DISSERTATION

ECOLOGICAL CHARACTERISTICS, ENVIRONMENTAL SERVICES AND THE POTENTIAL FOR CHANGE IN A SEMI-ARID AGRICULTURAL LANDSCAPE IN COLORADO, U.S.A.

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ABSTRACT

ECOLOGICAL CHARACTERISTICS, ENVIRONMENTAL SERVICES AND THE POTENTIAL FOR CHANGE IN A SEMI-ARID AGRICULTURAL LANDSCAPE IN COLORADO, U.S.A.

The conversion of native vegetation to cities and agriculture has caused a loss in habitat diversity with subsequent effects on plant and animal populations. Hydrologic modifications have increased land available for buildings and crops through drainage and irrigation, with largely negative effects on aquatic and wetland ecosystems. This work contributes to the growing literature of ecological benefits of anthropogenic ecosystems with two examples from the infrastructure and activities associated with irrigated agriculture, canal riparian habitat and nutrient mitigation processes of tailwater wetlands.

The prevalence of canals in agricultural areas and the immense amount of water used for agriculture have created a new stream system in parts of the western U.S. In my study area in semi-arid, northcentral Colorado 1,906 km of canals supply water to 67,606 ha of irrigated agriculture and several cities and towns. Riparian vegetation bordering the canals was statistically similar for canals and streams in agricultural areas for composition of functional plant groups, yet dissimilar for species composition. In residential areas species composition was statistically different, though the p-value was borderline ($p=0.05$) and the functional groups were more strongly separated ($p=0.013$). Temporary aquatic habitat also provides suitable conditions for macroinvertebrate communities to colonize with statistically similarity between canals and streams in both land uses. In addition to the similarity to natural ecosystems, the length of canals

can exceed that of natural streams in some regions, potentially creating more riparian habitat than would have naturally occurred and in new landscape positions. During high agricultural and municipal water use, more water could be flowing through artificial channels than natural streams, creating a paradigm shift and posing the concept that canals are the new rivers of the West.

Wetlands can trap and process agricultural water pollutants, but landscape position is crucial to accumulate polluted runoff or groundwater and maintain hydrologic conditions favorable to biogeochemical processes for pollutant removal. Tailwater wetlands receiving excess irrigation water and surface runoff intercepted nutrient rich waters with nitrate (NO_3-N) concentrations up to 54 mg/L, over five times the U.S. EPA and ten times the drinking water standard in Europe. Biotic, hydrologic, biochemical characteristics of tailwater wetlands were favorable for N transformation and uptake processes with shallow water tables creating anoxic conditions and sufficient organic matter and microbial communities for denitrification processes. Plant uptake was of greater importance at all wetland sites, especially those with *Typha latifolia*. The duration of saturated conditions supported wetland plants with high annual biomass for uptake of excess nutrients.

A decline in irrigated agriculture could change the type, extent and quality of ecosystems services associated with canal infrastructure and tailwater wetlands. A one-third decline in irrigated agriculture in the study area was modeled through drying up irrigated land using several spatial prioritizations. These resulted in different location and intensity of effects on canal riparian and wetland ecosystem services. Results from these three studies identify two ecosystem services of the infrastructure and activities associated with irrigated agriculture and the potential unintended consequences of changes to the timing and amount of water distributed across the landscape.

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DEDICATION

To the four generations of family that supported and believed in me.

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1. Introduction

Human activities can reduce ecosystem services at landscape and local scales. The conversion of native vegetation to crops has caused a loss in habitat diversity with subsequent effects on plant and animal populations. Human civilization has historically developed near reliable sources of freshwater leading to degradation of aquatic and riparian ecosystems. These ecosystems have positive values including wildlife habitat and corridors, biogeochemical cycling and landscape stability, with effects extending beyond their spatial footprint (Naiman et al. 1993; Patten 1998; Muehlbauer et al. 2014). Agriculture and urban land uses negatively affect natural habitats, nutrient cycles, and watershed scale hydrodynamics (Carpenter et al. 1998; Scanlon et al. 2007). The addition of key elements for growth (N, P, and C) has resulted in non-point source pollution of surface and groundwater resources across the world. Hydrologic modifications have increased arable land through drainage and irrigation, with negative effects on aquatic and wetland ecosystems (Bunn and Arthington 2002; Malmqvist 2002). In recent decades, there has been increased interest in the ecological elements and services of human modified landscapes (Moonen and Barberi 2008).

I present two examples of ecosystem services created by irrigated agriculture. First, riparian and aquatic ecosystems have formed along canals that support vegetation (Lopez-Pomares et al. 2015; Aspe and Jacque 2015), aquatic macroinvertebrate and fish communities similar to those of natural streams (Habit et al. 1998, 2005; Koetsier and McCauley 2015). In some agricultural regions, the length of irrigation canals can exceed that of natural streams, potentially creating more habitat than previously existed and in new landscape positions as water is diverted from rivers and delivered to and through upland areas. At some times of the year, more water is flowing in canals than natural rivers, and canals have become the new rivers of western U.S. The ecological characteristics of drainage ditches has been well studied (Herzon and Helenius 2008; Verdonschot et al. 2011; Shaw et al. 2015; Whatley et al. 2015; Leslie and Lamp 2017), yet other constructed channels, including irrigation canals have not received as much attention. In regions with altered landscapes, canal riparian zones could be providing similar quality habitat as degraded natural streams, in effect mitigating for the loss associated with diverting streamflow (Lopez-Pomares et al. 2015). Physical and vegetation management of irrigation canals can have strong effects on the composition and structure of biotic communities. The intensity and frequency of maintenance activities including burnings and spraying could be modified to improve ecological qualities including patch connectivity and habitat refugia.

Mitigation of excess agricultural N runoff by tailwater wetlands is the second ecosystem service studied herein. Excess N has led to degraded water quality in the Gulf of Mexico (Rabalais et al. 2002), the Chesapeake Bay (Boesch et al. 2001), and many other estuary and freshwater ecosystems (Smith 2003; Howarth and Marino 2006). Nitrates are highly soluble component of many fertilizers and easily leached from livestock waste (Gbolo and Gerla 2015). Nitrates move easily in surface and groundwater, leading to rapid contamination and downstream effects. Nutrient best management practices inform application rates, timing and methods to limit the amount NO_3-N runoff and leaching, however, NO_3-N concentrations in surface and groundwater in many agricultural regions are still above acceptable levels (Bauder et al. 2006).

Wetlands have formed in agricultural areas where irrigation runoff is concentrated and local groundwater rises due to irrigation canals and irrigation of crops. Wetlands can trap and process agricultural pollutants in water including N, P and sediment (Liehr and Kruzic 2007; Knox et al. 2008). Wetlands have been constructed to collect contaminated runoff and support ecological processes associated with nutrient pollution sequestration and transformation. Wetlands have been used with local success as components of wastewater treatment processes (Kivaisi 2001; Stottmeister et al. 2003; Vymazal 2014), passive filtration for storm water (Kao et al. 2001), and treating agricultural runoff (Kovacic et al. 2000; Tanner et al. 2005; Beutel et al. 2009; Budd et al. 2009; O'Geen et al. 2010; Diaz et al. 2012). Agricultural wetlands adjacent to and downslope of agricultural fields have the potential to intercept runoff containing high NO_3-N concentrations. Wetlands hydrodynamics can be linked to local and regional groundwater movement and canal seepage (Sueltenfuss et al. 2013; Denver et al. 2014).

Intensively managed landscapes can support ecosystem services such as habitat, food resources, and nutrient cycling. Some ecosystem services are directly linked to human activities, for example, riparian vegetation along canals are dependent on the canal being used to convey water. The reallocation of resources such as water and soil can cause negative effects such as alter streamflow from water extraction. Negative effects on natural streamflow including reduced peak flows and extreme low flows on natural aquatic and riparian ecosystems is well-studied (Merritt and Cooper 2000; Nilsson and Svedmark 2002; Poff and Zimmerman 2010; Pyne and Poff 2017). The conclusions of these studies could be used to understand potential impacts to canals if flow patterns change. Similarly, wetlands dependent on irrigation runoff or canal seepage could shrink, become drier, and support less diverse vegetation with changes to water delivery across a region (Peck and Lovvorn 2001). The direct and indirect consequences on natural and incidental ecological resources and should be considered in future water allocation and development plans. In the following chapters I investigate how infrastructure and activities that deliver water for municipal and agricultural use contribute to regional ecosystem services by creating riparian and aquatic habitat and wetlands that process excess agricultural N. The potential for riparian ecosystems bordering canals and temporary aquatic habitats of the channels to replace degraded natural stream ecosystems is explored in Chapter 2. The efficacy of tailwater wetlands to intercept, transform and trap excess agricultural N is tested in Chapter 3. Chapter 4 incorporates the findings from Chapters 2 and 3 to develop a spatially explicit model of current and future ecosystems services using predicted regional water distribution.

2. The New Rivers of the Western U.S.: Riparian and Aquatic Ecosystems of Agricultural Irrigation Canals

2.1 Introduction

Agriculture in arid and semi-arid regions requires irrigation water diverted from streams and transported in constructed canals. Streamflow reductions for hydropower generation, municipal and agricultural use affect rivers of various sizes across the globe (Graf 1999; Kingsford 2000; Poff et al. 2007; Stefanidis et al. 2016). The effects of dewatering streams on aquatic and riparian biota and ecosystems have been well studied in many regions of the world (Bunn and Arthington 2002; Malmqvist 2002). In many regions, the canals created to transport water to agricultural land support aquatic and riparian ecosystems, but relatively little is known about their biodiversity (Patten 1998), or similarity to communities found along natural streams and their riparian areas (Chester and Robson 2013).

Riparian ecosystems are highly productive and disproportionately diverse relative to the surrounding upland ecosystems in most regions (Naiman et al. 1993) and function to store, transform and cycle nutrients (Jacobs et al. 2007), organic matter (Tank et al. 2010) and sediment (Steiger et al. 2003) and water. They also link terrestrial and aquatic ecosystems at local and landscape scales (Fisher et al. 1998, Harvey and Gooseff 2015). In, and along, rivers hydrologic and geomorphic processes shape the physical landscape, and the disturbance regime controls the potential colonization and persistence of organisms (Shafroth et al. 2002; Katz et al. 2009). Canals are subject to many of the same hydrologic and geomorphic processes and human activities that shape riparian and aquatic ecosystems, yet differ in physical structure and flow characteristics.

Water diversions from streams for municipal, industrial and agricultural uses, reduce total and peak stream flows, and intensify low flow stressors on aquatic ecosystems such as increased water temperature and low dissolved oxygen (Bunn and Arthington 2002; Poff and Zimmerman 2010). At the watershed-scale, interactions with extraction and irrigation alter water balances (Hatfield 2015), have a cumulative effect on instream flows, and may degrade surface and ground-water by non-point source pollutants in agricultural return flows (Sprauge 2005).

Streams and riparian areas throughout the world have been directly modified by floodplain development and indirectly though watershed modification, stream flow alteration, and land use changes (Patten 1998) (Nilson and Svedmark 2002; Bunn and Arthington 2002). Human activities have altered the physical structure of rivers and their floodplains through dam and levee construction, channel straightening and bank stabilization. Impoundments have changed the historic sediment regime and reduced flood intensity, frequency and duration which have altered geomorphic processes and riparian plant communities (Merritt and Cooper 2000). For instance, *Populus* spp. dominated riparian forests rely on natural flood disturbance for health and regeneration with reduced growth, dieback and mortality under altered flow regimes or water table depths (Williams and Cooper 2005; Northcott et al. 2007, Schook et al. 2016).

Geomorphic metrics used to characterize streams, including overbank flooding, sinuosity, and channel migration, are not relevant for water conveyance canals. Canals are designed and built to minimize turbulent flow and maximize conveyance. They deliberately lack the hydrologic variability and spatial heterogeneity of landforms created by the fluvial processes of streams (Swamee 1995). Canal management including sediment and woody debris removal maintains the homogenization of canal riparian and aquatic ecosystems. In addition, canals are largely decoupled by berms from the surrounding landscape that limits sediment and organic matter inputs. However, colonization of riparian plants and aquatic macroinvertebrates through high flow dislodgement and drift is likely high through connectivity of surface waters between canals and streams (Ernegger et al. 1998; Koetsier et al. 2005). Indeed, canals have long been recognized as vectors for the introduction of non-native species through hydrochory (Egginton and Robbins 1920).

Aquatic macroinvertebrates are a key component of aquatic, riparian and some upland food webs (Polis et al. 1997; Nakano et al. 1999; Leigh et al. 2013), and the effects of flow variation can cascade through the food chain to adjacent uplands (Nakano and Murikami 2001; Muehlbauer et al. 2014). Aquatic macroinvertebrate populations are influenced by channel physical characteristics, food type and availability, and hydrologic patterns at relatively small scales (Resh et al. 1994; Williams and Feltmate 1992). Aquatic macroinvertebrates often use seasonal and hydrologic cues for their life history development (Bulter 1984). Flow variability in canals can be high, with rapid drawdowns and multiple high and low flow events per year, with unnatural timing that can affect aquatic macroinvertebrate growth and reproduction. Studies investigating agricultural drainage ditches illustrated selection of biota with life history adaptations to varying flow regimes (Whatley et al. 2015). Seasonal and random disturbance events are layered on top of the physical setting and influence the relative importance of colonization and competition in community composition.

Canals add channel length to existing stream networks and may increase total riparian habitat in a watershed above pre-settlement amounts. Canal riparian habitats potentially exist wherever humans have diverted water from streams, yet have been studied in few locations in Chile, Spain, France and the United States (see Habit et al. 1998; Lopez-Pomares et al. 2015, Aspe and Jacque 2015; Fernald et al. 2007). The prevalence of canals can be easily overlooked as their riparian vegetation may resemble that along natural streams (see Figure 2.1), even though they may lack some ecosystem functions such as nesting habitat for birds or food for wildlife (Cox and Franklin 1989; Chester and Robson 2013). However, studies on fish in irrigation canals showed increase size, more and diverse prey items while maintaining synchronous reproduction similar to river populations (Habit et al. 2005).

Most large rivers and streams in the western U.S. are affected by impoundments, reservoirs, diversions, and hydro-electric structures (Graf 1999; Barnett et al. 2008). Historic flow regimes of the larger regional rivers were dominated by early summer melt water from mountain snowpack. Streamflow on most rivers is now altered by engineered structures to provide water to the region's population and agriculture (Milliken 1988; Strange et al. 1999). To meet water demands trans-basin water diversions further alters streamflow patterns and sediment dynamics (Dennehy et al. 1993.

The contribution of canal riparian vegetation and aquatic invertebrates to the quality and resilience of local and regional ecosystems continues to be overlooked, although some agricultural drainage ditches and heavily impacted urban streams have received increased attention (Paul and Meyer 2001; Herzon and Helenius 2008; Vermonden et al. 2009; Verdonschot et al. 2011). Canals could support riparian and aquatic ecosystems comparable to natural streams. Irrigated agriculture has a 150-year history in the western U.S. (Eschner et al. 1983, Evans and Evans 1991) and plants and animals have colonized this new water distribution system, forming communities and ecosystems that influence the larger agro-ecosystem. In this section, I address the following questions; (1) how abundant are canal riparian ecosystems comparted to natural streams? (2) do natural streams and canals support similar riparian vegetation and aquatic macroinvertebrate communities? (3) how do functional groups of plants and aquatic macroinvertebrates vary between stream and canal sites?

2.2 Methods

2.2.1 Study Area and Data Collection

The study was conducted in northcentral Colorado (Figure 2.2) a semi-arid region with extensive irrigated agriculture. Precipitation averages 250 millimeters during the summer and 135 mm during winter (www.usclimatedata.com). Total annual precipitation is insufficient to support desirable crops such as corn, beans and vegetables (Schneekloth and Andales 2009). Water for irrigation is diverted from rivers fed by melting snow in the Rocky Mountains and applied to crops from late April through September.

South Platte River Basin Water District 3 diverts water from the Cache la Poudre River, which then flows primarily through earthen canals with riparian vegetation along canal margins. This District's network of canals has 1,063 km of channel length according to Colorado Decision Support System (CDSS) 2010 data on irrigation structure and land use (http://cdss.state.co.us), providing water to 67,606 hectares of irrigated crops. However, I mapped the network canals using aerial imagery (details section 2.2.2.1) after noticing discrepancies with current land use and missing segments. The updated network contained 1,968 km of canals which included all sizes of canals from 18 m wide to <1m laterals providing water to individual fields. Vegetation management along canals varies with adjacent land use. More frequent and intense management was observed in agricultural compared to urban and residential. Therefore, I separated sites as agricultural (Ag.) or residential (Res.) *a posterior* using the dominant on land use within 100 m of each site. I do not suggest that management activities are controlled within or between land uses, but field observations warranted distinction.

I included small peripheral canals not mapped by CDSS or U.S. Geological Survey (USGS) because significant riparian and temporary aquatic habitat occurs on some of these channels. These were generally not present in residential areas, as peripheral canals are designed to provide water to individual fields, and local agricultural irrigation is limited in residential areas (Fig. 2.3A).

Streams in the region have been altered from their pre-settlement condition, as humans have developed floodplains, straightened channels, diverted water for irrigation, managed flows for water storage, release treated wastewater into channels, and invasive nonnative plants have become abundant (Strange et al. 1999; Shieh et al. 1999, 2001). For instance, the flow- and sediment regime of the Cache la Poudre River is altered by low-head dams used to divert water into irrigation canals and by the addition of water through trans-basin flow augmentation (Bartholow 1991; Evans and Evans 1991). The floodplain has been constrained by gravel mining, roads and agriculture as many other rivers in the region (Strange 1999; Wohl 2001).

2.2.1.1 Site Selection and Layout

Vegetation along canals was stratified into five cover types: heavy tree canopy, light tree canopy, shrub, herbaceous, and concrete, using sub-meter resolution aerial and satellite imagery in ArcGIS v10 (ESRI 2010). Heavy tree canopy was defined as continuous woody canopy >50% cover over channel banks. Light tree canopy had <50% woody canopy cover. Shrub sites had woody vegetation less than 3 meters in height covering >50% of the bank vegetation. Each site included three transects oriented perpendicular to the channel at five bank full widths up and downstream from the central point. Each canal site was selected randomly with the allocation of points for each cover type determined by the proportion of each cover type in the canal network. Stream sites were selected to minimize hydrologic modifications within the reach and maximize undeveloped floodplain width to represent the least impacted natural stream reaches and are considered reference sites. Vegetation was sampled at 47 canals and 7 streams during summer of 2013. Aquatic macroinvertebrates were sampled (every three weeks, May-Aug) in 2015 for a maximum of 6 samples per site for 20 sites representing all substrate types, channel sizes, and flow regimes.

2.2.1.2 Environmental and Hydrologic Variables

Channel and floodplain physical characteristics, land use, network, and hydrologic variables were selected and modified from riparian and stream assessment methods (e.g. Innis et al. 2000; Munne et al. 2003; del Tanago and de Jalon 2011) to fit the range of channel forms expected (details in section 8.3). Streamflow data were from two USGS gages on the Cache la Poudre River for the period 1999-2015 (Fort Collins #06752260, and Greeley #06752500). Streamflow records for Spring Creek, Box Elder Creek and Dry Creek were only available for 2013-2015 and were obtained from the City of Fort Collins flood warning system. Owl Creek and Willow Creek are not gaged and Lone Tree Creek streamflow data were not available for the study period. Flow in irrigation canals is recorded at the point of diversion from the Cache la Poudre River and available from the CDSS (http://cdss.state.co.us/ONLINETOOLS/Pages/StructuresDiversions.aspx). Channels were separated by bank full width into large (>4 m) and small (<4 m) for some hydrologic comparisons.

2.2.1.3 Riparian Vegetation

Canopy cover by species was visually estimated using a modified Braun-Blanquet cover class scale (trace, <1%, 1-2%, 2-5%, 5-10%, 10-25%, 25-50%, 50-75%, 75-95%, >95%) (Bonham 1989) for five 1 m^2 plots on each bank and top/floodplain surfaces on each side of the channel along each transect for a total of 60 plots per site. The taxonomy used for plant species identifications follows Weber and Wittman (2012). Vertical structure was categorized for each species by estimating average crown height using classes (<1 m, 1-2 m, 2-5 m, 5-10 m, >10 m) (Merritt and Bateman 2012). Bare sediment was analyzed using the same categories as plant species, and it accounted for significant cover at many sites.

The mid-point of field estimated cover classes was used to calculate the average cover for each species. Plot level cover values were averaged for each site to scale the data to channel reach scale. Diversity metrics including total richness, native richness, Shannon-Wiener (log e) (Shannon and Weaver 1949) and Simpson's index (1-λ) (Simpson 1949) were calculated using all species. Sixty number of plots were sampled for each site, making direct comparisons of richness and diversity appropriate. For community analysis, species present in fewer than 5% of sites were removed (McCune and Grace 2002) reducing the number of plant species used in the analysis from 251 to 126.

The riparian quality index (RQI) of del Tanago and de Jalon (2006) was used to quantify river and riparian ecosystems condition. I calculated the RQI for sites following the updated protocol in del Tanago and de Jalon (2011). I developed a habitat quality index (HQI) that adds quantitative metrics for site level hydrologic, geomorphic and vegetation characteristics detailed in Appendix Table 8.2. The HQI is transferable between streams and canals across a range of sizes. Functional redundancy (FR) (Bruno et al. 2016) was used to identify the diversity within functional groups and is used as a component of an ecosystem's resilience to vegetation disturbance from insects, disease, or humans through biomass removal, herbicide application, etc. FR was calculated as species richness divided by functional group richness using the full plant species list. However, this metric is most useful for comparisons within a study as it is sensitive to the number of functional groups used, making comparisons to other studies difficult.

Plant species were placed into 22 functional groups (Appendix Table 8.1) defined *a priori* by origin (native or introduced, USDA Plants Database www.plants.usda.gov), National Wetland Inventory wetland indicator status (Lichvar et al. 2012), and growth form (e.g. grass, forb, from USDA Plants Database). Three categories of wetland indicator status were used; wetland (obligate and facultative wetland), mesic (facultative, facultative upland) and upland. *A priori* groups were used because little is known about plant species responses to fluvial disturbance including inundation, burial and physical damage for many observed species. Similar attributes have been suggested for use in riparian plant guild creation (Merritt et al. 2010), especially for woody taxa (Hough-Snee et al. 2015). Species were replaced by functional groups and cover summed to create a functional vegetation dataset. The functional groups were treated as species in statistical analysis. Additional vegetation metrics are described in Appendix 8.2.

2.2.1.4 Aquatic Macroinvertebrates

Aquatic macroinvertebrates were collected at each site every 3 weeks from May-August 2015 using a D-frame kick net with a 500-micron capture net. The top 5 cm of sediment and submerged vegetation were disturbed for three minutes. A sweeping motion under the water was used to simulate flow in stagnant pools. Micro-habitats including riffles, pools, banks, submerged and aquatic vegetation were pooled as one sample. Presence of submerged vegetation and aquatic macrophytes was recorded during each sample as present or absent.

All aquatic macroinvertebrates were removed from samples the same day and preserved in 80% ethanol. Volume based subsampling was used in the few cases where more than several hundred individuals occurred in a sample (Hickley 1975). Individuals were identified to genus for most groups; worms and leaches to family. A total of 97 samples were collected and analyzed and five samples contained no organisms. The number of samples collected at each site varied between three and seven, therefore abundance counts were averaged for each site as opposed to summed.

I calculated the traditional EPT index (Ephemeroptera [mayflies], Plecoptera [stoneflies], and Trichoptera [caddisflies] (Plafkin et al. 1989; Kerans and Karr 1994) for taxa richness (EPT_r) as well as the proportion of EPT taxa using abundance (EPT_a) (Rosenberg et al. 2008). The EPT_r index was calculated for each site as the number of EPT taxa divided by the total number of taxa present, the EPT_a index replaced richness with abundance in the proportional calculation. Temporal change in aquatic macroinvertebrate communities was calculating as distance between consecutive samples in 3-dimensional ordination space with axes scores taken from the non-multidimensional scaling plot created from the species dataset using Eq. 1.

$$D = \sqrt{(x_2 - x_1)^2 + (y_2 - y_1)^2 + (z_2 - z_1)^2} \qquad \text{Eq.1}$$

Physiological and ecological traits (Poff et al. 2006) of genera and families were used to create the functional aquatic dataset. Taxa abundances were combined using the functional groups. EPT_r and EPT_a indices and temporal change distances were tested for similarity between groups using Student's T-test.

2.2.2 Statistical Analyses

Community analyses for vegetation and aquatic macroinvertebrates were performed using Primer v.7 software (Clarke et al. 2014). Non-metric Multi-Dimensional Scaling (nMDS) was used to analyze the overall structure of communities and principal components analysis for physical characteristics of riparian and aquatic habitats. For all statistical tests an alpha < 0.05 indicated a significant result. Diversity metrics and habitat indices were tested for differences between all groups using ANOVA and between channels controlling for land use (i.e. ag. canal vs. ag. stream) using Student's t-test.

2.2.2.1 Environmental Variables

Environmental variables were normalized by subtracting the mean and dividing by the standard deviation for each variable (Clarke et al. 2014). Euclidean distance was used to calculate a similarity matrix for environmental variables. Environmental variables were related to both species and functional group datasets for vegetation and aquatic macroinvertebrates using distance-based linear modeling with the AICc selection criteria and by Spearman Rank correlation of environmental and biotic similarity matrices.

2.2.2.2 Ecological Variables

Abundance data for plants and macroinvertebrates were square-root transformed to down weight dominant taxa and included the predictive value of infrequent species and a Bray-Curtis similarity matrix was calculated. An nMDS was calculated to visualize the data and multivariate dispersion was tested using the PermDISP routine in Primer with distances measured to the centroid of the group. Permutational multivariate analysis of variance (PerMANOVA) was used to test the effects of adjacent land use, channel type, and their interaction on riparian vegetation composition. Due to the unbalanced sampling design, Type III sum of squares and unrestricted permutation of the raw data were used in the calculation with 999 permutations. Comparisons that resulted in fewer than 100 permutations were reported with Monte Carlo corrected p-values. These statistical analyses were performed on the species/taxonomic and functional datasets for vegetation and macroinvertebrates.

2.3 Results

Land use differed markedly between assessment areas. Representative areas of residential and agricultural land uses demonstrate the difference. The selected area in City of Fort Collins (Fig 2.3A) is 86.9% residential and Weld County (Fig 2.3B) is 89.7% agricultural. The length of irrigation canals in the residential area was 33 km compared to 25.5 km of streams, for a canal/stream length ratio of 1.3/1. In the selected agricultural area of Weld County 98.5 km of canals and 22.5 km of streams occurred for a ratio of 4.4/1. In Fort Collins, many canal reaches are piped underground resulting in the shorter canal length.

2.3.1 Hydrology

Flow varied by two orders of magnitude in all study channels (Figure 2.5). Precipitation driven peak flows (Fig. 2.5A) and snowmelt runoff (Fig. 2.5B) were most prevalent on the small and large streams, respectively. An early summer peak of short duration (Fig. 2.5C) and more sustained late summer flows (Fig 2.5D) were characteristic of small and large canals, respectively.

Flow variability assessed as daily coefficient of variation (CV) and the R-B Flashiness Index (Table 2.1) indicated that most streams had greater variability than canals; the exception being Dry Creek whose watershed has several storm water retention structures. The date of peak flow was 22 days later for large canals and 51 days earlier for small canals than similar sized streams. Large canals had a similar number of low flow events and days as the large river while small canals had more than small streams. The number of zero flow days during canal operation varied. The average number of days with flow from April 1-September 30 (183 days) was 174 (SE = 3.1) for the large canal, 47.8 (6.1) for small canals and 183 (0) for streams. This is illustrated by a small canal that began to flow later and ended earlier than all other channels (Fig. 2.5C).

2.3.2 Environmental Characteristics

Canals varied from 1 to 22 m wide and 0.4 to 2.5 m deep at bank full flow. Streams had a similar range, 1.5 to 30 m wide and 0.6 to 2.5 m deep at bank full flow. Residential canals and streams had a higher proportion of sites with a woody canopy (e.g. *Salix* x *fragilis* L. and *Populus deltoides* Marshall subsp. *wislizenii* (S. Watson) and submerged overhanging terrestrial vegetation (e.g. *Phalaroides arundinacea* (L.) Rauschert and *Carex emoryi* (Dewey)) while agricultural canals and streams more frequently contained macrophytes. The ratio of observed depth to bank full depth (*Ratio*) was highest in residential canals in June (mean = 0.79) with a negative correlation (r = -0.43) with Julian date. Agricultural canals had the opposite correlation (r = 0.43) with consistently higher flows in August with a mean *Ratio* of 0.97. Mean water temperature in canals in agricultural areas was 5-6 ° C warmer than residential streams and canals, and agricultural streams were also 5-6 ° C warmer than residential streams. *Temp* in canals was more positively correlated with month (r = 0.47) than distance to the point of diversion (r = 0.34) and was negatively correlated to woody canopy (r = -0.46).

2.3.3 Vegetation

A total of 3,247 plots were averaged to create vegetation cover for the 54 sites. Forbs accounted for approximately 50% of species, grasses 32% and shrubs and trees 14%. Native species comprised 51% of the total richness but contributed only 36% of total plant cover. Three vascular plant species *Bromopsis inermis* (Leysser), *P. arundinacea*, and one native, *C. emoryi* dominated canals and streams, in both urban and agricultural regions, combining for 49% of total cover. These three species were present at 87, 76, and 61% of sites. ANOVA indicated that diversity metrics and vegetation indices (Table 2.2) were not significantly different between canals and streams, likely due to the higher variance in streams. Stream and canal vegetation did not separate into distinct groups in non-metric multidimensional scaling (Fig. 2.6).

Multivariate dispersion was not significantly different by land use (agriculture vs. urban) (species: F = 0.84, p = 0.389; functional: F = 0.007, p = 0.944). The same was true for channels (canal and stream) (species: $F < .001$, $p = 0.981$; functional: $F = 0.09$, $p = 0.832$). Pairwise PerMANOVA tests indicated 3 of 4 comparisons had significantly different species composition (Table 2.3) with streams similar across land uses. The functional dataset indicated only two comparisons were significantly different (Ag. canal *vs*. Res. canal and Res. canal *vs*. Res. stream), with Ag. streams and Ag. canals being statistically similar.

Tree and shrub cover along canals and streams in agricultural areas was low with high cover of grasses and forbs compared to residential streams (Figure 2.7). Agricultural streams had a more open forest structure dominated by *P. deltoides* with only 2 shrub species compared to residential streams with a mixed canopy of *P. deltoides*, *S. fragilis* and *Fraxinus americana* L. and a more diverse woody understory of five species. Bare sediment occupied twice the cover along ag. canals with average 21% cover compared to other sites with 11% or less.

On average, sites contained 9.8 functional plant groups with no significant difference between streams and canals ($t = 1.68$, $p = 0.361$) or when comparing channel type within land uses (Ag.: $t = 1.7$, $p = 0.202$; Res.: $t = 1.73$, $p = 0.475$). The number of functional groups at each site did not change after removing rare species. Introduced upland forb was the most common functional group occurring at 98% of sites, and introduced wetland grasses comprised 24% of all vegetation cover, the most of any functional plant group. Forbs (including vines and cacti) were a relatively small proportion of total cover (18.5%), though they accounted for the highest species diversity of any group with 142 species, or 60% of all species in the study area.

The Riparian Quality Index (RQI) and Habitat Quality Index (HQI) metrics indicated similar patterns for different groups (Figure 2.8), though the magnitude of difference between groups was noticeable larger for the RQI and the correlation between the two indices was only $R = 0.59$. Agricultural canals had the lowest scores for both indices. Pairwise comparisons between

HQI values for agricultural streams were not significantly different from residential canals ($t = -1.00$, $p = 0.18$) or agricultural canals ($t = -2.07$, $p = 0.056$).

Environmental variables were poor predictors of species and functional group composition for riparian vegetation. Results of distance based linear modeling were not significant and did not exceed an $r^2 = 0.10$ using species or functional group datasets. A Spearman Rank correlation of the site environmental matrix to species and functional group matrices selected *Land* with a $\rho = 0.131$ and 0.137, respectively, though these were not statistically significant.

2.3.4 Aquatic Macroinvertebrates

Aquatic macroinvertebrates were relatively diverse across the study area with 108 taxa identified. The most diverse order collected was Diptera with 45 taxa, Ephemeroptera with 19, and Coleoptera with 16. Site level richness was higher for residential streams (36.5) and agricultural streams (28) than residential canals (20.3) and agricultural canals (15.3). Many taxa were infrequently collected with 25% collected only once. Thirty-one taxa were only found in streams and the same number for canals, and 47 in both streams and canals. There was overlap between site groups in Figure 2.9 for both functional and species datasets with PerMANOVA pairwise tests confirming similarity between site groups in Table 2.4.

Seventy aquatic macroinvertebrate taxa that accounted for 71% of collected individuals were classified by functional traits into six groups by kR means cluster analysis ($R = 0.94$). Functional group A included predators sensitive to degraded water quality including stoneflies *Isoperla* spp. and *Pteronarcella badia* (Hagen). Group B were moderately tolerant larger predators such as the predaceous diving beetle *Agabus* spp. and the stonefly *Claassenia sabulosa* (Banks). Sensitive collector gatherer taxa are split into two groups: C with more caddisfly taxa (ex: *Brachycentrus* spp., *Hydropsyche* spp.) and F with more mayfly taxa (ex: *Baetis* spp., *Ephemerella dorothea infrequens* (McDunnough)). Chironomid midges that were tolerant of poor water quality including the abundant species of *Chironomus* and *Cricotopus* (Poff et al 2006) were characteristic of Group D. Group E were also tolerant taxa with mostly aquatic adults such as the water bugs *Trichocorixa* sp. and *Belostoma* sp.

A permutational distance-based test for homogeneity of multivariate dispersions (PERMDisp) on site averaged data indicated no differences when sites were grouped by land use and channel type for the taxonomic ($F = 1.59$ $p = 0.833$, df1 = 3, df2 = 16) and functional datasets ($F = 3.78$, $p = 0.293$, df1 = 3, df2 = 16). Aquatic macroinvertebrate communities were significantly different between agricultural and residential canals when taxonomic and functional dataset were analyzed using PERMANOVA (Table 2.4). All other comparisons were not significant and the two datasets agreed.

The EPT_r index was similar between site groups but generally higher in residential areas (Table 2.5). The proportional abundance of EPT taxa, however, was on average more than double for streams. Channel size was a major contributor to EPT for streams where the large Cache la Poudre River sites had EPT_r and EPT_a of approximately 32 and 80, respectively while smaller streams were 13 and 3, respectively. The mayfly genus *Tricorythodes explicatus* (Eaton) is relatively tolerant of poor water quality and higher water temperatures and comprised the 45% of the EPT abundance in Ag. streams but was not common in Ag. canals. Residential canals and streams shared many EPT taxa including the mayflies *Baetis*, *Ephemerella* and *Heptagenia*.

The Pearson correlation between environmental and biotic similarity matrices was low for both species ($R = 0.12$, $p = 0.003$) and functional group datasets ($\rho = 0.096$, $p = 0.036$). Spearman rank correlation and distance based linear modeling (DistLM) using the AICc selection criteria identified similar sets of environmental variables, though the predictive power of each was relatively poor. Each selected *BF_Depth, Dist, Canopy* and *Macros* while DistLM (r^2 = 0.17) also included *Bed_Sub* and Spearman rank (ρ = 0.16) included *Land*. Riparian condition indices (RQI and HQI) were moderately to well correlated with the EPT_a index (Pearson R = 0.59, t = 3.06, p = .007; and R = 0.69, t = 4.05, $p < 0.001$, respectively). However, only the HQI variable was significantly correlated with EPT_r (R = 0.47, t = 2.28, p= 0.035).

Student t-tests on sample distances for each period did not show a difference in temporal variation between canals and streams, however early and late season patterns appear different (Figure 2.10). Large changes in communities (D > 2) occurred only in canals during the early part of the crop growing season when most canals are flowing. The range and median values were statistically similar between channel types in mid-late summer.

2.4 Discussion

Canals exceed streams in length with a 4.4/1 ratio in the agricultural area and 1.3/1 in res./urban area of the northern Colorado study area. Although irrigation canals are anthropogenic waterways with complete human management of flow and physical form, as a group they support similar species and functional groups as natural streams and they both were influenced by adjacent land use. Previous studies on the ecological attributes of constructed waterways have focused on agricultural drainage ditches (Herzon and Helenius 2008; Leslie and Lamp 2017) and transportation canals (Harvolk et al. 2013, Dorotovicova 2013) concluding that artificial waterways can support diverse biotic communities.

Hydrologic and geomorphic processes have acted on the study irrigation canals for over a century, but horizontal connectivity and in-channel physical heterogeneity is intentionally limited by engineering design and ongoing maintenance to clear material from the canals. Under normal canal operation, overbank flooding, local erosion, sedimentation and channel migration, characteristic of streams, do not occur or are rare processes (Depeweg and Mendez 2002). The design and construction of irrigation canals limits groundwater supported base flows or surface water inputs characteristic of a stream's interaction with its watershed.

Streamflow had distinct inter- and intra-annually variability. Peak flow in small canals occurred earlier than similarly sized streams and the opposite was true for large channels. A striking difference was that canals had measurable flow for only 147 days during the 183 day April-September study period while all study streams were perennial with flow on 183 days.

2.4.1 Vegetation

The hydrologic and geomorphic differences between streams and canals did not cause significant differences in species richness, Shannon diversity index, and Simpson's index. Diverse aquatic and riparian vegetation also occur in agricultural drainage ditches in temperate regions of the northern hemisphere (Herzon and Helenius 2008) including north-west Europe (Verdonschot et al. 2011). I found plant species occurrence and abundance differed between canals and streams in agricultural and residential landscapes. Species diversity alone does not equate to ecosystem health (Moonen and Barberi 2008). Functional plant groups or guilds have been used to characterize environmental filters in riparian ecosystems (Hough-Snee et al. 2015), specific habitat types for wildlife (Merritt and Bateman 2012), and ecological functionality of riparian ecosystems (see Merritt et al. 2010; Bruno et al. 2016). Analysis of the composition and abundance of functional plant groups largely supported species differences found between canals and streams. The exception was the vegetation along agricultural canals and agriculture streams had statistically similar functional groups.

Functional plant attributes varied between residential and agricultural landscapes. Fewer sites were characterized by woody vegetation (shrubs and trees) in agricultural streams and canals compared to residential channels. Vertical structure and species diversity were also significantly lower. This may be due to vegetation management along canals and limited protection of streams in agricultural landscapes. This has implications for wildlife that require structurally diverse riparian zones to provide cover and food (Meaney et al. 2003; Lopez-Pomares et al. 2015). Wetland plant cover was also lower on channels in agricultural landscapes, likely related to plant tolerance of vegetation management methods including burning and herbicide application as well as a potentially more variable and erratic stream flows.

The restoration of degraded riparian ecosystems along streams in agricultural landscapes has a long history in many regions (Osborne and Kovacic 1993; Zedler 2003; McTammany et al. 2007). The similarity of riparian vegetation along irrigation canals and streams in agricultural landscapes suggests that there is the potential to improve the ecological functioning of these artificial waterways if land use and management stressors are reduced. The habitat quality index (HQI) of residential canals was comparable to agricultural region streams indicating that irrigation canals in residential areas are providing riparian qualities like degraded agricultural streams, including species richness and vegetation structure. Lateral connectivity through flooding is a component of the HQI score that is absent from canals. This limits the extent and interaction with upland ecosystems affecting nutrient cycling through in the riparian and aquatic systems of canals similar to streams disconnected from their floodplains (Ward et al. 1999; Valett et al. 2005).

2.4.2 Aquatic Macroinvertebrates

Macroinvertebrate diversity and abundance are influenced by channel physical characteristics, food type and availability, and hydrologic patterns at relatively small scales (Resh et al. 1994). In artificial channels, such as canals, streamflow patterns and variability can be analogous to natural streams yet, channel maintenance and riparian vegetation management create disturbance and limit the formation of aquatic habitats and food resources. Irrigation canals in this study supported diverse macroinvertebrate communities that were similar to streams in agricultural and residential landscapes using both taxonomic and functional datasets. Habit et al. (1998) found similar species and biomass in irrigation canals and streams in Chile and Koetsier et al. (2015) in Idaho, U.S.A. Small lateral canals at terminal positions in the irrigation network had fewer taxa dominated by chironomids and *Simulium* likely in response to short duration of flows.

The USGS National Hydrography Dataset categorizes the flow regime of many local streams as “intermittent” or “ephemeral” and the spring and summer streamflow in canals could mimics these intermittent streams, thus species adapted to intermittent flow would be present in the regional species pool and available for colonization (Mackay 1992; Miller and Golladay 1996; Bogan and Boersma 2012). Studies investigating aquatic macroinvertebrates of drainage ditches showed a selection towards morphological or life history adaptations to varied flow regimes (Whatley et al. 2015, Leslie and Lamp 2017) and the same would be expected of canals with variable flows. Within and between year flow patterns in canals are highly variable (see Fig. 2.5) limiting taxa with strong seasonal life history patterns. The onset or delay of irrigation could impact canal physical and chemical environments, creating unfavorable conditions during critical life stages including emergence and oviposition (Mackay 1992). However, tolerant taxa such as

Belastoma, *Chironomous*, could be relatively unaffected by flow reductions (Brown et al. 2012) especially where decades of variable and intermittent flows have limited colonization by sensitive taxa.

With taxonomically and functionally similar communities, it appears that the physical conditions of aquatic ecosystems are similar between streams and canals during the spring and summer. Hypothesized changes to macroinvertebrate communities in natural streams due to climate change (Pyne and Poff 2017) could also occur in irrigation canals, but may be mitigated or intensified by human controlled flows in canals if sufficient water is available.

Aquatic macroinvertebrates use several life history strategies to inhabit temporary waters including adult colonization, rapid maturity, and desiccation resistance (Mackay 1992; Miller and Golladay 1996; Tronstad et al. 2007; Bogan and Boersma 2012; Whatley et al. 2015). This likely caused the significant changes in macroinvertebrate communities in irrigation canals during May and June when flows were initiated while July and August showed similar rates of change to perennial streams (Fig. 2.10).

The rapid riparian assessment approaches used in this study were moderately to well correlated to EPT_r and EPT_a as the influence of riparian condition on aquatic macroinvertebrate composition have been well studied (Briers and Gee 2004; Arnaiz et al. 2011; Greenwood et al. 2012). Physical and chemical characteristics of aquatic ecosystems such as water temperature, food resources, and in-channel woody debris are dependent in part on inputs from adjacent riparian ecosystems (Pozo et al. 1997; Lyons et al. 2000; Moore et al. 2005; Baxter et al. 2005). These can be even more important in agricultural landscapes where riparian buffers are being increasingly utilized to improve water quality, and aquatic habitat and terrestrial connectivity (Wooster and DeBano 2007). A by-product of riparian vegetation along irrigation canals could be improved aquatic habitat, due to increases in allochthonous carbon input and thermal buffering (Bunn et al. 1999). This could change the aquatic food web and linkages with the surrounding riparian and terrestrial ecosystems (Nakano et al. 1999; Henschel et al. 2001; Nakano et al. 2001; Sabo and Power 2002; Ballenger and Lake 2005). Not all riparian areas have woody canopies, and prairie streams were likely dominated by low herbaceous vegetation pre-settlement, with herbaceous canals acting as modern analogs (Dodds et al. 2004; Vandermyde and Whiles 2015).

2.4.3 Maintenance of Irrigation Canals

Canal maintenance, including mowing, burning, the application of herbicide, and the removal of woody plants affect riparian vegetation. Periodic removing of bed sediment to ensure consistent gradient and unobstructed flow affects aquatic macroinvertebrate communities. Andersen and Nelson (1997) suggested that vegetation management limited butterfly habitat along a canal in western Colorado and Meaney et al. (2003) noted that canal maintenance degraded riparian habitat for a threatened small mammal. Adjacent land uses influenced the type and intensity of maintenance activities. For example, the use of fire and large machinery to clear brush and woody debris from canals is not feasible in residential areas and could have allowed the growth of woody plants. In addition, public use of canals and maintenance roads for recreation could influence the type and frequency of vegetation management (Aspe and Jacque 2015). Conversely, intense and frequent vegetation management was observed along agricultural canals. Dredging canals to remove sediment occurred in both urban and agricultural landscapes and severely alters channel bed conditions for aquatic macroinvertebrates and negatively affected diversity and EPT indices in agricultural canals (Shaw et al. 2015). The intensity and frequency of vegetation and physical channel maintenance is expected to vary within and between land use categories according to need, resources and social preferences. Quantitative data describing management activities is critical for identifying the responses of riparian and aquatic ecosystems to management and to develop recommendations for improving the ecological functions of canals while maintaining the water conveyance purpose of the canal similar to the approach in Lyons et al. (2000).

2.5 Conclusions

A holistic view of canals as integrated elements of the landscape is necessary to recognize their support of local and regional ecosystems. The prevalence of canals in many semi-arid landscapes make them the dominant channel type; coupled with the volume of water diverted canals become a common if not dominant lotic ecosystems by length. The landscape setting influences the quality of streams and canals and their potential similarity. Canals in agricultural landscapes supported riparian vegetation and aquatic macroinvertebrate communities similar to streams. These results give support to the concept that canal networks have added habitats often typical of stream ecosystems. Riparian vegetation of canals in residential areas differed from streams, yet had relatively high habitat quality values.

It is critical to understand the physical and biotic attributes of these anthropogenic ecosystems as they support rare species (Meaney et al. 2003), support habitat connectivity, trophic subsidies and supporting regional biodiversity (Fernald et al. 2010; Lopez-Pomares et al. 2015). Pervasive human impacts to streams in agricultural and residential landscapes further highlight the potential importance of riparian and aquatic ecosystems of canals. Irrigation canals have been instrumental in supporting agriculture and I have shown that the ecological byproducts of created riparian and aquatic habitats occur in sufficient amounts and with similar biological communities as natural steams to considered an integral component to the agro-ecosystem of northcentral Colorado.

3. Nitrate removal potential of tailwater wetland ecosystems in an irrigated agricultural landscape.

3.1 Introduction:

Nutrient pollution is a widespread environmental problem often linked to agricultural practices, and causing significant impacts to ecosystem health and natural functions (Carpenter et al. 1998; Schroder et al. 2004; Collins and McGonigle 2008). Concentrated cropland and animal production facilities such as feedlots have been identified as contributing to diffuse non-point source pollution (Saintfort et al. 1991; Giupponi 1995; Kronvang et al. 1995). In many portions of the world, elevated concentrations of N and other agricultural nutrients degrade freshwater ecosystems through eutrophication and may be toxic to plants and animals (Carpenter et al. 1998).

The Food and Agriculture Organization of the United Nations reported that 20.1% of agricultural land was irrigated in 2008 with a total fertilizer (nitrogen (N), phosphate (P_2O_5), potassium oxide (K_2O)) consumption of 161.8 million tons per year, expected to increase to over 200 million tons by 2018 (FAO 2015). Precipitation and irrigation on agricultural land may create runoff and deep percolation, often transporting high concentrations of N, primarily as ammonia (NH_4^+) and nitrate (NO_3^-) ions. High levels of these and other nutrients negatively impact fresh and saltwater ecosystems and contaminate groundwater (Carpenter et al. 1998; Foster and Chilton 2003). Elevated concentrations of NO_3-N in drinking water poses a health threat to humans in many developing and developed nations (Powlson et al. 2008). High concentrations of NO_3-N in surface water can also cause algal blooms and acidification that degrade water quality, damage water supply infrastructure, and result in the mortality of aquatic organisms (Camargo and Alonso 2006).

Nutrient cycling is a critical ecosystem function performed by wetlands (Mitsch and Gosselink 2000; Zedler and Kercher 2005). Physiochemical and hydrologic conditions interact with site vegetation and microorganisms to regulate the assimilation, transformation, and availability of several important nutrients including N, sulphur (S), phosphorus (P) and iron (Fe) (Lamers et al. 2012). The chemical and biological processes in wetlands are controlled in part by cycles of aerobic and anaerobic conditions of soil and water (Hefting et al. 2004; Vyzamal 2007). Plant-mediated N cycling within wetlands occurs through uptake and growth, decay of biomass, root exudates, and seasonal within-plant translocation between shoots and roots (Woo and Zedler 2002; Bastviken et al. 2015). Nitrogen can be transformed into many chemical forms, including NO_3^-, NO_2^-, NH_3, NH_4^+, N_2, and N_20, through biological processes including nitrification, denitrification, ammonification, and gasification. The process of denitrification occurs in anaerobic conditions and converts NO_3-N into N_2 when a carbon source is readily available for microbial communities.

The capacity of wetlands to intercept, retain and transform pollutants has been well documented (Gregoire et al. 2009; O'Geen et al. 2010). Wetlands are often constructed to treat runoff and improve water quality from agricultural, urban, industrial and mine runoff (Diaz et al. 2012, Vymazal 2014, O' Sullivan et al. 1999) and have been incorporated into many wastewater treatment systems (Kivaisi 2001; Liehr and Kruzic 2007; Stottmeister et al. 2003). However, the use of treatment wetlands in agricultural landscapes has had mixed success (Vymazal 2007, Knox et al. 2008, Brauer et al. 2015). Studies of constructed and restored wetlands with fixed inlets and outlets (Beutel et al. 2009; Diaz et al. 2012; Brauer et al. 2015) are necessary to understand dominant processes, yet these concepts have not been widely tested on wetlands receiving high concentrations of NO_3-N in diffuse runoff or through shallow groundwater. The efficacy of constructed wetlands to remove, stabilize or transform pollutants varies with the physical and biological components of the wetland, land use in the contributing watershed, age of the wetland, and the inflow rates (Vyzamal 2007). The relative importance of plant uptake and denitrification for N mitigation varies with hydrologic condition, vegetation, and input dynamics (Edwards et al. 2006; Borin and Tocchetto 2007).

Wetlands have formed in many agricultural landscapes where surface or groundwater flow is concentrated creating saturated or ponded conditions from irrigation runoff and drainage (Budd et al. 2009). Irrigated agriculture may increase the timing, duration and extent of saturated conditions in natural wetlands (Peck and Lovvorn 2001). These wetlands are often considered waste land by producers because they cannot be easily drained or cultivated. However, the position and characteristics of some wetlands could be providing significant treatment of runoff containing NO_3-N.

Wetlands also form adjacent to irrigated croplands and their associated higher water tables can be linked to water application and management creating tailwater wetlands. Soil amendments, fertilizers and pesticides applied to crop and pasture land can be transported by water and deposits in or flows through tailwater wetlands that develop downslope of fields from surface runoff or shallow groundwater discharge (Budd et al. 2009). Riparian buffer strips occupy a similar landscape position between crops and streams, and have been used worldwide to intercept and abate impacts from water borne pollutants (Osborne 1993, Muscutt et al. 1993; Vought et al. 1994; Hickey and Dorn 2004, Zhang et al. 2010, Dosskey et al. 2010) sediment (Yuan et al. 2009), phosphorus (P) (Hoffman et al. 2009), pesticides (Arora et al. 2010), and NO_3-N (Kovacic et al. 2000, Mayer et al. 2007). Wetlands provide similar functions as vertical buffers between agricultural runoff and shallow groundwater recharge areas, but the extent and importance of this process at local and regional scales is not well understood.

Tailwater wetlands occupy a relatively small portion of agricultural landscapes, but their biogeochemical processes may sustain local and regional nutrient cycling (Blackwell and Pilgrim 2011). To clarify the role of these wetlands in nutrient cycling, I addressed three questions on the functioning of tailwater wetlands in agricultural landscapes: 1) Do incidental wetlands intercept surface and/or groundwater with elevated concentrations of NO_3-N, 2) What physical, chemical and biological characteristics contribute most NO_3-N transformation, and 3) How does N assimilation differ by wetland plant species? These questions address two primary biologically-mediated pathways of NO_3-N abatement by wetlands, assimilation and transformation. By understanding the rates of NO_3-N assimilation and transformation in different wetland types I aimed to identify key physical or biological factors that could be modified to improve NO_3-N abatement in wetlands.

To address my focal questions, I selected an area where groundwater NO_3-N pollution, irrigation and wetlands coincide. Weld County in northcentral Colorado has been irrigated for nearly 150 years (Eschnur et al. 1983; Strange et al. 1999) with impacts to groundwater including consistently elevated NO_3-N concentrations recorded since the 1960's (Waltz 1969). A recent study found 80% of irrigation and 45% of domestic wells exceeded the EPA drinking water standard of 10 mg/L (Bauder et al. 2006).

3.2 Methods

3.2.1 Study Area

The study area is in northcentral Colorado (Figure 3.1), a region representative of semi-arid western U.S. Great Plains with extensive irrigated agriculture. Precipitation averages 250 millimeters during the summer and 135 mm during winter (www.usclimatedata.com), which is insufficient to support crops without irrigation (Schneekloth and Andales 2009). Most irrigation water in the region is diverted from rivers fed by melting snow in the Rocky Mountains (Strange et al. 1999; CWCB 2015).

Wetlands were selected that appeared to be supported by drainage from irrigated fields as evidenced by land slope, linked surface erosion and sediment deposition. Wetland vegetation extended to the field edge and sometimes into the crop field. All sites were emergent wetlands with three sites were dominated by *Typha latifolia* L. and three sites were wet meadow wetland dominated by *Schoenoplectus pungens* (Vahl) Palla, *P. arundinacea*, *Carex nebrascensis* Dewey, *Juncus gerardii* Loiseleur and *J. arcticus* Willdenow using regional taxonomy following Weber and Wittmann (2012). Sites B1, B2 and F were located between agricultural fields and irrigation reservoirs (Figure 3.2). Sites S1, S2 were drained by small channels while site E was a shallow depression with no channelized outlet.

3.2.2 Wetland Type and Distribution

Wetlands were classified following the NWI classification (Cowardin et al. 1979). Nearest neighbor analysis was performed on aggregated wetland polygons using expected Euclidean distance to assess wetland distribution across the landscape. Nearest neighbor ratios, z-scores and p-values are reported.

3.2.3 Hydrology, Precipitation

Ground water monitoring wells were constructed to a depth of 125 cm, and casing the borehole with 6.35 cm i.d. slotted PVC pipe, packed with silica sand and annuli and sealed at the surface with bentonite. Wells were installed every 10 m along transects oriented perpendicular to wetland boundaries and instrumented with recording pressure transducers (Troll 100: In-Situ, Fort Collins, Colorado). Pressure measurements were made every 30 minutes during the growing season (April-September) and corrected using a BaroTroll (barometric pressure logger, In-Situ, Fort Collins, Colorado) and converted to depth to water below the surface. The number of days the mean groundwater depth was within 25 cm and 50 cm of the surface was calculated (reference). Precipitation and sprinkler irrigation were measured bi-weekly in the field with rain gages with mineral oil to impede evaporation.

3.2.4 Soil Characteristics and Microbial Biomass

One 100 cm deep soil pit was dug at each groundwater well in each study wetland. Soil characteristics were measured for the 0-25 and 25-50 cm layers by first collecting samples from the pit wall using a 5.5 cm i.d., 4 cm tall brass ring, drying them to a constant weight at 55˚ C, and calculating bulk density (g/cm^3). Organic matter content was measured by loss on ignition (Nelson and Sommer 1982) by burning approximately 4 g of dried samples at 550˚ C and determining mass difference between burned and unburned sample. Soil particle size distribution was estimated using the Stokes Law of settling time (Day 1965); 15 ml of loose soil was combined with 30 ml of water in 50 ml centrifuge tubes, shaken vigorously for three minutes. Volume of settled sediment was recorded at one minute for sand, 2 hours for silt and two days for clay.

The chloroform fumigation protocol was used to determine microbial biomass carbon and nitrogen following procedures of Beck et al. (1997) and Brookes et al. (1985). Liquid collected from filtration with Whatman #1 filter paper was diluted to 1/10 concentration to reduce the concentration of potassium salts. Liquid samples were run on a Shimadzu TOC-L with combustion type, Non-Dispersive Infrared (NDIR) detector for Total Organic Carbon (TOC) and results reported in standard units of μg C/gram dry soil.

3.2.5 Groundwater Chemistry

Groundwater samples were collected every 2-3 weeks for a total of seven possible sample dates per well. Samples were collected after purging two well volumes with a PVC bailer. Samples were filtered with a 0.02 micron PFE filter, stored on ice and delivered to the Colorado Department of Agriculture Biochemistry Lab on the same day. Groundwater pH, conductivity and temperature were tested using a multi-probe meter (Thermo Scientific Orion meter) in August 2015.

3.2.6 Vegetation, Plant Uptake

Absolute cover of wetland vegetation was visually estimated in 1 m^2 plots. Aboveground plant material was collected from 100 cm^2 subplots within 2 m of groundwater wells and soil pits in mid-August to measure dry biomass and C:N ratio. Live biomass and litter were collected separately. Samples were dried in a 55˚ C oven to a constant weight. Total biomass was recorded then subsamples were ground to a fine powder on a Retsch grinder and analyzed for carbon and nitrogen content using a LECO Tru-Spec CN dry combustion type analyzer (Leco Corp., St. Joseph, Michigan, USA) with infrared detection for carbon and thermal conductivity detection for nitrogen.

3.2.7 In-Situ denitrification experiment

The method utilized is comparable to intact core incubations described by Balderson et al. (1976), Yoshinari and Knowles (1976) and Sorenson (1978) where acetylene (C_2H_2) is added to a closed chamber containing wetland soil and incubated in the laboratory. C_2H_2 blocks the reduction of nitrous oxide (N_2O) to nitrogen gas (N_2) during denitrification and N_2O builds up in the chamber. The chambers utilized in this study were installed in the wetland and not removed during the sampling period (Groffman et al. 2011). Incubation chambers were constructed of 7.8

cm i.d. PVC tubes, 75 cm in length. A brass port, silicone tubing and in-line stopper were installed for gas sampling. The chambers were installed within 1 m of groundwater wells and soil pits (Figure 3.3). Caps were installed and sealed with silicone caulk and petroleum jelly for an airtight seal. Acetylene was added at a concentration of 10% of the headspace volume and allowed to incubate for 2-18 hours depending on water table depth. Air samples were collected at 1-2 hour intervals using a sterile syringe and injected into evacuated 20 ml glass scintillation vials with butyl septa. Samples were processed on a Shimadzu GC14B gas chromatograph with FID, methanizer ECD, and autosampler for C_2H_2, NO_2, carbon dioxide (CO_2) and methane (CH_4).

Denitrification rates were calculated using the ideal gas law to convert measured concentrations into a daily rate of g N/m^2/day. N_2O concentrations were corrected using dilution factors for multiple samples. CO_2 concentration was used as a surrogate to confirm air-tight seals. The rate was determined as the slope of the line connecting accumulated nitrous oxide (the intermediate N-species created during the acetylene block experiment) over time prior to an inflection in the slope. Experimental results from a 2014 pilot study were included to increase the number of successful experiments. All data was combined to determine average rates for individual wells.

3.2.8 Statistical Analyses

Pearson correlation coefficients were used to identify relationships between environmental and biological predictor variables and response variables. Best subset multiple linear regression and the AICc selection criteria were used to identity the most significant predictors of denitrification rate and N in aboveground biomass. Significance was reported when alpha < 0.05.

3.3 **Results**

3.3.1 Wetland Type and Distribution

Wetlands covered 2,032 ha, or 2.1% of the total area of the northern portion of the study area. The southern portion contained 1,447 ha of wetlands, or 3.8% of the total area. The ratio of irrigated agricultural land to wetland was 30:1 in the north and 13:1 in the south areas. Both regions were dominated by emergent wetland vegetation at more than 58%, while woody wetlands comprised less than 35%. Sites B1 and B2 (Figure 3.2) and F were characteristic emergent wetland vegetation bordering ponds. Wetlands were not evenly dispersed across the study areas. Nearest neighbor (N) analysis indicated that wetlands were clustered in both the north (N ratio = 0.64, z-score = -22.3, p-value < 0.001) and south (N ratio = 0.63, z-score = -12.4, p < 0.001).

3.3.2 Soil Characteristics and Microbial Biomass

Soil texture and carbon content varied by site, with fine grained, higher soil organic matter (SOM) at sites F, B1 and B2 compared to sites S1, S2 and E (% sand: t = -3.64, p = 0.001; SOM: t = 5.31, p < 0.001) (Table 3.1). Within a wetland SOM varied spatially and with depth (Figure 3.4), though a statistically significant relationship was not detected when data from all sites were combined. Sites with a relatively static wetland boundary, F and S2, had higher organic matter content in the upper 25 cm. At four sites, B1, B2, S1, and E, where the wetland-crop boundary varied between years, SOM was slightly higher at 25-50 cm depth, where plowing and sediment deposition were observed to bury plant material, likely increasing organic matter content of the deeper soil (Hang et al 2016). Microbial biomass was correlated to organic matter content at both depths (0-25 cm, Pearson r = 0.79, p = 0.001; 0-50 cm, r = 0.75, p = 0.003) and varied widely from 4.39 to 443.6 ug C/g dry soil. At four sites, microbial biomass was higher in the top 25 cm moving from the wetland boundary to wells at 10 m and 20 m into the wetland while one site had the opposite pattern.

3.3.3 Hydrology and Precipitation 2015

Water level dynamics in adjacent water bodies, precipitation and irrigation influenced water tables in the study wetlands. Water tables in wetlands adjacent to irrigation ponds were closely linked with levels of open water. Rapid changes in pond level resulted in rapid groundwater table change at sites B2 and F while site B1 had relatively stable pond and wetland water tables. For instance, the 4-ha irrigation pond adjacent to site F was filled on June 22, 2015 and groundwater levels responded by rising nearly 100 cm over the following three days (Figure 3.5). Conversely, groundwater levels at B1 varied only ~40 cm during the entire study period because the adjacent reservoir had stable water levels.

Regional groundwater levels rose in May in response to spring rains at sites S1, S2 and E. At site E, groundwater levels rose in response to irrigation events, followed by a gradual lowering in July and August. High late summer evapotranspiration created a diurnal water table pattern with a magnitude of several cm.

Spring of 2015 was an unusually wet period with 190–275 mm of rain, 76–110% of average summer total falling between April 1 and June 1 across the study area. This delayed planting of crops by 4–6 weeks in some areas and some producers changed crops or fallowed fields due to persistent wet soil. An additional 50-125 mm of rain fell between June 1 and Sept 1, 2015. Sprinkler irrigation added 416 mm of water to site E. The effects of irrigation and precipitation produced erosion and sediment movement that deposited as much as 6 cm of sediment at the edge of some wetlands.

3.3.4 Groundwater Chemistry

Sites B1, E and F had groundwater nitrate levels that averaged < 5 mg/L during 2015 (Table 3.2). The remaining 3 sites had different patterns of groundwater nitrate concentrations. Sites S1 and S2 had relatively stable NO_3-N concentrations that averaged 16.9 and 35.4 mg/L from June through August. In contrast, B2 had a NO_3-N peak of 54.2 mg/L in mid-July and much lower average concentrations in June of 3.8 mg/L and 0.3 mg/L in August. B2 also had the highest within site variation when comparing groundwater samples from wells along parallel transects. Wells in transect 1 averaged 27.7 mg/L higher NO_3-N than paired wells in transect 2.

Nitrate$^-$ concentrations in surface waters were < 0.03 mg/L in June and July adjacent to sites F, B1 and B2 at a time when concentrations should be highest due to high precipitation and irrigation application on fields that received pre-planting and early-season fertilizer applications. Similarly, NO_3-N levels from a channel adjacent to the wetland complex of S1 was below detection limits (0.01 mg/L) during July and August.

3.3.5 Vegetation, Plant Uptake

Total vegetated wetland area assessed at the six sites was 1.87 ha with species cover varying between sites. Wetlands were dominated by *T. latifolia*, *S. pungens*, *P. arundinacea, C. nebrascensis* and *J. gerardii*. The clonal rhizomatous growth character of these species produced monospecific patches at most sites with sharp boundaries between patches. *T. latifolia* biomass was positively correlated to mean water table depth for the April-Sept growing season ($r = 0.51$, $p = 0.018$) and late season water tables mean $r = 0.55$, $p= 0.011$ for July-Sept water table depth). Cover of *P. arundinacea* and annual forbs and grasses such as *Setaria viridis* (L.) Palisot de Beauvois, *Polypogon monspeliensis* (L.) Desfontaines and *Lactuca serriola* L. were negatively correlated to late season high water tables, mean r = -0.51, p = 0.02 and -0.42, p = 0.041 for July-Sept water table depth and were predominately located at the wetland-field boundary.

Aboveground biomass of living plant material varied by community: 2.91 (+/- 0.29) kg m^{-1} for *Typha latifolia*, 0.79 (+/- 0.11) for *S. pungens*, 0.76 (+/- 0.21) for annual grasses, 0.39 (+/- 0.06) for *P. arundinacea* and 1.26 (+/- 0.26) for *C. nebrascensis* /*J. gerardii*, *T. latifolia* reached a maximum height of 3.5 m at site B1 contributing to the high biomass while most other communities were ≤1 m tall. Percent dry biomass as N was highest for *P. arundinacea* (3.23, +/- 0.57) and lowest for *C. nebrascensis*/*J. gerardii* (1.46, +/- 0.10). N content in aboveground biomass was positively correlated with the percent of time groundwater was within 25 cm of the surface (Spearman Rank = 0.58, t = 2.75, p = 0.015). Total N contained in annual aboveground biomass varied significantly by site and vegetation within a site (Figure 3.7) with 30 times more N in *T. latifolia* at site B1 than *S. pungens* in site E. Litter mass and N content were not correlated to hydrologic metrics, nor living biomass.

3.3.6 Denitrification rates

Denitrification rates were four-fold higher for soils under *P. arundinacea* (42.9 kg N ha^{-1} yr^{-1}) and annual grass communities (50.2) than soils under *C. nebrascensis*/*J. gerardii* (4.2) and *T. latifolia* (11.8) as these were usually at the wetland-field boundary and were first to intercept surface water and shallow groundwater. Daily denitrification rates are reported at an annual scale (kg N ha^{-1} yr^{-1}) to match above ground biomass calculations, however, microbial activity would be expected to decline significantly during dry and cold winter months, thus reported rates likely overestimate the annual denitrification rate.

3.3.7 Variable Interactions

Total N in aboveground plant biomass was positively correlated with groundwater levels, in April ($r = 0.73$) and July ($r = 0.70$) and daily mean groundwater levels ($r = 0.59$). However, the relationship was non-linear. Microbial biomass in the top 25 cm of soil was positively correlated to organic matter content ($r = 0.83$) and August and September groundwater levels ($r = 0.83$ and 0.75, respectively). Deeper soil microbial biomass was only positively correlated to organic matter content ($r = 0.81$).

Several site-averaged environmental variables were correlated to biological variables (Table 3.3). Bulk density was negatively correlated with soil organic matter and microbial biomass. Aboveground plant biomass was positively correlated to percent time the water table was within 25 and 50 cm of the surface due to high *T. latifolia* cover in the wettest areas.

3.4 Discussion

Wetlands in agricultural landscapes are known to improve water quality across the world (Whigham et al. 1988; Otte et al. 2007; Thiere et al. 2009; Blackwell and Pilgrim 2011). Vegetation in tailwater wetlands was effective at N uptake with average aboveground biomass N content of 1,274 kg N ha^{-1} yr^{-1}. Transformation of NO_3-N by microbial denitrification was a minor process compared to plant uptake averaging 19.5 kg N ha^{-1} yr^{-1}, in contrast to many studies identifying this process as the dominant process for NO_3-N mitigation in wetlands (Bachand and Horne 1999; Karpuzcu and Stringfellow 2012; Maxwell et al. 2017). Tailwater wetlands in agricultural landscapes are not designed to improve nutrient retention and transformation. Physiochemical and vegetation characteristics developed naturally thus physical, biological, and hydrologic conditions favorable to trap and transformation NO_3-N vary over space and time (McClain et al. 2003; Bruland et al. 2006; Kjellin et al. 2007).

Small proportions of wetlands in agricultural watersheds can reduce NO_3-N concentrations (Stringfellow et al. 2013). Karpuzcu and Stringfellow (2012) estimated that only 3% of total land area in small agricultural watersheds would be necessary to reduce NO_3-N concentrations in agricultural drainage below 0.5 mg/L in surface water and Qui and Turner (2015) found water quality improvement with 6% wetland area. In this study wetlands occupied 5.3% (7,204 ha) of the predominately agricultural landscape, with 1,559 ha of wetlands within 35 m of irrigated and fertilized agricultural fields. The hydrologic connection to agricultural fields is critical for wetlands to intercept surface and shallow groundwater runoff with elevated nutrient concentrations. Wetlands with *T. latifolia* could uptake significant N but without human management this only acts as a temporal buffer, release N during fall and winter through liter decomposition. Denitrification, although found to be a smaller component of N mitigation, occurs without additional management activities.

3.4.1 Physical and Hydrologic Controls

Groundwater dynamics influence NO_3-N sequestration and transformation processes (Kjellin et al. 2007) and loading rates and flow paths affect its residence time (James et al. 2008; Lin et al. 2008). Wetlands in this study had shallow water tables (<50 cm) for an average of 80% of the growing season, that create anaerobic conditions favorable for denitrification (Hill et al. 2000; Seitzinger et al. 2006; Sirivehden and Gray 2006). The effect of adjacent ponds on wetland groundwater levels was to create stable levels at B1, a gradual rise at B2 and an increase in levels following pond filling (Figure 6). Ponds, drainage ditches and streams can control water levels in adjacent wetlands (Carter 1997; Karan et al. 2014; Wurster et al. 2003) and can control hydraulic gradients into the wetland that can affect NO_3-N loading rates and water residence times.

Physical (soil bulk density, organic matter content and hydrodynamics) and biological (microbial communities and vegetation) factors interact to determine N sequestration and transformation processes. Bulk density and SOM have been correlated to C and N cycling in restored (Meyer et al. 2008, Wolf et al. 2011B) and agricultural wetlands (Briar et al. 2007). In addition, bulk density and SOM are correlated to hydraulic conductivity (Hua et al. 2012), infiltration (Franzluebbers 2002) and runoff potential (Rhoton et al. 2002), which are important for understanding the movement of highly soluble NO_3-N in agricultural landscapes. I found that hydrologic and soil conditions were favorable for denitrification including soil organic matter and input NO_3-N concentrations (Waters 2012). Hydrologic conditions varied between stable, shallow water tables favorable for continued denitrification to fluctuating water tables favorable for paired nitrification/denitrification processes that could remove more total N from the system (Wolf et al. 2011A).

3.4.2. Seasonal NO_3-N *Concentrations*

Elevated groundwater NO_3-N levels in the shallow aquifer of Weld County, Colorado have been documented since the 1960's (Smith et al. 1964; Waltz 1969) with a median value in 2002 of 16 mg/L (Paschke et al. 2008). Shallow (< 1.2 m) groundwater in this study had a wide range of NO_3-N concentrations from 54.2 to < 0.01 mg/L, similar to values reported in previous studies in Colorado (Bauder et al. 2006; Paschke et al. 2008). I observed two patterns of NO_3-N concentrations in tailwater wetlands: 1) a peaked distribution with the highest concentration in mid- and late-July, and 2) a stable distribution in fallow fields indicating persistent NO_3-N concentrations. This second pattern could indicate a local residual effect of fertilization influences NO_3-N concentrations even in fallow years or the concentration in the regional aquifer, as water tables at these sites rose during the regional irrigation season while receiving no direct or indirect irrigation runoff.

Sites with mid-summer NO_3-N peaks corresponded to fine soil texture, flood irrigation and rapid runoff typical for pulsed nutrient release (Song et al. 2010). Constructed marsh wetlands can retain up to 41% of NO_3-N from episodic inflow events (Fink and Mitsch 2004). The efficacy of wetlands to remove nitrogen in pulsed hydrologic settings (Tanner et al. 2005) helps to explain lower NO_3-N levels at wells further into the wetland as uptake and denitrification are likely occurring along groundwater flow paths through the wetland. However, significant differences in denitrification were attributed to changes in C and NO_3-N availability and loading rates associated with pulse events (Song et al. 2010), suggesting the response could be site specific and vary at small scales.

Groundwater NO_3-N concentrations varied significantly between parallel transects suggesting that preferential subsurface and surface flow paths may influence N movement and denitrification rates (Casey et al. 2004; Kjellin et al. 2007). This could also be caused by spatial variability in microbial communities associated with heterogeneous carbon availability that can vary within 10 m even without differences in pH or soil moisture (Hill et al. 2000; Correa-Galeota et al. 2013; Ballantine et al. 2014). Other physical controls on denitrification of emergent wetlands including soil temperature, soil oxygen levels and dissolved oxygen in groundwater could also affect rates in space and time, creating locations or times of increased biochemical processing (McClain et al. 2003).

3.4.3 Plant Uptake and Transformation

N input to wetlands is from multiple sources, including N-enriched surface water and groundwater, N-fixation and atmospheric deposition. N is removed from the system primarily through denitrification, volatilization and plant uptake (if biomass is removed) (Woo and Zedler 2002; Bastviken et al. 2005). N is recycled through senescence and decomposition, mineralization, root storage and reuse (Woo and Zedler 2002; Kroger et al. 2007). The N concentration in aboveground biomass ranged from 1.4-3.2%; when coupled with biomass these values produced the significant differences in N content per unit area shown in (Figure 6). Harvest of aboveground biomass reduces the amount of N available for translocation to roots and reuse in subsequent growth, thus plants will need to uptake more N from the soil and groundwater.

Microbial biomass is a key indicator for potential NO_3-N removal in wetlands (Ballantine et al. 2014). If denitrifying microbes are present under anaerobic conditions in sufficient populations, they can respond to increased NO_3-N or C and significantly increase NO_3-N removal through denitrification (Jahangir et al. 2012). Most denitrifying bacteria require labile C for respiration, growth and reproduction (Beauchamp et al. 1989), thus the strong correlation between microbial biomass and soil organic matter content was anticipated. Organic matter content at my study sites was comparable to other natural wetlands yet higher than those reported for restored wetlands (Wolf et al. 2011A). Microbial biomass increased moving from agricultural field into the wetland area with the shallowest water tables. The potential for dissolved OC exports could extend the positive impact of wetlands denitrification into connected downstream surface waters or riparian areas (Hansen et al. 2016).

Denitrification is controlled by the presence of denitrifying bacteria, C content, oxygen levels and NO_3-N concentrations (Willems et al. 1997; Sirivehden and Gray 2006). Tailwater wetlands in this study had a range of values for these controlling factors and thus denitrification rates varied widely. The *in situ* acetylene block technique posed challenges related to diffusion efficacy of C_2H_2 into saturated soils, difficulty of maintaining airtight seals under harsh field conditions, and submerged portions of wetlands limiting the number of samples from lower positions. However, I achieved the goal of identifying the relative importance of denitrification vs. plant uptake on annual N cycling in irrigation dependent agricultural wetlands. The small number of successful experiments and the high variability in physical factors controlling denitrification that vary over space and time made statistically robust comparison difficult.

3.4.4 Management Implications

Hydrological and biological management of wetlands to enhance N storage or removal through transformation must balance the need for water and space with agricultural landscapes. Wetlands have been constructed to lower NO_3-N concentrations from agricultural and residential runoff and numerous studies conducted to understand how to optimize above and belowground sequestration and transformation (Kootatep and Polprasert 1997; Bachand and Horne 1999; Sirivehden and Gray 2006; Chavan et al. 2008; Wolf et al. 2011B).The addition of C as glucose and acetate to small scale bioreactors and restored wetlands enhanced microbial denitrification (Schipper et al. 2010; Ballantine et al. 2014; Bock et al. 2015) yet is not practical on a landscape scale. Similarly, the application of agricultural wastewater has been shown to add increase DOC in deeper soil layers creating denitrification hotspots (Jahangir et al. 2010, 2012). Periodically incorporating annual biomass into the wetland soil by mechanical means (hang et al 2016) is a potentially cost effective method to increase soil organic matter content and has been shown to be effective with *T. latifolia* (Ingersoll and Baker 1998). However, this method is limited because heavy equipment cannot operate easily in saturated sites during peak biomass and peak N content and may be most appropriate for seasonal wetlands and those with hydrological controls. Harvest of annual biomass has been used in commercial scale nutrient capture and biofuel production in Canada though not without challenges to harvesting during summer and fall (Grosshans et al. 2014). *Typha* ash has trace concentrations of NO_3-N (<0.01%), but it is unclear if this would be expected if burned *in situ*. Burning is a common practice for agricultural producers to control weeds, and nutrient removal could be an additional benefit of this cost-effective management action if timing and frequency are managed for several coincident goals.

The human health concern for consumption of drinking water with NO_3-N concentration exceeding 10 mg/L (Powlson et al. 2008). Municipalities can treat public drinking water to reduce the NO_3-N levels through blending with low NO_3-N water, reverse osmosis, ion exchange and electrodialysis, but these are economically or physically infeasible for domestic wells (Jensen et al. 2012). This endangers residents in rural areas and populations in developing countries that depend on domestic wells (Kivaisi 2001). Reducing NO_3-N levels in aquifers is often limited by microbial activity and available C sources (Starr and Gillham 1993), thus the focus should be to intercept and transform dissolved NO_3-N prior to deep percolation to limit further contamination. In rural areas with a dispersed population and significant agricultural land use, NO_3-N sequestration and transformation in wetlands is an attractive low cost option to limit further contamination of groundwater supplies (Shrimali and Singh 2001) by enhancing wetland ecosystem functions such as plant growth and nutrient transformation.

4. Potential Changes to Ecosystem Services of Irrigated Agriculture Under Three Water Transfer Scenarios

4.1 Introduction

The flow of water in many landscapes has been altered by human use at local, watershed and regional scales. Agriculture is the largest user of water in most regions of the world (Bennett 2000) especially semi-arid regions. Where water is diverted from rivers the interactions of rivers with their historic floodplain is reduced, limiting the spatial extent and functioning of many ecological processes (Nilsson and Svedmark 2002; Malmqvist 2002; Kuiper et al. 2014). Aquatic ecosystems are impacted by altered flow regimes that result in increased water temperature, limited natural flood disturbance, and altered sediment dynamics (Bunn and Arthington 2002; Ellery et al. 2003). Physical alterations to channels also impact floodplain connectivity and the condition and persistence of flood dependent riparian ecosystems (Richter et al. 1996; Poff and Zimmerman 2010). However, some agricultural practices, especially irrigation, may have environmental benefits that could partially offset impacts to the quantity and functioning of wetlands and riparian ecosystems caused by water extraction and land use change (Peck and Lovvorn 2001).

Wetlands and riparian areas in agricultural landscapes have long been recognized for protecting regional biodiversity, water quality and supply, and processing excess nutrients (Karr and Schlosser 1978; Mitsch and Gosselink 2000; Thiere et al. 2009). Irrigation networks create riparian areas along canals and incidental wetland ecosystems bordering storage reservoirs and agricultural fields through seepage from earthen canals and irrigation runoff (Lutton et al. 2010; Sueltenfuss et al. 2013). Incidental wetlands have the potential to mitigate environmental degradation associated with fertilizer runoff, pesticide use, habitat loss, and altered food webs.

For example, irrigation supported wetlands in the Ebro Basin in Spain occupy only 1.4% of land area but remove up to 50% of total nitrogen in agricultural runoff (Moreno et al. 2007).

Incidental wetlands in agricultural regions also provide wildlife habitat in fragmented landscapes (Uden et al. 2014). For example, songbirds use many canal corridors during the breeding season (Lopez-Pomares et al. 2015), and can occur in intensively irrigated landscapes in the western U.S. (Peck and Lovvorn 2001, Peck et al. 2005; Fernald et al. 2007), Spain (Lopez-Pomares et al. 2015) France (Aspe and Jacque 2015), and other countries. In some regions, irrigation supported riparian and wetland ecosystems can be more abundant than natural riparian and wetland ecosystems and have novel hydrogeomorphic and biological elements such as open water habitats where none previously existed. Irrigation dependent ecosystems also have created new spatial arrangements of riparian and wetland ecosystems. Historically all riparian and most wetland ecosystems occurred in valley bottoms, but now the landscape position has flip-flopped with many irrigation dependent wetlands in upland areas. These anthropogenic features may replace some ecological functions lost from natural riparian and wetland ecosystems (Roberts and Rahel 2011; Ferreira and Beja 2013). In regions where agriculture and suburban land use are integrated, the social benefits from recreation and aesthetics also occur (Brander et al. 2013; Aspe and Jacque 2015).

Irrigation canals and incidental wetlands do not replace natural streams and wetlands (DiNatale et al. 2008), but are additional features on the landscape that resemble natural streams and wetlands yet occupy formerly terrestrial areas. Thus, as natural riparian and wetland areas have been degraded by decades of water extraction and land use change (Strange et al. 1999), these new ecosystems could increase the total area of wetlands and restore or enhance some regional ecological functions. These include wildlife habitat (Lopez-Pomares et al. 2015), nutrient cycling (Moreno et al. 2007), and trophic interactions (Peck and Lovvorn 2001).

The environmental benefits of irrigated agriculture are also at risk from changes in climate, local economies, land management, and government policies. It has been predicted that climate change will alter water supplies in many parts of the world (Vorosmarty et al. 2000), including reductions in annual snowpack, strengthening of seasonal rainfall patterns with more pronounced wet and dry periods, and increases in evapotranspiration rates any of which can influence water availability for irrigation (Barnett et al. 2005; Barnett et al. 2008; Ray et al. 2008; Gao et al. 2011). Changes in the extent or type of irrigation could dramatically impact the location, type, and permanence of these ecosystems. Physical and vegetation maintenance activities such as burning, herbicide application, and sediment removal could limit their ecological functions, creating a variety of vegetation types (Figure 4.1).

Population growth is also likely to cause widespread changes in the area and location of irrigated agriculture as water is transferred to meet increasing municipal water demand (Qadir et al. 2003; Rosenzweig et al. 2004; Molle and Burkoff 2009). The Colorado Front Range has experienced significant increases in municipal water demand and water currently used for irrigated agriculture is viewed as an attractive option to meet this growing municipal demand (CWCB 2015, SPBIP 2015). Changes in irrigated land area, crop type, and efficiencies are likely to affect irrigation supported riparian and wetland ecosystems that sustain local and migratory wildlife, nutrient processes, and surface and groundwater quality functions (Peck et al. 2005; Chapter 2, 3). In some instances, reductions in water diverted for irrigation or efficiency improvements can improve natural instream habitat (Peck et al. 2005), but this potential tradeoff should be fully investigated. Predicted changes in water supply and demand create a complex set of economic, social, engineering and environmental problems with distinct and hidden consequences (Howe and Goemans 2003; SPBIP 2015).

Change in water allocation between sectors (e.g. agriculture to municipal) is likely along the Colorado Front Range to meet the expected municipal water supply gap (CWCB 2015). In this paper, I develop three water transfer scenarios designed as end points on a spectrum of economic, resource conservation and ecological integrity to identify the type and spatial arrangement of impacts to ecosystem services compared to present. I use information from two regional studies on irrigation canal riparian ecosystems (Chapter 2) and nitrogen dynamics in agricultural wetlands (Chapter 3) to assess the risk to these ecosystem services under potential water transfer scenarios in a highly irrigated area. I use spatially explicit data from present and three future water transfer scenarios to ask the following questions: 1) how is canal flow reduced, 2) what type of wetlands are impacted and where are they located, 3) how does riparian habitat change, 4) how are potential nitrogen mitigating wetlands affected and 5) how do ecosystem services change? The scenarios are meant to highlight the importance of water resource planning at a regional scale to account for indirect effects to the landscape.

4.2 Methods

4.2.1 Study Area

The study area (Figure 4.2) was in Weld County in northcentral Colorado. Land use was assessed using the National Land Cover Dataset (Homer et al. 2015). This data set indicated that 81.2% was in agriculture with 74% of agricultural land irrigated at least in part by surface water. Residential land use accounts for only 1.6% of the study area. Surface water is delivered to irrigated agriculture through a network of canals and reservoirs.

The irrigation method has shifted from 100% flood irrigation in 1956 to 64% flood and 34% sprinkler in 2010 (http://cdss.state.co.us/GIS/Pages/Division1SouthPlatte.aspx). Dominant crops include corn (47%), grass pasture (25%), and alfalfa (22%). The area includes a range of wetlands generally described as lakes, ponds, riverine, marshes and wet meadows totaling 4,312 ha. Emergent wetlands, defined by the National Wetlands Inventory as dominated by "erect, rooted, herbaceous hydrophytes" was particularly common at 29% of total along with open water lakes and ponds at 42% (Cowardin et al. 1979). These were further classified by the duration of saturated soil conditions that influences anaerobic soil characteristics and plant species composition as semi-permanently flooded (PEMF), seasonally flooded (PEMC), and temporarily flooded (PEMA) (Cowardin et al. 1979). Emergent wetlands in the study were dominated by several clonal species including *Typha latifolia*, *T. angustifolia* L., *S. pungens*, *P. arundinacea*, several species of *Juncus*, *Carex* and *Persicaria* Miller spp. distinguished on aerial photographs by differences in color and texture (Figure 4.3).

4.2.2 Spatially Explicit Water Transfer Scenarios

Irrigated agriculture is predicted to be reduced 33% (~18,700 ha) in the study area to meet the projected new water supply gap in 2050 (CWCB 2015). To model the impact of dry up on Individual farm parcels ranging from 1-40 ha were dried (dry-up parcels) in three spatial arrangements that represent single goals that represent: 1) maximized water conservation, 2) likely residential/urban development, and 3) maximized ecosystem services. I attempted to incorporate a complex set of values and decisions by individuals, communities, governments and businesses to develop the three water transfer scenarios described below:

<u>Prioritizing Water Conservation (WaterCon)</u>: Water is lost along irrigation canals through seepage and evapotranspiration. Thus, parcels at the furthest end of the canal network will have

the highest evapotranspiration and seepage losses. To maximize water conservation, parcels at the distal ends of irrigation service areas are prioritized for dry up and water transfers to M&I use, creating additional savings from transmission losses. The concentrates dry-up parcels in the eastern portion of the study area.

Residential and Urban Development (Res/Urban): The human population of the Colorado Front Range is rapidly increasing and small communities are experiencing residential and commercial growth. Agricultural land is converted and the water rights are likely to be converted to higher value municipal use. These land conversions were modeled to occur within 2 km of existing incorporated and unincorporated town and city boundaries to utilize existing sewer, power and transportation infrastructure.

Ecosystem Service Retention (EcoServ): Ecosystem services of wetland area, riparian habitat quality along irrigation canals and N-mitigation potential of agricultural wetlands adjacent to irrigated farm parcels were assessed using the public land survey system parcels (~2.5 km^2). The maintenance of current (2015) ecosystem services was prioritized and dry-up parcels were selected from areas with few local ecosystems services and the least cascading impacts along the irrigation canal network.

I made two assumptions for these scenarios: 1) water use is proportional to parcel area and equal across the study area, crop type and irrigation type; and 2) return flows and secondary uses are not inhibited by water transfers. The latter addresses a clause in Colorado water law stating that no change in water use or transfer or all or part of a water right will reduce water use by downstream users (Hutchins et al. 1972; MacDonnell et al. 1990; Hobbs 2004). This accounts for return flows not consumptively used by a crop, residence or industry and is subsequently used further downstream. Proving no injury adds significant transaction costs to water right transfer and lease agreements and is varies by location and complexity of surface and subsurface hydrology (Howe et al. 1990, Howe and Goemans 2003.)

4.2.3 Tracing Flow Reduction

Flow is commonly described as a volume of water per unit time (m^3/s or acre feet/year), however data on flow in irrigation canals was not available at the spatial resolution necessary for this study. Therefore, I used the concept of downstream service areas to assign flow reduction values. The percent flow reductions describe approximate reductions in flow, not the flow rate, total annual flow, or any measure of volume, as current flow rates, transmission losses and on farm use are not known. I used the total irrigated area serviced by a canal and reduced flow by the percentage of the service area dried up under a given scenario. Thus, if a canal provides water to 100 ha of irrigated land and 25 ha are selected for dry-up, then 75% of flow would remain in the canal.

USGS 1:24,000 topographic maps and aerial imagery were used in ArcGIS 10.4 (ESRI 2011) and Google Earth to identify water routing at the parcel scale. Percent flow reduction was traced through the irrigation canal network for each scenario. Figure 4.4 provides a hypothetical example of how this analysis was performed and how flow reduction was calculated at canal intersections.

4.2.4 Riparian Habitat Quality Index Scores

Canal segments were assigned a Habitat Quality Index (HQI) score developed from riparian surveys conducted in the study area in 2014 using the approach of del Tanago and de Jalon (2011) with methods and results described in detail in Chapter 2. With relevant statistical values presented in Table 4.1. The mean and standard deviation (when $n > 4$) for canals were calculated for each pairing of cover and land use. For groups with < 4 samples, I used the average standard deviation for channels with residential land use (0.73) and agricultural land use (0.60), regardless of cover type. HQI scores were assigned to canal segments using Eq. 1 below which used a random number *y* (between -10 and 10) as a multiplier for the standard deviation (*stdev*) to add to the mean.

$$HQI = \bar{x} + ((stdev/10) * y) \quad \text{Eq. 1}$$

Where:

$\bar{x}$ = mean HQI score for a canal cover type (e.g. Shrub, Herb)

stdev = standard deviation of the scores for a canal cover type

y = a random number between -10 and 10

A length weighted HQI score using Eq. 3 was calculated for the study area and public land survey system (PLSS) sections (~2.5 km^2) for assessing ecosystem services of riparian habitat. The length weighting follows the prevalence index used for wetland identification and condition assessments (Wentworth et al. 1988). Categories were scaled from a larger study from Chapter 2 as very poor (0-1.5), poor (1.5-2.5), fair (2.5-3.5), good (3.5-4.5, very good (4.5-5.5), excellent (5.5+).

A modeled HQI (HQI_m) was calculated for each canal segment under the three scenarios using Eq. 3. This assumes a linear relationship between proportional flow in a canal and the HQI score. A flow reduction of 20% was a conservative estimate for the normal range of variability and not expected to impact the riparian vegetation, thus the linear relationship begins at flow reductions greater than 20%.

$$HQI_M = HQI_I - (HQI_I * (|20 - FR| * 0.125)) \quad \text{Eq. 2}$$

$$HQI_L = \left(\frac{L_{VP} + 2L_P + 3L_F + 4L_G + 5L_{VG} + 6L_E}{TL}\right) \quad \text{Eq. 3}$$

Where:

L_x = length of canal in each HQI category (Very Poor – Excellent)

TL = total length of canals

HQI_M = modeled HQI score

HQI_I = initial HQI score

FR = flow reduction in percent

4.2.5 Nitrogen Mitigation by Agricultural Wetlands

Nitrogen (N) uptake and transformation were calculated for plant communities that were observed to correspond to three NWI wetland types PEMA, PEMC and PEMF (Chapter 3). PEMA wetlands are temporarily flooded at least two weeks a year but are the driest wetland type. PEMC are seasonally flooded, and PEMF are semi-permanently flooded and indicate marsh wetlands. N uptake was modeled as the N present in aboveground biomass produced in one year that could be removed through mechanical harvest or burning. Nitrate transformation to N via microbial denitrification is more difficult to estimate at the landscape scale due to high spatial and temporal variability within a wetland (McClain et al. 2003; Poe et al. 2003; Bruland et al. 2006; Chapter 3). Thus, N uptake and transformation rates (Table 4.2) calculated for wetland area within 15 m of irrigated parcels are estimates.

4.2.6 Wetland Area

Palustrine wetlands including palustrine emergent (PEM), palustrine woody (PSS, PFO) as well as ponds with or without aquatic plants (PAB, PUB, PUS) were selected from all wetlands present to assess the aerial extent of wetlands likely to be directly and indirectly impacted by water transfers. Lakes were excluded as these were unlikely to be indirectly impacted by changes in water allocation, although their use in the irrigation network makes them susceptible to infrastructure changes resulting from or in support of water transfers.

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4.2.7 *Combined Ecosystem Service Scores*

Three ecosystem services provided by irrigated agriculture were assessed using scores and values summed or averaged by public land survey system sections (section) (~2.5 km^2). The scores for N-mitigation and total wetland area were scaled 0-100 using maximum values in the study area described in Eq. 4 and 5. Each ecosystem service is treated equally with a possible maximum combined score of 300. Riparian Habitat Quality Index (HQI_M) was used as a quantitative value representing the community composition, structural components, and pressure of adjacent land use for riparian ecosystems. The possible score of 100 was split evenly between the scaled HQI_L term (Eq. 6) and HQI_{Ib} that incorporates total canal length (Eq. 7).

$$N_I = N_s * M_N \qquad \text{Eq. 4}$$

$$W_t = W_s * M_W \qquad \text{Eq. 5}$$

$$HQI_{Ia} = \left(\frac{L_{VP} + 2L_P + 3L_F + 4L_G + 5L_{VG} + 6L_E}{TL}\right) * (M_{Ha} * 0.5) \qquad \text{Eq. 6}$$

$$HQI_{Ib} = \left(\frac{L_{VP} + 2L_P + 3L_F + 4L_G + 5L_{VG} + 6L_E}{2 * M_{Hb}}\right) \qquad \text{Eq. 7}$$

$$ES = N_t + W_t + HQI_{Ia} + HQI_{Ib} \qquad \text{Eq. 8}$$

Where:

N_I = Total N uptake and transformation in kg ha^{-1} yr^{-1}

M_n = multiplier for N-mitigation term (3.3 for the study area)

M_{Hl} = multiplier for HQI_L score (3 for the study area)

M_{Ht} = multiplier for HQI_T score (160 for the study area)

M_w = multiplier for wetland area term (3 for the study)

WA = wetland area in hectares

4.3 Results

Vegetation along the canals was dominated by herbaceous plants (71.8%) or the canal was concrete lined (26.5%), and less than 2% of the vegetation cover was from woody plants. The three water transfer scenarios resulted in the drying of an average 18,724 (+/- 27) ha of irrigated agricultural land, or 33% of the total surface water irrigated agricultural land. The spatial distribution (Figure 4.5) of dried parcels was related to the central goal of each scenario. The WaterCon scenario dried up 18,711 ha to reduce or eliminate consumptive water use at the field and water lost through transmission along canals (Figure 4.5B). The Res/Urban scenario created a patchy distribution of 18,761 ha of dry-up area (Figure 4.5C). The EcoServ scenario dried up 18,699 ha in areas with low ecosystem service scores and with low network effects, also created a patchy distribution (Figure 4.5D). Overlap between all scenarios was 2,682 ha, only 14% of projected dry-up. Res/Urban and EcoServ shared 41% of dry-up parcels (7,774 ha), Res/Urban and WaterCon shared 29% (5,508 ha) and WaterCon and EcoServ shared 25% (4,419 ha).

4.3.1 Flow Reduction

The total length of canals projected to have flow reductions from drying agricultural parcels was lowest for the WaterCon scenario. The Res/Urban scenario had the greatest total length of canals with reduced flow and greatest length of 100% flow reduction canal. The spatial patterns of flow reduction (Figure 4.6) varied by the arrangement of agricultural parcels selected for drying (Figure 4.4). Impacts for the WaterCon scenario were most concentrated on the distant ends of the network with diluted effects along the main stem canals. Res/Urban impacts were scattered around 13 incorporated and unincorporated towns. EcoServ focused agricultural dry up in areas with low HQI_M, low wetland area, and low wetland N-mitigation potential and created a patchy distribution of impacted canals. Res/Urban and EcoServ impacted different canal sections, largely not affecting northeast and east areas (Figure 4.7).

4.3.2 Canal Riparian Ecosystem Change

Riparian ecosystem HQI_M scores had a net loss in canal length for all scenarios in Good and Fair categories and a net increase in Very Poor scoring canals (Figure 4.8). The current HQI length index (HQI_L) of the canal network was 2.18 and provided a baseline to assess changes in habitat quality in the canal network. When the full 1,778 km canal network was assessed, all scenarios scored lower than the current condition, with Res/Urban at 1.73, WaterCon at 1.83 and EcoServ at 1.87. When channels with projected 100% flow reduction were excluded, the HQI_L scores were closer to current conditions, Res/Urban = 2.02 for 1,281 km of canal, WaterCon = 2.02 for 1,460 km, and EcoServ = 2.13 for 1,372 km. Canal sections with woody canopy had the highest HQI yet were extremely rare (< 2%). Total length of impacted canals with woody canopy was lowest for the WaterCon scenario (4.2 km), however the woody canopy canal length between 20-100% flow reduction was equal to the EcoServ scenario at 2.8 km.

4.3.3 Wetland Area Change

Wetlands were expected to be impacted under all scenarios. The WaterCon scenario impacted the most wetland area (632 ha) similar to the Res/Urban scenario (607 ha). The number of wetlands was also very similar, 665 for WaterCon and 659 for Res/Urban. The EcoServ scenario had the fewest (433) and least area (394 ha) of wetlands impacted. Only 48.6 ha of wetlands were impacted by both WaterCon and Res/Urban scenarios or about 8% of impacted wetlands with a similar overlap between WaterCon and EcoServ. Res/Urban and EcoServ impacted more of the same wetlands at 28%.

4.3.4 N Mitigation Potential Change

Changes to nitrate mitigation from agricultural runoff through transformation (via microbial denitrification) and plant uptake varied by scenario and were linked to wetland type and area. EcoServ had the lowest total N-mitigation loss, about 45% lower than WaterCon and Res/Urban which were similar (Figure 4.9). Plant uptake had the largest impact on N-mitigation accounting for more than 95% of potential annual N-mitigation in all scenarios.

4.3.5 Change in combined environmental services

Irrigated acreage was reduced by 33% in each water transfer scenario yet the overall reduction and distribution of ecosystem service scores varied (Fig. 4.10). Current conditions had a mean ecosystem service value of 66.1 (+/- 22.9), and a total ecosystem service value of 24,070. The WaterCon scenario lost 23.5%, with a mean of 50.6 (+/- 34.2), an additional 11,396 ha of zero ecosystem services, and a total score of 18,408. The EcoServ scenario had no additional zero score area but a lower mean (46.1, +/- 15.4) and total score (16,776) for a 30.3% reduction of ecosystem services. The lowest ecosystem scores were measured on the Res/Urban scenario with a mean of 23.8 (+/- 22.6), an additional 6,216 ha of zero score area, a total score of 8,644 for a reduction of 64.1% of ecosystem services.

WaterCon and Res/Urban scenarios had less than a 20% reduction in N-mitigation services but wetland area scores were reduced for all scenarios by over 40%. Length and riparian HQI scores were maximized in the EcoServ scenario with only a 14.5% reduction in scores across the study area, however, wetland area scores were reduced 85.5% compared to current conditions.

4.4 Discussion

4.4.1 Flow reduction

Water right, irrigation method, crop type, topography and soil characteristics influence the structure, density and distribution of irrigation canals. The transfer of agricultural water to municipal use will affect the frequency, duration, and magnitude of flows in the canals. While consumptive use is the only transferable water of a water right, variable seepage, evaporation, surface and groundwater returns create a complex "leaky" network susceptible to unpredicted impacts. Landscape and land use variability produced distinct spatial patterns of modeled impacts from the three water transfer scenarios. For example, the Res/Urban scenario impacted the most canal length due to the higher density of canals in the dry-up parcels near towns while the WaterCon scenario impacted the least because water was supplied to many sprinkler-irrigated fields by pipelines (not included) instead of open irrigation canals, which may have high rates of leakage.

The intensity of flow reduction decreased moving from peripheral canals to main stem canals and towards the point of diversion. This created pockets of severe flow reduction or complete elimination of flow (red lines in Fig. 4.7) with less impact to main stem canals as flow was retained to support irrigation in other areas. Many main stem canals might have <20% flow reduction, our conservative estimate of annual flow variation, while smaller canals are more frequently >20% and could experience significant impacts. These impacts could include loss of sensitive aquatic taxa such as *Ephemerella* mayflies (Garcia-Roger et al. 2011; Chessman 2015), physiological stress and mortality of riparian vegetation (Smith et al. 1991; Stromberg and Patten 1996) and shifts in riparian vegetation composition (Merritt and Cooper 2000; Shafroth et al. 2002; Scott et al. 2010). While these impacts have not been studied in irrigation canals, we showed plant riparian communities and flow regimes of canals in this region are comparable to streams (Chapter 2) and similar responses are expected.

4.4.2 Riparian ecosystem change

Canal riparian ecosystems in semi-arid agricultural regions can provide habitat refuges and wildlife corridors in highly developed agricultural landscapes (DiNatale et al. 2008). For example, many bird species nest in vegetation along irrigation canals (Lopez-Pomares et al. 2015) and threatened mammals use the vegetation as replacement for habitat loss (Meaney et al. 2003). It is difficult to determine riparian habitat loss from land use change and water extraction but historical records describe significant morphological and ecological changes on streams in the study area (Strange et al. 1999). In this study, I quantify the amount and attempt to evaluate the quality of incidentally created riparian habitat along irrigation canals. In a coincident study (Chapter 2), canal riparian ecosystems and streams were found to have largely similar riparian biota that could be attributed to the degradation of streams and the colonization and survival of species tolerant of highly variable flow and frequent and intense disturbance. Thus, the riparian ecosystems along the 1,778 km of canals provide similar habitat value as remnant riparian areas, across a range of habitat quality.

The ecological consequences of flow reduction on riparian ecosystems including impaired plant recruitment and seedling establishment with lower peak flows (Williams and Cooper 2005) as well as lower plant cover, more terrestrial species, and increased plant mortality from changes in flood frequency and duration (Merritt and Cooper 2000; Poff and Zimmerman 2010). These impacts have been primarily studied on floodplains, yet canal banks often support similar vegetation (Chapter 2).

Field observations indicated that canals with less frequent or mild maintenance activities supported more diverse vegetation including woody species (Chapter 2), yet studies on the impacts of flow reduction or vegetation management on riparian flora of irrigation canals are lacking. Moving beyond biological inventories (*sensu* Habit et al. 1998; Lopez-Pomares et al. 2015; Chapter 2), these areas could be suitable for manipulative experiments to improve our understanding of establishment and survival thresholds of plant species, as well as the impacts to aquatic physiochemical conditions and biological communities (Koetsier and McCauley 2015.

4.4.3 Wetland Area change

Wetlands in many agricultural areas are created by canal seepage (Sueltenfuss et al. 2013), irrigation runoff from fields (Peck et al. 2005), and elevated water tables from regional irrigation (Chapter 3). It is likely that wetland area has increased in the study area due to irrigated agriculture, and the type and distribution of wetlands is different from the historically dominant floodplain wetlands (Sueltenfuss et al. 2013). The impacts to irrigation dependent wetlands from complete or partial flow reduction in canals and changes to field scale irrigation are not fully understood. Canal seepage, surface and sub-surface hydrologic inputs to incidental wetlands would likely be reduced and could lead to less wetland area, changes in plant communities, and loss of connectivity (Sueltenfuss et al. 2013). The location of wetland impacts varied by scenario, with the WaterCon scenario reducing wetland area in the eastern portion of the study area and the Res/Urban scenario impacting the western portion while the number and impacted area was similar between the two scenarios. The EcoServ scenario was developed to avoid impacting wetlands and succeeded by preserving over 200 ha more than the other scenarios, with impacts dispersed across the landscape. This could help to maintain connectivity between wetlands, an important quality in agricultural landscapes (Uden et al. 2014).

4.4.4 *N-mitigation change*

The ecological and economic value of wetlands in agricultural landscapes includes the cycling and transformation of nutrients and chemicals (Brander et al. 2013). Removal of agriculture contaminants in water can be a costly component of treating water for human consumption (Jensen et al. 2012). Wetlands have been constructed as a filtration and bioprocessing component of water treatment facilities (Kivaisi 2001; Liehr and Kruzic 2007; Vymazal 2014). Current wetlands comprise 4.3% of the study area, within the 3-5% range proposed as effective in removing N compounds from agricultural wastewater (Hammer 1992; Mitsch et al. 2001. However, these studies are focused on surface water contamination (Kovacic et al. 2000, Budd et al. 2009) where runoff water is concentrated in topographic depressions and drainages where ecological processes such as microbial denitrification and plant uptake occur. In contrast, the study area has shallow groundwater that can be contaminated by infiltration of irrigation water that passes through agricultural soils, thus the distribution of wetlands across the landscape and the proximity to irrigated agricultural parcels is critical for N-mitigation. The nitrate concentrations in shallow groundwater have been shown to decrease below the EPA drinking water standard within 10-20 m of the wetland/ag field boundary (Chapter 3).

N-mitigation varies over small distances and by plant community making the modeling of changes to this ecosystem service particularly challenging. Changes towards drier wetlands could dramatically reduce potential N-mitigation. Plant uptake was 95% of the total N-mitigation potential, with the highest biomass and thus N uptake occurring in wetter sites. To remove N, plant biomass must be removed through harvest (Grosshans et al. 2014) or burning. The latter is a common practice to control weeds and could enhance the productivity and biodiversity of wetlands (Hopple and Craft 2013). Farmers participating in this study were hesitant to the idea of using machinery in soft wetland soils and may be unlikely to mechanically harvest plant biomass.

Incidental wetlands adjacent to fertilized agricultural fields intercept surface and shallow groundwater flow and are critical to N-mitigation potential of wetlands (Blackwell and Pilgrim 2011). If wetland area shrinks, the field-wetland adjacency might me lost and N-rich water could infiltrate into the contaminated aquifer without passing through the wetland. It could be argued that a reduction in irrigated agriculture would also reduce N-inputs. However, the long-term effects of decades of manufactured and manure fertilizers could maintain groundwater with elevated N-levels for an unknown period (Bauder et al. 2006). In addition, areas with locally shallow groundwater can create wetlands that remove N from aquifer water. Studies using detailed groundwater flow maps and stable isotope tracers could identify these areas for focused protection and enhancement.

4.4.5 Combined ecosystem services

The ecological services and resources supported by irrigated agricultural are at risk from changes to the distribution, amount, and timing of irrigation activities, field scale efficiencies, and infrastructure condition. The three components of the ecosystem service score: canal riparian ecosystems, wetland area, and N-mitigation were modeled to decrease under all water transfer scenarios. Spatial arrangement had a large effect on ecosystem scores. For example, the Res/Urban scenario had the highest combined ecosystem service scores likely because many of the areas selected for drying contained low scoring riparian habitat and few wetlands.

Some researchers have attempted to quantify the economic benefits of ecosystem services provided by agricultural landscapes (see Brander et al. 2013, Crossman et al. 2010). I did not attempt to identify monetary values for riparian habitat, wetland area and N-mitigation as the economics of pollution, water treatment, wildlife habitat and land for development are rapidly changing and were likely to mis-represent future conditions. Ecosystem service calculations including economic value are often exclude small features that could cumulatively provide significant benefits (Blackwell and Pilgrim 2011). For instance, Brander et al. (2013) limited analyses to wetlands >5 km^2, which would exclude all wetlands from the current study, even though wetlands were calculated to transform or absorb over 120,000 kg of N. Thus, the distribution and arrangement of wetlands of all sizes is critical for understanding the potential changes from water transfers. The advantage to using area based ecosystem service scores is the production of a prioritization map for water managers that identifies areas where effective and the least environmentally damaging water transfers from agricultural to municipal use could occur.

The advantage of a simplistic ecosystem service calculation (Eq. 2) is the flexibility to scale to the range of resources, the ability to add terms, and apply coefficients to terms to adjust the weight of a service. In this study, wetland area was inherently weighted higher as it was used as an individual component and to calculate N-mitigation estimates.

4.5 Conclusion

Redistribution of agricultural water to municipal uses will affect the riparian vegetation that borders canals and wetland ecosystems that develop from canal seepage, irrigation runoff and elevated regional groundwater. Water transfers prioritizing water conservation and ecosystem service retention had the lowest impacts on riparian and wetland ecosystems in a semi-arid agricultural region. Conversion of irrigated land to residential and urban land uses is also a likely scenario as human communities expand with the greatest impact to incidental habitats and canal length. Incidental wetlands and riparian areas supported by irrigated agriculture provides ecosystem services in agricultural landscapes. These impacts should be incorporated in individual and regional water planning to account for potential changes to regional environmental resources and ecological processes.

5. Synthesis

The direct and indirect impacts of human land use and activities on natural ecosystems are widespread (Vitousek et al. 1997; Foley et al. 2005). Utilization of freshwater resources for generating energy, irrigating crops, and sustaining human populations has negatively impacted streams, lakes and wetlands across the world (Carpenter et al. 1998; Graf 1999; Allan 2004; Hatfield 2015). In this study, I contributed to the global literature on ecological services created by water infrastructure (riparian and aquatic ecosystems) and use (nitrate (NO_3-N) mitigation of irrigation dependent wetlands). The prevalence of canals in agricultural areas and the immense amount of water used for agriculture have created a new stream system in parts of the western U.S. Canal networks total thousands of kilometers bordered by riparian vegetation and with aquatic habitat. The hydrologic conditions and disturbances of these anthropogenic ecosystems are completely controlled by humans yet the species and functional composition of some canals resemble natural streams. The addition of these ecosystems to the landscape, in unnatural positions and orientations fundamentally changes the hydrodynamics of a landscape.

Wetlands created by irrigation runoff and regionally elevated water tables from extensive water activities were found to intercept, transform and trap excess NO_3-N through plant uptake at rates similar to natural and constructed wetlands. N transformation was less effective than pant uptake, but occurred without additional material or labor inputs. My results identified the benefits of two ecosystem services created by irrigated agriculture: creation of riparian and aquatic habitat and treatment of agricultural runoff. This work aligns with previous studies on ecosystem services in agricultural landscape (Zedler 2003; Moonen and Barberi 2008; Crossman et al. 2010; Brander et al. 2013) while expanding to include completely anthropogenic landscape features. Risks associated with the fundamental connection to human hydrologic modification can be modeled with the findings useful for regional water planning.

Key findings from this research identify beneficial elements of an irrigated agro-ecosystem. Chapter 2 described the similarity of irrigation canal riparian vegetation and aquatic macroinvertebrate communities to natural stream ecosystems. These anthropogenic ecosystems create significant ecological resources due to the prevalence of canals in agricultural areas. Temporary aquatic habitats in canals supported diverse macroinvertebrate communities with similar taxa and functional composition, likely from colonization through drift of eggs or larvae (Koetsier and McCauley 2015). Riparian vegetation bordering irrigation canals had similar species and functional composition to natural streams. Canal riparian ecosystems quality was high in residential/urban areas where other studies have noted wildlife and social values of canals (Aspe and Jacque 2015; Lopez-Pomares et al. 2015).

In Chapter 3 I concluded plant uptake outperformed microbial denitrification in NO_3-N mitigation in wetlands created and maintained by irrigation. Physical, hydrologic and biotic conditions of irrigation dependent wetlands to vary in space and time in agreement with many other studies (Hill et al. 2000; Kjellin et al. 2007; Correa-Galeota et al. 2013; Ballantine et al. 2014). Hydrodynamics of shallow wetland water tables were influenced by pond levels, irrigation runoff, precipitation, and regional groundwater. NO_3-N concentrations had two temporal patterns: 1) a mid-summer peak, likely associated with high irrigation rates and 2) stable background concentrations.

Rates of N transformation were highly variable within and between wetlands, but mean values were comparable to those reported for riparian buffers (Hanson et al. 1994) and some wetlands (Borin and Tocchetto 2007). Rapid changes to common limiting factors for denitrification including NO_3-N concentration, bioavailable carbon, favorable anoxic conditions and sufficient microbial activity lead to "hot spot" and "hot moment" characteristics of denitrification (McClain et al. 2003). These factors can be identified and potentially altered in irrigation dependent wetlands to improve denitrification of NO_3-N enriched runoff and groundwater. N transformation processed low amounts of N compared to plant uptake, yet is a passive process that occurs without human involvement. To eliminate N cycling within a wetland from aboveground plant biomass through litter decomposition, biomass must be removed through harvest or burning (Grosshans et al. 2015). Fire is utilized by agricultural producers to reduce weed populations and seed dispersal from adjacent non-cultivated areas, and could be incorporated into land management plans to enhance N removal.

Current and future spatial distribution of anthropogenic ecosystems and N-mitigation potential was modeled in Chapter 4 to understand the effects of different approaches to converting water from agricultural to municipal water use by drying irrigated land. Taken together and included in regional water planning, these conclusions about the amount and location of ecosystem services in a changing agricultural and urban landscape could help avoid unintended environmental degradation in the future.

Assessment of ecosystem services is a growing field, especially in agricultural regions (Otte et al. 2007; Thiere et al. 2009; Crossman et al. 2010; Blackwell and Pilgrim 2011). I used current infrastructure and irrigated area as a baseline to assess the changes to ecosystem services described in Chapters 2 and 3 for three water transfer scenarios. Water use in Colorado is expected to change sectors (from ag. to municipal) to support a growing population with the study area expected to lose 33% of irrigated acreage to fill the predicted gap (CWCB 2015; SPBIP 2015). The amount and location of irrigation water transfers to municipal use will affect the amount and location of ecosystem services of irrigation dependent ecosystems (DiNatale et al. 2008). The spatial distribution of ecosystem services and interconnectedness of canals, ponds and reservoirs make predictions about the location and extent of changes to water distribution difficult. For example, I found that drying 33% of irrigated land would result in flows being reduced in 37-53% of canals, depending on the spatial arrangement of dry-up areas.

Incorporating policy, river basin plans, public education into regional water development plans is necessary to account for incidental, often non-monetized, ecosystems services (e.g. Crossman et al. 2010; Downard and Endter-Wada 2013; Kovacs et al. 2016). The study area in Colorado, U.S.A. is an example of semi-arid agricultural landscapes across the world including Chile, Spain, France, Greece, Australia, India and China (Habit et al. 1998; Roy and Shah 2002; Foster and Chilton 2003; Cai and Rosegrant 2004; Baral et al. 2014; Aspe and Jacque 2015; Stefanidis et al. 2016). I found aquatic habitats of canals with similar composition of taxa and functional groups as natural streams yet riparian vegetation showed a mixed pattern of differences in plant species but some similarity of functional groups. Future research on the potential to manage canal maintenance to develop higher quality riparian and aquatic habitat on canals without impacting water conveyance would be an exciting continuation of this work. The potential for wetlands to mitigate excess NO_3-N from agricultural runoff by transformation and uptake needs further testing in more locations to identify patterns associated with soil types, fertilizer applications, and regional groundwater movement. Economic, environmental and social drivers of water use in the region may not prioritize maintaining the current distribution of ecosystem services, however it is important to understand the changes so that decisions can be made with the most complete information.

6. Tables and Figures

Table 2.1 – Hydrologic metrics used to characterize flow regimes of canals and streams calculated from April 1 to Sept 30. CV = coefficient of variation, R-B Flashiness index, date of peak flow (Julian days), number of high flow events (flows exceeding 90%), number of low flow events (flows less than 10%), number of days below 10%, and number of days with zero flow. Detailed definitions are in Appendix 8.5. [1] located in Larimer County, CO; [2] located in Weld County, CO.

Metric	*Cache la Poudre River (USGS gage 6752260)*[1]	*Cache la Poudre River (USGS gage 6752500)*[2]	*Spring Creek*[1]	*Box Elder Creek*[1]	*Large Canal*[1]	*Dry Creek*[1]	*Small Canal 1*[1]	*Small Canal 2*[1]
CV	1.77	1.6	0.89	1.2	1	0.19	0.81	0.79
R-B Index (Flashiness)	0.24	0.2	0.35	0.22	0.22	0.07	0.11	0.11
Date of Peak	150	155	186	211	174	148	135	128
# High Flow Events	1.6	1.3	5	1.6	3	4.6	1.6	2.3
# Low Flow Events	4.3	3.3	2	2	3.7	1.3	3	3.3
# Low Flow Days	18	18	17.6	17.3	30	20	13.3	34
Zero flow days	0	0	0	0	12.3	0.3	2.6	10.3

Table 2.2: Summary of values for diversity metrics and indices of wetland prevalence, conservative species cover, native richness and percent cover. ANOVA results (F-statistic and p-value) reported for each metric.

	Species Richness	*Native Richness*	*Shannon Diversity*	*Simpson's Index (1-λ)*	*Wetland PI*	*Cover Weighted Mean C*	*Native Cover %*
F-statistic	*0.38*	*0.94*	*0.34*	*0.49*	*0.29*	*1.12*	*0.85*
(p-value)	*(0.77)*	*(0.43)*	*(0.80)*	*(0.69)*	*(0.83)*	*(0.35)*	*(0.47)*
CANALS	22.2	10.9	1.6	0.7	2.87	1.61	37.3
Residential	23.5	12.8	1.61	0.71	2.90	1.63	37.7
Agricultural	21.3	9.6	1.59	0.69	2.84	1.59	37
STREAMS	23.8	14	1.44	0.62	3.21	1.11	26.1
Residential	26.3	14.5	1.56	0.61	3.05	1.88	39.5
Agricultural	22	13.3	1.35	0.63	3.33	0.53	16

Table 2.3: Selected pairwise comparisons of riparian vegetation composition for land use and channel groups. t-statistic and p-values reported. [1] indicates Monte Carlo corrected p-value.

	Species Dataset		Functional Dataset	
	T-stat	*p-val*	*T-stat*	*p-val*
Ag. Stream vs. Res. Stream	0.948	0.449[1]	1.118	0.257[1]
Ag. Canal vs. Res. Canal	2.052	0.001	1.768	0.008
Ag. Canal vs. Ag. Stream	1.083	0.027	1.008	0.392
Res. Canal vs. Res. Stream	1.344	0.05	1.752	0.013

Table 2.4: Pairwise comparisons aquatic macroinvertebrate communities for land use and channel groups using PERMANOVA. T-statistic and p-values reported. *indicates Monte-Carlo corrected p-value.

	Taxonomic Dataset		Functional Dataset	
	T-stat	*p-val*	*T-stat*	*p-val*
Ag. Stream vs. Res. Stream	0.78	0.657*	0.68	0.712*
Ag. Canal vs. Res. Canal	1.35	0.025	1.38	0.029
Ag. Canal vs. Ag. Stream	0.86	0.644*	0.70	0.857*
Res. Canal vs. Res. Stream	1.40	0.098*	1.41	0.113*

Table 2.5: Mean EPT Richness (% of total species) and mean proportion of abundance (%) of EPT taxa groups of canal and stream sites in Larimer and Weld Counties, Colorado.

Channel Type	EPT_r Index	EPT_a Index
CANALS	22.1	20.9
Residential	23.7	19.9
Agricultural	21.2	21.5
STREAMS	22.6	41.7
Residential	24.7	41.9
Agricultural	20.5	41.6

Figure 2.1: Visual comparison of streams and irrigation canals in agricultural and residential landscapes: A) agricultural stream, Willow Creek, Weld County, B) agricultural canal, Fort Collins, C) residential/urban canal, Fort Collins D) agricultural stream, Lone Tree Creek, Weld County.

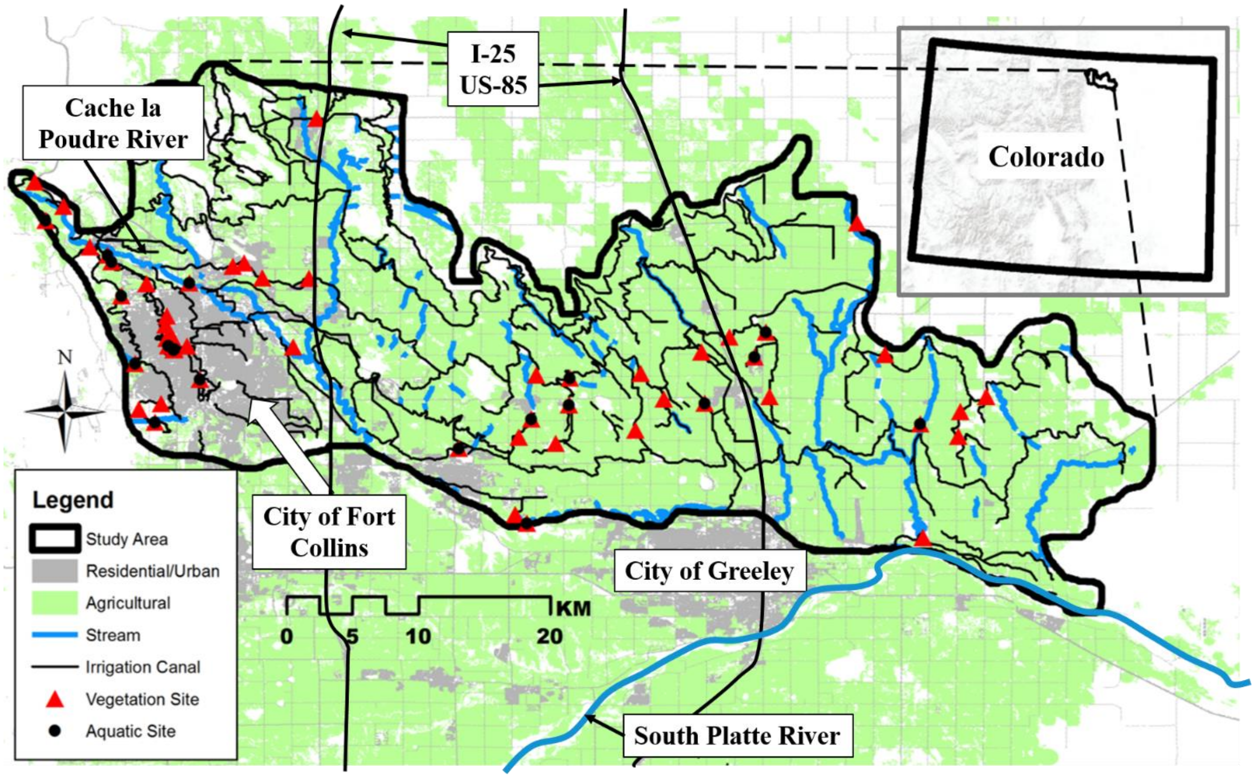

Figure 2.2: Study area in northcentral Colorado.

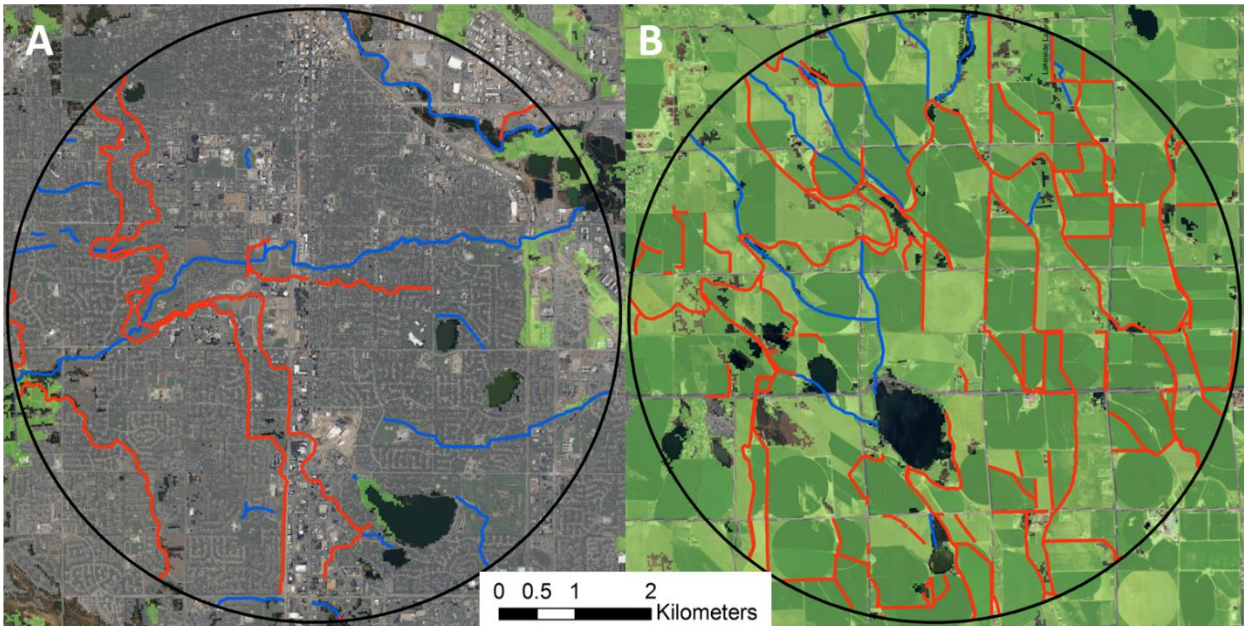

Figure 2.3: Comparison of land use (residential = gray; agricultural = green) and density of streams (blue) and irrigation canals (red) for representative 50 km^2 areas of the City of Fort Collins (A) and Weld County (B), Colorado.

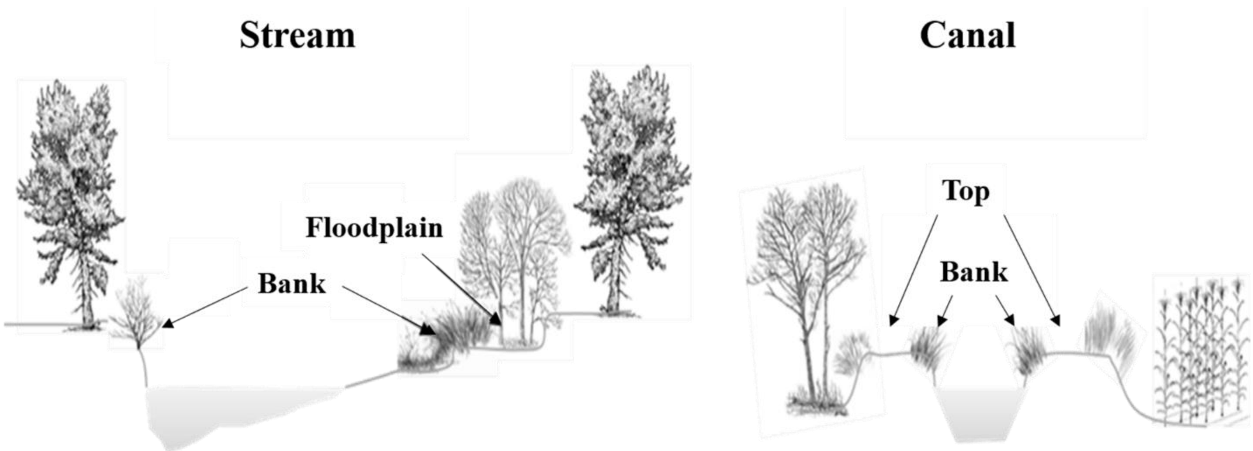

Figure 2.4: Cross section of typical stream and canal with two geomorphic surfaces (bank and top/floodplain).

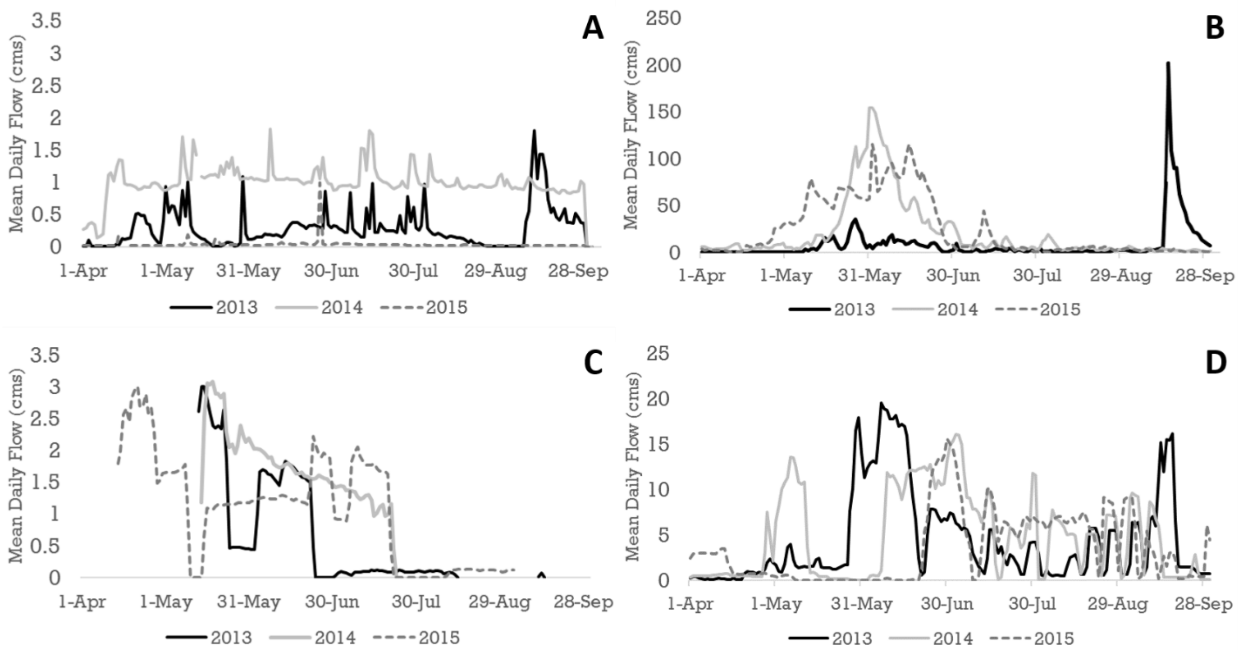

Figure 2.5: Mean daily streamflow (cms) for the April-Sept irrigation season for the years 2013-2015 in one small stream (Spring Creek in Fort Collins) (A), large stream (Cache la Poudre River at USGS gage 6752260) (B), small canal (C), and large canal (D). Note y-axis scale differences. All canal data are from gages at the point of diversion from the Cache la Poudre River.

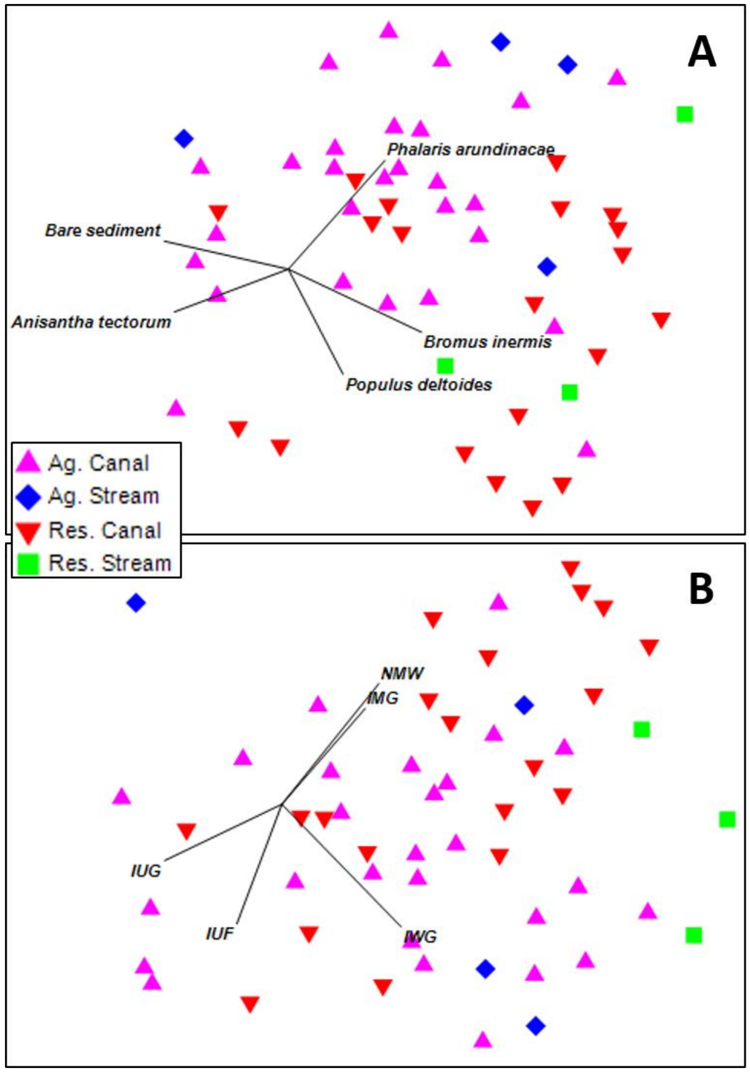

Figure 2.6: nMDS of vegetation averaged by site (n = 54). Panel A shows species level data with vectors for species with a Pearson r > 0.5, 2-D stress was 0.22. Panel B shows functional group data with vectors for functional groups with Pearson r > 0.5, (IUG = introduced upland grass, IUF = introduced upland forb, IWG = introduced wetland grass, NMW = native mesic woody, IMG = introduced mesic grass). 2-D stress was 0.21 for B.

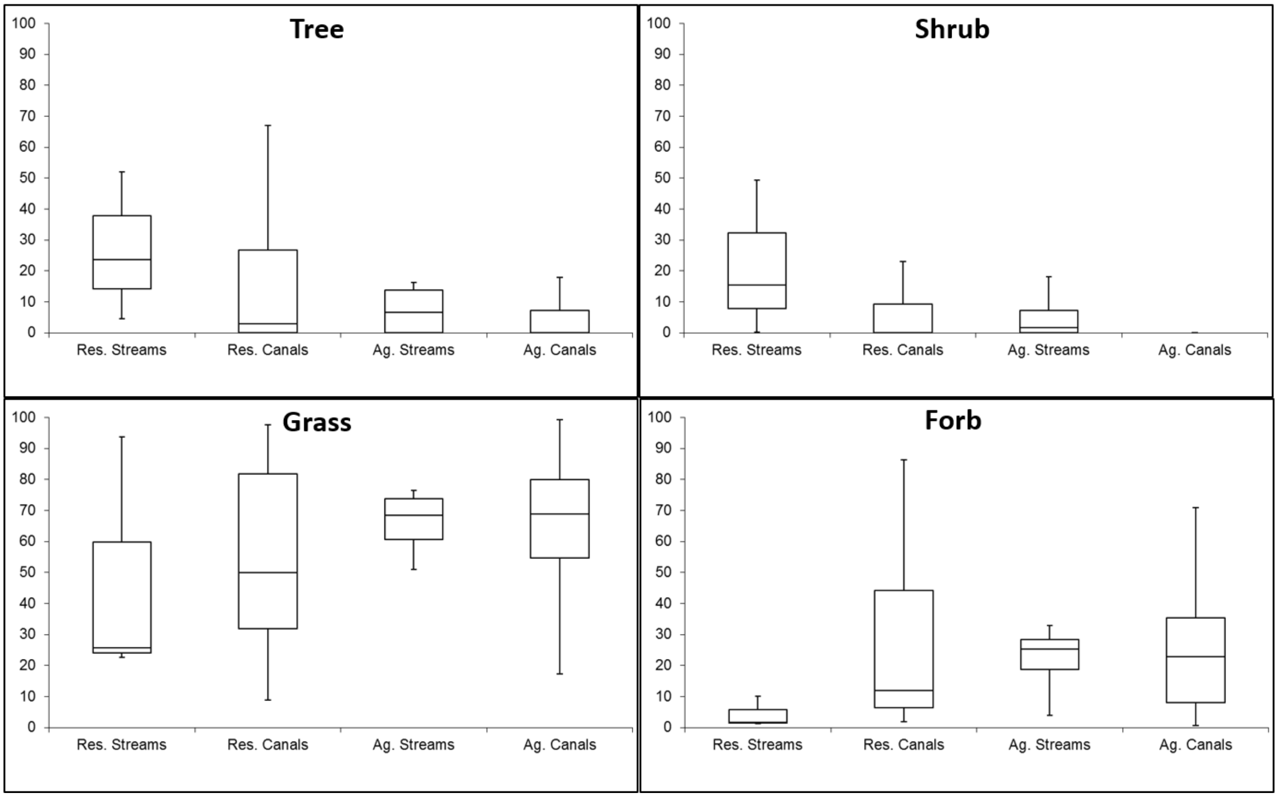

Figure 2.7: Box plot of average percent cover of major plant growth forms for each channel group. Endpoints indicate maximum and minimum values, the box indicates 25th an 75th percentiles with the mean median as the center line.

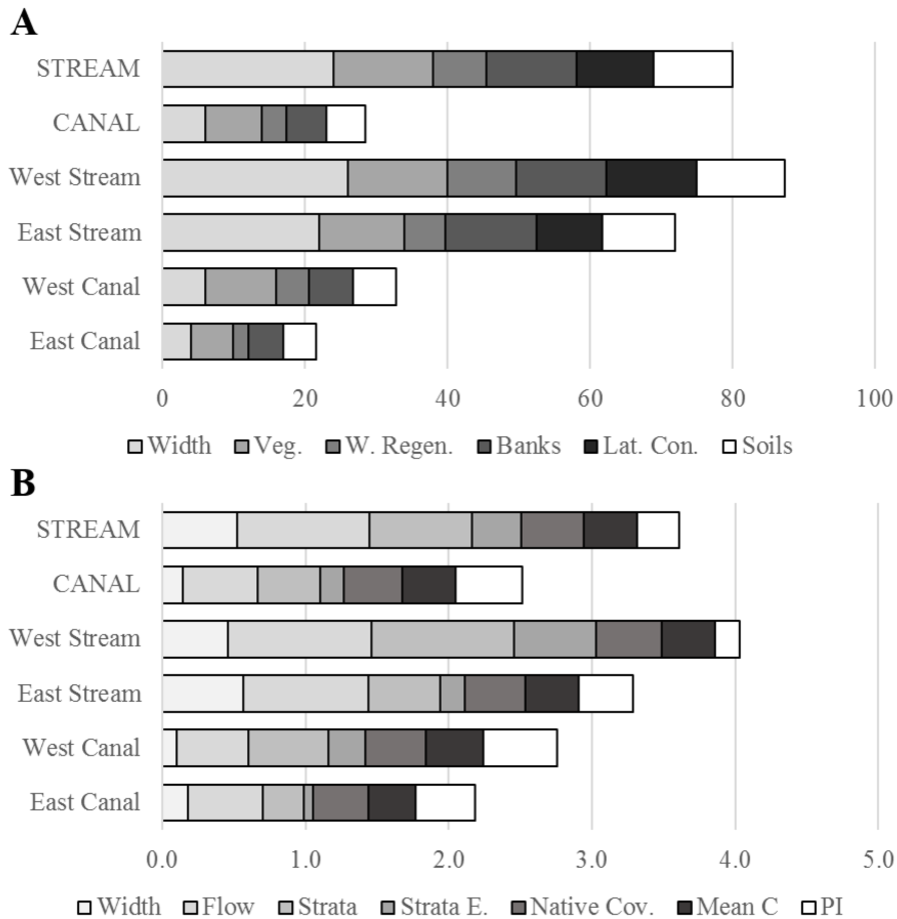

Figure 2.8: Breakdown of contributing values of each attribute used to score stream and canal sites in Larimer and Weld Counties, Colorado using the RQI (Panel A) and the HQI (Panel B). RQI attributes "Width" and "Veg." are the combine score for right and left banks. Average values used for each group.

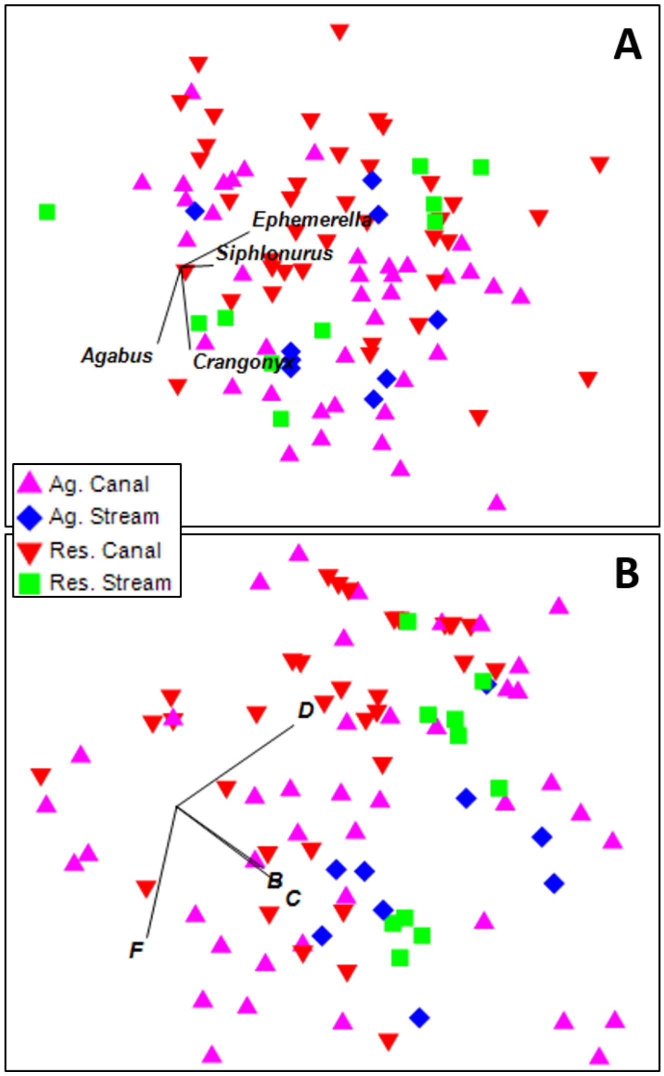

Figure 2.9: nMDS of macroinvertebrate samples (n = 97). Panel A shows taxa level data with vectors of selected taxa showing correlation to points, 2-D stress was 0.24. Panel B shows functional group data with vectors representing groups with Pearson correlation R > 0.4, 2-D stress was 0.18.

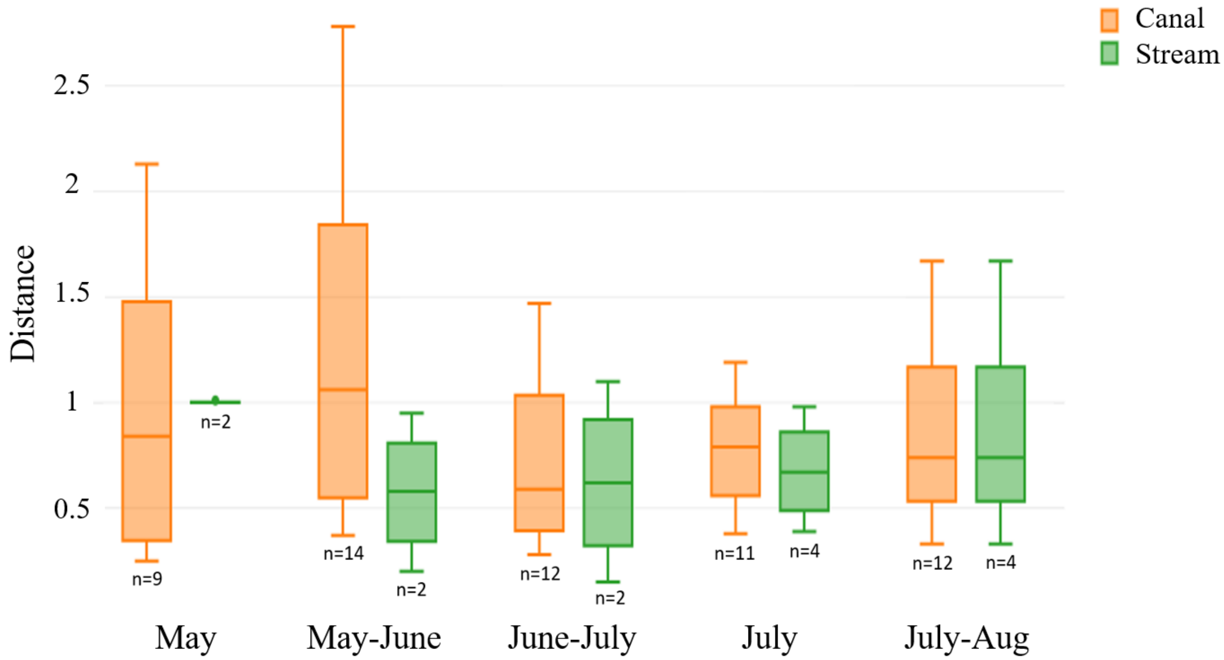

Figure 2.10: The magnitude of change (measured as Euclidean distance in 3-dimensional space) between aquatic macroinvertebrate communities from consecutive samples (3-weeks). Endpoints indicate maximum and minimum values, the box indicates 25th an 75th percentiles with the mean median as the center line.

Table 3.1: Soil characteristics for each site sampled near groundwater monitoring wells. Site name (e.g. B1, E), distance into wetland (e.g. *wb*=wetland boundary, 10 m). Dark gray shading indicates where data was not collected.

	Bulk Density (g/cm^3)		Sand (% volume)		Microbial Biomass (ug C/g dry soil)	
Well	*0-25 cm*	*25-50 cm*	*0-25 cm*	*25-50 cm*	*0-25 cm*	*25-50 cm*
B1 wb	1.21	1.29	7	3	153.95	39.72
B1 10m	1.03	1.13	12	5	62.87	82.59
B2 wb	1.1	1.05	1	18	94.32	66.51
B2 10m	0.93	0.92	6	5	145.95	70.14
B2 20m	0.76	1.11	10	18	200.65	7.73
B2 30m	0.61		16		443.62	
E wb	1.42	1.46	8	8	53.16	14.72
E 10m	1.19	1.45	17	7	60.99	13.3
E 20m	1.13	1.43	17	3	92.66	19.26
S1 wb	1.37	1.35	65	76	58.12	4.38
S1 10m	1.08	1.42	69	77	11.89	0.652
S1 20m	1.15	1.4	57	67	114.85	17.66
S2 wb	1.34	1.38	6	97	38.89	7.21
S2 10m	1.22	1.54	72	77	74.38	5.23
F wb	1.22	1.39	4	11		
F 10m	1.07	1.32	7	4		
F 20m	0.9	1.22	7	3		

Table 3.2: Groundwater NO_3-N concentration over time during June-August 2015. Labels indicate site (B1, E or F), wetland position (wb = wetland boundary, 10m = 10 m into wetland, 20m = 20 m into wetland) and transect 1 (^{1}) or 2 (^{2}). *BDL* = below detection limit of 0.01 mg/L. *Dry* indicates no measurable water was in the well. *Damaged* indicates where farm machinery compromised the integrity of the well which was subsequently replaced.

	2-Jun	26-Jun	29-Jun	14-Jul	26-Jul	6-Aug
B1 wb^1		0.298		1.62	2.24	0.242
B1 wb^2		4.38		1.98	0.635	0.116
B1 10m^1		0.014		0.0848	0.0405	0.0105
B1 10m^2		*BDL*		0.0169	*BDL*	0.0438
E wb^1			0.027	*BDL*	*BDL*	*BDL*
E wb^2			0.055	*Damaged*	*Damaged*	2.31
E 10m^1			0.113	*BDL*	3.48	3.28
E 10m^2			*BDL*	*BDL*	0.0911	0.47
E 20m^1			0.03	*BDL*	0.642	1.16
E 20m^2			*BDL*	*BDL*	1.23	0.854
F wb^1		Dry		*BDL*	*BDL*	*BDL*
F wb^2		Dry		*BDL*	*BDL*	*BDL*
F 10m^1		0.112		*BDL*	*BDL*	*BDL*
F 10m^2		0.088		0.244	0.262	0.0344
F 20m^1		0.271		*BDL*	*Dry*	*BDL*
F 20m^2		*BDL*		*BDL*	*Dry*	*BDL*

Table 3.3: Pearson correlation for selected site-averaged environmental and biological variables for study wetlands in Weld County, Colorado. Collections and measurements made in summer 2015. * indicates significance at alpha = 0.05.

	Bulk Density	*Soil Organic Matter*	*% Time < 25cm*	*% Time < 50cm*	*Microbial Biomass*	*Aboveground Biomass*
Bulk Density	1					
Soil Organic Matter	-0.99*	1				
% Time @ 25cm	-0.07	0.12	1			
% Time @ 50cm	0.16	-0.12	0.96*	1		
Microbial Biomass	-0.99*	0.99*	0.02	-0.25	1	
Plant Biomass	-0.29	0.35	0.89*	0.76*	0.28	1

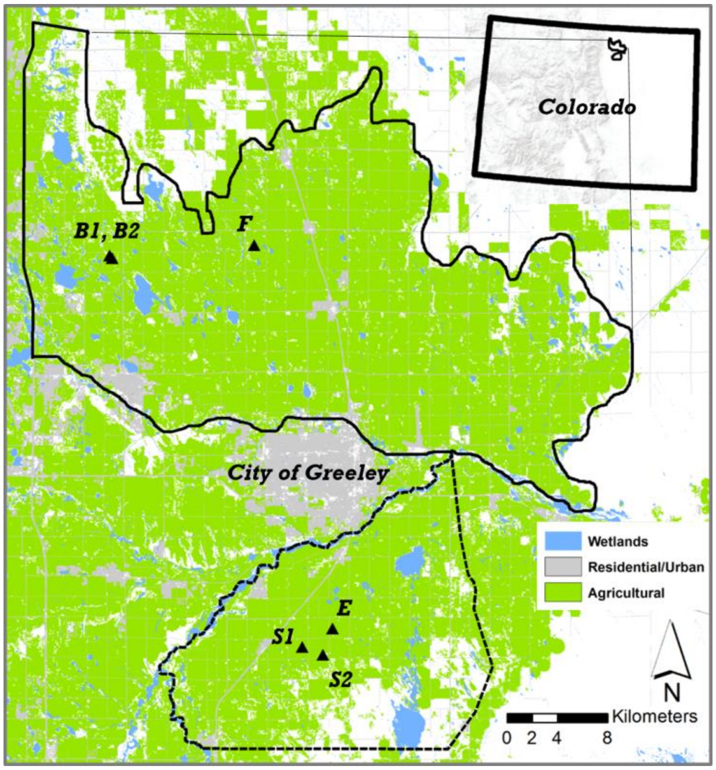

Figure 3.1: Study area map showing agricultural and residential land use, wetlands, site locations and northern (solid outline) and southern (dashed outline) portions of the study area in northcentral Colorado, U.S.A. Site locations are approximate to protect the privacy of landowners.

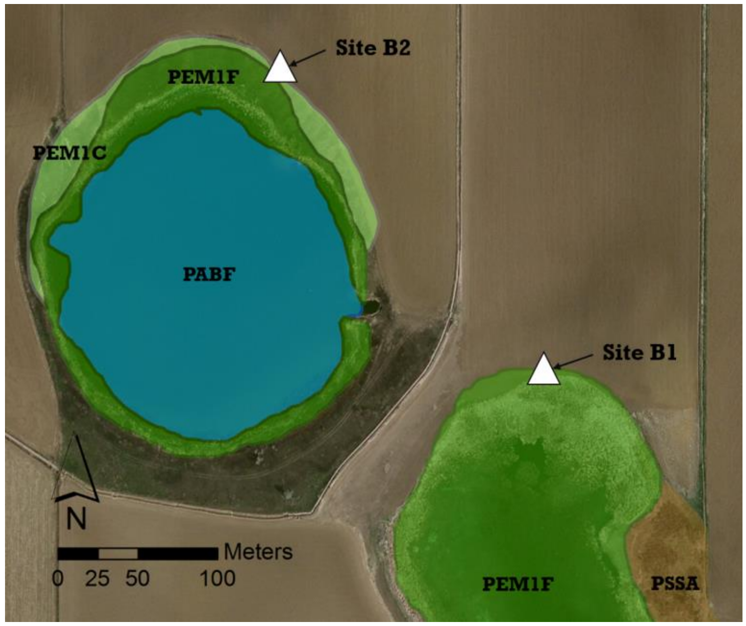

Figure 3.2. Typical emergent ("PEM") wetlands bordering ponds ("PAB") in the northern study area. NWI wetland classification codes indicate persistence of saturated conditions (PEM1F = semi-permanently flooded; PEM1C = seasonally flooded; PSSA = temporarily flooded shrubs; PABG = permanently flooded pond). Land slope is from north to south into sites B1 and B2.

Figure 3.3: Field installation of *in situ* denitrification experiment. Paired control and acetylene amended incubation tubes (left) with silicone tubing and stoppers for gas sampling. Groundwater well with sensor (center) and piezometer nest (right).

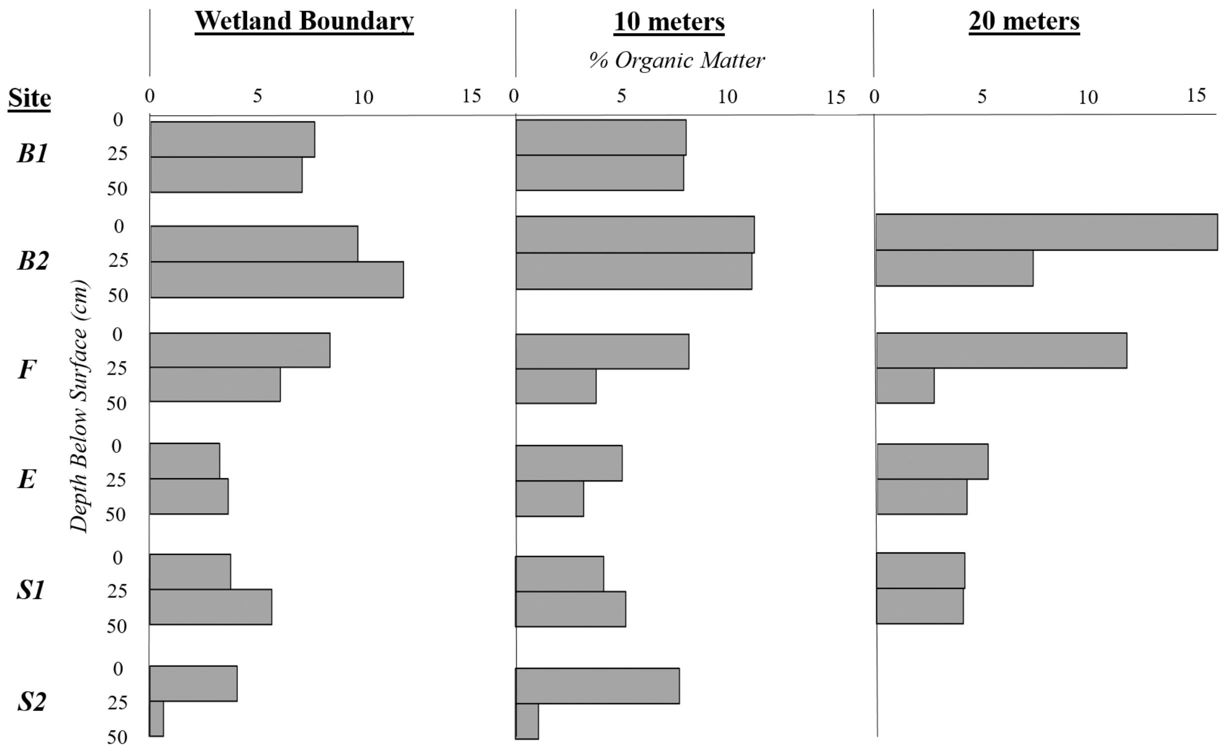

Figure 3.4: Soil organic matter content (%) determined by loss on ignition at wetlands sites in Weld County, CO for two depths (0–25 cm and 25–50 cm) and up to three locations from the wetland boundary (0, 10 and 20 m).

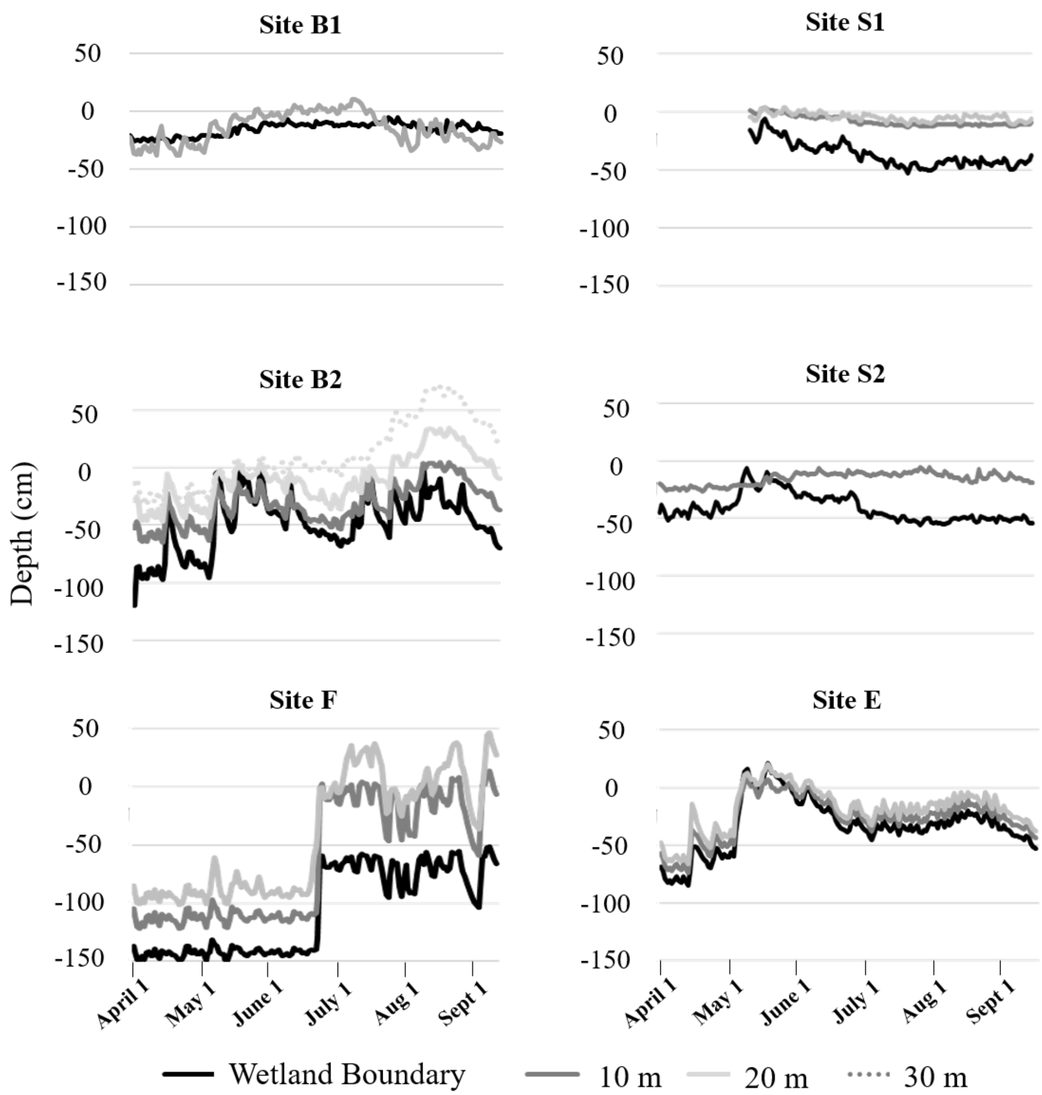

Figure 3.5: Groundwater levels at wetland sites from April to mid-September, 2015. Note site S1 was not instrumented until May 12, 2015 due to weather.

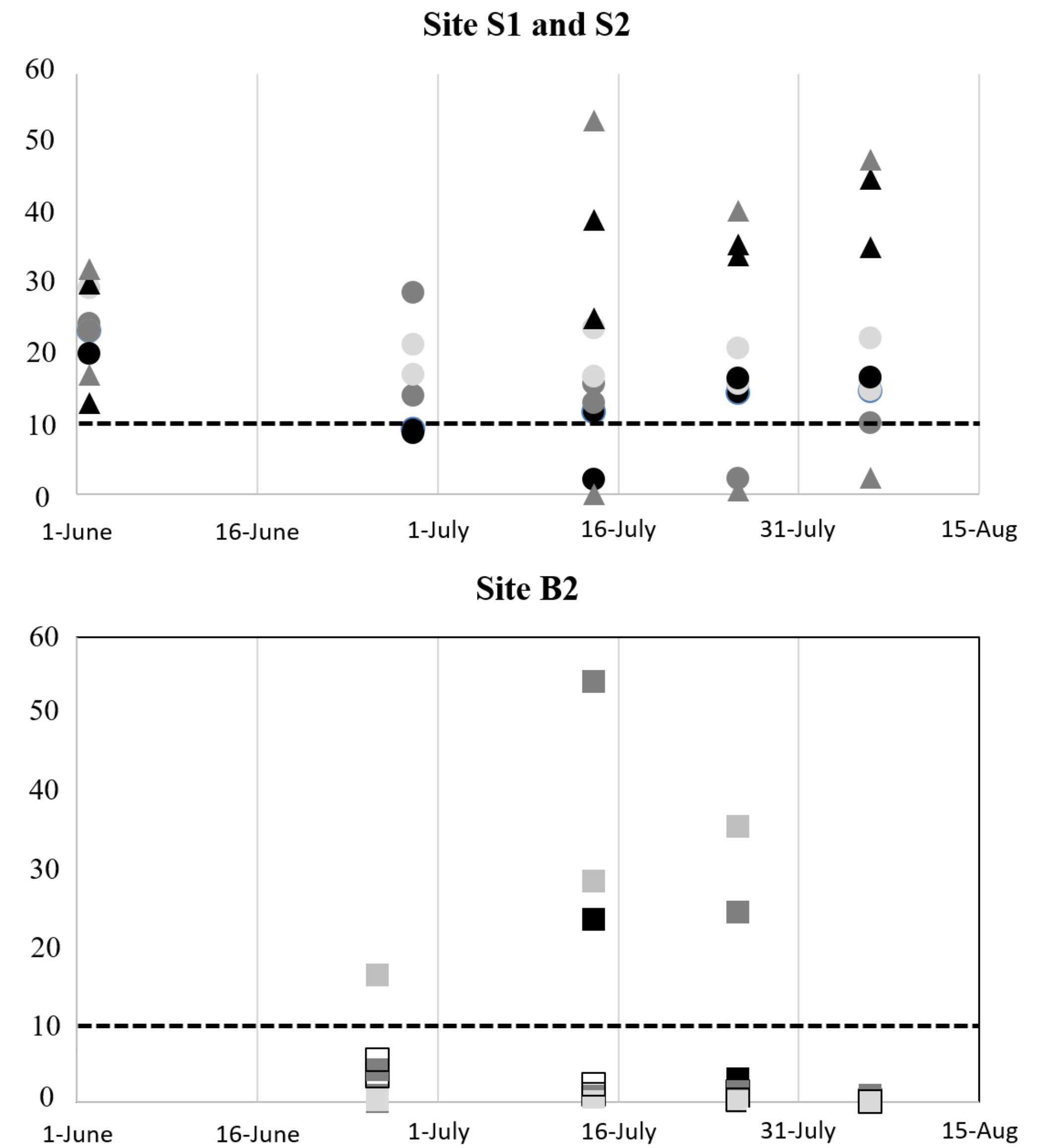

Figure 3.6: Groundwater nitrate concentration from June to August, 2015 from paired transects in study wetlands in Weld County, Colorado. Top panel site S1=circles, S2=triangles. Bottom panel site B1=squares. Distance from the wetland boundary is represented by a color gradient from black to gray. Open squares in bottom panel indicate sites 30 m into the wetland. EPA drinking water standard (10 mg/L of NO_3-N) indicated by dashed line.

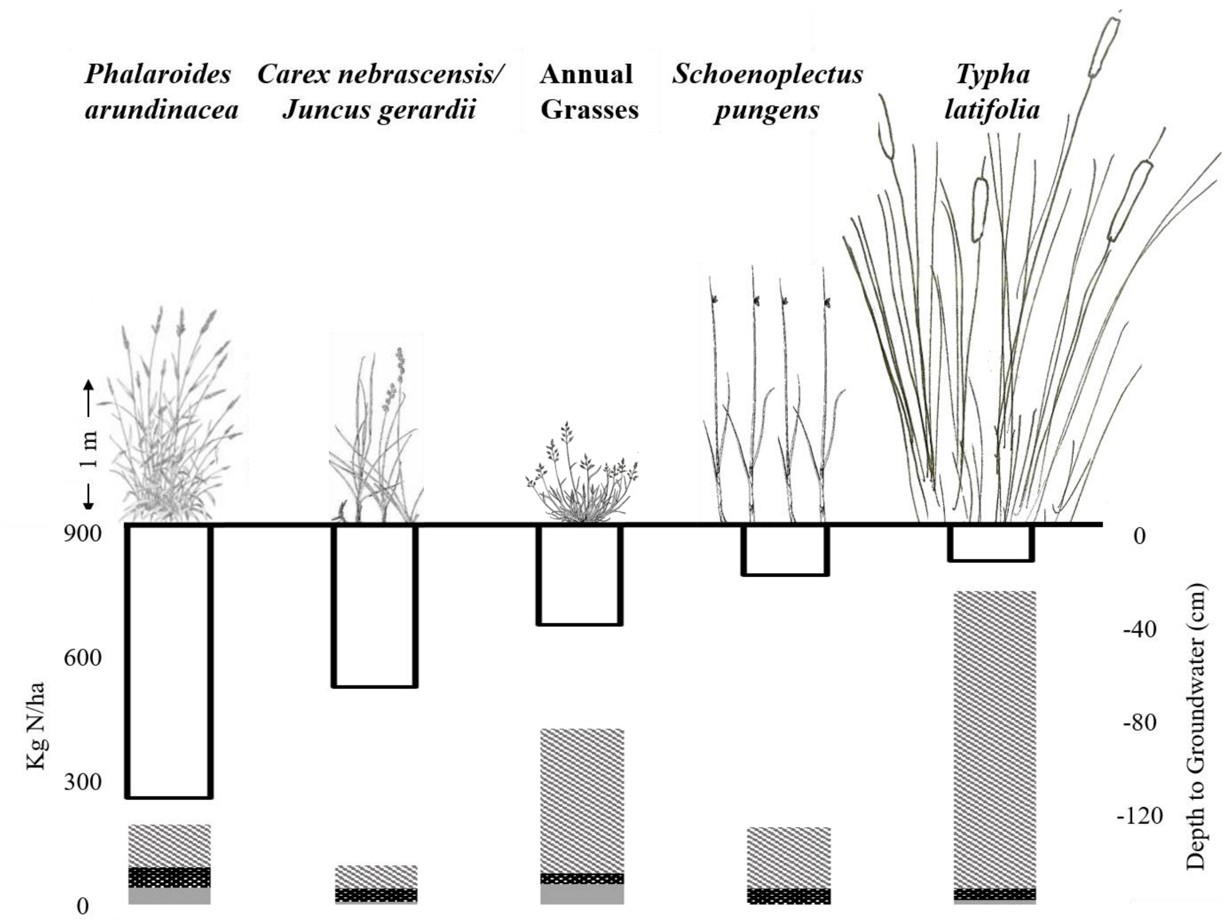

Figure 3.7: Relationship of mean water table depth and N uptake and transformation. Mean depth to groundwater (open bars) scaled on the right axis, concentrations of N in aboveground biomass (kg N/ha) (grey hatch), litter (black) and denitrification rate (gray) scaled on the left axis for each major plant community (kg N ha^{-1} yr^{-1}). Note denitrification was not measured under *Schoenoplectus pungens*.

Table 4.1: Canal cover and land use classification with number of samples, mean HQI score from previous study (Chapter 2) and standard deviation. * indicates standard deviation of the land use group applied when samples were too low to calculate the standard deviation.

Cover	Land use	Samples	Mean	Standard Deviation
Concrete	Agriculture	4	1.63	0.45
Concrete	Residential	1	2.31	0.73*
Herb	Agriculture	13	2.35	0.64
Herb	Residential	11	2.09	0.65
Shrub	Agriculture	1	3.46	0.60*
Light Canopy	Agriculture	2	3.02	0.60*
Heavy	Agriculture	1	3.41	0.60*

Table 4.2: N uptake and transformation rates for wetland types from field study conducted June-August 2015 in Weld County, Colorado (Chapter 3). Rates reported as total N. Wetland codes are from Cowardin et al. (1979) wetland classes.

Wetland Type (NWI Code)	Annual N Transformation ($kg\ ha^{-1}\ yr^{-1}$)	Annual N Uptake ($kg\ ha^{-1}\ yr^{-1}$)
Temporarily Flooded (PEMA)	23.2	199.0
Seasonally Flooded (PEMC)	2.1	150.1
Semi-permanently Flooded (PEMF)	5.9	939.9

Figure 4.1: An example of a main stem canal (A) with significant vegetation management for road access and water conveyance, and (B) a small lateral ditch in Weld County, Colorado.

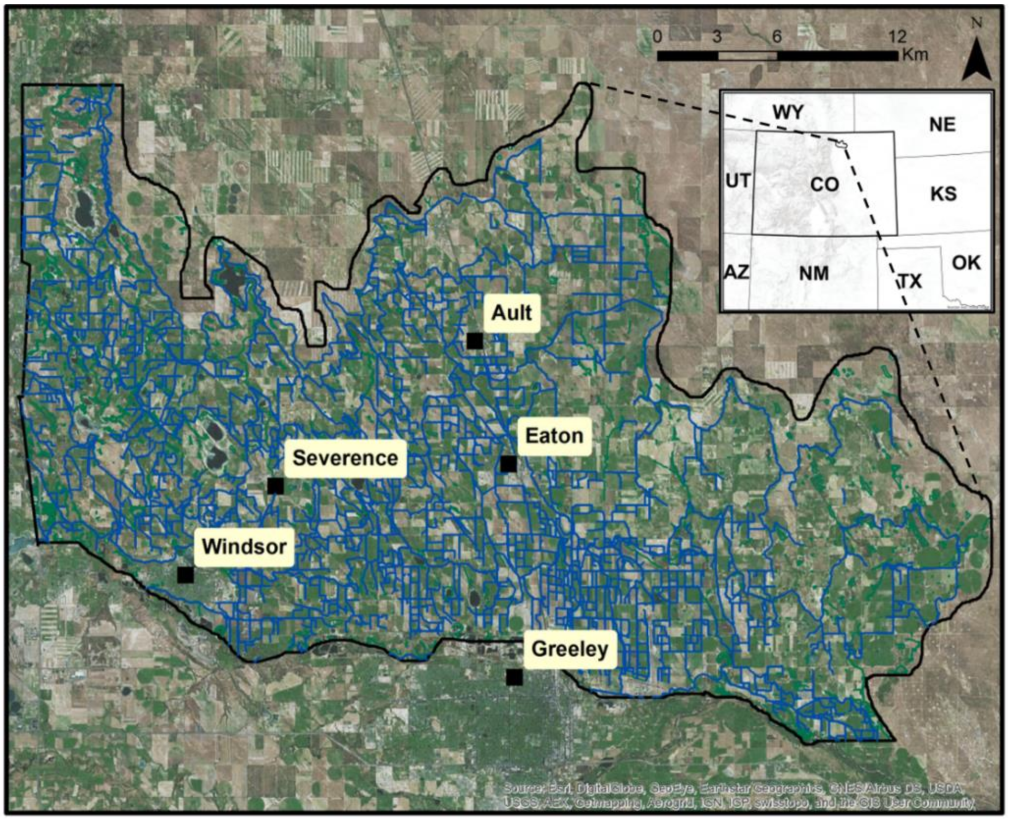

Figure 4.2: Study area in northcentral Colorado. The study area is approximately bounded by an Interstate-25 to the west, the Cache la Poudre River to the south and the service area boundaries of irrigation companies to the north and east.

Figure 4.3: An example of an agricultural wetland supported by irrigation runoff in Weld County, CO near site F. Water moves from left to right and top to bottom into the wetland. Multiple wetland plant species including *T. latifolia*, *C. nebrascensis*, *S. pungens* create clonal patches differentiated by color and texture.

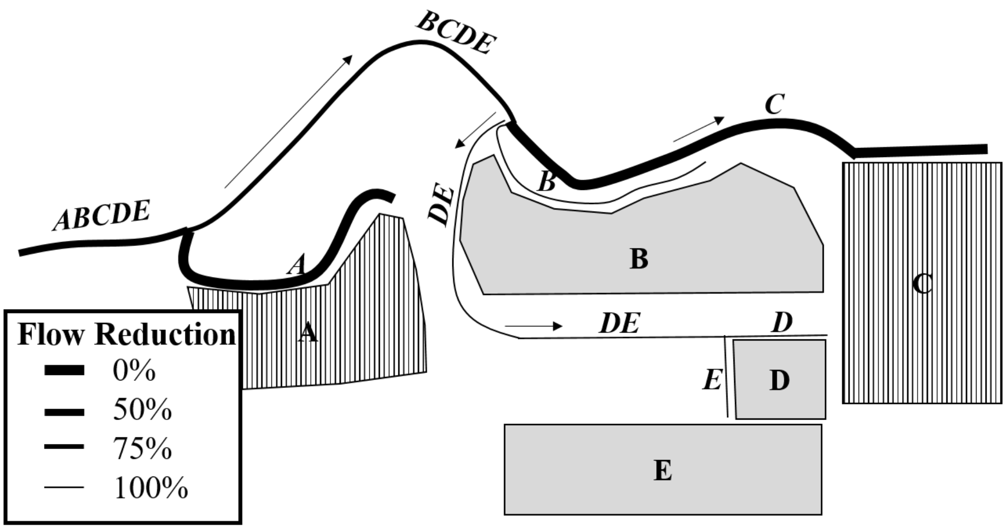

Figure 4.4: A hypothetical example of percent flow reductions through an irrigation network. Flow direction is indicated by arrows. Irrigated fields (polygons) are either continuously irrigated (stripped) or dried (grey). Fields are paired to canal segments by letters. Canal segments are labeled with all serviced parcels. Percent flow reduction indicated by the thickness of the line.

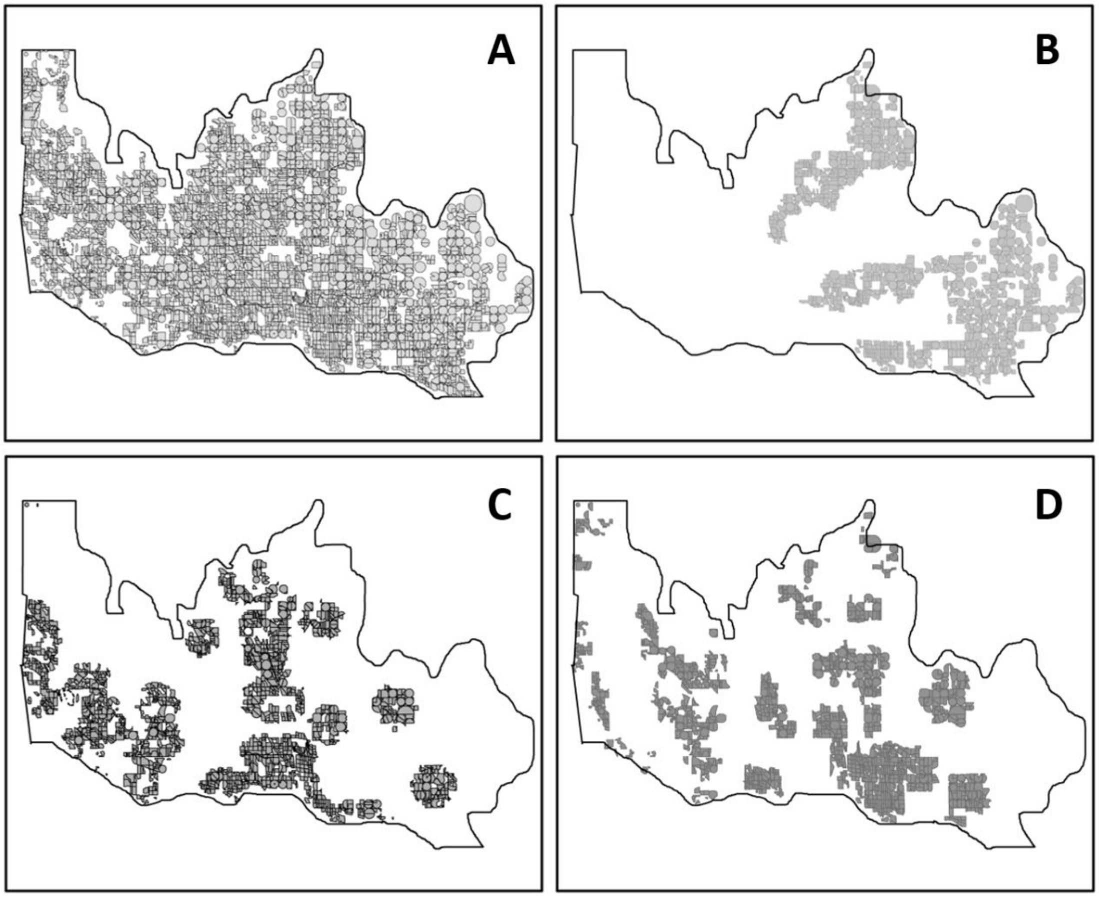

Figure 4.5: The distribution of surface water irrigated parcels (A) in Weld County, Colorado and parcels selected for dry-up under the scenarios for water conservation (B), residential/urban development (C), and conserving ecosystem services (D). Data from CDSS for 2010.

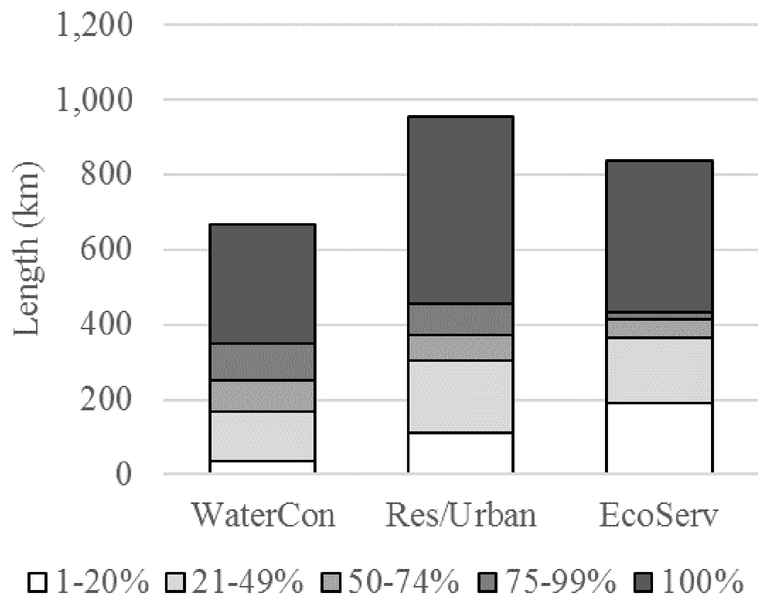

Figure 4.6: Distribution of flow reduction for canals for each water transfer scenario. Showing only canals with some level of flow reduction out of a total canal network length of 1,778 km in the study area in Weld County, Colorado.

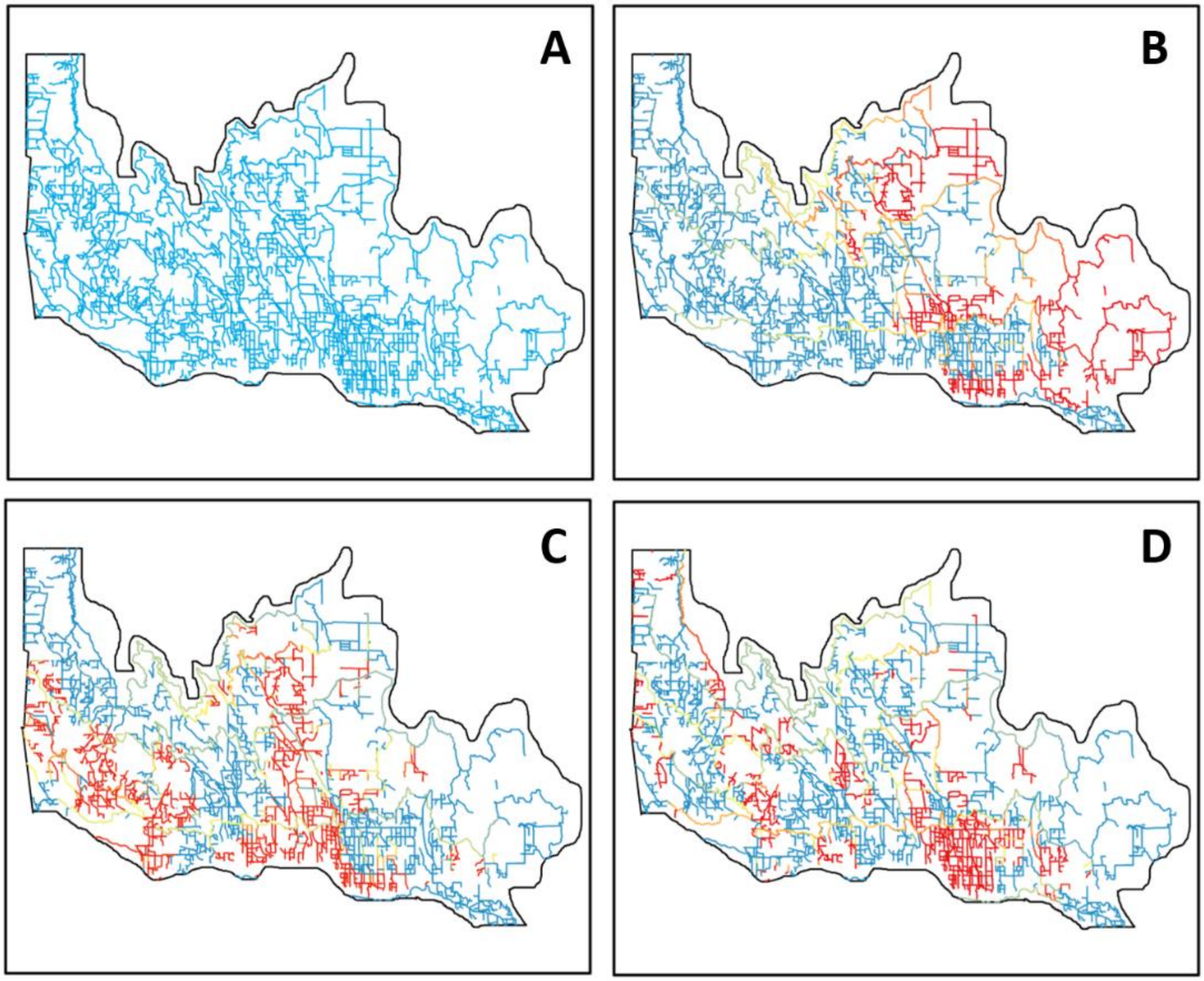

Figure 4.7: Spatial distribution of irrigation canals in the study area in Weld County, Colorado (A) and the reduction of flow associated with water transfer scenarios WaterCon (B), Res/Urban (C), and EcoServ (D). Flow reduction is scaled by color with red representing complete flow reduction and blue as no flow reduction.

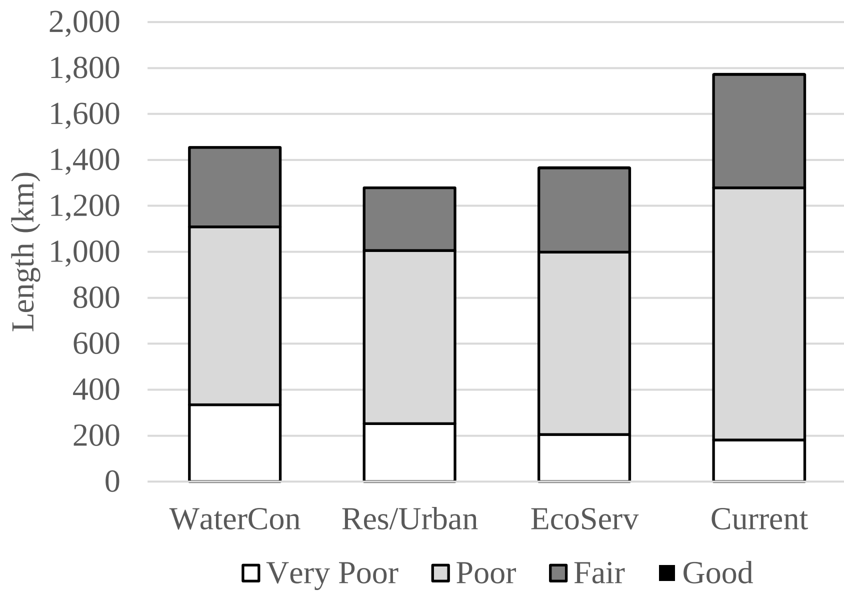

Figure 4.8: Distribution of HQI_M scores for riparian ecosystems in the study area in Weld County, Colorado. Scores grouped as Very Poor (HQI_M <1.5), Poor (1.5-2.5), Fair (2.5-3.5), Good (3.5-4.5). Canals with 100% flow reduction were not included because they are expected to have no riparian habitat value.

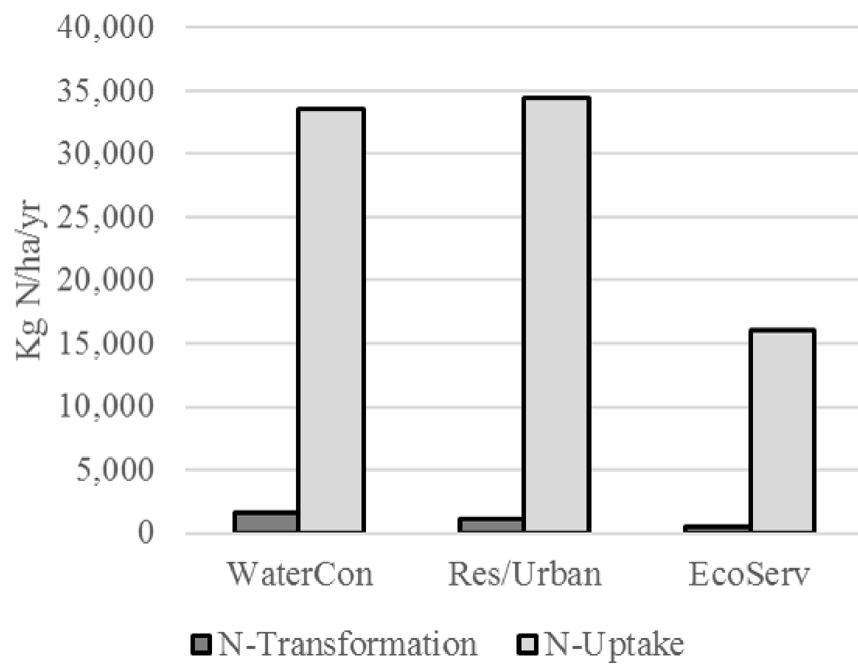

Figure 4.9: Estimated reduction in N transformation and uptake by wetlands within 15m of irrigated crops in the study area in Weld County, Colorado for three water transfer scenarios. Calculated using rates measured in the field in 2015.

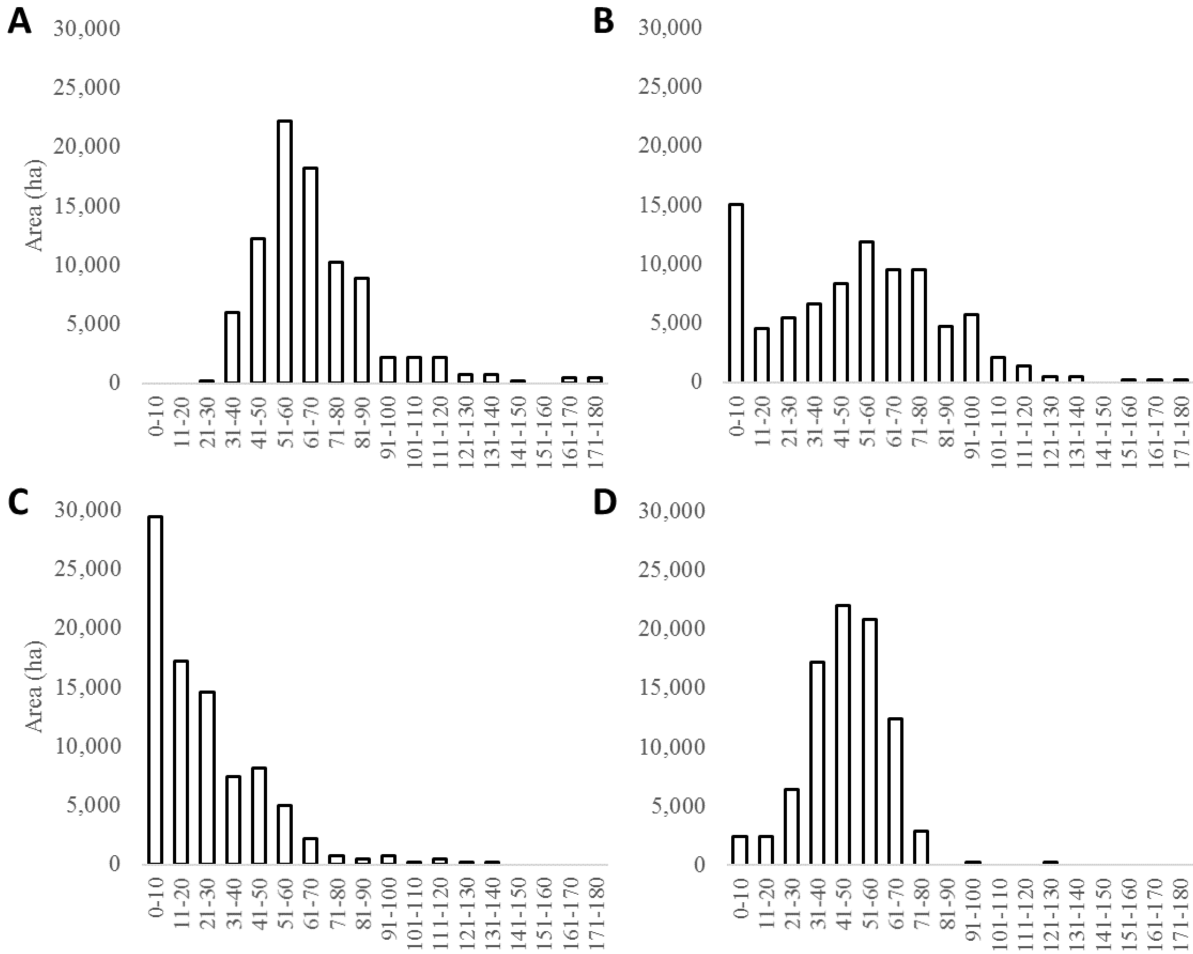

Figure 4.10: Distribution of the combined ecosystem scores calculated for the study area in Weld County, Colorado. Public Land Survey System sections for current conditions (A) and three water transfer scenarios WaterCon (B), Res/Urban (C), and EcoServ (D).

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8. APPENDIX

8.1 Additional Equations, Chapter 2

Eq. 1 $$Cover\ weighted\ mean\ C = \frac{(\sum C_j * MC_j)}{TC}$$

Eq. 2 $$Prevalence\ Index = \left(\frac{A_{OBL} + 2A_{FACW} + 3A_{FAC} + 4A_{FACU} + 5A_{UPL}}{TA}\right)$$

Where:
C_j = c-value for species j
MC_j = mean percent cover for species j
TC = total percent cover for all species
A = sum of percent cover for all species of specified wetland indicator status
TA = sum of percent cover for all species

8.2 Additional Vegetation Metrics, Chapter 2

Two metrics were calculated using the full species dataset to describe the character of the vegetation: the coefficient of conservatism (C-value) and the prevalence index of wetland vegetation (PI). The value of conservatism (C-value) gives each species a rating (0-10) on their requirements for undisturbed habitats with limited to no human interference in ecological processes (Wilhelm 1977, Rocchio 2007). Introduced species are given a value of 0 and endemic sensitive species values near 10. Cover weighted mean C formula (section 8.1) incorporates the ratings and mean cover of the species to calculate a single value for each site. The wetland prevalence index (PI) (section 8.1) follows a similar approach with National Wetland Indicator values (1-5) (Lichvar et al. 2012) categorizing the likelihood a species would be observed in a wetland (Wentworth et al. 1988). PI values < 3 indicate a wetland (FICWD 1989; SCS 1994).

8.3 Modifications to RQI (del Tanago & de Jalon 2010), Chapter 2

Longitudinal connectivity was not included as sample reaches were 30-150 m. The total score and riparian status interpretations were adjusted: 0-10 very bad, 10-30 bad, 30-50 poor, 50-70 moderate, 70-90 good, 90-120 very good. Metrics described in Table 8.1 were normalized on a 0-1 scale using the range of values measured in the study.

8.4 Physical and Landscape Variables, Chapter 2

Bankfull width (*BF_Width)* and depth (*BF_depth*) (Leopold and Maddock 1953; Leopold et al. 1964, Rosgen 1994), land use (*Land*) categorized into one of six groups following NLCD class definitions (Homer et al. 2015) with integer values representing an increase in disturbance and vegetation management: natural = 1, pasture/hay = 2, low intensity residential = 3, high intensity residential = 4, commercial/industrial = 5, and row crops/animal production = 6, for each side separately then summed for the site. Top/floodplain elevation (*Top_HT*) the vertical distance of the mean floodplain elevation or canal top above the bankfull boundary, angle of the bank (*Angle*) the gradient from the bankfull boundary to the top/floodplain surface, channel bed substrate (*Bed_Sub*), and distance from the point of diversion (*Dist*) for canals only. Additional variables included monthly depth and temperature observations (e.g. *May_Depth*, *June_Temp*), observed depth ratio (*Ratio*) used as a surrogate for flow, calculated as *Observed* *depth*/*BF_depth*, presence of macrophytes (*Macro*), and presence of submerged overhanging terrestrial vegetation (*Terr_Veg*).

8.5 Hydrologic Variables, Metrics, and Indices, Chapter 2

Seven flow variation metrics were calculated (*CV* and *R-B Index*), timing (*Peak_Date*) and frequency of high flow events (*High*), the number of low flow events (*Low*), number of days at low flow (*Low_Days*) and the number of days without flow (*Zero_Days*) during the period of canal operation or between the April 1 – Sept 30 period, whichever was shorter. CV is calculated as the average of the annual coefficient of variation calculated as the standard deviation/mean, using daily streamflow data. The R-B Index indicates trends in flow oscillations relative to total flow (Baker et al. 2004). High flows exceeded the 90^{th} percentile of flow and low flows were below the 10^{th} percentile calculated on 2013-2015 daily flow data. A single flow event was defined as a contiguous set of daily values within the range for high (> 90%) or low (< 10%) flows with at least 2 days separating the next day(s).

Table 8.1 Names and physical characteristics of canal and stream sites selected as field sites in Larimer and Weld Counties, Colorado. * indicates missing value due to high flow during visit.

Site Name	Channel Type	Land Use	Vegetation	Distance (km)	Width (m)	Depth (m)	Substrate
287	canal	Res.	herb	6.50	11.5	1.1	cobble
Arch	canal	Ag.	herb	51.62	14	1.2	mud
BigFlow	canal	Ag.	light canopy	8.67	4.5	0.6	gravel
Bryan	canal	Res.	light canopy	8.76	6	1.2	sand
Cemetery	canal	Res.	light canopy	1.00	4.5	0.8	cobble
Center	canal	Res.	herb	13.36	7	0.9	sand
Cinema	canal	Res.	herb	16.13	5	0.8	cobble
CoHabit	canal	Res.	shrub	4.50	5	0.7	cobble
CottonCoon	canal	Ag.	light canopy	91.01	5.2	0.5	sand
CountryClub	canal	Res.	herb	5.65	17	*	*
CowFly	canal	Ag.	herb	91.31	3	0.7	mud
Crestone	canal	Res.	heavy canopy	7.89	6	1	cobble
EatonClub	canal	Ag.	concrete	8.84	1.2	0.4	concrete
ELC	steam	Res.	light canopy	0.00	17	1.4	cobble
Fromme	canal	Res.	herb	32.80	1.2	0.4	sand
Geiss	canal	Ag.	herb	83.72	1.5	0.5	sand
Goat	canal	Res.	light canopy	14.91	4.5	0.9	sand
greeley	canal	Ag.	shrub	69.47	4.5	0.9	sand
HillCanal	canal	Res.	shrub	9.12	4	1	cobble
Kreykas	canal	Ag.	concrete	46.37	1.2	0.4	concrete
Landfill	canal	Res.	herb	34.36	2	0.4	sand
LawSchool	canal	Ag.	herb	2.15	1	0.2	mud
LawSlough	canal	Ag.	herb	0.49	2.5	0.4	mud
Lemay	canal	Res.	herb	4.78	16	*	*
LindenLake	canal	Res.	herb	7.25	16	*	*
LoneTreeCreek	steam	Ag.	light canopy	0.00	2.5	0.5	gravel
Mantis	canal	Ag.	herb	78.62	6	1.3	mud
McMurray	stream	Res.	heavy canopy	0.00	30	1.7	cobble
McNear	canal	Ag.	heavy canopy	55.50	2	0.8	sand
Metal Bridge	canal	Ag.	herb	91.31	3	1.3	mud
NiceFellas	canal	Ag.	herb	61.52	10	2.5	*
NoTouchCorn	canal	Ag.	herb	63.13	1.5	1.1	mud
OilWell	canal	Ag.	herb	41.42	11	2	mud
Parkway	canal	Ag.	herb	66.42	1	0.5	sand
Pawnee	stream	Res.	herb	0.00	4	0.7	mud
Peachleaf	canal	Ag.	light canopy	3.14	6	1.1	mud
PoplarLane	canal	Ag.	herb	66.30	4	1.1	sand
Road	canal	Ag.	herb	38.22	13	2	cobble
Rolland	canal	Res.	herb	12.51	6	1.1	sand
RollandUp	canal	Res.	herb	12.45	3.5	0.6	sand
SageCon	canal	Res.	concrete	30.71	1.5	0.7	concrete
Shed	canal	Ag.	concrete	42.54	1.5	0.5	concrete
Sheriff	canal	Ag.	herb	10.81	16	*	*
SpringCreek	stream	Res.	light canopy	0.00	2.5	0.7	sand

Stillwater	stream	Ag.	light canopy	0.00	24	2	gravel
StuartCon	canal	Res.	concrete	10.79	5	0.9	concrete
StuartLow	canal	Res.	heavy canopy	11.13	5	1.2	cobble
TedsPlace	canal	Ag.	herb	3.96	10	2	mud
Tennis	canal	Res.	heavy canopy	11.56	5	1.1	gravel
WaterWorks	canal	Res.	heavy canopy	1.58	4.5	1	cobble
WellSchool	canal	Ag.	herb	33.06	11	1.3	sand
WhiteFence	canal	Ag.	concrete	54.31	2.5	0.5	concrete
WillowCreek	stream	Ag.	herb	0.00	1.5	0.7	sand
Windsor	canal	Ag.	herb	7.30	22	*	sand

Table 8.2: Description of metrics used to assess Habitat Quality Index (HQI) of riparian areas.

Metric	*Description*	*Calculation*
Width_R	Ratio of riparian area width to bankfull width	$(Rip\_width - BF\_width) / BF\_width$
Mean-C	Mean cover-weighted C-value (Rocchio 2007): a measure of the conservative nature vegetation, higher values correspond to species sensitive to human disturbance and/or with limited distributions and specific habitat requirements.	$\frac{(\sum C_j * MC_j)}{TC}$
PI	Prevalence Index of wetland vegetation (Wentworth et al. 1988): ranks species per the likelihood they would be observed in a wetland (1, likely – 5, unlikely). PI values below 3 are considered to identify a wetland (FICWD 1989; SCS 1994).	$\left(\frac{A_{OBL} + 2A_{FACW} + 3A_{FAC} + 4A_{FACU} + 5A_{UPL}}{TA}\right)$
Strata	Number of vegetation strata present	
Strat_Even	Evenness of cover in each vegetation strata present.	$\left(\frac{1}{3 * \left(\sum \left(\frac{Cover_T}{TC}\right)^2 + \left(\frac{Cover_S}{TC}\right)^2 + \left(\frac{Cover_H}{TC}\right)^2\right)}\right)$
Land	Adjacent land use: using categorical values for land use: natural areas = 1, pasture/hay fields = 2, light residential = 3, dense residential = 4, urban/commercial = 5, and row crops/animal production = 6.	$\text{Landuse}_L + \text{Landuse}_R$
% Native	% Native cover of riparian vegetation	$Cover_N$ / TC
Flow	Coarse flow regime categorization: perennial (3), intermittent (2), ephemeral (1).	

Table 8.3: List of plant species functional group assignment using taxonomy from Weber and Wittmann (2012). Information on origin, wetland indicator status and growth form from USDA Plants Database (https://plants.usda.gov).

Family	Species	Origin	Wetland Indicator	Form	Functional Group
Aceraceae	*Acer saccharinum*	Introduced	FAC	Tree	IMT
Asteraceae	*Achillea millefolium*	Native	FACU	Forb	NUF
Poaceae	*Achnatherum hymenoides*	Native	FACU	Grass	NUG
Asteraceae	*Acosta diffusa*	Introduced	UPL	Forb	IUF
Euphorbiaceae	*Agaloma marginata*	Native	FACU	Forb	NUF
Asteraceae	*Agoseris glauca*	Native	FACU	Forb	NUF
Poaceae	*Agropryon cristatum*	Introduced	UPL	Grass	IUG
Poaceae	*Agrostis gigantea*	Introduced	FACW	Grass	IWG
Alliaceae	*Allium textile*	Native	UPL	Grass	NUG
Poaceae	*Alopecurus aequalis*	Native	OBL	Grass	NWG
Amaranthaceae	*Amaranthus retroflexus*	Introduced	FACU	Forb	IUF
Asteraceae	*Ambrosia psilostachya*	Native	FACU	Forb	NUF
Asteraceae	*Ambrosia tomentosa*	Native	FACU	Forb	NUF
Asteraceae	*Ambrosia trifida*	Introduced	FAC	Forb	IMF
Fabaceae	*Amorpha fruticosa*	Native	FACW	Forb	NWF
Poaceae	*Andropogon gerardii*	Native	FACU	Grass	NUG
Ranunculaceae	*Anemonidium canadense*	Native	FACW	Forb	NWF
Poaceae	*Anisantha tectorum*	Introduced	FACU	Grass	IUG
Apocynaceae	*Apocynum cannabinum*	Native	FAC	Forb	NMF
Apocynaceae	*Apocynum sibiricum*	Native	FAC	Forb	NMF
Asteraceae	*Arctium minus*	Introduced	FACU	Forb	IUF
Poaceae	*Aristida purpurea*	Native	UPL	Grass	NUG
Asteraceae	*Artemisia frigida*	Native	UPL	Forb	NUF
Asteraceae	*Artemisia ludoviciana*	Native	UPL	Forb	NUF
Asclepiadaceae	*Asclepias speciosa*	Native	FAC	Forb	NMF
Asparagaceae	*Asparagus officinalis*	Introduced	FACU	Forb	IUF
Boraginaceae	*Asperugo procumbens*	Introduced	UPL	Forb	IUF
Chenopodiaceae	*Atriplex canescens*	Native	UPL	Forb	NUF
Chenopodiaceae	*Atriplex dioica*	Introduced	FACW	Forb	IWF
Chenopodiaceae	*Atriplex heterosperma*	Introduced	FAC	Forb	IMF
Poaceae	*Avena fatua*	Introduced	FACU	Grass	IUG
Chenopodiaceae	*Bassia sieversiana*	Introduced	FACU	Forb	IUF
Asteraceae	*Bidens frondosa*	Native	FACW	Forb	NWF
Poaceae	*Bouteloua curtipendula*	Native	FACU	Grass	NUG
Asteraceae	*Breea arvensis*	Introduced	FACU	Forb	IUF
Poaceae	*Bromopsis inermis*	Introduced	UPL	Grass	IUG

Poaceae	*Bromus japonicus*	Introduced	FAC	Grass	IMG
Poaceae	*Buchloe dactyloides*	Native	UPL	Grass	NUG
Brassicaceae	*Camelina microcarpa*	Introduced	UPL	Forb	IUF
Campanulaceae	*Campanula rotundifolia*	Native	FAC	Forb	NMF
Brassicaceae	*Capsella bursa-pastoris*	Introduced	FACU	Forb	IUF
Brassicaceae	*Cardaria chalepensis*	Introduced	FACU	Forb	IUF
Brassicaceae	*Cardaria draba*	Introduced	FACU	Forb	IUF
Brassicaceae	*Cardaria latifolia*	Introduced	FACW	Forb	IWF
Cyperaceae	*Carex aquatilis*	Native	OBL	Grass	NWG
Cyperaceae	*Carex emoryi*	Native	OBL	Grass	NWG
Cyperaceae	*Carex lanuginosa*	Native	OBL	Grass	NWG
Cyperaceae	*Carex nebrascensis*	Native	OBL	Grass	NWG
Cyperaceae	*Carex praegracilis*	Native	FACW	Grass	NWG
Cyperaceae	*Carex stipata*	Native	OBL	Grass	NWG
Cyperaceae	*Carex utriculata*	Native	OBL	Grass	NWG
Poaceae	*Catabrosia aquatica*	Native	OBL	Grass	NWG
Ulmaceae	*Celtis reticulata*	Native	UPL	Tree	NUT
Rosaceae	*Cerasus pensylvanica*	Native	FACU	Tree	NUT
Euphorbiaceae	*Chamaesyce geyeri*	Native	FACU	Forb	NUF
Euphorbiaceae	*Chamaesyce serpyllifolia*	Introduced	FACU	Forb	IUF
Chenopodiaceae	*Chenopodium album*	Introduced	FACU	Forb	IUF
Chenopodiaceae	*Chenopodium berlandieri*	Native	FACU	Forb	NUF
Chenopodiaceae	*Chenopodium glaucum*	Introduced	FAC	Forb	IMF
Poaceae	*Chondrosum gracile*	Native	UPL	Grass	NUG
Asteraceae	*Chyrsothamnus nauseosus*	Native	UPL	Forb	NUF
Asteraceae	*Cirsium vulgare*	Introduced	UPL	Forb	IUF
Ranunculaceae	*Clematis ligusticifolia*	Native	FACU	Forb	NUF
Apaiaceae	*Conium maculatum*	Introduced	FACW	Forb	IWF
Concolvulaceae	*Convolvulus arvensis*	Introduced	FACU	Vine	IUH
Asteraceae	*Conyza canadensis*	Introduced	FACU	Forb	IUF
Rosaceae	*Cotoneaster acutifolius*	Introduced	UPL	Shrub	IUS
Rosaceae	*Crataegus macracantha*	Native	FAC	Tree	NMT
Poaceae	*Critesion jubatum*	Native	FACW	Grass	NWG
Cucurbitaceae	*Cucurbita foetidissima*	Native	FACU	Vine	NUH
Poaceae	*Cynodon dactylon*	Introduced	FACU	Grass	IUG
Boraginaceae	*Cynoglossum officinale*	Introduced	FACU	Forb	IUF
Poaceae	*Dactylis glomerata*	Introduced	FACU	Grass	IUG
Fabaceae	*Dalea purpurea*	Native	FACU	Forb	NUF
Poaceae	*Deschampsia cespitosa*	Native	FACW	Grass	NWG
Brassicaceae	*Descurainia sophia*	Introduced	UPL	Forb	IUF
Poaceae	*Digitaria sanguinalis*	Native	FACU	Grass	NUG

Dispacaceae	*Dipsacus fullonum*	Introduced	FACU	Forb	IUF
Poaceae	*Distichlis stricta*	Native	FACW	Grass	NWG
Poaceae	*Echinochloa crus-galli*	Introduced	FACW	Grass	IWG
Cucurbitaceae	*Echinocystis lobata*	Native	FAC	Vine	NMH
Elaeagnaceae	*Eleaegnus angustifolia*	Introduced	FACU	Tree	IUT
Cyperaceae	*Eleocharis palustris*	Native	OBL	Grass	NWG
Poaceae	*Elymus elymoides*	Native	UPL	Grass	NUG
Poaceae	*Elymus trachycaulus*	Native	FACU	Grass	NUG
Poaceae	*Elytrigia repens*	Introduced	FAC	Grass	IMG
Equisetaceae	*Equisetum arvense*	Native	FAC	Grass	NMG
Poaceae	*Eragrostis cilianensis*	Introduced	FACU	Grass	IUG
Poaceae	*Eragrostis trichodes*	Introduced	UPL	Grass	IUG
Geraniaceae	*Erodium cicutarium*	Introduced	UPL	Forb	IUF
Poaceae	*Festuca arizonica*	Native	UPL	Grass	NUG
Oleaceae	*Forsythia X*	Introduced	UPL	Shrub	IUS
Oleaceae	*Fraxinus pensylvanica*	Native	FACU	Tree	NUT
Fumaraceae	*Fumaria vaillantii*	Introduced	UPL	Forb	IUF
Onagraceae	*Gaura coccinea*	Native	UPL	Forb	NUF
Onagraceae	*Gaura neomexicana*	Native	FAC	Forb	NMF
Fabaceae	*Gleditsia tricanthos*	Introduced	FACU	Tree	IUT
Asteraceae	*Glycyrrhiza lepidota*	Native	FAC	Forb	NMF
Asteraceae	*Grindelia hirsutula*	Native	UPL	Forb	NUF
Asteraceae	*Grindelia squarrosa*	Native	UPL	Forb	NUF
Asteraceae	*Grindelia subalpina*	Native	UPL	Forb	NUF
Asteraceae	*Helianthus annus*	Native	FACU	Forb	NUF
Asteraceae	*Helianthus nuttallii*	Native	FACW	Forb	NWF
Poaceae	*Heteropogon contortus*	Introduced	UPL	Grass	IUG
Poaceae	*Heterostipa comata*	Native	UPL	Grass	NUG
Asteraceae	*Heterotheca canescens*	Native	UPL	Forb	NUF
Equisetaceae	*Hippochaete hyemalis*	Native	FACW	Grass	NWG
Equisetaceae	*Hippochaete laevigata*	Native	FACW	Grass	NWG
Iridaceae	*Iris missouriensis*	Native	FACW	Grass	NWG
Asteraceae	*Iva axillaris*	Native	FAC	Forb	NMF
Hydrangeaceae	*Jamesia americana*	Native	FACU	Shrub	NUS
Juncaceae	*Juncus arcticus*	Native	FACW	Grass	NWG
Juncaceae	*Juncus compressus*	Native	FACW	Grass	NWG
Juncaceae	*Juncus confusus*	Native	FACW	Grass	NWG
Juncaceae	*Juncus longistylis*	Native	FACW	Grass	NWG
Asteraceae	*Lactuca serriola*	Introduced	FAC	Forb	IMF
Asteraceae	*Lactuca tatarica*	Native	FAC	Forb	NMF
Lamiaceae	*Lamium amplexicaule*	Introduced	FAC	Forb	IMF

Boraginaceae	*Lappula marginata*	Native	FAC	Forb	NMF
Asteraceae	*Leucanthemum vulgare*	Introduced	UPL	Forb	IUF
Scrophulariaceae	*Linaria genistifolia*	Introduced	UPL	Forb	IUF
Poaceae	*Lolium perenne*	Introduced	FACU	Grass	IUG
Caprifoliaceae	*Lonicera morrowii*	Introduced	UPL	Shrub	IUS
Caprifoliaceae	*Lonicera tatarica*	Introduced	FACU	Shrub	IUS
Lamiaceae	*Lycopus asper*	Native	OBL	Forb	NWF
Asteraceae	*Lygodesmia juncea*	Native	UPL	Forb	NUF
Primulaceae	*Lysimachia ciliata*	Native	FACW	Forb	NWF
Asteraceae	*Machaeranthera canescens*	Native	FAC	Forb	NMF
Convallariaceae	*Maianthemum stellatum*	Native	FAC	Forb	NMF
Rosaceae	*Malus sylvestris*	Introduced	FACU	Tree	IUT
Malvaceae	*Malva neglecta*	Introduced	UPL	Forb	IUF
Fabaceae	*Medicago lupulina*	Introduced	FACU	Forb	IUF
Fabaceae	*Medicago sativa*	Introduced	UPL	Forb	IUF
Fabaceae	*Melilotus albus*	Introduced	FACU	Forb	IUF
Fabaceae	*Melilotus officinale*	Introduced	FACU	Forb	IUF
Lamiaceae	*Mentha arvensis*	Native	FACW	Forb	NWF
Boraginaceae	*Mertensia lanceolata*	Native	FAC	Forb	NMF
Nyctaginaceae	*Mirabilis multiflora*	Native	FAC	Forb	NMF
Moraceae	*Morus alba*	Introduced	FAC	Tree	IMT
Poaceae	*Muhlenbergia asperifolia*	Native	FACW	Grass	NWG
Aceraceae	*Negundo aceroides*	Native	FAC	Tree	NMT
Solanaceae	*Nicandra physalodes*	Introduced	UPL	Forb	IUF
Onagraceae	*Oenothera villosa*	Native	FAC	Forb	NMF
Asteraceae	*Oligosporus dracunculus*	Native	UPL	Forb	NUF
Cactaceae	*Opuntia macrorhiza*	Native	UPL	Forb	NUF
Rosaceae	*Padus virginiana*	Native	FACU	Shrub	NUS
Poaceae	*Panicum capillare*	Native	FAC	Grass	NMG
Poaceae	*Panicun virgatum*	Native	FAC	Grass	NMG
Vitaceae	*Parthenocissus vitacea*	Native	FACU	Vine	NUH
Poaceae	*Pascopyrum smithii*	Native	FACU	Grass	NUG
Scrophulariaceae	*Penstemon angustifolius*	Native	UPL	Forb	NUF
Scrophulariaceae	*Pentstemon virens*	Native	UPL	Forb	NUF
Polygonaceae	*Persecaria lapathifolia*	Introduced	FACW	Forb	IWF
Polygonaceae	*Persecaria maculosa*	Introduced	FACW	Forb	IWF
Polygonaceae	*Persecaria pensylvanica*	Native	FACW	Forb	NWF
Poaceae	*Phalaroides arundinacae*	Introduced	FACW	Grass	IWG
Poaceae	*Phleum pratense*	Introduced	FACU	Grass	IUG
Solanaceae	*Physalis hederifolia*	Native	UPL	Forb	NUF
Solanaceae	*Physalis virginiana*	Native	UPL	Forb	NUF

Pinaceae	*Picea pungens*	Native	FAC	Tree	NMT
Pinaceae	*Pinus ponderosa*	Native	UPL	Tree	NUT
Plantaginaceae	*Plantago major*	Introduced	FAC	Forb	IMF
Poaceae	*Poa bulbosa*	Introduced	FACU	Grass	IUG
Poaceae	*Poa compressa*	Native	FACU	Grass	NUG
Poaceae	*Poa palustris*	Introduced	FACW	Grass	IWG
Poaceae	*Poa pratensis*	Introduced	FAC	Grass	IMG
Poaceae	*Poa secunda*	Native	FACU	Grass	NUG
Asteraceae	*Podospermum laciniatum*	Introduced	UPL	Forb	IUF
Euphorbiaceae	*Poinsettia dentata*	Introduced	FACU	Forb	IUF
Polygonaceae	*Polygonum aviculare*	Introduced	FACU	Forb	IUF
Polygonaceae	*Polygonum erectum*	Introduced	UPL	Forb	IUF
Poaceae	*Polypogon monspeliensis*	Introduced	FACW	Grass	IWG
Salicaceae	*Populus angustifolia*	Native	FACW	Tree	NWT
Salicaceae	*Populus angustifolia X*	Native	FACW	Tree	NWT
Salicaceae	*Populus deltoides*	Native	FAC	Tree	NMT
Salicaceae	*Populus tremuloides*	Native	FAC	Tree	NMT
Portulacaceae	*Portulaca oleracea*	Introduced	FACU	Forb	IUF
Rosaceae	*Prunus americana*	Native	UPL	Tree	NUT
Fabaceae	*Psoralidium tenuiflorum*	Native	FACU	Forb	NUF
Fagaceae	*Quercus alba*	Introduced	UPL	Tree	IUT
Solanaceae	*Quincula lobata*	Native	UPL	Forb	NUF
Ranunculaceae	*Ranunculus macounii*	Native	OBL	Forb	NWF
Ranunculaceae	*Ranunculus repens*	Introduced	FACW	Forb	IWF
Ranunculaceae	*Ranunculus uncinatus*	Native	FACW	Forb	NWF
Asteraceae	*Ratibida columnifera*	Native	FACU	Forb	NUF
Grossulariaceae	*Ribes aureum*	Native	FAC	Shrub	NMS
Grossulariaceae	*Ribes cereum*	Native	FAC	Shrub	NMS
Grossulariaceae	*Ribes inerme*	Native	FACW	Shrub	NWS
Rosaceae	*Robinia psuedoacacia*	Introduced	FACU	Tree	IUT
Brassicaceae	*Rorippa sinuata*	Native	FACW	Forb	NWF
Rosaceae	*Rosa woodsii*	Native	FACU	Shrub	NUS
Rosaceae	*Rosa.spp ornamental*	Introduced	UPL	Shrub	IUS
Rosaceae	*Rubus idaeus*	Native	FACU	Forb	NUF
Asteraceae	*Rudbeckia hirta*	Native	FACU	Forb	NUF
Polygonaceae	*Rumex altissimus*	Native	FACW	Forb	NWF
Polygonaceae	*Rumex aquaticus*	Native	OBL	Forb	NWF
Polygonaceae	*Rumex crispus*	Introduced	FAC	Forb	IMF
Cupressaceae	*Sabina scopulorum*	Native	UPL	Tree	NUT
Cupressaceae	*Sabinia monosperma*	Native	UPL	Tree	NUT
Salicaceae	*Salix amygdaloides*	Native	FACW	Tree	NWT

Salicaceae	*Salix exigua*	Native	FAC	Shrub	NMS
Salicaceae	*Salix fragilis*	Introduced	FAC	Tree	IMT
Chenopodiaceae	*Salsola australis*	Introduced	FACU	Forb	IUF
Chenopodiaceae	*Salsola collina*	Introduced	FACU	Forb	IUF
Carophyllaceae	*Saponaria officinalis*	Introduced	FACU	Forb	IUF
Cyperaceae	*Schoenoplectus pungens*	Native	OBL	Grass	NWG
Cyperaceae	*Scirpus microcarpus*	Native	OBL	Grass	NWG
Cyperaceae	*Scirpus pallidus*	Native	OBL	Grass	NWG
Scrophulariaceae	*Scrophularia lanceolata*	Native	FACU	Forb	NUF
Poaceae	*Setaria pumila*	Introduced	FAC	Grass	IMG
Poaceae	*Setaria viridis*	Introduced	FACU	Grass	IUG
Carophyllaceae	*Silene noctiflora*	Introduced	FACU	Forb	IUF
Brassicaceae	*Sisymbrium altissimum*	Introduced	FACU	Forb	IUF
Solanaceae	*Solanum americanum*	Native	FACU	Forb	NUF
Solanaceae	*Solanum physalifolium*	Introduced	FACU	Forb	IUF
Solanaceae	*Solanum rostratum*	Introduced	FACU	Forb	IUF
Asteraceae	*Solidago canadensis*	Native	FACU	Forb	NUF
Asteraceae	*Sonchus asper*	Introduced	FAC	Forb	IMF
Malvaceae	*Sphaeralcea coccinea*	Native	FACU	Forb	NUF
Poaceae	*Sporobolos cryptandrus*	Native	FACU	Grass	NUG
Cornaceae	*Swida sericea*	Native	FACW	Shrub	NWS
Caprifoliaceae	*Symphoricarpos occidentalis*	Native	UPL	Shrub	NUS
Asteraceae	*Symphytotrichum lanceolatus*	Native	FAC	Forb	NMF
Tamaricaceae	*Tamarisk chinensis*	Introduced	FAC	Shrub	IMS
Asteraceae	*Taraxacum officinale*	Introduced	FACU	Forb	IUF
Thalictraceae	*Thalictrum dasycarpum*	Native	FAC	Forb	NMF
Fabaceae	*Thermopsis divaricarpa*	Native	FAC	Forb	NMF
Poaceae	*Thinopyrum intermedium*	Introduced	FACU	Grass	IUG
Euphorbiaceae	*Tithymalus uralensis*	Introduced	FACU	Forb	IUF
Anacardiaceae	*Toxicodendron rydbergii*	Native	FACU	Forb	NUF
Melanthiaceae	*Toxicoscordion veneosum*	Native	FAC	Forb	NMF
Asteraceae	*Tragopogon dubius*	Introduced	UPL	Forb	IUF
Zygophyllaceae	*Tribulus terrestris*	Introduced	UPL	Forb	IUF
Fabaceae	*Trifolium pratense*	Introduced	FACU	Forb	IUF
Fabaceae	*Trifolium repens*	Introduced	FACU	Forb	IUF
Poaceae	*Triticum aestivum*	Introduced	UPL	Grass	IUG
Typhaceae	*Typha angustifolia*	Native	OBL	Grass	NWG
Typhaceae	*Typha latifolia*	Native	OBL	Grass	NWG
Ulmaceae	*Ulmus pumila*	Introduced	UPL	Tree	IUT
Urticaceae	*Urtica gracilis*	Native	FAC	Forb	NMF
Scrophulariaceae	*Verbascum thapsus*	Introduced	UPL	Forb	IUF

Verbenaceae	*Verbena bracteata*	Introduced	FACU	Forb	IUF
Scrophulariaceae	*Veronica americana*	Native	OBL	Forb	NWF
Scrophulariaceae	*Veronica catenata*	Native	OBL	Forb	NWF
Caprifoliaceae	*Viburnum lentago*	Introduced	FACU	Shrub	IUS
Fabaceae	*Vicea americana*	Native	FACU	Vine	NUH
Asteraceae	*Virgulus campestris*	Native	FAC	Forb	NMF
Vitaceae	*Vitus riparia*	Native	FAC	Vine	NMH
Asteraceae	*Xanthium strumarium*	Introduced	FAC	Forb	IMF
Agavaceae	*Yucca glauca*	Native	UPL	Forb	NUF
Poaceae	*Zea maize*	Introduced	UPL	Grass	IUG

Table 8.4: List of aquatic macroinvertebrate taxa collected from streams and canals in Larimer and Weld Counties, Colorado during April-August 2014 and 2015 using taxonomy from Merritt et al. (2008).

GENUS	FAMILY	TRIBE	GENUS	FAMILY	TRIBE
Ablabesmyia	Tanypodinae	Pentaneuriini	*Erioptera*	Tipulidae	
Acentrella	Baetidae		*Eukiefferiella*	Chironominae	Orthocladiinae
Aeshna	Aeshnidae		*Ferrissia*	Ancylidae	
Agabus	Dytiscidae		*Gerris*	Gerridae	
Ameletus	Ameletidae		*Glossiphoniidae*	Glossiphoniidae	
Anacaena	Hydrophilidae		*Glyptotendipes*	Chironominae	Chironomini
Anopheles	Culicidae		*Gyrinus*	Gyrinidae	
Antocha	Tipulidae		*Haliplus*	Haliplidae	
Aquarius	Gerridae		*Helisoma*	Planorbidae	
Arctopsyche	Hydropsychidae		*Hemerodromia*	Empididae	
Atherix	Athericidae		*Heptagenia*	Heptageniidae	
Attenella	Ephemerellidae		*Hetaerina*	Calopterygidae	
Baetis	Baetidae		*Heterolemus*	Elmidae	
Belostoma	Belostomatidae		*Hexatoma*	Tipulidae	
Berosus	Hydrophilidae		*Hydrobaenus*	Chironominae	Orthocladiinae
Bezzia	Ceratopogonidae		*Hydrocanthus*	Noteridae	
Brachycentrus	Brachycentridae		*Hydrochara*	Hydrophilidae	
Brillia	Chironominae	Orthocladiinae	*Hydropsyche*	Hydropsychidae	
Caenis	Caenidae		*Isoperla*	Perlodidae	
Callibaetis	Baetidae		*Isotomidae*	Isotomidae	
Cardiocladius	Chironominae	Orthocladiinae	*Labiobaetis*	Baetidae	
Chaetocladius	Chironominae	Orthocladiinae	*Laccobius*	Hydrophilidae	
Cheumatopsyche	Hydropsychidae		*Lepidostoma*	Lepidostomatidae	
Chironomus	Chironominae	Chironomini	*Leptohyphes*	Leptohyphidae	
Chrysops	Tabanidae		*Libellula*	Libellulidae	
Cinygmula	Heptageniidae		*Limnephilus*	Limnephilidae	
Claassenia	Perlidae		*Limnophila*	Tipulidae	
Cladotanytarsus	Chironominae	Tanytarsini	*Limnophora*	Muscidae	
Crangonyx	Crangonyctidae		*Limnoporus*	Gerridae	
Cricotopus	Chironominae	Orthocladiinae	*Limonia*	Tipulidae	
Cryptochironomus	Chironominae	Chironomini	*Liodessus*	Dytiscidae	
Culex	Culicidae		*Lirceus*	Asellidae	
Culicoides	Ceratopogonidae		*Lumbriculidae*	Lumbriculidae	
Culiseta	Culicidae		*Lymnaea*	Lymnaeidae	
Diamesa	Diamesinae		*Macrostemum*	Hydropsychidae	
Dicrotendipes	Chironominae	Chironomini	*Mesovelia*	Mesoveliidae	
Dixa	Dixidae		*Micropsectra*	Chironominae	
Doncricotopus	Chironominae	Orthocladiinae	*Microtendipes*	Chironominae	
Drunella	Ephemerellidae		*Microvelia*	Veliidae	
Dubiraphia	Elmidae		*Naididae*	Naidadae	
Dugesia	Planariidae		*Nanocladius*	Chironominae	Orthocladiinae
Empididae	Empididae		*Narpus*	Elmidae	
Endochironomus	Chironominae	Chironomini	*Nectopsyche*	Leptoceridae	
Enallagma	Coenagrionidae		*Notonecta*	Notonectidae	
Enochrus	Hydrophilidae		*Odontomyia*	Stratiomyidae	
Epeorus	Heptageniidae		*Oecetis*	Leptoceridae	
Ephemerella	Ephemerellidae		*Ophiogomphus*	Gomphidae	
Ephydridae	Ephydridae		*Optioservus*	Elmidae	

GENUS	FAMILY	TRIBE
Orconectes	Cambaridae	
Ormosia	Tipulidae	
Orthocladius	Chironominae	Orthocladiinae
Paracladius	Chironominae	Orthocladiinae
Paracricotopus	Chironominae	Orthocladiinae
Paracymus	Hydrophilidae	
Parakiefferiella	Chironominae	Orthocladiinae
Paraleptophlebia	Leptophlebiidae	
Parametriocnemus	Chironominae	Orthocladiinae
Paratanytarsus	Chironominae	Tanytarsini
Paratendipes	Chironominae	Chironomini
Pedicia	Tipulidae	
Peltodytes	Haliplidae	
Phaenospectra	Chironominae	Chironomini
Physa	Physidae	
Planorbidae	Planorbidae	
Polypedilum	Chironominae	Chironomini
Procladius	Tanypodinae	
Prodiamesa	Prodiamesinae	
Protonerus	Noteridae	
Protoplasa	Tanyderidae	
Psectrocladius	Chironominae	Orthocladiinae
Psuedocloeon	Baetidae	
Psychoglypha	Limnephilidae	
Pteronarcella	Pteronarcyidae	
Radotanypus	Chironominae	Macropelopiini
Rhagovelia	Veliidae	
Rhantus	Dytiscidae	
Rheocricotopus	Chironominae	Orthocladiinae
Rheotanytarsus	Chironominae	Tanytarsini
Rhithrogena	Heptageniidae	
Serratelia	Ephemerellidae	
Sigara	Corixidae	
Simulium	Simuliidae	
Siphlonurus	Siphlonuridae	
Skwala	Perlodidae	
Stenochironomous	Chironominae	Chironomini
Stenonema/ Mccaffertium	Heptageniidae	
Stictochironomous	Chironominae	Chironomini
Stictotarsus	Dytiscidae	
Suwallia	Chloroperlidae	
Sweltsa	Chloroperlidae	
Sympetrum	Libellulidae	
Synorthocladius	Chironominae	Orthocladiinae
Tabanus	Tabanidae	
Tanypus	Chironominae	Tanypodini
Tanytarsus	Chironominae	Tanytarsini
Thienemannimyia	Chironominae	Pentaneuriini
Tipula	Tipulidae	
Trichocorixa	Corixidae	
Tricorythodes	Leptohyphidae	
Triznaka	Chloroperlidae	
Tropisternus	Hydrophilidae	
Tvetenia	Chironominae	Orthocladiinae
Zaitzevia	Elmidae	

Table 8.5: List of aquatic macroinvertebrate taxa included in functional group assessment of macroinvertebrate communities at canal and stream sites in Larimer and Weld Counties, Colorado. Information on life history, mobility, morphology and ecology from Poff et al (2006) and tolerance values (https://thewatershed.org/pdf/Science/Resources/Hilsenhoff%20FTV.pdf).

Taxa	Functional Group	Voltinism	Development	Synchronization	Life Span Adult	Adult Exit	Desiccation	Female Dispersal	Adult Flight	Drift	Crawl Rate	Swim Ability	Attachment	Armoring	Shape	Respiration	Size	Rheophiliy	Thermal Preference	Habitat	FFG	Tolerance
Ablabesmyia	D	2	1	2	1	2	1	2	1	3	1	1	1	1	2	2	1	2	2	3	4	8
Acentrella	F	3	1	1	1	1	1	1	1	2	1	3	1	1	1	2	1	2	2	5	1	4
Aeshna	B	1	2	2	3	1	1	2	2	1	3	3	1	2	2	2	3	2	2	2	4	5
Agabus	B	1	2	1	3	2	1	2	2	1	3	3	1	3	1	3	2	2	2	5	4	8
Ameletus	F	2	1	2	2	1	1	1	1	1	3	3	1	1	1	2	1	2	1	5	1	0
Arctopsyche	C	2	2	2	2	1	1	1	1	1	2	1	2	1	2	2	2	3	1	4	2	1
Athrix	A	2	2	2	1	1	1	1	1	1	2	1	1	2	2	1	2	2	2	3	4	2
Attenella	F	2	2	2	1	1	1	1	1	1	2	2	1	1	2	2	2	3	2	4	1	2
Baetis	F	3	1	1	1	1	1	1	1	3	1	3	1	1	1	2	1	2	2	5	1	5
Belostoma	E	2	1	2	3	2	2	2	2	1	3	3	1	3	1	3	3	1	3	2	4	8
Brachycentrus	C	2	2	2	2	1	1	1	1	1	1	1	3	3	2	2	2	3	2	4	2	1
Caenis	F	3	2	2	1	1	1	1	1	1	2	2	1	1	2	2	1	1	3	3	1	7
Callibaetis	F	3	1	1	1	1	1	1	1	1	2	3	1	1	1	2	1	1	2	5	1	9
Cardiocladius	D	2	1	2	1	2	1	2	1	3	1	1	1	1	2	2	1	2	1	1	1	5
Chironomus	D	3	1	2	1	2	1	2	1	3	1	1	1	1	2	2	1	1	2	1	1	10
Cinygmula	F	2	1	2	1	1	1	1	1	2	2	2	1	1	1	2	1	2	1	4	3	4
Claassenia	B	1	3	1	2	1	1	2	2	1	3	2	1	2	1	2	3	3	1	4	4	3
Cricotopus	D	2	1	2	1	2	1	2	1	3	1	1	1	1	2	2	1	2	1	1	1	7
Cryptochironomus	D	3	1	2	1	2	1	2	1	3	1	1	1	1	2	2	1	1	2	1	1	8
Diamesa	D	2	1	2	1	2	1	2	1	3	1	1	1	1	2	2	1	2	1	3	1	5
Dicrotendipes	D	3	1	2	1	2	1	2	1	3	1	1	1	1	2	2	1	1	2	1	1	8
Drunnella	F	2	2	2	1	1	1	1	1	1	2	2	1	1	2	2	2	2	2	4	3	0
Dubiraphia	A	1	3	1	3	1	2	1	1	2	2	1	1	2	2	1	1	2	2	4	1	6
Empididae	A	2	2	2	1	2	1	1	1	1	2	1	1	1	2	1	2	2	2	3	4	6
Endochironomus	D	3	1	2	1	2	1	2	1	3	1	1	1	1	2	2	1	1	2	1	1	10
Epeorus	F	2	1	2	1	1	1	1	1	1	2	2	1	1	1	2	2	2	1	4	1	0
Ephemerella	F	2	2	2	1	1	1	1	1	2	2	2	1	1	2	2	2	2	2	4	1	1
Eukiefferiella	D	2	1	2	1	2	1	2	1	3	1	1	1	1	2	2	1	2	1	1	1	8
Glyptotendipes	D	3	1	2	1	2	1	2	1	3	1	1	1	1	2	2	1	1	2	1	1	10
Heptagenia	F	2	1	2	1	1	1	1	1	2	2	2	1	1	1	2	2	2	1	4	3	4
Heterolemus	A	1	3	1	3	1	2	1	1	2	2	1	1	2	2	1	1	2	2	4	1	4
Hexatoma	A	2	2	1	2	1	2	1	1	1	2	1	1	1	2	2	2	2	2	3	5	2
Hydrobaenus	D	2	1	2	1	2	1	2	1	3	1	1	1	1	2	2	1	2	1	1	1	8
Hydropsyche	C	2	2	2	2	2	1	2	2	2	2	1	2	1	2	2	2	3	2	4	2	4
Isoperla	A	2	2	2	1	1	2	1	1	2	3	2	1	2	2	1	2	2	2	4	4	2
Limonia	A	2	2	1	2	1	2	1	1	1	2	1	1	1	2	2	2	2	2	3	5	6
Maccafertium	F	2	1	1	2	1	1	1	1	2	2	2	1	1	1	2	2	2	2	4	3	2
Macrostemum	C	2	2	2	2	1	1	2	2	2	2	1	2	1	2	2	2	3	3	4	2	6
Micropsectra	D	3	1	2	1	2	1	2	1	3	1	1	1	1	2	2	1	1	2	1	1	7
Microtendipes	D	3	1	2	1	2	1	2	1	3	1	1	1	1	2	2	1	1	2	1	1	6
Narpus	A	1	3	1	3	1	2	1	1	2	2	1	1	2	2	1	1	2	2	4	1	4
Nectopsyche	C	2	2	2	2	1	1	2	1	1	1	2	2	2	2	2	2	2	2	2	3	3
Optioservus	A	1	3	1	3	1	2	1	1	2	2	1	1	2	2	1	1	2	2	4	1	4
Ormosia	A	2	2	1	2	1	2	1	1	1	2	1	1	1	2	2	2	2	2	3	5	3
Orthocladius	D	2	1	2	1	2	1	2	1	3	1	1	1	1	2	2	1	2	1	1	1	6

Paracricotopus	D	2	1	2	1	2	1	2	1	3	1	1	1	1	2	2	1	2	1	1	1	5
Paraleptophlebia	F	2	1	1	1	1	1	1	1	2	2	2	1	1	1	2	2	2	2	5	1	4
Paratendipes	D	3	1	2	1	2	1	2	1	3	1	1	1	1	2	2	1	1	2	1	1	8
Pedicia	A	2	2	1	2	1	2	1	1	1	2	1	1	1	2	2	2	2	2	3	5	1
Polypedilum	D	3	1	2	1	2	1	2	1	3	1	1	1	1	2	2	1	1	2	1	1	6
Procladius	D	2	1	2	1	2	1	2	1	3	1	1	1	1	2	2	1	2	2	3	4	9
Psuedocloeon	F	3	1	1	1	1	1	1	1	3	2	2	1	1	1	2	1	2	2	5	1	4
Pteronarcella	A	1	2	2	2	1	1	1	1	1	2	1	1	2	2	2	2	2	2	4	5	0
Rhagovelia	E	3	1	2	3	2	2	1	1	1	3	3	1	1	2	3	1	2	2	6	4	6
Rheocricotopus	D	2	1	2	1	2	1	2	1	3	1	1	1	1	2	2	1	2	1	1	1	6
Rheotanytarsus	D	3	1	2	1	2	1	2	1	3	1	1	1	1	2	2	1	1	2	1	1	6
Rhthrogna	F	2	1	2	1	1	1	1	1	2	2	2	1	1	1	2	2	2	1	4	1	0
Psectrocladius	D	2	1	2	1	2	1	2	1	3	1	1	1	1	2	2	1	2	1	1	1	8
Seratella	F	2	2	2	1	1	1	1	1	2	2	2	1	1	2	2	2	2	2	4	1	2
Simulium	C	3	1	2	1	1	1	1	2	2	2	1	2	1	2	1	1	3	2	4	2	6
Siphlonurus	F	2	2	1	1	1	1	1	1	2	3	3	1	1	1	2	2	1	2	5	1	7
Stenochironomous	D	3	1	2	1	2	1	2	1	3	1	1	1	1	2	2	1	1	2	1	1	9
Synorthocladius	D	2	1	2	1	2	1	2	1	3	1	1	1	1	2	2	1	2	1	1	1	2
Tanytarsus	D	3	1	2	1	2	1	2	1	3	1	1	1	1	2	2	1	1	2	1	1	6
Thienemannimyia	D	2	1	2	1	2	1	2	1	3	1	1	1	1	2	2	1	2	2	3	4	7
Tipula	A	2	2	1	2	1	2	1	1	1	2	1	1	1	2	2	2	2	2	3	5	4
Trichocorixa	E	3	1	2	3	2	2	2	2	1	3	3	1	3	1	3	1	1	2	5	3	8
Tricorythodes	F	2	1	1	1	1	1	1	1	3	2	2	1	1	2	2	1	1	2	3	1	4
Tvetenia	D	2	1	2	1	2	1	2	1	3	1	1	1	1	2	2	1	2	1	1	1	5
Zaitzevia	A	1	3	1	3	1	2	1	1	2	2	1	1	2	2	1	1	2	2	4	1	4