THESIS

A COMPARATIVE ANALYSIS OF WETLAND AND RIPARIAN VEGETATION ON BUREAU OF LAND MANAGEMENT LAND IN THE WESTERN US.

Submitted by

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ABSTRACT

A COMPARATIVE ANALYSIS OF WETLAND AND RIPARIAN VEGETATION ON BUREAU OF LAND MANAGEMENT LAND IN THE WESTERN US

In 2011, the BLM deployed its first of three Assessment, Inventory, and Monitoring (AIM) programs as a large-scale, standardized ecological monitoring effort across the agency's land. The first two programs, known as Terrestrial AIM and Lotic AIM, were designed to sample all terrestrial and river ecosystems throughout the landscape. In 2019, the agency piloted its third AIM program, specifically targeting riparian areas and wetlands. This study addressed two main questions: 1) How do wetland and riparian areas sampled with the Terrestrial AIM program compare to those sampled with the Riparian and Wetland (R&W) AIM program, and 2) What are the drivers of plant community composition of the wetlands and riparian areas sampled on BLM land? I developed a set of criteria to identify sites sampled with Terrestrial AIM that had characteristics of wetlands or riparian areas. I then compared vegetation cover, floristic quality metrics, and species composition using nonmetric multidimensional scaling (NMDS) to those sites sampled with R&W AIM. R&W AIM sites had much greater foliar cover, hydrophytic species cover, and perennial cover, but Terrestrial sites had slightly higher floristic metric values. I similarly analyzed the R&W sites on their own, incorporating wetland-specific data that is collected with the new program. I found that sites that met the criteria to be classified as wetlands in the Terrestrial data were a distinct population from the sites sampled with R&W AIM. The main drivers of plant community composition among sites sampled with R&W AIM were elevation and the distribution of surface water, but impacts of grazing were also

apparent. All sites assessed by both AIM programs had floristic quality metrics characteristic of highly impacted wetland systems. This study indicates the value of the new R&W AIM program for its ability to perform wetland-specific ecological monitoring, provide valuable data on the health of wetlands, and provide baseline condition that can help guide land management practices into the future.

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1. CHAPTER ONE – INTRODUCTION

Wetlands provide numerous unique ecosystem services on both regional and global scales. Regionally, they provide critical support to biodiversity (Lind et al., 2019), provide flood and wildfire mitigation, extend water availability both seasonally and spatially (Wohl et al., 2021), and provide human recreational opportunities. Globally, wetlands have received increasing attention for their carbon storage as greenhouse gases continue to increase and impact the livelihood of humans and biodiversity throughout the world (W. J. Mitsch et al., 2013).

Despite the critical services they provide, wetlands are highly sensitive to novel disturbance regimes and have historically been exploited by humans for the access to water and fertile land (Larsen, 2019; Maltby, 2022). Dahl (1990) estimated that roughly 53% of wetlands in the lower 48 states have been lost since US colonization, predominantly due to agriculture. Groundwater extraction has also increased as human populations have continued to grow around the world, lowering water tables and thus reducing water inputs that are critical to the existence of wetlands (de Graaf et al., 2019). Climate change is putting further pressure on these systems, as climatic patterns become more variable and extreme, and temperatures and evaporation rates increase (R. G. Taylor et al., 2013).

In recent years, as the value of wetlands becomes more widely understood while water simultaneously becomes increasingly scarce in many parts of the world, there has been an increase in demand for information about the status, health and trend of wetland systems (Dahl & Watmough, 2007; Maltby, 2022). As a result, regional and national scale monitoring programs have been developed around the world (Clarkson et al., 2003; Dahl & Watmough, 2007; Funkenberg et al., 2014; Yussuf et al., 2023).

The United States alone has developed numerous monitoring programs targeted specifically towards monitoring the status and trend of wetlands. For instance, the US Fish and Wildlife Service (USFWS) began mapping wetlands in 1974 in order to track wetland quantity throughout the nation (Dahl, 2011; Tiner, 2017). However, it soon became clear that knowing the quantity of wetlands throughout the nation meant little without knowledge of the ecological health of those wetlands (Dahl & Watmough, 2007). In response, the US Environmental Protection Agency (EPA) initiated the National Wetland Condition Assessment (NWCA) in 2011 as a nation-wide wetland monitoring program (Lomnicky et al., 2019). Similarly, the US Forest Service developed protocols for assessment of ground water dependent systems in 2012 (Coles-Ritchie et al., 2012, 2022) and a riparian monitoring protocol in 2017 (Merritt et al., 2017). The National Park Service has also developed some wetland- and riparian-specific monitoring protocols throughout the 2010s, though they are on smaller, regional scales (Gage et al., 2018; Schweiger et al., 2015; Starkey et al., 2011).

The Bureau of Land Management is the latest agency to develop a monitoring program specifically for riparian and wetland areas. Prior to implementing the Riparian and Wetland Assessment, Inventory, and Monitoring (R&W AIM) program in 2019, the agency had no quantitative, standardized monitoring program with which to sample wetlands on their land (Reynolds et al., 2021). As a result, little was known about the health and status of wetlands on BLM land.

Additionally, despite the growing momentum of wetland-specific monitoring throughout the world, little research has been done on the necessity of such programs. For instance, the BLM implemented two other AIM programs prior to their development of the R&W program: a Terrestrial program and an Aquatic program, which focused on monitoring upland systems and stream systems, respectively. The Terrestrial program is a landscape-scale program that uses random site selection to ensure that all BLM land is being sampled equally (Herrick et al., 2021). In theory, such a sample design would inevitably sample wetlands in proportion to their occurrence on the landscape. However, prior to this study, no analysis had been conducted on the quantity or quality of wetlands that may have been sampled by the program.

The BLM's AIM program has been implemented in a large and efficient manner, providing extensive data of Terrestrial systems throughout the western U.S. In conjunction with their new R&W program, the AIM data provides a unique opportunity to assess the presence and ecological health of wetland and riparian areas in an unprecedented manner.

2. CHAPTER TWO – A COMPARATIVE ANALYSIS OF WETLAND AND RIPARIAN VEGETATION ON BUREAU OF LAND MANAGEMENT LAND IN THE WESTERN US

2.1 Introduction

The impacts from altered climate patterns and increasing human development are putting critically important freshwater dependent systems, such as wetlands and riparian areas, at risk in the West. These aquatic ecosystems have been well demonstrated to provide valuable, multifaceted benefits, many of which are more pronounced in semi-arid and arid climates due to their scarcity. For instance, while wetlands and riparian areas are known as biodiversity hot spots throughout the world (Lind et al., 2019), they are particularly valuable for supporting biodiversity and a number of threatened species in the semi-arid west, where they have been estimated to support 60-80% of biodiversity (Belsky et al., 1999; Lemly et al., 2000; Patten, 1998). Along with biodiversity support, these systems also provide many functions important to humans. Riparian-wetland complexes, or wetlands that form due to channel complexity in floodplains, have been found to increase resiliency to drought, fires and erosion, and can increase the residence time of water and nutrients in alluvial systems (Wohl et al., 2021). In the Intermountain West, where water is driven by snow melt that peaks in the spring, networks of wetlands and riparian areas serve as sponges that can both reduce the effects of flooding and store water for slow release throughout the relatively dry summer and fall seasons (Hubert, 2004; Kudray & Schemm, 2008). Finally, wetlands are frequently touted for their ability to store large amounts of carbon in their soil (W. J. Mitsch et al., 2013; Nahlik & Fennessy, 2016), an issue that is becoming increasingly pertinent as atmospheric CO_2 concentrations continue to climb.

Unfortunately, wetland systems are also highly sensitive to novel disturbance regimes and climatic variability (Larsen, 2019; Schlesinger et al., 1990; Thomas et al., 1979). It is estimated that more than 90% of wetlands in some parts of the semi-arid west have been lost since US colonization (Dahl, 2011; Lemly et al., 2000), seriously impacting a region where wetlands only accounted for a few percent of the landscape to begin with (Dahl, 1990). Extirpation of beavers and human-induced landscape modification have not only led to a loss of riparian-wetland complexes, but have also resulted in oversimplified, channelized and incised river systems (Wohl et al., 2021). Channel incision and human extraction of groundwater have lowered water tables, leading to disconnection of ground water and further loss of wetlands (de Graaf et al., 2019; R. G. Taylor et al., 2013). In 2006, the Environmental Protection Agency (EPA) estimated that 49% of the riparian areas in arid regions of the Western U.S. had poor to fair vegetation cover and 84% of riparian areas showed signs of medium to high human-influenced disturbance (U.S. Environmental Protection Agency, 2006). Decreases in wetland-adapted vegetation in these systems can increase rates of erosion, lead to further incision, and decrease the functionality of wetlands and riparian-wetland complexes (Belsky et al., 1999).

In the Intermountain West, where wetlands are particularly vulnerable due to aridification (Overpeck & Udall, 2020), rapidly growing populations, and long-term, intensive livestock grazing, management towards resilient and functioning ecosystems on public lands could result in significant improvements at an immense geographic scale. The Bureau of Land Management (BLM), for instance, manages more land than any other federal agency in the United States, with oversight of 248 million acres of land (Havlick et al., 2015; Vincent et al., 2017). Just shy of 100% of BLM land falls in Alaska and semi-arid and arid parts of the contiguous western states (Vincent et al., 2017), making up about 23% of the total land area in those parts of the country

(Havlick et al., 2015). As a result, appropriate, large-scale management of wetland systems has the potential to make an incredible difference. In recent years, the agency has begun to grow its efforts towards ecological conservation and specifically the protection and restoration of wetlands and riparian areas (Silverman et al., 2019), making such a difference a real possibility.

However, to date, no one has conducted a robust, large-scale analysis of existing data on BLM lands that would be necessary to inform a cohesive wetland management plan (Reynolds et al., 2021). Past research conducted on wetland, riparian, and mesic areas on public lands have predominantly consisted of case studies of individual watersheds or small regions (Donnelly et al., 2016; Silverman et al., 2019). The BLM lacked any standardized, quantitative resource monitoring efforts until the early 21st century (Taylor et al., 2014). Consequently, assessments were conducted by land managers on small scales with a variety of monitoring protocols and data were stored in unstandardized formats in offices at different jurisdictional levels, preventing comparison of data across time or space (Taylor, 2014). To address this issue and to meet compliance under the Federal Land Policy and Management Act of 1976, the BLM implemented the Assessment, Inventory, and Monitoring (AIM) program in 2011 (Toevs et al., 2011; U.S Department of Interior & Bureau of Land Management, 2022). The AIM program was designed to collect data that can be applied to multiple scales, a diversity of management goals, and jurisdictions across the agency (Toevs et al., 2011). Since then, the program has successfully been deployed, as of 2022 more than 55,000 sites had been sampled throughout the West and Alaska.

The AIM program is intended to address the BLM's goals of promoting multiple uses of the land it manages, while ensuring the future health of its watersheds, wildlife, and natural resources (BLM, 2001). As part of their Fundamentals of Rangeland Health guidelines, the agency is required to assess, inventory, and monitor those resources to ensure their protection or improvement over time (BLM, 2001). The AIM program's large scale, standardized methods can provide data that can be used to assess quantity, status, and trends of resources, such as wetlands, at regional, state, or national scales. The quantitative data can then be used for the development and prioritization of adaptive land management practices at numerous scales (Toevs et al., 2011).

The AIM strategy was originally developed with the intention to sample a diversity of land-based systems with a single sampling protocol. However, after a few years of sampling, the BLM acknowledged a need to develop a separate protocol for sampling riparian and wetland areas, particularly areas beyond the immediate banks of rivers and streams, which are sampled through other BLM sampling programs (BLM, 2021; Burton et al., 2011). The BLM subsequently partnered with wetland ecologists at the Colorado Natural Heritage Program (CNHP) in 2017 to further develop their ideas into a draft sampling protocol. This Riparian and Wetland ("R&W" hereafter) AIM sampling protocol is predominantly modeled after the Terrestrial protocol, but is specifically tailored to sampling a wide variety of vegetated systems that are influenced by consistent hydrologic regimes, such as wetlands, mesic areas, and the fringes of lakes and rivers. The R&W program seeks to assess vegetation diversity and composition, hydrologic drivers, soil characteristics, water quality, and the use of wetlands by humans, livestock, and wildlife. Fortunately, due to significant similarities between the two protocols, much of their data is directly comparable, though, prior to this study, no analyis of the data from the R&W program had been completed. CNHP conducted three pilot years of sampling from 2019 to 2021 to ensure the protocol would meet the BLM's goals and AIM guidelines (Reynolds et al., 2021).

Between 2019 and 2020, data was collected across BLM land in Colorado, Utah, and Idaho and provides the first quantitative, standardized, and large-scale data on wetlands on BLM land to date. However, because early iterations of the Terrestrial AIM program were designed with the intention to include riparian and/or wetland systems, it is likely that Terrestrial crews sampled sites that fall into the target population of the new R&W program. Despite this, no one had conducted a formal search for such sites, so it was not known how many of these sites had been sampled or how they might be characterized. Specifically, the two protocols collect similar information on plant communities.

Plant communities can provide valuable information about the health and functions of wetlands because they both react to and drive conditions in wetland and riparian areas. Plant communities drive ecosystem functions in most systems via net primary production and nutrient availability (Avolio et al., 2019; Britson et al., 2016). In riparian systems, vegetation can also influence stream flow regimes and increase beneficial riparian-corridor complexity by providing resistance to water flow and erosion and increasing sediment deposition (Han & Brierley, 2020; Heffernan, 2008; Larsen, 2019; Wohl et al., 2021). However, plant communities in wetlands are also sensitive to disturbances and can therefore serve as indicators of unfavorable conditions. For instance, emergent vegetation is very sensitive to increases in erosion (Larsen, 2019), changes to the depth of the water table (Hammersmark et al., 2008; Silverman et al., 2019), and grazing intensity (Cubley et al., 2021).

Floristic metrics have been shown to be correlated with disturbance in wetlands (Jones, 2005; Mack, 2004; Rocchio et al., 2007; Smith et al., 2020). For instance, C values, or coefficients of conservatism, are a floristic metric that represents the ability of a given plant species to tolerate disturbance (Swink & Wilhelm, 1979). C values of zero indicate low quality, weedy species that are likely to occur in highly disturbed areas, while high values are assigned to rare, high-quality species that tend to only occur in unaltered areas (Smith et al., 2020; Swink & Wilhelm, 1979). Mean C values have been found to be negatively correlated with disturbance in wetlands (Rocchio et al., 2007; Smith et al., 2020). Similarly, the floristic quality assessment index (FQAI) was developed as a way to capture more of the complexity that exists in plant communities in a quantitative and comparable way by incorporating both mean C value and species richness of a site (Andreas & Lichvar, 1995; Wilhelm & Masters, 1995). These metrics have been shown to vary predictably with intensity of disturbance to wetlands and can therefore be used as proxies for degree of impact due to grazing, pollution, or agriculture (Mack, 2004).

Prior to this study, there have been no large-scale analyses of wetlands on BLM land (Reynolds et al., 2021). Thus, there was no baseline for the status of wetland systems, what variables are influencing their plant community composition, or how they may be reacting to disturbance.

In this study, I address two questions in order to establish baseline information on wetland, riparian, and mesic areas from pilot data collected by the Wetland AIM program:

1. **How do the wetland and riparian areas sampled with the Terrestrial AIM program compare to those sampled with the Riparian and Wetland AIM program?**
2. **What are the drivers of plant community composition of wetlands and riparian areas on BLM land?**

2.2 Methods

2.2.1 Datasets

The Terrestrial AIM database used in this study included data collected from 2011 to 2020. It contained a total of 36,232 sites located throughout BLM land in all contiguous western states (states including and west of the continental divide), plus North Dakota, South Dakota, and

Alaska. The data included sites that were randomly selected as well as sites that were handpicked by regional land managers. The Riparian and Wetland (R&W) AIM data used in this study was pilot data that was collected in Colorado, Utah, and Idaho in 2019 and 2020. There were a total of R&W 133 sites, also selected either randomly or handpicked by local land managers.

Both datasets include GPS location, elevation, aspect, and soil data for each site, as well as similar line-point-intercept (LPI) data that included cover of vegetative species, plant litter, rocks, soil surface, and water. The standard LPI methodology for both programs includes measurements every half meter along three 25-meter transects (Herrick et al., 2021; Reynolds et al., 2021). I excluded any sites that did not have exactly three transects and 50 measurements per transect, resulting in a total of 115 R&W sites and 250 Terrestrial sites.

The R&W data additionally had wetland specific data for each site. This included wetland classifications within three different classification systems: a colloquial classification system referred to as "Wetland type," Cowardin Classification, and hydrogeomorphic classification (Reynolds et al., 2021). The wetland type classification system was adapted from CNHP's Ecological Systems of Colorado and was developed with the intention of being readily identifiable via both remote imagery and field visits (Decker et al., 2020; Reynolds et al., 2021). I consolidated sites classified as the wetland type "vegetated drainageway" with those classified as "spring/seep" because they were the only two groups without statistically significant differences in species composition as indicated by the PERMANOVA. The Cowardin Classification system has been the industry standard for the National Wetland Inventory mapping system since 1976 and was developed as a way to categorize and compare wetlands on a national scale (Cowardin, 1979; Federal Geographic Data Committee, 2013). I consolidated Cowardin classifications to palustrine emergent (PEM) and palustrine scrub-shrub (PSS). While there were nine sites that were categorized as non-wetland riparian rather than palustrine, these sites were excluded from Cowardin Classification analyses for simplicity. The HGM classifications were based off of the geomorphic location of a wetland, its main water source, and the energy and direction of water flow (Brinson, 1993).

The R&W data also included predominant water sources, “extent surface water,” which is a quantitative estimate of surface water coverage throughout an entire plot, percent of graminoids grazed along each transect, and soil alteration, or the number of hoof prints present within 30-centimeters of each transect. “Extent surface water” is distinguishable from “total water cover,” in that “total water cover” is a measurement of the number of LPI pin drops that hit surface water along the transects. Field technicians also identified the major soil type for each site and documented the presence of applicable hydric soil indicators. Major soil types were classified as clayey/loamy, sandy, or organic, but I consolidated them to just mineral and organic. Hydric soil indicators are similarly recorded as specific indicators, but to reduce noise, I consolidated them into the functional groups of Fe/Mn reduction/accumulation, organic matter accumulation, sulfate reduction, and none.

2.2.2 Terrestrial AIM Site Selection

I developed a set of two main criteria to identify and isolate Terrestrial sites with wetland or riparian characteristics that would likely have also been assessed by the R&W program. The first criteria aligns directly with the R&W protocol, which defines target sites as areas that have greater than 50% relative hydrophytic species cover (species rated as Obligate (OBL), Facultative Wetland (FACW), or Facultative (FAC), Lichvar et al., 2012; Reynolds et al., 2021).

However, in order to capture sites that demonstrate other evidence of hydrologic influence, I developed a second criteria for identifying wetlands as sites that had between 25 and 50% relative hydrophytic species cover **and** met one of the following secondary criteria:

1. intersected with a National Wetland Inventory (NWI) polygon,
2. the plot center was within 50 meters of a body of water such as a river, lake, or pond that is mapped with the National Hydrography Dataset (NHD),
3. fell within the “open water” and “riparian” vegetation categories LANDFIRE’s Biophysical Settings raster dataset **or**,
4. had greater than 2% surface water along their transects.

For the site selection process, I calculated relative species cover quantities using line point intercept data from all Terrestrial AIM sites sampled through 2020. I chose relative cover over absolute cover to be more inclusive, acknowledging that Terrestrial AIM sites did not specifically target wetlands. Consequently, Terrestrial sites that included wetlands were likely to cross wetland boundaries, rather than be centered entirely within a wetland. I calculated relative cover as the proportion of times an individual species was hit on the Line-Point-Intercept (LPI) transect in relation to the total number of pin drops that intersected a plant. For example, for a transect with plants hit at 100 total LPI pin drops, a species hit 25 times would have 25% relative cover. However, for a transect with 40% bare ground (no plants hit for 40 out of 100 pin drops), a species would only have to be hit 15 times to have 25% relative cover (15/60 = 25%).

To determine if sites intersected an NWI polygon, as required for the second criteria of selection, I created 30-meter buffers around each plot center point to represent the entirety of each plot. Any site whose 30-meter buffer intersected an NWI polygon was selected for inclusion. I downloaded NWI data by state from the US Fish and Wildlife Service data download website (USFWS, n.d.). I used an identical process to find sites near NHD polygons, except I used a 50-meter buffer from the plot center instead of a 30-meter buffer to account for some lack of spatial accuracy in the NHD dataset. I accessed NHD data using the get_nhdplus() function from the nhdplusTools package using R Statistical Software (v4.1.2, 2021).

The final category of sites included those that were selected as part of a riparian strata within the Terrestrial AIM program. Despite the deliberate inclusion of these sites, they were not labeled as riparian, making it impossible to search for them in the Terrestrial database. Therefore, I identified sites that met the original selection criteria for the riparian strata by locating sites that occurred within a raster cell categorized as “open water” or “riparian” in LANDFIRE’s Biophysical Settings (BPS) dataset. The Terrestrial AIM team has used the BPS raster to classify and stratify all Terrestrial sample designs to ensure proportional sampling occurs based on an ecosystem’s prevalence on the landscape. I completed all spatial analyses in ArcGIS Pro 2.9.0 except for the NHD analyses.

I additionally chose to include sites that had greater than 2% surface water along their transects because permanent and ephemeral surface water are “lower layer” and “ground code” options for LPI in the Terrestrial AIM protocol (Herrick et al., 2021). While 2% may seem like a small amount, it is not uncommon for sites sampled with the Wetland protocol to have zero percent standing water along the transects at the time of sampling. Furthermore, while the Wetland protocol has other ways of documenting standing water, the Terrestrial protocol does not have a designated way to document such data outside of the LPI transects. After completing these calculations in R, I combined all sites that met any of the four secondary criteria and removed duplicates that occurred from sites meeting multiple criteria. While my initial queries of the Terrestrial AIM database were completed on all sites in the lower 48, I ultimately analyzed only data from Colorado, Utah, and Idaho to match the states where sampling occurred for the R&W 2019 and 2020 pilot years.

Prior to performing statistical analyses, I removed species that occurred in less than five percent of sites as well any sites that had only those removed species present. The subsequent dataset used for analyses were 85 Terrestrial sites and 115 R&W sites. I removed an additional site from the R&W only NMDS, as it was an extreme outlier in the ordination, resulting in a total of 114 sites for that analysis.

2.2.3 Terrestrial and R&W Comparison

The metrics I used to compare the selected Terrestrial AIM sites to the R&W AIM sites included perennial foliar cover, annual foliar cover, noxious foliar cover, herbaceous foliar cover, woody foliar cover, hydrophytic foliar cover, foliar upland cover, total foliar cover, elevation, species richness, species evenness, mean C value, Floristic Quality Assessment Index (FQAI), and percent "intolerant" species. All metrics were calculated from LPI data.

I calculated foliar cover as the percent of LPI pin drops within a transect that hit a species of a given metric. For instance, I calculated herbaceous foliar cover as the number of pin drops that had an herbaceous species hit. Although multiple herbaceous species may be hit at a single LPI location, only one hit was counted as one for each LPI location. I calculated hydrophytic species cover as the foliar cover of species that had wetland indicator statuses of OBL, FACW or FAC and upland species cover as the foliar cover of species that had wetland indactor statuses as FACU or UPL. I used the community_structure() function from the "codyn" package to calculate evenness, using the default Evar method from Smith and Wilson 1996. I calculated mean C value as the mean C value of all plants occurring along all three transects at a given site. FQAI was calculated as the mean C value multiplied times the square root of species richness. Finally, percent intolerant is a metric that represents the percent of species present that have a C-value greater than or equal to seven. A high percent intolerant value indicates a large number of species that are intolerant to disturbance. Because none of the quantitative variables in the datasets were normally distributed, I compared the quantitative variables between the two groups using the nonparametric Wilcoxon Rank Sum test.

2.2.4 NMDS Analysis

I analyzed species composition of sites using nonmetric multidimensional scaling (NMDS) with the "vegan" package in R. I performed one NMDS to compare the foliar species composition of the selected Terrestrial sites to the composition of the R&W sites, and one to assess the drivers of composition of just the R&W sites. I used the envfit() function to fit continuous environmental variables to the ordination matrices and identify significant relationships with plant community composition (J. Oksanen, 2013). I used the simper() function from the "vegan" package to identify the contribution of individual species to the overall Bray-Curtis dissimilarity matrix between the AIM programs for the ordination that included all sites, and between wetland types for the ordination that included only R&W sites (A. J. Oksanen et al., 2019). I used the adonis2() function to perform permutational multivariate analysis of variance (PERMANOVA) to test for significant differences in the species composition among groups. If a PERMANOVA indicated a significant difference in an analysis with more than two groups, I used the pairwise.adonis2() function to perform pairwise comparisons between all groups.

2.2.5 R&W Community Analyses

For the R&W ordination, I compared differences among five groups: wetland type, hydrogeomorphic (HGM) classification, Cowardin classification, major soil type, and hydric soil indicator functional groups. I visually assessed the distribution of these categories across the

NMDS ordination using ellipses and performed PERMANOVAs to identify significant differences in species composition among the different groups in each category.

I then performed analyses to test for statistical differences in individual quantitative variables. For categories with three or more groups, I used the lm() function in R to fit a linear model, then tested the model's residuals for normality using the Shapiro-Wilks test. If residuals were normal, I used Levene's test to test for equal variances. If both of these assumptions were met, I used the anova() function to perform an ANOVA on the linear model. If the assumptions were not met, I used the kruskal.test() function to perform Kruskal Wallis nonparametric test for differences between group medians. For categories with just two groups, I tested the data for normality using the Shapiro-Wilks test. If data was found to be normal, I used the t_test() function from the "rstatix" package to test for differences between the groups. If the assumption of normality was not met, I used the Wilcoxon Rank Sum test.

2.3 Results

Out of the 35,878 total Terrestrial sites sampled in the lower forty-eight states, only 250 sites within the matched the criteria outlined for wetland and riparian areas (Figure 1a). These sites had a mean percent hydrophytic cover of 35.96% and had highly significantly greater hydrophytic cover than the Terrestrial dataset as a whole (t-test, $p = 4.52e{-114}$; Figure 2). In 2011 there was only one site that met the criteria, while in 2017 there were 61 sites that met the criteria. Just over 1% of all the Terrestrial sites visited in 2017, 2018, and 2020 were selected, while in all other years less than 1% of sites were considered riparian or wetland.

Terrestrial sites meeting the criteria occurred in all states where sampling occurred. Colorado had the highest number of qualifying sites, where 57 sites met the selection criteria, accounting for 1.5% of all sites sampled. North Dakota and Washington had the fewest qualifying sites, with only one and two sites, respectively. Qualifying sites made up less than one percent of all sites sampled in all states other than Colorado. When narrowed down to just Colorado, Utah, and Idaho, there were 85 sites from the Terrestrial database that met the wetland criteria (Figure 1b).

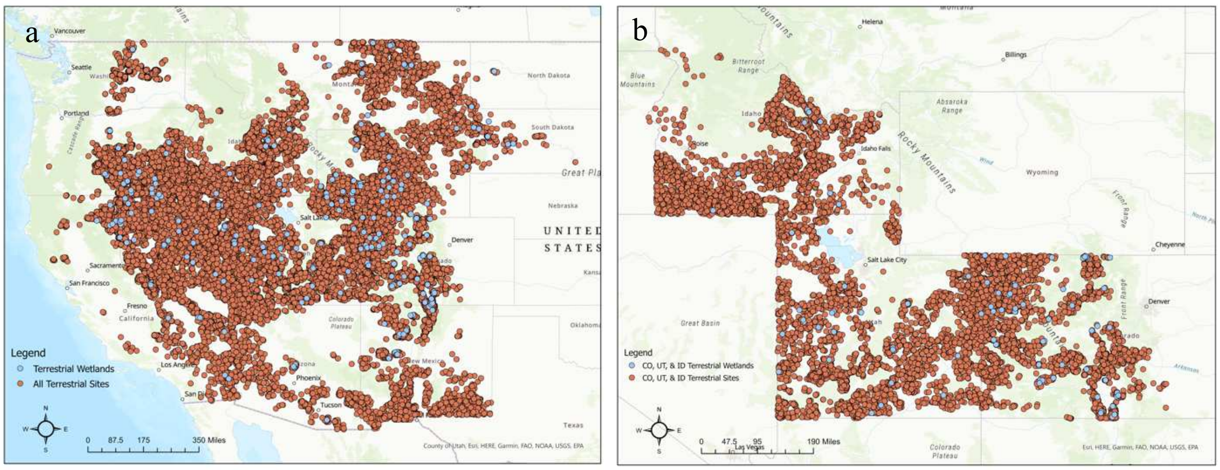

Figure 1. Spatial distribution of a) all Terrestrial AIM sites sampled in the lower 48 between 2011 and 2020 with sites that met criteria for selection in this study highlighted in blue, and b) all Terrestrial AIM sites sampled in Colorado, Utah, and Idaho between 2011 and 2020 with sites that were selected for this study highlighted in blue.

R&W sites had much greater total foliar cover (median = 93.3), than the Terrestrial sites (median = 62.0, p = 1.44e-12, Figure 3). R&W sites also had much less variable foliar cover (interquartile range = 13.667) than the Terrestrial sites (interquartile range = 50.333). However, the two groups had very similar median annual cover values and distributions (T median = 0.67, R&W median = 1.33, p = 0.82). Additionally, Terrestrial sites had greater upland species cover (median = 23.0) than R&W sites (median = 8.0, p = 3.26e-07). The Terrestrial group also had more evenly distributed upland cover data than the R&W sites. Similarly, woody species cover was much greater in the Terrestrial group (median = 18.7) than the R&W group by a factor of nearly seven (median = 2.7, p = 7.92e-05).

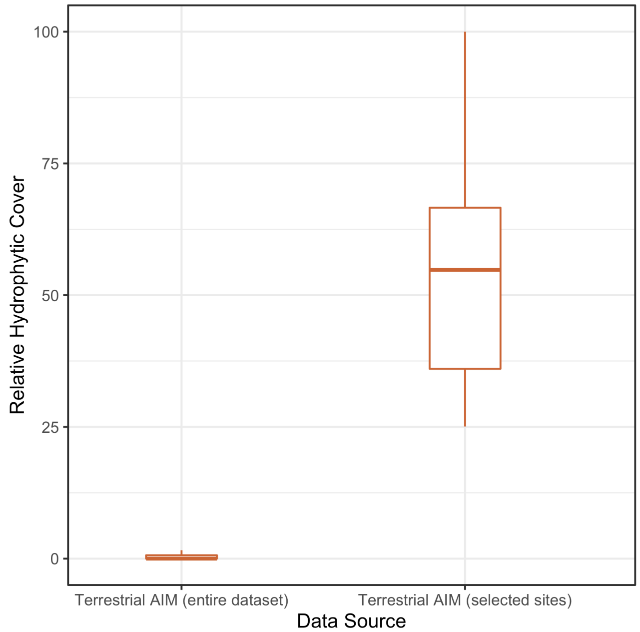

Figure 2. Relative hydrophytic species cover of all Terrestrial AIM sites sampled in the lower 48 states between 2011 and 2020 (n = 35879) and the Terrestrial AIM sites selected for this study (n = 250).

Areas where R&W sites did have higher cover were perennial species cover (p = 9.85e-15), herbaceous species cover (p = 6.96e-17), and hydrophytic species cover (p = 1.73e-25, Figure 3). R&W sites had much greater perennial cover (median = 91.3) than the Terrestrial sites (median = 51.0). 90 percent of R&W sites had perennial cover of 73.3 or greater, compared to the Terrestrial sites, 90 percent of which had perennial cover of 17.7 or greater. The R&W sites had a median (88.67) more than twice the herbaceous cover (median = 88.67) of the Terrestrial sites (median = 37.00). Once again, herbaceous cover was less variable for the R&W group (interquartile range = 21.0) than the Terrestrial group (interquartile range = 57.5).

The difference in hydrophytic cover between the two groups was even more pronounced. The median for the R&W sites (95.3) was more than twice the median for the Terrestrial sites (37.3). R&W sites also had significantly greater noxious species cover (p = 7.74e-05), though it was very low for both groups. Both groups had median noxious cover of 0, but R&W sites had a mean noxious cover of 4.7 compared to a mean of 0 for the Terrestrial group.

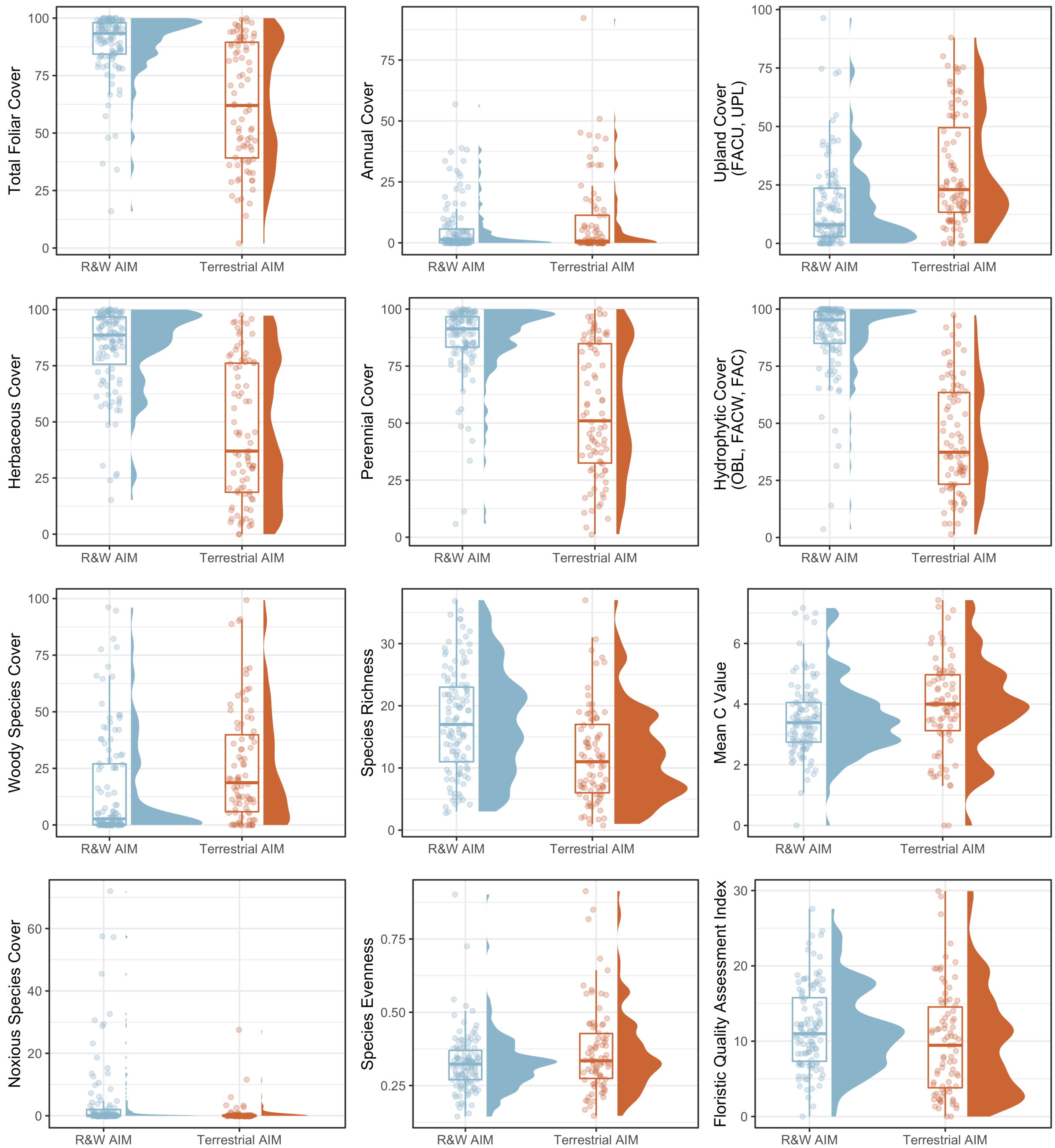

Figure 3. Distribution of plant species composition metrics for R&W AIM and selected Terrestrial AIM sites. All cover metrics represent foliar cover.

Median species richness for R&W sites was 17 species, which was significantly greater than that of Terrestrial sites (median = 11, $p = 1.59e-06$; Figure 3), but there was no difference in evenness between the two groups ($p = 0.0805$), nor was evenness significantly correlated with the ordination (p = 0.534). Terrestrial sites had significantly greater mean C values (median = 4.00) than R&W sites (median = 3.39, p = 0.007), though neither means (T = 3.92, R&W = 3.5) nor medians were far from each other. FQAI values were significantly higher in the R&W data than in the Terrestrial data (p = 0.0196), but both groups had FQAI values that are associated with low floristic quality (Means: T = 7.081431, R&W = 11.708165, Medians: T = 9.457, R&W = 10.994, Rocchio et al., 2007; Wilhelm & Masters, 1995). Less than 10% of sites in each group had values of 20 or above. R&W sites also had lower percent intolerant species (median = 0, mean = 6.70) than the Terrestrial sites (median = 0, mean = 12.7), but, again, values were very low for both groups (p = 0.00889).

The NMDS ordination with the selected Terrestrial sites and the R&W sites converged with two dimensions and a final stress of 0.132. The NMDS demonstrated that the species composition of the wetland sites found in the Terrestrial AIM database are distinctly different from the sites sampled with the R&W AIM program (Figure 4;p = 0.001). Species that contributed the most to average dissimilarity between the two groups and were more characteristic of the Terrestrial sites were *Pascopyrum smithii* (avg. diss. = 0.016, p = 0.053), *Ericameria nauseosa* (avg. diss. = 0.0135, p = 0.17), and *Bromus tectorum* (avg. diss. = 0.010, p = 0.121). Species that were associated with the R&W and contributed the most to average dissimilarity were *Juncus arcticus* (avg. diss. = 0.0415, p = 0.001), *Poa pratensis* (avg. diss. = 0.0225, p = 0.084), *Distichlis spicata* (avg. diss. = 0.0224, p = 0.521), and *Carex nebrascensis*

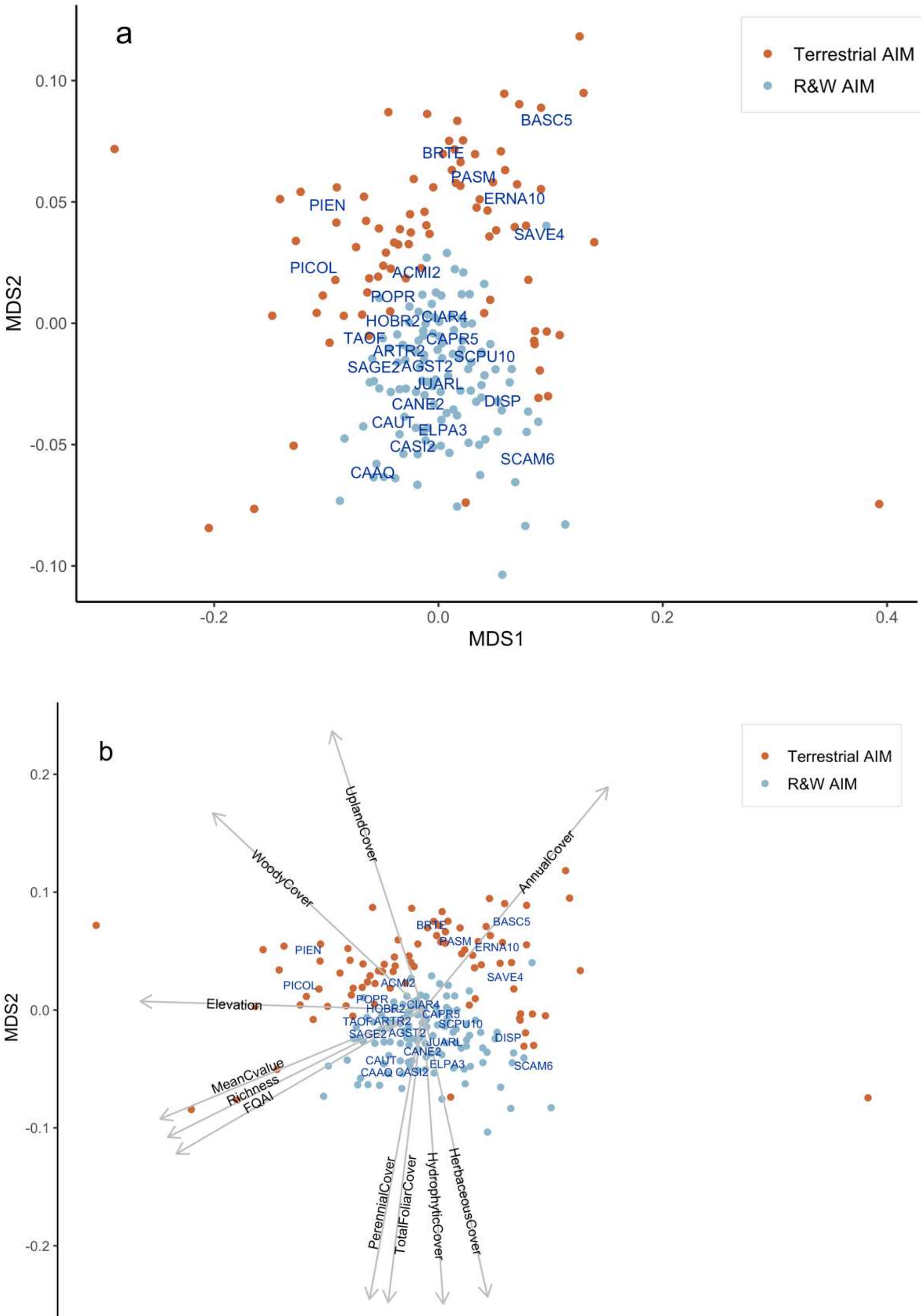

Figure 4. NMDS ordination of wetlands identified in the Terrestrial AIM database that were sampled within CO, UT, or ID, and wetlands that were sampled with the R&W AIM program. a) Species with the highest contribution to dissimilarity in species composition between the two datasets were *Juncus arcticus* ("JUARL"), *Poa pratensis* ("POPR"), *Distichlis spicata* ("DISP"), *Carex nebraskensis* ("CANE2"), Pascopyrum smithii ("PASM"), *Carex*

utriculata ("CAUT"), *Eleocharis palustris* ("ELPA3"), *Ericameria nauseosa* ("ERNA10"), *Carex aquatilis* ("CAAQ"), *Taraxacum officinale* ("TAOF"), *Carex praegracilis* ("CAPR5"), *Schoenoplectus americanus* ("SCAM6"), *Hordeum brachyantherum* ("HOBR2"), *Achillea millefolium* ("ACMI2"), *Carex simulata* ("CASI2"), *Circium arvense* ("CIAR4"), *Bromus tectorum* ("BRTE"), *Pinus contorta* ("PICOL"), *Salix geyeriana* ("SAGE2"), *Schoenoplectus pungens* ("SCPU10"), *Agrostis stolonifera* ("AGST2"), *Artemisia tridentata* ("ARTR2"), *Sarcobatus vermiculatus* ("SAVE4"), *Bassia scoparia* ("BASC5"), and *Picea engelmannii* ("PIEN"). b) Vectors represent quantitative data that was significantly associated with the species composition ordination. The direction and magnitude of the arrows indicate the direction of increasing values and the rate at which they increase. Metrics significantly correlated with the species composition were annual species foliar cover ("AnnualCover"), upland species foliar cover ("UplandCover"), woody species foliar cover ("WoodyCover"), elevation, mean C value ("MeanCvalue"), species richness ("Richness"), floristic quality index ("FQAI"), perennial species foliar cover ("PerennialCover"), total foliar cover ("TotalFoliarCover"), hydrophytic species foliar cover ("HydrophyticCover"), and herbaceous species foliar cover ("HerbaceousCover").

(avg. diss. = 0.0202, $p = 0.047$). *Carex aquatilis* was present at R&W sites that were least similar to the Terrestrial sites, while *Bassia scoparia* was present at Terrestrial sites farthest from the R&W sites.

The variables that were significantly correlated with the species composition ordination were perennial foliar cover ($r^2 = 0.2547$, $p < 0.001$), annual foliar cover ($r^2 = 0.1327$, $p < 0.001$), herbaceous foliar cover ($r^2 = 0.2675$, $p < 0.001$), woody foliar cover ($r^2 = 0.1935$, $p < 0.001$), hydrophytic foliar cover ($r^2 = 0.3824$, $p < 0.001$), upland foliar cover ($r^2 = 0.1781$, $p < 0.001$), elevation($r^2 = 0.4184$, $p < 0.001$), species richness ($r^2 = 0.0891$, $p < 0.001$), mean C value ($r^2 = 0.3895$, $p < 0.001$), FQAI ($r^2 = 0.3515$, $p < 0.001$), total foliar cover ($r^2 = 0.1627$, $p < 0.001$), noxious species cover ($r = 0.0304$; $p = 0.038$) and percent intolerant species ($r^2 = 0.1627$, $p < 0.001$; Figure 4b). Elevation was the most highly correlated variable with NMDS axis 1 (-0.999), and hydrophytic cover was the most highly correlated with NMDS axis 2 (-0.998). The elevation vector does not clearly point in the direction of one group over the other, indicating that elevation is not a determining variable in the difference between the two groups. Sites with more wetland characteristics, such as greater hydrophytic species cover, greater perennial species cover, greater herbaceous species cover, and total foliar cover occur towards the bottom of the plot, in the direction of the cluster of R&W sites. Sites with greater upland species cover, woody species cover, and annual species cover are towards the top of the plot (Figure 4b). Vectors for mean C value, species richness, and FQAI point to the bottom left. While they point predominantly in the direction of the R&W cluster, they specifically point towards three Terrestrial sites that are on the opposite side of the plot than the main Terrestrial site cluster. The annual cover vector points strongly in the direction of the Terrestrial cluster and roughly in the opposite direction from the mean C value, FQAI, and species richness vectors.

A cluster of Carex species occurs towards the bottom left of the plot, including *Carex aquatilis*, *Carex utriculata*, and *Carex simulata* (Figure 4a). This cluster falls between the two vectors indicating greater FQAI and perennial cover. *Bromus tectorum* and *Pascopyrum smithii* fall towards the top of the plot, between the vectors for upland cover and annual cover. *Bassia scoparia* falls in the top right of the plot, along the annual cover vector. Two pine species, *Picea engelmannii* and *Pinus contorta,* fall near the woody cover vector.

The NMDS ordination with species cover data from just the R&W sites converged with three dimensions and a final stress of 0.171 (Figure 5). The metrics that were significantly correlated with the species composition ordination were extent surface water ($r = 0.088$, $p = 0.016$), maximum water depth ($r = 0.063$, $p = 0.047$), elevation ($r = 0.466$, $p < 0.001$), slope ($r = 0.132$, $p = 0.002$), total cover of litter or thatch along transects ($r = 0.0611$, $p = 0.036$), total water cover along transects ($r = 0.074$, $p = 0.012$), average litter or thatch depth ($r = 0.222$, $p <0.001$), mean specific conductance ($r = 0.265$, $p <0.001$), mean water temperature ($r = 0.222$, $p < 0.001$), hydrophytic species foliar cover ($r = 0.157$, $p <0.001$), upland species foliar cover ($r = 0.172$, $p < 0.001$), noxious species foliar cover ($r = 0.087$, $p = 0.011$), herbaceous species foliar cover

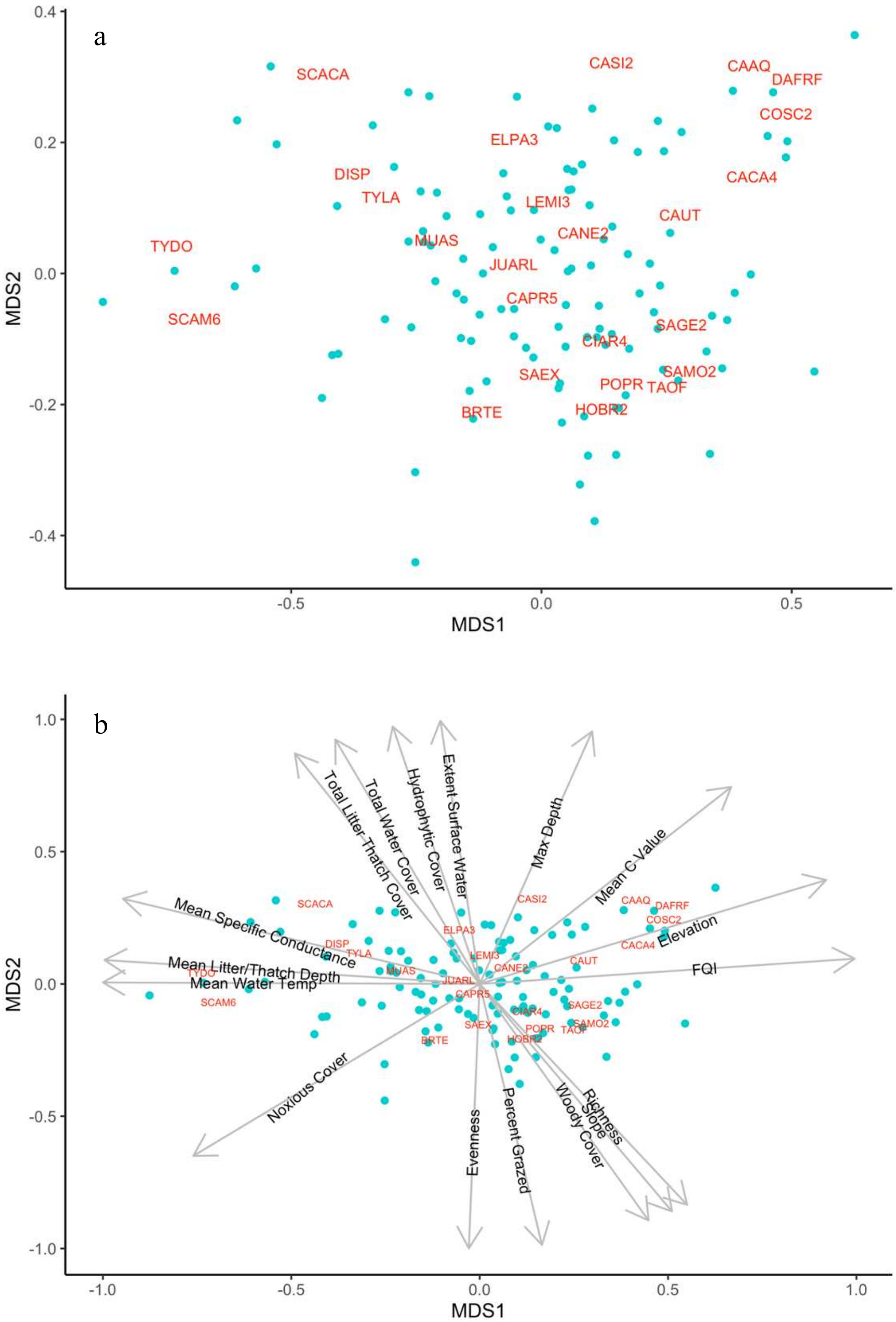

Figure 5. NMDS ordination of R&W sites sampled in 2019 and 2020. a) Species with the highest contribution to dissimilarity in species composition between the two datasets were *Carex aquatilis* ("CAAQ"), *Juncus arcticus* ("JUARL"), *Schoenoplectus americanus* ("SCAM6"), *Schoenoplectus acutus* ("SCACA"), *Carex simulata* ("CASI2"), *Carex utriculata* ("CAUT"), *Typha domingensis* ("TYDO"), *Distichlis spicata* ("DISP"), *Calamagrostis canadensis* ("CACA4"), *Dasiphora fruticosa* ("DAFRF"), *Salix exigua* ("SAEX"), *Carex nebraskensis* ("CANE2"), *Poa pratensis* ("POPR"), *Taraxacum officinale* ("TAOF"), *Hordeum brachyantherum* ("HOBR2"), *Eleocharis palustris* ("ELPA3"), *Circium arvense* ("CIAR4"), *Salix monticola* ("SAMO2"), *Salix geyeriana* ("SAGE2"), *Carex*

praegracilis (“CAPR5”), *Muhlenbergia asperifolia* (“MUAS”), *Bromus tectorum* (“BRTE”), *Lemna minor* (“LEMI3”), *Conioselinum scopulorum* (“COSC2”), and *Typha latifolia* (“TYLA”). b) Vectors represent quantitative data that was significantly associated with the species composition ordination. The direction and magnitude of the arrows indicate the direction of increasing values and the rate at which they increase. Metrics significantly correlated with the species composition were estimated extent of surface water throughout the plot (“Extent Surface Water,” hydrophytic species foliar cover, total water cover, calculated as LPI pin drops with surface water (“Total Water Cover”), total litter or thatch cover (“Total Litter/Thatch Cover”), mean specific conductance (“Mean Specific Conductance”), mean litter or thatch depth (“Mean Litter/Thatch Depth”), mean water temperature (“Mean Water Temp”), noxious species foliar cover (“Noxious Cover”), species evenness (“Evenness”), percent of graminoids that were grazed (“Percent Grazed”), woody species foliar cover (“Woody Cover”), slope (“Slope”), and species richness (“Richness”).

($r = 0.126$, $p < 0.001$), woody species foliar cover ($r = 0.168$, $p < 0.001$), mean C value ($r = 0.462$, $p < 0.001$), floristic quality index ($r = 0.488$, $p < 0.001$), percent of intolerant species present ($r = 0.374$, $p < 0.001$), and percent of graminoids grazed ($r = 0.076$, $p = 0.011$; Figure 5b). Herbaceous species foliar cover and upland species foliar cover were also significantly correlated with the ordination, but were removed from the plot due to redundancy with woody cover and hydrophytic cover, respectively. Percent intolerant was also significantly correlated, but I considered it to be similar to mean C value and was also removed from the plot for clarity. Mean water temperature was the most strongly correlated metric with the NMDS1 axis (-0.999), while species evenness was the most strongly correlated metric with the NMDS2 axis (-0.999), despite its low correlation with species composition overall (0.10259691).

The PERMANOVA analyses revealed significant differences in species composition among wetland types ($p < 0.001$), Cowardin classifications ($p < 0.001$), HGM classifications ($p < 0.001$), major soil types ($p < 0.001$), and hydric soil indicator functional groups ($p < 0.001$).

2.3.1 Wetland Types

All wetland types had significantly different species composition from one another ($p < 0.001$). Within wetland types, riparian shrublands had a mean species richness of 24.12, which was greater than fens (mean = 13.0, $p = 0.0079$) and marshes (mean = 8.636, $p < 0.0001$).

Table 1. Summary of results of statistical analyses of different variables among wetland type, Hydrogeomorphic (HGM) classification, and Cowardin classification groups.

	Wetland Type	**HGM Classification**	**Cowardin Classification**
Vegetative cover			
Total foliar cover	Significantly different	No difference	No difference
Herbaceous cover	Significantly different	No difference	Significantly different
Hydrophytic cover	No difference	No difference	No difference
Noxious cover	Significantly different	No difference	Significantly different
Upland cover	Significantly different	Significantly different	Significantly different
Woody cover	Significantly different	Significantly different	Significantly different
Annual cover	Significantly different	No difference	No difference
Perennial cover	No difference	No difference	No difference
Total litter/thatch cover	No difference	No difference	No difference
Bare soil cover	No difference	No difference	No difference
Site Characteristics			
Aspect	No difference	No difference	No difference
Elevation	Significantly different	Significantly different	No difference
Slope	Significantly different	Significantly different	No difference
Floristic Quality			
FQAI	Significantly different	Significantly different	Significantly different
Mean C value	Significantly different	No difference	No difference
Percent intolerant	Significantly different	No difference	Significantly different
Species evenness	Significantly different	No difference	Significantly different
Species richness	Significantly different	Significantly different	Significantly different
Annual Grazing			
Percent grazed	Significantly different	No difference	No difference
Average soil alteration	Significantly different	No difference	No difference
Water quality			
Mean water temp	No difference	No difference	Significantly different
Specific conductance	Significantly different	Significantly different	Significantly different
Mean pH	Nearly significant ($p = 0.59$)	Significantly different	No difference
Total nitrogen	Significantly different	No difference	No difference
Total phosphorus	Significantly different	No difference	Nearly significant ($p = 0.50$)
Water quantity			
Average water depth	Significantly different	Significantly different	No difference
Depth to saturated soil	No difference	Significantly different	No difference
Average channel width	Significantly different	Significantly different	Significantly different
Soil pit water depth	Significantly different	Significantly different	No difference
Total water cover	Significantly different	No difference	Significantly different
Extent surface water	Significantly different	No difference	No difference
Max water depth	No difference	No difference	No difference

Marshes also had significantly lower species richness than spring/seeps, (mean = 20.133, $p < 0.0001$) and wet meadows (mean = 20.045, $p = 0.0011$. Spring/seeps were the wetland type with the greatest evenness (median = 0.346), which was significantly greater than that of fens (median = 0.309, $p = 0.0052$), marshes (median = 0.250, $p = 0.0004$) and wet meadows (median = 0.313, $p = 0.0028$), but not riparian shrublands (median = 0.340, $p = 0.0766$).

Riparian shrublands had the greatest upland species cover (median = 27.26%), which was significantly greater than that of fens (median = 2.04%, Dunn Test, $p = 0.0000$) and marshes (median = 0.0%, $p = 0.0000$), but wasn't significantly different from spring/seeps (median = 16.54%) or wet meadows (median = 11.67%). However, spring/seeps also had significantly greater upland cover than fens ($p = 0.0008$) and marshes ($p = 0.0006$). Riparian shrublands also had the highest median woody species cover (56.67%), which was significantly greater than that of fens (median = 1.0%, Dunn Test, $p = 0.0000$), marshes (median = 0.0%, $p = 0.0000$), spring/seeps (median = 4.00%, $p = 0.0000$) and wet meadows (median = 0.67%, $p = 0.0195$). Spring/seeps had the second greatest woody cover, which was significantly greater than wet meadows ($p= 0.0195$) and marshes ($p = 0.0002$).

Fens had significantly greater mean C values than all other groups (mean = 5.62, $p < 0.001$). Marshes FQAI values averaged 6.845, and were lower than fens (mean = 15.86, $p = 0.0004$), riparian shrublands (mean = 14.87, $p = 0.0002$), and spring/seeps (mean = 11.88, $p = 0.0196$), but not wet meadows (mean = 14.87, $p = 0.1096$). Fens also had FQAI values that were nearly significantly greater than wet meadows ($p = 0.0602$). Fens had significantly lower annual species cover (median = 0.0%) compared to riparian shrublands (median = 1.33%, Dunn Test, $p = 0.0013$), spring/seeps (median = 1.33%, $p = 0.0008$), and wet meadows (median = 3.34%, $p = 0.0002$), but not marshes (median = 0.67%, $p = 0.0387$). Fens also had the highest percent intolerant species (median = 20.83%), which was significantly greater than marshes (median = 0.0%, Dunn Test, $p = 0.0002$), spring/seeps (median = 0.0%, $p = 0.0003$), and wet meadows (median = 0.0%, $p = 0.0004$). Riparian shrublands had the second highest percent intolerant species (median = 7.63%) and were not significantly different than fens ($p = 0.2153$) but were significantly greater than marshes ($p = 0.0005$), spring/seeps ($p = 0.0008$), and wet meadows ($p = 0.0010$). Similarly, fens had the lowest noxious species cover of any group (median = 0.00%, mean = 0.00%), which was different from riparian shrublands (median = 1.08% mean = 10.98%, Dunn Test, $p = 0.0002$), spring/seeps (median = 0.00%, mean = 5.98%, $p = 0.0104$), and wet meadows (median = 0.00%, mean = 2.94%, $p = 0.0105$). Riparian shrublands had the greatest noxious cover and were also significantly different than marshes (median = 0.00%, mean = 1.81%, $p = 0.0044$).

Wet meadows had the lowest median total water cover (3.0%), which was significantly different than fens (median = 30.42%, $p = 0.0019$), marshes (median = 27.50%, $p = 0.0016$) and spring/seeps (median = 8.67%, $p = 0.0108$). Fens had the greatest extent surface water (median = 40%), which was significantly greater than wet meadows (median = 10%, Dunn Test, $p = 0.0026$) and spring/seeps (median = 10%, $p = 0.0091$). Marshes and riparian shrublands had the greatest average water depth (medians = 9.67 cm, 11.67 cm), and were not significantly different from each other (Dunn Test, $p = 0.2789$). Marshes had significantly greater average water depth (median = 9.669 cm) than spring/seeps (median = 5.00 cm, $p = 0.0198$). Riparian shrubland average water depth was significantly different than fens (median = 5.43 cm, $p = 0.0139$) and spring/seeps ($p = 0.0007$). Spring/seeps had the lowest average water depth which was also significantly different from wet meadows (median = 8.05 cm, $p = 0.0208$).

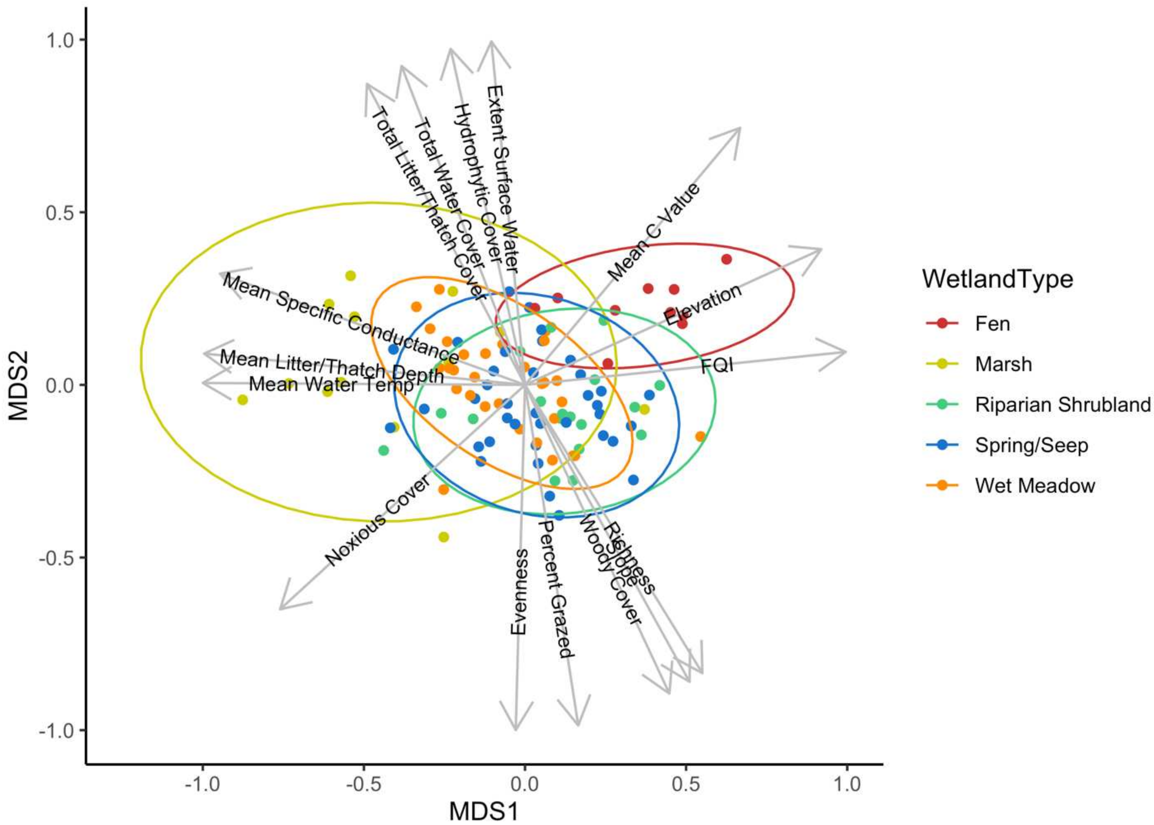

Figure 6. NMDS ordination of R&W sites sampled in 2019 and 2020. Colors correspond to the different wetland types as they were classified in the field. Ellipses were manually added to aid in comparisons among groups.

Marshes had the greatest specific conductance (median = 1271.50 μS), which was greater than riparian shrublands (median = 203.33 μS, Dunn Test, p = 0.0006), spring/seeps (median = 375.00 μS, p = 0.0177) and fens (median = 166.08 μS, p = 0.0003). Fens had the lowest mean specific conductance, which was lower than spring/seeps (p = 0.0207) and wet meadows (median = 396.00 μS, p = 0.0125).

Spring/seeps had the greatest percent grazed graminoids (median = 34.62), which was significantly greater than that of fens (median = 2.58, Dunn Test, p = 0.0129), marshes (median = 0.00, p = 0.0028), and riparian shrublands (median = 0.93, p = 0.0019). There was no significant difference in percent grazed between spring/seeps and wet meadows (median = 17.78, p = 0.0827), despite a two-fold difference in medians. A similar pattern existed among the wetland types in average soil alteration. Spring/seeps had greater average soil alteration (median = 0.67) than fens (median = 0.05, Dunn Test, $p = 0.0058$), marshes (median = 0.10, $p = 0.0122$), and riparian shrublands (median = 0.16, 0.0084), but not wet meadows (median = 0.50, $p = 0.2971$). Fens also had significantly lower soil alteration values than wet meadows ($p = 0.0179$). Spring/seeps (median = 669.00 μg/L, $p = 0.0015$) and marshes (median = 580.15 μg/L, $p = 0.0073$) had significantly greater total nitrogen concentrations than riparian shrublands (median = 202.00 μg/L), which had the lowest levels. Spring/seeps also had the greatest total phosphorus concentration (median = 51.7 μg/L), which was greater than that of marshes (median = 18.37 μg/L, Dunn Test, $p = 0.0116$) and wet meadows (median = 15.00 μg/L, $p = 0.0012$).

2.3.2 HGM Classification

Within the HGM classification groups, slope wetlands had significantly different species composition than riverine wetlands ($p = 0.003$), lacustrine fringe wetlands ($p = 0.036$), and depressional wetlands ($p = 0.001$). Riverine wetlands were also significantly different than depression ($p < 0.001$, but not lacustrine fringe wetlands ($p = 0.12$).

The NMDS shows that the species composition of depressional wetlands was most influenced by elevation and mean water temperature (Figure 7). Slope wetlands were similarly influenced by all of the variables significantly associated with the ordination. Riverine wetlands were most strongly influenced by elevation and tended to have greater woody cover and greater slope. While slope wetlands and riverine wetlands have a lot of overlap in the NMDS, slope wetlands include a wider range of plant communities, while riverine wetlands trend towards having greater woody cover, slope, and species richness.

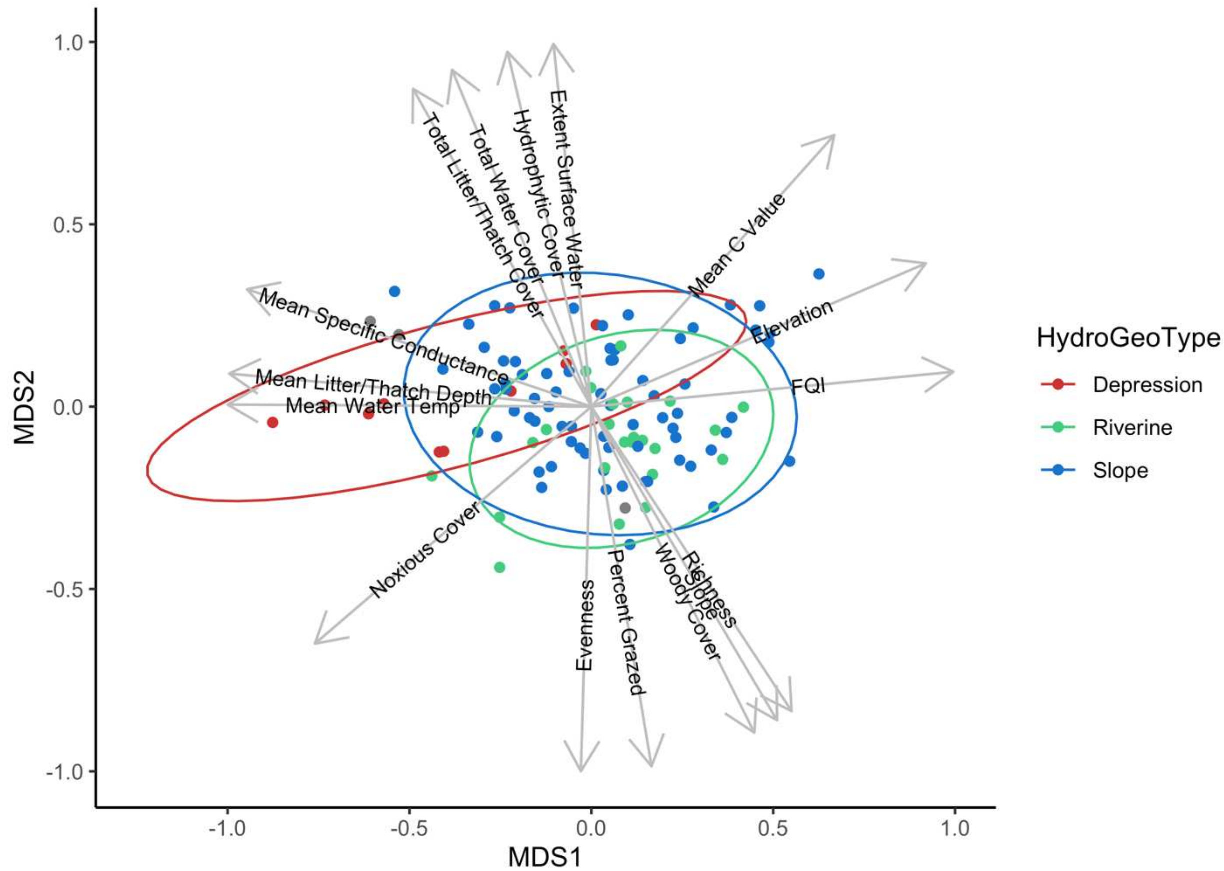

Figure 7. NMDS ordination of R&W sites sampled in 2019 and 2020. Colors correspond to the different hydrogeomorphic classifications as they were assigned in the field. Ellipses were manually added to aid in comparisons among groups.

The HGM classification system did not show significant differences in evenness among the groups, but there were differences in species richness. Riverine wetlands had the greatest richness (median = 21 species) compared to depressional wetlands (median = 6, Dunn Test, p = 0.000) and slope wetlands (median = 19, p = 0.0198). Depressional wetlands had the lowest species richness of all of the HGM groups, which was significantly different from slope wetlands (p = 0.0010). Riverine wetlands (median = 20.67, Dunn Test, p = 0.0000) and slope wetlands (median = 8.33, p = 0.0002) both had significantly greater upland species cover than depression wetlands (median = 0.00). The three groups also had significant differences in woody species cover. Riverine wetlands had significantly greater woody cover (median = 32.34) than slope wetlands (median = 2.34, Dunn Test, $p = 0.0000$) and depression wetlands (median = 0.00, $p = 0.0003$). Although the median woody cover in slope wetlands was much closer to depressional wetlands than riverine, woody cover in slope wetlands was significantly greater than depression wetlands ($p = 0.0045$). Riverine sites had the greatest FQAI values (median = 13.62) and were significantly larger than those of depressional wetlands (median = 6.16, $p = 0.0025$), despite there being no significant differences among the mean C values of the HGM classification groups. Depressional wetlands had significantly larger mean specific conductance (median = 2089 µS) compared to riverine (median = 317 µS, Dunn Test, $p = 0.0010$) and slope wetlands (median = 354.83 µS, $p = 0.0014$). Riverine wetlands (median = 0.00, mean =-6.773 cm) had significantly greater depth to saturated soil than both depressional wetlands (median = 0.00, mean = -4.111 cm, $p = 0.0135$) and slope wetlands (median = 0.00, mean = -2.306 cm, $p = 0.0070$).

2.3.3 Cowardin Classification

Cowardin classification groups were narrowed down to just palustrine emergent (PEM) and palustrine scrub shrub (PSS). The NMDS indicated that PSS sites fell almost entirely within the range of species composition that exists for PEM sites, but that PSS sites trended towards having greater slope, woody species cover, upland species and species richness than PEM sites (Figure 8). They also tended to have low litter/thatch depth and cover, low mean specific conductance, and low mean water temperature.

PSS sites had greater species richness (mean = 24.96 species) than PEM sites (mean = 16.23, $p = < 0.0001$). PSS sites also had greater evenness (median = 0.350, Wilcoxon Test, $p = 0.00752$), upland cover (median = 23.245%, Wilcoxon Test, $p = 0.00044$), woody cover (median

= 48.67%, Wilcoxon Test, p = 6.9e-16), and FQAI (median = 16.308, Wilcoxon Test, p = 0.000247).

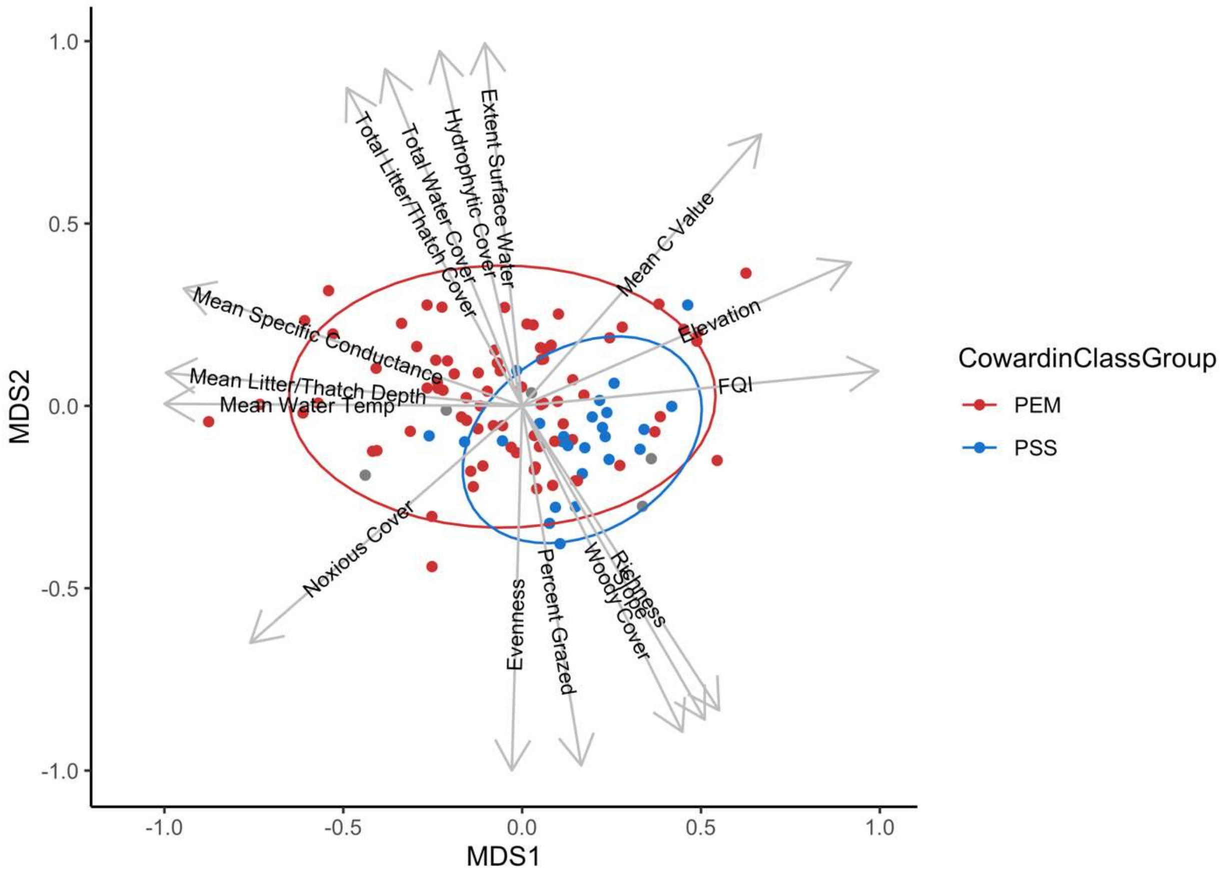

Figure 8. NMDS ordination of R&W sites sampled in 2019 and 2020. Colors correspond to the different Cowardin classifications assigned to each site in the field. Ellipses were manually added to aid in comparisons among groups.

2.3.4 Major Soil Type

Species composition of sites with organic soil was strongly correlated with elevation and water temperature, while composition of sites with mineral soil was most highly correlated with specific conductance, water temperature, and slope (Figure 9). Analyses of major soil types and hydric soil indicator groups both showed significant overlap with wetland type analyses in that sites with organic soils and/or organic matter accumulation aligned with trends of fens.

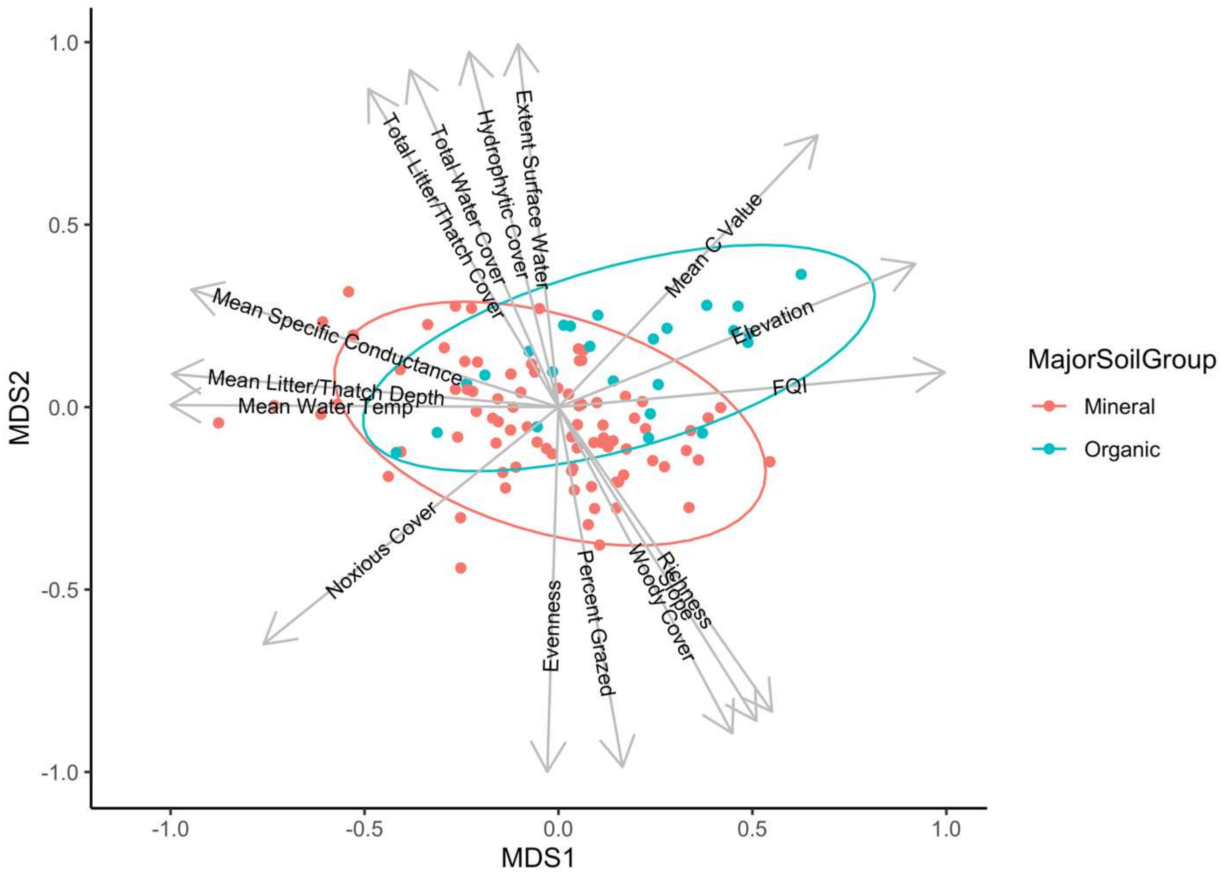

Figure 9. NMDS ordination of R&W sites sampled in 2019 and 2020. Colors correspond to the major soil types at each site. Ellipses were manually added to aid in comparisons among groups.

Sites with organic soils had greater FQAI values (median = 12.16, p = 0.028), greater mean C values (median = 4.21, p = 0.0002), and greater percent intolerant species (median = 2.941, p = 0.0137) than sites with mineral soils. Sites with mineral soils also had significantly greater upland species cover (median = 14.00, p = 0.002). Sites with mineral soils had greater annual species cover (median = 2.0, p = 0.0225), but there was no difference in noxious cover between the two groups (medians = 0, p = 0.199). Sites with organic soils (median = 97.34) had nearly significantly greater hydrophytic cover than sites with mineral soils (median = 94.00, p = 0.0584).

2.3.5 Hydric Soil Indicator Groups

Sites with organic matter accumulation were strongly influenced by elevation and water temperature (Figure 10). Sites with no indicators were influenced by slope and specific conductance, while sites with sulfate reduction fell towards the left of the plot, with higher specific conductance values, warm water temperatures, and higher thatch depth and cover. Sites with Fe/Mn reduction or accumulation fall largely in the middle of the plot, with influence from specific conductance and water temperature. There is a lot of overlap among all four groups in the center of the plot. Sites with sulfate reduction had significantly greater hydrophytic cover (median = 98.00) compared to sites with Fe/Mn reduction or accumulation (median = 92.22, $p = 0.0029$).

Among hydric soil indicator functional groups, sites with organic matter accumulation were significantly different than sites with no hydric soil indicators ($p = 0.003$), sites with sulfate reduction ($p = 0.006$), and sites with Fe/Mn reduction/accumulation ($p = 0.021$). Sites with Fe/Mn reduction/accumulation were also significantly different than sites with sulfate reduction ($p = 0.031$), but not sites with no indicators ($p = 0.33$). Sites with sulfate reduction were not significantly different than sites with no indicators ($p = 0.073$).

Sites with organic matter accumulation had significantly lower upland species cover (median = 6.67) than sites with Fe/Mn reduction or accumulation (median = 14.00, $p = 0.014$) and sites with no hydric soil indicators (median = 22.58, $p = 0.0136$), but there were not any other significant differences in upland cover among the different hydric soil indicator groups. There were no significant differences in woody species cover, species evenness, or species richness among the different hydric soil indicator groups.

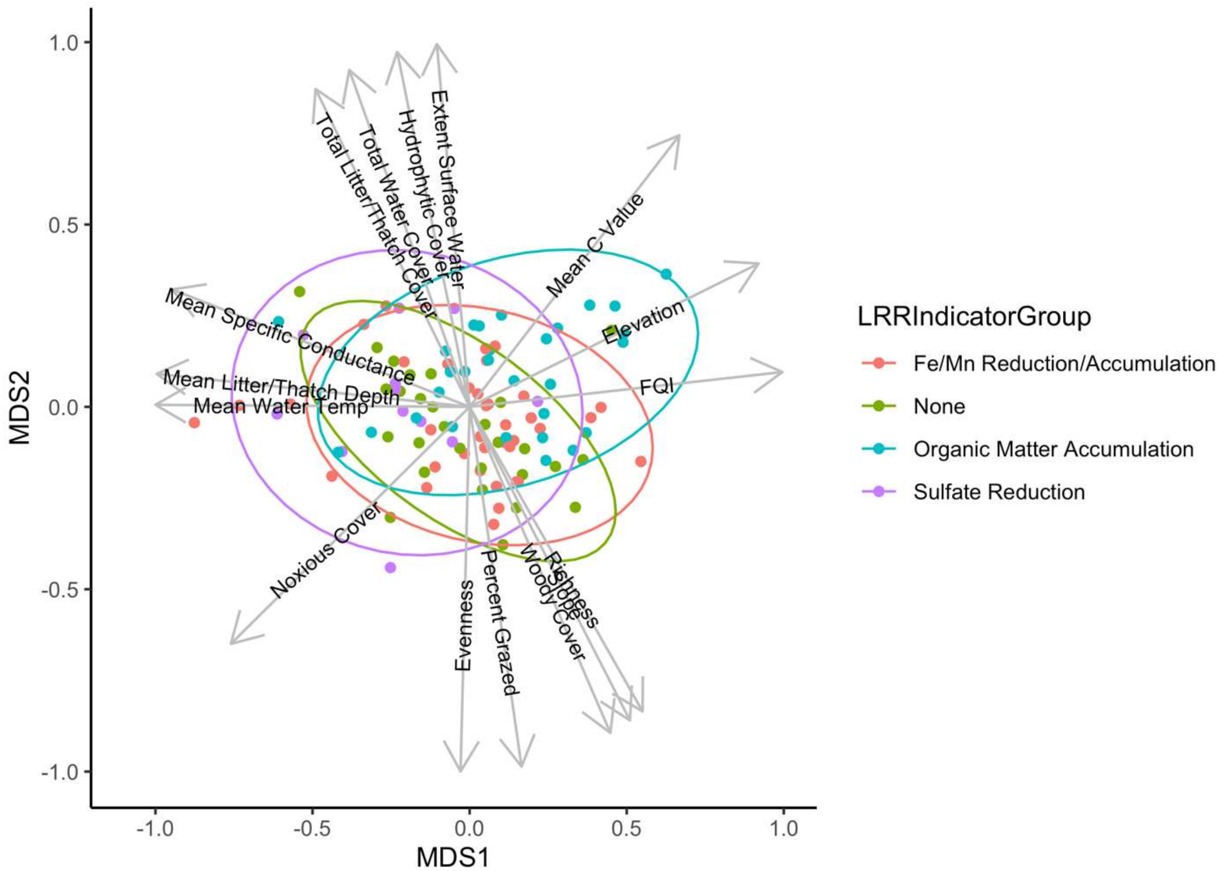

Figure 10. NMDS ordination of R&W sites sampled in 2019 and 2020. Colors correspond to the main hydric soil indicator found at each site. Ellipses were manually added to aid in comparisons among groups.

2.4 Discussion

The results of this study demonstrate that the wetlands and riparian areas sampled by the Terrestrial AIM program are not only rare outliers, but also represent a distinctly different population than the sites sampled with the R&W AIM program. The Terrestrial sites that met the criteria to be considered wetlands represented less than 1% of total sites sampled between 2011 and 2020, even in years that included riparian-specific site selection. The proportion of sites that met the criteria never accounted for even 2% of the sites sampled per year. Later years (2016-2020) had roughly 1% of sites qualify per year, while years prior to 2016 all had less than 1% of sites meet the criteria, despite known intent to sample riparian areas during that time. This is less than would be expected under normal error and demonstrates that despite the fact the Terrestrial AIM program sample design selects sites from across the entire BLM owned landscape, a more targeted approach is needed to sample riparian and wetland areas on a scale appropriate for resource monitoring.

Moreover, the Terrestrial sites that were captured in CO, UT, and ID were distinctly different from the sites sampled with the R&W program. The NMDS plot clearly depicts two distinct populations with little overlap. The drastic difference in foliar cover between the two groups can be attributed to the R&W sites having much greater perennial, herbaceous, and hydrophytic species cover. The higher foliar cover in the R&W sites becomes more pronounced when we take into consideration the similarity between the two groups in annual species cover and that the Terrestrial sites had greater woody and upland species cover. The perennial, herbaceous and hydrophytic cover in the R&W sites must overtake those in woody and upland cover to account for the vast difference in foliar cover.

The greater woody cover in Terrestrial sites was somewhat surprising, as many wetlands and riparian areas can have dense willow cover. However, this discrepancy is likely due to the inclusion of both herbaceous wetlands and riparian shrublands in the R&W sites. The majority of R&W sites have little to no woody cover (Figure 3). R&W had a median woody cover of 2.7 percent, while the terrestrial sites had a median woody cover of 18.7 percent (Figure 3). This is in agreement with other shrub cover estimates of rangelands in the west (Blaisdell et al., 1982; Davies et al., 2006; Kleinhesselink et al., 2023), especially in areas that have been historically grazed, where preferential consumption of grasses and forbs allowed unpalatable shrubs to flourish and establish dominance (Anderson & Inouye, 2001).

The differences in cover of these functional groups indicate that the R&W sites are wetter than those selected from the Terrestrial data. In the west, wetlands are often highly productive compared to the surrounding upland systems, largely due to water availability and nutrients that are transported via surface or groundwater flows (Belsky et al., 1999; Kauffman & Krueger, 1984; Wohl et al., 2021). Wetlands are frequently dominated by perennial species of sedges, rushes and wetland adapted grasses that are rhizomatous and often form dense mats (Allen-Diaz, 1991; Culver & Lemly, 2013; Martin & Chambers, 2001; Ramstead et al., 2012). As a result, these areas would likely have higher foliar cover than surrounding upland areas, which are predominantly arid and semi-arid systems (Kleinhesselink et al., 2023) that have been estimated to have an average bare ground cover of 45% (Rigge et al., 2020).

The differences in floristic metrics between the Terrestrial sites and the R&W sites indicate that there may be disparities in the levels of disturbance occurring between the two groups. The distinction between herbaceous, perennial, and hydrophytic vegetation between the two groups suggests that the R&W sites tend to have more lush, herbaceous vegetation than the wetlands found in the Terrestrial database, and therefore may be more intensely used by livestock. Livestock have also been shown to increase the spread of invasive species, through seed dispersal and defoliation of native species (Fleischner, 1994), which may explain why the R&W sites had greater noxious species cover than the Terrestrial sites. The fact that the Terrestrial sites had lower noxious cover, greater mean C values, and greater percent intolerant species may be an indication that they are less impacted than the R&W sites, as these metrics have been demonstrated to be negatively correlated with levels of disturbance (Chipps et al., 2006; Jones, 2005; Mack, 2004; Rocchio et al., 2007). Previous research on livestock grazing patterns has also demonstrated preferential use of riparian and wetland areas due to higher forage quality than surrounding uplands (Gillen et al., 1984; Harris et al., 2002).

An alternative hypothesis for these differences in floristic quality is that the Terrestrial sites may be more disconnected from their floodplains than the R&W sites, impacting species' dispersal abilities. Perennial cover was higher in R&W sites, while annual cover was similar between the two groups. Previous research has demonstrated that perennial wetland species are dispersal limited compared to annual species (O'Connell et al., 2013). Perennial species tend to produce relatively few seeds when compared to annual species, and perennial seeds tend to be dispersed by water or small mammals while annual seeds are dispersed more widely by waterfowl and wind (Chang et al., 2005; O'Connell et al., 2012; Shipley & Parent, 1991). In systems that are disconnected from their floodplains, hydrologic isolation further inhibits perennial seed dispersal by water. Similar to the results presented here, O'Connell (2013) also found no differences in annual species richness between isolated wetlands and well-connected reference conditions.

A final possible explanation for R&W sites appearing wetter may just be the result of the inclusive approach used to select Terrestrial sites for this study. In attempting to capture sites that may have had portions of wetland but were not fully centered in a wetland, the resulting data may characterize plots that crossed wetland-upland boundaries. Hence, hydrophytic species cover would be inherently lower than if a plot had been centered on the wetland. This may not mean that the wetland itself is drier or of lower quality, but just that it was not fully captured, and the data was skewed by the inclusion of adjacent upland systems. Additionally, standard wetland identification and delineation requires three attributes to be characteristic of wetlands: hydrology, vegetation, and soils (Tiner, 1991). While I tried to capture sites that may have had evidence of inundation of water by selecting sites with greater than 2% water cover and/or a proximity to mapped water bodies, there was no way to identify sites with hydric soil indicators in the Terrestrial dataset. As a result, I predominantly relied on vegetation as the identifier of wetlands, which is inherently limited (Tiner, 1991).

Although the differences in mean C and FQAI were statistically significant between the two datasets, the actual value of those differences was small. For instance, C values are on a scale from 0 to 10, but the difference between the medians of the two groups was only 0.69. Such a small difference is unlikely to be an ecologically meaningful discrepancy. Similarly, median FQAI values, which ranged from 0 to 30 for this data, differed by 0.5, indicating that the groups are ecologically very similar, despite the statistically significant difference. Furthermore, a mean C value around 4 and an FQAI around 30 are indicative of highly disturbed systems (Andreas & Lichvar, 1995; Rocchio et al., 2007). Thus, despite the groups having statistically different values, the metrics indicate that both groups are indicative of highly disturbed systems.

2.4.1 Drivers of Riparian and Wetland Areas Sampled with the R&W AIM Program

The cover and distribution of water appears to be a major determinant of certain floristic metrics. Sites with greater extent of surface water or total water cover were consistently associated with greater perennial species cover, lower species richness, and lower upland species cover than sites with less consistent water cover throughout the plot. However, sites with greater extent of surface water also had lower evenness than sites with a lesser extent.

Sites with a greater variety of geomorphic features, such as riparian shrublands, have more heterogeneous surface water patterns (Wohl, 2016). Riverine systems are inherently geomorphically complex due to their disturbance regimes (Naiman et al., 2005), such as flooding and channel migration (Choné & Biron, 2014). Regular disturbance in these systems naturally increases heterogeneity of substrate size, nutrient availability, and connection to the floodplain (Baniya et al., 2020; Wohl et al., 2021). For instance, Choné and Biron (2014) found that channel migration is strongly linked to habitat diversity. Such diversity allows more opportunities for colonization by different species, both hydrophytic and upland, and reduces the dominance of mat-forming perennial hydrophytes. This heterogeneity promotes species richness, as it provides variation in depth to the water table, distance to a channel, and heterogeneity in nutrient and light availability.

Sites like fens and marshes, on the other hand, tend to occur in flatter or depressional locations that result in more homogenous water coverage. This more consistent water coverage led to greater coverage of hydrophytic, perennial vegetation, which tends to be rhizomatous, patchy and mat forming (Hoag et al., 2001). These sites therefore tend to be dominated by a few dominant species (McIlroy & Allen-Diaz, 2012), which consequently reduces species evenness. This homogenous water coverage likely also limits species colonization to hydrophytic species that are tolerant of anaerobic soil conditions (W. Mitsch & Gosselink, 2015) and inhibits growth of species that cannot tolerate consistently shallow water tables (Martin & Chambers, 2001), which may result in lower species richness (Wohl, 2016).

Fens and marshes had surprisingly similar characteristics, but ultimately differ due to large differences in elevation. For instance, the two wetland types had similar species evenness, species richness, and upland species cover. However, the vast differences in species composition are likely due to the large difference in elevation between the two groups, where fens are located at higher elevations. While there were no statistically significant differences in water temperatures among wetland types, median water temperatures for fens were 8.8 degrees Celsius lower than that of marshes. This likely corresponds to the difference in elevation, as ambient air temperatures are highly correlated with elevation (Collados-Lara et al., 2021), as well as differences in hydraulic conductivity between the wetland types. Fens have consistent water flowing through them, while depressional marshes tend to have smaller throughflow of water. The NMDS provides further evidence of this trend, as marshes tend to fall in the direction of increasing mean water temperature, while fens tend to be in the opposite direction. Lower temperatures hinder microbial decomposition and therefore contribute to accumulation of organic matter in fen soils (Marschner, 2021), which demonstrates one reason why fens occur at higher elevations than marshes.

It is likely that some, if not many, of the marshes were constructed depressional wetlands that have either been impounded or excavated to provide easy access to water for livestock and wildlife, a common practice on BLM land (Bull et al., 2001). Prior research has indicated that constructed marshes in Colorado may take over 50 years to meet floristic metric values similar to those of natural reference marshes and tend to have less than half the species richness of reference marshes (Gutrich et al., 2009). Marshes also had high specific conductance, impacting the floristic quality of these sites. Specific conductance increases with increasing temperatures and surface water, common for these lower elevation wetlands, because higher temperatures increase evaporation, which in turn increases the salinity of the water (Belsky et al., 1999). This combination of factors limits the floristic quality of marshes in this region.

Although water table depth has been shown to be one of the strongest determinants of species composition in wetlands (Allen-Diaz, 1991; Hammersmark et al., 2010; Loheide & Gorelick, 2007; McIlroy & Allen-Diaz, 2012), metrics associated with water table depth (soil pit water depth and depth to saturated soil) were not significantly correlated with species composition in this study. A potential explanation for this is that data for the R&W program is collected as a single snapshot in time; technicians visit a site once per year, and timing is heavily dictated by sampling efficiency across the landscape rather than ideal timing for hydrologic characterization for a given site. The pilot data used in this study had no repeat data on water table depth. Previous research on the connection between water table and species composition tracked water table depths at multiple locations throughout a growing season (Allen-Diaz, 1991; Hammersmark et al., 2010; Loheide & Gorelick, 2007; McIlroy & Allen-Diaz, 2012). Within this dataset, surface water coverage may be a better proxy for water table dynamics at a site than a one-time water table depth measurement.

Differences in disturbance metrics indicate that spring/seeps may receive more pressure from livestock and ungulates than other wetland types. Spring/seeps had greater percent grazed and soil alteration values compared to all other groups except wet meadows. Spring/seeps also had elevated total nitrogen and total phosphorus levels, which are positively correlated with disturbance (Herlihy et al., 2019). While the NMDS showed significant similarities between species composition of spring and seeps and riparian shrublands, riparian shrublands had significantly greater woody cover than spring/seeps. Research has demonstrated that cattle preferentially use wet meadows and riparian areas that have greater forage quality than surrounding uplands. However, specific grazing patterns are highly dependent on ease of access, particularly slope and lack of thick brush or other obstructions (Harris et al., 2001; Johnson et al., 2016). While spring and seeps in this dataset had significantly greater slope than riparian shrublands, the slope documented by technicians is the slope in the predominant direction of water flow at each site. In a riparian setting, this would not capture the slope of banks down to a channel, which are known to be highly influential to cattle grazing patterns. Spring and seeps likely have gentler cross-sectional slopes, allowing cattle easier access to water. Additionally, the fact that they have lower woody cover than riparian shrublands indicates that they likely have less dense brush that discourages cattle from accessing water. These spring and seeps may provide easy access to water and high-quality forage, while the moderate amount of woody species cover provides shade that is generally less abundant in wet meadows.

Interestingly, this potentially greater amount of utilization did not lead to any differences in floristic metrics between springs and seeps and the other wetland types. There are a few potential explanations for this. First, previous research has found that species composition in wetlands is not affected by low to moderate grazing regimes (McIlroy & Allen-Diaz, 2012), especially when there is ample water supply (Allen-Diaz & Jackson, 2000). This could indicate that spring and seeps are not being utilized to a degree that is detrimental and that water levels were enough to buffer against detrimental effects of grazing. It is also possible that metrics that are less directly connected to the effects of utilization by livestock/ungulates are not being picked up in the wetland type comparisons. For instance, Allen-Diaz (2000) found that herbaceous cover was sensitive to grazing intensity in springs, especially under drought conditions. However, there are likely inherent differences in herbaceous cover among the different wetland types, meaning a signal of decreased herbaceous cover due to grazing would be difficult to detect in such a comparison across ecosystem types.

It is also important to note that the metrics that showed the signal that springs and seeps may be more heavily utilized than other wetland types are considered "annual use" metrics, as they are heavily dependent on the timing and intensity of grazing during the growing season. If a site is sampled by field technicians early in the season prior to cattle arriving in the area, the site would likely have low percent grazed and limited soil alteration. However, if technicians visit a site during or right after cattle are present at a site, these metrics would likely be much more pronounced. Further analysis of these metrics over multiple years would help determine if the signal detected in this study persists over multiple years, or if the high percent grazed and soil alteration in this dataset is merely due to springs and seeps being sampled more frequently after cattle grazed a site than other wetland types simply due to happenstance.

Alternatively, it is possible that all sites sampled were impacted to a degree that any disproportionate disturbance to springs had an inconsequential effect on the floristic metrics. FQAI values and mean C values for all wetland types were indicative of highly impacted systems (Miller & Wardrop, 2006; Rocchio et al., 2007). It is possible that any additional pressure on spring/seeps is not resulting in further detriment to the floristic quality of the sites.

These findings are consistent with recent findings from data collected through the NWCA monitoring program. These studies showed that wetlands in the west were the most highly impacted (Magee et al., 2019), and had the most pervasive disturbance of wetlands in any other region in the nation (Lomnicky et al., 2019). In line with these results, wetlands in the west also had some of the worst vegetative quality of wetlands on a national scale (Magee et al., 2019). Furthermore, the data used for this study was collected exclusively on BLM lands, which tend to be more disturbed than surrounding lands as a result of the agency's historical focus on grazing, mining, and oil and gas production (U.S Department of Interior & Bureau of Land Management, 2022). As a result, the findings in this study are consistent with what would be expected based on other large-scale wetland monitoring programs and known historical management of BLM lands.

2.4.2 Takeaways

Sites from the Terrestrial database that were identified as wetlands were distinctly different than sites sampled with BLM's new R&W monitoring program, indicating that the

Terrestrial AIM program did not effectively sample wetlands in its first 10 years of operation. On average, Terrestrial AIM sites had much lower foliar cover and less cover of hydrophytic plant species. This study demonstrates that in order to sample riparian and wetland areas on BLM land in a way that provides sufficient data for management decisions, a unique monitoring program that specifically targets riparian and wetland areas is necessary. Although the Terrestrial program is sampling wetlands at a rate proportional to the distribution of these areas across the landscape throughout the American west, it has not provided a thorough characterization of the range of variation existing within these sites. Wetlands and riparian areas provide unique services that far outweigh their prevalence on the landscape and must be assessed and monitored as such.

Of the riparian and wetland areas sampled with the R&W program, the main drivers of plant community composition were elevation and the degree of heterogeneity of water coverage. Utilization by cattle or native ungulates also appears to be influential to these systems, decreasing the C value of the plant communities, though further investigation may be necessary to fully understand its impact on species composition. There is evidence that spring/seeps are receiving more grazing pressure from livestock and/or native ungulates than other wetland types, but the preliminary data does not indicate that they are of lower vegetative quality as a result. However, vegetative data of all the sites assessed in this study indicate that the wetlands sampled on BLM land are representative of highly impacted wetlands.

This study provides the first analysis of pilot data collected by the Riparian and Wetland AIM program. The analyses presented here serve as a baseline for the status of wetlands and riparian areas sampled and identifies the various important drivers of plant communities. Further research is needed to fully understand the effects of annual water fluctuations, regional aridification, and utilization by livestock on the health of these systems.

3. CONCLUSION

The results from this study provide evidence that the BLM's wetland- and riparian-specific AIM program is necessary to effectively monitoring the quantity, ecological health, and trend of wetland and riparian areas on BLM land. The Terrestrial program, as a landscape scale monitoring program, only sampled wetlands or riparian areas less than one percent of the time. While this may be nearly representative of the true distribution of wetlands and riparian areas on BLM land in the lower 48 (Dahl, 1990), the sites captured represented a population that was distinctly different from the R&W sites.

Additionally, prior research has demonstrated that, in the West, wetland and riparian areas are utilized more intensely by livestock than their surrounding uplands (Gillen et al., 1984; Harris et al., 2001). As a result, it is important to include sampling methods that capture the disturbance by livestock to provide context to plant community composition, soil, and water quality data. The R&W data on the percent of graminoids grazed and the number of soil altering hoof prints at a site were significantly correlated with the NMDS distribution, demonstrating the influence of grazing on riparian and wetland systems. Similarly, water-specific sampling provided insight into the influence of variables such as water quantity and distribution, specific conductance, and concentrations of nitrogen and phosphorus on these systems, data which was unavailable using the Terrestrial data. These findings indicate the necessity for specific riparian- and wetland-focused monitoring efforts, particularly in a region like the American West, where they are rare and require a targeted sampling design to be effective.

The results of this study further demonstrated that elevation and the distribution of water were the most influential variables on plant community composition of R&W sites, though disturbance of sites by livestock and native ungulates was apparent in the data. This was particularly true of spring/seep systems, which had evidence of additional pressure of grazing when compared to other wetland types. Despite lack of evidence of degradation due to this additional pressure, land managers may want to pay particular attention to spring/seep systems to ensure future degradation does not occur, particularly under drought conditions when water may not be available to serve as a buffer (Allen-Diaz & Jackson, 2000).

Finally, floristic metrics for majority of the sites sampled suggest that they are characteristic of highly disturbed systems (Mack, 2004; Rocchio et al., 2007). This is consistent with other research on wetlands in the west that has found them to be some of the most highly impacted (Lomnicky et al., 2019) and lowest quality wetlands in the nation (Magee et al., 2019). As the first large-scale assessment of wetlands and riparian areas on BLM lands, the findings of this study can not only serve as a baseline for wetland condition for future research, but also inform agency-wide management towards functioning and resilient ecosystems.

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