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Effects of Landscape Age and Salinity on Plant Community Composition and Productivity in Opportunistic and Constructed Wetlands in the Athabasca Oil Sands Region, Alberta

by

Ashlee Dawn Mombourquette

A THESIS
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Abstract

Wetlands comprise 65% of the Athabasca Oil Sands Region (AOSR) surface mineable area and thus support diverse flora (approximately 400 species in Alberta). Due to increased anthropogenic land disturbance activities such as bitumen extraction, reclamation of surface mineable areas will also increase. The resulting reclaimed areas will tend to be sodium-enriched compared to pre-disturbance landscapes.

In this thesis, forty young (<40 years old) stratified-randomly selected wetlands were sampled on reclaimed landscapes at Syncrude's Mildred Lake lease and from reference wetlands in adjacent areas in the AOSR to determine how salinity and age influence the vegetation community composition, and the biomass of six dominant wetland plant species (*Carex aquatilis, Calamagrostis canadensis, Carex atherodes, Carex utriculata, Schoenoplectus tabernaemontani,* and *Typha latifolia*) found on reclaimed and reference landscapes. Wetland vegetation communities on reclaimed landscapes differed from those on reference landscapes; however, landscape type had no impact on the biomass of the dominant plant species. Vegetation communities varied along a salinity gradient; species richness was negatively associated with salinity, and vegetative species abundance (percent cover) was lowest in freshwater wetlands and highest in moderately brackish wetlands. Species richness did not differ among wetland classes. However, vegetative species abundance was significantly different among age classes. The biomass produced by each dominant species did not vary with respect to a gradient of salinity or among wetlands of different age classes. These findings may provide a useful frame of reference against which to compare vegetation communities that may be observed in wetlands forming on reclaimed landscapes.

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Preface

This thesis is original, unpublished, and independent work by the author, Ashlee Dawn Mombourquette.

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Chapter 1: General Introduction and Thesis Overview

Introduction

The Athabasca Oil Sands Region (AOSR) comprises approximately 66% of the total area (142,000 squared kilometers (km^2)) of land in Alberta underlain by oil sands. Of the total area, approximately 4800 km^2 is surface mineable. The remaining reserves are recovered through in-situ processes such as steam-assisted gravity drainage, which heats the reservoir where bitumen is located, allowing it to flow to a vertical or horizontal wellbore (Government of Alberta (GoA), 2017). Oil sands extraction is a multibillion-dollar industry in Alberta, employing many Canadians and providing energy in the form of crude oil to domestic and international markets (Foote, 2012).

Oil sands are comprised of inorganic material (sand, silt, and clay), bitumen and water (Kannel and Gan, 2012). Bitumen is extracted from the sands via the Clark process. The Clark process uses caustic soda (NaOH) and hot water to separate the bitumen from inorganic material (Kannel and Gan, 2012). After the bitumen is extracted a tailings slurry remains, commonly known as oil sands process water (OSPW), which is comprised of sands, silts, clays, water, dissolved ions (primarily sodium and sulfate), trace metals, and other organic (i.e., naphthenic acids, and polyaromatic hydrocarbons) and inorganic compounds (BGC, 2010; Kannel and Gan, 2012). Due to the large volume of freshwater required to extract bitumen from the sands, there is a need to recycle OSPW to reduce water demand (BGC, 2010). However, this practice also increases the concentration of constituents of concern in OSPW. The tailings slurry is largely stored in large ponds, which allow the sands to rapidly settle relative to the fines (silt and clay particles, and bitumen), leaving the fluid fine tailings to settle, which can take decades or centuries to form a semi-solid material with a surface that is trafficable (BGC, 2010).

Tailings reclamation practices consists of aquatic closure, which uses pit lakes designed to store and remediate OSPW through bioremediation and dilution, and terrestrial closure, which uses consolidated tailings (tailings sand, fine tailings, and gypsum) capped with sand to create an upland area (BGC, 2010).

In this thesis I explore how salinity and age of wetlands influence plant community composition and productivity. The overall question of my thesis is: how do age and salinity influence vegetation community composition and aboveground biomass in newly formed wetlands on reclaimed and reference landscapes?

Wetlands

A wetland is "land that is saturated with water long enough to promote formation of water altered soils, growth of water tolerant vegetation, and various kinds of biological activity that are adapted to wet environments" (Alberta Environment and Sustainable Resource Development (ESRD), 2015). Wetlands form and are sustained by interactions among precipitation, surface water and groundwater, and evapotranspiration, soil moisture and chemistry, predominant wetland vegetation species, and external factors that interact to promote wetland persistence (Little-Devito et al., 2020). Wetlands are multifaceted systems that provide a variety of ecological functions including but not limited to flood and drought mitigation, by storing and slowing the movement of water, and acting as a water filtration system, which improves water quality (GoA, 2013; Volik et al., 2020). They also store and sequester carbon, which regulates the climate (GoA, 2013). Wetlands also provide habitat for flora and fauna (Ducks Unlimited Canada, 2015). Approximately 600 species of plants are found in wetlands across Canada (Ducks Unlimited Canada, 2015).

There are a variety of wetland classification systems used in the boreal zone (Ducks Unlimited Canada, 2015; ESRD, 2015; Stewart and Kantrud, 1971). In my thesis, the Alberta Wetland Classification System (AWCS) was used to identify wetland classes within the study area (ESRD, 2015). The AWCS identifies five broad classes of wetlands: bogs, fens, swamps, marshes, and shallow open water. Bogs and fens are considered peat forming wetlands, which have more than 40 cm of peat accumulation. Marshes, and shallow open water systems are mineral wetlands and have less than 40 cm of accumulated peat. Swamps can be considered peatlands or mineral wetlands depending on the depth of peat accumulated (ESRD, 2015). Mineral wetlands are most common in areas where there is a fluctuating water table or where salinity reduces the establishment of many freshwater vascular and non-vascular species (ESRD, 2015). For this study only marshes and shallow open water wetlands will be investigated.

Boreal marshes and shallow open water wetlands are often characterized as having three distinct concentric vegetation zones or communities whose distribution is determined by water depth or soil saturation. The three zones are (1) open water (supporting submergent and floating-leaf vegetation; ESRD, 2015), (2) emergent (comprised of erect plants whose roots are typically submerged; ESRD, 2015), and (3) wet meadow (areas where the soil is intermittently saturated with water; ESRD, 2015). Wet meadow and emergent zones are the focus of this thesis.

Wet meadow zones are characterized by water-saturated soils with a shallow (<2 cm) fluctuating water depth (Roy et al., 2016). Wet meadow zones are dominated by herbaceous vegetation during the initial stages of wetland succession. As the wetland ages, other distinctive vegetation seres (communities) form, with the wet meadow zone remaining on the fringe of the wetland (Stewart and Kantrud, 1971; van der Valk, 1981). Wet meadow zones commonly border marshes and shallow open water wetlands, where the soils are periodically saturated, and have short periods of shallow inundation (ESRD, 2015). In natural areas of the AOSR, the wet meadow zone is comprised of sedge, grass, and forb species (Raab and Bayley, 2012).

The emergent zone is situated between the wet meadow zone and the shallow open water zone (Raab and Bayley, 2012) and is typically inundated with water. Emergent zones are characterized by vegetation that can withstand variable water levels and prolonged flooding, such as cattails, rushes, sedges, and grasses (ESRD, 2015; Raab and Bayley, 2013). Vegetation composition also varies depending on salinity levels and often includes species that are tolerant to saturated conditions. Species composition can also be influenced by wetland type and permanence (ESRD, 2015; Roy 2014; Steward and Kantrud, 1971).

Northern Alberta Wetlands and Reclamation in the Oil Sands

Almost all of wetlands in the Athabasca Oil Sands Region (AOSR) surface minable oil sands area are peat-forming (i.e., fens and bogs; Hawkes et al., 2020). However, human activities, including resource extraction and forestry operations (Alberta Biodiversity Monitoring Institute (ABMI), 2020; Hawkes et al., 2020) have had various impacts to the environment, ranging from altering hydro-ecology to the full removal of wetlands within these areas (Nwaishi et al., 2015). Further, the extent of AOSR landscape disturbance (and reclamation of disturbed areas) is projected to increase over time due to continued surface mining and in-situ (below ground) bitumen extraction activities (Government of Canada, 2016).

Mined areas in the AOSR must be reclaimed to equivalent land capabilities (GoA, 2017). “Equivalent land capabilities” means that after area has been reclaimed, the land must support uses that existed prior to mining activities, such as providing habitat for wildlife (Raab and Bayley, 2013). Reclamation activities include a ‘dry option’, consisting of filling former open pits with oil sand process material (OSPM), capping the areas with tailings sand and overburden, and configuring the surface to produce uplands   The primary goal of early upland reclamation efforts is to create forested landscapes (Kelln et al., 2009; Magas 2020; Rowland et al., 2016). However, several pilot investigation studies have been conducted to determine the feasibility of constructing watersheds that can promote wetland formation. Three examples include Sandhill watershed

(Hartsock et al., 2021; Vitt et al., 2016; Wytrykush et al. 2012) at Syncrude Mildred Lake, Wapisiw Marsh (Daly et al., 2012) and Nikanotee wetland (Ketcheson et al., 2016) areas at Suncor Base Plant. Upland forest reclamation practices have become relatively well established (Kelln et al., 2009; Rowland et al., 2016), but recent surveys have also documented that uplands are also sites of newly formed 'opportunistic' wetlands (Little Devito et al. 2019), occupying up to 18% of reclaimed upland landscapes (Hawkes et al., 2020). However, limited research has been conducted on the formation or characteristics of opportunistic wetlands on reclaimed landscapes (Hawkes et al., 2020; Kovalenko et al., 2013).

Opportunistic wetlands develop where hydrological features and water balance together result in a newly wetted area on disturbed landscapes (Daly, 2011). Knowledge of opportunistic wetlands is limited because of their recent origins in the AOSR. The oldest reclaimed area in the AOSR is Gateway Hill which was created in the 1980s (Magas 2020). Additionally, early succession occurs over decades, often beyond funded scientific study periods, which further limits knowledge of early wetland succession.

Functional success of constructed and opportunistic wetlands so far is promising (Borkenhagen and Cooper, 2019; Hawkes et al., 2020; Vitt et al., 2016). Borkenhagen and Cooper (2019) noted that transplanting bryophytes and vascular plants into a constructed wetland allowed successful establishment of vegetative species, indicating fen species can successfully introduced into post-mined landscapes.

Newly Formed Wetlands

Newly formed wetlands are wetlands that have been built or have otherwise naturally formed on a disturbed landscape (Daly, 2011). Opportunistic wetlands (OW, herein) can form on reclaimed landscapes due to slight irregularities in soil placement and settlement, which can result in wet to dry microsites (Hawkes et al., 2020). Reclaimed areas are capped with a peat-mineral mix or fine textured soils; the peat-mineral mix can absorb surface water from precipitation, whereas the fine texture soils that are less permeable promote surface saturation (Hawkes et al., 2020). Wet microsites can form due to pooling of water in areas that have differentially settled (Daly, 2011; Little-Devito et al., 2019) and as subsurface hydrological networks develop (Hawkes et al., 2020), these wet microsites can provide habitat suitable for the establishment of hydrophilic vascular vegetation and non-vascular bryophytes. Common wetland classes that have formed on reclaimed landscapes include shallow open water, marshes, and swamp wetlands (Hawkes et al., 2020). Constructed wetlands, on the other hand, are wetlands that have been built with the intent of having sustainable freshwater and nutrient sources and inputs into the wetland (Daly, 2011). To date, constructed wetlands have primarily been shallow open water and marsh wetlands (Hawkes et al., 2020).

A challenge with reclaimed landscapes in the AOSR is the level of salinity that may develop and remain in surface waters that occur in the post-mining landscape due to the use of naturally saline soils, and tailings materials created during oil sands bitumen extraction processes (Glaeser et al., 2015; Mollard et al., 2015; Trites and Bayley, 2009a). Many native boreal plant species are sensitive to elevated salinity levels, which can result in different community structures and composition, especially in OW compared to reference boreal areas (Glaeser et al., 2015; Purdy et al., 2005; Trites and Bayley, 2009a). The community composition of OW is similar to salt-tolerant plant communities found in sodic wetlands of the Great Plains (Purdy et al., 2005).

Wetland Age and Influence on Community Composition and Aboveground Biomass

During the earliest successional stage in wetlands (i.e., wetlands less than three years of age), vegetation zonal formation is closely associated with the height of the water table (Vitt et al., 2016). As wetlands age, a sequence of vegetation seres (community types) is expected to occur (van der Valk, 1981), producing distinct vegetation zones (shallow open water, emergent, and wet meadow zones) defined by Stewart and Kantrud (1971) for prairie pothole wetlands and observed in ESRD (2015).

Along with the development of zones in wetlands between 3-35 years old, Raab and Bayley (2013) found that aboveground biomass in reclaimed wetlands was similar to that found in non-anthropogenically disturbed wetlands that had existed for over 1000 years.

Distribution of Salinity in the Environment

Salinity describes the quantity of salts present in a substance, expressed as a concentration (Alberta Environment, 2001). The resulting reclaimed landscape varies in salinity due to the distribution of naturally high saline concentrations. Salts may occur naturally in soils and groundwater prior to mining activities associated with a variety of influencing factors such as estuarine and marine sediment deposits, deep saline aquifers, and from post mining extraction processes of bitumen from the oil sands (Trites and Bayley, 2009a; Vitt et al., 2020; Roy et al., 2016). Sodium, calcium, and sulphate are the principal salts contributing to the elevated salt concentrations in the post mining landscape (Biagi et al., 2019). Conductivity is used as a proxy for salinity in water and can be a general indicator of productivity of wetland systems (ESRD, 2015). ESRD (2015) has classified wetlands into six specific conductance classes (Table 1.1).

Table 1.1 Alberta Environment and Sustainable Resource Development Wetland Salinity Classification System.

Wetland Classification	Specific Conductance (µS/cm)
Freshwater	<500
Slightly Brackish	500 – 2,000
Moderately Brackish	2,000 – 5,000
Brackish	5,000 – 15,000
Subsaline	15,000 – 45,000
Saline	>45,000

Newly formed wetlands on reclaimed landscapes will likely have elevated sodium, chloride, and sulfates, with a likely conductivity range of 4700 – 7800 µS/cm (Trites and Bayley, 2009a), which are classified as moderately brackish to brackish wetlands (ESRD, 2015). Understanding how plant community composition and productivity vary with salinity may indicate ecosystem recovery on a post-mined landscape (Phillips et al., 2016). As reclaimed landscapes were built using sodic overburden newly formed wetlands that form on these landscapes may also have large ranges in salinity (Trites and Bayley, 2009a), which is important because saline wetlands are relatively uncommon in the boreal region and with newly formed wetlands on reclaimed landscapes expected to be more saline the presence of saline wetlands on the post-mining landscape is expected to increase.

Salinity Influences on Plant Community Composition and Aboveground Biomass

Plant species composition can often be used as an indicator of environmental gradients such as salinity (ESRD, 2015). Sodium, calcium, and sulphate salts can result in ecological effects such as vegetation stress, which can influence plant growth (Biagi et al., 2019; Howart 2000). However, species can adapt to tolerate elevated salinity levels, resulting in unique wetland community compositions (ESRD, 2015; Stewart and Kantrud, 1971). Plant species use osmotic tolerance, which changes the level of salinity within the cells and around the roots to exclude salts from their roots and tissues. They also develop tissue tolerance – the tolerance of plant tissues to the accumulation of salts when elevated salt concentrations occur (Munn and Tester, 2008).

Several studies conducted in the AOSR have investigated if the community composition differs between OW and constructed wetlands created following oil sands mining, compared to natural areas (Borkenhagen and Cooper, 2016; Raab and Bayley, 2013; Raab and Bayley, 2014; Rooney and Bayley, 2011; Roy et al., 2016; Trites and Bayley, 2009;). Trites and Bayley (2009) studied marsh wetland systems on reclaimed landscapes and in natural areas. Many vegetation communities on reclaimed landscapes resembled that of natural reference areas, where marsh wetlands were dominated by *Typha latifolia*, *Carex aquatilis* or *Carex atherodes*, and *Lemna minor*. However, some natural wetland species (primarily submergent aquatic vegetation species and salt tolerant species) were absent from reclaimed landscapes. The absence of some submergent aquatic vegetation may be due to limited dispersal mechanisms (water flow, and/or wildlife and avian activity). There is potential for reclaimed landscapes to lack salt tolerant plants as the landscape may not be salty enough for the species to successfully establish. Roy et al. (2016) compared community composition in natural and reclaimed shallow open water and marsh systems. They found differences in emergent and wet meadow zone vegetation between natural and reclaimed wetlands. Emergent zone vegetation was influenced by physical conditions of the wetland: basin slope, water level and zone width, all were primary drivers for differences in community types between natural and reclaimed wetlands. For example, *Schoenoplectus tabernaemontani* and *T. latifolia* occurred more frequently and in higher abundances in the emergent zones of wetlands on reclaimed landscapes than in natural wetlands due to the physical conditions of the wetland. Community composition in emergent and wet meadow zones on reclaimed landscapes also differed from one another, possibly due to an abrupt transition from saturated soils to dry upland soils (Roy et al., 2016). Emergent and wet meadow zones of natural wetlands were similar and dominated by *Carex* species.

A species' tolerance to salinity can be reflected through its growth response (Munn and Tester, 2008). Mollard et al. (2013) and Vitt et al. (2020) studied how two characteristic wetland plant species, *T. latifolia* and *C. aquatilis*, performed when exposed to OSPW and elevated sodium concentrations. Mollard et al. (2013) noted that *T. latifolia* exposed to OPSW displayed reduced growth when compared to *T. latifolia* in natural wetlands. The leaves of *T. latifolia* exposed to OSPW were significantly smaller than *T. latifolia* growing in natural wetlands, and had fewer leaves per shoot (Mollard et al., 2013). *T. latifolia* growing in wetlands on reclaimed landscapes had higher sodium concentrations in their leaves and lower potassium and calcium concentrations, which are essential macronutrients for cell function. *T. latifolia's* ability to exclude sodium from its tissues created an energetic trade off resulting in reduced leaf growth (decreased biomass; Mollard et al., 2013). A similar response is found in *C. aquatilis*. Vitt et al. (2020) investigated the effects of increasing sodium concentrations on the structural and functional responses of *C. aquatilis*. *C. aquatilis*, a species commonly found in boreal fen and marsh wetlands, had a wide tolerance to sodium levels (17 to 1079 mg L^{-1}), but a clear threshold was observed between 1079 mg L^{-1} and 2354 mg L^{-1} where productivity strongly decreased. This was observed in reduced above- and below- ground biomass, stomatal conductance, transpiration, and rates of photosynthesis. Vitt et al., (2020) also noted that in natural wetlands, *C. aquatilis* is not present above sodium concentrations of 1900 mg L^{-1}, which is an important consideration as reclaimed landscapes have large ranges in salinity (Purdy et al., 2005; Trites and Bayley, 2009a).

Knowledge Gaps

The creation of a variety of newly reclaimed areas in the AOSR presents a unique opportunity to observe the establishment of vegetation in newly formed wetlands. Knowledge around ecological function of newly formed wetlands on reclaimed landscapes is limited due to the relative newness of reclaimed landscapes and to an absence of a frame of reference for evaluating these ecosystems. Characterizing reference wetland vegetation communities and their productivity can provide a frame of reference against vegetation community composition and productivity in reclaimed mining lease areas. This frame of reference will contribute to understanding how salinity and age of newly formed wetlands influence vegetation communities and their productivity. These findings will provide insights into wetland reclamation practices in the AOSR and how wetland vegetation communities change across a gradient of salinity and as they age into mature wetlands which is important for determining reclamation success.

Study Location

I assessed the community composition and productivity of plant species and associated water chemistry in 40 newly formed (opportunistic and constructed) wetlands on reclaimed landscapes and in reference areas in the Athabasca Oil Sands Region, Alberta. The newly formed wetlands included in this study were located on Syncrude Mildred Lake Lease where the landscape has been reclaimed following mining activities, also known as the reclaimed landscape, herein. Of the 40 wetlands studied 20 were on the reclaimed landscape. The remaining 20 wetlands were on a less disturbed ‘reference’ landscapes. Stoddard et al. (2006) defined three operational definitions of reference conditions: minimally disturbed condition, least disturbed, and best attainable. Minimally disturbed conditions exist in the absence of significant human disturbance, recognizing that some natural variability in indicators will occur due to other environmental factors (i.e., climate). The least disturbed reference condition represents the best available physical, chemical, and biological habitat conditions currently present and what is “best” for evaluating data collected at selected sites based on pre-define criteria. The best attainable reference condition is equivalent to what is expected for ecological conditions of least disturbed sites when the best management practices are in place for a duration of time. In this study, I apply the “least disturbed reference condition” as the measure of what the reference landscape is considered to be. Examples of newly formed wetlands on the reference landscape are locations where ponding occurs upstream of newly built roads (Saraswati, et al., 2020a; Saraswati et al., 2020b) or other hydrological obstructions that result in the formation of newly formed opportunistic wetlands. Of the 20 newly formed wetlands on the reclaimed landscapes, 17 were opportunistic and three were constructed wetlands. On the reference landscape, seven of the newly formed wetlands were opportunistic wetlands and 13 were constructed wetlands. All newly formed wetlands were assessed in July and August 2021 (Figure 1.1).

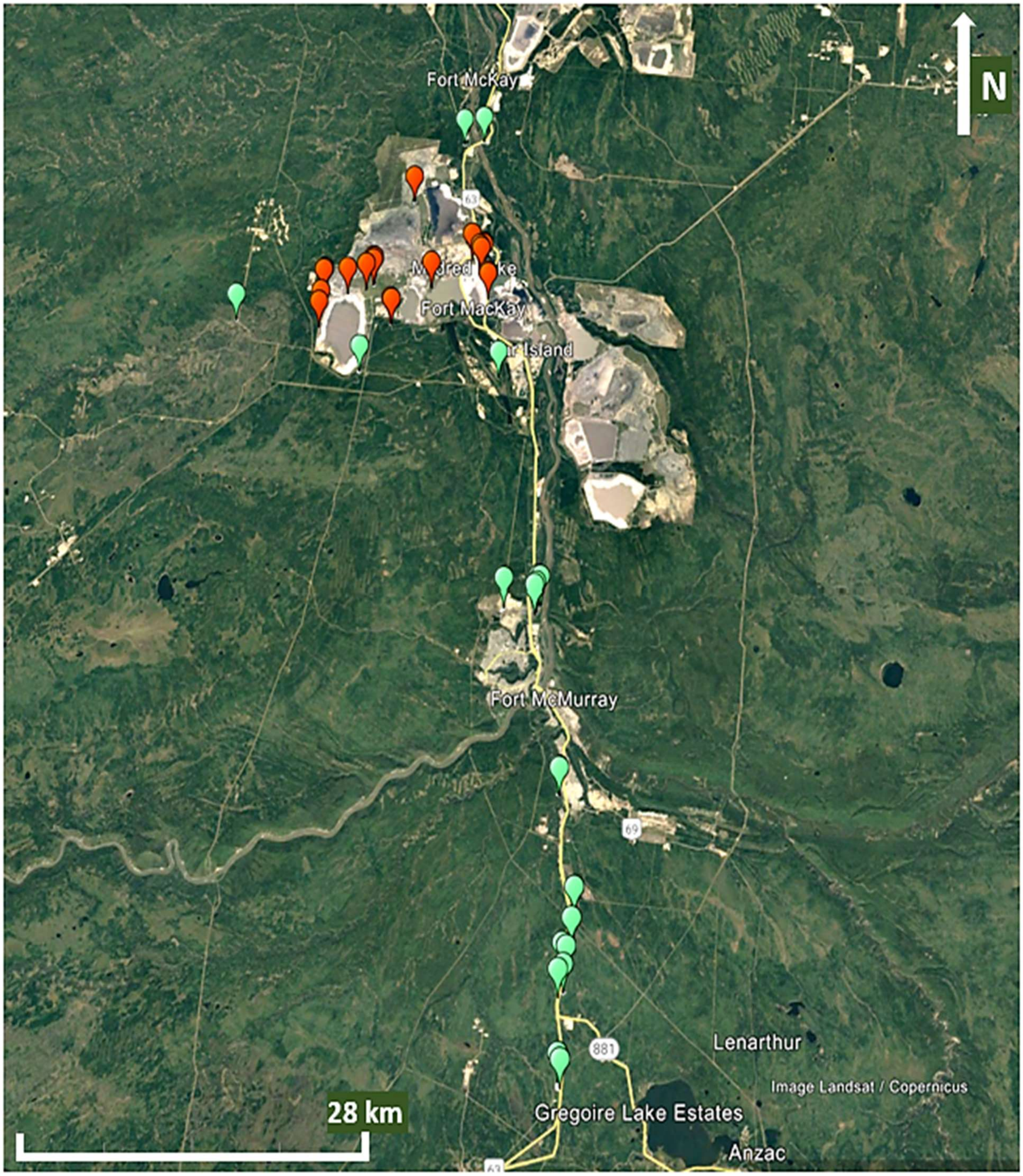

Figure 1.1 Map of wetland study sites on reclaimed landscapes (red markers; n = 20) and on reference landscapes (green markers; n = 20) in the Athabasca Oil Sands Region (imagery from Google Earth Pro, 2023).

Thesis Objectives and Outline

The overall question for this thesis is: how do age and salinity influence vegetation community composition and aboveground biomass in young wetlands on reclaimed and reference landscapes? To answer this overall question the thesis is divided into two sub-objectives:

1) Compare the vegetation community composition in newly formed wetlands of various ages and across a gradient of salinity on reclaimed and reference landscapes; and
2) Assess how age and a gradient of salinity influence dominant plant species' aboveground biomass in wetlands on reclaimed and reference landscapes.

The remaining chapters of my thesis are:

Chapter 2: Vegetation Establishment in Young Wetlands Along Gradients of Age and Salinity

In this chapter, the first objective was addressed by determining the community composition of forty wetlands of various ages and across a salinity gradient. This chapter compared wetland vegetation communities in wetlands of various age, and across a gradient of salinity among wetlands on reclaimed and reference landscapes.

Chapter 3: Influence of Age and Salinity on Plant Productivity (aboveground biomass)

Observations of biomass (as an indicator of productivity) of the dominant plant species were examined to determine how biomass varies across age and salinity gradients in young wetlands of reclaimed and reference landscapes. The results addressed the second objective.

Chapter 4: Synopsis, Conclusion, Applications and Implication, Future Research

I reviewed the major findings of how salinity gradients influence community composition and productivity of wetlands ranging in age (years). I suggested ways in which the findings can be used as a part of future guidance for wetland reclamation practices in oil sand landscapes. I suggest future research that can be undertaken to determine which species should be planted on reclaimed saline landscapes to promote the development of newly formed wetlands due to their ability to tolerate saline environments.

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Chapter 2: Vegetation Establishment in Wetlands Along Gradients of Salinity and Age on Reclaimed and Reference Landscapes

Introduction

Petroleum resources, especially bitumen extracted from oilsands, are an important energy source for the province of Alberta. Bitumen is extracted in two ways: through surface mining when deposits are relatively near the surface, and by employing below ground in-situ processes for deeper deposits (Government of Alberta (GOA), 2017). Bitumen near the surface (that is surface mineable) underlies approximately 4800 km^2 of the Athabasca Oil Sands Region (AOSR), which is located in northeastern Alberta.

Approximately 800 km^2 of the AOSR has been disturbed due to oil sands open pit mining (Hartsock et al., 2021). Open pit mining involves the removal of surface and subsurface materials to a depth of up to 100 m. Provincial regulation requires that disturbed land is reclaimed to equivalent land capabilities. This is to support land uses that follow post-mining reclamation, such that the function of the land surface is the same as it was prior to mining activities (Raab and Bayley, 2013). However, despite reclamation practices, a result of oil sands mining activities, composite tailings (oil-depleted sands and clay and extraction by-products) and salty water (i.e., water enriched in bicarbonate, sodium, sulfate, calcium, and chloride ions) are produced (Hartsock et al., 2021; Kannel and Gan, 2012). Overburden and soil are placed on top of these products to reclaim these areas. The areas are then either contoured and planted to support upland boreal forests (Hawkes et al., 2020; Little-Devito et al., 2019) or constructed to create watersheds that will support constructed wetlands (e.g., Sandhill watershed (Hartsock et al., 2021; Lukenbach et al., 2019; Wytrykush et al. 2010; Vitt et al., 2016), Nikanotee wetland (Ketcheson et al., 2016), and Wapisiw wetland (Daly et al., 2012)). The soil used to construct these landscapes is either directly placed (harvested from one area and placed in another) or stockpiled for future use, which can further alter soil properties. Stockpiling soils alters root systems and disturbs seed banks, reducing the abundance of propagules that will be present in these reclaimed landscapes once the soils are applied (Dhar et al., 2016). Stockpiling materials can also reduce the success of regeneration from plant propagules and seed germination viability (Dhar et al., 2016). These effects can result in decreased species diversity in the seedbanks on reclaimed landscapes (Cooper 2004). Plant community establishment and community development on a reclaimed landscape will be influenced by the type of capping material used (i.e., overburden, tailings sand, peat-mineral soil mix, or forest floor materials) and whether it was stockpiled or directly placed (Dhar et al., 2016).

While revegetation of reclaimed landscapes relies on germination of native species from propagules present in reclamation soils, colonization of native species via natural processes (i.e., dispersal by wind or animals), and planting with upland boreal forest species can also occur (Farnden, 2021). Revegetation processes are similar in natural areas that have experienced disturbance. Buried seeds may germinate, living vegetation may regenerate, or propagules may arrive from surrounding undisturbed areas (Dhar et al., 2016). However, propagules in reclamation soils are almost completely lost after as little as 16 months, leaving species that have colonized the stockpile surface as a substantially diluted propagule source (Cooper 2004; Farnden, 2021). Therefore, most reclaimed landscapes are planted with upland boreal forest species (Farnden, 2021; Hawkes et al., 2020), which may create differences between vegetation communities on reclaimed landscapes compared to reference landscapes that have experienced some level of disturbance (Dhar et al., 2016).

Upland boreal forest landscapes reclaimed from bitumen open pit mining contain naturally forming depressions whose poorly drained fine-textured soils cause water to pool. This promotes the formation of wetland soils and habitat suitable for vegetation species that can withstand saturated conditions. Such areas are colloquially called “opportunistic” wetlands (Little-Devito, 2019; Hawkes et al., 2020). Knowledge of newly formed wetlands, primarily concerning the formation of opportunistic wetlands on mine reclaimed landscapes is limited relative to constructed wetlands, which have been more intensively studied. Little-Devito et al. (2019) noted that opportunistic wetlands cover approximately 8% of the overall reclaimed landscape on Syncrude's Mildred Lake lease in the AOSR, and Hawkes et al. (2020) estimated that opportunistic wetlands covered approximately 17% of a 1209-ha reclaimed study area on Suncor's Base Plant lease in the AOSR. Swamps and marshes are the predominant opportunistic wetland type that have formed, with shallow open water wetlands being less dominant but still present (Little-Devito, 2019). Dominant vegetation community types found in these opportunistic wetlands are sedges (*Carex* species (spp)), bluejoint grass (*Calamagrostis canadensis*), slough grass (*Beckmannia syzigachne*), bluegrass (*Poa* spp), horsetail (*Equisetum* spp), and willow (*Salix* spp) (Little-Devito et al., 2019). The same vegetation assemblages have been successfully planted in constructed wetlands on reclaimed landscapes (Borkenhagen and Cooper, 2019; Daly et al., 2012; Hartsock et al., 2021).

The water of wetlands forming on landscapes reclaimed from oil sands mining varies greatly in salinity, reflecting the reclamation practices employed (Biagi and Carey 2019; Rooney and Bailey, 2011; Roy et al., 2015; Trites and Bayley, 2013). However, little is known about the vegetation assemblages of saline marshes and shallow open water wetlands in the boreal region, as these wetlands are relatively young (Hawkes et al., 2020). In this study, community composition in the emergent and wet meadow vegetation zones of marshes and shallow open water newly formed wetlands (opportunistic and constructed wetlands) were identified and compared between waterbodies situated on a reclaimed landscape and those developing on less-disturbed reference landscapes. Vegetation community composition was assessed by comparing the species richness and vegetative species abundance (percent cover) between newly formed wetlands on reclaimed and reference landscapes. The purpose was to determine if wetlands forming on reclaimed landscapes follow trajectories of post-disturbance regeneration species similar to those of reference disturbance sites not on a mining landscape. Providing evidence that these areas are returning to equivalent land capabilities and function, shows promise for returning the land to equivalent land capabilities, analogous to reference wetland vegetation communities. The objectives of this study were to: 1) assess how disturbance classes (reclaimed versus reference landscapes) influence vegetation community composition; 2) compare wetland vegetation community composition across a gradient of salinity on reclaimed and reference landscapes; and 3) compare wetland vegetation community composition in wetlands of various ages on reclaimed and reference landscapes.

Methods

Study Area

The study area was in the Regional Municipality of Wood Buffalo (RMWB), north and south of Fort McMurray, Alberta (Chapter 1. Figure 1.1) and in the Boreal Plains ecozone. This region is characterized by long cold winters and short warm summers, with a daily mean winter air temperature in January of -17.4^{o}C, and mean temperature in July of 17.1^{o}C. Mean annual precipitation in Fort McMurray was 419 millimeters (mm), with a mean annual rainfall of 316 mm and snowfall of 134 cm (1981 to 2010, WMO climate normal from the weather station at 56^{o}39N, -111^{o}13W; Environment Canada 2023a). The study was conducted during July and August 2021. Air temperatures were slightly above normal for that period, with warmer temperatures observed in July (18.7^{o} C) than in August, (16.1^{o} C). The total annual precipitation in 2021 was 326 mm, (daily data report for 2021 from the weather station at 56^{o}39’12N, -111^{o}13’24W; Environment Canada 2023b).

The AOSR is dominated by organic Gray Luvisolic soils (Government of Alberta, 2023) overlain by boreal forest and wetlands. Peatlands comprise 62% of the surface mineable area, with mineral wetlands comprising another 3% (Borkenhagen and Cooper, 2016; Hawkes et al.

2020). Upland forest comprises the remainder of the natural surface mineable area. Energy extraction has disturbed 2.3% (2,154 km^2) of the Athabasca oil sands surface mineable (ABMI 2020).

Study Design

The research presented entails one aspect of a proof-of-concept evaluation of the Reclamation Assessment Approach (RAA; Ciborowski, 2021), a 5-year study designed to assess the ecological condition of wetlands in the AOSR on reclamation landscapes and to develop measures of reclamation success. The hydroperiod (permanence/impermanence), topography (disturbance status), water quality (salinity, nutrients, cations and anions) and biological features (vegetation, invertebrates, avifauna) of 120 wetlands will continue to be assessed over a 3-year period.

A total of 40 newly formed wetlands were selected within the Athabasca Oil Sands Region, Alberta. Of these, 20 wetlands were on Syncrude's Mildred Lake lease (57° 2' 37.9134" N, -111° 35' 26.1636" W) in upland reclaimed areas ('on lease'). The 20 remaining 'reference' wetlands were situated in areas that were 'off lease' in the surrounding region (56° 28' 30.072" N, -111° 18' 3.9918" W (furthest south site) to 57° 8' 2.259" N, -111° 36' 8.514" W (furthest north site)). Reference wetlands were operationally defined as naturally occurring wetlands that may be subject to varying extents of anthropogenic disturbance, but that are not found on previously mined areas (e.g., forming along a newly (<40 years old) built road).

The opportunistic wetlands studied were identified by locating candidate waterbodies in recent high spatial-resolution satellite images (Google Earth Pro, 2021) prior to ground truthing. The age of wetlands (number of years since formation) was inferred using the time lapse tool in Google Earth Pro (2021) to determine the year in which persistent open water was first visible.

Wetland size (open water area and wet meadow fringe, determined from 2021 Google Earth imagery using the polygon tool) ranged from 0.04 hectares (ha) to 13.50 ha, with a mean (SE) of 1.7 (2.7) ha, and a median of 0.5 ha. A similar size range was determined for both the reclaimed wetlands and the reference wetlands (0.04–13.5 ha and 0.1–8.4 ha, respectively).

An important consideration in identifying study wetlands was to create a factorial study design representing the range of variation in wetland ages and salinities. To select suitable candidate wetlands for this study, 103 wetlands were initially visited, and preliminary measurements of water quality, including specific conductance (µS/cm) were recorded (where specific conductance (µS/cm) at each wetland was used as an indicator of salinity (ESRD, 2015)). Each wetland was then assigned to an age x salinity class. Wetlands were assigned to one of four salinity classes following the Alberta Wetland Classification System (ESRD 2015): freshwater (class 1; <500 µS/cm), slightly brackish (class 2; 500 – 2,000 µS/cm), moderately brackish (class 3; 2,000 – 5,000 µS/cm), brackish (class 4; 5,000 – 15,000 µS/cm). Wetlands were also assigned to one of five age classes: 1 - 4 years (class 1), 5 - 9 years (class 2), 10 - 14 years (class 3), 15 - 20 years old (class 4), and 21 - 40 years (Class 5). Wetland age class ranges varied as there were many younger wetlands and relative few older wetlands in the study region, reflecting the chronology of reclamation efforts in the AOSR. The use of a balanced factorial design maximized the power to assess differences in vegetation associated with salinity and wetland age, resulting in the 40 selected wetlands.

Field Data Collection

Wetlands selected for detailed study were re-visited between mid-July and mid-August to assess the vegetation community composition. To characterize the community composition across the wetland, each wetland was divided into five 72-degree radial sectors (Figure 2.1). Sampling was dependent on the accessibility of all sides of the wetland (e.g., terrain not suitable such as being too soft to walk across, or the water waist deep and was not accessible via boat) (Figure 2.1).

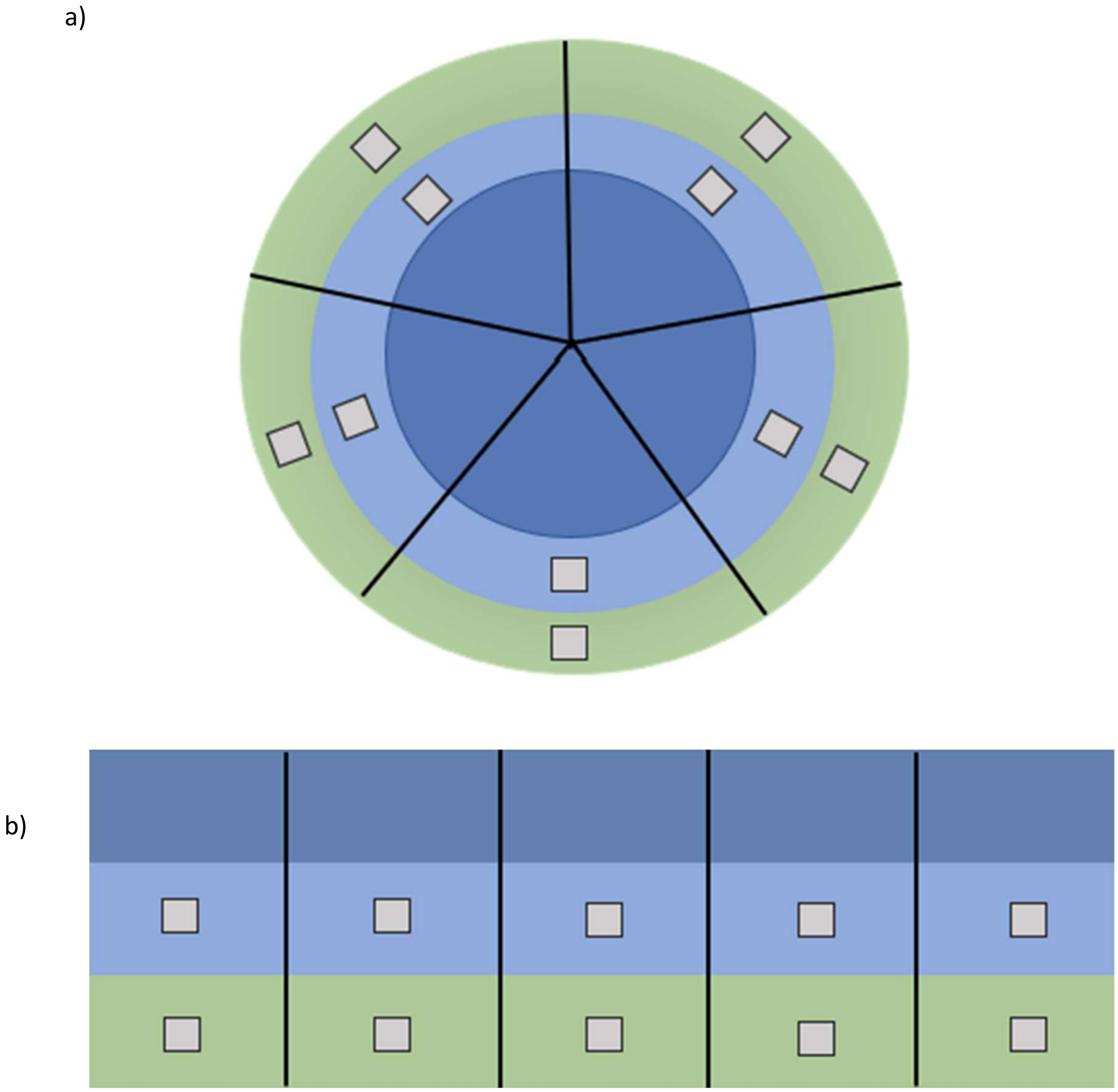

Figure 2.1 General overview of wetland sampling zones across the wetlands a) sampled based on five 72-degree radial sectors b) sampling using five contiguous rectangular sectors. The black lines delineate the boundaries of the sectors within which sample plots were assigned. Dark blue shading indicates the submergent water zone, light blue indicates the emergent zone, and green indicates the wet meadow zone. Grey boxes indicate 1 m x 1 m vegetation community composition plots. Each transect was situated randomly within each sector. Each community composition plot location was situated at the mid point of each transect.

This study focused on the community composition in the wet meadow zone and the emergent zone of each wetland. The wet meadow zone was characterized by having relatively shallow water (<2 cm) or saturated soils (Roy et al., 2016; Trites and Bayley, 2009). The emergent zone was comprised of a fringe of vegetation around the open water zone of the wetland (Roy et al., 2016; Trites and Bayley, 2009). The open water zone is an expanse of open water that typically supports submersed or floating vegetation (ESRD, 2015). The vegetation zones were visually delineated (Bayley and Rabb, 2012; Trites and Bayley, 2009)

Within each vegetation zone a transect was oriented perpendicular to the shoreline, or area of pooled water if water was present (Roy et al., 2016). To better understand the variability of community composition across the width of each zone (per wetland), zone widths perpendicular to the shoreline were first measured. A 1-meter (m) x 1-m vegetation plot was then placed at the midpoint of the emergent and wet meadow zones at which vascular plant community composition of each zone was recorded. Plots were placed at the midpoint of the wet meadow and emergent zones of each transect to avoid sampling in transitional areas between the zones (Roy et al., 2016). The plots were placed approximately 2 m away from the transect line to reduce the risk of the vegetation being trampled. Plot locations were determined using a handheld Garmin Global Positioning System (GPS) device.

Vascular species present and their associated percent cover were recorded per plot per zone, in each wetland. Mean vegetative species abundance (percent cover) in each wetland was determined by averaging the percent cover of the species in each wetland. This value was termed 'vegetative species abundance.' Species nomenclature follows the Alberta Conservation Information Management System (ACIMS 2018). Where individual plants could not be identified to species, they were recorded to genus level.

A maximum of 10 plots were placed, and species present in each plot were recorded for each wetland (5 emergent zone and 5 wet meadow zone community composition plots). Due to the shape and slope of some of the wetlands studied, some vegetation zones were not present. In such cases, a plot was not placed in one or more of the five transects per zone.

To determine variability in wetland moisture associated with availability of water within the wetland, depth to water table was measured near each vegetation plot. To determine water table depth, a small soil pit (<30 cm deep) was dug adjacent to the vegetation plot during community composition sampling. The soil pit is dug to a maximum depth of 30 cm due to ground disturbance permitting; at which point the whole would being to have water pool at the bottom depending on soil saturation and pore water availability. The distance from the soil surface to the surface of the pooled water was measured with a meter stick to the nearest cm (Table S1).

Water quality parameters, including specific conductance (µS/m), were assessed *in situ* adjacent to the plot using a YSI Proplus multiparameter meter (Table S1). Water quality parameters were assessed to understand and quantify the variability in salinity (specific conductance) among wetlands. In addition, water samples were collected for laboratory analysis of cations, anions, nutrients, and metal content. Analyses were performed by the Natural Resources Analytical Laboratory, University of Alberta. Parameters assessed included but were not limited to sodium, calcium, magnesium, nitrate, phosphate and sulphate (Table S1).

Statistical Analysis

Univariate analyses were performed in Tibco Statistica version 14.0.1. All multivariate analyses were conducted in Primer 7, version 7.0.21, with PERMANOVA +1 (Primer-E Ltd; Anderson et al. 2008).

Species Richness

To understand and characterize the vegetation community and the environmental drivers of primary interest (age and salinity), species richness (the total number of species observed in plots at each wetland) was tabulated. Species richness is useful because it provides a simple measure of plant diversity within a wetland and of inter-wetland variability. A student t-test was conducted to determine if species richness differed significantly between the two disturbance classes - reclaimed and reference landscapes. Comparing community composition between disturbance classes can indicate if there is a difference between community composition in wetlands on a reclaimed landscape compared to a reference landscape. This comparison is important because the degree of similarity of plant communities between reclaimed and reference communities can be used as an indicator of reclamation success (Farnden, 2021).

Species associations among wetlands – NMDS ordination

Multivariate analysis using non-metric multidimensional scaling ordination (NMDS) analysis was performed to determine how wetlands ordinate based on the similarities in community composition among wetlands. Community composition (plant vegetative species abundance (percent cover)) from all plots, was averaged per wetland, relativized and square root transformed to downweight the contribution of dominant species (Anderson et al. 2008; McCune and Grace 2002). Wetlands were then compared using Bray-Curtis similarity (Bray and Curtis 1957) and ordinated using NMDS. A permutational multivariate analysis of variance (PERMANOVA; Anderson et al. 2008) was conducted to assess whether there were statistically significant multivariate differences in community composition between disturbance classes (reclaimed vs. reference landscapes), among salinity classes (freshwater to brackish; ESRD 2015), and among age classes (1 – 4 years to 21 – 40 years).

To determine if there is variation in community composition relative to salinity and wetland age classes (as defined in Section 2.1), a PERMANOVA was conducted. The PERMANOVA was used to assess the interaction between salinity class and disturbance class, and age class and disturbance class on community composition. A pairwise PERMANOVA, which compares each class to one another, was then run to assess the multivariate differences in community composition between salinity classes, and age classes.

Wetland Assemblage according to species composition – Cluster analysis:

Vegetation communities (assemblages) were identified using a group average cluster analysis (unweighted pair group method with arithmetic mean (UPGMA)) with a cut- off level of 32% similarity, determined by a similarity profile analysis (SIMPROF) on the Bray-Curtis similarities matrix used for the NMDS analysis. A similarities percentage analysis (SIMPER) identified the species contributing the most to the similarities between pairs of wetland sites (Anderson et al. 2008). The SIMPER analysis was used to identify the dominant species (highest percent cover) in each community (identified from the cluster analysis). The SIMPER analysis was performed to indicate whether wetlands of different disturbance classes (reclaimed or reference landscapes) fell within the same or distinct vegetation-based assemblages.

RESULTS

Newly formed wetlands (opportunistic and constructed) on reclaimed landscapes ranged from 0.04 ha (hectares) to 13.5 ha, with a mean (SE) size of 1.4 (0.7) ha. The NFW on reference landscapes ranged from 0.1 ha to 8.44 ha, with a mean (SE) size of 1.9 (0.5) ha. Wetlands on the reclaimed landscape were 2 to 28 years old, with a mean (SE) age of 9.2 (1.5), while wetlands on reference landscapes were slightly older with a mean age of 12.6 (2.0) with an age range of 3 – 40 years old (Table S1). Wetlands on a reclaimed landscape were also more saline, ranging from 615 µS/cm to 8134 µS/cm (slightly brackish to brackish) with a mean salinity of 2849 (497) µS/cm. Reference wetland salinities ranged from freshwater to slightly brackish, with salinity ranging from 277 µS/cm to 1690 µS/cm, with a mean salinity of 823 (84.80) µS/cm (Table S1).

Community Composition and Disturbance Classes

The mean (SE) species richness on the reference landscape was 20 (2.0) and on reclaimed landscapes was 13 (2). There was a significant difference in species richness between disturbance classes (reclaimed and reference landscapes). Significantly fewer species occurred in wetlands on reclaimed landscapes than in wetlands on reference landscapes (*t*=2.98, p<0.005, df = 38; Figure 2.2).

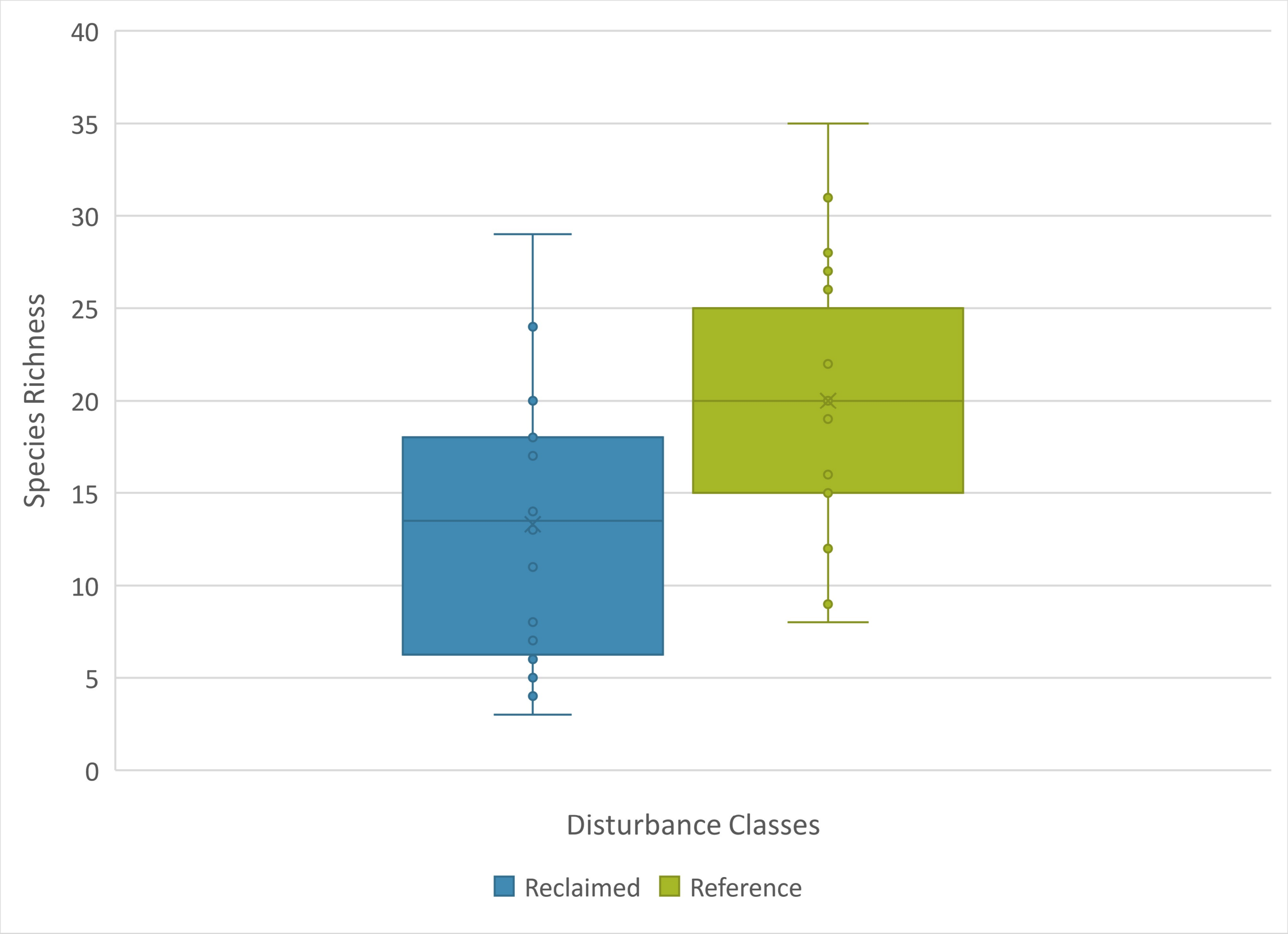

Figure 2.2 Species richness in newly formed wetlands on reference and reclaimed landscapes (n = 20 for both disturbance classes). The mean (SE) species richness in newly formed wetlands on the reference landscape was 20 (2.0) and the reclaimed landscape was 13 (2.0).

To determine how the 40 wetlands on reclaimed and reference landscapes ordinated based on their similarities in community composition (vegetative species abundance) from the Bray-Curtis similarities analysis, an NMDS analysis was performed (Figure 2.3). A solution for the NMDS ordination was reached in 50 iterations with a stress value of 0.22 in 2-dimensions, and 0.16 in 3-dimensions. Vegetative species abundance (percent cover) was significantly different between reference and reclaimed landscapes ($p<0.01$; Table 2.4). However, there are areas in ordination space where wetlands on reclaimed and reference landscapes, assemblage were interspersed, indicating wetlands whose vegetative species abundances are similar to one another (Figure 2.3).

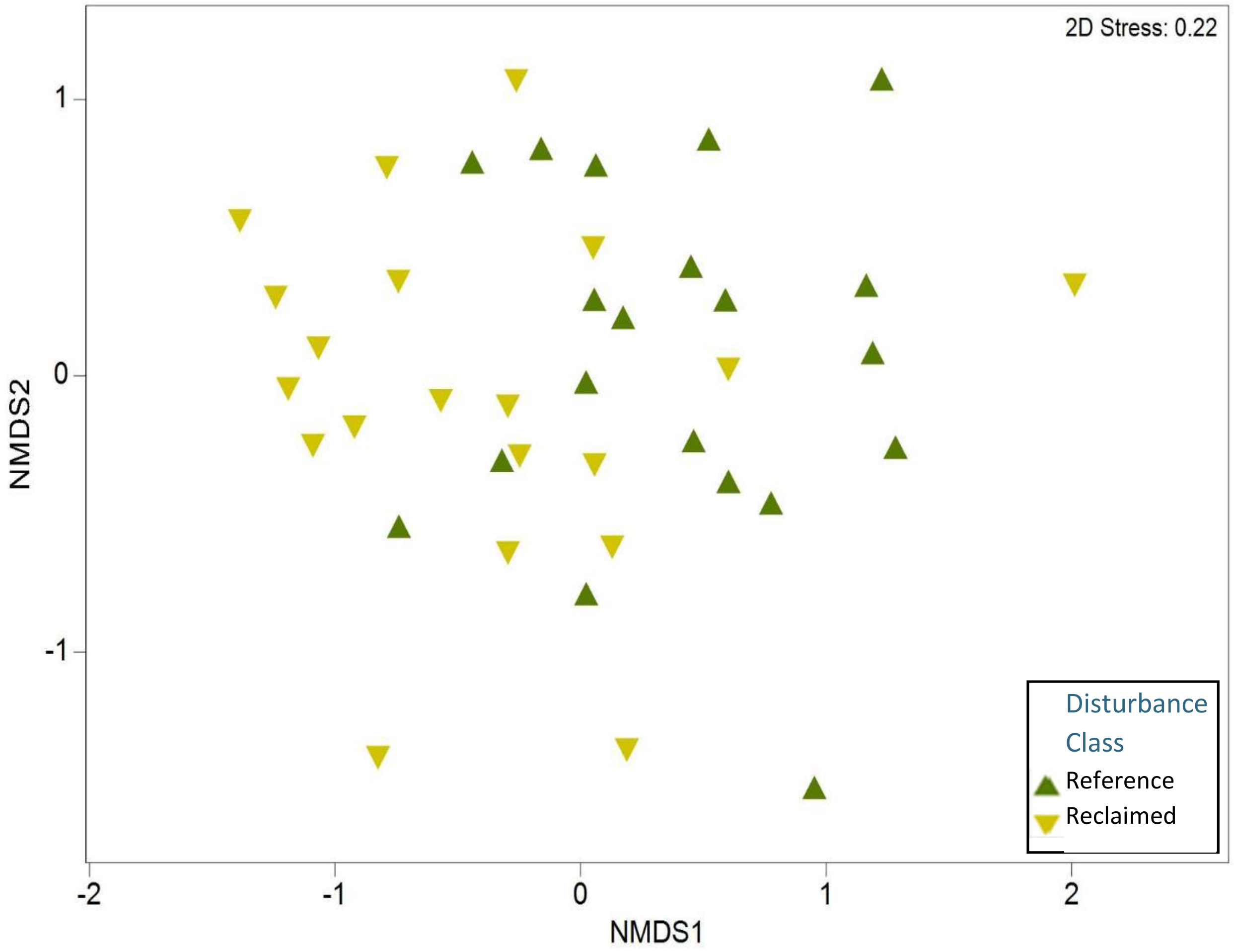

Figure 2.3 Non-metric multidimensional scaling (NMDS) ordination of mean relativized, and square root transformed vegetative species abundance (percent cover) data from wetlands on a reclaimed landscape (n = 20) and on a reference landscape (n = 20).

Environmental variables and associated vectors were overlaid on the NMDS scatterplot relative to the ordination axes to determine how environment variables interact, to potentially influence vegetative species abundance (Pearson's correlation score of >0.17; Figure 2.4). A Pearson's correlation score of >0.17 was selected so age (correlation of 0.17) was included in environmental variables displayed (Figure 2.4). The first and second ordination axes were plotted. Disturbance class most strongly correlated with NMDS axis 1, with wetlands on the reclaimed landscape displaying more negative scores, and wetlands on the reference landscape displaying more positive scores. No single variable was clearly strongly associated with NMDS axis 2 (Figure 2.4; Table 2.). However, specific conductance, magnesium, sodium, and sulphate correlated more strongly with wetlands on the reclaimed landscape and that have opportunistically formed.

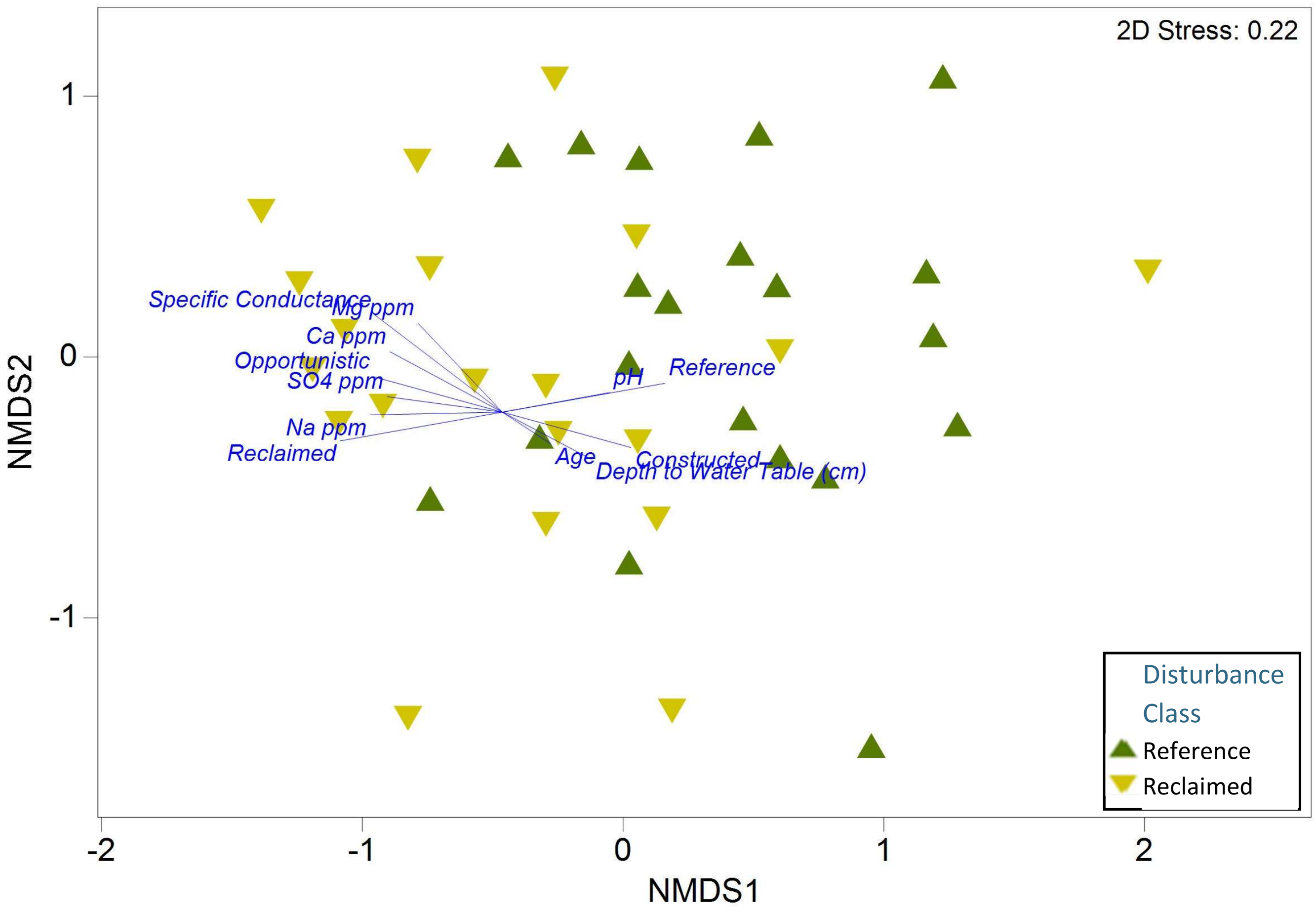

Figure 2.4 Non-metric multidimensional scaling (NMDS) ordination of relativized and square root transformed vegetative species abundance (percent cover) data from wetlands on a reclaimed landscape (n = 20; apex-upward triangles) and on a reference landscape (n = 20; apex-downward triangles). Variables with a Pearson's correlation coefficient score of >0.17 relative to ordination axes are shown with vectors.

Based on the Bray-Curtis similarities analysis, a group average cluster analysis (UPGMA) and a similarities percentage (SIMPER) analysis were performed to determine which wetlands clustered together and which species contributed the most to the assemblage. The cluster analysis identified eight assemblages of wetlands (Assemblages C, D, E, G, H, I, J, K). Three wetlands were so distinct that they were not assigned to any of assemblages (points A, B, and F; Figure 2.5; Figure S1; Table 2.3).

Assemblage C (*Calamagrostis-Rubus* assemblage) was dominated by wetland graminoid species, *Calamagrostis canadensis* and *Poa* species, and two upland species *Rubus idaeus* and *Lotus corniculatus* (Table 2.5). The wetlands comprising Assemblage C were a mixture of reference and reclaimed shallow open water opportunistic and constructed wetlands aged 13 – 15 years old with slightly brackish water (n = 2; Table S1).

Assemblage D (*Calamagrostis-C. atherodes* assemblage) was dominated by *Calamagrostis canadensis, Carex atherodes,* and *Persicaria amphibia* (Table 2.5). The wetlands comprising the *Calamagrostis-C. atherodes* assemblage were a mixture of reference and reclaimed shallow open water opportunistic wetlands aged 14 – 15 years old with slightly brackish water (n = 2; Table S1).

Assemblage E (*C. aquatilis-Typha* assemblage) was dominated by *Carex aquatilis* and *Typha latifolia* (Table 2.5). The wetlands comprising the *C. aquatilis-Typha* assemblage were a mixture of marsh and shallow open water wetlands aged 3 – 15 years old that were slightly brackish to brackish in nature on a reclaimed landscape (n = 8; Table S1).

Assemblage G (*C. utriculata-Trifolium* assemblage) was dominated by wetland species such as *Carex* species, *Typha latifolia*, *Acorus calamus*, and *Hordeum jubatum*, with an upland forb, *Trifolium hybridum* present (Table 2.5). The wetlands comprising the *C. utriculata-Trifolium* assemblage were shallow open water constructed wetlands aged 4 – 13 years old with freshwater to slightly brackish water on a reference landscape (n = 3; Table S1).

Assemblage H (*C. aquatilis-Typha-C. canascens* assemblage) was dominated by wetland species *Carex aquatilis*, *Typha latifolia*, *Carex canescens,* and *Triglochin maritima*, with an upland forb, *Atriplex prostrata* present (Table 2.5). The wetlands comprising the *C. aquatilis-Typha-C. canascens* assemblage were opportunistic marshes that were less than 4 years old, that were moderately brackish to brackish in nature on a reclaimed landscape (n = 3; Table S1).

Assemblage I (*Salix-C. aquatilis-Typha* assemblage) was dominated by *Salix* species, *Carex aquatilis*, *Typha latifolia*, and *Equisetum palustre* (Table 2.5). The wetlands comprising the *Salix-C. aquatilis-Typha* assemblage were shallow open water opportunistic and constructed wetlands on a reference landscape aged 4 – 13 years old, and freshwater to slightly brackish in nature (n = 3; Table S1).

Assemblage J (*Carex-Typha-Equisetum* assemblage) was dominated by wetland and upland species. Dominant wetland species are *Carex aquatilis, Typha latifolia, Equisetum arvense, Poa* species*, Carex utriculata,* and dominant upland species are *Sonchus arvensis* and *Agrostis scabra* (Table 2.5). The wetlands comprising the *Carex-Typha-Equisetum* assemblage were slightly brackish to moderately brackish opportunistic marshes, and shallow open water opportunistic and constructed wetlands on reclaimed and reference landscapes. The age of these wetlands was primarily between 10 to 14 years old (n = 9), with three wetlands that are 3, 6, and 8 years old (Table S1).

Assemblage K (*Typha-Carex* assemblage*)* was dominated by wetland and upland species. Dominant wetland species are *Typha latifolia, Carex atherodes, Carex utriculata, Carex aquatilis, Equisetum arvense, Equisetum fluviatile,* and the dominant upland species was *Agrostis scabra* (Table 2.5). The wetlands comprising the *Typha-Carex* assemblage were slightly brackish opportunistic marshes, and shallow open water opportunistic and constructed wetlands on reclaimed and reference landscape. The age of these wetlands was primarily between 5 to 9 years old (n = 3), with two older wetlands that are 18 and 28 years old (Table S1).

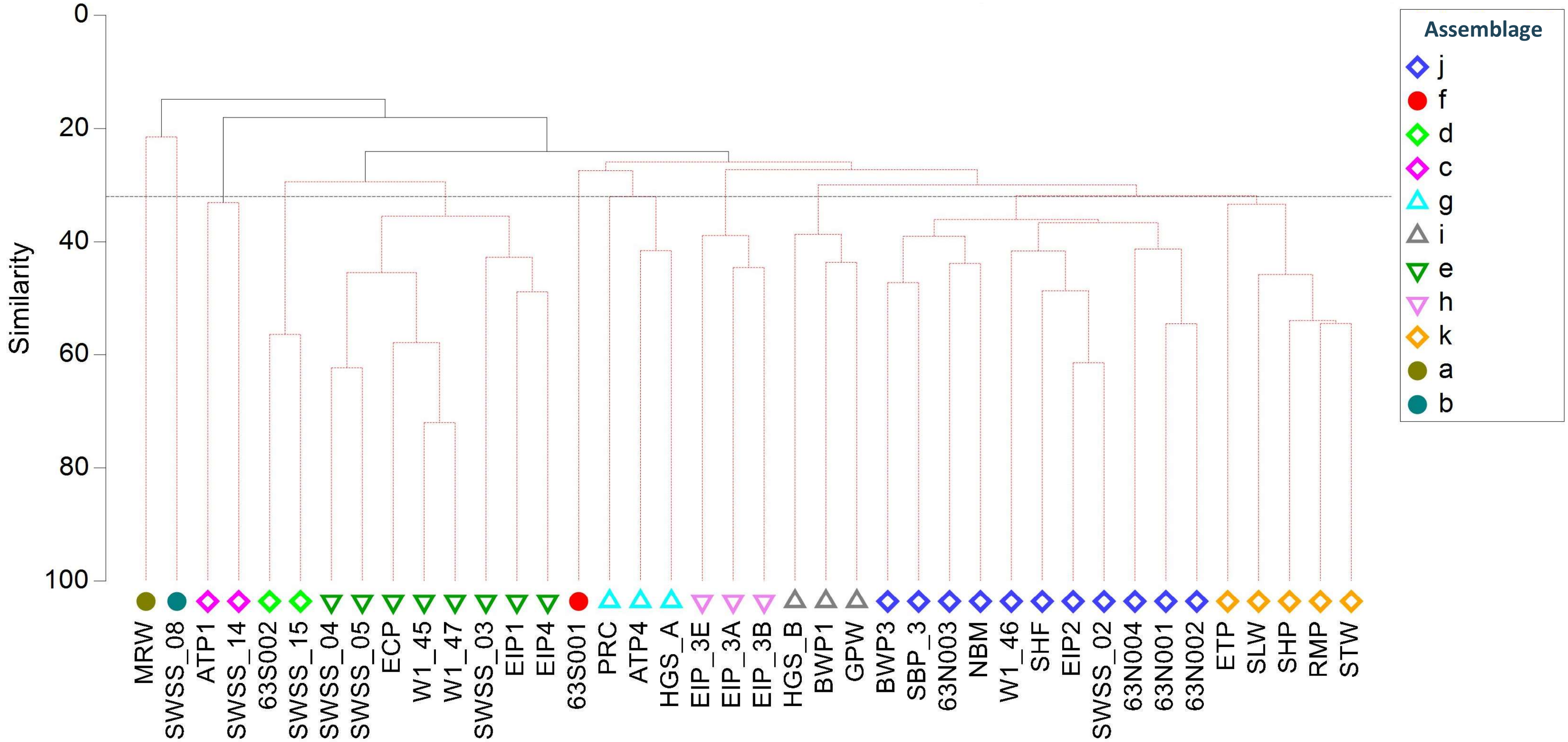

Figure 2.5 Group average cluster analysis of relativized and square root transformed vegetative species abundance (percent cover) data from wetlands on reclaimed (n = 20) and reference landscapes (n = 20). Colours represent assemblages that were determined by grouping average cluster analysis with a cut of level of 32% similarity. Eight assemblages were identified (Assemblages C, D, E, G, H, I, J, K) with three outliers (A, B, and F) that did not cluster with more than 1 other wetland. Symbols represent disturbance class: apex-down triangles represent assemblages whose wetlands occurred only on reclaimed landscapes, diamonds are assemblages containing wetlands occurring on both reclaimed and reference landscapes, apex upward triangles are Assemblages whose wetlands occurred only on reference landscapes, and circles represent wetlands that were outliers.

Community Composition in Relation to Salinity

Species richness slightly declined across a salinity gradient; however, species richness did not differ significantly among wetland salinity classes (p=0.13; Table 2.6; Figure 2.6). Freshwater wetlands on a reference landscape had a mean (SE) species richness of 23 (5). The mean species richness for slightly brackish wetlands on a reference landscape was 19 (1) and was 28% higher than slightly brackish wetlands on a reclaimed landscape, as the mean species richness on a reference landscape was 14 (3). The mean species richness for moderately brackish wetlands on a reclaimed landscape was 13 (2). Brackish wetlands on a reclaimed landscape had mean species richness of 12 (1).

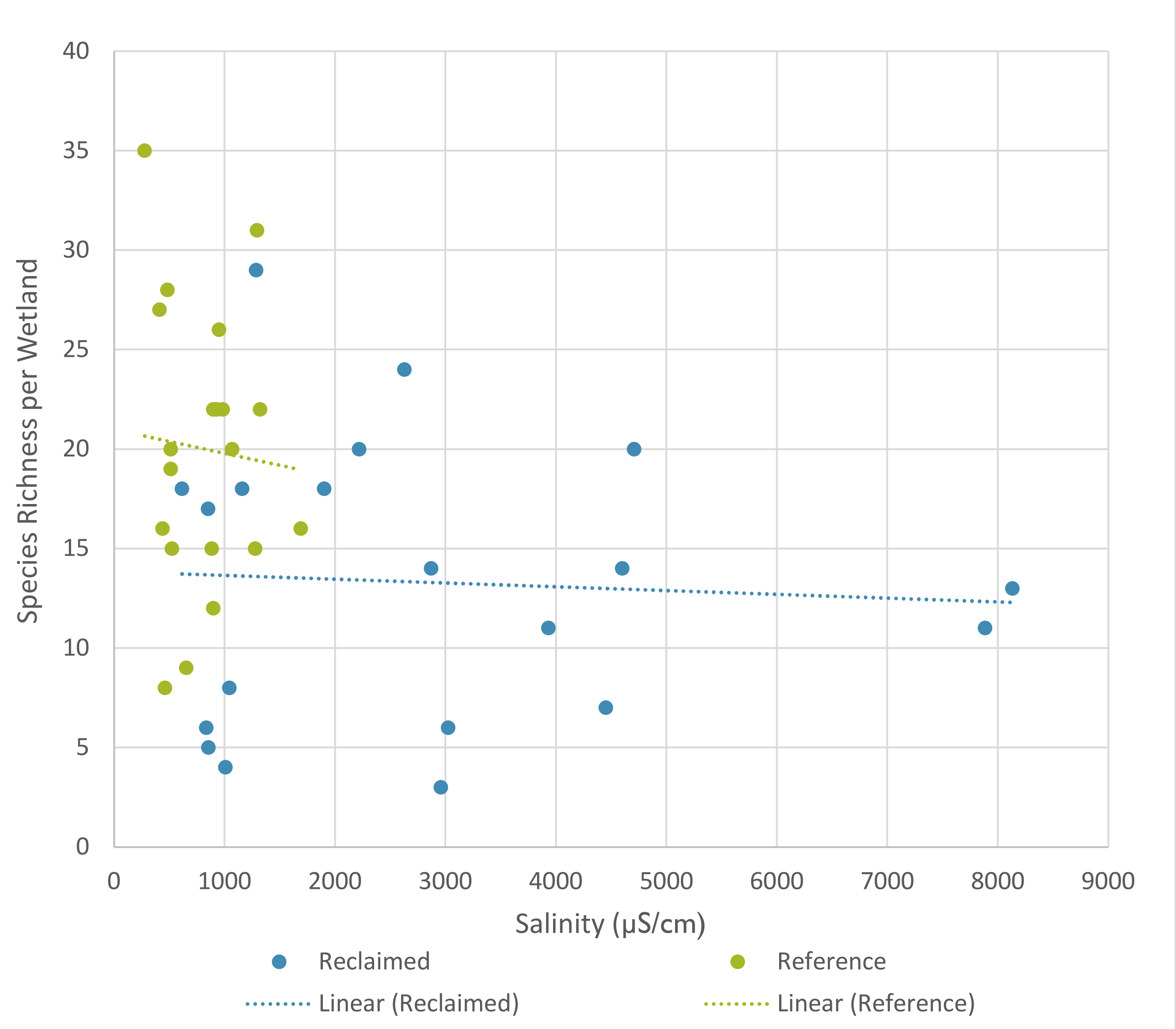

Figure 2.6 Species richness per newly formed wetland on a reference landscape (n = 20) and reclaimed landscape (n = 20) along a gradient of salinity.

PERMANOVA analysis revealed that plant community composition (mean vegetative species abundance) differed among salinity classes ($p<0.001$; Table 2.7). Mean vegetative species abundance (percent cover) increased across a gradient of salinity (from low salinity to high salinity) (Figure 2.7). Mean vegetative species abundance (percent cover) in freshwater wetlands was 26% lower compared to vegetative species abundance in slightly brackish wetlands ($p<0.05$; Table 2.7). Freshwater wetlands mean vegetative species abundance was 49% lower compared to moderately brackish wetlands ($p<0.01$; Table 2.7;). Mean vegetative species abundance between freshwater wetlands and brackish wetlands was similar ($p=0.20$; Table 2.7). Slightly brackish wetlands' vegetative species abundance was lower compared to moderately brackish wetlands by 31% ($p<0.01$; Table 2.7), but only 7% lower compared to brackish wetlands ($p<0.05$; Table 2.7). The vegetative species abundance in moderately brackish and brackish wetlands were similar ($p = 0.45$; Table 2.8). Wetlands across different salinity classes had a community composition that were on average 28% similar to one another (Table 2.10).

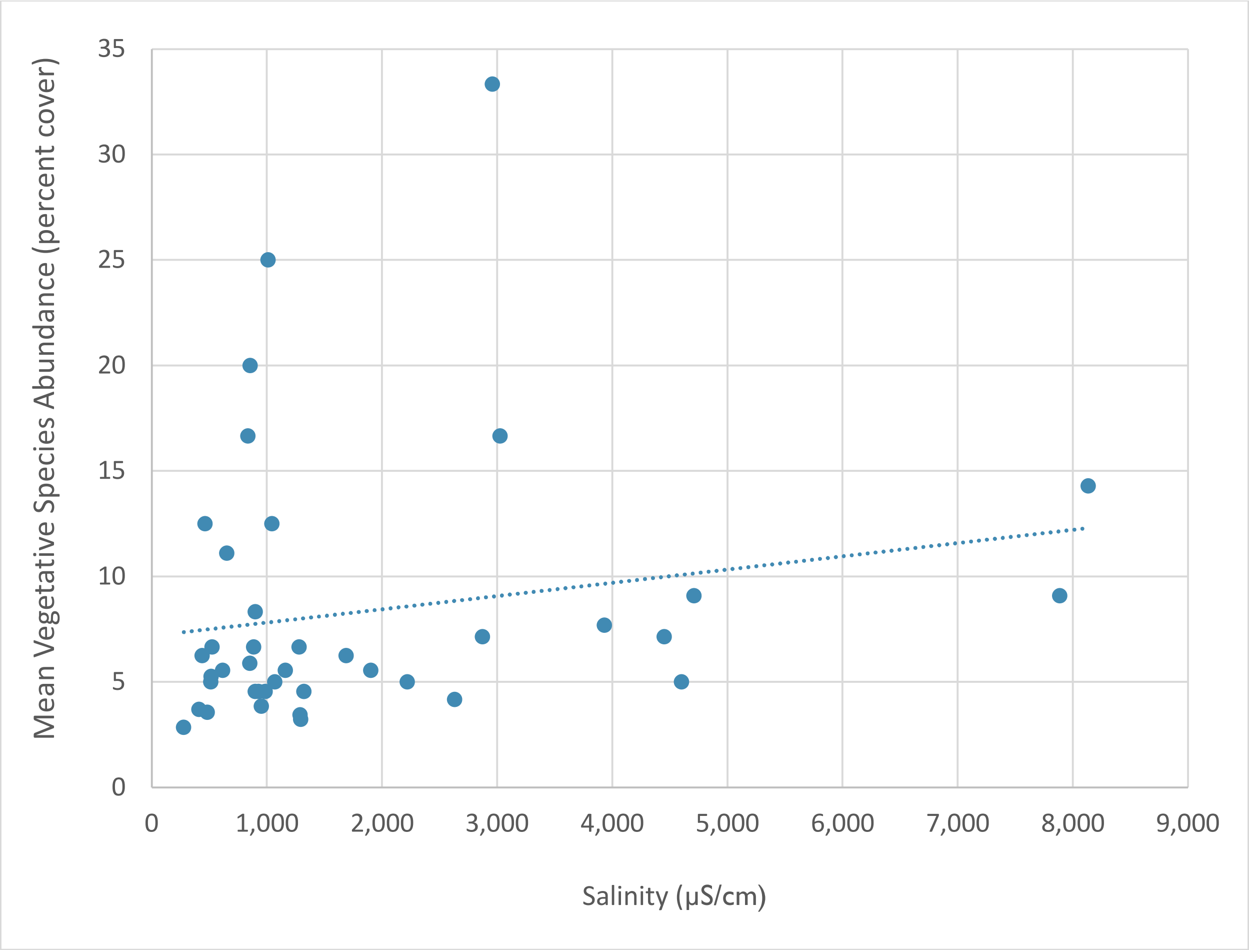

Figure 2.7 Relativized and standardized mean (SE) vegetative species abundance (percent cover) across a gradient of salinity in newly formed wetlands on reference and reclaimed landscapes (n = 40).

Overall, vegetative species abundance (percent cover) across the forty wetlands along a gradient of salinity was greater in wetlands on reclaimed landscapes than reference landscapes (Figure 2.8).

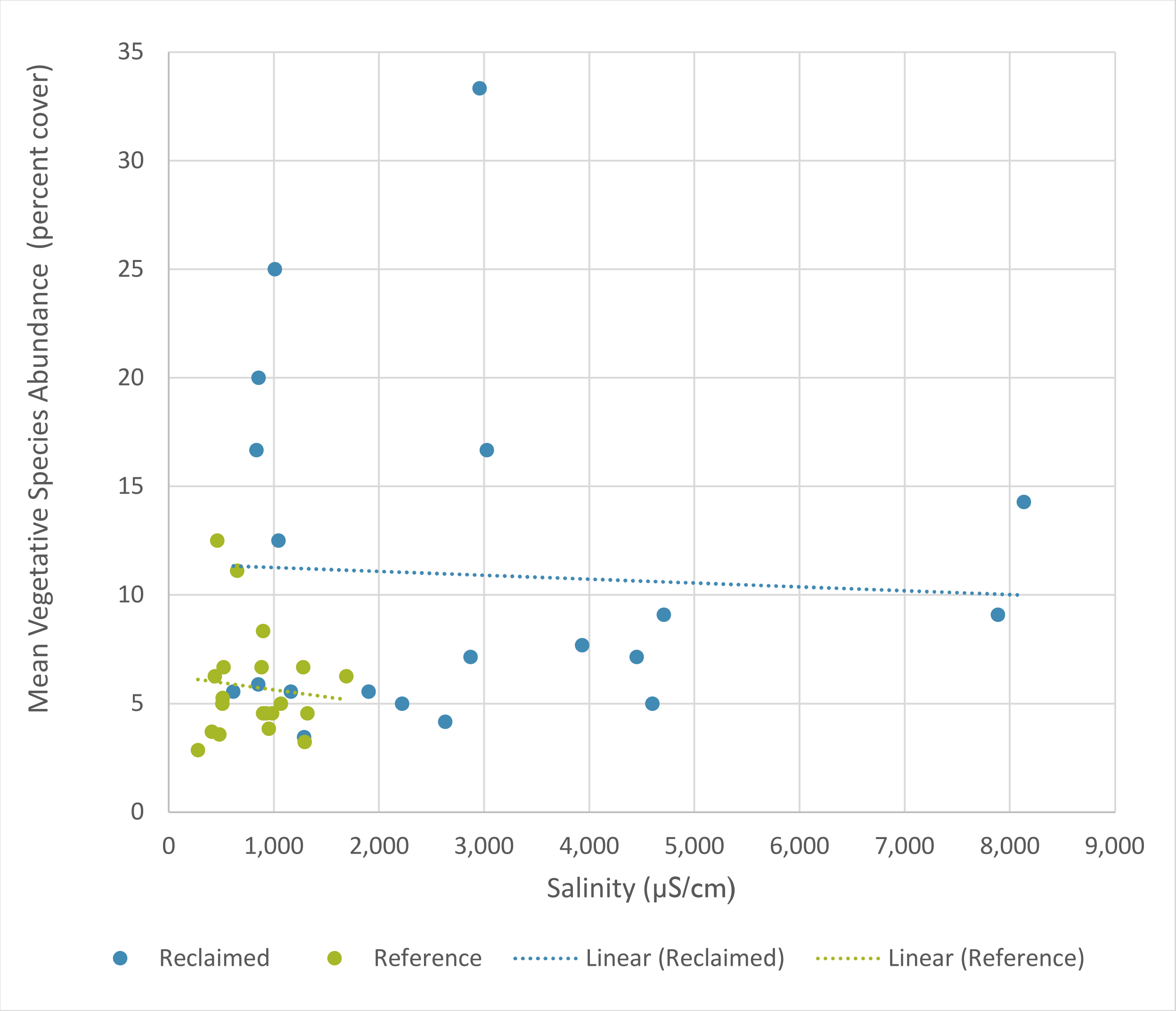

Figure 2.8 Relativized and standardized mean (SE) vegetative species abundance (percent cover) along a gradient of salinity in newly formed wetlands on reference and reclaimed landscapes (n = 20 for both reclaimed and reference landscapes).

Community Composition in Response to Age on Reference and Reclaimed Landscapes

Species richness did not differ among age classes (p=0.96; Table 2.10). However, wetlands of all ages situated on reference landscapes (except those 21 – 40 years old), had higher species richness than wetlands of equivalent age situated on a reclaimed landscape (Figure 2.9). Species richness of wetlands that were 1 – 4 years old on a reference landscape was 32% higher than wetlands on a reclaimed landscape (Table 2.10). Wetlands that were 5 – 9 years old on a reference landscape had 80% more species than wetlands of the same age on a reclaimed landscape (Table 2.10). Species richness of wetlands that were 10 – 14 years old on a reference landscape was 25% higher than wetlands on a reclaimed landscape (Table 2.10). Wetlands that were 15 – 20 years old on a reference landscape had 35% more species than wetlands of the same age on a reclaimed landscape (Table 2.10). Wetlands 21 – 40 years old on a reclaimed landscape and reference landscape had the same species richness (Table 2.10).

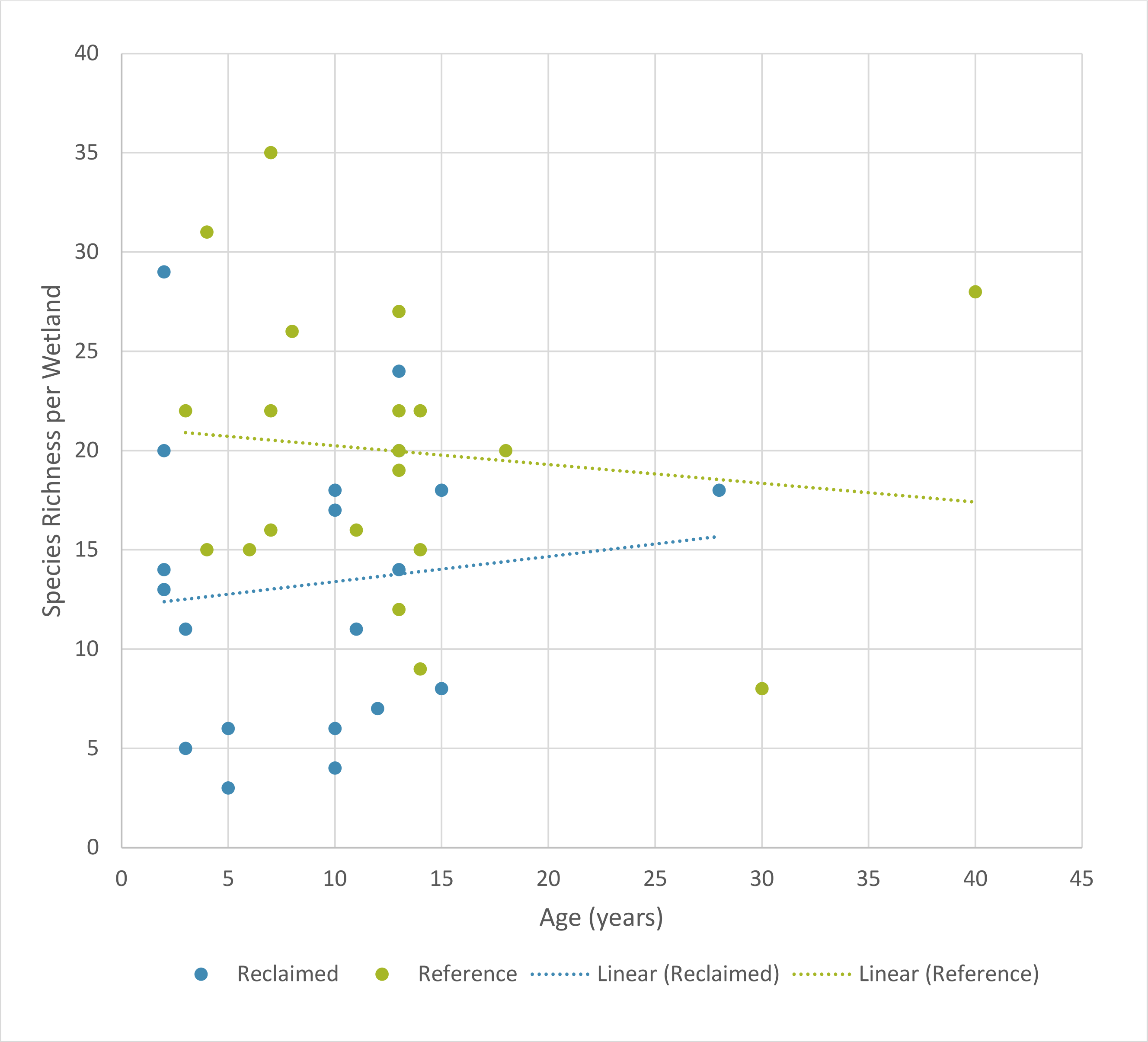

Figure 2.9 Species richness per newly formed wetland on a reference landscape (n = 20) and reclaimed landscape (n = 20) along an age gradient.

Plant community composition (mean vegetative species abundance) differed significantly among age classes and between disturbance classes (PERMANOVA, p<0.002 and p<0.001, respectively; Table 2.11). There was no statistically significant interaction between age class and disturbance class (Table 2.11). However, the number of wetlands contributing to the age classes differs substantially as there is only a 6% decrease in mean vegetative species abundance between wetlands 10-14 years old and 15 – 20 years old (n = 18, and n = 3, respectively; Table 2.11). Wetlands across different age classes had a community composition that were on average 25% similar to one another (Table 2.13).

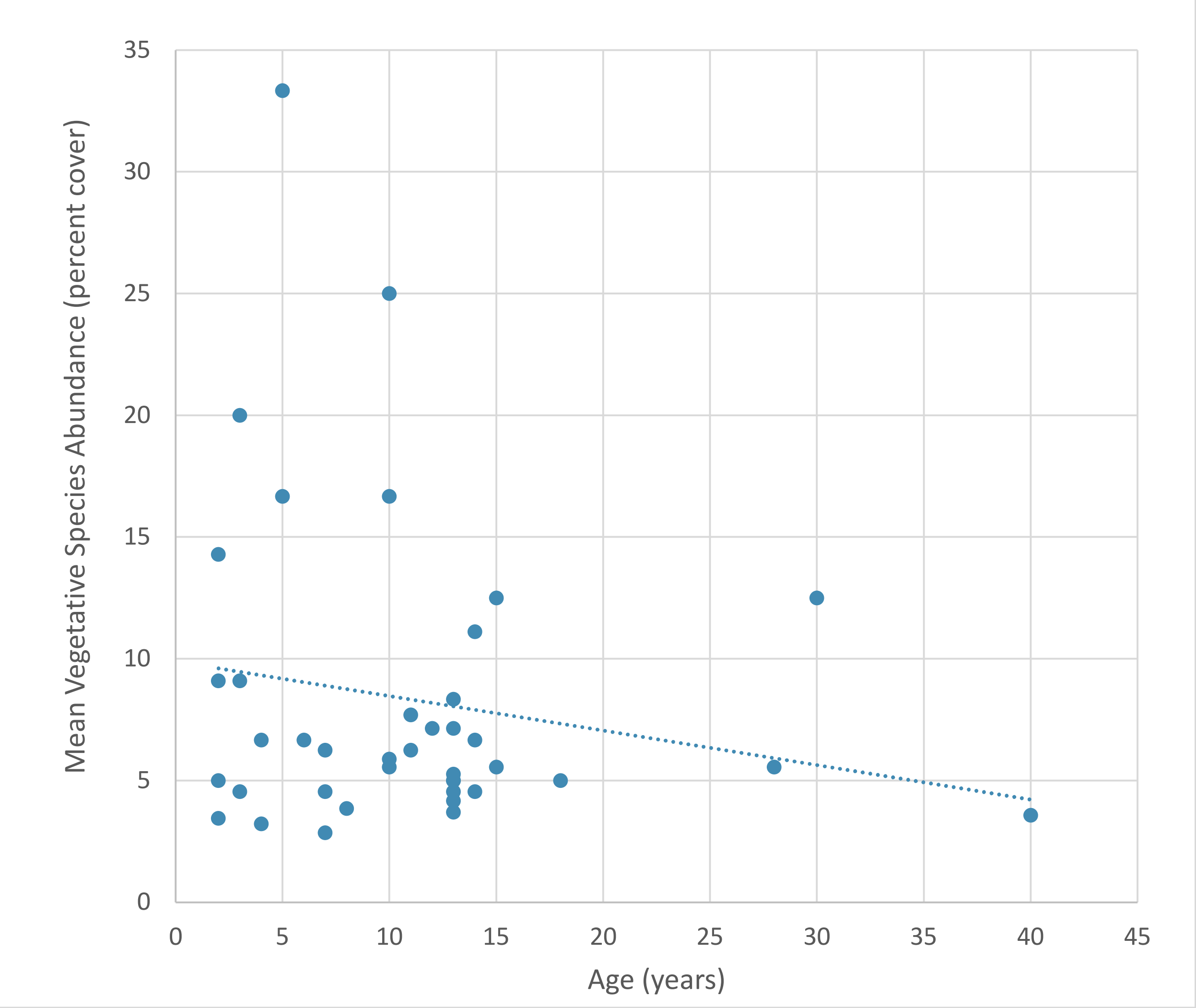

Figure 2.10 Relativized and standardized mean (SE) vegetative species abundance (percent cover) along an age gradient (n = 40).

Overall vegetative species abundance (percent cover) was higher on reclaimed landscapes than reference landscapes for every age class, except for wetlands that were 21 – 40 years old (Figure 2.11). Vegetative species abundance in wetlands between the ages of 1 – 4 was 45% lower in wetlands on a reference landscape compared to a reclaimed landscape. Wetlands that were 5 – 9 years old on reclaimed landscapes had 4 times the percent cover for vegetative species abundance than wetlands on a reference landscape. Vegetative species abundance in wetlands 10 – 14 years old was 40% higher in wetlands on a reference landscape compared to a reclaimed landscape. Wetlands that were 15 – 20 years old on reclaimed landscapes had 45% higher vegetative species abundance than wetlands on a reference landscape. Although wetlands that were 21 – 40 years old had 20% lower vegetative species abundance on a reclaimed landscape, the difference between the two disturbance classes were less than other age classes.

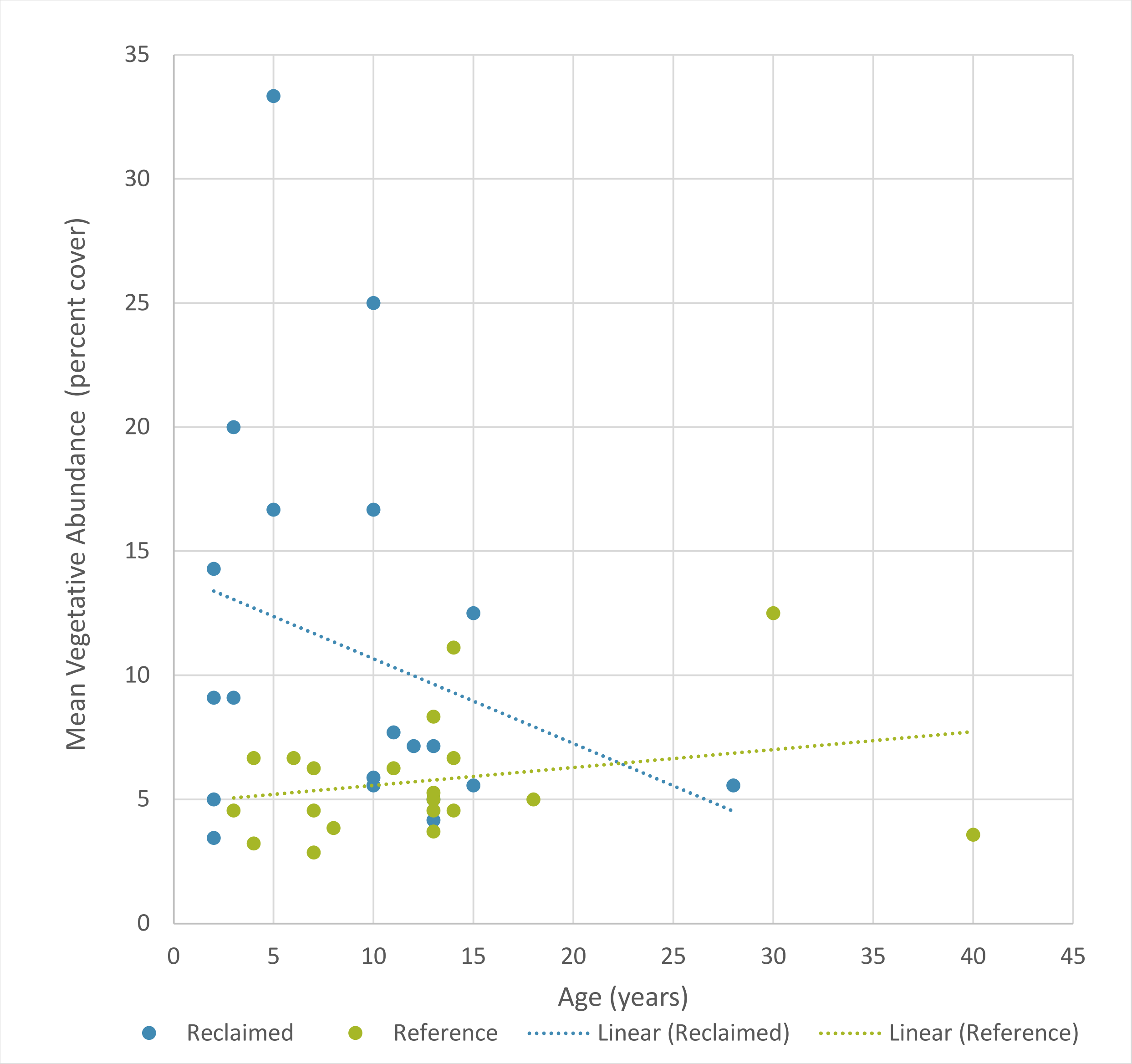

Figure 2.11 Relativized and standardized mean (SE) vegetative species abundance (percent cover) along an age gradient in reference and reclaimed wetlands (n = 20 for both reclaimed and reference).

DISCUSSION

Community Composition in Wetlands on Reference and Reclaimed landscapes

Eight vegetation assemblages were identified among 40 opportunistic and constructed marsh and shallow open water wetlands. These assemblages reflected differences in community composition among the wetlands studied and grouped wetlands that were most similar in community composition (based on percent cover). Four of the assemblages contained wetlands from reference and reclaimed landscapes (Assemblages C, D, J and K). However, two assemblages were exclusively comprised of wetlands on reclaimed landscapes (Assemblages E and H), and the remaining two assemblages were made up of wetlands on reference landscapes (Assemblages G and I). The disturbance class (reclaimed or reference) largely accounted for these differences. Age and salinity had less of a role in accounting for differences in community composition (Figure 2.4). This indicates that there are likely other environmental variables not investigated in this study that are more strongly correlated with the differences in community composition than the two main variables of salinity and age. Other candidate variables to investigate are soil placement material, wetland origin (constructed or natural; Roy et al., 2016), hydrologic regime (Cooper et al., 2006), and depth to water table (Borkenhagen et al., 2023; Cooper et al., 2006).

The finding that plant communities on reference and reclaimed landscapes of equivalent age are similar to one another is a promising sign that perhaps wetlands forming on reclaimed landscapes are developing in a fashion similar to newly formed wetlands on reference landscapes; and both are possibly returning to equivalent land capabilities. There were 138 plant species observed in the emergent and wet meadow zones of wetlands on reclaimed and reference landscapes, although significantly fewer species were present in wetlands on reclaimed landscapes than in wetlands on reference landscapes. Species richness and community composition (vegetative species abundance) were significantly different across a gradient of salinity and among wetlands of different ages. Although there has been limited literature studying the effects of salinity and age on community composition in opportunistic and constructed wetlands, similar results were observed in several studies (Raab and Bayley, 2013; Rooney and Bayley, 2011; Roy et al., 2016; Trites and Bayley, 2009). The focus of reclamation practices should be on community composition so there is continuity in species across the landscapes, which was observed prior to disturbance.

Community Composition in Relation to Salinity on Reference and Reclaimed Landscapes

Vegetation community composition varies along a gradient of salinity (House et al., 2022; Purdy et al., 2005; Rooney and Bayley, 2011). I found that species richness was negatively associated with increasing salinity levels from freshwater to brackish conditions. The observed relationship could reflect structural and functional differences among plant species and their assemblages in relation to salinity levels. For example, species such as *Carex aquatilis, Typha latifolia,* and *Carex atherodes,* are common species comprising the wetland assemblages observed in this study. These species can tolerate a wide range of salinity (Glaeser et al., 2021; Mollard et al., 2012; Mollard et al., 2013; Vitt et al., 2020), which could contribute to species differences between the assemblages. The differences in community composition observed between wetlands on a reclaimed landscape compared to those on a reference landscape can be attributed to the gradient of salinity observed as freshwater wetlands were only observed on reference landscapes and more saline wetlands were only observed on reclaimed landscapes.

Although species richness was reduced in the more saline wetlands, vegetative species abundance (percent cover) was lowest in the freshwater wetlands, higher in moderately brackish wetlands and somewhat reduced from the maximum in brackish wetlands. The increasing vegetative species abundance observed across the gradient of salinity (from freshwater to moderately brackish) may reflect the increasing dominance of salt tolerant species at the intermediate salinity classes until moderately brackish conditions are exceeded, at which point vegetative species abundance is reduced. Lower vegetative species abundance could be attributed to a reduction in energy available for growth by non-halophytes as they expend energy to maintain osmotic balance by excluding salts from the tissues in the plant (Vitt et al., 2020). Mechanisms of salt exclusion from plant tissues include osmotic tolerance (the change of salinity levels within the cells and around the roots), ability to exclude Na+ (Na+ is excluded from the leaves by the roots) and tissue tolerance (tolerance of the plant tissue to the accumulation of Na+; Munns and Tester, 2008).

Community Composition in Relation to Age on Reference and Reclaimed Landscapes

Species richness did not differ among wetland age classes. However, vegetation community composition was significantly different among age classes. Species richness tended to be progressively lower in wetlands on reference landscapes of increasing age, whereas species richness of wetlands on reclaimed landscapes were greater in older wetlands than their younger counterparts. Species richness in wetlands on reference landscapes was higher across all age classes compared to wetlands on reclaimed landscapes. Differences in species richness between disturbance classes could be attributed to seed bank availability, abundance of propagules in the soil and their associated viability as these are factors that negatively impact species richness in wetlands on reclaimed landscapes (Cooper 2004; Dhar et al., 2016). There is limited knowledge around the effects of wetland age on species richness and community composition. However, Rooney and Bayley (2011) did not notice a difference in wetland age and species richness or community composition (vegetative species abundance) and attributed differences in species richness and community composition between wetlands on reclaimed and reference landscapes to time for colonization to occur.

Limitations and Future Research

Of the 40 wetlands investigated in this study, few (n = 3) wetlands were older than 21 years, more than half of the wetlands were between the age of 10 and 18 (n = 21), and the remainder were younger than 9 years old (n = 16). Additionally, the only wetlands in the freshwater salinity class (n = 5) were situated reference landscapes, and wetlands in the moderately brackish (n = 9) and brackish (n = 2) classes were limited to the reclamation landscapes. Slightly brackish wetlands (n = 24) were observed both on the reference and reclaimed landscapes. The inability to locate wetlands representing all classes of the disturbance x salinity class study design meant I could not test for a statistical interaction between possible effects of salinity and disturbance class. Additional older wetlands on reclaimed and reference landscapes, and wetlands that are more saline in nature on reference landscapes and freshwater wetlands on reclaimed landscapes should be studied in future monitoring years of the Boreal Wetland Reclamation Assessment Program.

Although it was not a planned part of this study, a PERMANOVA was performed to examine the effects of type of soil placement material (natural (reference wetlands), overburden, saline sodic overburden, tailings sand, and saline sodic consolidate tailings) on vegetative species abundance. The type of soil placement material used significantly influenced community composition (Table S3). Future studies should investigate how the differences in soil placement materials affect community composition. Future work should entail investigating how wetland origin (constructed vs opportunistic), soil placement material, and soil quality, and depth to water table influence community composition (species richness and vegetative species abundance).

CONCLUSIONS

Community composition differed between newly formed wetlands on reclaimed and reference landscapes, and among classes representing a gradient of salinity, and among wetlands of different ages. Disturbance class (reclaimed or reference) appears to account for the differences in communities seen between landscapes in these newly formed wetlands. However, differences in the communities could be more strongly ascribed to differences in wetland salinity and age. Other environmental variables such as soil placement material, depth to water table, and wetland origin (constructed or opportunistic) may also account for the differences observed in community composition. Despite differences in salinity observed between landscape types, some wetlands on reclaimed landscapes displayed a similar community composition to those observed on reference landscapes. Such evidence is consistent with an expectation that communities should exhibit parallel successional trends of species accrual and replacement in reference and reclaimed landscapes, ideally leading to equivalent landscape capabilities.

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TABLES

Table 2.1 Plant species identified in opportunistic (OW) and constructed (CW) wetlands on a reclaimed (RC) and reference (RF) landscapes. Certain species were identified to genus-level only.

Species	OW	CW	RC	RF
Achillea millefolium	x	x	x	x
Achillea sibirica	x			x
Acorus calamus	x	x	x	x
Agropyron trachycalum	x			x
Agropyron unilaterale	x			x
Agrostis scabra	x	x	x	x
Alisma plantago-aquatica	x	x		x
Alisma triviale		x		x
Alopecurus aequalis		x	x	
Aster borealis		x		x
Aster ciliolatus		x	x	
Aster conspicuus		x	x	
Aster puniceus	x	x	x	x
Astragalus canadensis	x		x	x
Astragalus species	x	x	x	x
Atriplex prostrata		x	x	
Beckmannia syzigachne	x	x	x	x
Bidens cernua	x	x	x	x
Bromus ciliatus		x	x	
Bromus inermis		x	x	
Calamagrostis canadensis	x	x	x	x
Calamagrostis inexpansa		x	x	
Calamagrostis strictata		x	x	
Calla palustris		x		x
Caltha natans		x	x	
Carex aquatilis	x	x	x	x
Carex atherodes	x	x	x	x
Carex aurea	x			x
Carex bebbii	x	x	x	x
Carex canescens	x	x	x	x
Carex chordorrhiza		x	x	x
Carex disperma	x			x
Carex limosa		x	x	
Carex media	x			x
Carex pseudocyperus	x	x		x
Carex raymondii	x			x
Carex retrosa	x	x		x
Carex species	x	x	x	x

Species	OW	CW	RC	RF
carex stipata		x		x
Carex utriculata	x	x	x	x
Cerastium nutans		x		x
Cicuta maculata	x	x		x
Cicuta virosa	x	x	x	x
Cirsium arvense		x	x	
Comarum palustre		x		x
Deschampsia caepitosa	x	x	x	x
Eleocharis palustris	x	x	x	x
Elymus canadensis	x			x
Elymus repens		x	x	
Epilobium angustifolium	x	x	x	x
Epilobium glandulosum	x			x
Equisetum arvense	x	x	x	x
Equisetum fluviatile	x	x	x	x
Equisetum hyemale		x		x
Equisetum palustre	x	x	x	x
Equisetum pratense	x	x	x	x
Equisetum scirpoides	x	x	x	x
Equisetum sylvaticum	x	x	x	x
Fragaria vesca	x	x	x	
Fragaria virginiana	x	x	x	x
Galeopsis bifida		x	x	
Galium boreale	x		x	
Galium trifidum	x	x	x	x
Glyceria grandis	x	x	x	x
Habenaria hyperborea	x	x	x	x
Hieracium umbellatum		x	x	
Hordeum jubatum	x	x	x	x
Juncaceae	x			x
Juncus alpinoarticulatus	x			x
Juncus bufonius	x			x
Juncus confusus	x			x
Juncus nodous	x	x		x
Juncus tenuis	x			x
Juncus vaseyi		x	x	
Lamiaceae	x			x
Lathyrus venosus	x			x
Ledum palustre	x	x		x
Lotus corniculatus	x	x	x	x
Luzula multiflora	x			x
Lycopus uniflorus	x			x
Melilotus alba	x	x	x	x

Species	OW	CW	RC	RF
Mentha arvensis	x	x	x	x
Parnassia palustris	x	x	x	x
Persicaria amphibia	x	x	x	x
Petasites frigidus	x			x
Petasites palmatus	x	x	x	x
Phalaris arundinacae	x	x	x	x
Phleum pratense	x	x		x
Picea glauca	x	x		x
Plantago major	x			x
Poa palustris	x	x	x	x
Poa pratensis	x	x	x	x
Poa species	x	x	x	x
Polygonum lapathifolium	x	x	x	x
Populus balsamifera	x	x	x	x
Populus tremuloides	x	x		x
Potentilla gracilis		x	x	
Ranunculus sceleratus	x	x	x	x
Rhianthus minor	x			x
Rhinanthus borealis	x			x
Ribes hudsonianum	x			x
Rosa acicularis	x	x		x
Rubus arcticus	x			x
Rubus chamaemorus		x	x	
Rubus ideus	x	x	x	x
Rubus pubescens	x	x	x	x
Rumex martitmus		x	x	x
Rumex occidentalis		x	x	
Rumex salicifolius	x			x
Salix bebbiana	x	x	x	x
Salix brachycarpa	x			x
Salix exigua	x	x	x	x
Salix glauca	x	x	x	x
Salix pedicellaris		x	x	
Salix petiolaris	x	x		x
Salix planifolia	x	x	x	x
Salix species	x	x	x	x
Schizachne pupurscens	x			x
Scirpus cyperinus	x	x	x	x
Scirpus maritimus	x	x	x	x
Scirpus microcarpus	x	x	x	x
Scirpus validus	x	x	x	x
Scripus cyperinus	x			x
Scutellaria galericolata		x	x	

Species	**OW**	**CW**	**RC**	**RF**
Solidago canadensis	x	x	x	x
Sonchus arvensis	x	x	x	x
Stachys palustris	x			x
Stellaria longifolia	x	x	x	
Tanacetum vulgare	x	x		x
Taraxacum officinale	x			x
Trifolium hybridum	x	x	x	x
Trifolium pratense	x		x	
Trifolium species		x		x
Triglochin maritima	x	x	x	x
Triglochin palustris	x	x	x	x
Typha latifolia	x	x	x	x
Urtica dioica		x	x	
Vicia americana	x	x	x	x

Table 2.2 Summary table for a student t-test comparing the mean species richness among wetlands on a reference and reclaimed landscapes.

	Mean	SD	n	t-Value	DF	p-value	F-ratio	p Variances
Reference	20	7.06	20	2.976	38	**0.005**	1.032	0.946
Reclaimed	13	7.18	20					

Table 2.3 Wetland sites and associated vegetation assemblage determined by the group average Assemblage analysis and associated non-metric multidimensional scaling analysis (NMDS) scores.

Wetland Name	Assemblage	NMDS1	NMDS2	NMDS3
Muskeg River Wetland	a	0.900	0.741	0.970
SWSS 08	b	1.794	0.438	0.242
AOSTRA Pit 1	c	0.838	-1.379	-0.434
SWSS 14	c	0.254	-1.232	0.299
63 S 002	d	-0.65	-0.423	0.462
SWSS 15	d	-0.673	-1.253	0.499
Eric’s Pond	e	-0.842	0.044	0.338
EIP 1	e	-1.05	-0.102	-0.237
EIP 4	e	-0.767	0.200	-0.325
SWSS 03	e	-0.555	-0.264	-0.363
SWSS 04	e	-0.989	0.160	0.537
SWSS 05	e	-1.280	0.618	-0.052
W1 45	e	-1.035	0.525	0.334
W1 47	e	-1.066	-0.081	0.118
63 S 001	f	1.039	0.541	-0.127
AOSTA Pit 4	g	0.741	-0.366	-0.419
Hangingstone A	g	1.121	0.214	-0.43
Parsons Creek	g	1.111	-0.073	0.549
EIP 3 A	h	-0.260	-0.117	-1.072
EIP 3 B	h	-0.745	0.474	-0.612
EIP 3 E	h	-0.023	-0.134	-0.694
Borrow Pit 1	i	-0.127	0.843	-0.004
Gravel Pit	i	-0.339	0.856	-0.117
Hangingstone B	i	0.068	0.397	0.529
63 N 001	j	0.131	0.254	-0.377
63 N 002	j	0.476	0.008	-0.648
63 N 003	j	0.669	-0.360	0.080
63 N 004	j	0.002	0.449	-0.804
Borrow Pit 3	j	0.500	-0.296	-0.019
EIP 2	j	-0.237	-0.172	-0.145
North Base Mine	j	0.547	0.187	-0.455
Small Borrow Wetland 3	j	0.457	0.438	-0.006

Sandhill Fen	j	-0.181	-0.402	0.077
SWSS 02	j	0.051	-0.416	-0.28
W1 46	j	-0.228	-0.717	-0.003
East Pond	k	0.344	0.850	0.339
Ramp Wetland	k	-0.222	-0.114	0.419
South Pond	k	0.025	0.128	0.329
Shallow wetland	k	0.095	-0.090	0.790
Southern Wetland	k	0.110	-0.371	0.712

Table 2.4 Multivariate permutation analysis of variance (PERMANOVA) of community composition (vegetative species abundance) in response to disturbance class (reclaimed or reference landscape). Significant differences are bolded.

Source	df	SS	MS	Pseudo-F	p-value	*P(MC)*
Disturbance Class	1	8974.7	8974.7	3.4259	**0.001**	**0.002**
Residuals	38	99546	2619.6			
Total	39	108521				

Table 2.5 Results from Similarity Percentage (SIMPER) analysis with the average within group similarity and the species that contributed in the highest abundance to the similarity among wetlands comprising the assemblage.

Assemblage	Mean Group Similarity (%)	Species	Functional Group	Typical habitat	Mean Abundance Cover (SD)	Mean Similarity	Contribution (%)
A	-	-	-	-	-	-	-
B	-	-	-	-	-	-	-
C	33.1	*Calamagrostis canadensis*	Graminoid	Marsh - Swamp	8.2	8.1	24.6
		Rubus idaeus	Low shrub	Upland	6.8	6.3	19.0
		Poaceae	Graminoid	Marsh - Upland	7.2	5.3	16.1
		Lotus corniculatus	Forb	Upland	7.6	3.8	11.6
D	56.4	*Calamagrostis canadensis*	Graminoid	Marsh - Swamp	23.3	18.9	33.4
		Carex atherodes	Graminoid	Marsh - Swamp	19.8	18.2	32.3
		Persicaria amphibia	Forb	Marsh - Swamp	12.8	11	19.5
E	42.5	*Carex aquatilis*	Forb	Marsh - Swamp	24.9 (4.2)	20.1	47.1
		Typha latifolia	Forb	Marsh	17.1 (2.5)	12.6	29.6
F	-	-	-	-	-	-	-
G	35.2	*Carex* species	Graminoid	Marsh - Swamp	8.6 (36.5)	6.46	18.4
		Trifolium hybridum	Forb	Forb	4.7 (17.0)	4.5	12.8
		Typha latifolia	Graminoid	Marsh	4.7 (15.5)	4.14	11.8
		Carex utriculata	Graminoid	Marsh - Swamp	5.5 (1.4)	3.61	10.3

Assemblage	Mean Group Similarity (%)	Species	Functional Group	Typical habitat	Mean Abundance Cover (SD)	Mean Similarity	Contribution (%)
		Acorus calamus	Graminoid	Marsh	3.9 (5.0)	3.38	9.6
		Hordeum jubatum	Graminoid	Marsh	4.7 (4.6)	3.0	8.6
H	40.8	*Carex aquatilis*	Graminoid	Marsh - Swamp	14.3 (6.9)	11.1	27.6
		Typha latifolia	Graminoid	Marsh	8.1 (12.3)	6.4	15.7
		Carex canescens	Graminoid	Marsh - Swamp	8.3 (2.4)	6.1	14.8
		Triglochin maritima	Forb	Marsh	3.6 (13.8)	3.4	8.4
		Atriplex prostrata	Forb	Upland	5.3 (0.6)	2.3	5.5
I	40.3	*Salix* species	Shrub	Swamp - Fen	10.3 (46.0)	9.9	24.5
		Carex aquatilis	Graminoid	Marsh - Swamp	10.8 (7.7)	8.6	21.4
		Typha latifolia	Graminoid	Marsh	8.6 (9.4)	7.8	19.2
		Equisetum palustre	Forb	Swamp - Fen	5.5 (19.2)	5.1	12.7
J	38.5	*Carex aquatilis*	Graminoid	Marsh - Swamp	10.7 (2.8)	8.4	21.7
		Typha latifolia	Graminoid	Marsh	7.9 (2.7)	5.8	15.0
		Equisetum arvense	Forb	Marsh - Upland	6.9 (3.1)	5.7	13.9
		Poa species	Graminoid	Marsh - Upland	5.4 (0.8)	2.8	7.2
		Sonchus arvensis	Forb	Upland	4.0 (1.3)	2.5	6.5
		Carex utriculata	Graminoid	Marsh - Swamp	3.7 (0.9)	2.1	5.2
		Agrostis scabra	Graminoid	Upland	3.3 (1.2)	1.9	5.0
K	43.3	*Typha latifolia*	Graminoid	Marsh	9.2 (6.6)	8.1	18.5
		Carex atherodes	Graminoid	Marsh - Swamp	9.1 (1.1)	5.8	13.5
		Carex utriculata	Graminoid	Marsh - Swamp	5.9 (1.1)	3.9	8.9
		Carex aquatilis	Graminoid	Marsh - Swamp	6.1 (1.0)	3.3	7.7
		Equisetum arvense	Forb	Marsh - Upland	6.0 (1.4)	3.3	7.6
		Equisetum fluviatile	Forb	Marsh - Swamp	5.1 (1.2)	3.3	7.5
		Agrostis scabra	Graminoid	Upland	3.5 (8.4)	2.9	6.7

Table 2.6 Summary table for a one-way Analysis of Variance (ANOVA) test comparing the mean species richness among salinity classes.

	SS	DF	MS	F-Value	p-Value
Salinity Class	341.786	3	113.929	2.017	0.129
Error	2033.314	36	56.481		

Table 2.7 Multivariate permutation analysis of variance (PERMANOVA) of community composition (based on species percent cover) in response to salinity and disturbance class. Salinity was treated as an ordinal variable (salinity) in one analysis and as a fixed categorical variable (salinity class) in a second analysis. Salinity classes are as defined by the Alberta Wetland Classification System (ESRD 2015). Disturbance class is a binary variable indicating whether the wetland was on a reference (0) or reclaimed landscape (1). Significant differences are bolded.

Source Effect	**df**	**SS**	**MS**	**Pseudo-F**	**p-value**	***P(MC)****
Salinity	38	1.07E+05	2810.6	1.6362	0.087	0.153
Residuals	1	1717.8	1717.8			
Total	39	10852				
Salinity Class	3	14521	4840.3	1.8537	**0.001**	**0.002**
Residuals	36	94000	2611.1			
Total	39	10852				
Salinity Class x Disturbance Class						
Disturbance Class	1	5655.8	5655.8	2.2407	**0.002**	**0.017**
Salinity Class	3	11202	3734	1.4793	**0.014**	**0.035**
Interaction	0	0		No test		
Residuals	35	88344	2524.1			
Total	39	10852				

* MC is an additional Monte Carlo test ran concurrently with the PERMANOVA

Table 2.8 Pairwise multivariate permutation analysis of variance (PERMANOVA) results to test for significant differences in community composition among salinity classes (freshwater (< 500 µS/cm); slightly brackish (500 – 2,000 µS/cm); moderately brackish (2,000 – 5,000 µS/cm); brackish (5,000 – 8,500 µS/cm)). Significant differences are bolded. Comparisons with low unique permutation counts that are in red, therefore the Monte Carlo (MC) p-value is looked at.

Salinity Class Comparisons	t	p-value	Unique permutations	P(MC)*
Freshwater vs Slightly Brackish	1.3629	**0.017**	989	**0.038**
Freshwater vs Moderately Brackish	1.5801	**0.001**	780	**0.025**
Freshwater vs Brackish	1.2303	0.197	21	0.241
Slightly Brackish vs Moderately Brackish	1.4133	**0.009**	998	**0.028**
Slightly Brackish vs Brackish	1.3241	0.031	312	0.073
Moderately Brackish vs Brackish	1.0091	0.446	55	0.412

* MC is an additional Monte Carlo test ran concurrently with the PERMANOVA

Table 2.9 Summary table of the degree of similarity from a Pairwise multivariate permutation analysis of variance (PERMANOVA) in terms of community composition between/within wetland salinity classes (freshwater (< 500 µS/cm); slightly brackish (500 – 2,000 µS/cm); moderately brackish (2,000 – 5,000 µS/cm); brackish (5,000 – 8,500 µS/cm)). Salinity ranges with a similarity greater than 25% are in blue, age ranges with a similarity less than 25% are in yellow.

Salinity Class	Freshwater	Slightly Brackish	Moderately Brackish	Brackish
Freshwater	25.46			
Slightly Brackish	22.50	27.08		
Moderately Brackish	21.53	26.94	32.97	
Brackish	25.26	23.30	35.27	43.16

Table 2.10 Summary table for a one-way Analysis of Variance (ANOVA) test comparing the mean species richness among age classes.

	SS	DF	MS	F-Value	p-Value
Age Class	43.552	4	10.888	0.163	0.96
Error	2331.548	35	66.616		

Table 2.11 Multivariate permutation analysis of variance (PERMANOVA) of community composition (based on vegetative species abundance) in response to age and disturbance class. Age was treated as an ordinal variable (age) in one analysis and as a fixed categorical variable (age class) in a second analysis. Age classes are as defined: 1 – 4 years old (n = 9); 5 – 9 years old (n = 7); 10 – 14 years old (n = 18); 15 – 20 years old (n = 3); 21 – 40 years old (n = 3). Disturbance class is a binary variable indicating whether the wetland was on a reference (0) or reclaimed landscape (1). Significant differences are bolded.

Source Effect	df	SS	MS	Pseudo-F	p-value	*P(MC)**
Age	16	55605	3475.3	1.511	**0.001**	**0.001**
Residuals	23	52915	2300.7			
Total	39	10852				
Age Class	4	16510	4127.4	1.57	**0.004**	**0.013**
Residuals	35	92011	2628.9			
Total	39	10852				
Age Class x Disturbance Class						
Age Class	4	16695	4173.7	1.740	**0.002**	**0.004**
Disturbance Class	1	9159.8	9159.8	3.818	**0.001**	**0.001**
Interaction	4	10881	2720.3	1.134	0.220	0.261
Residuals	30	71970	2399			
Total	39	10852				

* MC is an additional Monte Carlo test ran concurrently with the PERMANOVA

Table 2.12 Pairwise multivariate permutation analysis of variance (PERMANOVA) results to test for significant differences in community composition among age classes (1 – 4 years old (n = 9); 5 – 9 years old (n = 7); 10 – 14 years old (n = 18); 15 – 20 years old (n = 3); 21 – 40 years old (n = 3)). Significant differences are bolded. Comparisons with low unique permutation counts that are in red, therefore the Monte Carlo (MC) p-value is looked at.

Age Range Comparisons	t	p-value	Unique permutations	P(MC)*
1 – 4 vs 5 – 9	1.0809	0.280	964	0.334
1 – 4 vs 10 – 14	1.2701	0.051	999	0.091
1 – 4 vs 15 – 20	1.4457	0.004	218	0.051
1 – 4 vs 21 – 40	1.1663	0.129	220	0.245
5 – 9 vs 10 – 14	1.108	0.201	998	0.249
5 – 9 vs 15 – 20	1.286	0.112	120	0.162
5 – 9 vs 21 – 40	1.0364	0.356	120	0.362
10 – 14 vs 15 – 20	1.4544	**0.007**	708	**0.035**
10 – 14 vs 21 – 40	1.2683	0.047	677	0.096
15 – 20 vs 21 – 40	1.4195	0.118	10	0.136

* MC is an additional Monte Carlo test ran concurrently with the PERMANOVA

Table 2.13 Summary table of the degree of similarity from a Pairwise multivariate permutation analysis of variance (PERMANOVA) results in terms of community composition between/within wetland age classes (1 – 4 years old (n = 9); 5 – 9 years old (n = 7); 10 – 14 years old (n = 18); 15 – 20 years old (n = 3); 21 – 40 years old (n = 3)). Age ranges with a similarity greater than 25% are in blue, age ranges with a similarity less than 25% are in yellow.

Age Class	1 – 4	5 – 9	10 – 14	15 – 20	21 – 40
1 – 4	25.43				
5 – 9	25.66	28.23			
10 – 14	25.81	27.90	29.67		
15 – 20	19.11	24.64	23.17	32.28	
21 – 40	22.58	26.35	24.15	18.59	25.97

Chapter 3: Influence of Age and Salinity on Plant Productivity (aboveground biomass) in Newly Formed Wetlands

INTRODUCTION

Petroleum extraction activities covers 6,859 km^2 in the Boreal Forest Natural Region in Northern Alberta (Alberta Biodiversity Monitoring Institute (ABMI), 2020). Roughly 140,000 km^2 of Northern Alberta is underlain with oil sands deposits (Government of Alberta (GoA), 2023). Open pit bitumen mining is a form of petroleum extraction from the landscape, and roughly 500 km^2 of 140,000 km^2 is currently undergoing surface mining activities (GoA, 2023). Open pit mining involves the removal of vegetation, soil, and subsoil (including bituminous sand) to a depth of up to 100 meters (Trites and Bayley, 2009), transforming the landscape resulting in the loss of all surface features, including pre-existing wetlands – such as shallow open water wetlands, marshes, and peatlands (Mollard et al., 2013). After bitumen has been mined, the large (several km^2 in area), deep pits are gradually filled with composite tailings, which are high in salts (such as sodium, calcium, magnesium, bicarbonate, and sulphates; Biagi et al., 2019) due to bitumen extraction processes and natural sources (House et al., 2022). Provincial legislation requires that following mine closure the landscape be returned to "equivalent land capability", which supports land uses that existed prior to mining activities (Alberta Environment, 2006). Prior to mining, 64% of the surface area was classified as wetlands, primarily peatlands, and it is projected that a total of 4800 km^2 of the boreal forest (Mollard et al., 2015) and 12,414 km^2 of peatlands (Rooney et al., 2012) in Northern Alberta will be disturbed by mining activities (Mollard et al., 2015) further disturbing wetlands on the landscape.

The most common wetland cover type was fen vegetation, whereas shallow open water, swamp, and marsh wetland habitat were scarce (Rooney et al., 2012). However, due to the natural history of Northern Alberta's landscape (underlain by estuarine and marine sediment deposits; Trites and Bayley, 2009) and bitumen extraction processes (Giesy et al., 2010), the reclaimed landscape is anticipated to contain elevated concentrations of salts (Trites and Bayley, 2009). Therefore, wetlands that are constructed and those that naturally form on these reclaimed landscapes are likely to be more saline than their predecessors (House et al., 2022; Trites and Bayley, 2009). As many marsh plant species tolerate a range of salinity levels better than other wetland plant species (Stewart and Kanturd, 1972), the construction of marsh wetland systems has been the focus of oil sands wetland reclamation practices (Alberta Environment, 2008; Daly et al., 2012; Rooney and Bayley, 2011a). However, areas that have been reclaimed to upland boreal forests have marsh systems that are opportunistically forming.

Wetlands form and are sustained by interactions among precipitation, surface and groundwater hydrology, and evapotranspiration, soil moisture, predominant wetland vegetation species, and external factors that together promote wetland persistence (Hawkes et al., 2020; Little-Devito et al., 2019).

Open pit mining dramatically alters the hydrology in the surrounding landscape. Hydrological disturbances in wetlands can alter soil hydrophysical properties that regulate the water table position, discharge and recharge patterns that govern wetland community spatial configuration and ecosystem processes (i.e., evapotranspiration, and nutrient cycling; Volik et al., 2020). As research on constructed wetlands and the formation of opportunistic wetlands on reclaimed landscapes in the AOSR is relatively new, there is limited knowledge of the function or ecological condition of opportunistic wetlands (Kovalenko et al. 2013; Raab and Bayley, 2013; Roy et al. 2016; Trites and Bayley 2009;) and full scale constructed watersheds (Borkenhagen and Cooper, 2019; Hartsock et al., 2021; Vitt et al., 2016). Measures of wetland plant community ecological function include features such as productivity, biodiversity support, and nutrient cycling (Zelder, 2000). The success of reaching equivalent land capability for wetlands on reclaimed landscapes has been assessed relative to natural wetland analogues using vegetation attributes such as peak aboveground biomass ('biomass' herein; Raab and Bayley 2013). Biomass is an important measure of productivity that may be used to determine if reclamation practices are successful as mining activities and reclamation processes alter ecological conditions (i.e., hydrological conditions, and nutrient availability) that vegetation communities are influenced by (Raab et al., 2014).

Naturally saline marshes occur in the boreal region of Western Canada, including Alberta, but they are infrequent and scattered (Purdy et al., 2005). However, elevated salinity is relatively common in newly formed wetlands such as constructed wetlands (Borkenhagen and Cooper, 2019; Daly et al., 2012; Hartsock et al., 2021), and opportunistic wetlands forming on upland boreal forest reclaimed landscapes (Hawkes et al., 2020; Little-Devito, 2019). Wetlands in northern Alberta can range from freshwater to saline conditions (ESRD, 2015). In this study, the biomass of six dominant emergent and wet meadow species common in 40 young marshes and shallow open water wetlands was assessed to identify the extent to which salinity and the wetlands' age influenced the biomass of these species. Wetlands assessed were newly formed, opportunistic and constructed, and on a reclaimed (mining lease) landscape and in reference areas (off a mining lease). The objectives of this chapter were to: 1) compare how disturbance classes (reclaimed vs. reference landscapes) influence biomass of the dominant plant species; 2) assess how salinity influences the biomass of the dominant plant species; and 3) assess the influence of age on biomass of the dominant plant species.

METHODS

Study Area

The study wetlands were located in the Boreal Highlands Natural Subregion (BHNS; Natural Regions Committee (NRC) 2006), in the Regional Municipality of Wood Buffalo, north and south of Fort McMurray, Alberta (Chapter 1. Figure 1.1). The BHNS is comprised of coniferous and mixedwood forests on rolling hills, with wetlands at the base of the slopes and in nearby topographically low-lying areas and lakes that are scattered across the subregion (NRC 2006). The BHNS is characterized by long cold winters and short warm summers with the coldest temperatures occurring in January (mean = -17.4°C) and the warmest temperatures occurring in July (mean = 17.1°C). The total annual rainfall in Fort McMurray was 316 mm with a total annual snowfall of 134 cm (1981 to 2010 climate normals from the weather station at 56°39 N, -111°13 W; Environment Canada 2023a). The study was conducted during July and August 2021. Air temperatures during July and August were slightly above normal (daily data report for 2021 from the weather station at 56°39'12 N, -111°13'24 W; Environment Canada 2023b). The daily average (SE) temperature in July was 18.7 (1.1) °C, and 16.1 (1.0) °C in August. The total annual precipitation in 2021 was 326 mm (Environment Canada 2023b).

In this study, 40 wetlands were sampled. Twenty wetlands were on Syncrude's Mildred Lake lease (57° 2' 37.9134" N, -111° 35' 26.1636" W) in topographically upland reclaimed areas, while the remaining wetlands were in reference areas, off lease in the surrounding region (see Chapter 2 Section 2.1 for more details). Reference wetlands are defined as naturally occurring wetlands that may have varying extents of anthropogenic disturbance, but are not situated in previously mined areas, such as along a newly (<40 years old) built road.

High resolution satellite imagery (Google Earth Pro, 2021) was used to identify 103 candidate wetlands. The age of sites (number of years since formation) was determined using the timelapse feature in Google Earth Pro (2021). Wetlands that were ground-truthed were ultimately picked based on accessibility around the perimeter of the wetland, specific conductance (µS/cm), and general age of the wetland. To compare site age with salinity, 75 of the 103 sites were visited, and preliminary measurements of specific conductance (µS/cm) of the surface water present in the wetland was recorded in the emergent zone of each wetland. Specific conductance was used as an indicator of salinity (herein described as 'salinity'; ESRD, 2015). Of the 75 wetlands visited, 40 candidate wetlands were selected for sampling according to their age and corresponding salinity. Wetland size of the selected candidates wetlands was determined from 2021 Google Earth imagery using the polygon tool to delineate the area (hectares) of open water area and wet meadow fringe. The overarching program (Boreal Wetland Reclamation Assessment Program) entails three years of field work with the goal of sampling 120 wetlands total (40 wetlands each year).

Data Collection and Processing

To determine variability in biomass with age and salinity, species composition per wetland was determined. To do this, the species present in each wetland were tabulated (ascertained from the vegetation surveys described in Chapter 2 Section 2.2; Table 2.2) and each species' frequency of occurrence among the wetlands was determined. Six characteristic wetland plant species were selected (ESRD 2015): *Carex aquatilis* (water sedge; present at 35 wetlands)*, Carex atherodes* (beaked sedge; present at 13 wetlands)*, Carex utriculata* (small bottle sedge; present at 18 wetlands)*, Calamagrostis canadensis* (bluejoint grass; present at 15 wetlands)*, Schoenoplectus tabernaemontani* (soft-stem bulrush; present at 17 wetlands)*,* and *Typha latifolia* (cattail; present at 36 wetlands).

Above ground biomass ('biomass' hereafter) of the six dominant species in each wetland was determined by clipping and collecting plants in mid August at the approximate time of peak biomass to obtain a measure of each wetland's productivity (Rooney and Bayley, 2011). The species, if present in each wetland, were harvested from the community composition plot that had the greatest percent cover of the single dominant species (Chapter 2 Section 2.2; modified method from Mollard et al., 2013). A 0.25-meter (m) x 0.25 m plot was used to harvest the biomass of each dominant species (one plot per species; up to a maximum of six plots total per wetland). The intent of collecting biomass in the plots where each of the species were densest was to provide an idea of the maximum potential for biomass accumulation for each wetland (Mollard et al., 2013; Rooney and Bayley 2011). The stems of all plants in a plot were clipped 1 cm above the substrate surface. Stems of each of the dominant species were separated from the rest of the harvested material. Harvested vegetation was oven dried at 60 ^{o}C until a constant mass was reached, the sample was then weighed to the nearest 0.1 g and expressed as areal dry biomass (g/m^2).

Statistical Analysis

A student t-test was conducted to determine whether differences in total biomass of a species existed between disturbance classes (reclaimed and reference landscapes). To determine how salinity, and age influence the biomass of each species a multiple regression was performed. To meet the assumptions of normal distribution for a t-test and a multiple regression, species biomass, salinity, and age were log transformed (McCune and Grace, 2002).

RESULTS

Newly formed wetlands (NFW; opportunistic and constructed wetlands) on reclaimed landscapes ranged in area from 0.04 ha (hectares) to 13.5 ha, with a mean (SE) size of 1.4 (0.7) ha. NFW on reference landscapes ranged from 0.1 ha to 8.44 ha, with a mean (SE) size of 1.9 (0.5) ha. Wetlands on the reclaimed landscape were 2 to 28 years old, with a mean (SE) age of 9.2

(1.5), while wetlands on a reference landscape were slightly older with a mean age of 12.6 (2) with an age range of 3 – 40 years old (Table S1). Wetlands on a reclaimed landscape were also more saline, ranging from 615 µS/cm to 8134 µS/cm (slightly brackish to brackish) with a mean salinity of 2849 (497.1) µS/cm. Reference wetlands were freshwater to slightly brackish, with salinity ranging from 277 µS/cm to 1690 µS/cm, with a mean salinity of 823 (84.8) µS/cm (Table S1).

Wetlands on a reference landscape ranged from 3 – 40 years old, with a mean age of 12.6 (2.0), while wetlands on the reclaimed landscape were slightly younger, with a mean age of 9.2 (1.5), with an age range of 2 – 28 years (Table S1). Reference wetlands were freshwater to slightly brackish in nature ranging from 277 µS/cm to 1690 µS/cm, with a mean salinity of 823 (84.8) µS/cm (Table S1). While wetlands on a reclaimed landscape were more saline in nature, from slightly brackish to brackish, with a specific conductance ranging from 615 µS/cm to 8134 µS/cm, with a mean salinity of 2849 (497.1) µS/cm. Wetlands on reclaimed and reference landscapes both had different origins (opportunistic or constructed). There were 18 opportunistic and 2 constructed wetlands studied on the reclaimed landscape, and 7 opportunistic and 13 constructed wetlands on the reference landscape. Wetland size ranged from 0.04 hectares (ha) to 13.50 ha, with a mean (SE) of 1.7 (2.7) ha, and a median of 0.5 ha. A similar size range was captured for both the reclaimed wetlands and the reference wetlands (0.04–13.5 ha and 0.1–8.44 ha, respectively).

Aboveground Biomass in Reclaimed and Reference Wetlands

The mean (SE) total (sum of species) biomass of the all the dominant plant species in wetlands on reclaimed landscapes was 152.8 (13.5) g m^{-2} (n=20) and on reference landscapes was 146.6 (24.1) g m^{-2} (n=20) Table 3.1; Figure 3.1). There was no significant difference in mean total biomass of the dominant plant species between wetlands in reclaimed and reference landscapes (p=0.33; Table 3.2).

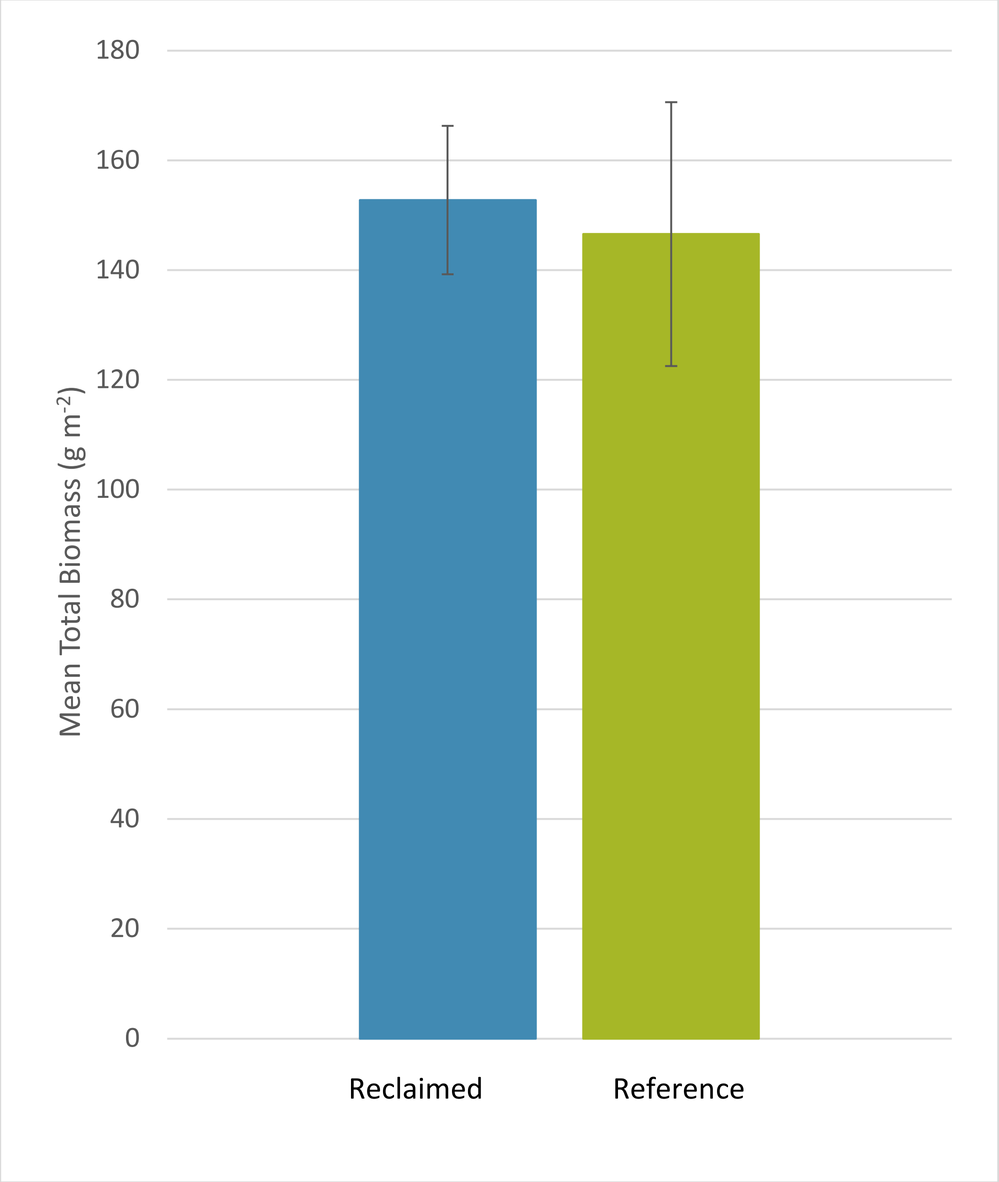


Figure 3.1. Mean (SE) total biomass (g m^{-2}) on a reclaimed and reference landscape (n = 20 for both reclaimed and reference sites). Values in the figure were not log transformed (Table 3.1).

Typha latifolia had the greatest amount of biomass in wetlands on reference and reclaimed landscapes compared to other dominant wetland graminoids (Table 3.1). *Calamagrostis canadensis*, *Carex atherodes,* and *Carex utriculata* produced more biomass in wetlands on reference landscapes than wetlands forming on reclaimed landscapes (Table 3.1). However, differences in the amount of biomass produced was not significantly different ($p > 0.05$; Table 3.2). Of the species that had greater biomass in wetlands on a reference landscape, *C. utriculata* exhibited the greatest difference in biomass between wetlands on a reference landscape and a reclaimed landscape (difference of 35%; Table 3.1). The difference in biomass of *C. canadensis* closely followed that of *C. utriculata* between wetlands on a reference and reclaimed landscape. The biomass of *C. canadensis* was 28% lower in wetlands on a reclaimed landscape compared to wetlands on a reference landscape (Table 3.1). *Carex atherodes* had the smallest difference of the three species, having 9% less biomass in wetlands on a reclaimed landscape compared to wetlands on a reference landscape (Table 3.1). However, the biomass of *Carex aquatilis* and *Schoenoplectus tabernaemontani* was 14% and 38% higher, respectively, on a reclaimed landscape than a reference landscape (Table 3.1; Figure 3.3). However, these differences were not significantly different ($p > 0.05$; Table 3.2).

Table 3.1 Summary table of total biomass and mean (SE) species biomass (g m^{-2}) of the dominant wetland species among wetlands on a reclaimed and reference landscape.

		Disturbance Classes		Sample Size	
		Reclaimed	***Reference***	***Reclaimed***	***Reference***
Mean Biomass (g m^{-2})	***Carex aquatilis***	40.9 (3.8)	35.8 (4.9)	18	17
	Calamagrostis canadensis	43.2 (8.2)	60.2 (11.3)	10	5
	Carex atherodes	48.9 (5.4)	53.7 (12.6)	7	6
	Carex utriculata	19.9 (4.4)	30.8 (8.8)	8	10
	Schoenoplectus tabernaemontani	45.1 (10.8)	32.7 (7.6)	9	8
	Typha latifolia	51.6 (6.4)	66.5 (9.8)	19	17
Total Biomass (g m^{-2})		152.8 (13.5)	146.6 (24.1)	-	-

Table 3.2 Summary table for a student t-test comparing the log transformed total biomass and mean species biomass of the dominant wetland species among wetlands on a reference and reclaimed landscapes (n = 40 for all species and total biomass).

	Regression Coefficient	SE	t-value	p-value	Partial R^2
Total biomass					
Intercept	2.009	0.991	2.027	0.050	
Age	-0.224	1.015	-0.221	0.826	<0.01
Salinity	-0.015	0.294	-0.049	0.961	<0.01
Age x Salinity	-0.224	1.015	-0.221	0.826	0.051
				Total	0.057
Carex aquatilis					
Intercept	2.981	2.208	1.350	0.186	
Age	-1.969	2.261	-0.871	0.390	0.019
Salinity	-0.638	0.656	-0.973	0.337	0.004
Age x Salinity	0.770	0.703	1.095	0.281	0.089
				Total	0.112
Calamagrostis canadensis					
Intercept	-0.346	3.408	-0.102	0.920	
Age	0.085	3.490	0.024	0.981	<0.01
Salinity	0.235	1.013	0.232	0.818	<0.01
Age x Salinity	0.048	1.085	0.044	0.965	<0.01
				Total	0.012
Carex atherodes					
Intercept	-2.916	3.243	-0.899	0.375	
Age	3.567	3.321	1.074	0.290	<0.001
Salinity	1.180	0.964	1.225	0.229	0.009

	Regression Coefficient	SE	t-value	p-value	Partial R^2
Age x Salinity	-1.244	1.033	-1.204	0.236	0.039
				Total	0 .078
Carex utriculata					
Intercept	-5.490	2.708	-2.028	0.050	
Age	4.963	2.773	1.790	0.082	
Salinity	1.720	0.805	2.138	0.039	N/A*
Age x Salinity	-1.377	0.862	-1.596	0.119	
				Total	0.140
Schoenoplectus tabernaemontani					
Intercept	5.013	3.172	1.581	0.123	
Age	-1.335	0.942	-1.416	0.165	0.021
Salinity	-3.428	3.248	-1.055	0.298	0.010
Age x Salinity	1.049	1.010	1.039	0.306	<0.001
				Total	0.066
Typha latifolia					
Intercept	1.669	2.214	0.754	0.456	
Age	-0.109	2.267	-0.048	0.962	0.135+
Salinity	-0.218	0.658	-0.332	0.742	
Age x Salinity	0.225	0.705	0.319	0.751	0.124+
				Total	0.138

*Significant (p<0.05) only in combination with other independent variables.

+ Significant (p<0.05) only if used as the only independent variable.

Relationship between Aboveground Biomass, Salinity, and Age

Despite the range of salinity observed there was no significant difference in mean total (sum of species) biomass and in species biomass across a salinity gradient (Table 3.3). There was no significant difference in mean total biomass and species biomass with age (Table 3.3). Mean total biomass was slightly lower at higher concentrations of salinity than lower concentrations; however, biomass slightly increased as wetlands aged (Figure 3.2). Biomass of *Carex aquatilis* was greater in older wetlands, but biomass was relatively invariant across a gradient of salinity and among ages (Figure 3.3). *C. aquatilis biomass* was lower in younger wetlands that were more saline than older wetlands with similar salinity conditions. However, the difference in biomass when accounting for age and salinity were not significant (Table 3.3). *Calamagrostis canadensis* biomass was similar across a gradient of salinity, but progressively increased in older wetlands (Figure 3.4). However, *C. canadensis* was more commonly observed in younger wetlands. *Carex atherodes* biomass was lower in saline than in freshwater wetlands, whereas *C. atherodes* biomass was higher in younger wetlands than older wetlands (Figure 3.5). However, *C. atherodes* was more commonly observed in younger wetlands. *Carex utriculata's* biomass was progressively lower in wetlands with increased salinity, whereas *C. utriculata's* biomass was higher in younger wetlands than older wetlands (Figure 3.6). However, *C. utriculata's* was more commonly observed in younger wetlands. *Schoenoplectus tabernaemontani*'s biomass was progressively greater across the gradient of salinity, but progressively decreased with age (Figure 3.7). *Typha latifolia*'s biomass was slightly lower in wetlands with higher salinity levels, but biomass tended to be higher in older wetlands (Figure 3.8).

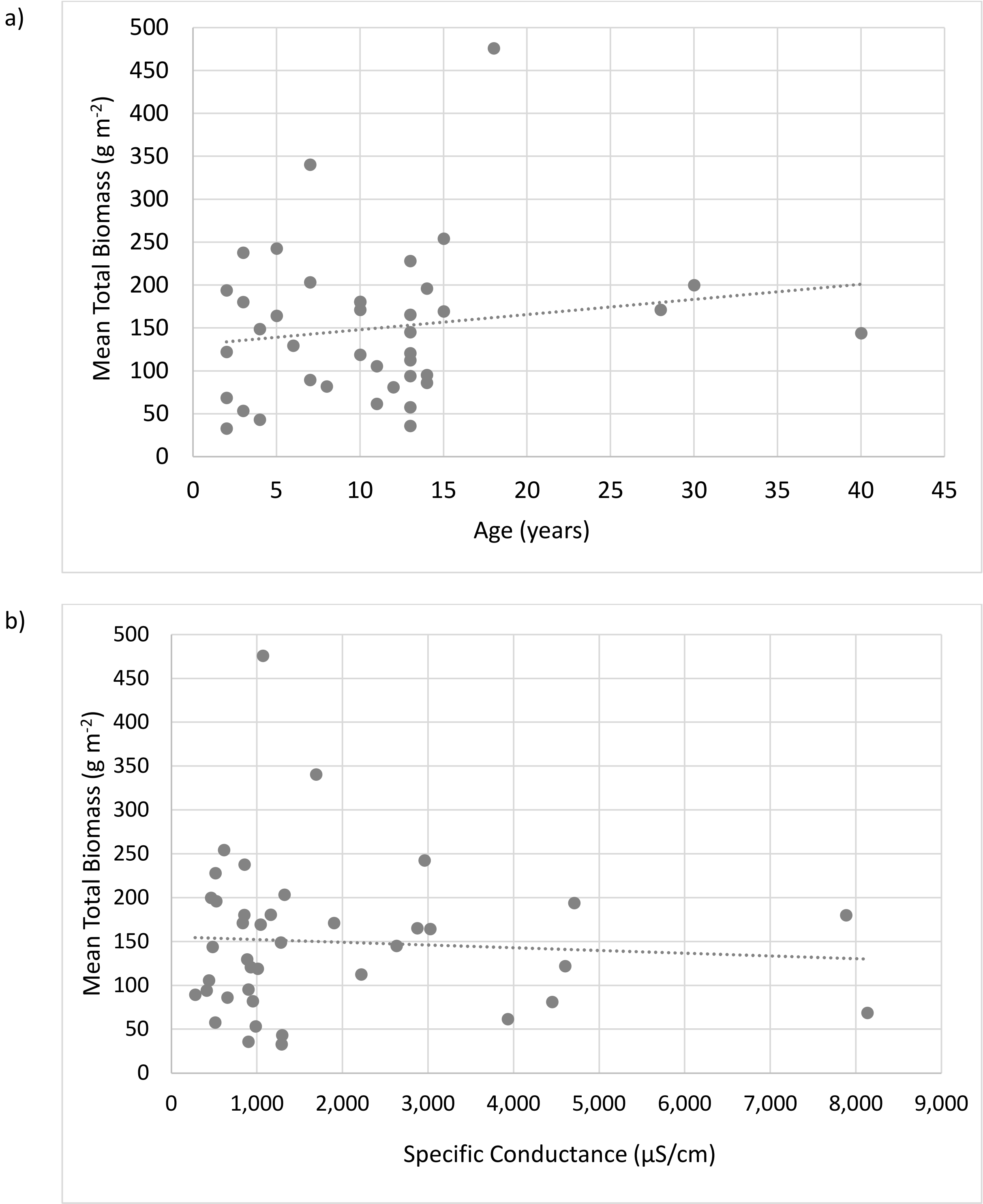


Figure 3.2 The relationship between **a)** age (years) and **b)** salinity (µS/cm) with total biomass (g m^{-2}) for the six dominant plant species. Values in figure are not transformed (n = 40).

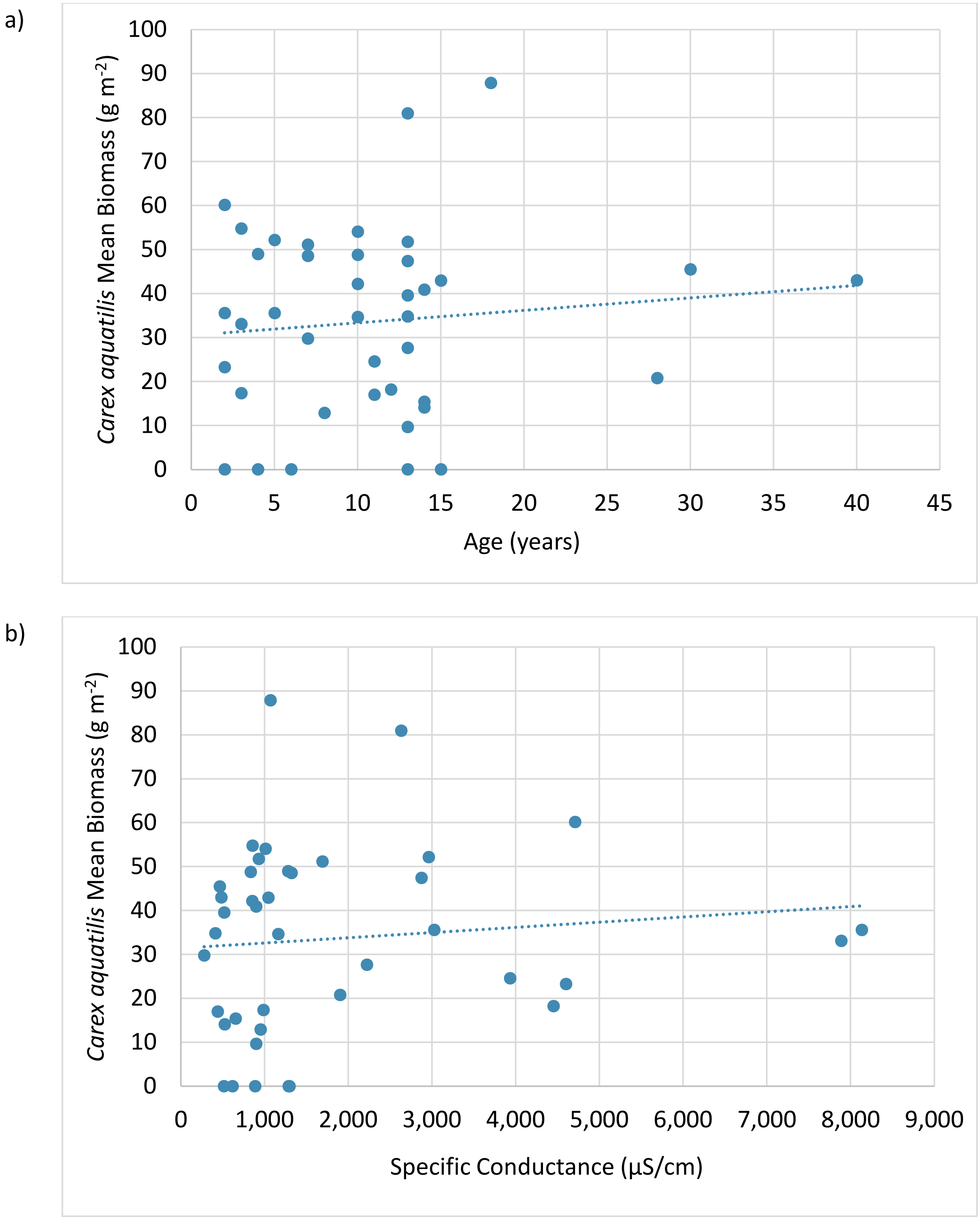


Figure 3.3 The relationship between **a)** age (years) and **b)** salinity (µS/cm) with *Carex aquatilis* biomass (g m^{-2}). Values in figure are not log transformed (n = 40).

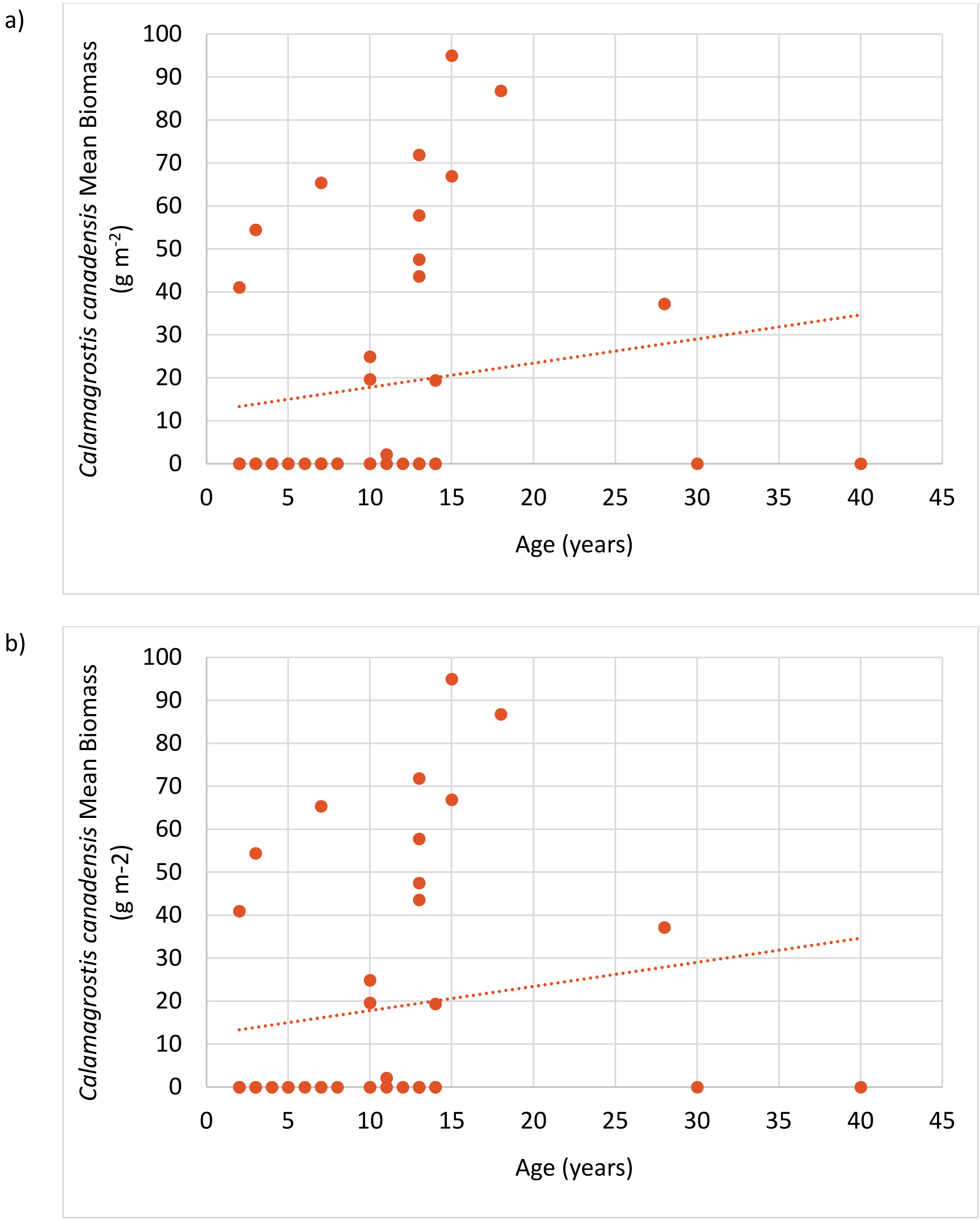
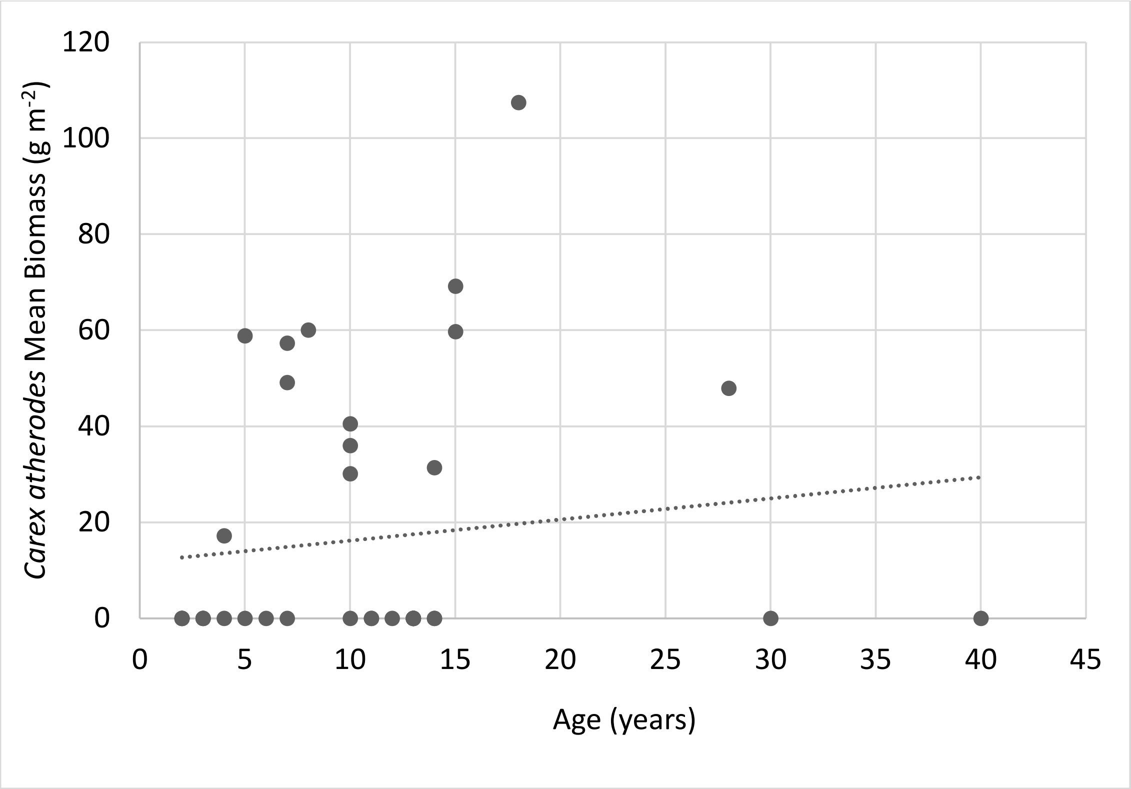


Figure 3.4 The relationship between **a)** age (years) and **b)** salinity (µS/cm) with *Calamagrostis canadensis* biomass (g m^{-2}). Values in figure are not log transformed (n = 40).

a)



b)

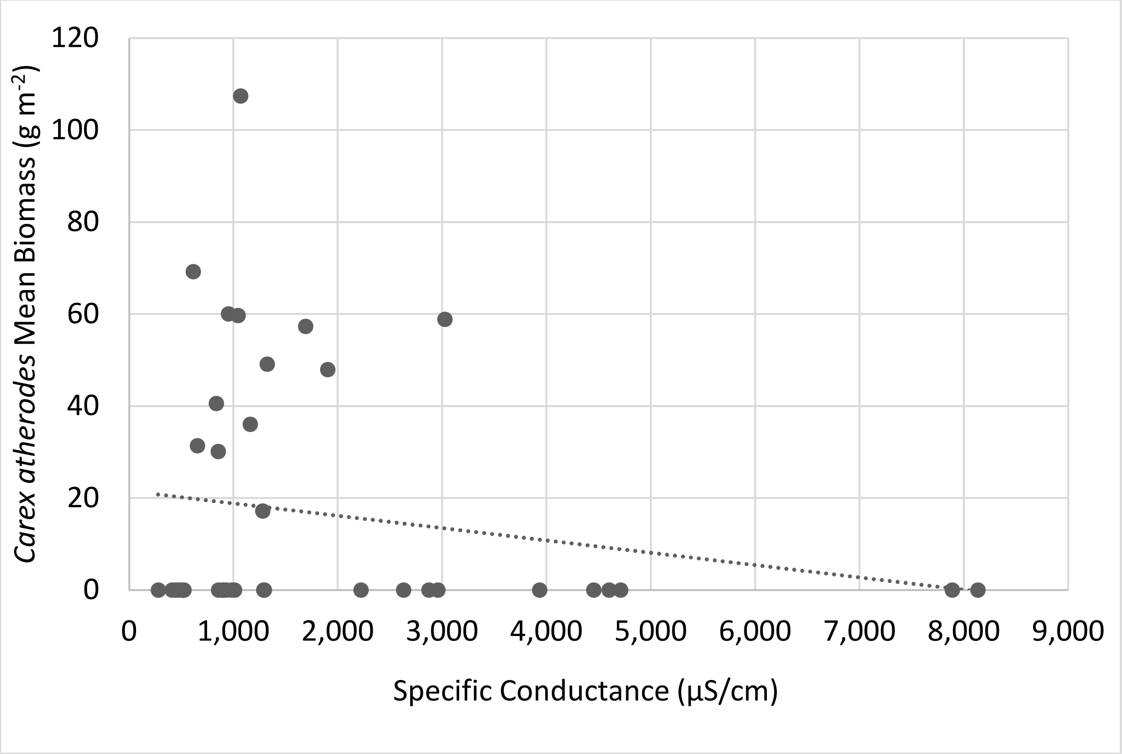
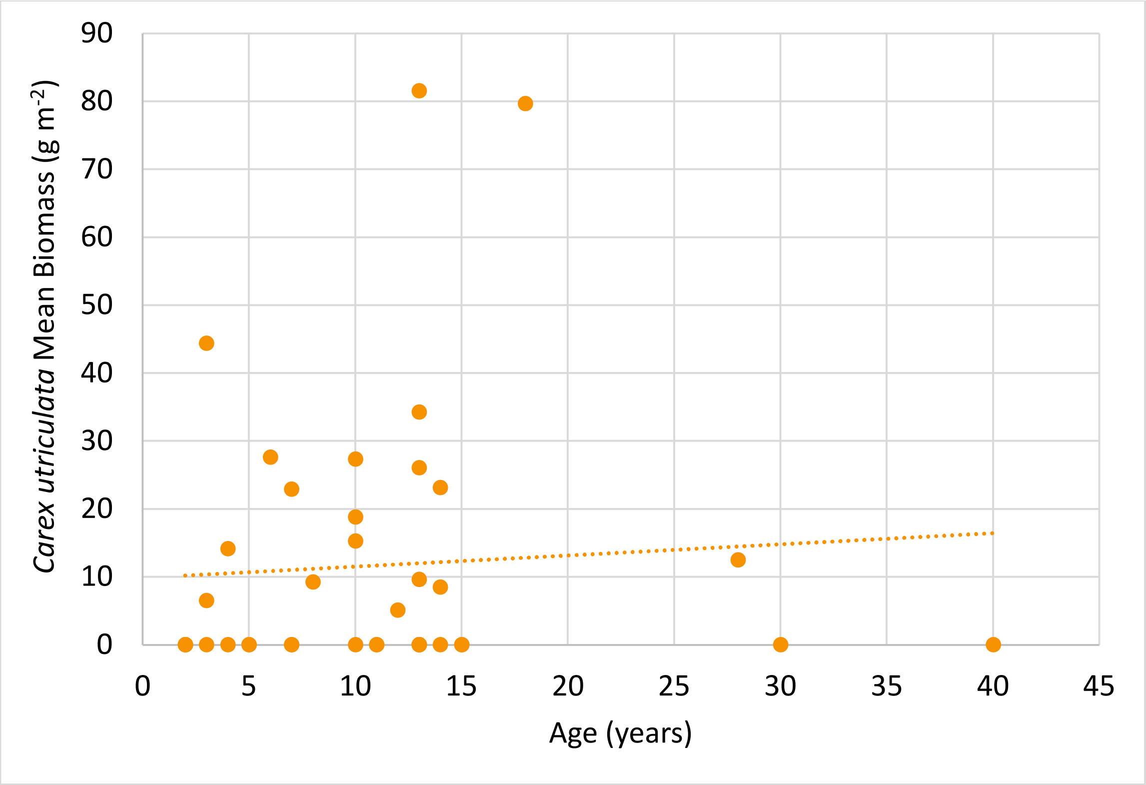


Figure 3.5 The relationship between **a)** age (years) and **b)** salinity (µS/cm) with *Carex atherodes* biomass (g m^{-2}). Values in figure are not log transformed (n = 40).

a)



b)

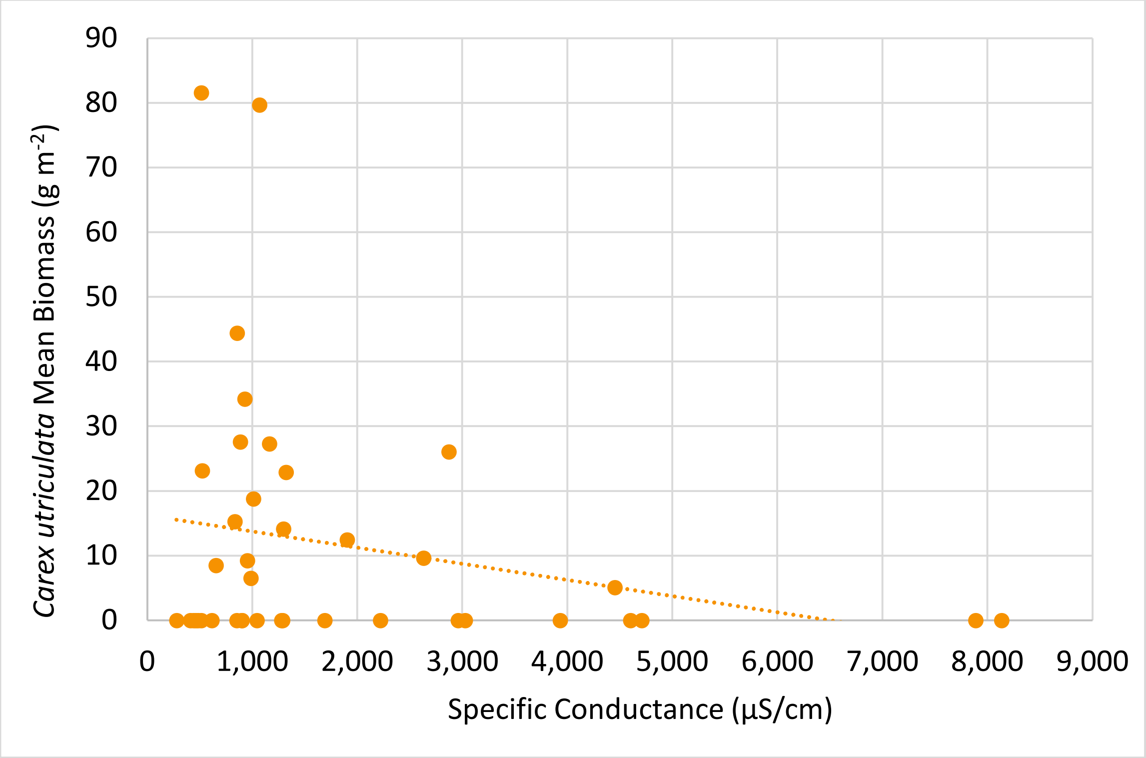
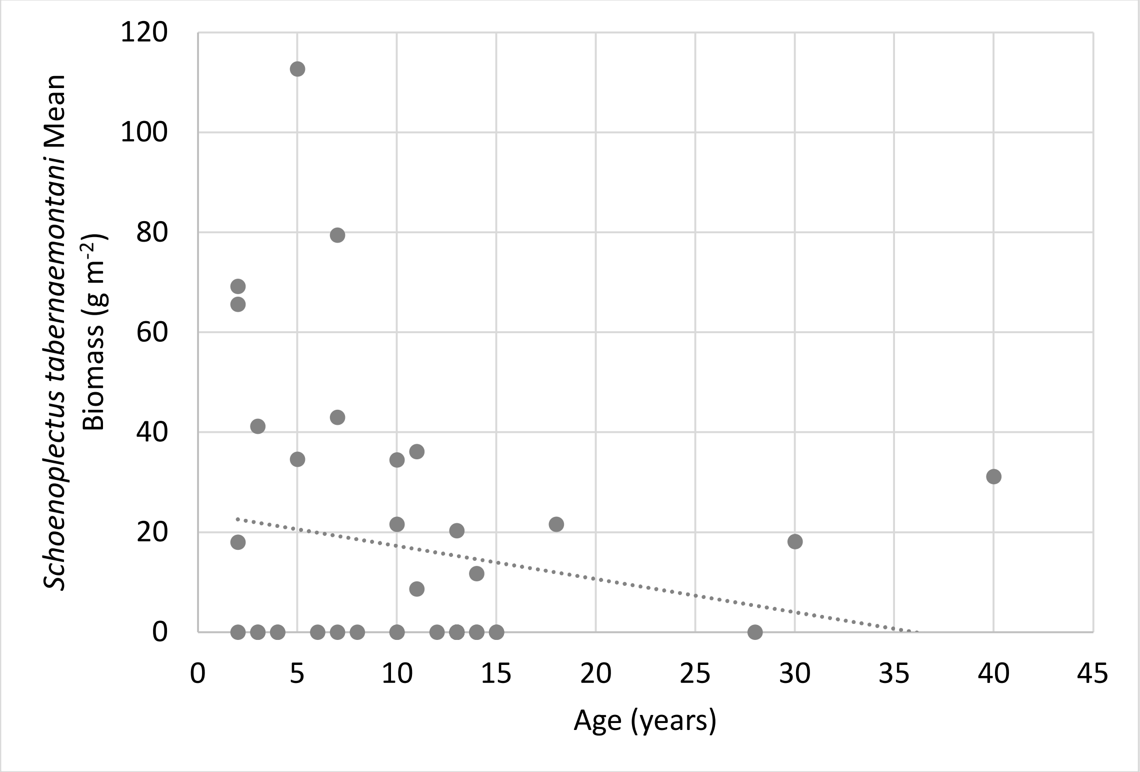


Figure 3.6 The relationship between **a)** age (years) and **b)** salinity (µS/cm) with *Carex utriculata* biomass (g m^{-2}). Values in figure are not log transformed (n = 40).

a)



b)

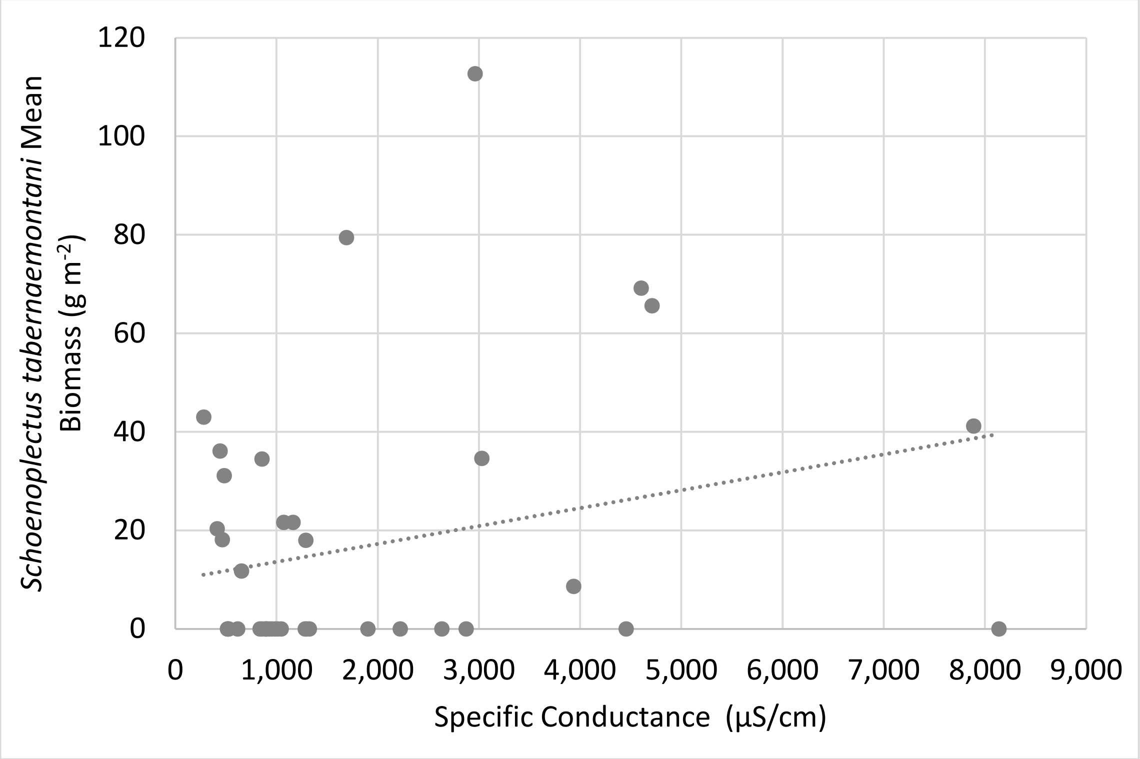


Figure 3.7 The relationship between **a)** age (years) and **b)** salinity (µS/cm) with *Schoenoplectus tabernaemontani* biomass (g m^{-2}). Values in figure are not log transformed (n = 40).

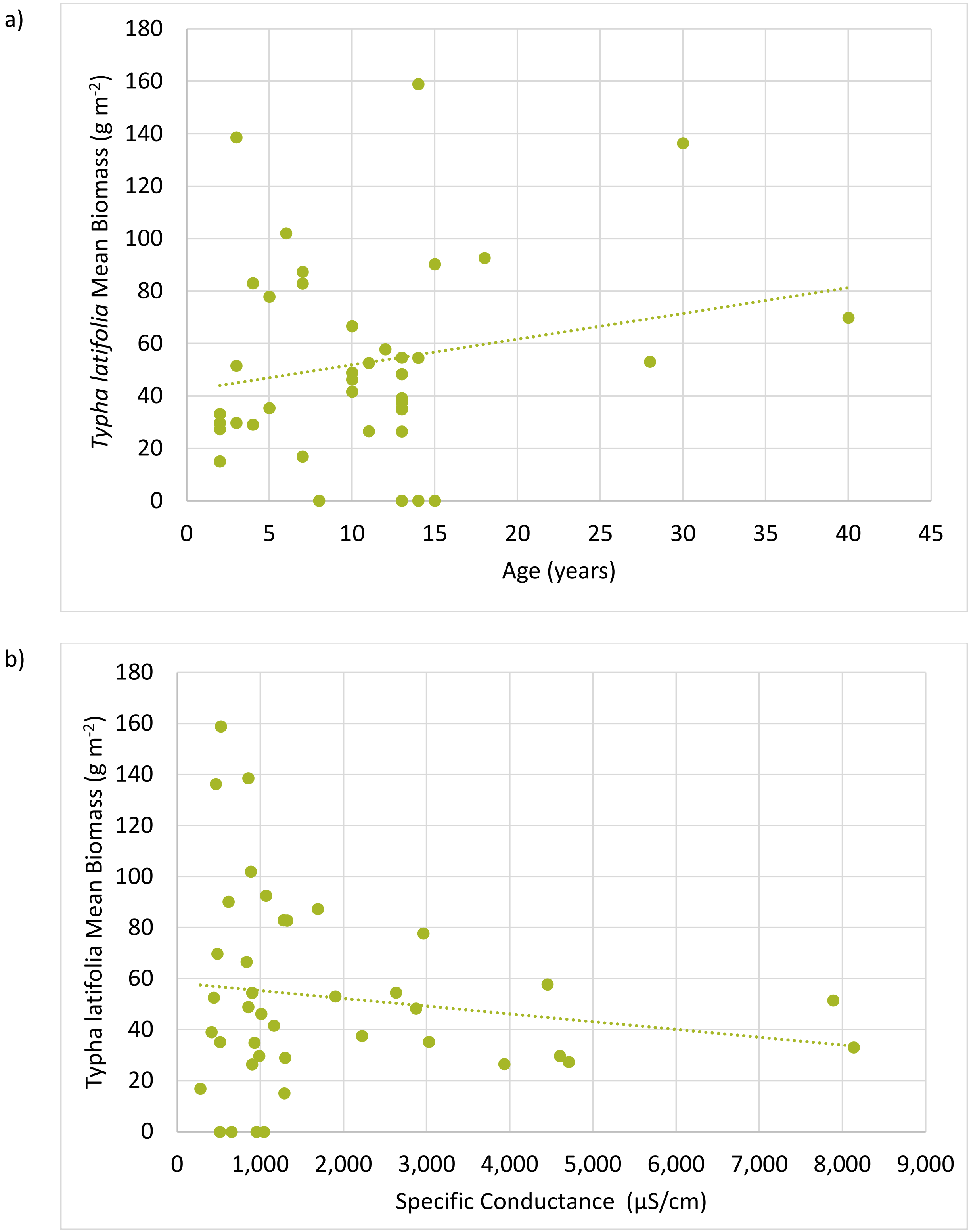


Figure 3.8 The relationship between **a)** age (years) and **b)** salinity (µS/cm) with *Typha latifolia* biomass (g m^{-2}). Values in figure are not log transformed (n = 40).

Table 3.3 Summary results of the multiple regression displaying the relationship between biomass and salinity, age, and their interaction. Biomass, salinity, and age values were log transformed.

	R^2	F-value	Regressions Correlation Coefficient	t-value	p-value	N
Total biomass						
Age	0.047	0.591	0.54	0.383	0.704	40
Salinity			0.18	0.401	0.691	
Age x Salinity			-0.26	-0.209	0.835	
Carex aquatilis						
Age	0.251	3.472	-2.02	-1.554	0.13	35
Salinity			-0.78	-1.863	0.072	
Age x Salinity			2.17	1.856	0.073	
Calamagrostis canadensis						
Age	0.077	0.305	-0.2	-0.068	0.947	15
Salinity			0.06	0.066	0.948	
Age x Salinity			0.47	0.179	0.861	
Carex atherodes						
Age	0.08	0.261	-1.05	-0.402	0.697	13
Salinity			-0.14	-0.151	0.883	
Age x Salinity			0.75	0.353	0.732	
Carex utriculata						
Age	0.212	1.255	-3.35	-1.299	0.215	18
Salinity			-0.99	-1.37	0.192	
Age x Salinity			2.94	1.394	0.185	
Schoenoplectus tabernaemontani						
Age	0.265	1.563	-5.23	-1.56	0.143	17
Salinity			-2.37	-1.608	0.132	
Age x Salinity			5.26	1.667	0.119	
Typha latifolia						
Age	0.01	0.105	0.54	0.357	0.723	36
Salinity			0.24	0.467	0.644	
Age x Salinity			-0.5	-0.371	0.713	

DISCUSSION

Disturbance Class and the influence on Dominant Species Biomass

Using wetland reference sites as a baseline measure against which to assess ecological conditions and functions, such as productivity allows one to assess whether the practices used to reclaim landscapes are successful relative to that frame of reference. Total biomass of the dominant species averaged across the 20 wetlands on reclaimed landscapes was not significantly different than total biomass averaged across the 20 wetlands studied from reference landscapes. The biomass of six dominant species was measured for *Carex aquatilis, Calamagrostis canadensis, Carex atherodes, Carex utriculata, Schoenoplectus tabernaemontani,* and *Typha latifolia*, which are all common species seen on reclaimed and reference landscapes (Daly et al., 2012; Hartsock et al., 2021; Glaeser et al., 2019; Mollard et al., 2012; Mollard et al., 2013; Roy et al., 2016; Vitt et al., 2020;). Of these six species *T. latifolia* produced the most biomass on both reclaimed and reference landscapes, while *C. utriculata* had the lowest biomass in wetlands on reclaimed and reference landscapes. The biomass of *C. canadensis* and *C. atherodes* was greater in wetlands on reference landscapes than on reclaimed landscapes, whereas the biomass of *C. aquatilis* and *S. tabernaemontani* was higher on reclaimed landscapes. *Carex aquatilis* withstands a broad variety of environmental conditions and is tolerant to saline conditions and relatively dry soils (depth to water table) (Mollard et al., 2013; Vitt et al., 2020), likely explaining why *C. aquatilis* biomass is higher in wetlands on a reclaimed landscape. Similarly, *S. tabernaemontani* establishes better in saline conditions than freshwater conditions (Batistel et al. 2021), which could explain the greater biomass observed between the reference and reclaimed landscapes. Even though some species performed better on reference landscapes than reclaimed landscapes and *vice versa* the difference in biomass produced per species was not significantly different between disturbance classes (reclaimed vs reference landscape). Similar findings were noted by Raab and Bayley (2013) when investigating biomass of wetland vegetation communities on reclaimed and reference landscapes.

Differences in overall biomass of the six dominant species, and of individual species biomass between disturbance classes may reflect environmental conditions of the wetland such as water chemistry (Vitt et al., 2020; Roy et al., 2016; Raab and Bayley, 2013), nutrient availability, age of the wetland (Raab and Bayley, 2013) or fluctuating water table levels (Vitt and Hartsock, 2022). The substrate type or soil placement material may also be important, as materials differ between wetlands on reclaimed and reference landscapes (Roy et al., 2016; Table S2).

Biomass of Dominant Species Response to Salinity

Even though wetlands on reclaimed landscapes were more saline than those in reference areas, mean species biomass was not significantly different among of the classes of salinity (freshwater to brackish conditions). The dominant species studied can are tolerant to a wide range of salinity concentrations (ESRD, 2015; Mollard et al., 2012; Mollard et al., 2013; Glaeser et al., 2019; Vitt et al., 2020) possibly explaining why there was no difference observed in biomass produced across a gradient of salinity.

Although the dominant plant species investigated in this study are common on both reference and reclaimed landscapes, *C. aquatilis* and *T. latifolia* are more common species studied investigating the effects of salinity (sodium) on plant function and physiological attributes (Koropchak and Vitt, 2013; Mollard et al., 2013; Vitt et al., 2020). *Carex aquatilis* biomass was similar across salinity classes. However, Vitt et al. (2020) noted that biomass of *C. aquatilis* decreases as sodium concentrations exceed 1079 mg L^{-1}, which corresponds to brackish conditions (Table S2 & S1), corroborating my observation that biomass of *C. aquatilis* was the lowest in brackish wetlands among all salinity classes studied. *Typha latifolia* produced more biomass in freshwater and slightly brackish wetland classes than in other classes; biomass was lower in moderately brackish wetlands. Koropchak and Vitt (2013) noted a similar trend when investigating the effects of sodium on *T. latifolia* growth. Biomass appeared to be substantially lower at sodium concentrations of 600 mg L^{-1}. Similar concentrations were observed in moderately brackish wetlands (Table S2 & S1). Indicating that the biomass of *T. latifolia* may be impeded by the saline conditions that are more typical of wetlands on reclaimed landscapes than on reference landscapes (Mollard et al., 2013). *Carex atherodes,* another common wetland species, produced more biomass than *C. aquatilis* when exposed to sodium concentrations of 789 mg L^{-1} (Glaesar et al., 2021), which is similar to moderately brackish conditions (Table S2 & S1) and biomass declines as sodium concentrations increase (Glaeser et al., 2021). Findings presented by Glaesar et al. (2021) support the results found in this study as *C. atherodes* had the highest amount of biomass in moderately brackish wetlands. Similar conclusions likely apply to the other dominant wetland plant species investigated in this study.

Biomass of Dominant Species Relative to wetland Age

There is limited research documenting how biomass production of plant species varies as a function of wetland age. Biomass is more commonly used to assess forest stands (Farden, 2021) than wetland carbon accrual. Raab and Bayley (2013) noted that age can limit biomass production of *Carex* species, as age affects the extent of belowground biomass, which can serve as a source of biomass regeneration between years. However, there was no difference in total biomass and mean species biomass with wetland age.

Limitations and Future Research

Most of the wetlands surveyed in this study were between the ages of 10 – 18 years (n = 21). Few wetlands were between the ages of 20 – 40 years (n = 3) and the remaining wetlands were less than 9 years old (n = 16). Additionally, most of the wetlands were slightly brackish in nature on both the reclaimed and reference landscapes, and only freshwater wetlands were observed on reference landscapes while only moderately brackish and brackish wetlands were observed on reclaimed landscapes. Therefore, older wetlands on reclaimed and reference landscapes, and wetlands that are more saline in nature on reference landscapes and freshwater wetlands on reclaimed landscapes should be studied in future monitoring years of the Boreal Wetland Reclamation Assessment Program.

Additionally, to better understand how other environmental variables contribute to biomass production as observed in other studies (Vitt et al., 2020; Raab and Bayley, 2013; Vitt and Hartsock, 2022; Roy et al., 2016) depth to water table, soil placement material, and water and soil chemistry specifically sodium, magnesium, calcium, and sulphate should be studied.

CONCLUSIONS

Biomass did not differ among wetlands among classes representing a gradient of salinity, or among wetlands of different ages. Biomass of the most frequently occurring species was similar in wetlands on reclaimed and reference landscapes, indicating that the biomass of the dominant species established in newly formed wetlands on the young reclaimed landscapes studied are reaching and exceeding the biomass produced on the reference landscape examined.

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Chapter 4: General Discussion

Overview and Objectives

The Athabasca Oil Sands Region is the third largest bitumen reserve globally, containing 168 billon barrels of recoverable bitumen (Government of Alberta (GoA), 2014). Provincial regulation requires that disturbed land is reclaimed to equivalent land capabilities to enable land uses following post-mining reclamation, similar to the uses that existed before mining (Raab and Bayley, 2013), consisting of a mosaic of upland forests and wetlands (Hawkes et al., 2020). However, wetlands forming on these reclaimed landscapes are expected to vary greatly in salinity as salts in the reclaimed landscape rise to the surface (Purdy et al., 2005; Rooney and Bayley, 2011). As mine reclaimed landscapes are relatively new there is limited knowledge around the influence of salinity and age on vegetation community composition and productivity in wetlands that have opportunistically formed in upland forest areas. To better understand these relationships, wetlands on reclaimed landscapes were compared to reference wetlands, which are operationally defined as naturally occurring wetlands that may be subject to varying degrees of anthropogenic disturbance (e.g., forming along a newly a newly built road), but that are not found on previously mined areas.

Wetlands on the reference landscape can provide a frame of reference against which to compare vegetation community composition and biomass (as a measure of productivity) in wetlands situated on reclaimed landscapes. Few studies have compared wetlands that have formed on reclaimed landscapes to wetlands of equivalent age on reference landscapes, and the associated effects to water quality, including salinity, on vegetation communities and biomass (Raab and Bayley, 2013; Roy et al., 2016; Trites and Bayley, 2009). These studies found that wetland vegetation communities on reclaimed landscapes differed from those on reference landscapes and attributed some of these differences to salinity. Oilsands landscape effects of salinity can be confounded with possible effects of other mine byproducts (i.e., naphthenic acids, alkaline conditions, etc.) and historical marine sediment deposits present that are naturally high in salts (Raab and Bayley, 2013). Raab and Bayley (2013) contrasted naturally saline wetlands with industrial saline wetlands to determine community composition and productivity; but there, age was a confounding factor as the age of the naturally saline wetlands are unknown. Raab and Bayley (2013) noted that a wetland's age can limit estimates of biomass production as some species such as *Carex*, use rhizomes (below ground biomass) as a mechanism to persist in an area year after year. Additionally, several studies have examined the effect of salinity on the dominant wetland plant species investigated in this study such as *Carex aquatilis* (Mollard et al, 2012; Raab and Bayley, 2013; Vitt et al., 2020) and *Typha latifolia* (Koropchak and Vitt, 2013; Mollard et al., 2013), less commonly studied species are *Carex atherodes* (Glaesar et al. 2021) and *Schoenoplectus tabernaemontani* (Batistel et al. 2021). All studies noted that the aboveground biomass produced by each species was not inhibited by saline conditions unless conditions exceeded moderately brackish levels (Environment of Sustainable Resource Development, 2015). Similar conclusions likely apply to the remaining two species (*Carex utriculata* and *Calamagrostis canadensis*) investigated in this study.

The objective of this study was to summarize the effects of salinity and age on vegetation community composition and productivity in wetlands on reclaimed and reference landscapes in the Athabasca Oil Sands Region. In particular, the aim was to compare the wetland vegetation community composition in wetlands of various ages, and across a gradient of salinity among wetlands on reclaimed and reference landscapes; and assess how these parameters influence dominant plant species aboveground biomass in wetlands on reclaimed and reference landscapes.

Major Findings and Applications

The wetlands examined on reclaimed landscapes were more saline than those on reference landscapes, which resulted in different community compositions between the two landscapes. However, four of eight species assemblages had wetlands that were compositionally similar (based on vegetative species abundance) in the same assemblages, with four of the eight assemblages containing mixtures of wetlands from reference and reclaimed landscapes. This finding is promising because the community composition in newly formed wetlands on reclaimed landscapes are similar to some newly formed wetlands on reference landscapes.

Community composition differed between newly formed wetlands on reclaimed and reference landscapes, and among salinity classes representing a gradient of salinity, and among wetlands of different ages. Disturbance class (reclaimed or reference landscapes) appears to be a strong predictor of the differences observed in community composition seen in these newly formed wetlands between landscape types. However, the correlation between age and salinity with vegetation communities was weak. Other environmental variables likely account for the differences observed in community composition. Variables noted that play a role in determining community composition differences include available nutrients (Raab and Bayley, 2013), wetland origin (constructed or natural) and soil placement material used during reclamation practices (Roy et al., 2016), water depth and pH (Trites and Bayley, 2009). Thus, although salinity plays a role in community composition other possibly equally important variables could be considered when developing reclamation practices.

Biomass did not differ between wetlands on reclaimed and reference landscapes, and across a gradient of salinity, or among wetlands of different ages. Even though wetlands on a reclaimed landscape are more saline than those on a reference landscape, the productivity (biomass) of wetlands on a reclaimed landscape is unaffected by the level of salinity present. This is important as it is a promising sign that perhaps wetlands forming on reclaimed landscapes provide suitable habitat for species, as the amount of biomass produced does not differ between disturbance classes.

Study Limitations and Future Research

The Boreal Wetland Reclamation Assessment Program (BWRAP) is a 5-year study designed to assess the ecological condition of wetlands in Athabasca oil sands reclamation landscapes and to develop measures of reclamation success. Processes and characteristics being investigated include hydroperiod (permanence verses impermanence), topography (disturbance class), water quality (salinity, nutrients, cations and anions) and biological features (vegetation, invertebrates, avifauna) of 120 wetlands younger than 40 years in age, assessed over a 3-year period. This study investigated 40 wetlands during the first year of field collections. However, the wetlands included in this study were primarily wetlands that were 10 – 15 years old, with more younger wetlands than older wetlands, as there were only three wetlands between the ages of 21 – 40. This limits the knowledge around community composition and biomass in older wetlands. Therefore, the BWRAP should locate and incorporate additional candidate wetlands that are older (21 – 40 years old) to provide a greater understanding on the effects of salinity and age on wetland vegetation community composition and biomass. Additionally, the most saline (moderately brackish and brackish) wetlands studies were located only on reclaimed landscapes, whereas freshwater wetlands were commonly located on reference landscapes. Saline marshes do exist in northern Alberta in reference areas (Rooney and Bayley, 2011), although identifying newly formed systems may be difficult. Nevertheless, locating more saline marshes on a reference landscape as natural analogues for comparison to saline wetlands on reclaimed landscapes will help identify similarities and differences in community composition and biomass. Considering other variables in addition to salinity and age will be important to characterize what is driving the differences seen across reclaimed and reference landscapes. Other driving variables to investigate are soil placement material, wetland origin (constructed or natural; Roy et al., 2016), hydrologic regime (Cooper et al., 2006), and depth to water table (Borkenhagen et al., 2023; Cooper et al., 2006).

Conclusions

Despite the limitations associated with this study, this research has described the community composition and productivity (biomass) of young (<40 year old) newly formed wetlands (opportunistic and constructed wetlands) on reclaimed and reference landscapes. Community composition was different between wetlands on reclaimed and reference landscapes, and among classes representing a gradient of salinity, and among wetlands of different aged. Biomass of the six species was not different between wetlands on reclaimed and reference landscapes, and across a salinity gradient, and with age. These findings provide a frame of reference for assessing the role of salinity and age in opportunistic and constructed wetlands on reclaimed post-mining landscapes.

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Appendix A. Supplementary Data

Table S1. Summary of wetland site characteristics and associated environmental variables (RF = reference landscape; RC = reclaimed landscape; CW = constructed wetland; OW = opportunistic wetland).

Wetland Name	Age (years)	Age Class	Specific Conductance (µS/cm)	Salinity Class*	Depth to Water Table (cm)	Disturbance Class	Design	Wetland Classification*
63 N 001	14	10 - 15	523	Slightly Brackish	8.85	RF	CW	Shallow open water
63 N 002	14	10 - 15	899	Slightly Brackish	23.11	RF	CW	Shallow open water
63 N 003	13	10 - 15	928	Slightly Brackish	18.8	RF	CW	Shallow open water
63 N 004	13	10 - 15	899	Slightly Brackish	6.94	RF	CW	Shallow open water
63 S 001	13	10 - 15	410	Fresh water	3.12	RF	CW	Shallow open water
63 S 002	14	10 - 15	653	Slightly Brackish	-37.96	RF	OW	Shallow open water
Aostra Pit 1	13	10 - 15	512	Slightly Brackish	40	RF	CW	Shallow open water
Aostra Pit 4	13	10 - 15	514	Slightly Brackish	15.67	RF	CW	Shallow open water
Borrow Pit 1	11	10 - 15	439	Fresh water	-6.14	RF	CW	Shallow open water
Borrow Pit 3	8	5 - 9	951	Slightly Brackish	26.09	RF	CW	Shallow open water
Erics Pond	10	10 - 15	834	Slightly Brackish	7.1	RC	CW	Shallow open water
EIP 1	12	10 - 15	4451	Moderately Brackish	-1.26	RC	OW	Shallow open water

Wetland Name	Age (years)	Age Class	Specific Conductance (µS/cm)	Salinity Class*	Depth to Water Table (cm)	Disturbance Class	Design	Wetland Classification*
EIP 2	11	10 - 15	3932	Moderately Brackish	10.7	RC	OW	Shallow open water
EIP 3 A	2	1 - 4	4600	Moderately Brackish	40	RC	OW	Marsh
EIP 3 B	2	1 - 4	8134	Brackish	39.76	RC	OW	Marsh
EIP 3 E	2	1 - 4	4708	Moderately Brackish	40	RC	OW	Marsh
EIP 4	3	1 - 4	7886	Brackish	36.78	RC	OW	Marsh
East Pond	6	5 - 9	884	Slightly Brackish	-8.61	RF	OW	Marsh
Gravel Pit	30	21 - 40	462	Fresh water	-7.36	RF	OW	Shallow open water
Hanging Stone A	4	1 - 4	1295	Slightly Brackish	1.74	RF	CW	Shallow open water
Hanging Stone B	4	1 - 4	1279	Slightly Brackish	-21.08	RF	CW	Shallow open water
Muskeg River Wetland	40	21 - 40	482	Fresh water	-12.05	RF	OW	Shallow open water
North Base Mine	13	10 - 15	2630	Moderately Brackish	1.29	RC	OW	Shallow open water
Parsons Creek	7	5 - 9	277	Fresh water	33.08	RF	CW	Shallow open water
Ramp Wetland	7	5 - 9	1690	Slightly Brackish	-12.4	RF	OW	Shallow open water
Small Borrow Wetland 3	3	1 - 4	984	Slightly Brackish	11.09	RF	OW	Marsh
Sandhill Fen	10	10 - 15	1161	Slightly Brackish	-8.29	RC	CW	Shallow open water

Wetland Name	Age (years)	Age Class	Specific Conductance (µS/cm)	Salinity Class*	Depth to Water Table (cm)	Disturbance Class	Design	Wetland Classification*
South Pond	7	5 - 9	1321	Slightly Brackish	-11.25	RF	CW	Shallow open water
Shallow wetland	28	21 - 40	1901	Slightly Brackish	-2.75	RC	CW	Shallow open water
Southern Wetland	18	15 - 20	1068	Slightly Brackish	-8.52	RF	OW	Shallow open water
SWSS 02	13	10 - 15	2220	Moderately Brackish	18	RC	OW	Shallow open water
SWSS 03	13	10 - 15	2872	Moderately Brackish	12.6	RC	OW	Marsh
SWSS 04	5	5 - 9	3025	Moderately Brackish	-15.09	RC	OW	Marsh
SWSS 05	5	5 - 9	2958	Moderately Brackish	-8.38	RC	OW	Marsh
SWSS 08	2	1 - 4	1287	Slightly Brackish	40	RC	OW	Marsh
SWSS 14	15	15 - 20	615	Slightly Brackish	37.5	RC	OW	Shallow open water
SWSS 15	15	15 - 20	1043	Slightly Brackish	36.5	RC	OW	Shallow open water
W1 45	10	10 - 15	1009	Slightly Brackish	-8.59	RC	OW	Shallow open water
W1 46	10	10 - 15	852	Slightly Brackish	5.46	RC	OW	Shallow open water
W1 47	3	1 - 4	854	Slightly Brackish	1.3	RC	OW	Marsh

Note: * indicates the classification follows the Alberta Wetland Classification System (ESRD 2015).

Table S2. Select water quality and water quantity parameters, and associated substrate type for each wetland. Substrate types are as follows: Natural = wetlands that are present on a reference landscape, SSOB = saline sodic overburden, SCCT = sand capped consolidated tailings, OB = overburden, TS = tailings sand (information on substrate types (SSOB, SCCT, OB, TS) was provided by C. Wytrykush, Syncrude Canada Ltd., pers. commun.). NR: no record.

Wetland Name	pH	Calcium (ppm)	Potassium (ppm)	Magnesium (ppm)	Sodium (ppm)	Phosphorous (ppm)	Chloride (ppm)	Sulphate (ppm)	Depth to Water Table (cm)	Soil Placement Material
63 N 001	8.30	36.9	3.488	11.846	54.805	0.015	53.724	9.586	8.85	Natural
63 N 002	8.93	14.015	3.136	9.353	220.483	0.019	124.466	8.806	23.11	Natural
63 N 003	8.11	44.763	3.689	13.109	126.065	0.027	173.492	12.218	18.8	Natural
63 N 004	8.53	55.293	2.922	20.351	113.773	NR	100.045	58.46	6.94	Natural
63 S 001	8.09	22.612	1.795	20.532	32.16	0.019	24.800	NR	3.12	Natural
63 S 002	7.30	58.544	2.589	17.431	52.167	0.019	46.897	NR	-37.96	Natural
Aostra Pit 1	8.46	25.498	5.627	10.106	74.898	NR	4.902	26.815	40	Natural
Aostra Pit 4	7.96	40.108	3.214	20.43	45.246	0.021	13.468	15.336	15.67	Natural
Borrow Pit 1	7.94	34.67	3.762	23.622	23.394	NR	18.053	1.931	-6.14	Natural
Borrow Pit 3	8.55	51.702	4.095	19.254	117.023	NR	153.983	21.504	26.09	Natural
Erics Pond	7.63	74.406	4.999	46.798	85.416	0.024	11.655	94.465	7.1	SSOB
EIP 1	7.92	80.567	17.56	39.597	847.178	0.057	524.135	265.346	-1.26	SCCT
EIP 2	8.25	79.516	10.796	52.885	607.099	0.031	642.978	234.296	10.7	SCCT
EIP 3 A	NR	NR	NR	NR	NR	NR	NR	NR	40	SCCT
EIP 3 B	7.39	363.171	22.651	115.014	914.341	0.037	446.683	813.507	39.76	SCCT
EIP 3 E	7.72	68.525	4.191	46.154	850.072	0.179	831.454	106.782	40	SCCT
EIP 4	7.61	151.651	8.853	86.213	1186.363	0.06	1470.313	821.063	36.78	SCCT
East Pond	7.54	80.183	4.14	14.221	21.773	0.241	5.899	6.14	-8.61	Natural
Gravel Pit	8.05	34.123	1.446	26.041	15.333	NR	16.667	3.071	-7.36	Natural
Hanging Stone A	7.73	80.872	6.511	32.674	122.873	NR	221.868	33.531	1.74	Natural
Hanging Stone B	8.17	64.081	6.61	34.2	128.84	0.027	234.558	35.812	-21.08	Natural

Wetland Name	pH	Calcium (ppm)	Potassium (ppm)	Magnesium (ppm)	Sodium (ppm)	Phosphorous (ppm)	Chloride (ppm)	Sulphate (ppm)	Depth to Water Table (cm)	Soil Placement Material
Muskeg River Wetland	7.49	51.069	1.575	189.389	20.294	0.013	20.689	150.202	-12.05	Natural
North Base Mine	8.17	119.848	10.837	59.416	288.769	0.039	215.559	196.037	1.29	OB
Parsons Creek	8.45	25.259	3.121	9.627	24.395	0.031	2.579	11.271	33.08	Natural
Ramp Wetland	7.44	68.114	5.891	31.923	128.468	0.03	177.890	53.652	-12.4	Natural
Small Borrow Wetland 3	7.15	30.572	0.499	8.22	7.706	0.092	2.586	NR	11.09	Natural
Sandhill Fen	6.98	157.759	8.631	49.999	348.451	0.038	240.135	185.585	-8.29	SCCT
South Pond	7.52	90.354	6.475	40.109	194.371	0.019	194.381	99.821	-11.25	Natural
Shallow wetland	7.59	51.259	6.459	23.301	101.323	0.031	27.429	NR	-2.75	OB
Southern Wetland	7.23	63.786	4.333	24.255	74.354	0.031	106.655	36.433	-8.52	Natural
SWSS 02	7.82	51.432	5.866	35.734	387.34	0.028	260.960	49.353	18	TS
SWSS 03	7.80	123.036	9.66	62	369.86	0.087	183.126	245.669	12.6	TS
SWSS 04	8.04	66.437	11.068	34.169	545.876	0.024	301.399	173.185	-15.09	TS
SWSS 05	8.21	63.66	11.493	33.577	559.731	0.02	297.224	168.492	-8.38	TS
SWSS 08	8.39	75.077	4.977	59.121	171.913	0.071	108.174	95.171	40	TS
SWSS 14	7.55	46.333	3.089	24.224	63.21	0.13	18.143	19.238	37.5	TS
SWSS 15	7.72	44.324	2.09	42.724	115.799	0.096	17.790	75.22	36.5	TS
W1 45	7.57	83.824	7.714	42.987	87.052	0.037	11.200	77.081	-8.59	SSOB
W1 46	8.20	62.77	14.364	27.8	60.385	0.668	15.596	68.649	5.46	SSOB
W1 47	7.31	29.167	8.462	30.405	112.397	0.028	15.788	24.969	1.3	SSOB

Table S3. Multivariate permutation analysis of variance (PERMANOVA) of community composition (vegetative species abundance) in response to soil placement materials (natural, saline sodic overburden, overburden, saline sodic consolidated tailings, tailings sand). Significant differences are bolded.

Source	df	SS	MS	Pseudo-F	P (Perm)	*P(MC)*
Soil Placement Material	4	19612	4903	1.93	**0.001**	**0.001**
Residuals	35	88909	2540.2			
Total	39	10852				