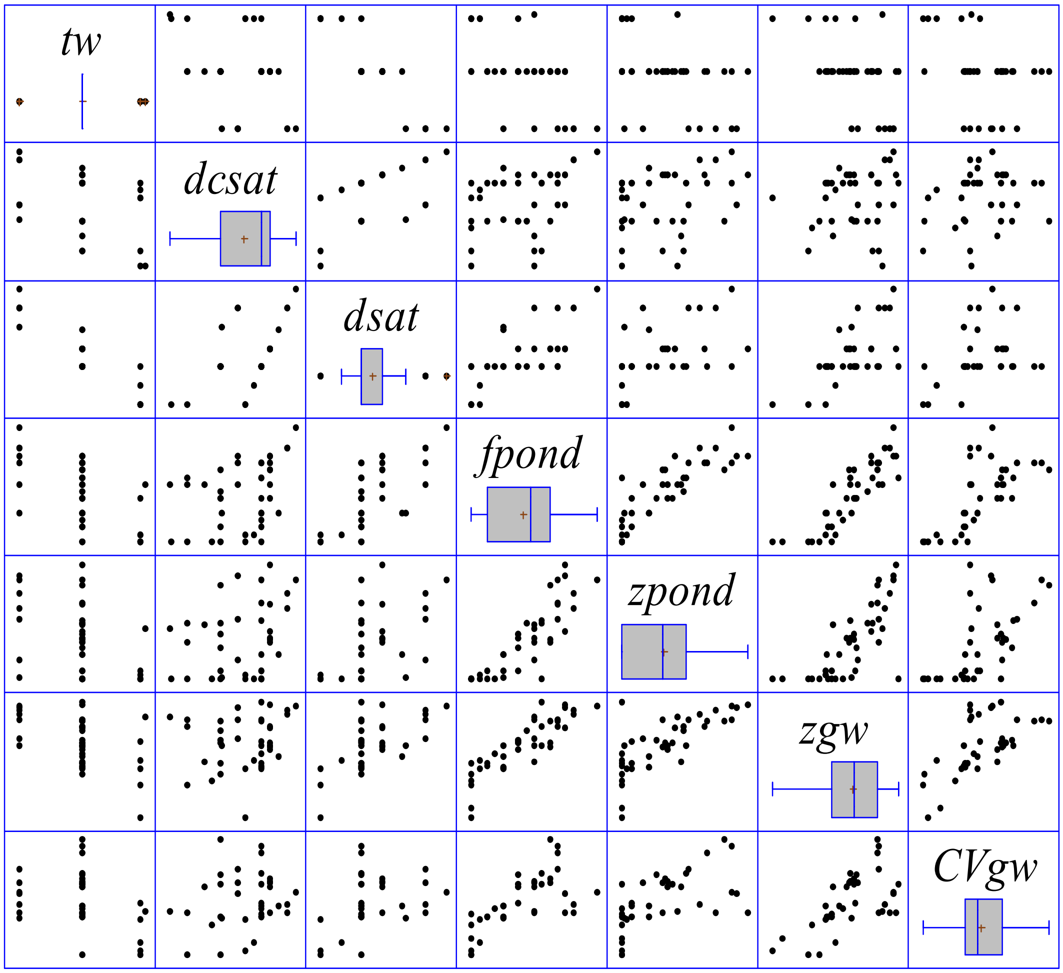

Figure 3. Scatterplot matrix for associations between seven hydroperiod metrics. Metrics describing the depth, frequency, and variability in shallow surface ponding and groundwater were most strongly correlated. Please see Table 4 for correlation coefficients.

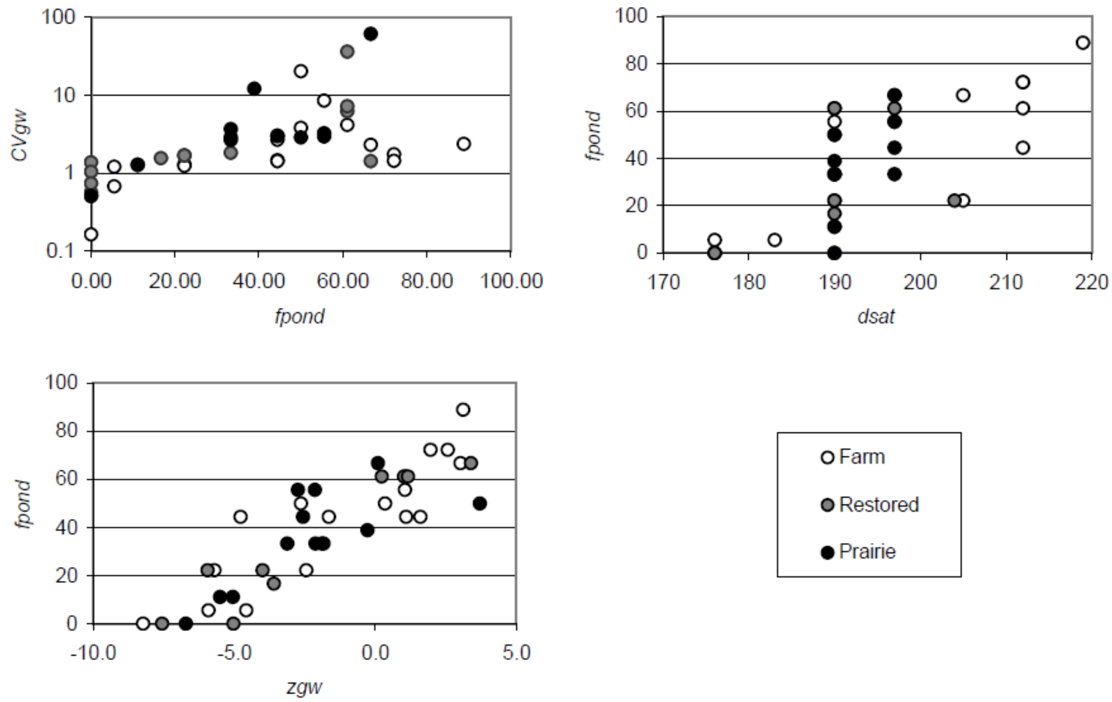

Figure 4. Relationships between highly correlated hydroperiod metrics, by land use.

Table 1. Site Descriptions

Site (Ownership)	Size (ha)*	Primary management	Years since plowing/discing	Mapped Soil Series
Finley Prairie (USFWS)	30	Fire	-	Dayton
Finley Restored (USFWS)	11	Fire	10	Dayton
Finley Farm (Private)	32	Mowing, discing	1	Dayton
Amazon Prairie (TNC)	7	Fire	-	Dayton/Natroy
Amazon Restored (City of Eugene)	16	Fire, mowing	5	Dayton/Natroy
Amazon Farm (Private)	8	Mowing	6	Dayton
Coyote Prairie (USACOE)	29	Fire; brush trimming	-	Natroy
Coyote Restored (City of Eugene)	11	Mowing	3	Natroy
Coyote Farm (City of Eugene)	33	Mowing, discing	1	Natroy

*size includes all mapped Dayton or Natroy hydric soil within the same land management area of the site

Table 2. Variables used in statistical analyses, and transformations applied to data.

Explanatory Variables	Units	Definition	Transformation Applied
Elev	m	plot elevation relative to lowest plot in field	log10
ρ_d	g/cm^3	dry soil bulk density	none
SOM	percent	percent organic matter in soil	none
Clay	percent	percent clay in soil	log10
Silt	percent	percent silt in soil	none
Sand	percent	percent sand in soil	none
BC	percent	percent basal cover	none
BG	percent	percent bare ground	log10
LT	percent	percent surface litter	none
Forb	percent	percent forb cover	log10
Grass	percent	percent grass (all types)	log10
Response Variables			
I	m/hr	Steady state infiltration rate	log10
t_w	day	Timing of wet-up (day from October 1) within upper 30-cm of soil profile	none
t_{csat}	day	End of continuous saturation (since Oct. 1) in upper 30-cm of soil profile	none
d_{csat}	days	Duration of continuous saturation in upper 30-cm of soil profile	none
d_{sat}	days	Duration of all saturation (to date of last observed 30-cm saturation in spring)	none
f_{pond}	none	Frequency of surface ponding from November 1 - July 1	none
z_{pond}	cm	Mean surface ponding depth	none
z_{gw}	cm	Mean groundwater elevation from t_w to t_{csat}	none
CV_{gw}	none	Coefficient of variation in groundwater elevation from t_w to t_{csat}	none

Table 3. Descriptive statistics for infiltration rates and potentially associated soil variables across all nine study wetlands.

	Infiltration Rate (I) in m/hr				*% Clay*		*ρ_d (g/cm^3)*		*SOM*	
Site	Mean	Min	Max	sd	Mean	sd	Mean	sd	Mean	sd
Finley Prairie	1.26	0.22	6.08	0.05	11.19	2.40	0.91	0.05	8.60	1.68
Finley Restored	0.22	0.12	0.31	0.01	18.46	4.19	1.06	0.09	5.84	0.75
Finley Farm	0.04	0.02	0.10	0.02	19.80	3.70	1.17	0.04	5.51	0.87
Amazon Prairie	1.27	0.84	3.06	0.02	17.48	2.66	1.07	0.05	7.77	2.32
Amazon Restored	0.25	0.17	0.43	0.01	14.93	4.09	1.34	0.06	3.55	0.40
Amazon Farm	0.76	0.26	1.46	0.02	8.08	1.73	1.11	0.10	4.89	0.94
Coyote Prairie	3.51	1.07	5.69	0.02	37.55	3.90	0.97	0.05	8.31	1.63
Coyote Restored	0.75	0.44	1.55	0.02	22.35	6.97	1.05	0.08	8.81	2.20
Coyote Farm	2.51	0.42	4.44	0.03	38.51	4.05	0.91	0.04	9.89	0.70

sd = Standard deviation

Table 4. Hydroperiod metrics by site.

	t_w		t_{csat}		d_{csat}		f_{pond}		z_{pond}		z_{gw}		CV_{gw}	
Site	Mean	sd	Mean	sd	Mean	sd	Mean	sd	Mean	sd	Mean	sd	Mean	sd
Finley Prairie	64	0.0	225	14.3	161	14.3	27.8	18.6	2.52	3.0	-2.05	3.7	2.24	1.3
Finley Restored	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Finley Farm	64	9.5	203	13.8	139	21.2	50.9	8.9	3.17	1.3	0.39	2.7	1.97	0.6
Amazon Prairie	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Amazon Restored	64	0.0	255	2.9	191	2.9	38.0	28.4	2.39	2.6	-1.50	3.8	1.51	0.1
Amazon Farm	73	7.2	237	30.3	163	34.1	13.9	19.5	0.71	1.1	-8.63	8.1	1.27	1.3
Coyote Prairie	64	0.0	260	2.9	196	2.9	44.4	19.9	1.67	0.8	-2.66	1.8	2.6	0.8
Coyote Restored	83	19.1	252	23.7	169	39.6	19.4	24.8	0.86	1.4	-6.28	5.3	1.17	0.6
Coyote Farm	49	0.0	239	27.4	190	27.4	60.2	23.7	2.71	1.8	0.77	2.3	2.32	1.1

Table 5. Pearson correlation coefficients for associations among wetland hydrologic metrics across seven sites.

Metric	t_w	t_{csat}	d_{csat}	d_{sat}	f_{pond}	z_{pond}	z_{gw}	CV_{gw}
I	-0.18	0.28[c]	**0.34**[b]	0.25	-0.20	**-0.36**[b]	-0.31[c]	-0.23
t_w		-0.05	**-0.36**[b]	**-0.91**[a]	**-0.59**[a]	**-0.46**[a]	**-0.51**[a]	**-0.42**[a]
t_{csat}			**0.92**[a]	0.21	0.20	0.10	0.03	0.24
d_{csat}				**0.53**[a]	**0.38**[b]	0.24	0.17	**0.36**[b]
d_{sat}					**0.71**[a]	**0.53**[a]	**0.64**[a]	**0.45**[a]
f_{pond}						**0.87**[a]	**0.87**[a]	**0.69**[a]
z_{pond}							**0.74**[a]	**0.60**[a]
z_{gw}								**0.62**[a]

Statistically significant values are in bold, and are of $p<0.05$ at a 95% confidence level.

[a] $p < 0.01$

[b] $0.01 < p < 0.05$

[c] $0.05 < p < 0.10$

Table 6. Pearson correlation coefficients for associations between plot characteristics and wetland hydrologic metrics across all sites.

Metric	*Elev*	ρ_d	*SOM*	*Clay*	*Silt*	*Sand*	*BC*	*BG*	*LT*	*Forb*	*Grass*
I	0.08	**-0.59**[a]	**0.65**[a]	0.27[c]	**-0.32**[b]	0.03	**0.36**[a]	**-0.33**[b]	0.04	**0.42**[a]	0.03
t_w	-0.05	0.26	-0.31[c]	**-0.50**[a]	0.07	**0.33**[b]	0.11	0.30[c]	-0.30[c]	**0.38**[b]	**0.45**[a]
t_{csat}	-0.11	0.08	0.07	0.24	**-0.58**[a]	**0.31**[b]	0.22	-0.22	-0.05	0.22	0.10
d_{csat}	-0.04	0.02	0.11	**0.36**[b]	**-0.61**[a]	0.25	0.16	-0.29[c]	0.05	0.08	-0.03
d_{sat}	0.05	**-0.37**[b]	**0.44**[a]	**0.70**[a]	-0.18	**-0.44**[b]	-0.24	**-0.48**[a]	**0.44**[a]	-0.27	**-0.66**[a]
f_{pond}	0.01	-0.01	0.04	**0.42**[a]	-0.02	-0.29[c]	-0.18	-0.15	0.20	**-0.40**[a]	-0.24
z_{pond}	-0.06	0.19	-0.11	0.20	-0.06	-0.07	-0.15	-0.01	0.09	**-0.42**[a]	-0.19
z_{gw}	-0.02	0.04	-0.09	0.21	0.14	-0.28[c]	-0.14	-0.09	0.11	**-0.57**[a]	-0.06
CV_{gw}	0.01	0.15	-0.06	0.29[c]	0.00	-0.20	-0.25	-0.28[c]	**0.37**[b]	-0.25	-0.12

Statistically significant values are in bold, and are of $p<0.05$ at a 95% confidence level.

[a] $p < 0.01$

[b] $0.01 < p < 0.05$

[c] $0.05 < p < 0.10$

$n = 42$ for hydroperiod metrics

CHAPTER 3. EFFECTS OF LAND USE ON SEASONAL GROUNDWATER-SURFACE WATER DYNAMICS IN MINERAL FLAT WETLANDS

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To be submitted for publication

Abstract

We used stable isotope tracers and hydrometric measurements to examine the effects of land use on seasonal surface and groundwater dynamics in mineral flat wetlands. For wetlands under least-altered prairie, farmed grass field, and restored prairie, we examined whether land use (1) affects overall wetland hydroperiod, (2) influences wet-season groundwater-surface water mixing processes, and (3) affects water loss processes for surface water, shallow groundwater, and deeper groundwater storage pools at the end of the wet season. Our results indicate that, in spite of land use differences and slight variations in soil stratigraphy, many similarities exist in overall wetland hydroperiod, water sources and evaporation rates for mineral flat wetlands in the Willamette Valley lowlands. Isotopic evidence suggests that while downward hydraulic gradients in the upper 2 m of the soil profile persist throughout much of the wet season, groundwater below the impeding layer is likely recharged during the fall and early winter rains, with little mixing or flushing of newer meteoric water as the wet season progresses. The greatest degree of groundwater-surface water mixing was observed in the upper 0.5 m of the saturated soil profile.

Introduction

Mineral flat wetlands were once common on lowland interfluves and floodplain terraces in temperate regions of the United States (Brinson et al. 1995). Drainage associated with agricultural land development has greatly diminished the area of least-altered wetland flats across the US (Daggett et al. 1998; Brinson and Malvarez 2002; Rheinhardt et al. 2002; Taft and Haig 2003; Dahl and Allord 2004). In regions such as the Willamette Valley, Oregon, former wetland flats often persist as seasonally saturated or inundated areas on farmed land (Daggett et al. 1998; Taft and Haig 2003). Riparian and depressional wetlands in agricultural catchments may provide critical ecological functions such as water storage and delay, and nutrient retention at the catchment scale (e.g. Lowrance et al. 1984; Zedler 2003; Montreuil and Merot 2006; DeSteven and Lowrance 2011; Fennessy and Craft 2011). Wetland flats may also be important water and nutrient storage features in agricultural catchments if they maintain wetland hydrology during all or part of the wet season.

With increasing public and private resources dedicated to restoring farmed mineral flat wetlands through initiatives like the Wetlands Reserve Program (De Steven and Lowrance 2011), there is a need to evaluate how farmed and restored wetland hydrology compares to that of least-altered wetlands. The hydrology of forested wetland flats has been documented by several studies along the Gulf Atlantic Coastal Plain of the U.S. (McCarthy et al. 1991; Sun et al. 1998; Sun et al. 2001; Xu et al. 2002; Ehrenfeld et al. 2003; Stander and Ehrenfeld 2009). Still, little information exists on field-scale hydroperiod timing and duration, groundwater-surface water interactions, and relative seasonal contributions of precipitation, groundwater recharge, surface runoff, and evaporation to the hydrologic budget of grassland wetland flats.

In lowland agricultural catchments of the Willamette Valley, Oregon, a unique research opportunity exists to study mineral flat wetlands under least-altered prairie, farmed grassland, and restored prairie land use. Mineral flats in the Willamette Valley and the Gulf Atlantic Coastal Plain share nearly-flat topography, precipitation as a dominant water source, and surface runoff, groundwater recharge, and evapotranspiration as primary water loss pathways (Brinson et al. 1995). Willamette Valley flats experience wet winters and summer droughts, and have a shallower restrictive layer, more silt and clay in the near-surface soil profile, and generally lower soil permeability than their eastern counterparts.

Transitions between dry and wet seasons within wetland-dominated catchments are closely tied to soil and groundwater recharge, and the onset of surface runoff generation (Lindsay et al. 2004). With the first fall rain storms, Willamette Valley flat soils transition from dry, cracked conditions to field saturation and shallow inundation at the soil surface. Research in higher elevation hillslopes of the Willamette River basin has shown that tightly-bound soil water is recharged during the first fall rains, and stored until it is used by trees via transpiration in the spring (Brooks et al. 2010). Given the low topographic gradient of mineral flats in the Willamette lowlands, it is possible for shallow groundwater recharged in the fall to be stored in the soil profile until spring, or vertically or laterally displaced by successive rainfall events during the winter. Shallow confining and semi-confining layers in wetland flats are generally assumed to limit groundwater recharge, and may minimize leaching of anthropogenic nutrients and pesticides to the regional aquifer in farmed catchments (Thomas et al. 1992). Still, preferential flow paths may facilitate groundwater recharge and solute transport through a shallow confining layer (e.g. D'Amore et al. 2000; Ronkanen and Klove 2007; van der Kamp and Hayashi 2009; Janssen et al. 2010), particularly in shrink-swell clay soils where soil cracks may remain open into the early wet season (D'Amore et al. 2000).

Once precipitation recharges shallow groundwater in the fall, the water table remains at or near the ground surface throughout the wet season (Finley 1995). Partitioning of wet season storm flow between soil and surface water pools has important implications for nutrient processing, particularly nitrate reduction on farm sites (e.g. Wigington et al. 2003), as well as the volume and quality of water exported downstream. For many East Coast flats, surface infiltration rarely limits water storage during the wet season, and saturation excess overland flow is the most common runoff generation process (Hernandez et al. 2003; Martinez et al. 2008). Infiltration rates in farmed Willamette Valley wetland flat soil may be low enough to cause Hortonian excess overland flow (Horton 1933) during the transition from dry to saturated soil conditions in the fall, in drier periods during the winter, and as soil transitions to unsaturated conditions in the spring (Marshall and Tullos in prep).

As summer approaches, vertical drainage and evapotranspiration gradually remove water from the root zone of Willamette Valley flats. The timing and duration of saturation during the growing season dictate a wetland's legal protection in the U.S. (USACE 1987), and directly influence soil biological and biogeochemical processes, along with plant community structure (Mitsch and Gosselink 2000). In seasonal wetlands, the degree of connectivity between the shallow subsurface water pool and deeper groundwater has been shown to affect the timing of de-saturation in the upper soil profile (e.g. Skalbeck et al. 2009). Depth to the water table also influences evaporative losses and plant-available water. The extinction depth, or depth in the soil profile where groundwater evapotranspiration is near-zero, may reach approximately 4 to 5 m below ground in fine-textured soils (Shah et al. 2007).

When combined with hydrometric measurements, stable environmental isotopes, such as $\delta^{18}O$ and $\delta^{2}H$, provide insight into evaporation and other key wetland surface and groundwater processes (e.g. Ladouche and Weng 2005; Ronkanen and Klove 2007; Sikdar and Sahu 2009). In this study, we used stable isotope tracers and hydrometric measurements to examine the effects of land use on seasonal surface and groundwater dynamics in mineral flat wetlands. For wetlands under least-altered prairie, farmed grass field, and restored prairie, we asked: does land use (1) affect the overall wetland hydroperiod (timing and duration), (2) influence wet-season groundwater-surface water mixing processes, and (3) affect water loss processes for surface water, shallow groundwater, and deeper groundwater storage pools at the end of the wet season? We hypothesized that least-altered wetlands would reach saturation later in the water year and have a longer hydroperiod than farmed and restored wetlands; farmed and restored sites would have less overall interaction between shallow and deeper groundwater pools during the wet season; and that evaporation would be a dominant water loss process for all sampled water pools, across all land uses in late spring.

The results of this work contribute to a better understanding of how land use affects hydrologic processes governing overall wetland hydroperiod, biogeochemistry, plant community structure, and wetland contribution to surface and subsurface water quality and quantity in adjacent wetlands and uplands (Mitsch and Gosselink 2000). Additionally, we provide information on the field-scale hydrologic resiliency of wetland flats to agricultural and restoration-related anthropogenic disturbance.

Methods

Study Area

Our research focused on seven mineral flat wetlands of similar soil and hydrogeomorphic type, but different land management (least-altered wetland prairie, non-irrigated, farmed grass field, and restored prairie) in the southern Willamette Valley lowlands of Oregon (Fig. 1, Table 1). Sites are located in paired groups within the Amazon Creek, Coyote Creek, and Muddy Creek (Finley National Wildlife Refuge area) catchments (Fig. 1). Herein, each site is labeled as the site group name, followed by the land use (e.g. Coyote Prairie, Coyote Farm, Coyote Restored). The catchment area above each site is <200 km². Mean annual temperatures in the study area are 3°C in January and 19°C in July (WRCC 2011). Between 70% and 80% of the 102-127cm of average annual precipitation, and 41 cm of mean annual groundwater recharge in the region, occurs between November and April (PNW-ERC 2002; Conlon et al. 2005). Evapotranspiration rates in the Willamette lowlands are highest between April and July, and estimated annual evapotranspiration from lowland areas where the water table is within 3 m of the soil surface is approximately 40-50 cm (Lee and Risley 2002).

The seven study sites are located on Dayton and Natroy hydric soils, which are historically associated with mineral flat wetland prairie ecosystems. Dayton soil (fine, smectitic, mesic, Vertic Albaqualfs), which formed on Pleistocene glaciolacustrine deposits overlain by Holocene Willamette River alluvium, is extensive in the Willamette Valley lowlands on nearly flat prairie terraces (Balster and Parsons 1968; NRCS 2006). Natroy soil (very-fine, smectitic, mesic, Xeric Endoaquerts) intergrades with Dayton soil, and is common on alluvial terraces and fans in the southern portion of the Willamette Valley lowlands (NRCS 2006). In the upper 20 to 40 cm of the soil profile of both Dayton and Natroy soil, an abrupt textural change between the surface A/E and lower B horizons results in a perched groundwater table in the fall. An apparent water table forms during the November- April wet season, and shallow surface water may be present between December and April (Austin 1994; James and

Baitis 2003; NRCS 2006). The near-surface and perching soil horizons of the Dayton and Natroy soils contain smectite clay minerals, resulting in surface crack formation as these soils transition from saturated winter conditions to dry summer conditions. Current land cover on former prairie soils commonly includes ryegrass and other crops that tolerate soil saturation into the early part of the growing season (PNW-ERC 2002; Taft and Haig 2003). Farmed wetlands are typically disced and plowed, and lack characteristic native vegetation (Taft and Haig 2003).

Field Instrumentation and Monitoring

From October-August 2009-2011, we monitored biweekly groundwater elevations, surface ponding, and ditch water levels. Groundwater and surface water elevations were used to estimate vertical and lateral hydraulic gradients with distance from wetland edges with ditches. We obtained climate data from the National Climate Data Center weather station located within the Finley National Wildlife Refuge.

Each study site was instrumented with three nests of 2.5-cm diameter slotted PVC piezometers immediately above, within, and below the suspected perching horizon (approximately 0.5, 1, and 2 m). At Amazon Restored, we were only able to install the deepest piezometers to 170 cm, where we encountered extensive gravel in the soil matrix. Screen lengths were 30 cm for all 0.5-m piezometers, and 15 cm for the 1-m and 2-m depths. All piezometers were constructed and maintained according to technical standards for wetland hydrologic monitoring established by the US Army Corps of Engineers (2005). Nests were installed during fall, 2009, followed a transect normal to the dominant topographic gradient on site, and were located approximately 5, 35, and 65 m from each wetland's surface water outlet (Fig. 1). Two additional transects of 0.5-m and 1-m piezometers were installed at each site approximately 15 m away and parallel to the main piezometer nest transect.

To establish boundary and site conditions, topography within and adjacent to the area bounded by the well transects, locations of adjacent streams or ditches, and elevations and locations of all wells and piezometers were established using RTK-GPS with >1.5 cm accuracy. Soil stratigraphy was documented during the piezometer installation process and soil samples were collected from each soil horizon for hydrometer particle size analysis (Day 1965; Gee and Bauder 1986; Gee and Or 2002) in the lab.

Water Sample Collection and Laboratory Analysis

During the 2010-2011 water year, we collected surface water and groundwater samples from all sites on January 20-21, 2011 and April 22-23, 2011. Weekly isotopic values for local rainfall were obtained from the USEPA National Health and Environmental Effects Laboratory—Western Ecology Division (WED), located approximately 15 km from the Finley sites and 60 km from the Coyote and Amazon sites in Corvallis, Oregon. If surface water was present, water samples were collected from ditches upstream and downstream of each site, as well as standing surface water near the piezometer nests. Surface water was sampled before groundwater was pumped out of piezometers to avoid contamination. Groundwater samples were collected from 0.5-m, 1-m, and 2-m piezometers using a plastic marine hand pump. After measuring the current water level, we purged each piezometer of water and allowed fresh groundwater to re-enter the screen. We also pulled a minimum of one liter of sample water through the pump prior to sample collection to avoid contamination between piezometers. Water samples were collected in 20-ml glass scintillation vials and capped immediately with poly-seal cone caps to minimize isotopic fractionation during sampling, transport, and storage. Isotope samples were pipetted into glass vials and run during the following 24-36 hours with a LGR Liquid-Water Isotope Analyzer.

Isotope data were post-processed using LGR LWIA Post Analysis Software (LGR 2010), and are reported as per mil (‰) $\delta^{18}O$/ $\delta^{16}O$ relative to Vienna Standard Mean Ocean Water (VSMOW). Isotope values were compared across sites and water storage pools, and with the Global Meteoric Water Line (GMWL; Craig 1961). The GMWL equation (Craig 1961):

$$\delta^2H = 8\ \delta^{18}O + 10$$

describes the relationship between δ^2H and $\delta^{18}O$ isotope values in precipitation samples from around the world. $\delta^{18}O$ values are referred to as more "depleted" or "enriched" moving along the GMWL toward lower $\delta^{18}O$ and δ^2H or higher $\delta^{18}O$ and δ^2H values, respectively. Water becomes isotopically fractionated, and departs from the GMWL as lighter water isotopes ($\delta^{16}O$ and δH) are preferentially removed with water vapor during evaporation (Kendall and Caldwell 1998). Evaporative fractionation occurs at the highest rate when ambient humidity approaches zero (Kendall and Caldwell 1998).

Results

Precipitation

Total precipitation for the southern Willamette Valley lowlands, obtained from the weather station at Finley National Wildlife Refuge, was 1005 mm in 2009-2010 and 1027 mm in 2010-2011 from October 1-July 1. Precipitation for the month of October was 69 mm in 2009 and116 mm in 2010. Average weekly rain isotope values obtained from WED were -8.61‰ $\delta^{18}O/\delta^{16}O$ (range -5.22‰ to -11.66‰) between November 1, 2010 and May 1, 2011 (USEPA 2011). The average isotopic composition of weekly rainfall during the fall 2010 wet-up (October 26-December 1) was -8.78 ‰ $\delta^{18}O/\delta^{16}O$ (range -7.13‰ to -10.0‰).

Hydroperiod timing and duration across land use

During both water years, we observed perched water tables within the top 0.5 m of the soil profile across all sites, and seasonal water tables across all sites fluctuated between >2 m below the ground surface (September-October) and at or above the ground surface (December-April; Fig. 2). Compared to 2009, greater October precipitation in 2010 was associated with earlier perched water table formation and a more rapid rise in deeper groundwater levels across all sites (Fig. 2). Once sites became fully saturated above the impeding layer, saturation excess, contributing surface runoff from wetland fields to ditches, occurred under all land use conditions. For both monitoring years, downward hydraulic gradients between 0.5-m and 2-m piezometers were indicative of groundwater recharge for all sites during most of the wet season (Fig. 2). With the exception of the Finley area sites, downward hydraulic gradients generally decreased with time during the 2010-2011 wet season, (Fig. 2). In both water years, early June storms caused the otherwise-declining water table to rebound to within 0.5 m of the ground surface across all sites. Amazon Restored and Coyote Restored had the greatest vertical hydraulic gradients (groundwater recharge) and the shortest hydroperiods of all seven sites during both the 2009-2010 and 2010-2011 water years.

Coyote Creek Catchment

In 2009, we observed a perched water table within 30 cm of the soil surface beginning on November 18, 2009 at Coyote Farm—two weeks before we observed any shallow groundwater at Coyote Prairie and three weeks before Coyote Restored (Fig. 2). A perched water table formed at Coyote Prairie on December 3, following a storm on November 26-27 (27.5mm of precipitation). In 2010, the Coyote Creek sites had the earliest wet-up of all seven sites, and individual Coyote sites wet up approximately a month earlier than the 2009 wet-up date. As in 2009, we observed perched water tables at Coyote Farm before both Coyote Prairie and Coyote Restored in fall, 2010.

During both water years, we observed the most rapid rise in 2-m groundwater at Coyote Prairie, followed by Coyote Farm (Fig. 2). Prairie 2-m groundwater reached the ground surface in early March in 2010, and early January in 2011. Farm site and Restored site 2-m groundwater did not reach the ground surface in either water year, and Restored site 2-m piezometers were dry for most of the wet season in both years. In late spring, Restored site groundwater had the most rapid water table drawdown, with piezometers at all depths dry by early July. Prairie site groundwater recession was slightly faster than Farm site water table recession in both water years.

Amazon Creek Catchment

We first observed a perched water table at Amazon Farm and Amazon Restored on December 3, 2009, following the same November 26-27 storm that led to wet-up at several sites in the other study catchments (Fig. 2). In fall, 2010, perched water tables formed at the Amazon Creek sites approximately two weeks earlier than the fall, 2009 wet-up date. For the 2010-2011 water year, 2-m groundwater was first detected in 2-m piezometers approximately two months earlier than in 2009-2010 (Fig. 2). Piezometric data (Fig. 2) indicate that 2-m groundwater reached the ground surface at Amazon Farm in late June during spring, 2010 and in early March in spring, 2011. 2-m groundwater did not reach the ground surface at Amazon Restored in either water year. In late spring, Amazon Restored had a much faster rate of water table recession than Amazon Farm in both years.

Finley Sites

Near-surface saturation was first observed on December 3, 2009, at Finley Farm, with deeper groundwater reaching the ground surface in early to mid January, 2010. With the exception of piezometers nearest the ditch, 2-m groundwater at Finley Prairie did not reach the ground surface during the 2009-2010 water year. In summer, 2010, Finley Prairie groundwater receded at a more rapid rate than Finley Farm, and the upper 2 m of the soil profile was unsaturated by early August. The Farm site soil profile did not reach unsaturated conditions until late August.

In fall, 2010, Finley-area sites had an approximately 1 week earlier wet-up than in 2009 (Fig. 2), and Finley Farm became saturated in the upper 2 m of the soil profile approximately one week earlier than Finley Prairie. Both Finley sites had 2-m groundwater at or near the ground surface by mid-November, 2010.

Groundwater-surface water interaction across land use

All isotope samples collected from shallow groundwater and surface water plotted to the right of the GMWL on the $\delta^{18}O$ vs. $\delta^{2}H$ plot, indicating an evaporated water component across water pools, sites, and seasons (Fig. 3). When we combined the data for all sites, we observed a strong correlation between $\delta^{18}O$ and $\delta^{2}H$ values for the January sampling date (slope = 4.58 on the $\delta^{18}O$ vs. $\delta^{2}H$ plot; R^2= 0.84) and the April sampling date (slope = 4.67; R^2= 0.93; Fig. 3). Most samples became more enriched in $\delta^{18}O$ from winter to spring (Fig. 3). For individual sites, $\delta^{18}O$ vs. $\delta^{2}H$ water line slopes ranged between 4.11-6.12 in January and 3.73-6.88 in April (Fig. 3). A decrease in water line slope was observed at all farmed and restored sites between January and April, indicative of increased evaporation. Prairie sites had a slight increase in water line slope between winter and spring, indicative of a greater meteoric water contribution compared to restored and farmed sites. Overall, water from all sampled wetlands appears to follow a mixing line between more $\delta^{18}O$-depleted meteoric water stored in the soil at the beginning of the wet season and more evaporated, $\delta^{18}O$-enriched water in the spring (Fig. 3).

Surface water isotopic composition reflected greater meteoric water inputs during the winter and evaporation in the spring. Across the surface water and groundwater storage pools, we observed the greatest range in $\delta^{18}O$ in spring surface water (4.03‰ for ditch water and 4.52‰ for field surface water), and the smallest range in winter and spring 2-m groundwater (1.81‰ and 1.58‰, respectively; Table 2). Winter field surface water was similar across all sites (range 1.98‰ for all sites; Table 2). Field surface water was the most depleted in $\delta^{18}O$ for the winter sampling period (mean -9.01‰ ± 0.63 s.d.) and most enriched in $\delta^{18}O$ (mean -5.18‰ ± 1.68 s.d.) during the spring, compared to ditch water and all sampled groundwater pools (Table 2).

For the groundwater storage pools, we observed the greatest variation in the isotopic composition of 0.5-m groundwater across sites and seasons (winter mean -8.82‰, range 3.19‰; spring mean -7.87‰, range 3.59‰; Table 2). Where surface water was present, the isotopic composition of 0.5-m shallow groundwater was generally intermediate between surface water and 1-m groundwater (Fig. 4). Of the individual sites, Coyote Farm and Amazon Restored had the greatest range (3.16‰

and 2.55‰, respectively) in 0.5-m groundwater isotopic composition between winter and spring, while Finley Prairie and Coyote Prairie had the smallest range (0.52‰ and 0.75‰, respectively; Fig. 4). An overall shift to more enriched $\delta^{18}O$ values from winter to spring indicated that evaporated water was well-mixed with the 0.5-m groundwater pool as the study wetlands began to dry out in the spring (Fig. 4).

Groundwater within the impeding layer, as sampled from the 1-m and 2-m piezometers, showed little change in isotopic composition between winter and spring (Fig. 4). Average site $\delta^{18}O$ values for 2-m groundwater ranged from -7.86‰ to -8.90‰, with Coyote Prairie and Finley Prairie at each end of the range, respectively. Across sites, 1-m and 2-m groundwater isotopic values plotted closest to the meteoric water line, suggesting minimal evaporative influence during the sampling period.

Discussion

Hydroperiod timing and duration across land use

We observed the most pronounced difference in the timing of wet-up across land use between the Farm, Prairie, and Restored sites at Coyote Creek, and to some extent between the wet-up of Finley Farm and Finley Prairie. Later wet-up at the prairie sites, corresponding with a more rapid rise in deeper groundwater, suggests that preferential flow may be an important process at these least-disturbed wetland sites. We suspect that the higher clay content in the A/E and B soil horizons, along with a shallower impeding layer, led to a more rapid wet-up, as well as a slower rate of water table recession at the Coyote sites. The slower drainage rate of these sites is also likely tied to higher antecedent moisture conditions in the soil profile at the onset of fall run-off. For the Coyote and Amazon sites, we did not see any strong relationships between field infiltration rates and rates of wet-up or drainage across land use and study site groups.

Groundwater data from the Amazon and Coyote Restored sites suggest that recharge-dominated flats with less connectivity between shallow and deeper groundwater pools have a shorter hydroperiod, and effectively dry out faster in the spring than wetlands with greater groundwater mixing. In the case of drawdown of a regional aquifer, or stream channel incision (as is the case with the streams and ditches around Amazon Restored), our results suggest that mineral flat wetlands may maintain wetland hydrology, but have a shorter hydroperiod.

Wet-up and drainage processes across land use

The meteoric water line plots (Figs. 3) indicate that, regardless of land use and slight variations in soil properties with depth, many similarities exist in water sources and evaporation rates for mineral flat wetlands in the Willamette Valley lowlands. This isotopic evidence, in combination with observations of shallow water table dynamics for these sites, suggests that farmed grass seed and hayfield mineral flat wetlands in the study area are very similar to their least-altered and restored counterparts in terms of dominant water sources and evaporative losses. The increased slope of the spring water line on the $\delta^{18}O$ vs. $\delta^{2}H$ plot for Finley Prairie suggests that this wetland may have incorporated a higher amount of meteoric water than the other study wetlands. Also, the increase in water line slope from winter to spring at farmed and restored sites, vs. the decrease for the least-altered sites, suggests that the least-altered prairie sites may incorporate (store) more meteoric water throughout the winter than the more disturbed wetlands. We hypothesize that, similar to observations in other grassland wetlands (van der Kamp et al. 2003), anthropogenic soil disturbance may restrict infiltration of new water into the soil profile throughout the wet season, as well as reducing available soil water storage space (Bodhinayake and Si 2004).

Overall isotopic similarity across the different water storage pools during winter and spring suggests similar dominant hydrologic inputs and processes across our study sites, regardless of land use. Isotopic variation in 0.5-cm groundwater across sites and seasons suggests both incorporation of surface water into the near-surface soil water pool and different degrees of mixing across sites. Differences in spring surface water isotopic values across sites may indicate variation in evaporation rates and mixing of surface water and near-surface groundwater

Our findings suggest that while downward hydraulic gradients in the upper 2 m of the soil profile persist throughout much of the wet season, groundwater below the impeding layer is likely recharged during the fall and early winter rains, with little mixing or flushing of newer meteoric water as the wet season progresses. Similarities in 2-m spring groundwater isotopic values across sites suggest a similar water source for deeper groundwater at all the wetlands, most likely precipitation from the fall and early winter wet-up (Brooks et al. 2010). The greatest degree of groundwater-surface water mixing appears to occur in the upper 0.5 m of the saturated soil profile, where occasional drier periods during the winter may allow the water table to drop to the point where soil water storage is available for successive rainfall inputs.

Our isotopic data suggest that, near the end of the hydroperiod (January-April), evaporation is an important water loss process for surface water and groundwater above the impeding layer for wetland flats across a variety of land uses. Other studies have suggested that evapotranspiration is an important process controlling water table recession in Gulf Atlantic wetlands (e.g. McCarthy et al. 1991). We did not see a pronounced evaporation signal in isotope samples from 1-m and 2-m groundwater, as we had hypothesized. Had we sampled deeper groundwater later in the spring or summer, it is possible that deeper groundwater may have seen some influence from movement of near-surface groundwater through the soil profile. A more detailed study of Finley Prairie and Finley Farm isotopic composition throughout the wet season (Marshall, in preparation), however, suggested that wetland surface water isotope values began to become more enriched in $\delta^{18}O$ in the beginning in March, which would have allowed several months for more evaporated water to percolate to 1-m and 2-m depths.

Conclusions

Our findings suggest that mineral flat wetlands under a variety of grassland land uses, ranging from least-altered prairie to farmed grass fields and restored prairie, can have a similar hydroperiod and degree of shallow groundwater-surface water interaction. We observed the greatest differences between farmed and least-altered sites at the beginning of the wet season, and shallow water table formation at farm sites occurred earlier than least-altered wetlands. Sites with high clay content in the near-surface soil profile had the most pronounced time lag between farm site and least-altered site wet-up.

Environmental stable isotopes were a useful tool in understanding key wetland seasonal water inputs and losses, along with groundwater-surface water interactions. Mineral flat wetlands across all studied land uses had a minimal amount of seasonal variation in the isotopic composition of groundwater below the perching layer, suggesting that the deeper part of the soil profile absorbs and retains water from the early part of the wet season. Shallow groundwater, however, showed greater seasonal variation in isotopic composition, switching from $\delta^{18}O$ values closest to meteoric inputs during the winter to values departing from the meteoric water line and indicative of evaporation in the spring.

Investigation of deeper groundwater behavior, along with soil profile analysis, is important in understanding the near-surface hydrology of wetland flats under different land uses, as well as selecting restoration sites that will sustain a hydroperiod similar to their least-altered counterparts. Further examination of soil water isotope values, throughout the year, would help shed light on unsaturated processes within these wetlands, and the degree to which mobile groundwater mixes with water already stored in the soil profile during successive parts of the wet season.

Acknowledgements

We thank Laura and Jim Marshall, Alicia Omlid, Leah Wilson, Terrence Morris, Diana Sharps for assistance with the installation and monitoring of water table wells and piezometers. We are grateful for feedback from Jeff McDonnell and Jack Istok related to the study design, particularly with respect to stable isotope hydrology and sampling, and groundwater hydraulics and monitoring techniques. We thank Renee Brooks from the US Environmental Protection Agency for sharing precipitation isotope data. Thanks also to Jock Beall, Jarod Jebousek, and Molly Monroe from the US Fish and Wildlife Service, Diane Steeck, Trevor Taylor, and Paul Gordon from the City of Eugene, Wes Messinger from the US Army Corps of Engineers, Chuck Stolsig, and Tom Danelski for help with field logistics, site background information, permitting, and access to wetland study sites. This material is based upon work supported under a National Science Foundation Graduate Research Fellowship. Any opinions, findings, conclusions or recommendations expressed in this publication are those of the authors and do not necessarily reflect the views of the National Science Foundation.

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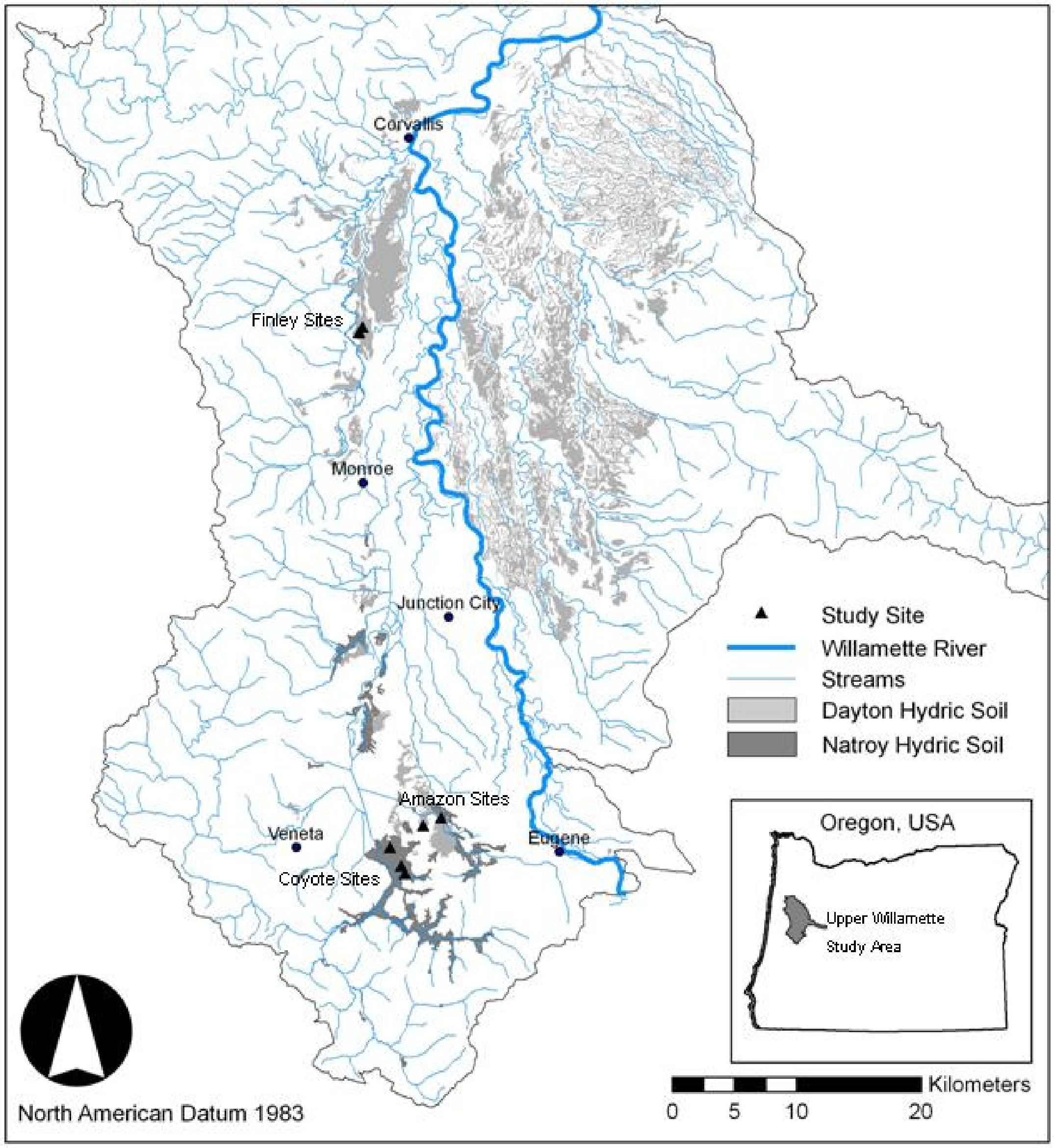

Figure 1. Study Area, including hydric soils associated with historic mineral flat wetlands in the Upper Willamette River lowlands, Oregon. From North to South, Finley sites include Finley Farm and Finley Prairie; Amazon sites include Amazon Restored and Amazon Farm; and Coyote sites include Coyote Prairie, Coyote Farm, and Coyote Restored.

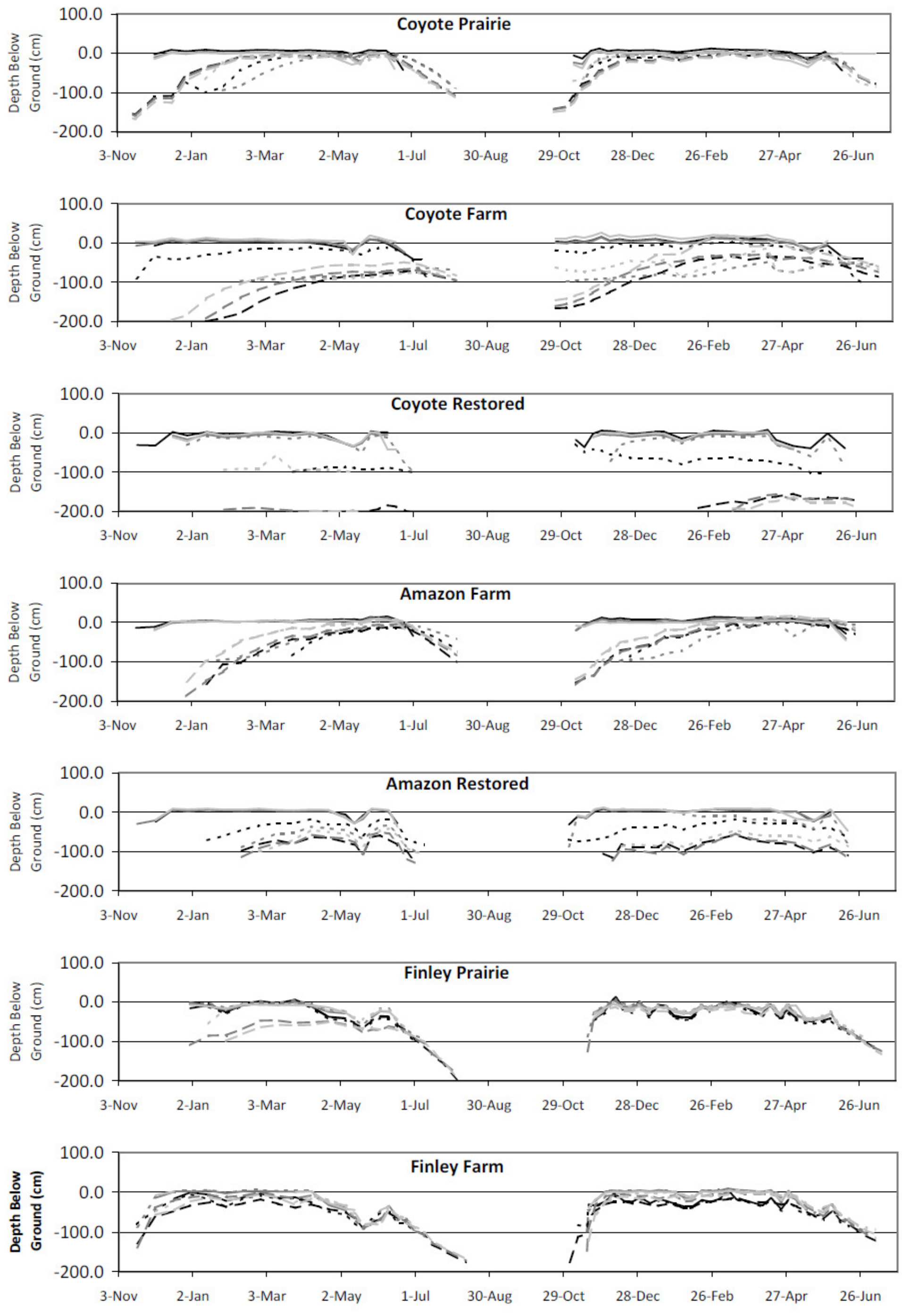

Figure 2. 2009-2010 and 2010-2011 water table elevations within the top 2-m of the soil surface across all seven study sites. MWT = 0.5-m monitoring well, P1 = 1-m piezometer, P2 = 2-m piezometer; 5, 35, and 65 indicate piezometer nests located 5, 35, and 65-m from the downslope field edge, respectively.

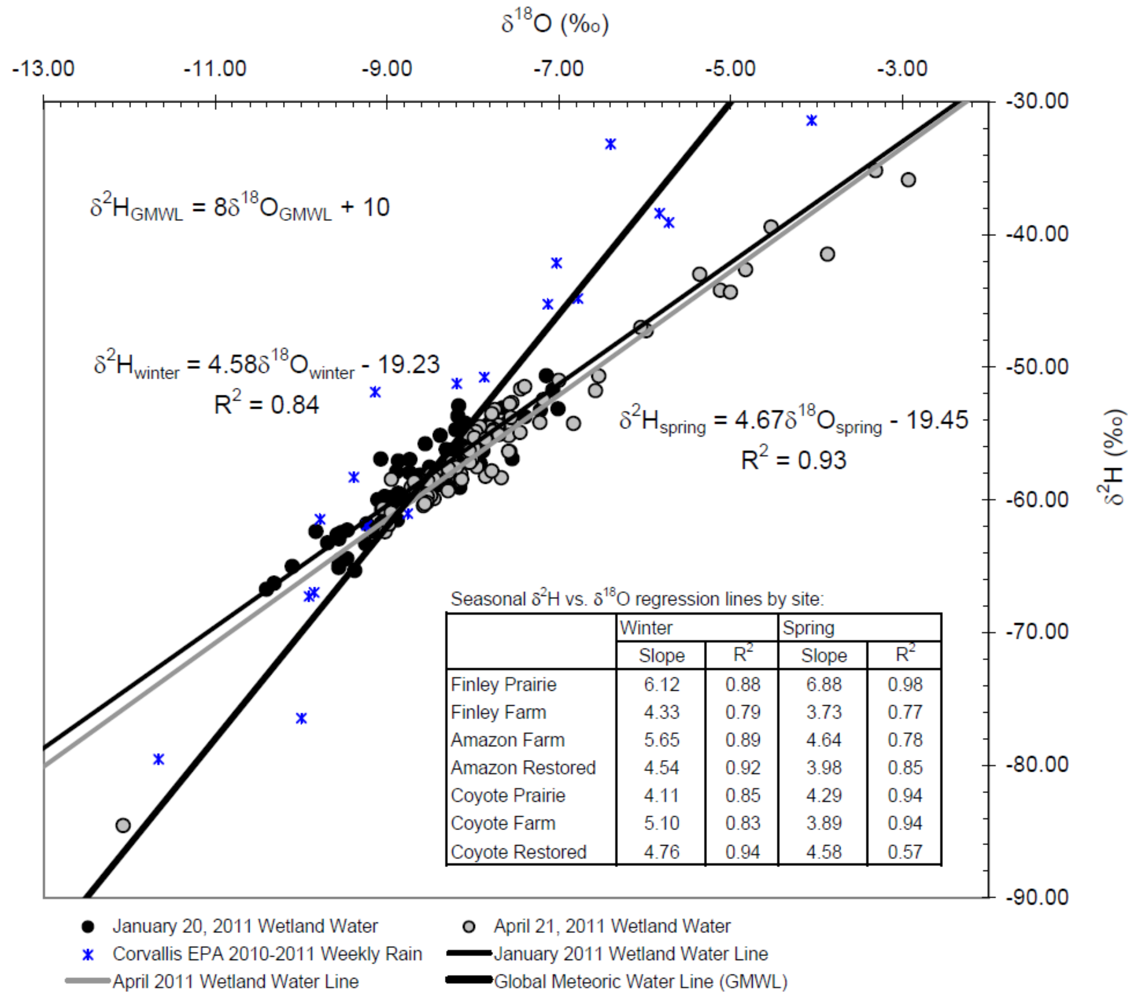

Figure 3. Relationships between δ^2H and δ^{18}O across seven mineral flat wetlands in the southern Willamette Valley lowlands during the winter (January 20, 2011 sample date) and spring (April 21, 2011 sample date) seasons. Inset table shows water line slopes for individual sites by season. δ^2H_{winter} and δ^2H_{spring} are the composite water line equations for all sites during the winter and spring, respectively.

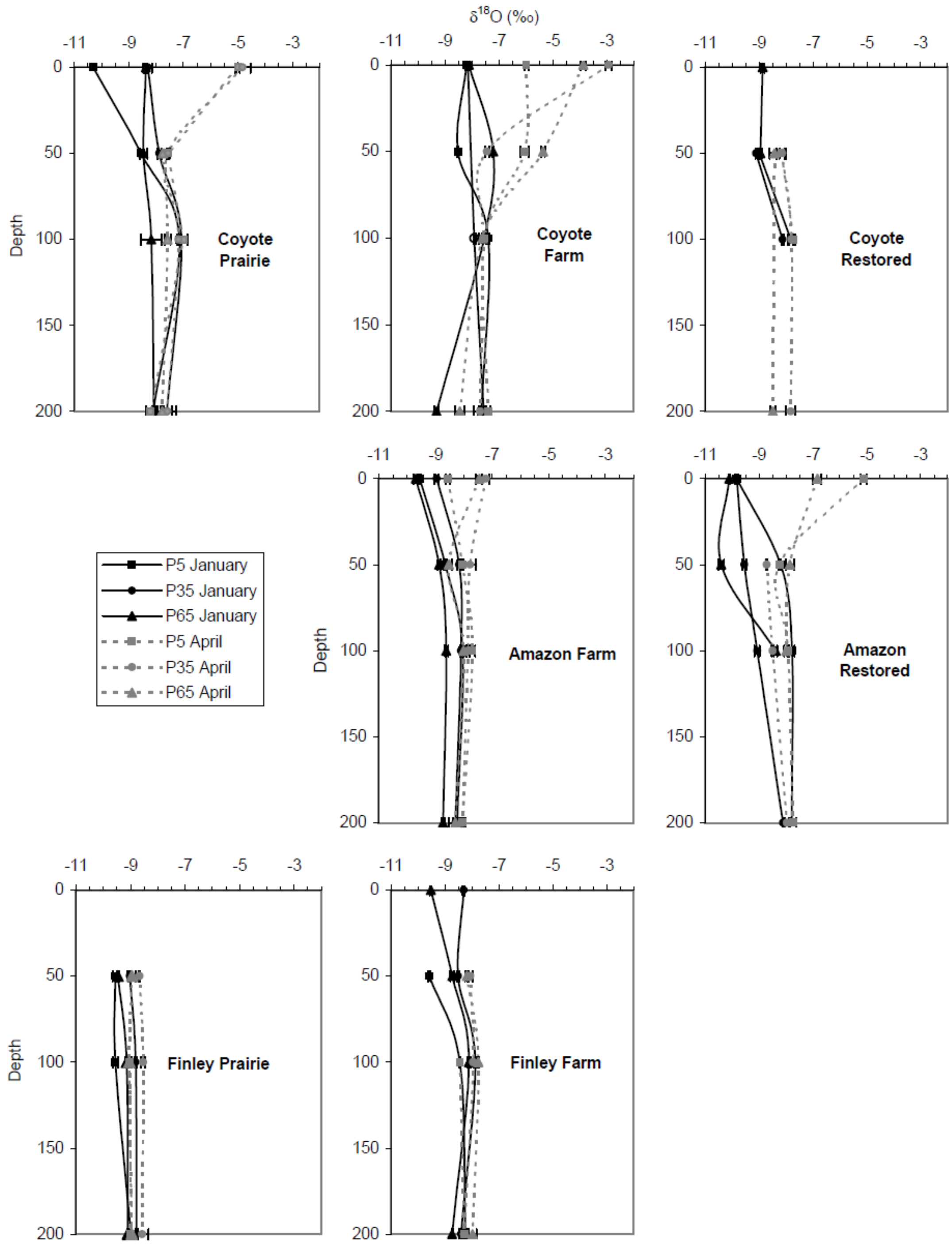

Figure 4. Seasonal groundwater $\delta^{18}O$ profiles with depth (cm) in the soil profile. Standing surface water, if it was present during sampling, is shown at a depth of zero. P5, P35, and P65 are piezometers located 5, 35, and 65-m from the downslope edge of each wetland.

Table I. Site Characteristics.

Catchment	Site Name (Ownership)	Land Cover	Size (ha)	Mapped Soil Series (NRCS)	Average Infiltration Rate (m/hr)	Average Bulk Density (g/cm^3)
Muddy Creek	Finley Prairie (USFWS)	Least-Altered Prairie	30	Dayton silty clay loam	1.26	0.91
	Finley Farm (Private)	Annual Ryegrass	32	Dayton silty clay loam	0.04	1.17
Amazon Creek	Amazon Restored (City of Eugene)	Restored Prairie; Former Ryegrass	16	Dayton silty clay loam	0.25	1.34
	Amazon Farm (Private)	Hay Field	8	Dayton silty clay loam	0.76	1.11
Coyote Creek	Coyote Prairie (USACOE)	Least-Altered Prairie	29	Natroy silty clay	3.51	0.97
	Coyote Farmed (City of Eugene)	Annual Ryegrass	11	Natroy silty clay	2.51	0.91
	Coyote Restored (City of Eugene)	Restored Prairie; Former Ryegrass	33	Natroy silty clay	0.75	1.05

Table 2. Seasonal variation in $\delta^{18}O$ values across the seven study sites.

	Sample Source	n	min (‰)	max (‰)	range (‰)	mean (‰)	standard deviation
January	0.5-m Groundwater	20	-10.40	-7.22	3.19	-8.82	0.70
	1-m Groundwater	20	-9.56	-7.07	2.49	-8.16	0.65
	2-m Groundwater	17	-9.38	-7.57	1.81	-8.33	0.55
	Ditch	14	-10.31	-8.19	2.12	-8.92	0.59
	Field Surface Water	14	-10.10	-8.12	1.98	-9.01	0.63
April	0.5-m Groundwater	19	-8.95	-5.36	3.59	-7.87	0.87
	1-m Groundwater	20	-9.05	-7.00	2.05	-7.94	0.54
	2-m Groundwater	19	-8.98	-7.40	1.58	-8.14	0.43
	Ditch	10	-8.56	-4.53	4.03	-7.02	1.40
	Field Surface Water	9	-7.45	-2.93	4.52	-5.18	1.68

CHAPTER 4. SEASONAL VARIATION IN RUNOFF SOURCES AND NUTRIENT TRANSPORT FROM NATIVE PRAIRIE AND FARMED MINERAL FLAT WETLANDS

Sarah M. Marshall and Desireé D. Tullos

To be submitted for publication

Abstract

Hydrologic connections between seasonal mineral flat wetlands, groundwater, and downstream surface water in lowland agricultural catchments are poorly understood, particularly under different land uses. We used a multiple source mass-balance analysis of δ^2H and $\delta^{18}O$ stable isotope tracers, water quality analysis, and shallow groundwater and surface water observations to investigate how land use affects seasonal variation in runoff sources and water chemistry in paired native prairie and farmed, former prairie mineral flat wetland catchments in an agricultural area of the Willamette Valley, Oregon. Our objectives were to examine whether surface runoff generated from prairie and farmed wetland sites comes from different storage pools (surface water, shallow soil water, or deeper groundwater) at the beginning, middle, and end of the wet season; and 2) how nitrate-nitrogen and other nutrient concentrations in downstream surface runoff are related to the timing and source of runoff. With the exception of a low-rainfall period in mid-winter, farm site runoff was isotopically similar to field surface water for most of the wet season, indicating that saturation excess was an important runoff generation process. Prairie site runoff was isotopically similar to upstream water throughout the winter, and briefly similar to shallow groundwater and surface water within the wetland in mid-spring. Throughout the wet season, elevated nitrate, sulfate, and chloride concentrations were observed in groundwater and surface water at the farm site, and deeper groundwater at the prairie site. Upstream-downstream runoff chemistry remained similar throughout the wet season at the prairie site. Farm site runoff chemistry reflected the dominant water source within the farm field throughout the wet season. Our findings suggest that, while surface water pathways dominate runoff from wetland flats under farm land use, large wetland flat fields (>10 ha) still have a high potential to absorb, store, and process nutrients and agrochemicals from on-site and nearby off-site chemical inputs. Mineral flats that maintain wetland hydrology in spite of farm use represent a unique balance between agricultural production and preservation of some of the water storage and delay functions, as well as water

quality-related ecosystem services once provided at a much larger scale in the Willamette Valley lowlands.

Introduction

Land use change in the United States during the last four centuries has led to the conversion, fragmentation, and loss of over half of pre-European settlement wetland area (Dahl and Allord 2004). Lowland prairie, savanna, forest, and emergent marsh wetland ecosystems have been especially impacted by drainage for agricultural land use (Dahl and Allord 2004). Mineral flat wetlands, a hydrogeomorphic class of wetland located on nearly flat floodplain terraces, interfluves, and large former lake bottoms (Brinson et al. 1995), were regularly converted to agricultural and silvicultural use due to their flat topography and seasonal drying cycles. Following centuries of fire suppression and land use conversion, least-disturbed mineral flat wetlands, including the wet pine flatwoods, pine savanna, and hardwoods of the Gulf Atlantic Coastal Plain and wetland prairie in the Willamette Valley, Oregon, are now some of the rarest ecosystems in their respective regions (Rheinhardt et al. 2002; PNW-ERC 2002).

While land use change has converted many mineral flats to uplands, some flats maintain wetland hydrology under agricultural (Taft and Haig 2003) or silvicultural (Sun et al. 2001; Xu et al. 2002) land use. Several studies have examined the hydrology of flats managed for timber production (McCarthy et al. 1991; Sun et al. 1998; Sun et al. 2001; Xu et al. 2002) and in urban areas (Ehrenfeld et al. 2003; Stander and Ehrenfeld 2009), but we currently lack information on how farmed mineral flats compare with their least-altered counterparts in terms of hydrology and water quality (Rheinhardt et al. 2002). Farmed flats that maintain wetland hydrology, in spite of drainage ditches and other local hydrologic modifications, may a) store and delay wet-season surface runoff as seasonally farmed wetlands, or b) be targeted by state, federal, and private land stewards for restoration to native wetland plant communities (De Steven and Lowrance 2011). In light of efforts to conserve and restore wetlands at the catchment scale, and better understand the hydrologic and biogeochemical ecosystem services provided by these wetlands under a variety of land uses, there is a need investigate the processes governing how mineral flat wetlands affect downstream water quality and quantity in lowland agricultural catchments.

Wetland flats are fed primarily by precipitation and dominated by vertical flow, though near-surface, low-hydraulic conductivity soil horizons may impede water movement through the soil profile (Brinson et al. 1995). Water is lost from mineral flats via evapotranspiration, saturation-excess overland flow, and groundwater recharge (Brinson et al. 1995; Ehrenfeld et al. 2003; Marshall and Tullos in prep a). Both seasonal surface water and groundwater connections may play an important role in how mineral flats affect downstream water quantity and quality (Tiner 2003; Whigham and Jordan 2003). Under least-altered conditions, mineral flat wetlands are not likely to export substantial quantities of nutrients downstream. Least-altered flats have diverse microtopography and seasonal wet and dry cycles that allow for a variety of soil biogeochemical environments, and are adapted to rapid nutrient cycling following regular fires (Rheinhardt et al. 2002). Under farm use, wetland flats may store and process or rapidly deliver agricultural chemicals to streams depending on rates of soil water infiltration and recharge, timing and duration of field saturation, and the degree of connectivity between fields and receiving surface waters (e.g. Wigington et al. 2003). Within agricultural catchments, hydrologic connectivity between mineral flats under both land uses and nearby streams may be artificially enhanced by drainage ditches and other anthropogenic extensions of the natural drainage network (Wigington et al. 2005).

Several hydrologic processes may be important in controlling the timing and source of wetland runoff from mineral flats under prairie and farmed land cover, including infiltration excess overland flow, saturation excess overland flow, and lateral flow in the shallow subsurface. Near the soil surface, soil and plant cover disturbance associated with agricultural land use has been shown to reduce near-saturated infiltration (e.g. van der Kamp et al. 2003) and soil water storage capacity, and may increase runoff generation relative to natural wetlands (Bodhinayake and Si 2004). When precipitation exceeds the infiltration rate of the soil, infiltration excess overland flow may occur (Horton 1933). Once soil is saturated to the ground surface, additional precipitation inputs generate saturation excess overland flow (Dunne and Black 1970). In fine-textured soils, saturation excess runoff may be compounded by seasonally shallow water table levels that respond more rapidly to precipitation due to capillary fringe effects (Gillham 1984). Saturation excess overland flow has been documented as a key wet-season runoff generation process in mineral flat wetlands along the Gulf Atlantic Coastal Plain (Hernandez et al. 2003; Martinez et al. 2008), and in Canadian boreal and subarctic catchments dominated by organic flat wetlands (Quinton and Roulet 1998).

Lateral flow has been identified as an important flow path in urban (Ehrenfeld et al. 2003) and forested (McCarthy et al. 1991) mineral flat wetlands, and may hydrologically connect mineral flat wetlands and adjacent uplands, wetlands, and streams. In the subsurface, lateral flow commonly occurs when a higher permeability soil layer overlies a low-permeability rock or soil layer (Kirkby 1969). Surface drainage ditches and subsurface tile drainage above low-permeability soil horizons, commonly employed in draining wetlands for agricultural use across the United States (Dahl and Allord 2004), have the potential to increase lateral flow from wetland flats and change the volume, timing, and chemistry of runoff (Havens et al. 2001). Drainage ditches have been shown to affect water table dynamics in areas still designated as wetlands. For example, Phillips et al. (2010) observed that ditches affected lateral drainage within 4 to 15 m of an adjacent mineral flat wetland in North Carolina depending on ditch elevation, while the water table further into the field was nearly flat during the wet season. Lateral soil drainage to drainage ditches has also been shown to account for up to 42% of water loss from managed forested mineral flats during seasons when evapotranspiration rates are lowest (McCarthy et al. 1991).

Lateral subsurface, vertical subsurface, and surface flow paths each bring water in contact with a distinct isotopic and biogeochemical environment. Water moving along a predominantly lateral flow path has greater contact with the groundwater/soil water and chemical pool, whereas direct or diffuse field-surface runoff will most likely reflect surface water chemistry and isotopic composition. In this study, we used a multiple source mass-balance analysis of δ^2H and $\delta^{18}O$ stable isotope tracers, water quality analysis, and shallow groundwater and surface water observations to investigate how land use affects seasonal variation in runoff sources and water chemistry in paired native prairie and farmed, former prairie mineral flat wetland catchments in an agricultural area of the Willamette Valley, Oregon. We asked: 1) does surface runoff generated from native prairie and farmed wetland sites come from different storage pools (surface water, shallow soil water, or deeper groundwater) at the beginning, middle, and end of the wet season; and 2) how are nitrate-nitrogen and other nutrient concentrations in downstream surface runoff related to the timing and source of runoff? We hypothesized that surface runoff from farm sites would be most isotopically and chemically similar to field surface water, whereas prairie runoff would be most similar to shallow groundwater.

Our results provide a better understanding of how land use affects seasonal runoff generation processes in mineral flat wetlands, and the water quality implications of modifying groundwater and surface water connectivity between mineral flats and surrounding surface drainage networks.

Methods

Study Area

Our research focused on two small, mineral flat wetland catchments of similar soil and hydrogeomorphic type, but different land management (least-altered wetland prairie vs. ryegrass farm) in the southern Willamette Valley of Oregon (44°26’ N, 123°18’ W; Fig. 1). Mean annual temperatures in this region are 3°C in January and 19°C in July (WRCC 2011). Average annual precipitation is between 102 and 127cm, 70-80% of which falls during November and April (PNW-ERC 2002). Most of the estimated 41 cm of mean annual groundwater recharge in the lowlands also occurs between November and April (Conlon et al. 2005).

Wetland prairie flats, typified by low-permeability mineral soils and seasonal saturation and inundation, once played a key role in intercepting and storing precipitation and runoff on the Willamette Valley floor (PNW-ERC 2002). Today, less than 1% of the estimated 121,488 ha of original wetland prairie remains in the Willamette Valley (Christy 2000, PNW-ERC 2002, Taft and Haig 2003). Grass and grass seed crops, along with some pasture and hay production, account for approximately 66% of the 569,000-ha farm area in the Willamette River Basin (PNW-ERC 2002), and are common on former wetland soils. Farmed wetland soils are typically plowed and lack characteristic native vegetation, but may retain similar hydrologic properties to least-altered prairie wetlands (Taft and Haig 2003).

The two study sites are part of the same large Dayton soil unit, and are separated only by current and abandoned gravel service roads and associated drainage ditches. Dayton soils (fine, smectitic, mesic, vertic albaqualfs), formed on Pleistocene glaciolacustrine deposits overlain by Holocene Willamette River alluvium, are extensive in the Willamette Valley lowlands on nearly flat prairie terraces (Balster and Parsons 1968; NRCS 2006). An abrupt textural change between the silt loam surface A/E and lower silty clay Bt horizons in the upper 0.5 m of the soil profile (herein referred to as the perching layer) results in a perched groundwater table in the fall, saturation throughout the soil profile between the months of November and April, and some surface water ponding between December and April (Austin 1994; NRCS 2006).

Prairie catchment

Finley Prairie is a 30-ha site located within the Willamette Floodplain Research Natural Area of the William L. Finley National Wildlife Refuge, and is part of the largest remaining wetland prairies in the Willamette Valley. Hydric soils and wetland hydrology support a characteristic native wetland prairie plant community consisting of a variety of native grasses (including *Deschampsia caespitosa*, *Hordeum* *brachyantherum, beckmannia syzigachne*, and *Alopecurus geniculatus*), several species of sedge (*Carex spp.*) and rush (*Juncus spp.*), scattered trees and shrubs (*Fraxinus latifolia, Crataegus douglasii*), and seasonally abundant wildflowers (including *Camassia leichtlinii*, *Saxifraga oregana*, *Geranium oreganum*, *Wyethia angustifolium*, and *Clarkia amonea*). Introduced *Rosa eglanteria*, *Agrostis spp., Holcus lanatus,* and *Hypericum perforatum* are also prevalent throughout the prairie.

Like other wetland prairie ecosystems in the Willamette Valley lowlands, Finley Prairie was historically maintained by fire, and has been grazed and burned over the last century to prevent the encroachment of shrubs and trees (Johannessen et al. 1971; Clark and Wilson 2001). There is no evidence of past plowing, though the site experienced some cattle and other livestock grazing through the mid-1960s (Streatfeild 1995). Average soil bulk density for the prairie is 0.91 g/cm^3 (0.57-1.12 g/cm^3 range) and average infiltration rate is 1.26 m/hr (0.22-6.08 x 10^1 m/hr range) (Marshall in prep b).

The ground surface at Finley Prairie is nearly level, with the exception of several large, slightly raised mounds, several slight depressions, and a shallow swale running from southeast to northwest across the prairie (Fig. 1). Hydrologic modifications to the site are minor, with shallow ditches flowing along the east and west side of the prairie, and a series of two deeper ditches between the north edge of the prairie, an old service road, and a gravel road that runs through the refuge. The intermittent (December-April) surface water flow path between the downstream ditch at the prairie site and Muddy Creek (Fig. 1) is approximately 530 m in length, with a 0.10 % average slope.

Farmed catchment

The 32-ha Finley Farm site has been used for seasonal waterfowl hunting and annual ryegrass (*Lolium multiflorum*) production since the 1960's. Historic aerial photos indicate that this site was farmed as early as 1936, and that drainage ditching was not prevalent in the area until 1970 (Streatfeild 1995). The site lacks drainage tiles or other subsurface drainage, but is surrounded by surface drainage ditches and gravel roads to the north, south, and west, a gravel driveway to the east, and an artificial pond created to attract waterfowl to the southeast (Fig. 1). Ditch water from the farm site flows into several culverts and vegetated surface ditches (average slope 0.20%) before flowing into the riparian area of Muddy Creek approximately 450 m from the downstream edge of Finley Farm. Average soil bulk density for the farm site is 1.17 g/cm^3 (1.00-1.31 g/cm^3 range) and infiltration rate is 0.04 m/hr (0.02-0.10 m/hr range), as measured in summer, 2010 (Marshall in prep b).

In 2010, the farm was enrolled in the USDA NRCS Wetlands Reserve Program. The field was last plowed, disced, fertilized, and planted with ryegrass in summer, 2009, and hay was last harvested from the site in spring, 2010. No new anthropogenic nitrogen or other fertilizer inputs have been added to the site since 2009. Estimates for past nutrient additions, based on average nitrogen application rates for non-irrigated Willamette Basin grass seed crops, are between 130-280 kg N/ha/year and 56-170 kg N/ha/year for other hay crops (Hinkle 1997). Also in the summer of 2010, a low berm was constructed around the inner portion of the farm field to create a shallow pond (approximately 0.1-0.5 m deep) during most of the wet winter season. Construction of the berm resulted in some surface soil disturbance, though the farm field remained under ryegrass cover and was not disturbed in the vicinity of our study area.

Field Instrumentation and Monitoring

Climate data were obtained from the National Climate Data Center weather station located within the wildlife refuge and approximately 1.5 km from the study area. Weekly isotopic values for local rainfall were obtained from the USEPA National Health and Environmental Effects Laboratory—Western Ecology Division (WED), located approximately 15 km from the study area in Corvallis, Oregon at an elevation of 76.2 m, approximately 3 m above the study area. Each rainfall isotope value represents the isotopic composition of the previous week's cumulative precipitation.

From October to August 2009 to 2011, we examined seasonal groundwater and surface water elevations with distance from the northwest corner of each wetland field. Study sites were instrumented with three nests of 2.54-cm diameter slotted PVC piezometers immediately above, within, and below the suspected perching horizon (~50, 100, and 2 m). Screen lengths were 30 cm for the 0.5-m piezometers, and 15 cm for the 1 and 2-m depths. All piezometers were constructed and maintained according to technical standards for wetland hydrologic monitoring established by the US Army Corps of Engineers (2005). Nests were installed during fall, 2009, followed a transect normal to the dominant topographic gradient on site, and were located approximately 5, 35, and 65 m from each wetland's surface water outlet (Fig. 1). Two additional transects of 50 and 100 cm piezometers were installed parallel to, and approximately 15 m away from the main piezometer transect. We monitored water table elevations biweekly during the 2009-2010 water year and weekly during the 2010-2011 water year. Piezometers were developed at the beginning of each monitoring season to ensure that screens were not clogged with fine sediment.

To establish boundary and site conditions, topography within and adjacent to the area bounded by the well transects, locations of adjacent streams or ditches, and elevations and locations of all wells and piezometers were established using RTK-GPS with accuracy >1.5 cm. We utilized soil stratigraphy and observations of seasonal groundwater elevations from the 2009-2010 water year (Marshall in prep a) to inform isotopic sampling.

Water Sample Collection and Laboratory Analysis

Environmental isotope tracers and hydrometric measurements have been successfully employed in a variety of wetland environments to determine water and nutrient flow paths, sources of surface runoff, and surface water interactions with underlying groundwater (e.g. Burns et al. 2005; Ladouche and Weng 2005; Ronkanen and Klove 2007; Sikdar and Sahu 2009). We collected water samples for environmental isotope and water quality analysis from groundwater, field surface water, and ditch surface water upstream and downstream of each Finley-area site (Fig 1). Groundwater was sampled from 0.5-m, 1-m, and 2-m piezometers using a plastic marine hand pump. Prior to sampling, water levels were measured, and then piezometers were purged of water and allowed to re-fill with fresh groundwater. When samples were collected, at least one liter of fresh well water was pulled through the pump prior to sampling to avoid contamination between piezometers. Water isotope samples were collected in 20-ml glass scintillation vials and capped immediately with poly-seal cone caps to minimize isotopic fractionation during sampling, transport, and storage. Water quality samples were filtered in the field or within 24 hours of collection using a vacuum pump with a glass microfibre prefilter and a nucleopore glass fibre 0.45-μm final filter. All filtration equipment was rinsed with deionized water prior to new sample collection to minimize cross-contamination. At least 50 ml of each filtered sample was poured into an acid-washed 125-ml sample bottle and frozen at -4° C until processed in the lab.

Water quality samples were brought to room temperature and then processed in the lab with a Dionex ICS-1500 ion chromatograph. Each sample run included one duplicate sample per ten water quality samples, check standards, and blanks to assure quality. When sample constituents exceeded the detection range of the chromatograph, original samples were diluted to bring each chemical within the detection range and then re-run. Isotope samples were pipetted into glass vials and run during the following 24-36 hours with a LGR Liquid-Water Isotope Analyzer. Isotope data were post-processed using LGR LWIA Post Analysis Software (LGR 2010), and are reported as per mil (‰) $\delta^{18}O/\delta^{16}O$ relative to Vienna Standard Mean Ocean Water (VSMOW). Isotope values were compared across sites and water storage pools, and with the Global Meteoric Water Line (GMWL; Craig 1961). The GMWL equation (Craig 1961):

$$\delta^2H = 8*\delta^{18}O + 10$$

describes the relationship between δ^2H and δ^{18}O isotope values in precipitation samples from around the world. δ^{18}O values are referred to as more "depleted" or "enriched" moving along the GMWL toward lower δ^{18}O and δ^2H or higher δ^{18}O and δ^2H values, respectively. Water becomes isotopically fractionated, and departs from the GMWL as lighter water isotopes (δ^{16}O and δH) are preferentially removed with water vapor during evaporation (Kendall and Caldwell 1998). Evaporative fractionation occurs at the highest rate when ambient humidity approaches zero (Kendall and Caldwell 1998).

End Member Mixing and Statistical Analysis

Multiple source mass-balance analysis for isotope samples was performed using the USEPA's IsoSource Version 1.3.1 mixing model software (Phillips and Gregg 2003). IsoSource calculates the range of feasible source contributions to a mixture (0 to 100% in user-specified small source increments) when the number of potential sources is too high for a unique solution to the mixing model to be developed (Phillips and Gregg 2003). Our input for the IsoSource model consisted of δ^2H and δ^{18}O isotope values for each surface water and shallow groundwater pool potentially contributing to downstream ditch runoff at the Finley Prairie and Finley Farm sites. We chose a source increment of 1%, and selected the lowest mass balance tolerance value within a range of 0.05%- 0.50% that produced at least one feasible solution for the mixing model. We reported the mean, standard deviation, and first and 99th percentiles for each potential source to capture the range of potential solutions. All other statistical analyses, including correlation analyses and a Principal Components Analysis (PCA; Pearson 1901; Hotelling 1933) were conducted with Statgraphics Centurion XVI.

Results

Seasonal Runoff Sources

Total precipitation for the study area, obtained from Finley National Wildlife Refuge was 834 mm from November 1-May 1, 2010-2011. Precipitation for the month of October was 116 mm in 2010. Average weekly rain isotope values obtained from WED were -8.61‰ $\delta^{18}O/\delta^{16}O$ (range -5.22‰ to -11.66‰) between November 1, 2010 and May 1, 2011 (USEPA 2011). Average precipitation values prior to the first isotope sampling date, and contributing to soil water recharge during the wet-up, were -8.78‰ $\delta^{18}O/\delta^{16}O$ (range -7.13‰ to -10.00‰).

Isotopic data for Finley Prairie and Finley Farm generally followed the Global Meteoric Water Line (GMWL; Fig. 2) during the study period. Beginning in mid-March, surface water samples from both sites diverged to the right of the GMWL during the spring, indicative of increased direct surface water evaporation (Fig. 2; Fig. 3a). Average groundwater isotopic composition at both sites was similar to average precipitation recorded during the sample period. Finley Prairie groundwater samples were tightly grouped around an average value of -8.80‰ $\delta^{18}O/\delta^{16}O$ (± 0.46 standard deviation) and -59.96‰ $\delta^{2}H/\delta^{1}H$ (± 3.09 sd). Finley Farm groundwater samples were clustered around a slightly less depleted value of -8.26‰ $\delta^{18}O/\delta^{16}O$ (± 0.64 sd) and -57.59‰ $\delta^{2}H/\delta^{1}H$ (± 3.31 sd).

Overall, isotope values for paired water pools (field surface, ditch, 0.5-m, 1-m, and 2-m groundwater) across Finley Prairie and Finley Farm were strongly correlated (n=123, Pearson $r = 0.83$, $p<0.01$). Throughout the sampling period, upstream ditch water samples were most strongly correlated across land uses (n=9, $r = 0.97$, $p<0.01$), and downstream ditch samples were correlated to a lesser degree (n=9, $r = 0.58$, $p<0.10$). Finley Prairie swale water and Finley Farm pond water were also highly correlated (n=8, $r = 0.93$, $p<0.01$). We did not observe any statistically significant ($p<0.10$) correlations between 0.5-m, 1-m and 2-m groundwater samples ($r < 0.20$ for each pool) across sites during the study period.

Finley Prairie

We first observed runoff in the Finley Prairie ditch following a storm on November 17-18, 2010. The shallow water table was within the upper 0.5 m of the soil profile at all Finley Prairie piezometer nests on November 24, but was not detected within 30 cm of the ground surface until December 2 (Fig. 3). On December 2, ditch water sampled downstream of Finley Prairie (-8.83‰ $\delta^{18}O / \delta^{16}O \pm 0.01$ standard deviation) was within 0.24‰ of 0.5-m groundwater nearest the ditch (-8.59‰ $\delta^{18}O / \delta^{16}O \pm 0.09$ sd), and nearly identical to water in the ditch along the west side of the prairie (1st percentile and 99th percentile = 1.00 in IsoSource). By mid-December, runoff sampled from the ditch below Finley Prairie was almost entirely derived from upstream ditch water (1st percentile, 0.96; 99th percentile, 0.97; Table 1).

Between January 19 and February 6, 2011, cumulative rainfall was below 2.2 mm. The shallow water table reached its lowest winter elevation around February 4 nearest the ditch, and on February 10 further into the field (Fig. 3). As the dry period progressed, Finley Prairie ditch water transitioned from mostly upstream water (1st percentile 0.89; 99th percentile, 0.97 on January 20) to a mixture of water from upstream and the swale running southeast to northwest across the prairie (1st percentile 0.55; 99th percentile, 0.55 for upstream water, 1st percentile 0.44; 99th percentile 0.44 for swale water on February 4; Table 1).

On March 18, downstream ditch water was most isotopically similar to 0.5-m groundwater (1st percentile 0.44; 99th percentile 0.95), corresponding with a low ditch water level similar to the dry period in January-February (Table 1; Fig. 3). On April 8, swale water was again an important contributor to ditch runoff (1st percentile 0.35; 99th percentile 0.60). When the swale became dry in mid-April, downstream ditch water was isotopically most similar to upstream ditch water for the remainder of the sampling period (Table 1) and became progressively more enriched in $\delta^{18}O$.

Finley Farm

We first observed shallow groundwater in Finley Farm piezometers within 5 m of the drainage ditch on November 10, 2010. A single storm event from November 17-18 led to soil saturation within the upper 0.5 m of the soil profile and the first observed fall runoff in the ditch downstream of the farm. Nearly all of the 0.5-m monitoring wells nearest the ditch had water within 30 cm of the ground surface on November 24 (Fig. 3), approximately one week before Finley Prairie reached jurisdictional wetland hydrologic criteria (<30 cm of the soil surface). The earliest isotope samples, collected on December 2, indicate that $\delta^{18}O/\ \delta^{16}O$ values for ditch water near the farm field outlet at Finley Farm (-8.64‰ ± 0.01 sd) were within 0.29‰ of water sampled from shallow monitoring wells nearest the ditch (-8.35‰ ± 0.02 sd). Surface water was not observed on the farm field until December 17, two weeks after surface water appeared on the Finley Prairie site.

After the onset of fall wet-up, on December 17, Finley Farm runoff was composed primarily of field surface water (1st percentile, 0.49; 99th percentile, 0.91) with a smaller contribution from upstream water (Table 1). As the wet season progressed and the farm field became more thoroughly saturated, field surface water impounded by the low berm was the primary contributor to downstream ditch water during wetter periods (Table 1). During the January 19-February 6, 2011 dry period, the shallow water table at Finley Farm reached its lowest winter elevation around February 4 nearest the ditch (Fig. 3b), and on February 10 further into the field (Fig. 3d). In the middle of the dry period, Finley Farm ditch water originated from a mixture of sources, with shallow groundwater having the highest likely mean contribution (1st percentile 0.34; 99th percentile, 0.67 on January 20; 1st percentile 0.42; 99th percentile 0.58 on February 4), along with smaller contributions from pond water and upstream ditch water (Table 1). When water table elevations were again within 30 cm of the soil surface after the dry period (Fig. 3), the contribution of pond water to downstream runoff increased between April 8 and April 23, but declined on

May 6 (Table 1). During this same spring period, the contribution of 0.5-m groundwater to downstream runoff gradually increased (Table 1).

Seasonal trends in water chemistry

At both Finley Farm and Finley Prairie, nitrate, chloride, and sulfate were detected in the 2-m groundwater at higher levels than other sampled water pools (Table 2). Nitrate-nitrogen, chloride, and sulfate levels at both sites were below the USEPA Drinking Water Standards for all sampled water pools, and all surface and groundwater P-PO_4^{3-} concentrations were below the USEPA-recommended level for preventing excess aquatic plant growth in streams (0.10 mg/L; USEPA 1986).

Across all measured water pools and sites, with time throughout the wet season, chloride in 0.5-m Finley Prairie groundwater was strongly correlated with 2-m groundwater ($r = 0.76$), field surface water ($r = 0.90$), upstream ditch water ($r = 0.81$), and downstream ditch water ($r = 0.88$) at the prairie site, along with upstream ditch water from the farm site ($r = 0.90$; $n = 7$ and $p< 0.05$ for all samples). Field surface water at Finely Prairie was closely related to upstream and downstream ditch water ($r = 0.98$ and $r = 1.00$, respectively, with $p<0.001$) and upstream Finley Farm ditch water ($r = 0.88$, $p<0.01$). With the exception of a strong correlation between Finley Farm field surface water and Finley Farm downstream ditch water ($r = 0.92$, $p <0.01$), we did not observe any statistically significant associations between Finley Farm field surface water, downstream ditch water, or 0.5-m, 1-m, and 2-m groundwater and other sampled water at Finley Farm and Finley Prairie.

Our PCA revealed two components that explained 81.7% of the variance in the water quality data for the entire sampling period (Fig. 4). The first component explained 57.6% of the variance and corresponded with a concentration gradient from low to high values for chloride, sulfate, nitrate, bromide, and fluoride (component weights of 0.50, 0.50, 0.47, 0.40, and 0.34, respectively). The second component explained 24.1% of the variance and corresponded with a concentration gradient for phosphate, fluoride, and bromide (component weights of 0.73, 0.51, and -0.35, respectively). Finley Prairie 0.5-m groundwater chemistry was most similar to surface water at both Finley Prairie and the Finley Farm sites for both PCA components (Fig. 4). Finley Farm 0.5-m groundwater was more similar to surface water at both sites than 2-m farm groundwater. Differences in Finley Farm groundwater and Finley Prairie Groundwater, especially from the 2-m sampling depths, followed the first and second component axes, respectively (Fig. 4).

Finley Prairie

At Finley Prairie, both chloride and sulfate concentrations in 2-m groundwater and sulfate concentrations in 0.5-m groundwater decreased throughout the wet season (Table 2). Across the sampling period, 2-m groundwater chloride concentrations were approximately one order of magnitude higher than 0.5-m groundwater, swale water, and ditch water (Table 2). Concentrations of sulfate in 2-m groundwater were nearly two orders of magnitude higher than other sampled water pools. When we tested 1-m piezometer groundwater along with 0.5-m and 2-m groundwater on April 8, 2011, we found a concentration gradient of increasing chloride with depth at the prairie site, but found no evidence of a similar trend with sulfate.

Throughout the wet season, chloride concentrations in Finley Prairie ditch water generally increased moving downstream (1.7-44%; Table 2). Ditch water sulfate concentrations increased slightly or remained the same moving downstream with the exception of February 4 and April 8 (37% and 76% decrease, respectively). Swale water had slightly higher chloride concentrations than upstream ditch water, and downstream ditch water chloride values were between swale and upstream concentrations (Table 2). Relative to Finley Farm, Finley Prairie had slightly higher phosphate concentrations in all surface and groundwater pools. On the January 7 and February 4 and 18, 2011, sampling dates, phosphate concentrations in Finley Prairie swale water were the highest for the sampling period (Table 2). These dates correspond with periods of decreased water table elevation and re-wetting following a drier period at Finley Prairie (Fig. 3).

Finley Farm

Finley Farm groundwater sulfate concentrations were nearly an order of magnitude higher than Finley Prairie during the early part of the winter wet season, and decreased with time during the sampling period (Table 2). Sulfate levels in 1-m groundwater (90.6 mg/L) were intermediate between 0.5-m and 2-m groundwater concentrations (Table 2). Nitrate-nitrogen levels showed a slight increase with depth between 1-m (2.9 mg/L) and 2-m groundwater (Table 2). Within the Finley Farm soil profile, 1-m groundwater chloride levels (26.8mg/L) were nearly double that of 2-m groundwater and nearly four times levels found in 0.5-m groundwater (Table 2). Chloride concentrations in 2-m groundwater remained relatively similar throughout the sampling period (Table 2).

Of the surface water quality constituents we sampled, chloride and sulfate had the highest concentrations. Nitrate concentrations were $\leq$0.05 mg/L and phosphate was <0.02 mg/L for the entire study period. Farm field surface water, a major contributor to downstream ditch runoff throughout the wet season, had higher chloride and lower sulfate concentrations than upstream ditch water. With one exception on April 8, ditch runoff from the Finley Farm showed an increase in chloride (40-150%) and a decrease in sulfate (1-78%) with distance downstream throughout the sampling period (Table 2). On April 8, chloride decreased by 11% and sulfate increased by 44% moving downstream (Table 2). At the same time we observed an increase in sulfate moving downstream on April 8, IsoSource mixing model output indicated an elevated contribution of 1-m groundwater to ditch water (1st percentile 0.01; 99th percentile 0.35; Table 2).

Discussion

Seasonal runoff sources

Our results suggest that land use, seasonal variations in precipitation, and antecedent field moisture conditions are all associated with variation in the source of wetland runoff for the mineral flat wetlands we studied. While flat topography and similar shallow water table elevations throughout the wet winter season might suggest consistent runoff sources and chemical composition, our isotope and chemical data indicate that small variations in precipitation and water table elevation are associated with shifts in dominant runoff sources and runoff water quality from farm sites, and to some extent, their least-altered prairie counterparts.

Early in the wet season, differences in isotopic composition of runoff between Finley Farm and Finley Prairie sites suggest that Finley Prairie was storing most precipitation inputs, and that the onset of runoff generation directly from the field was delayed relative to the Finley Farm site. Some of this delay may be accounted for by higher field infiltration rates (e.g. van der Kamp et al. 2003) and greater microtopographic variation in the Finley Prairie field, such as ovoid, low mounds and shallow pools at a larger scale as well as smaller-scale bunchgrass, sedge, and ant mound hummocks that increase roughness and tortuosity of surface flow paths. Finley Farm, potentially due to a combination of infiltration excess and saturation excess overland flow observed in other mineral flat wetlands (Hernandez et al. 2003; Martinez et al. 2008), along with reduced microtopographic complexity (Tweedy and Evans 2001), exported a greater proportion of field surface runoff and chemicals downstream.

Finley Farm runoff was isotopically similar to field surface water for most of the wet season, indicating that saturation excess was an important runoff generation process. Exceptions included the dry period in late January-early February, when shallow groundwater (likely via lateral flow) was the primary contributor to downstream runoff at Finley Farm. Groundwater contributions to ditch water also increased in late spring, likely because of reduced precipitation limiting surface ponding and runoff, and increased lateral hydraulic gradients between field groundwater and the ditch (e.g. McCarthy et al. 1991). Unlike Finley Farm, ditch water downstream of Finley Prairie remained isotopically similar to upstream water for most of the wet season. For approximately two months (March and April) during late winter/early spring, Finley Prairie runoff was isotopically similar to field surface water and shallow groundwater. The increase in contributions from field water at Finley Prairie corresponded with water table recession induced by a low-rainfall period, followed by spring storms re-wetting the field.

Seasonal trends in water chemistry

Through the course of our study, we found higher concentrations of nitrate, sulfate, and chloride in groundwater and surface water at Finley Farm. The presence and concentrations of these agriculture-associated chemicals suggests both short-term storage of agrochemicals in the deeper portion of the soil profile below farmed mineral flats and lateral movement of agrochemicals in the deeper, more permeable portion of the shallow, unconfined aquifer. Elevated nitrate, sulfate, and chloride concentrations in 2-m piezometers at the least-altered site suggest lateral migration of agrochemicals in shallow groundwater below the perching layer. Agrochemicals detected at both sites may have originated from other farm areas to the east and south of the study area. Phosphate, which is commonly associated with agricultural land use, was detected at higher concentrations in Finley Prairie groundwater and swale water, suggesting elevated background concentrations unrelated to local farm use. Given the high correlation observed between chloride concentrations in swale and shallow groundwater at the Prairie site, the adsorption of phosphate to clay particles (Froelich 1988), and the 0.45-μm filter size we used for water samples, it is likely that we sampled phosphate bound to clay particles moving in solution from shallow groundwater to the swale.

The decrease in groundwater chloride and sulfate concentrations at Finley Prairie throughout the season suggests either flushing or dilution of these chemicals in the shallow aquifer. Nitrate and sulfate concentrations in 2-m Finley Farm groundwater remained fairly consistent throughout the wet season, suggesting that these nutrients are primarily being stored, rather than exported or diluted from the farm system. Our findings of elevated chemical concentrations in the perching layer (1-m groundwater), may indicate that chloride and nitrate are stored in the perching layer and slowly leached into deeper groundwater. Unlike chloride and nitrate, sulfate concentrations increased with depth in the groundwater profile, suggesting removal of sulfate by plant uptake, leaching, and/or soil adsorption (Mitsch and Gosselink 2002) within and above the perching layer.

Elevated nitrate, chloride, and sulfate in Finley Farm groundwater, and very low concentrations in downstream ditch water suggest that a) farmed wetland flats may function as short or long-term sinks for some agrochemicals and b) that groundwater flow paths may be more important than surface water flow paths for transporting chemicals from mineral flat wetland fields to streams during the wet season. We should note that isotope and chemical data suggested a high contribution of field surface water to Finley Farm runoff throughout the wet season, and that nitrate and other chemicals stored at the soil surface may be readily transported to ditches and receiving streams without adequate soil contact (e.g. Wigington et al. 2003). If agrochemicals had been applied during the growing season preceding this study, we would expect to see elevated chemical concentrations in surface runoff (e.g. Hubbard and Sheridan 1989). Our findings of increased chemical transport corresponding with shifts to greater shallow groundwater input suggest that, as in many East Coast mineral flats (Thomas et al. 1992), lateral flow along a shallow perching layer can result in enhanced delivery of agrochemicals stored in the soil to nearby surface water.

Low concentrations of agrochemicals in ditch runoff may also be the result of several other factors. First, a reducing environment and slow groundwater recharge through the wet season has likely facilitated the removal of nitrate from the upper soil profile of the farm field (e.g. Wigington et al. 2003). Second, the study wetlands only have moderate artificial surface drainage. If the Finley Farm wetland had artificial subsurface drainage or more intensive surface drainage, we would expect increased downstream export of both water (McCarthy et al. 1991) and agricultural chemicals (Havens et al. 2001). Finally, we suspect that the overall flow path length, slope, and presence of wetland vegetation along ditches between both study sites and Muddy

Creek facilitates the removal of non-conservative chemicals like nitrate-nitrogen in surface runoff before reaching the creek.

Conclusions

Our results provide initial insight into how surface runoff generated from native prairie and farmed mineral flat wetland sites originates from different water storage pools within each wetland depending on time of year, recent precipitation, and antecedent moisture conditions in the field. Our results also illustrate how nitrate-nitrogen and other nutrient concentrations in downstream runoff relate to the timing and source of surface runoff within each wetland. Overall, our findings suggest that, while surface water pathways dominate runoff from wetland flats under farm land use, larger wetland flat fields have a high potential to absorb, store, and process nutrients and agrochemicals from on-site and nearby off-site chemical inputs. Mineral flats that maintain wetland hydrology in spite of farm use represent a unique balance between agricultural production and preservation of some of the water storage and delay, and water quality-related ecosystem services once provided at a much larger scale in the Willamette Valley lowlands. Land managers in the southeastern United States have recognized the need for some of the ecosystem services provided by mineral flat wetlands in agricultural landscapes, and have begun to adopt strategies such as drainage water management, where subsurface and surface drainage outlets are temporarily closed to allow soil saturation or shallow inundation during the wet season in the most heavily drained fields (De Steven and Lowrance 2011). Our results provide useful information for land use planning decisions and the assessment of the local and catchment-scale ecological services provided by seasonally wet, former wetland flats in agricultural areas, many of which are undocumented on national and local wetland inventories.

Use of stable isotopes and water chemistry analysis were critical to distinguishing land use effects, as well as identifying dominant storage pools and transport pathways for water and nutrients in a relatively flat, valley bottom agricultural setting. In order to fully characterize catchment-scale runoff generation processes in mineral flat-dominated agricultural catchments, additional storm event-based sampling of stable isotopes, runoff volume, and water chemistry would be important for confirming our findings on other farmed wetlands with hydric soils, including those with subsurface drainage systems. Further research is also warranted on the effectiveness of low-gradient, vegetated ditches transferring runoff from mineral flats to other wetlands and streams with respect to nutrient-removal. A better understanding of the cumulative water quality and quantity impacts of mineral flat wetlands, along with seasonal, artificial extensions of the natural drainage network in agricultural landscapes (Wigington et al. 2005), is important for evaluating how anthropogenically isolated wetlands are linked to downstream navigable waterways and their tributaries at a larger catchment scale.

Acknowledgements

We thank Laura and Jim Marshall, Alicia Omlid, Leah Wilson, Terrence Morris, and Diana Sharps for assistance with the installation and monitoring of water table wells and piezometers. We are grateful for feedback from Jeff McDonnell, Scott Bridgham, Jack Istok, and Kathy Motter related to the study design, particularly with respect to stable isotope hydrology and sampling, wetland biogeochemistry, groundwater hydraulics and monitoring techniques, and water quality sample collection and analysis techniques, respectively. We thank Renee Brooks from the US Environmental Protection Agency for sharing precipitation isotope data. Thanks also to Jock Beall, Jarod Jebousek, and Molly Monroe from the US Fish and Wildlife Service, and Tom Danelski for help with field logistics, site background information, permitting, and access to wetland study sites. This material is based upon work supported under a National Science Foundation Graduate Research Fellowship. Any opinions, findings, conclusions or recommendations expressed in this publication are those of the authors and do not necessarily reflect the views of the National Science Foundation.

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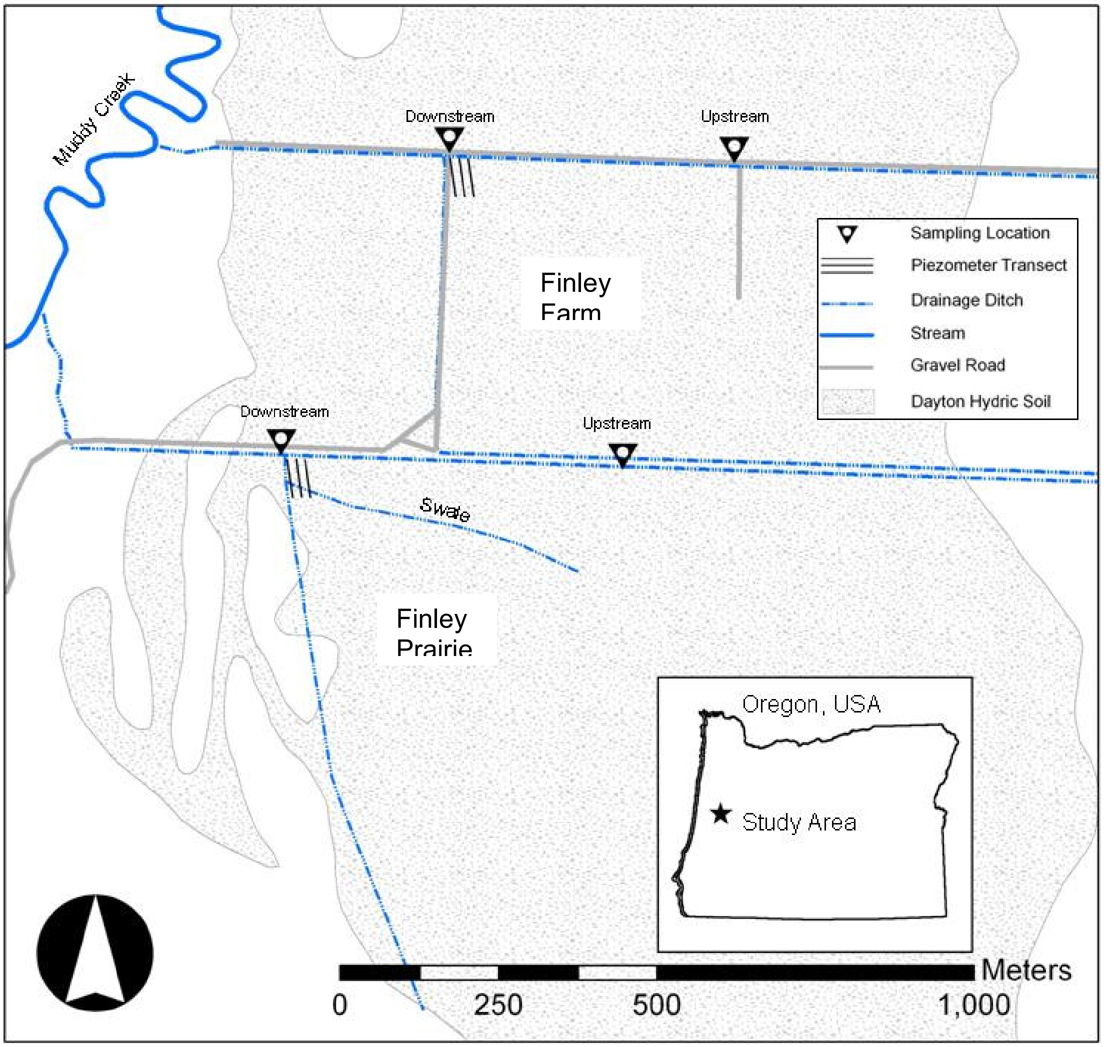

Figure 1. Study Area, showing the extent of the Dayton wetland soil unit and the artificial drainage network. Ditches drain to the northwest and Muddy Creek flows north.

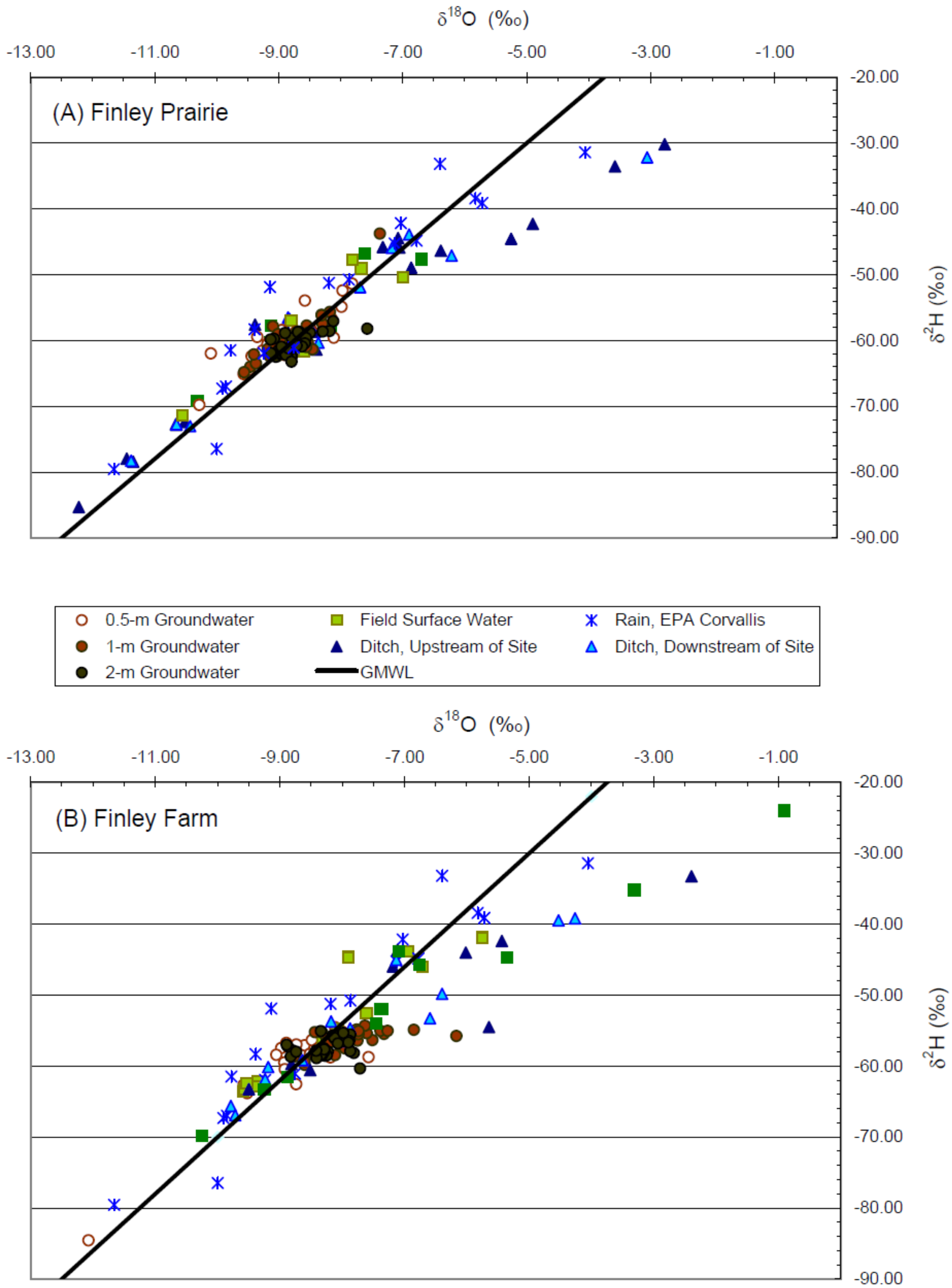

Figure 2. Finley Prairie (A) and Finley Farm (B) site isotopic comparisons. The equation for the Global Meteoric Water Line (GMWL) is: $\delta^2H = 10 + 8*\delta^{18}O$.

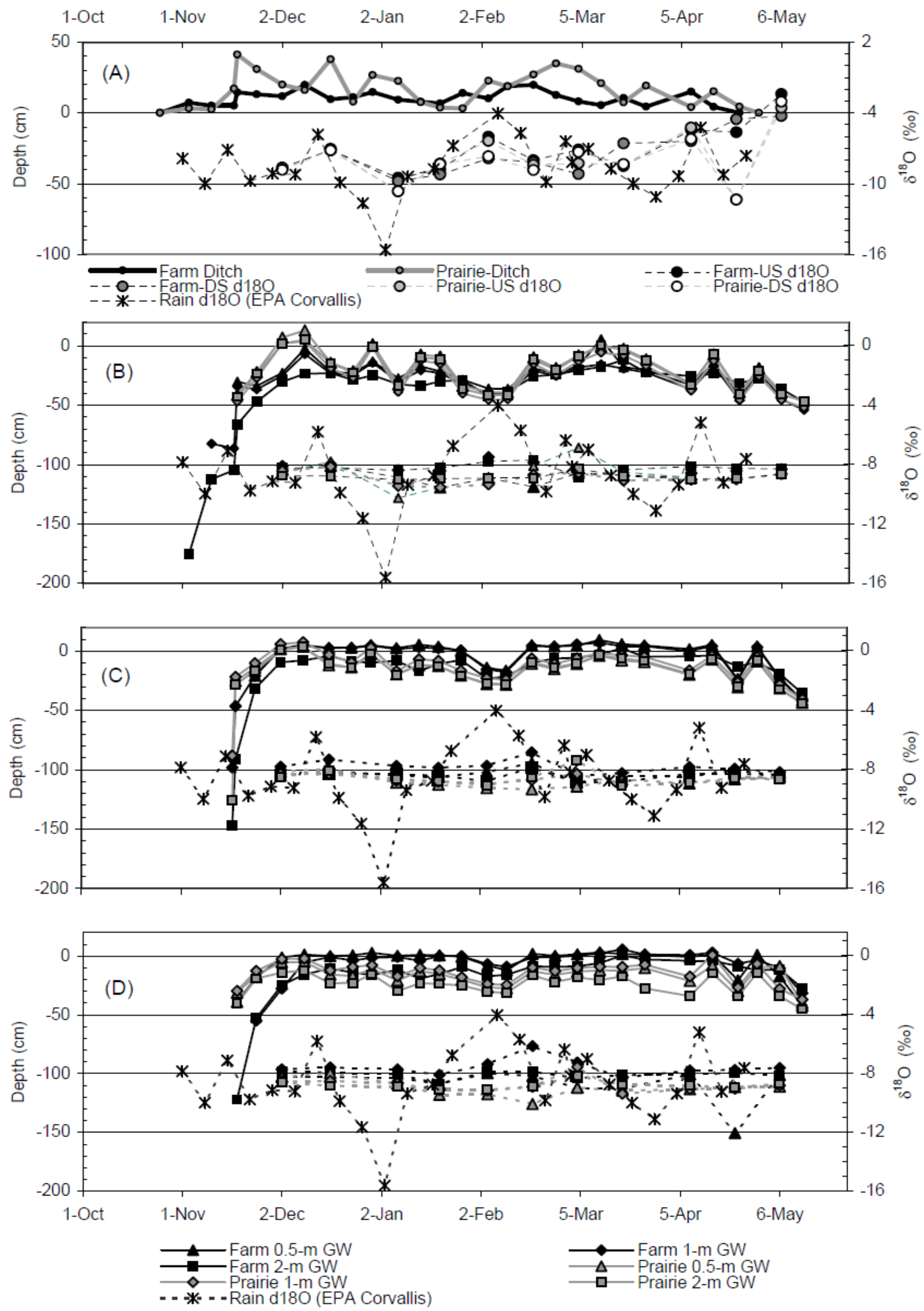

Figure 3. Water levels with respect to ground surface and isotopic composition of water in ditches and 0.5-m, 1-m, and 2-m groundwater. (A) shows data from ditches, and (B), (C), and (D) represent sampling points 5, 35, and 65 m away from the ditch, respectively. Isotope values are connected with dotted lines.

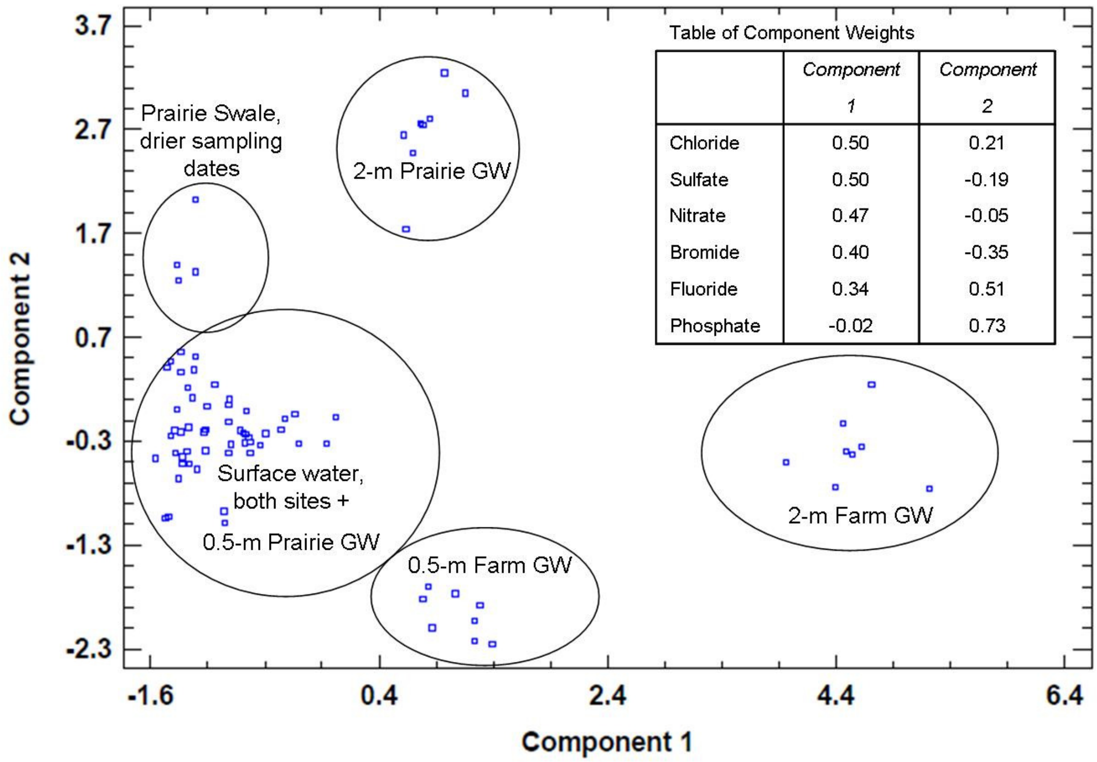

Figure 4. Principal Components Analysis for all water quality data. Inset table shows component weights for Component 1 and Component 2. GW = groundwater samples.

Table 1. IsoSource mass balance model output for feasible water source contributions to ditch water downstream of Finley Prairie and Finley Farm wetland sites.

	Upstream Ditch Water		Field Surface Water 1		Field Surface Water 2		0.5-m Groundwater		1-m Groundwater		2-m Groundwater	
Date/Site	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD
12/2/2010												
Prairie	0.000	*	0.000	*	**1.000**	*	0.000	*	0.000	*	0.000	*
Farm	-	-	-	-	-	-	-	-	-	-	-	-
12/17/2010												
Prairie	**0.962**	0.004	0.018	0.014	0.013	0.014	0.007	0.009	0.000	0.000	0.000	0
Farm	0.169	0.124	**0.767**	0.103	n.a.	n.a.	0.019	0.018	0.026	0.021	0.019	0.018
1/7/2011												
Prairie	**0.987**	0.010	0.002	0.004	0.010	0.013	0.002	0.004	0.000	0.000	0.000	0
Farm	0.205	0.152	**0.392**	0.020	0.323	0.145	0.042	0.037	0.020	0.019	0.019	0.018
1/20/2011												
Prairie	**0.934**	0.017	0.021	0.021	0.024	0.023	0.005	0.007	0.005	0.007	0.012	0.013
Farm	0.087	0.071	0.260	0.073	0.041	0.035	**0.499**	0.080	0.058	0.049	0.055	0.046
2/4/2011												
Prairie	**0.550**	*	0.440	*	n.a.	n.a.	0.010	*	0.000	*	0.000	*
Farm	0.249	0.028	0.124	0.094	n.a.	n.a.	**0.500**	0.036	0.066	0.051	0.062	0.049
2/18/2011												
Prairie	-	-	-	-	-	-	-	-	-	-	-	-
Farm	0.010	0.014	**0.794**	0.009	0.184	0.015	0.012	0.013	0.000	0.000	0.000	0
3/4/2011												
Prairie	0.011	0.012	**0.446**	0.220	0.232	0.096	0.255	0.172	0.044	0.036	0.013	0.014
Farm	0.010	0.012	**0.942**	0.015	0.007	0.009	0.011	0.013	0.015	0.016	0.015	0.016
3/18/2011												
Prairie	0.111	0.097	0.023	0.023	0.013	0.015	**0.794**	0.118	0.055	0.029	0.004	0.006
Farm	0.009	0.011	**0.950**	0.010	0.006	0.008	0.010	0.012	0.013	0.014	0.013	0.015
4/8/2011												
Prairie	0.409	0.030	**0.490**	0.056	0.067	0.056	0.011	0.012	0.012	0.012	0.012	0.013
Farm	0.008	0.010	**0.646**	0.014	0.003	0.005	0.092	0.077	0.235	0.082	0.016	0.017
4/23/2011												
Prairie	**0.990**	0.000	n.a.	n.a.	n.a.	n.a.	0.003	0.006	0.003	0.006	0.003	0.006
Farm	0.000	0.000	**0.780**	0.000	n.a.	n.a.	0.200	0.000	0.015	0.007	0.005	0.007
5/6/2011												
Prairie	**0.930**	0.000	n.a.	n.a.	n.a.	n.a.	0.070	0.015	0.050	0.015	0.040	0.014
Farm	0.037	0.030	**0.525**	0.023	n.a.	n.a.	0.299	0.072	0.068	0.059	0.070	0.062

SD = standard deviation, * indicates standard deviation could not be computed in IsoSource because there was only one solution generated

Bold values = sources with highest likely contribution to downstream ditch water

Field surface water 1 = pond water for Finley Farm, swale water for Finley Prairie; Field surface water 2 = ponded water at the Finley Farm, water in the west ditch at North Prairie

Table 2. 2010-2011 water chemistry data for field groundwater (GW), field surface water (prairie swale/ farm pond), and ditch water upstream (US) and downstream (DS) of Finley Prairie and Finley Farm. Hyphen indicates sample was below ion chromatograph detection limit; n.a. indicates sampling point was dry at time of sampling.

	Finley Prairie					**Finley Farm**				
	0.5-m GW	2-m GW	Swale	US Ditch	DS Ditch	0.5-m GW	2-m GW	Pond	US Ditch	DS Ditch
Date					**$N\text{-}NO_3$ (mg/L)**					
1/7/2011	0.00	0.03	0.00	0.00	0.00	0.04	3.97	-	0.00	0.02
1/20/2011	0.01	0.01	0.00	-	0.00	0.02	3.14	0.00	0.04	0.03
2/4/2011	0.01	0.01	0.02	0.00	0.02	0.01	3.48	-	-	0.02
2/18/2011	0.00	0.01	0.00	0.00	0.00	0.01	3.79	-	0.19	0.05
3/4/2011	-	0.01	0.00	0.00	0.00	0.00	3.56	0.00	0.02	0.00
3/18/2011	0.00	0.01	0.00	0.01	0.00	0.00	3.86	0.00	0.00	0.01
4/8/2011	0.00	0.02	0.00	0.01	0.00	0.01	3.92	0.00	0.04	0.00
4/23/2011	0.00	0.00	n.a.	0.00	0.00	0.01	4.27	0.00	n.a.	0.00
					Cl^- (mg/L)					
1/7/2011	3.82	14.55	3.76	3.54	3.60	8.53	19.12	5.48	4.42	6.18
1/20/2011	3.36	14.26	2.83	2.25	2.50	9.28	16.97	5.93	2.65	6.68
2/4/2011	3.75	13.88	3.99	3.53	3.74	8.13	18.09	9.55	4.07	9.03
2/18/2011	3.09	13.14	2.72	2.39	2.56	6.49	19.81	8.02	2.83	7.13
3/4/2011	2.67	13.44	2.23	1.79	2.02	6.92	17.73	4.87	2.38	4.79
3/18/2011	2.62	11.98	2.71	2.53	2.52	7.27	18.87	4.18	2.29	3.94
4/8/2011	2.86	11.53	2.22	1.59	1.94	7.71	18.93	4.12	2.90	2.59
4/23/2011	3.48	12.71	n.a.	0.62	0.89	5.57	19.98	4.42	n.a.	2.05
					SO_4^{2-} (mg/L)					
1/7/2011	0.35	14.95	0.10	0.50	0.57	71.91	136.64	0.46	10.49	3.36
1/20/2011	0.35	15.33	0.07	0.12	0.14	70.18	128.16	0.07	6.61	1.45
2/4/2011	0.32	15.26	0.12	0.57	0.36	65.80	129.97	0.04	3.50	3.20
2/18/2011	0.32	13.64	0.24	0.15	0.17	57.23	134.57	0.16	7.96	2.73
3/4/2011	0.27	13.12	0.05	0.14	0.12	57.78	120.36	0.19	4.19	1.06
3/18/2011	0.17	10.36	0.06	0.14	0.13	59.16	124.09	0.11	3.02	2.79
4/8/2011	0.18	11.89	0.02	0.29	0.07	64.02	121.75	0.04	7.02	10.13
4/23/2011	0.15	11.10	n.a.	0.12	0.14	47.95	124.48	0.07	n.a.	9.65
					$P\text{-}PO_4^{3-}$ (mg/L)					
1/7/2011	0.03	0.06	0.05	0.01	0.02	-	0.01	0.01	0.01	0.01
1/20/2011	0.04	0.05	0.00	0.03	0.00	-	0.01	0.00	0.01	0.00
2/4/2011	0.03	0.06	0.07	0.01	0.01	-	0.02	-	0.01	0.01
2/18/2011	0.03	0.05	0.06	0.02	0.02	-	0.01	-	0.02	0.01
3/4/2011	0.03	0.06	0.04	0.02	0.01	-	0.02	0.00	0.01	0.01
3/18/2011	0.06	0.05	0.04	0.01	0.01	-	0.01	-	0.01	0.01
4/8/2011	0.03	0.04	0.03	0.01	0.01	-	0.01	0.00	0.01	0.01
4/23/2011	0.03	0.03	n.a.	0.01	0.02	0.00	0.02	-	n.a.	0.01

CHAPTER 5. CONCLUSIONS

We compared infiltration, wetland hydroperiod, groundwater recharge dynamics, surface runoff generation, and water quality in mineral flat wetlands using a combination of soil and hydrometric measurements, stable isotope tracers, and water chemistry analysis. Our overarching objectives were to examine, for mineral flat wetlands under native prairie, farmed grassland, and restored prairie land cover: 1) how different land management influences infiltration and wetland hydroperiod at the plot scale, 2) the effects of land use on seasonal groundwater-surface water dynamics at the field scale, and 3) seasonal variation in runoff sources and nutrient transport from native prairie and farmed wetlands at the small catchment scale. Key findings of this research are:

1) Land use does not necessarily dictate the overall hydrologic behavior of mineral flat wetlands, particularly if land management practices minimize mechanical soil disturbance and compaction. If we are to manage for wetlands with high infiltration rates, and thus a greater potential to store and delay storm runoff, land management practices that minimize soil compaction, increase soil organic matter content, and promote diverse plant cover are most suitable for mineral flat wetlands.
2) Mineral flat wetlands under a variety of grassland land uses, ranging from least-altered prairie to farmed grass fields and restored prairie, can have a similar hydroperiod and degree of shallow groundwater-surface water interaction. Across all studied land uses, flats had a minimal amount of seasonal variation in the isotopic composition of groundwater below the perching layer, suggesting that the deeper part of the soil profile absorbs and retains water from the early part of the wet season. $\delta^{18}O$ values in shallow groundwater above the perching layer reflected meteoric inputs during the winter and evaporation in the spring.
3) Surface runoff generated from native prairie and farmed mineral flat wetland sites originates from different water storage pools within each

wetland depending on time of year, recent precipitation, and antecedent moisture conditions in the field. While surface water pathways dominate runoff from wetland flats under farm land use, larger wetland flat fields have a high potential to absorb, store, and process nutrients and agrochemicals from on-site and nearby off-site chemical inputs.

Further examination of soil water isotope values, throughout the year, would help shed light on unsaturated processes within mineral flat wetlands, and the degree to which mobile groundwater mixes with water already stored in the soil profile during successive parts of the wet season. To fully characterize catchment-scale runoff generation processes in mineral flat-dominated agricultural catchments, additional storm event-based sampling of stable isotopes, runoff volume, and water chemistry would also be important for confirming our findings on other farmed wetlands with hydric soils, including those with subsurface drainage systems. Finally, research is warranted on the effectiveness of low-gradient, vegetated ditches transferring runoff from mineral flats to other wetlands and streams with respect to nutrient-removal. A better understanding of the cumulative water quality and quantity impacts of mineral flat wetlands, along with seasonal, artificial extensions of the natural drainage network in agricultural landscapes is critical in evaluating how anthropogenically isolated wetlands are linked to downstream navigable waterways and their tributaries at a larger catchment scale.

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