Marsh reclamation in the oil sands of Alberta: providing benchmarks and models of vegetation development

by

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Abstract

A key objective of the Alberta oil sands industry is to reclaim the post-mined landscape to “equivalent land capability” (Harris 2007). Vitt and Bhatti (2012) proposed a restoration framework for boreal disturbances. They suggested that to increase chances of achieving ecosystem equivalency and sustainability, created sites must have 1) species composition similar to natural reference sites, 2) species performance based on natural benchmarks, and 3) ecological processes similar to reference sites. To provide reclamation benchmarks to which created marshes of the Fort McMurray region can be compared and reclamation practices adjusted, my work follows the rationale developed by Vitt and Bhatti (2012). In Chapter One, I provided an introduction to the major paradigms of community ecology. In Chapter Two, I identified, described and compared environmental and plant assemblage patterns present in different types of created and natural marshes. In Chapter Three, I examined the degree to which the addition of peat-mineral mix (PM) to different types of oil sands process materials (OSPM) affects *C. aquatilis* performance. I also tested the effects of oil sands process water (OSPW) on *C. aquatilis* performance. In Chapter Four, I defined and compared natural and created marsh zone area variation over time and identified abiotic factors that influence the patterns observed.

My results revealed that created and natural marshes were characterised by distinct environmental conditions and that the vegetation composition of some created sites was dissimilar to natural reference sites. The addition of PM to OSPM significantly increases *C. aquatilis* survival, below and aboveground biomass. The use of OSPW significantly reduced *C. aquatilis* belowground biomass and affected its physiological performance. Amending created marshes with PM may enhance plant performance but its effect at the community level remains to be tested. Unlike natural marshes the total areas of created marshes were dominated by stable submersed aquatic vegetation zone (SAVZ) in all years. Mean maximum temperature and annual total snow were identified as the simplest ways to predict SAVZ area within natural marshes for a given year. The ratio of marsh area to volume described SAVZ area variation and provided prescriptive guidance for construction of reclaimed marshes.

À ma famille.
Pour mon père qui a su me transmettre entre deux billots d'épinette, avec peu de mots, un peu de sa sagesse mais surtout son amour pour le bois. Pour ma mère qui, en s'assoyant dans l'escalier par les soirs d'été pour écouter la grive solitaire chanter, m'a transmise son adoration pour la nature. Pour mes deux grands frères, pour m'avoir « tapper la trail » et ébranché les sentiers à coups d'hachette et de bonheurs naïfs. Pour mes deux petites nièces et leurs petites joues roses, pour me donner une raison de croire en un avenir meilleur.

Aux épinettes de Saint-Alfred qui m'ont vu grandir et m'ont offertes le hurlement des coyotes, le chant du merle, les chevreuils sous le pommier et les écureuils dans le grenier.

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To Piaf, for never leaving my side, for your gentle soul full of joys whom reminds me that life is beautiful and simple.

To Tamara, for believing in me and to always make the best adventure out of everything.

Preface

My thesis bridges the practice of ecological restoration with theoretical paradigms developed in community ecology. Thus, in Chapter One, I provide an introduction and synthesis of the major paradigms of community ecology including past and current accomplishments in this field. Using the oil sands of Alberta as a model, I identified how reclamation can provide a fruitful setting to test theories and hypotheses of community ecology. I also outlined my chapter objectives and identified how they contribute to the field of community ecology.

To provide reclamation benchmarks to which created marshes of the Fort McMurray region can be compared and reclamation practices adjusted, my work followed the rationale developed by Vitt and Bhatti (2012):

1) Created sites must have species composition similar to natural references sites (Vitt and Bhatti 2012).

In Chapter Two, I examined whether created marshes were characterised by environmental conditions and plant species composition that were similar to their natural analogues. I identifed, described and compared environmental and plant assemblage patterns present in different types of created (i.e. created-tailings, created-unamended, and created-peat-amended) and natural marshes. Environmental conditions (water and soil chemistry, and physical conditions), as well as plant community measures (richness, diversity, composition) were collected in 51 marshes of the Fort McMurray region. Univariate and multivariate analyses were used to compare them, and differences in environmental and vegetation composition among marsh types were found. My results suggested that oil sands amendments (oil sands process water (OSPW) and oil sands process material (OSPM)) influence vegetation composition. Based on my results, the next logical step was to determine if and how OSPW and OSPM influence marsh vegetation performance.

2) Species performance in created sites must be based on natural benchmarks (Vitt and Bhatti 2012).

A sedge community dominated by *Carex aquatilis* was identified by Raab and Bayley (2013) as a desirable late-succession community for the wet-meadow zone of oil sands-created marshes. However, when compared to *C. aquatilis* growing in natural marshes, *C.aquatilis* performance was inhibited in oil sands created marshes (Mollard et al. 2013). In Chapter Three, in a two-factor experiment, I examined the degree to which the addition of peat-mineral mix (PM) to different types of OSPM (Consolidated-tailings (CT),, Tailings-sand (TS)) and OSPW affect *C. aquatilis* performance, I also tested the effects of oil sands process water (OSPW) on *C. aquatilis*. I assessed survival, below- and aboveground biomass, and physiological responses (chlorophyll *a* fluorescence). My results revealed that the addition of PM to OSPM significantly increases *C. aquatilis* survival as well as altering below and aboveground biomass. The use of OSPW significantly reduced *C. aquatilis* belowground biomass and affected its physiological performance.

3) The development of created site function must be similar to reference sites (Vitt and Bhatti 2012).

In Chapter Four, I quantified the variation of the zone area of natural marshes over time to provide a range of acceptable benchmarks that can guide reclamation. Reclaiming wetland hydrological processes may be the first of the factors essential to reclaiming wetland structure and functions. The aim of my study was thus to define and compare natural and created marsh zone area and vegetation biomass variation over time and to identify abiotic factors that influence the patterns observed. I used a series of historical aerial photos and ground-based methods to quantify zone area and vegetation aboveground biomass variation over time. I demonstrated that unlike natural marshes, created marsh SAVZ occupied a greater proportion of the total marsh area than the VZ. Both the SAVZ and VZ surfaces of created marshes were more stable over time than those of natural marshes. Mean maximum temperature and annual snow were identified as the simplest ways to predict SAVZ area within natural marshes for a given year. The ratio of marsh area to volume described SAVZ area variation and provided prescriptive guidance for construction of reclaimed marshes.

In Chapter Five, I reviewed my general objectives, the main questions investigated, and the conclusions reached by my work.

I would like to acknowledge the co-authors who dedicated their time and thoughts to the development of the manuscript that appears as Chapter Three in this dissertation. The manuscript(Chapter three) has been modified to fit within the dissertation as a whole.

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Glossary of Terms and Definitions

Additive or removal perturbation method: Used to test for competition among two (or more) species. It compares the responses of two (or more) species before and after the removal or addition of disturbances (Wilson and Keddy 1986).

Allogenic succession: Community change through time as brought about by the environment (Morin 2011).

Autogenic succession: Community change through time as brought about by the biota (Morin 2011).

Assembly rules: Ecological restrictions on the observed patterns of species presence or abundance that are based on the presence or abundance of one or more other species or group of species (not simply the response of species to environment) (Weiher and Keddy 2001).

Alternative Stable State: Apparent differences in the composition of communities in otherwise comparable environment. The alternate stable states should correspond to different possible outcomes of species interactions rather than the result of different environmental conditions (Lewontin 1969).

Chronosequence: Method in ecology used to study a process over time (such as vegetation development) where time is substituted for space (Johnson and Miyanishi 2008).

Community assembly: The process by which species from a regional pool colonise and interact to form local communities (HilleRisLambers et al. 2012).

Community Ecology: The study of patterns and processes involving at least two species at a particular location (Morin 2011).

Ecology: Coined in 1869 by Enrst Haeckle, it is defined as the science that studies the specific interactions among organisms and their living and non-living environment (Begon et al. 2008).

Ecological succession: The process of temporal change in community composition (Morin 2011).

Ecological restoration: An intentional activity that initiates or accelerates the recovery of an ecosystem with respect to its health, integrity and sustainability (SER 2004).

Exploitative competition: Occurs between two organisms of the same or different species whenever a valuable and limited resource, such as space, food or nutrients and light, is shared between them (Harper 1977).

Facilitation: The positive effect of plants on the establishment or growth of other plants (Homlgren et al. 1997).

Guild: Group of functionally similar species in a community (Mitsch and Gosselink 2011).

Inhibition: Affecting the rate of succession through competitive interaction between species or through allelochemicals that may inhibit the growth of another species (Morin 2011).

Interference competition: Involves active denial of access to a resource by one competitor to the other (Begon et al. 1986).

Interspecific Competition: Any mutually negative interaction between two or more species that does not involve mutual predation (Morin 2011).

Life history traits: Factors such as the size, rate of growth, seed size, and dispersal ability of an organism (plant) (Loreau 2010).

Marsh: A wetland dominated by emergent, herbaceous vascular plants, where the vegetation is primarily non-woody (Mitsch and Gosselink 2011).

Mixed planting method: Used to test for competition among two (or more) species. This method is used to compare the response (growth rate, height etc.) of two species grown individually (in monoculture) to treatments where they are grown together (McCreary 1991).

Non-equilibrium state: Communities with fluctuating species composition and extensive variation in population dynamics (Morin 2011).

Population: A set of organisms from the same biological species in a given area (Loreau 2010).

Primary Succession: Community development on newly exposed (thus sterile and inorganic substrates) lands, typically generated by volcanism or glaciation (Morin 2011).

Pulse Stability: Ecosystem of community in dynamic equilibrium with abiotic forces (Mitsch and Gosselink 2011).

Secondary succession: Occurs after disturbance disrupts established communities without completely eliminating all life (Morin 2011).

Replacement series method: An experimental design used to test for competition amongst two (or more) species in which each species is grown in varying proportions, while maintaining a constant overall stand density (Moen and Cohen 1989).

Restoration ecology: The scientific discipline of developing and/or applying theory to guide restoration activities (Palmer et al. 2006).

Stable equilibrium state: Communities with a relatively constant species composition (Morin 2011).

Wetland: A shallow-water ecosystem characterised by hydric soils and where hydrophytes grow. Wetlands include marshes, bogs, fens, vernal pools and seagrass beds (Zedler 2000; Mitsch and Gosselink 2011).

Wetland functions: Biological processes such as productivity, biodiversity support, nutrient cycling and floodwater storage (Zedler 2000).

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List of Abbreviations

CP: constructed-peat marsh
CT: constructed-tailing marsh or consolidated-tailings
CU: constructed-unamended marsh
DO: dissolved oxygen concentration
EC: electrical conductivity
EZ: emergent zone
LOI: loss on ignition
MFT: Mature fine tailings
N: natural
ORP: oxidation-reduction potential
OSPM: oil sands process material
OSPW: oil sands process water
PM: peat-mineral mix
PPT: parts per thousand
SAVZ: submersed aquatic vegetation zone
TN: total nitrogen
TP: total phosphorus
TS: tailings-sand
VZ: vegetated zone (WMZ and EZ combined)
WMZ: wet-meadow zone

CHAPTER 1

Title: Linking community ecology and reclamation in the oil sands of Alberta: an introduction.

1.1. Introduction

Here, I provide a contextualizing introduction and synthesis of the major paradigms of community ecology to review past and current accomplishments of this field of study. Using the oil sands of Alberta, I identify how reclamation can provide a fruitful setting in which to test theories and hypotheses of community ecology. I also outline my chapter objectives. Though this is a general overview of community ecology and reclamation, specific examples taken from the field of aquatic plant community ecology are related to different approaches, concepts and applications of the theories.

How do communities form? This simple question, debated over hundreds of years, remains incompletely answered (Loreau 2010). Ecologists have identified a wide variety of processes operating at diverse spatiotemporal scales but the details influencing community assembly still fuel debates and generate countless questions (HilleRisLambers et al. 2012). Stochasticity, historic, abiotic and biotic processes are all known to influence the assembly of communities (Ricklefs 2004; Gotzenberger et al. 2012; HilleRisLambers et al. 2012). These processes act as “filters” on a regional pool of species, thereby constraining its full expression at a local scale. A regional pool of species contains numerous potential colonisers to a local site. Due to chance and dispersal limitations, only a subset can even reach the site to colonise. The subset, once on site, is further constrained by both the local environment and the complex interactions (positive or negative; intra or interspecific) among species. A local community therefore reflects the cumulative effects of stochasticity and a variety of abiotic, biotic and feedback processes (Gotzenberger et al. 2012; HilleRisLambers et al. 2012).

Having progressed from a descriptive discipline, community ecology now seeks to understand the mechanisms behind observed patterns (Bramwell 1989; Hagen

1989). Striving for generalities and laws, community ecologists rely on experiments to test hypotheses and further elaborate theories (Simberloff 2004). Ecological restoration, which relies heavily on the paradigms developed in community ecology (such as succession theory) provides opportunities for elaborate experiments which may support or challenge theories about community assembly (Zedler 2000). In my thesis, I explore the science of community ecology and its relationship with the discipline of restoration. In this introductory chapter I summarise the science of community ecology and ecological restoration. I clarify descriptive and mechanistic approaches to community ecology and position my research. I compare the paradigms and guiding philosophies of ecology, particularly succession theories that have shaped our understanding of community assembly. Finally, I demonstrate how community ecology relates to reclamation of the oil sands of Alberta by using my thesis research.

1.2. Community ecology

The term community[1] ecology was first coined by Karl Möbius in 1877, but it is only recently that scientists have agreed on the specific objectives that this science aims to achieve (Golley 1993). For most of the 19th and 20th centuries, community ecology was a name loosely attributed to any studies that focused on more than one organism at a scale smaller than the landscape (Weiher and Keddy 2001). Lewontin (1974) criticised community ecology for its lack of apparent progress and described the discipline as being in a state of “agony” resulting from its poorly defined aims and its lack of clear boundaries. Since then, numerous authors have attempted to define community ecology. Hence, while Weiher and Keddy (2001) define the science by the central question it aims to answer: “how one [species] gets from the pool of species to the community?”, Morin (2011) describes community ecology as “the study of patterns

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[1] The term *community* has different meanings to different ecologists. Despite the debate, most definitions include the idea of a collection of species found in a given location (Morin 2011). The debates have been over whether or not communities can be described as definable units that are distinct, discrete and defined by interactions (Palmer et al. 1996; Morin 2011). For the purpose of my thesis, and without digressing into this debate, the term community is used within the restricted taxonomic context of *plant community* and is defined as an assemblage of plant species found in a particular place that forms a functional unit, whose members interact (Putman 1994).

and processes involving at least two species at a particular location”. In addition to being criticised for its lack of clear focus, community ecology has been described as a “weak” and “soft” science because it often fails to provide general laws (Peter 1991; Lawton 1999). Simberloff (2004), in response to Lawton (1999) insists that community ecology cannot be judged by the traditional nomothetic approach that defines natural sciences such as physics. Simberloff (2004) agreed that generalisations are rare in community ecology, but he argued that although research is often local, experimental, and reductionist, it is crucial in understanding many environmental problems. Hence, community ecology strives to identify and understand mechanisms and processes influencing and explaining the presence of species in complex and idiosyncratic communities. Although general laws are rare, it is the application of community ecology’s framework to specific, and often local, cases that gives this science its strength and utility.

1.2.1. Population, community and ecosystem ecology

Community ecology is an integrative discipline nestled among and connected to other branches of ecology such as evolutionary ecology, population and ecosystem ecology (HilleRisLambers et al. 2012). The distinction between these disciplines of ecology is arbitrary and boundaries are set by ecologists to study operations of specific levels of ecological organisation (Palmer and White 1994; Palmer et al. 1996). Contrary to community ecology, which is interested in diversity, dynamic, and interspecific interactions of the biological components of ecosystems, population ecology adopts a more demographic study approach and instead focuses on interactions among individuals of a single species and their relationship with their environment (Loreau 2010; Morin 2011). Despite this inherent difference, population ecology provided the conceptual and methodological foundations for community ecology and this is said to be its ancestor (McIntosh 1985; Sheail 1987; Loreau 2010). For example, the *competition theory* describing interspecific competition among individuals of different species is an extension of the *density dependence theory* specific to intraspecific competition among individuals of one species as elaborated in population ecology (Loreau 2010). On the other hand, ecosystem ecology addresses the functioning of entire ecosystems using flux measurement of material or energy between the different compartments of ecosystems (Morin 2011). In addition, to be compatible with population and ecosystem ecology, community ecology also relies on evolutionary ecology. Hence evolutionary ecology provides to community ecology the historical context that has influenced the presence and composition of a pool of species from which a community will eventually emerge (Weiher and Keddy, 2001). These different branches of ecology address issues at different hierarchal levels of organisation and hence at different spatial and temporal scales. The mutual enrichment arising from the association of these fields is, however, proof that they cannot be dissociated from one another (Loreau 2010).

The combined benefit of improved understanding of ecology that arises from the association of different fields is accentuated by their common approach based on the scientific method (induction, deduction and hypothetico-deductive methods). Hence, these disciplines provide opportunities to develop ecological theories that transcend their own boundaries and that might lead to the establishment of the general laws so desired and valued in science (Lawton 1999; Temperton et al. 2004). It is indeed when tested, through a hypothetico-deductive approach, that research hypotheses can become laws or generalities (Romesburg 1981). The science of restoration ecology provides an especially suitable situation in which to use the scientific method to develop theories and test hypotheses that may emerge from community ecology (Palmer et al. 1996; Temperton et al. 2004; Holzel et al. 2012).

1.2.2. The role of community ecology in ecological restoration

While the practice of ecological restoration benefits from ecological theory, community ecology can use restoration as a opportunity to gain insights into how communities assemble and function (Palmer et al. 1996; Grainer and van Aarde 2012 When the aim is to reestablish lost or degraded multi-species assemblages, knowledge derived from community ecology is required (Palmer et al. 1996; Templeton et al. 2004; Young et al. 2001). Restoration ecology, to initiate or accelerate community development, relies on assembly and succession theories that are central paradigms of community ecology (Palmer et al. 1996; Young et al. 2001; Temperton et al. 2004; Young et al. 2005; Hobbs et al. 2007; Vitt and Bhatti 2012). Determining how species within a community arrive in an area, establish, survive and interact with one another is at the heart of both community and restoration ecology (Temperton et al. 2004; Young et al. 2005). Using the assembly theory as a framework to understand membership in a community, ecologists identify constraints that act on a species pool (Drake 1991; Wilson 1995; Wilson & Whittaker 1995; Weiher and Keddy, 2001), and understanding these “filters” has important implications for successful restoration. The predictive and deterministic models proposed by succession theory are often used in restoration to evaluate the development and an endpoint (and therefore the success) of restoration efforts. Consequently, community ecology paradigms are extremely relevant to restoration ecology and restoration ecology provides the laboratory to test community ecology theories (Bradshaw 1983). The post-mined landscapes existing in the oil sands region of Alberta, Canada, offer valuable opportunities to test important hypotheses in unique ways.

1.3. The oil sands of Alberta

The province of Alberta is known for its oil sands deposits and the subsequent extraction activities. Since 1967, Alberta’s oil sands development activity has been altering the boreal landscape and companies are required to address its reclamation (Alberta Government 2012). The Oil Sands Administrative Area (OSAA) covers 142,200 km^2 of north and eastern Alberta (Alberta Government, 2013) (Figure 1). Depending on the depth of the bitumen deposit in the ground, oil sands industries are using two types of technique: *in situ* or *surface mining.* The in situ technique extracts bitumen deposits that are more than 75m deep by a combination of drilling wells and injecting heated fluids to help liquify petroleum, while surface mining requires the complete removal of the ground top layers to access the subsurface bitumen deposits (Alberta Environment 2013). Over the past 60 years, surface mining has had a large impact on the regional landscape; mining activities have created a post-mined landscape of bare mineral ground dotted with tailings ponds and 100-meter deep holes (Grant et al. 2008). It is estimated that approximately 3,000 km^2 of oil sands post-mined landscape will need to be reclaimed (Rooney et al. 2012). The largest land category (up to 60% by area) of the mined area was fen wetlands (Foote 2012).

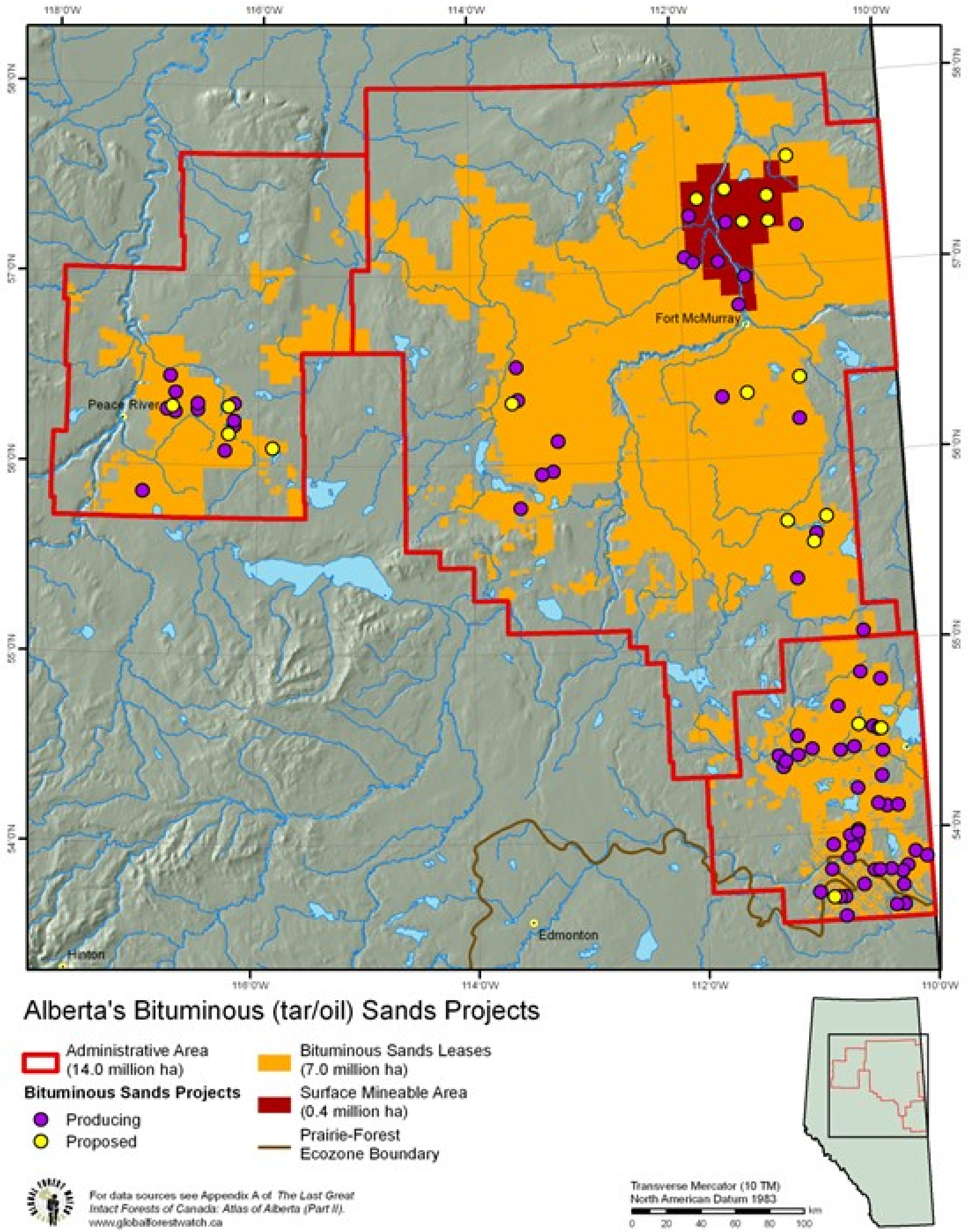

Figure 1- 1: Maps of the oil sands projects in Alberta (figure adapted from Global Forest Watch Canada, 2009).

1.3.1. The Alberta oil sands reclamation requirements and associated challenges

The oil sands industry of Alberta has the objective of reclaiming the post-mined landscape to "equivalent land capability" (Harris 2007). "Successful" reclamation is further defined as "the reclaimed soils and landforms are capable of supporting a self-sustaining, locally common boreal forest, regardless of the end land use" (Government of Alberta, 2007a, 2007b; Rooney 2011). Hence, the oil sands industries must reclaim rather than restore the post-mined landscape[2]. In addition to these reclamation objectives, Vitt and Bhatti (2012), proposed a theoretical framework for the reclamation of the post-mined landscape that translates into four main points:

> 1) Site development utilising natural analogues; 2) species selected from comparable natural settings; 3) species performances based on clear natural benchmarks; and 4) development of community stabilisation, species richness, and ecosystem function, again based on natural analogues. (p.9)

Wetland reclamation in the Fort McMurray region faces numerous challenges. Of the post-mined landscape, current reclamation guidelines for the oil sands propose that an area of approximately 660 km^2 will be returned to wetlands (Harris 2007). Reclaiming wetlands is challenging due to the physical and chemical characteristic of the post-mined landscape. The majority of the bitumen deposits are found in the McMurray Formation (Conly et al. 2002). This layer was formed during the Cretaceous period by river and ocean processes (Conly et al. 2002). By using surface mining, and removing the top ground layer, the oil sands industry uncovers the McMurray Formation. This newly exposed layer of shale, sandstone provides difficult conditions for effective reclamation due to its high salinity and limited amount of organic matter and lack of seed bank and propagule (Harris 2007).

To extract bitumen from the subterranean sands, Suncor Energy and Syncrude Canada use hot water processes referred to as the Clark Hot Water Extraction Process (Clark 1939; BGC 2010). To separate bitumen from sand, water is used. Sodium

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[2] The objective of restoration is usually to return a degraded site to specific conditions (i.e. biological, chemical, and physical) very similar to pre-disturbance conditions. The goal of reclamation is usually to give back to the disturbed land the ability to support various desirable land uses that may or may not be similar to ones that existed pre-disturbances. The reclaimed land uses in the post-mined landscape is not required to be identical to pre-disturbance uses (Government of Alberta, 1993).

hydroxide (NAOH) is occasionally added to water to improve bitumen separation (Allen 2008). The by-products of bitumen extraction are called tailings, which are stored in tailings ponds (settling basin). Tailings are slurries of process-water, sand, silt and clay, as well as soluble compounds such as salts, naphthenic acids (NAs) as well as polycyclic aromatic hydrocarbons (PAHs) (Schramm et al. 2000; Fedorak et al. 2003, BGC 2010). Tailings constituents are divided into two broad categories representing 1) the solid phase resulting from the sedimentation and consolidation of fine tailings in settling basins and 2) the liquid phase resulting from the released waters from tailings sediments during consolidation. The solid phase of tailings is referred as Oil Sands Process Material (OSPM; or more recently "fine fluid tails-FFT) and is often composed of Mature Fine Tailings (MFT) or Consolidated Tailings (CT) (see Fedorak et al. and Syncrude 2011 for more details). The liquid phase of tailings is Oil Sands Process Water (OSPW) (Syncrude 2011). This method of extraction produces approximately 262,000m^3 of tailings per day, and its storage in settling basins and management constitutes one of the main environmental challenges facing the oil sands industry (Alberta Environment 2011). In 2011, the oil sands region was estimated to host 170 km^2 of tailings ponds containing approximately 840 million m^3 of fine tailings (ERCB 2011). Due to the no-net-water discharge policy, the water used to extract bitumen is recycled numerous times before being stored in tailings ponds. By recycling water, the oil sands industry increases the salinity content of the OSPW. In addition to salts, the OSPW and OSPM contain NAs and PAHs that are toxic to aquatic and terrestrial species. NAsoccur naturally in a variety of petroleums and are assumed to be a by-product of aerobic microbial degradation of petroleum hydrocarbons (Brient et al. 1995). NAs are complex mixtures of predominately alkyl-substituted cycloaliphatic carboxylic acids and small amounts of acyclic acids (Brient et al. 1995) that have been identified as acutely toxic to aquatic species (Headley and McMartin. 2004; Scott et al. 2005). The PAHs result from the presence of unrecovered bitumen in tailings. Numerous studies have demonstrated that PAHs have toxic effect on fish and birds for example (Albers, 2003).Vast reclamation areas are connected directly or indirectly to those tailings ponds and the constituents that they contain.

Prior to industrial activity, the oil sands mineable area was rich in wetlands. The area occupied by wetlands was estimated to cover approximately 64% of the landscape (NRC 2002; Rooney et al. 2012). Of this area, 62% supported peatland vegetation, while the 2% remaining area was covered by marsh vegetation. Despite the large area of peatland lost, the oil sands industries have mainly focused their efforts on the restoration of marshes (NRC 2002; Rooney et al. 2012). The harsh environment found in the post-mined landscape and the sensitivity of peatland vegetation to these conditions have delayed peatland restoration attempts and favored the creation of marshes (Rooney et al. 2012).Furthermore, because peatlands only accumulate several mm of peat per year and to qualify as an Alberta peatland, sites must support a minimum of 40 cm of peat, the accumulation time component makes peatlands intractable targets for near-term reclamation.

To understand the potential effects of tailings constituents and different amendments on biota, including hydrophytic macrophytes, the oil sands companies have constructed pilot marshes in the mined landscape. Some of these marshes are referred to as created[3]-tailings (CT) and are directly or indirectly (runoff or seepage from tailings ponds) amended with MFT or CT as substrate and/or filled with OSPW. Other pilot marshes are referred as created-peat (CP) and created-unamended (CU). CU marshes haven't received any type of amendment while CP marshes are capped with a layer of peat-mineral mix (PM). As is the case for CU marshes, CP marshes are not affected directly or indirectly by tailings constituents. In Chapter Two, I described the specifics of each marsh types.

The large spatial scale and the environmental conditions of the post-mined landscape therefore offer unique research opportunities for both community ecologists and reclamation science. Given the challenge of reclaiming ecosystems at the landscape level, this vast area allows large-scale experimentation and hypothesis testing at different scales and levels. The complete removal of the top layers of ground

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[3] Marshes refered to as "created" can be either constructed or opportnunistic. Opportunistic marshes are unplanned landscape features, occurring in depressions or in dynamic drainage areas resulting from human activities (see Chapter Two for more details). Unlike natural marshes, the origin of created marshes is the result of direct or indirect anthropogenic actions and/or interventions. The term created was preferred over the term "reclaimed" or "restored" to avoid the assumption that these marshes have attained their full ecological maturity or targeted conditions.

to access the bitumen and the disturbances of the surrounding landscape (e.g. roads, cutlines, etc.) alter colonisation dynamics and in some instances, deprives the landscape of its seeds and other propagules. These conditions provide rare opportunities to test and study early ecosystem development. Where wetlands are filled with heat-sterilised substrates, near-primary succession conditions are available for experimentation over large areas.

1.4. Community ecology: describing patterns

1.4.1. Do patterns of species co-occurrence exist?

Recognising patterns is essential to science and is the stepping stone to the scientific method (Romesburg 1981). It was Alexander von Humboldt, a plant geographer, who was the first to recognise that plants tend to occur in repeatable assemblages (i.e. communities) (McIntosh 1985; Golley 1993). Through induction he later associated the observed patterns with environmental factors (Golley 1993). Important theories, such as natural selection by Wallace and Darwin, were first initiated by recognising patterns. In his famous manuscript titled "The Geographical Distribution of Animals", Wallace (1876) observed that certain islands of the Indian islands are "characterised by peculiar fauna", distinct from the ones generally observed. McArthur and Wilson (1967) developed the theory of island biogeography based on the uneven distributions of birds observed on islands of varying sizes and degrees of isolation. Hence, as mentioned by McArthur (1972), "to do science is to search for patterns", and describing and recognising assemblages is fundamental to the development of theory in science.

1.4.2. Defining patterns of wetland plant communities in the oil sands landscape

In Chapter Two I identified and compared the occurrence of environmental (i.e. chemical and physical) and vegetation patterns in natural and created marshes of the Fort McMurray region. Created marshes investigated had been amended with different substrates (i.e. OSPM (either CT or MFT), no amendment and PM), and capped with different types of water (OSPW and natural surface water). The origin (i.e. created or natural) and the types (i.e. sediment and water amendments) of marshes have been observed to result in different environmental conditions which in turn lead to different plant community assemblages.

Contemporary with the work of Trites and Bayley (2009), Rooney and Bayley (2011), Raab and Bayley (2012), and Slama (2010), Chapter Two was fundamental to the development of my work because it constituted the building block to my thesis by identifying topics of uncertainty, by identifying reclamation challenges, and by generating a series of hypotheses and research questions. Furthermore, Chapter Two provides reclamationists with a comparative model of species composition from natural conditions in the region.

1.4.3. From describing patterns to identifying processes

Community ecology has evolved over the last centuries from a descriptive discipline to a science that aims to explain the observed patterns (Shipley 2011). Understanding the mechanisms and processes behind observed patterns has become increasingly important and is presently the main focus of community ecology (Weiher and Keddy, 2001; Shipley 2011). Scientists recognise and agree that patterns arise at the community level and that those patterns result from processes and mechanisms that, if identified, can be used to predict which species will emerge locally from a given regional species pool (Weiher and Keddy 2001; Shipley 2011). Thus the central aim of community ecology is to determine “how species get from the pool to the community” (Weiher and Keddy 2001) or in other words, to determine “why patterns occur?” (Weiher and Keddy 2001).

1. 5. Community ecology: identifying processes

In her 1896 book entitled "Oecology of Plants: an Introduction to the Study of Plant Communities", Warming established the central questions that still define the focus and aims of plant community ecology (McIntosh 1985). With questions such as “How do species congregate to form characteristic communities?” community ecology shifted its focus away from simply documenting species associations, and began to focus on the causes of the observed patterns.

During this same period, scientists began to describe communities as dynamic entities. In 1899, with his studies on sand dunes of Lake Michigan, Cowles described the temporal change in community composition over time and set the groundwork for the theory of succession. Debates to determine if species associations were the result of deterministic or stochastic processes fed the conceptual development of succession throughout the 20th and 21st centuries.

1.6. Communities are dynamic entities: influential succession theories

Influenced by the work of Cowles, Clements (1916) developed a deterministic model of succession. He described the process of succession as a series of predictable seres[4] that culminate in a single and stable (at equilibrium) pre-defined endpoint or "climax" (Table A.1 and A.2). Clements identified many processes, such as initial site conditions and species interactions (primarily facilitation), that are still recognised today as important factors influencing community assembly (Pickett et al. 2009). This deterministic view of succession, however, was fiercely criticised by Gleason (1917) who proposed a stochastic approach to explain community development that pivoted on the responses of individual plants according to their life histories. According to Gleason (1917), plant communities are coincidental assemblages resulting from the migration and the physiological ability of individual species to exploit a given environment (Pickett et al. 2009). The two views have fed numerous debates and inspired numerous studies and theories (Pickett et al. 2009).

Adopting the concept of Clements, Egler (1954) viewed succession as the successive replacement of one group by another, each representing a distinct "stage". In his view, one stage prepares the habitat for the introduction of the next group by enhancing, for example, the soil properties (nutrient additions etc.). Later, Connell and Slatyer (1977) termed this process facilitation. Also inspired by Clements' view, Odum (1969) proposed that the attributes of a community (i.e. biomass, nutrients, species

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[4] Seres or successional stages

richness and evenness, life cycles, community stability etc.) vary over time in a predictable manner. Although criticised for its tautological argumentation (Patten 1993), Odum (1969) explained the predictable development of the community through modifications of the environment that were driven by the developing community itself.

In contrast to the single climax state proposed by Clements, a multiple-equilibrium perspective in ecology emerged during the 1970s. Levin (1974; 1976) argued that spatial patchiness and heterogeneity may support alternative and locally stable communities. Drury and Nisbet (1973) also challenged Clements' and Egler's views by suggesting that early colonisers do not predictably influence and facilitate later immigrant establishment and survival. Rather, they agree with Gleason's concept and proposed that communities result from the differential abilities of plant species to colonise and survive along environmental gradients. According to the different life-history characteristics of plants (such as stress-tolerance), species have different competitive abilities and are specialised to exploit different sets of conditions along these environmental gradients. Other studies, including those of Walker and Chaplin (1987) and Pickett and McDonell (1989), describe the importance of various processes that shape community development including seed dispersal, availability of propagules on site, stochastic events, and disturbances to name few. These studies also recognised that vegetation development may not always result in a predictable climax as defined by Clements. Hence, the idiosyncratic nature of community composition is hypothesised to result from a hierarchy of interacting factors unique to each system under study.

As this review indicates, the succession models that have been proposed through time have been numerous and each has contributed differently to our understanding of community development (Palmer et al. 1996; Young et al. 2001; Temperton et al. 2004) (see Table 1a, 1b, Appendix-A). Despite the numerous models of succession, the deterministic view remains the most influential, with self-design theory being one example of this influence (Zedler 2000). Under this theory, it is hypothesised that once the physical and chemical conditions that favor desired species are re-established at a given site, species will 'find' suitable habitats and that the community development will culminate in a state that is equivalent to that which existed pre-disturbance (Zedler 2000). Hence, the deterministic model forms in many cases the conceptual basis of restoration and is extensively used to evaluate the vegetation development and endpoint of a created ecosystem (Young et al. 2005).

1.6.1. Ecological restoration heavily relies on the succession theory

The succession theory and its assumptions provide to restorationists a framework to assess the development and success of their work (Palmer et al. 1996; Temperton et al. 2004). The models provided by succession theories are useful because they predict the structure, function, and structure-function relationships at different stages of a developing ecosystem (Walker and del Moral 2008). Despite the wide use of, and reliance upon, succession models of vegetation development, few studies have taken advantage of the restoration setting to directly test the assumptions of the succession theory (Walker and del Moral 2008; Grainer and van Aard 2012). Those who have taken this approach have tested these assumptions in terrestrial ecosystems, including coastal forest (Grainer and van Aard 2012), gravel pits (Rehounkova and Prach 2008) and sand dunes (Sykora et al. 2004), but studies examining succession theory in wetland ecosystems are rare. More recently, Noon (1996) and Mitsch et al. (2012) investigated primary succession and have described soil, water and vegetation changes in restored wetlands over time. Although they do not compare their results to any succession models, their research provides insights on this process in wetlands (see Table A.3). The study conducted by Noon (1996) predominantly supports the initial floristic composition model of Egler (1954), while the community characteristics described by Mitsch et al. (2012) support the succession models of Clements (1916), Gleason (1917) or Odum (1969) (Table A.1 and A.2). Hence, despite the fact that restorationists mainly rely on succession models, it remains unclear which models best describe and predict vegetation development in marshes. Restoration settings therefore provide the laboratory to test succession assumptions and develop our understanding of the structure and functions of ecosystems over primary and secondary succession.

1.6.2. Interacting processes influencing vegetation community succession

There are numerous factors operating at different spatiotemporal scales that may render succession unpredictable (Grainer and van Aard 2012). Variables such as seed dispersal (Kettenring and Galatowitsch 2011) and propagule availability (Matthew and Spyreas 2010), topographic heterogeneity (Cutler 2010), disturbances (Turner et al. 1998), priority effect (Connell and Slatyer 1977), predation (Carlsson et al. 2004) and many more may continuously or sporadically influence vegetation communities over time (HilleRisLambers et al. 2012). Hence, over the last two centuries, ecologists have identified a number of interacting processes that shape community patterns, including: 1) environmentally mediated patterns (abiotic filters); and 2) assembly rules (i.e. patterns due to interaction between species or biotic filters) (Wilson 2001; HilleRisLambers et al. 2012). Each of these two categories of filters includes numerous types of processes and mechanisms that influence community composition.

Community ecologists recognise that vegetation patterns result from the complex interaction of both abiotic and biotic variables. Building on previous process-oriented models, Wisheu and Keddy's (1992) concept of centrifugal organisation is an example of a model that reconciles the role of abiotic and biotic processes in shaping wetland vegetation communities. Hence, Wisheu and Keddy (1992) have explained vegetation patterns as a result of both individual species' 'tolerance along environmental gradients and their ability to compete. This concept assumes that the core habitat of wetlands is characterised by low disturbance and high productivity, and is therefore dominated by few competitive species forming dense canopies. At the other extreme of the gradient, peripheral habitats present different kinds and combinations of infertility and disturbance, thereby supporting distinctive, less competitive and therefore more diverse flora (Wisheu and Keddy 1992). Local communities at given sites are assumed to reflect the cumulative effects of both abiotic and biotic processes (HilleRisLambers et al. 2012).

The framework in which important community ecology theories (including succession) can be tested in restoration is provided in Table A.4, along with a list of potential research questions, and examples of related methods with which to test each question. Highlighted in grey are the questions that I will investigate in my thesis.

1.7. Abiotic processes influencing vegetation composition

Early during the 20th century scientists such as Warming (1896) and Shreve (1917), associated certain plant species assemblages with environmental variables such as climate and altitude. Following this approach, in 1981, van der Valk proposed an important paradigm for community assembly. Building on the stochastic succession theory of Gleason (1917), van der Valk's theory focused on the role of "environmental sieves" or filters that select individuals from a pool of potential plant species that possesses the traits required for survival under particular conditions (van der Valk 1981). Today, identifying processes that cause spatial variation in species composition along environmental gradients continues to be a central objective in community ecology (e.g. Belyea and Lancaster 1999, Willis et al. 2010; Trites and Bayley 2009; Gotzenberger et al. 2012; Pausas and Austin 2001; Foster et al. 2011). Ecologists have found numerous environmental variables influencing vegetation structure and functions that act at different spatial and temporal scales (Levin 1992). Their influence on vegetation can be sporadic or continuous, influencing primary, secondary or both stages of vegetation community development. Vegetation development is also influenced by local and/or landscape level environmental variables of various intensity (Levin 1992). Relating environmental variables to community structure and function has allowed the development of numerous concepts and has fed countless hypotheses in ecology. Understanding the influence of abiotic processes on vegetation community development also has important applications in the field of restoration ecology.

1.7.1. Understanding the role of abiotic processes in an ecological restoration context

Identifying, understanding and manipulating environmental processes that influence vegetation communities are essential to accelerate or facilitate the recovery of an ecosystem (Temperton et al. 2004). Using knowledge about plant species traits and life history (Shipley 2011), restorationists can establish the physical conditions that will favor the presence of specific species adapted to those conditions. Furthermore, environmental variables are, from a restoration point of view, easier to manipulate than processes relating to the interactions between species. Hence, understanding which and how environmental variables influence community structure and functions is essential to ensure the succession of restoration attempts (Temperton et al. 2004). Abiotic conditions such as hydrology, water and soil characteristics and topography, have been identified as important variables influencing vegetation development and composition in restored marshes (Douma et al. 2012). For example, sediment characteristics such as low nutrient and organic content, and grain size can affect wetland community development and composition (Langis et al. 1991; Rooney and Bayley 2011).

In Chapter Three and Chapter Four I investigated how chemical and physical conditions influence plant and marsh functions. More precisely, in Chapter Three, I experimentally tested the response (i.e. survival and biomass) of *Carex aquatilis* to amendments with PM on oil sand sediments (CT and Tailing-Sand (TS)). I further test the influence of OSPW and natural water on *C. aquatilis*. In Chapter Four, I quantified and compared water level amplitude of fluctuation in created and natural marshes over time and relate it to marsh functions. The results from Chapter Three and Four were then related to reclamation objectives.

1.8. Biotic processes influence vegetation composition

Among the biotic interactions that influence community development, facilitation and competition have generated the most vigorous debates in plant community ecology. While some ecologists believe that biotic interactions play a minor role in the assembly of communities (McCreary 1991), others have shown that resource competition (e.g. Goldberg and Barton 1992; Navas and Fayolle 2012) and facilitation (e.g. Callaway 1994; Levine 1999; Butterfield et al. 2010) have considerable effects on community assemblages. Although those two interactions were initially studied separately and often in a laboratory under controlled environments, a shift in thinking has suggested that biotic interactions interact in the community assembly and that interactions vary along temporal and spatial gradients (Holmgren et al 1997; Wilson 2001). For instance, a “nurse plant” can facilitate the establishment of other plants in early succession by enhancing soil properties (such as increasing organic matter and nutrients), but that same plant prevents the growth of young seedlings by providing shading at a later stage of community development (Nobel 1989; Holmgren et al 1997; Bailey et al. 2007). Furthermore, it is recognised that biotic interactions are modulated by abiotic gradients (Wilson and Keddy 1991; Brooker and Callaghan 1998). For example, Wilson and Keddy (1986) showed that in the absence of disturbance on lake shores, macrophyte growth rate and standing crop increase, leading to a higher plant nutrient demand that enhances competition among plants consequently reducing species diversity. Biotic interactions influencing the assembly of communities are numerous and have been extensively investigated by ecologists. For the purpose of this synthesis, only competition and facilitation are discussed.

1.8.1. Testing biotic interactions in restoration

Testing the role of biotic interactions in the assembly of aquatic communities has been less common than testing their effects on terrestrial communities (McCreary 1991). It is argued that water, a relatively uniform environment, may alleviate heterogeneous conditions that often mimic the assembly rules under investigation (McCreary 1991; Wilson 2001). Furthermore, the zonation of hydrophytes along water gradients and the phenological separation of species may simplify the interplay of interactions under study (McCreary 1991). In addition to providing an opportunity to test biotic interactions in a relatively simple environment, the oil sands created marshes provide conditions amenable to the use of manipulative experiments that would otherwise be not acceptable in a natural setting due to the often intrusive and disruptive techniques used. Furthermore, the initial conditions imposed by newly created marshes allow one to study these interactions in both primary and secondary succession. Moreover, the harsh initial conditions of created marshes (such as low soil organic matter and high water salinity) may be suitable to test plant facilitation at the community level along stress gradients using, for example, spatial patterns analysis.

1.9. Conclusions

The science of community ecology has evolved from a descriptive discipline to a science in search of laws and rules to explain the assembly of communities. Ecological restoration, with its aim to recover lost or damaged ecosystems, has developed with few overlaps and almost independently from community ecology (Zedler et al. 1997; Temperton et al. 2004). Despite the anomalous results obtained in ecological restoration and the desire of community ecologists to find generalities, I have demonstrated that both disciplines are intrinsically connected (Palmer et al. 1996; Young et al. 2001). The problem of specialisation and the lack of cohesion between the disciplines can therefore be overcome by what Fukami (2010) termed “theoretic bridging”. The created marshes in the Alberta oil sands provide the circumstances in which both disciplines can be unified based on a common currency i.e. the theory of vegetation development.

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Appendix-A

Table A. 1: Predictions of community structure and functions development over time according to the main succession theories (see table A.2).

	Alternative Hypotheses *(continued in table 1b)*	
	Clements (1916)	**Gleason (1917)**
Community Characteristics	*Pseudo-organismic theory (Deterministic)*	*Stochastic (stochastic)*
Species richness	Increase over time	
Causes	Increase habitat complexity over time leads to increase in niche-space availability	
Species evenness	Increase over time	
Causes	Increase habitat complexity over time leads to increase in niche-space availability	
Species turnover	Rate of species replacement is fastest between community transitions	Not mentioned
Causes	Plant facilitation speed up succession Each developmental sieve provides a new set of conditions	
Species traits	Pioneers species adapted to harsh conditions are replaced by competitive species Pioneers cannot replace pioneers	
Causes	Resources decrease over time and become limited	
Community Function*		
Soil organic matter	Increase over time	
Causes	As plants die soil organic matter increase	
Soil Inorganic Nutrients	Increase over time	
Causes	As plants die soil nutrients increase	
Plant above-ground biomass	Increase over time	
Causes	Succession: herbaceous species are replaced by shrubs, and ultimately by trees	
Structure-Function		
Type of succession	Both allogenic and autogenic but (dominantly autogenic)	Both allogenic and autogenic (dominantly allogenic)
Water, soil chemistry and organic matter determine community composition	Physical environment influences vegetation development: adaptive physiology of individual plant species	Physical environment influences vegetation development: plant species respond differently according to their requirements
Disturbances	Climax versus Disclimax: influence vegetation patterns at local scale and species adapted to stress dominate over the normal anticipated climax community Create new substrates (Nudation)	Retrogressive succession: environment can deteriorate and decrease species richness and growth

Table A. 2: Predictions of community structure and function development over time according to the main succession theories (Table A.1 continued).

Community Characteristics	**Alternative Hypotheses** *(continued)*	
	Egler (1954)	**Odum (1969)**
Community structure	*Initial Floristic Composition hypothesis*	Change in Ecosystem Properties
Species richness	Decrease over time	Increase over time
Causes	Environment filters out species over time	Increase in potential niches
Species evenness	Decrease over time	Increase over time
Causes	Environment filters out species over time	Increase in interspecific competition over time
Species turnover	Not mentioned directly Slower than the turnover rate defined by Clement (Wilson et al. 1992)	Decrease over time
Causes	Inhibited by preemption of plants already established	Change from short and simple to long and complex life histories
Species traits	All plant species (traits) are present initially (as seeds) Short-lived species slowly replaced by long-lived ones Long-lived species dominate in later stages	Early stages: small size, and short and simple life histories Later stages: large size, and long and complex life history
Causes	Turnover of life history trait may occurs during secondary succession following the process described by Clement	Energy flow is higher at climax and can support more complexity
Community Function		
Soil organic matter	Increase over time	
Causes	As plants die soil organic matters increase	
Soil Inorganic Nutrients	Increase over time	Early stages: extrabiotic Later stages: Intrabiotic
Causes	As plants die soil nutrients increase	Tend to be tied up in biomass
Plant above-ground biomass	Increase over time	
Causes	Succession: herbaceous species are replaced by shrubs, and ultimately by trees	Energy flow is higher at climax and can support more biomass Organism increase size and have long life history Consequence of competition for resources (Loreau 1998) Communities acquire species over time and biomass accumulate is long-lived species (Connell and Slatyer 1977))
Structure-Function		
Type of succession	Both allogenic and autogenic (dominantly allogenic)	Dominantly autogenic
Water and soil chemistry and organic matters determine community composition	Physical environment influences vegetation development: plant species respond differently according to their requirements	Modification of the physical environment by the community: as species die organic matter accumulate
Disturbances	Disturbance can arrest succession if it occurs in early stages and affects woody species that would otherwise dominate in later stages	Pulse stability: keep community in a young stage: organisms are adapted to the frequency and intensity of the disturbance

Table A. 3: Comparing two studies describing succession in created wetlands to some of the most influential succession theories (see table 1 and 1b for reference).

Community Characteristics	**Noon (1996)**	**Mitsch et al. (2012)**
Community structure		
Species richness	Decrease over time	Increase over time (higher in planted versus unplanted treatments) Highest rate of increase occurring over the 5 first years Level off after 5 years
Support the view of	Egler	Clements, Gleason and Odum
Species eveness	Increase over time Became less diverse after 4 years	Evenness decrease over time in unplanted treatments Evenness increase over time in planted treatments
Causes	Egler	Clements, Gleason, Egler, and Odum
Species turnover	Not mentioned	
Support the view of	NA	
Species traits	Annual dominate the 3 first years Perennials dominate after 3 years Trees and shrubs decrease over time	Not mentioned
Support the view of	Clements, Gleason, Egler and Odum	NA
Community Functions		
Organic matter increase over time	Increase over time Lower then in natural wetlands after 11 years	Double over 10 years Triple over 15 years
Support the view of	Clements, Gleason, Egler and Odum	Clements, Gleason, Egler and Odum
Plant above-ground biomass increase over time	Not mentioned directly but percentage cover increases over time	Increase over time
Support the view of	NA	Clements, Gleason, Egler and Odum
Soil Inorganic Nutrients	Increase of electric conductivity Not outside of range of natural wetlands	Decrease removal over 15 years for phosphorus and nitrogen
Support the view of	NA	Odum
Structure-Function		
Type of succession	Allogenic and autogenic	Allogenic and autogenic
Water and soil chemistry and organic matters determine community composition	Physical environment influences vegetation development: plant species respond differently according to their requierments	Physical environment influences vegetation development: plant species respond differently according to their requirements
Disturbances	Not mentioned	Predation (herbiory) by muskrat decrease measured productivity and influence Typha spp. dominance Hydroperiod and flood pulsing influence vegetation composition

Table A. 4: Proposed framework adapted from Smith (2012) and Grainger and van Aarde (2012) to test important paradigms of succession and assembly theories in a restoration context. Selected alternative hypotheses are given with potential research questions to test the hypotheses. In grey are some of the research questions that will be investigated in my thesis.

Framework	Hypotheses	Potential research questions	Potential research methods
1. Identifying patterns			
Is there a discontinuity in vegetation composition among created wetlands?	Yes No	*1) Does vegetation composition vary among sites and types of site?*	Compare vegetation structure and functions in created and natural marshes Compare observed vegetation to null models to identify if patterns occur (Chapter Two)
2. Identifying processes			
A) Does difference in community demonstrate spatial patterns across the landscape?	No		Using community data collected in the field
	Yes: seed dispersal limitation	*1) Are sites autocorrelated?*	Use of aerial photos to determinate wetland location in the field
	Yes: seed dispersal limitation	*2) Are species present reflect long-distance dispersal ability?*	Using ARGIS to measure distance between sites
	Yes: landscape patchiness and heterogeneity	*3) Does differences in physical conditions match autocorrelation observed?*	Compare environmental conditions among autoccorelated sites
B) What patterns of development characterised the vegetation community? *(see table 1 for a summary of the main succession theories)*	Directional (Clement) Stochastic (Gleason) Alternative Stable State (Levin) Change in Ecosystem Properties (Odum) Initial floristic composition (Egler) No change	***Structure*** *1) Does species richness and evenness increase or decrease over time?* *2) Does the rate of species turnover decrease over time as wetland develop?* *3) Is there a trend in the replacement of species from pioneers to competitive species?* *4) Does vegetation structure of created wetlands converge or diverge over time?* *5) Is there presence of alternative stable states?* *6) Do wetlands reach an (or many) equilibrium state (s)?* ***Functions*** *7) Does biomass increase over time?* *8) Does soil organic content and nutrients content increase or decrease over time?* *9) Does richness and functions are related?* *9.a. Does species richness is correlated to biomass?* *9.b. Does the rate of species richness and biomass accumulation is the same over time?*	Using community data collected in the field Follow sites over time or use of chronosequence approach (substituting time by sites' age) Follow same site from early to late succession

		9.c. Do they level off at the same time?	
C) What are the drivers of the observed changes?	Light	*1) Do (and which) abiotic variables influence vegetation structure and functions?*	Using community and environmental data collected in the field
Abiotic *(Environmentally mediated patterns)*	Nutrients	*2) Do soil organic matters and nutrients influence the rate of species turnover?*	Comparing different soil and water amendments and species composition and function (Chapter Two and Three)
	Space	*3) Does soil development is related to vegetation structure development?*	Using community and environmental data collected in the field
	Disturbances	*4) Do (and which) disturbances (e.g. water fluctuation) characterise a system?*	Quantify and compare marshes water level fluctuation over time and identify causes (Chapter Four) Compare marsh with similar biotic and abiotic conditions being submitted to different level of disturbances
		5) Do and how much (frequency, intensity) disturbances is necessary to reach another desired stable state?	Comparing community composition pre- and post-disturbances
Biotic *(Assembly rules)*	Competitive effect/response	*6) Does a species performance affected by the performance of another?*	Reciprocal replacement series, additive or removal perturbations, and character displacement studies
	Limiting similarity	*6.a. Do species that co-occur have less niche overlap than expected at random?*	Compare species co-occurrence based on selected and similar characters of the plants and compare to a null-model
	Priority effect	*7) Can introducing plants in a specific order lead to different community composition (alternative states)?*	Control experiment where treatments represent different selected plants introduced in different order
	Priority effect	*7.a. Does planting desired species early in succession preclude later invasive species establishment?*	Control experiment where treatments represent different planting strategies compared to control treatments
	Stress Gradient	*8) Do and how facilitative effects of nurse-plant influence community composition along stress gradient?*	Compare vegetation composition under the effect of different treatments (nurse-plant species versus unplanted) along environmental gradient (e.g.salinity)

CHAPTER 2

Title: Defining marsh vegetation composition patterns in the oil sands of Alberta post-mined landscape; the influence of oil sands process materials and organic amendment.

2.1. Introduction

Community ecology attempts to explain the processes responsible for the patterns of distribution, interaction and abundance of species (Weiher and Keddy 2001). Identifying and describing assemblages in nature is therefore a fundamental step to provide insights on the potential processes responsible for the differences segregating communities into distinct assemblages (Gotzenberger et al. 2012; HilleRisLambers et al. 2012). The presence of patterns may reveal that plants do not assemble randomly but are rather constrained by the cumulative effects of various variables acting at different spatio-temporal scales (Gotzenberger et al. 2012). In plant community ecology stochasticity, abiotic, biotic, spatial (dispersal), and historical processes have been identified as “filters” influencing the composition of local plant communities (Gotzenberger et al. 2012). These filters represent the limitations to growth or, as used in this chapter, “constraints”.

The post-mined landscape of oil sands in Alberta provides a unique opportunity to study the active and ongoing assembly of plant communities. The post-mined areas to be reclaimed present a wide range of environmental conditions constraining and shaping the establishment and development of the biota. Patterns in the assembly of plant communities are expected to reflect the constraints present in the post-mined conditions. Thus, describing plant community patterns under a set of reclaimed conditions is a fundamental step to identify constraints shaping these communities. Furthermore, identifying plant community patterns that converge toward or deviate from the desired reclamation endpoint (or reference sites), provides some ability to predict the outcome from the current reclamation conditions and strategies and adjust reclamation practices.

2.1.1.The reclamation context of the oil sands of Alberta

Vitt and Bhatti (2012) suggest that to successfully reclaim boreal disturbances, created sites must have species composition similar to natural references sites. However reclamation of marsh composition equivalent to natural sites in the Fort McMurray region present numerous challenges. Successfully reclaiming marsh vegetation composition and functions is difficult due to the chemical environment conditions resulting from oil sands activities. To understand the potential effects of different amendments on biota, including hydrophytic macrophytes, the oil sands companies have constructed three types of pilot marshes in the mined landscape. The first type of created marshes are referred to as created-tailings (CT) and are directly or indirectly (runoff or seepage from tailings ponds) amended with mature fine tailings (MFT)[5] or consolidated-tailings (CT) as substrate (i.e. oil sands process material (OSPM)) and/or filled with oil sands process water (OSPW). Other pilot marshes are referred as created-peat (CP) and created-unamended(CU). CU marshes have not received any type of amendment while CP marshes are capped with a layer of peat-mineral mix (PM)[6]. As is the case for CU marshes, CP marshes are not affected directly or indirectly by tailings constituents. Marshes of different origin (i.e. created versus natural) and of different types (i.e. amendments) are hypothesised to be characterised by dissimilar environmental conditions. Differences of vegetation communities among marshes are expected to reflect the wetlands' different environmental conditions.

2.1.2. Created marsh water and sediment quality

Due to their chemistry, OSPM and OSPW constituents are hypothesised to affect the terrestrial and aquatic vegetation, of the reclaimed landscape. Although the presence of naphthenic acids and hydrocarbons in tailings has been shown to negatively affect plant functions (Kamaluddin and Zwiazek 2002; Mollard et al. 2011),

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[5] MFT and CT are also refer collectively as 'fliud fine tails' (FFT)

[6] PM is a term used by the oil sands industry and refers to the material salvaged during the top-soil removal preceding surface mining. PM results of a mixture of drained peatlands and some underlying mineral soil ((Hemstock et al. 2010). The constituents of PM may vary depending on the harvesting location but PM usually contains between 25 to 50% (by volume) of mineral material (Hemstock et al. 2010). PM is stored in on-site stockpiles before being used for reclamation purposes as a source of organic matter (Luong,1999).PM was hypothesised to be beneficial to reclamation because it is a source of organic matter and seeds and propagules.

high alkalinity, low nutrient content of the water and the sediment have been identified as limiting factors to the success of mosses and aquatic macrophyte community colonisation in the post mined landscape (Trites and Bayley 2009; Rooney and Bayley 2011; Raab and Bayley 2011).

Oil sands created marshes' initial soil conditions are typically a coarse mineral, clay and sandy (e.g. tailings-sand (TS)) substrate supporting only pioneer wetland vegetation (Wieglieb and Felink 2001). Following mining, the bare mineral soil of the post-mined landscape is generally characterised by low penetrability, low nutrient content and the lack of seeds and propagules (Bradshaw 1997, Wieglieb and Felinks 2001). To enhance soil properties that favor vegetation community development, companies have amended created marshes with a PM (Harris 2007) collected during the top-soil removal preceding surface mining. The amendment of PM to created marshes was assumed to be beneficial to plants because it generally contains higher amount of micro (e.g. magnesium, iron) and macro (phosphorous nitrogen and carbon) nutrients than TS and CT sediments (see Table 3-1 for a summary of the chemistry CT, TS and PM)(Harris 2007, Luong 1999). Compare to TS and CT, PM bulk density is assumed to be lower. The lower bulk density of PM is hypothesised to facilitate root penetrability, improve soil aeration and increase microbial habitat (Luong 1999; Sutton-Grier et al. 2009). PM is assumed to have greater water retention than CT and TS (Harris 2007). Its addition to created marshes is also hypothesised to reduce potential pH fluctuation and to increase absorption of toxic compounds (Kovalenko et al. 2012).The effect of organic matter (and PM amendment in the created marshes of the oil sands of Alberta) have been investigated by many studies (e.g. Luong 1999; O'Brien and Zedler 2006; Bailey et al. 2007; Sutton-Grier et al. 2009; Ballantine et al. 2012; Kovalenko et al. 2012). Benefits of amendments to newly restored marshes have however been mostly demonstrated on soil properties and ecosystem functions (e.g. Bruland and Richardson 2004; Ballantine et al. 2012), and few studies have shown its positive effect on community composition (community richness and/or diversity) (but see Cooper 2004), (Sutton-Grier et al. 2009; Ballantine et al. 2012). Thus, it remains unclear if amending created marshes on the oil sands of Alberta leads to more desirable community composition.

In addition to chemical conditions present in the reclaimed landscape, physical conditions such as hydrology and basin morphology, need to be finely tuned to support the biological and ecological requirements of aquatic plants (Olson and Barker 1979; Olson 1981; Rumble 1989; Raab and Bayley 2011; Rooney and Bayley 2011). Water level is the primary variable influencing vegetation composition within wetlands (Mitsch and Gosselink 2011). Studies have also shown that water depth and its permanence within the zone of wetland vegetation influences light penetration and oxygen availability and thus, the photosynthetic capability of submerged vegetation (Olson 1981). Furthermore, created wetlands with steep basin slopes were characterised by having restricted littoral zones with little or no drawdown period (Rumble et al. 1985; Zampella and Laidig 2003). Thus, the steep basin slope observed in many created marshes in oil sands represents a physical characteristic leading to low habitat quality to the aquatic vegetation, hence, influencing vegetation composition (Raab 2011).

The intent of this study is to describe vegetation community patterns in different types of created marshes (i.e. CT, CU and CP) and secondly, to investigate the effect of amendment (i.e. OSPW/OSPM, CT/PM/no amendment) on plant community composition. To achieve these objectives, the vegetation of CT, CU and CP marshes was compared with one another and to natural marshes of the Fort McMurray region. The two specific objectives of this study are to:

1) characterise and compare sediment and water chemistry and physical characteristics of the four marsh types,

2) define vegetation patterns in each marsh zone and type (i.e. submersed aquatic vegetation (SAVZ), emergent (EZ) and wet-meadow (WMZ) zones) and determine if the vegetation composition is significantly different among marsh types, and relate differences to water and sediment differences among marsh types.

Due to the small number of created marshes in the post-mined landscape that have received an amendment of PM (n=4), their environmental conditions and vegetation composition will only be compared qualitatively to other marsh types. Their environmental conditions and vegetation composition were presented in analyses only to depict their tendency and characteristics in relation to other marshes and hence provides some insights on the role of PM amendment in reclamation.

2. 2. Research Site

This study was conducted in the Fort-McMurray Region of northeastern Alberta on the mining leases of Syncrude Canada Ltd. and Suncor Energy, Inc. A total of 51 marshes were selected of which 16 were natural, 14 were CT, four were CP and 17 were CU (see Appendix C for details about each selected marsh). Natural and created marshes were randomly selected to represent their respective populations. The natural marshes represent the typical marsh-type wetlands found in the boreal plain ecozone (see Locky et al. 2005 for more details). A marsh was defined as “natural” if it met four criteria: 1) its origin did not result from anthropologic intervention, 2) its past and present internal and surrounding conditions showed no major evidence of anthropologic disturbance, 3) it exhibited evidence of past or present beaver activities, and 4) it was permanent and contained at least three vegetative zones (i.e. submersed aquatic vegetative zone (SAVZ), emergent (EZ) and wet-meadow zone (WMZ)). A CU marsh was defined by: 1) anthropogenic origins, 2) no direct or indirect addition of OSPW and or OSPM, 3) constructed in the post-mined landscape or not and 4) supporting the three vegetative zones described above for natural marshes. CT marshes differed from CU ones by 1) the direct or indirect addition of OSPW and or OSPM during their construction and 2) being located solely in the post-mined landscape. Finally, CP wetlands were defined by the same criteria as CU with the exception that they were only found on-site and their mineral soil has been capped with a layer of PM. Created marshes selected were over 15 years of age (with the exception that one that was 7 years old). Although natural marsh ages were unknown, most were believed to be >1000 years old (Raab and Bayley 2012), the study of historical aerial photos ensured that natural marshes were present on the landscape in 1957 (more than 60 years ago). The average size of CT marshes studied was 18926 m^2 (ranging from 400 m^2 to 36,416 m^2), CP was 1628 m^2 (ranging from 600 m^2 to 4,000 m^2), CU was 24,181 m^2 (ranging from 878 to 226,000 m^2) and natural was 15,272 m^2 (ranging from 19,100 m^2 to 1,510,000 m^2) (Nakhaie 2013). In addition to the criteria defined above, sites were selected based on accessibility with an attempt to encompass the greatest variation of landscape diversity while selecting for homogeneity within wetland types. Mean annual precipitation in the Fort-McMurray region is 455.5 mm, with an average temperature of 13.2°C in summer and 13.5°C in winter (Strong and Laggat 1992). The topography of the area is flat to slightly rolling (Locky et al. 2005).

2.3. Methods

2.3.1. Data collection

Empirical data were collected during the period of peak aboveground standing crop at the end of August in 2008 to 2012. To capture all the vegetation and environmental variables variation, each sampled marsh was visually stratified into three zones: 1) SAVZ, 2) EZ, and 3) WMZ (Figure 1). In this study, the SAVZ was an open area of a maximum depth of 1.5 meters[7] and qualified by the presence of only submerged and floating vegetation. The EZ was characterised by having submersed and emergent vegetation concentrically distributed in a fringe around the SAVZ. The WMZ was characterised by a water-saturated (hydric) soil but where the water depth was very shallow (<2 cm). Prior to sampling, an aerial photograph of each marsh was used to divide each marsh into four quadrants of same approximate size. Six transects per marsh were positioned following a stratified randomised design. Within each quadrant,transects were randomly positioned. Each transect perpendicularly crossed the three zones. Along each transect, zone width was measured. Along each transect and within each sampled zone (SAVZ, EZ and WMZ), one randomly-positioned one-m^2 plots were assessed. In total, 18 plots were surveyed in each marsh (six plots per zone). In each plot, water depth and slope were measured and a percentage cover was attributed to each plant species identified. In each marsh, a 10-minutes walk-around[8] survey was conducted simultaneously by two researchers to identify less common species that may not have been seen in plots. Taxa were identified to the species level.

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[7] The maximum depth was restricted to 1.5 m to comply with the safety rules emitted by the oil sands companies.

[8] The amount of time spent for the walk-around was determined using a species-area curve analysis. As the cumulative time spent searching for new species increases, the number of species detected increased and then begins to plateau at approximately 8 minutes (with two researchers walking in opposite direction; one surveying the SAVZ and the EZ, one surveying the WMZ).

Aquatic mosses and the macroalga *Chara spp.* could not be identify at the species level. Submersed vegetation was collected in a 0.25 m^2 area from the plot, emptied into a water-filled pan, and then sorted by species (Bayley and Prather 2003). Submerged vegetation was then oven dried and weighed for biomass. Biomass by species, as determined in the lab, was transformed into five classes: 1) >0 to 5 g; 2) >5 to15 g; 3) >15 to 30 g; 4) 30 to 50 g and 5) >50 g. A sub-sample of 28 marshes (CT=7, CT=9, N=8 and CP=4) was randomly selected and assessed for soil chemistry. Within each plot, overlying vegetation was removed and three soil samples were collected from the vegetation rooting zone (top 10 cm) using a small plastic shovel. The samples were approximately 10 cm long and 3 cm in diameter. The samples were composited per zone and sent to the Natural Resources Analytical Laboratory of the University of Alberta (Alberta, Canada) for physical and chemical analyses. Loss on ignition (LOI) to estimate the percentage of organic material present, electrical conductivity (EC), soil moisture, total phosphorus (TP) and total nitrogen (TN) analyses were thus performed. Marsh water chemistry including pH, oxidation-reduction potential (ORP), salinity, dissolved oxygen concentration (DO), and specific conductance (adjusted for temperature and hydrogen ions) were determined using a hand held oxygen, conductivity, salinity, and temperature System (YSI model 85). Measurements were taken in the SAVZ. Plots assessed for species composition were then combined so that each of the three marsh zones became the sampling unit.

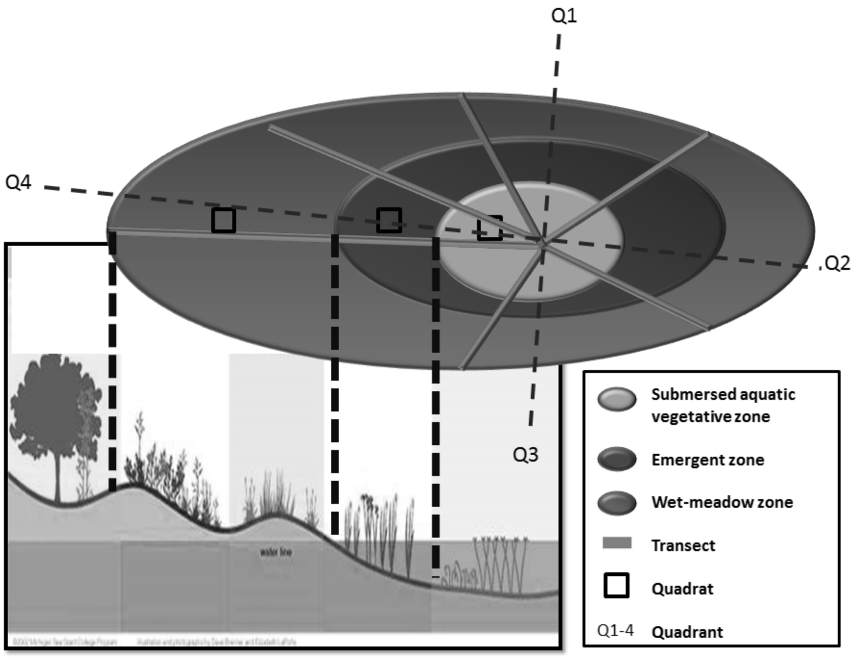

Figure 2- 1: Created and natural marshes were assessed following the same sampling design.

2.3.2. Data and statistical analysis

To determine the number of plots required to be sampled to find significant differences among marsh type vegetation richness and percent cover, the data collected in 2008[9] were used to perform a power analyses at α level 0.05 and indicated a minimum power of 0.6. Sample size was adjusted to six plots in each of the three marsh zones. A sub-sample of marshes (n=10) sampled in 2008 was re-sampled in 2012 to ensure that differences observed were not attributable to differences among sampling years.

Missing values for environmental variables were few and randomly distributed among marsh types. Thus, a missing value was corrected by using the median of the variable within the marsh type (McCune and Grace 2002). Water, soil and physical measurements measured at the zone level were averaged for each marsh to simplify the comparison of marsh types. For the statistical analysis, pH measurements were converted into hydrogen ions for analysis and then reconverted to pH for graphical

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[9] The year 2008 was the year during which the methodology was developed and tested. A total of 18 marshes were sampled (CT= 5, CU= 5, N=5, and CP=3) following the same methodology described in this chapter.

representation and discussion. Descriptive statistics for the environmental data set were first examined with summary statistics to check for errant entry or decimal place errors. Using R v. 2.10.1, each environmental variable was inspected for normality, skewness and kurtosis. Q-Q plots were used to visualize data distribution and variances and Shapiro-Wilk normality test was also performed on the data. Descriptive statistics of vegetation community data were also examined. The degree of variability in rows and columns was measured with the coefficient of variation (CV) to determine if data transformation was necessary. Prior to cluster and ordination analyses, and based on methods of Vitt et al. (1995) and Locky et al. (2005), percentage cover by species as determined in the field was transformed into four classes to increase multivariate normality: 1) Rare (cover ≤1%); 2) Few (2–10% cover); 3) Common (11–74% cover); and 4) Abundant (75–100% cover). Outliers in the environmental and vegetation community data sets were identified using the outlier analysis in PC ORD v. 5.10 (Euclidean distance, cutoff=2) (McCune and Grace 2002). For each type of univariate and multivariate analysis performed with the environmental and vegetation community data, an outlier analysis was performed. The remaining marshes were used to describe and compare vegetation richness and composition as well as water chemistry and the other physical conditions including marsh area, water depth, zone width, and zone slope. Due to the smaller sample sizes and compositing of collected soil chemistry data (n=25), these variables were analyzed separately from the other environmental variables.

To compare environmental variables among marsh types, all environmental variables measured at the zone level (i.e. physical conditions and sediment chemistry) were averaged so that one summary measurement could be obtained for each marsh sampled. To determine if the sediment and water chemistry, and water physical characteristics averaged at the marsh level were significantly different among marsh types, two Principal Component Analyses (PCA) were performed[10] using R v. 3.0.3. The PCAs were used to express the covariation of the sediment and water chemistry and

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[10] Ordination of the sediment and water chemistry and physical characteristic was performed at at the marshes level only (i.e. One PCA including water chemistry and physical characteristics (n=50) variables and one PCA including the sediment chemistry variables (n=28)).

physical characteristics as a smaller number of composite variables. The marshes scores in the reduced space were then used to perform the ANOVAs[11] (with Scheffe test adjustment to account for unequal sample sizes). Prior to the PCAs, the non-normal environmental variables were log-transformed and then standardized to account for unequal variable units. After standardization variables are expressed with the same units. To determine the number of axes to interpret a scree plot was produced and ecological meaning inferred (McCune and Grace 2002).

The gamma diversity was calculated as the total number of species identified across all marsh types sampled in the Fort McMurray region. Alpha diversity was measured as the mean plant species richness per marsh type and further developed as the average plant species richness in each marsh zone. Beta diversity is a measure of variability in species composition among marsh types and among a marsh's zones. Beta diversity is measured without reference to a specific gradient and was measured using the Sørensen's Similarity Index expressed as $\beta=2c/(S1+S2)$ where c was the number of species common to both communities and S1 and S2 were the total number of species in each community, respectively. Alpha, beta and gamma diversity were measured following the concept developed by Whittaker (1972, 1977) and further explained by McCune and Grace (2002). To determine if species richness of each zone was significantly different among marsh types an ANOVA (with Scheffe post hoc test to account for unequal sample size) was performed.

To examine if vegetation composition tend to differ among zones (regardless of marsh types), using R.3.0.3., a Non-metric Multidimensional Scaling Analyses (NMDS) was used. To determine and constrast patterns of vegetation composition among marsh types, another set of NMDS were performed at the zone level. The preliminary run for each NMDS has the following parameters: the distance measure is Sørensen (Bray-Curtis), the initial number of axes was 6, the instability criterion was of 0.00001. The maximum number of iterations was set at 350 and a random start configuration was

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[11] A MANOVA was first used to determine if the multivariate test statistic (Wilk's Lambda) was significant but an error message appeared because there were more unknowns than measurements. So to work around this limitation the method described here was preferred.

selected. Fifty runs were conducted with real data followed by 50 runs with randomised data (McCune and Grace 2002). A scree plot was then evaluated to determine the minimum number of axes to interpret. A Monte Carlo test was performed to seek a better-than random solution. Stability of the solution was then evaluated by the final stress value (McCune and Grace 2002). The results from the initial run were then used to perform a second run with the selected parameters. Thus, a NMDS with 2 to 3 axes, using a random start and no Monte Carlo test was performed to ordinate wetland zones in the species space and to ordinate marsh types (each zone separatly) in the species space.

To determine in each zone (i.e. SAVZ, EZ, and WMZ) which species can be used as indicators of marsh types (i.e. CT, CU, CP and natural), a series of indicator species analyses were preformed using R 3.0.3 (*indicspecies* package) (Dufrene and Legendre, 1997). The statistical significance of the relationships were tested using a permutation test (number of permutations=999). The indicator value obtained for each indicator species was further developed in two components i.e. specificity and fidelity (Dufrene and Legendre, 1997).

To determine whether there was a significant difference in terms of vegetation composition among the three marsh types (CT, CU and N), a Multiple Response Permutation Procedure (MRPP) was performed with PC ORD v.5 using a Sorensen (Bray-Curtis) distance measure and (n/Σ(n)) to weigh the groups for each marsh zone (SAVZ, EZ and WMZ). This distance measure was selected so that both the NMDS and the MRPP were performed with the same distance measure. Because groups were of unequal size, each group was randomly reduced to n=15 following the method described by Peck (2010 p.106).

2.4. Results

2.4.1. Environmental variables differences among marsh types

Natural marshes, CT, and CU marshes were each characterised by distinct environmental conditions (Table 1, Table 2, Table 3, Figure 1, and Figure 2). The PCA results for the sediments chemistry suggested that two axes were useful for meaningful interpretation (Table 2, Figure 2). PC1 was correlated with sediment moisture, TN, TP, and LOI (Table 3). PC2 was correlated with EC. Based on the interpretation of the PCA results and assisted by a scree plot, 2 axes were interpreted for the water chemistry and the physical conditions (averaged at the marsh level)(Table2, Figure 2). PC1 was correlated with water salinity, ORP, conductivity, DO and zone width (Table 3). The PC2 was correlated with pH, water depth, basin slope, and zone width.

The analyses revealed that significant differences in sediment and water chemistry and physical characteristics (averaged at the marsh level) existed among marsh types (sediment chemistry $\text{ANOVA}_{\text{PC1}}$, $F_{2,26,}$=6.41, p<0.05:, $\text{ANOVA}_{\text{PC2}}$, $F_{2,26}$=0.05, p=0.81; water chemistry and physical characteristics: $\text{ANOVA}_{\text{PC1,}}$ $F_{2,44}$=71.20, p<0.001, $\text{ANOVA}_{\text{PC2}}$, $F_{2,44}$=78.13, p=0.38)(Table 3). Due to insufficient sample size, CP marshes were not included in the analyses but nevertheless included in tables and figures to provide a point of comparison with other marsh types.

The results suggested that differences in sediment chemistry were mainly attribuated to the origin (i.e. created or natural) of the marshes rather than caused by their type of amendment (Figure 1, Table 1 and Table 2 in Appendix B to see sediment chemistry at the zone level). Sediment chemistry of CT and CU was not found significantly different (Scheffe test, p=0.6). CT and CU marshes had significantly (Scheffe test, p=0.01, p=0.02 respectively) less moisture, TN, TP, and organic content (LOI) than the natural marshes.

The differences in water chemistry among marshes were related to their type of amendment rather than to their origin. As expected, the results suggested that the addition of OSPW and/or OSPM proposes water chemistry that is atypical from the ones of natural marshes of the regions. When compared to CU and natural marshes, the CT marshes had significantly (Scheffe test, p<0.001, p<0.01 respectively) higher water salinity, conductivity, and DO, but lower ORP. The zone width (averaged at the marsh level) of CT marshes was also significantly shorter than the one measured in CU and natural marshes. The tests revealed that CU and natural marshes were not significantly different in term of water chemistry (averaged at the marsh level) (Scheffe test ,p=0.8).

Table 2-1: Averaged sediment chemistry, water chemistry, and physical characteristics of marshes and their associated ±CI (95%). Variables in bold were significantly different (ANOVA, $p<0.05$) among marsh types. LOI=lost of ignition, TN= total nitrogen, TP= total phosphorus, EC= electrical conductivity, DO= dissolved oxygen concentration, ORP= Oxidation reduction potential).

	Variables	CT	CU	CP	N
Sediment	**Moisture (%)**	50.0	43.5	52.1	201.1
	CI	22.9	32.0	30.1	337.0
	LOI (%)	4.3	4.0	4.2	11.4
	CI	1.3	2.8	1.2	8.4
	TN (mg/Kg)	3.9	3.7	3.7	11.9
	CI	2.4	2.7	2.3	9.8
	TP (mg/Kg)	0.8	0.7	0.5	0.9
	CI	0.2	0.4	0.2	0.4
	EC (µS/cm)	522.6	231.6	215.0	157.6
	CI	248.8	188.1	294.3	119.1
Water	pH	8.0	7.9	7.6	7.4
	CI	0.2	0.2	0.5	0.2
	DO (mg/L)	7.4	4.4	7.4	4.8
	CI	0.6	0.5	0.5	0.5
	Salinity (ppt)	1.1	0.2	0.5	0.1
	CI	0.2	0.0	0.7	0.0
	ORP (mV)	64.0	153.4	60.3	120.0
	CI	11.4	18.5	24.1	8.7
	Conductivity (mg/L)	2123.8	365.5	1375.5	224.0
	CI	380.3	55.9	1711.2	76.4
	Depth (cm)	30.2	44.2	32.5	29.8

	CI	8.9	36.8	53.1	16.9
Physical	Basin Slope (^{o})	17.8	13.6	6.8	3.6
	CI	7.2	10.3	9.8	2.7
	Zone Width (m)*	1.8	3.9	8.6	2.1
	CI	0.2	0.2	1.1	0.3

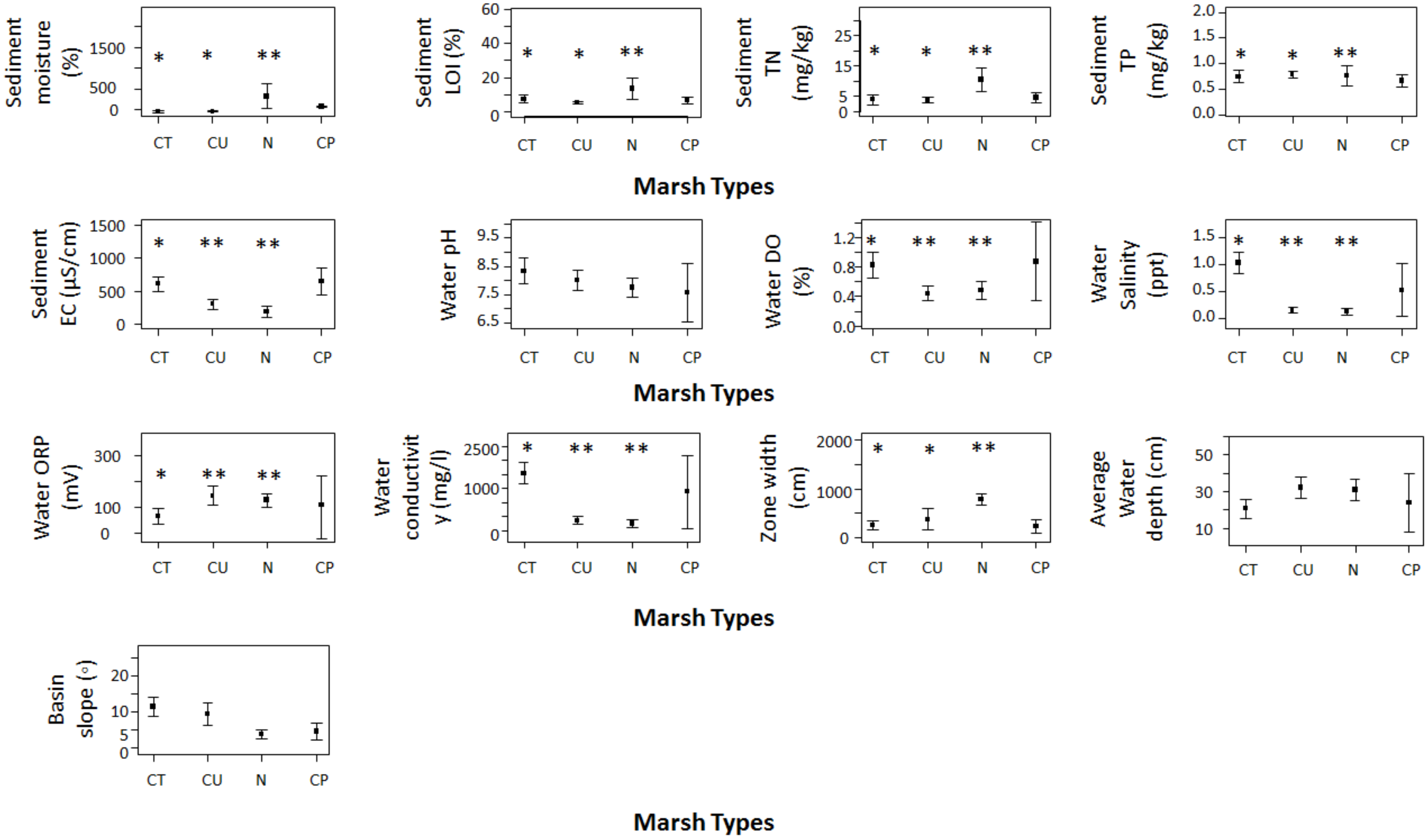

Figure 2- 2: Distributions of the selected environmental variables characterising the four marsh types. The values represent the averaged sediment and water chemistry, and physical characteristics (±CI). (*) indicates significant differences (Scheffe post hoc test, α=0.05).

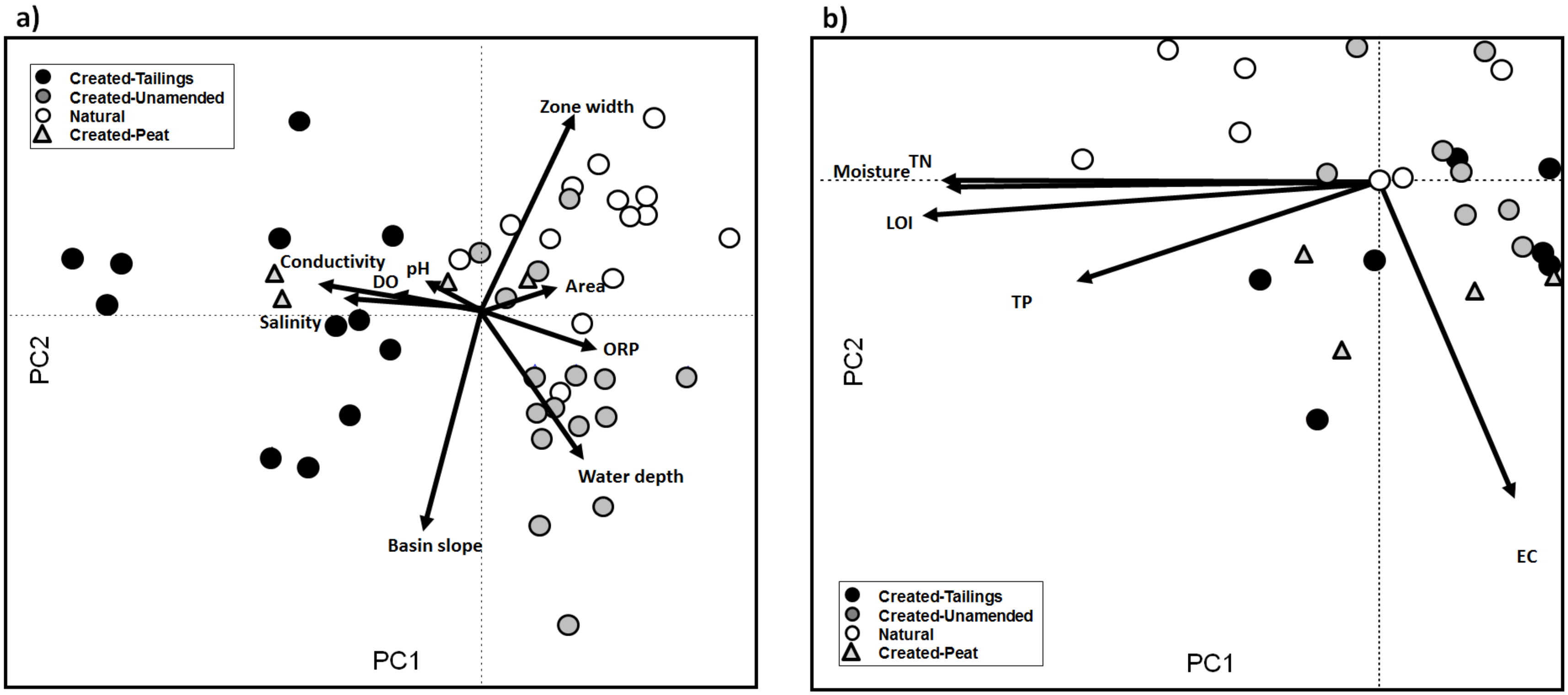

Figure 2- 3: The two PCAs performed using the variables averaged at the marsh level. The two dimensions solution is graphically presented. In **a)** ordination of the marshes based on the water chemistry and physical characteristics, and **b)** ordination of the marshes based on the sediment chemistry.

Table 2- 2: Summary for the two PCA analyses. Sediment and water chemistry analysis and physical characteristics were performed using the averaged of the measured data at the marsh level. PCA analyses using sediment chemistry data were also performed using the average measurement taken at the zone level.

	Variance explained (%)		
PCA	PC1	PC2	Cumulative
Marsh level			
Sediment chemistry	61.7	19.5	81.3
Water chemistry and physical characteristics	37.7	19.3	70.9

Table 2- 3: Summary of the variables' loadings for the two PCA analyses. Sediment and water chemistry analysis and physical characteristics were performed using the averaged of the measured data at the marsh level.

Sediment chemistry variables (marsh level)	**PC1**	**PC2**
Moisture	-0.53	0.01
LOI	-0.54	-0.1
TP	-0.35	-0.3
TN	-0.52	0.01
EC	0.16	-0.95
Water chemistry variables (marsh level)	**PC1**	**PC2**
ORP	0.37	-0.15
Zone width	0.32	0.50
Water depth	0.18	-0.51
pH	0.14	0.40
Basin slope	-0.25	-0.48
DO	-0.38	0.21
Salinity	-0.50	0.11
Conductivity	-0.50	0.11

2.4.2. Characterising and comparing marsh species composition

In total, among all four marsh types (n=51) 142 plant species (excluding shrub and tree species) were identified (Table 3 in Appendix B). In the four marsh types, a total 19 plant species composed of both submersed and floating species were identified in the SAVZ, 80 submersed and emergent plants species were found in the EZ, and 92 hydrophytes and mesophytes were identified in the WMZ.

As corroborated by Trites and Bayley (2009), an increasing trend in species richness was observed from the SAVZ to the WMZ for all three marsh types (Figure 4). The mean species richness of the SAVZ of CT marshes was 1.3; for the EZ it was 3.9, and for the WMZ was of 9.7 (Figure 3) (Table 4). CU marsh mean species richness of SAVZ was 3.9; of the EZ, 5.9; and of the WMZ, 9.3. Natural marsh mean species richness for SAVZ was 4.6; EZ, 8.6; and WMZ, 9.9.

Table 2- 4: Mean ±CI (95%) number of plant species per zone in each marsh type (n=sample size, T=total)

Alpha	CT (n=11)				CU (n=15)				N (n=14)				CP (n=4)			
	SAVZ	EZ	WMZ	T	SAVZ	EZ	WMZ	T	SAVZ	EZ	WMZ	T	SAVZ	EZ	WMZ	T
Averaged	1.3	3.9	9.7	20.5	3.9	5.9	9.3	25.0	4.6	8.6	9.9	30.4	2.0	4.3	8.3	18.0
CI	0.3	0.8	1.1	1.7	0.7	0.9	0.8	1.8	0.8	1.0	0.9	1.7	0.3	0.7	1.0	1.8

Species richness in the SAVZ of CT marshes was significantly lower than the one characterising CU and Natural marshes (Table 5, Figure 4). The species richness in CT and CU marshes EZ was not significantly different whereas species richness was significantly lower in CT and CU marshes than in natural marshes. The WMZ's species richness was not found to be statistically different between all three marsh types.

Table 2- 5: Results from the ANOVAs (with Scheffe post hoc test, α=0.05) to test for significant differences in term of species richness among marsh zones of different marsh types. Significant differences are in bold.

	SAVZ			EZ			WMZ		
Pairwise	CT-CU	CT-N	CU-N	CT-CU	CT-N	CU-N	CT-CU	CT-N	CU-N
p-value	**0.001**	**0.001**	0.93	0.28	**0.002**	**0.05**	0.80	0.87	0.99
$F_{2,44}$	0.0005			0.0028			0.8071		

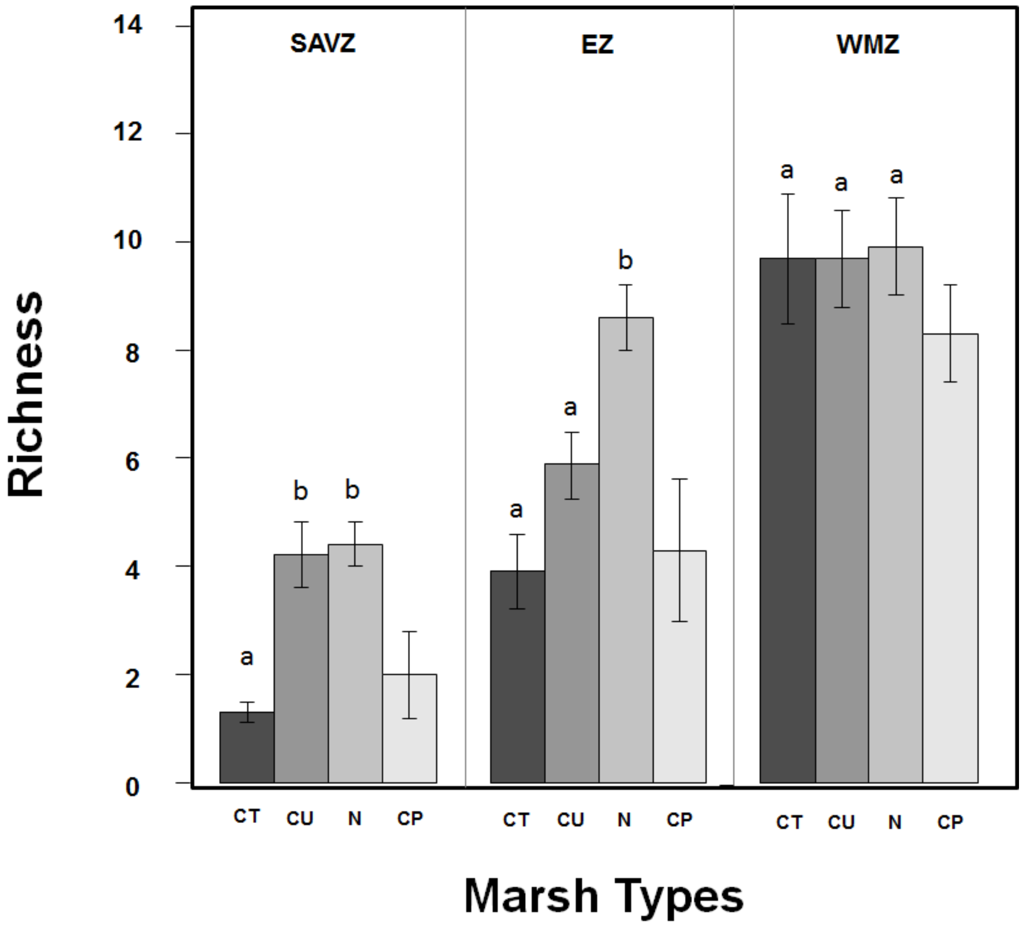

Figure 2- 4: Average number of plant species (richness) in each zone of each marsh types and their associated ±CI (95%). Letters indicate significant differences among marshes (CP marshes were not included in the statistical analysis).

The degree of similarity among marsh types and among marsh types' zones was measured using the Sørensen's Similarity Index (Sørensen, 1957; Magurran, 2004) where β=0 suggests that no species were shared by two marsh types whereas β=1 indicates that two marsh types were composed of the same species. Thus, the results suggest that few species were shared between CT and natural marshes (β=0.3), whereas CT and CU share some species (β=0.5) (Table 6). CU and natural marshes were the two types sharing the most species (β=0.7). When comparing zones of different marsh types, the SAVZ of CU and natural were the most similar (β=0.8), whereas CT and natural SAVZ shared the least similarity in species composition (β=0.3). The SAVZ of CT and CU exhibited an intermediate level of similarity where many species are shared between both marsh types (β=0.6). The EZ of CT and CU were characterised by the same species presence (β=1), the EZ of CU and natural marshes had a high degree of similarity (β=0.8) whereas CT and natural marshes shared some similarity (β=0.5). The WMZ of CT and CU marshes, as well as CU and natural shared the most similarity (β=0.7). The WMZ of CT and natural marshes shared some plant species similarity (β=0.5). The CP marshes shared most similarity with CT (β=0.5) and less with both CU and natural (β=0.4). At the SAVZ level, CP shared many species with CT (β=0.8) and few with both CU (β=0.3) and natural (β=0.4). The EZ of CP shared few plant species with CT (β=0.4), CU (β=0.3), and natural (β=0.2) marshes. The WMZ of CP shared some similarity with CT (β=0.5) and few with both CU (β=0.4) and natural (β=0.3)

Table 2- 6: Degree of similarity in term of species composition among marsh types and their three zones. Beta diversity was measured with a) the average of species found in each marsh type, b) the average of species found in the SAVZ of each marsh type, c) the average of species found in EZ of each marsh type, d) the average of species found in the WMZ of each marsh type.

a.	**Marsh**				**b.**	**SAVZ**			
	CT	CU	N	CP		CT	CU	N	CP
CT					CT				
CU	0.5				CU	0.6			
N	0.3	0.7			N	0.3	0.8		
CP	0.5	0.4	0.4		CP	0.8	0.3	0.4	
c.	**EZ**				**d.**	**WMZ**			
	CT	CU	N	CP		CT	CU	N	CP
CT					CT				
CU	1.0				CU	0.7			
N	0.5	0.8			N	0.5	0.7		
CP	0.4	0.3	0.2		CP	0.5	0.4	0.3	

2.4.3. Characterising vegetation composition in each marsh zones

The NMDS performed among zones (i.e. SAVZ, EZ, and WMZ) initial run suggested that the final best solution contains 2-dimensions and presented a mean stress of 18.9. From a Monte Carlo test I determined that this result was significantly lower than what would have occurred randomly (p=0.01). The rerun provided a mean stress of 14.1 and a final instability of >0.0001 after 350 iterations. The cumulative correlation between ordination distances and distances in the original space was high (r^2=0.75) with NMDS1 r^2=0.32, and NMDS2 r^2=0.43. The NMDS results for the SAVZ initial run suggested that the final best solution contains 2-dimensions and presented a mean stress of 19.9. From a Monte Carlo test it was determined that this result was significantly lower than what would have occurred randomly (p=0.02). The rerun provided a mean stress of 16.4 and a final instability of >0.0002 after 350 iterations. The cumulative correlation between ordination distances and distances in the original space was high (r^2=0.80) with NMDS1 r^2=0.27, NMDS2 r^2=0.35 and axes 3 r^2=0.19. The NMDS results for the EZ initial run suggested that the final best solution contains 2-dimensions with a mean stress of 19.1 and a Monte Carlo test p-value of 0.02. The rerun provided a mean stress of 17.3 and a final instability of >0.0001 after 350 iterations. The cumulative correlation between ordination and original distances was of 0.56 with NMDS1 r^2=0.21, and NMDS2 r^2=0.35. The NMDS results for the WMZ initial run identified 2- dimensions for the best final solution with a mean stress of 19.1 and a Monte Carlo test p-value of 0.004. The rerun resulted in a mean stress of 19.4 and a final instability of 0.0001 after 350 numbers of iteration. The cumulative correlation between ordination and original distances was of 0.70 with NMDS1 r^2=0.12, and NMDS2 r^2=0.58.

Each zone had vegetation composition distinctive from each other (figure 5). Most SAVZ sampled were clustered together in the ordination space and their vegetation composition showed limited overlap with the vegetation composition of EZ and no overlap with the one of the WMZ. Despite that the vegetation composition of the EZ and the WMZ partially overlapped in the ordination space, the EZ had plant species that were not found in the WMZ of some marshes. The ordination performed to illustrate the patterns of vegetation composition among zones provided the necessary reasoning for further investigating the vegetation composition of each zone individually.

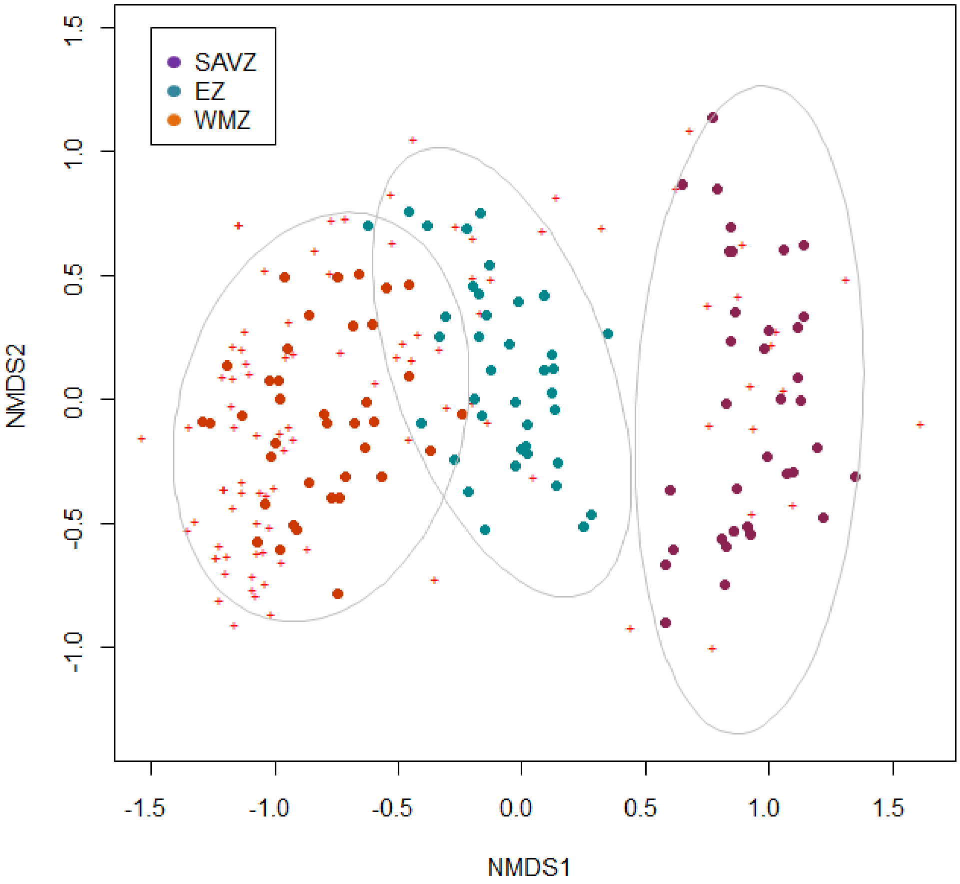

Figure 2- 5: NMDS of the three zones (SAVZ, EZ and WMZ) based on their vegetation composition. Each point represented the zone of a marsh. An individual marsh can be represented by three different points (i.e. zones). Species are represented by the red "+" but were not labeled to facilitate the visual interpretation of the ordination. The ellipses were drawn around each zone type to include at least 95% of the marshes associated with that zone.

The vegetation composition of the SAVZ of CT and CP marshes was less diverse than the one of CU and natural marshes. The NMDS results supported and corroborated the richness and diversity analyses ((Figure 6). In addition, CT and CP marshes SAVZ were composed of different vegetation composition than CU and natural marshes (Figure 6, Table 7). Most CT marshes were associated with only two species: *Chara spp.* and *R. cirrhosa* (Table 7). The SAVZ of most CT and CP marshes was dominated by monospecific stands of *Chara spp*.. Although *Chara spp.*was negatively correlated with *Utricularia macrorhiza*, *U. minor*, *Stuckenia pectinata, Ceratophyllum demersum*, and *Myriophyllum sibiricum* for example, it was also found in monospecific stands in few CU marshes*. Chara spp*. was also present in a limited amount of natural marshes where it was usually growing among other species (Figure 6, Table 7). Both CU and natural marshes shared most of the diversity which includes, for example, C*. demersum*, *M. sibiricum*, *U. macrorhiza, Moss spp. Lemna minor*, and many species of the *Potamogeton* genus (Figure 6, Table 7).

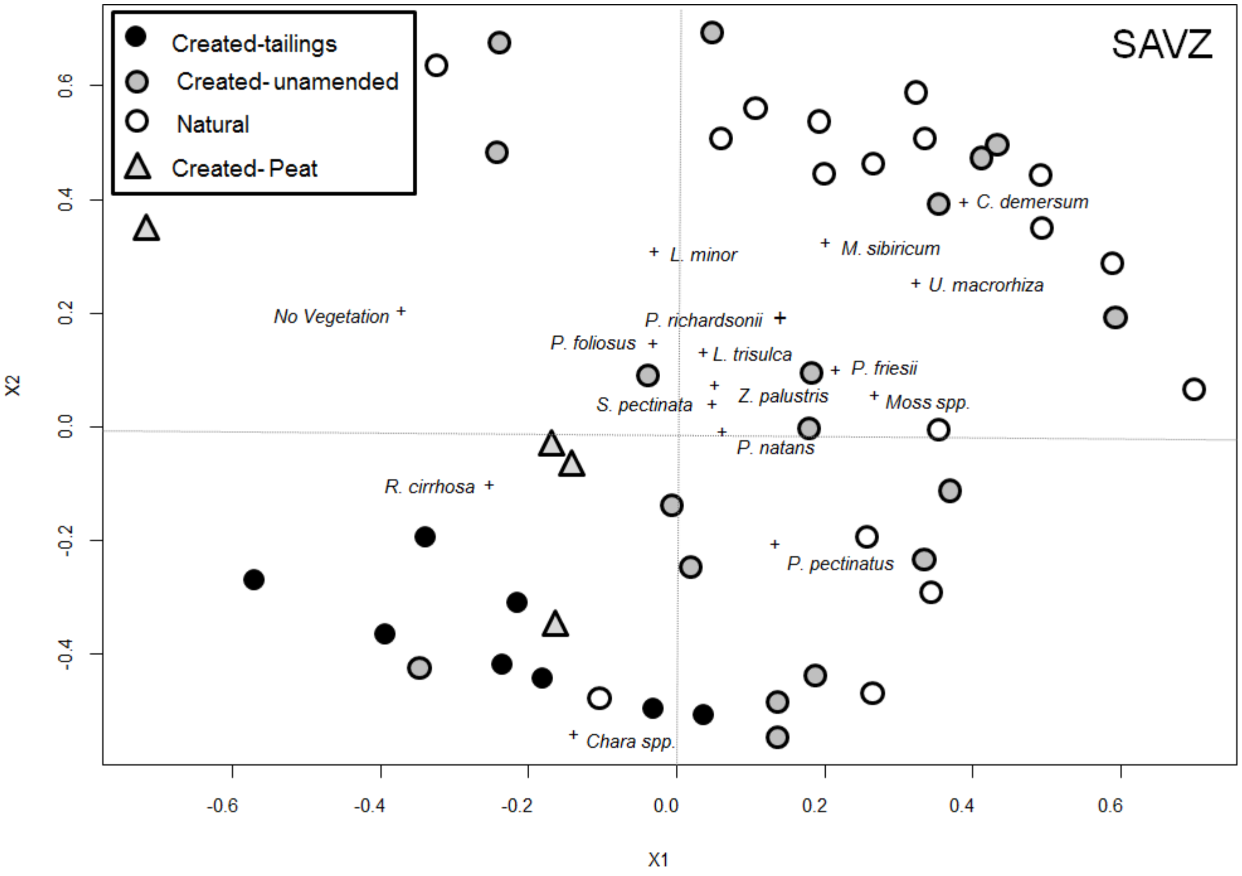

Figure 2- 6: NMDS for the SAVZ where the four marsh types are ordinated in species space. Each point represented a marsh.

Table 2- 7: List of species and their corresponding loading in relation to the NMDS performed (see Figure 6, 7, and 8) for each of the three zones.

SAVZ				
Species	**NMDS1**	**NMDS2**	**r**	**p-value**
P.natans	**1.00**	0.00	0.47	**0.01**
P.foliosus	**0.97**	-0.24	0.50	**<0.001**
P.pectinatus	**0.95**	-0.32	0.14	0.66
P.richardsonii	**0.93**	-0.36	0.48	**<0.01**
Mosses	**0.80**	0.61	0.29	0.19
P.friesii	**0.78**	0.63	0.17	0.60
L.minor	**0.71**	**0.71**	0.60	**<0.001**
C.demersum	0.24	**0.97**	0.64	**<0.001**
M.sibiricum	0.20	**0.98**	0.43	**0.02**
Z.palustris	0.15	**0.99**	0.34	0.07
L.trisulca	0.05	**1.00**	0.39	**0.02**
Chara.spp	-0.11	**-0.99**	0.73	**<0.001**
U.macrorhiza	-0.38	**0.93**	0.32	0.13
U.minor	-0.52	**0.85**	0.09	0.89
S.pectinata	-0.68	**0.73**	0.13	0.74
R.cirrhosa	**-0.99**	-0.16	0.53	**<0.01**
EZ				
Species	**NMDS1**	**NMDS2**	**r**	**p-value**
E.acicularis	**-1.00**	-0.06	0.54	**0.003**
L.parviflora	**-0.99**	-0.16	0.39	0.042
P.natans	**-0.95**	0.31	0.43	**0.019**
E.arvevensis	**-0.93**	0.36	0.10	0.849
E.fluviatile	**-0.93**	0.37	0.31	0.165
Mosses	**-0.86**	-0.51	0.61	**0.002**
B.syzigachne	**-0.81**	-0.59	0.16	0.620
C.aquatilis	**-0.74**	-0.68	0.52	**0.007**
P.pectinatus	-0.39	**0.92**	0.24	0.397
C.canadensis	-0.37	**-0.93**	0.32	0.143
H.vulgaris	-0.31	**-0.95**	0.36	0.081
E.palustre	-0.27	**-0.96**	0.06	0.955
S.tabernaemontani	-0.27	**0.96**	0.24	0.370
G.trifidum	-0.23	**-0.97**	0.62	**0.001**
Chara.spp.	-0.17	**0.98**	0.58	**0.002**
S.suave	0.00	**-1.00**	0.33	0.153
R.cirrhosa	0.08	**1.00**	0.44	**0.023**
M.sibiricum	0.12	**-0.99**	0.38	0.067
S.pectinata	0.14	**-0.99**	0.23	0.401
L.minor	0.36	**-0.93**	0.75	**0.001**

T.latifolia	0.56	**0.83**	0.70	**0.001**
U.macrorhiza	**0.76**	-0.65	0.43	**0.023**
G.grandis	**0.79**	0.61	0.31	0.162
C.utriculata	**0.86**	-0.51	0.62	**0.001**
C.demersum	**0.95**	-0.31	0.47	**0.014**
WMZ				
Species	**NMDS1**	**NMDS2**	**r**	**p-value**
C.canadensis	**1.000**	-0.011	0.542	**0.001**
L.parviflora	**0.993**	-0.117	0.326	0.12
P.arundinacea	**0.964**	0.265	0.218	0.46
C.aquatilis	**0.907**	-0.422	0.200	0.46
E.fluviatile	**0.822**	-0.570	0.369	0.07
P.amphibium	**0.733**	-0.680	0.499	**>0.01**
Mosses	**0.733**	-0.567	0.470	**0.01**
G.trifidum	0.621	**-0.784**	0.820	**0.001**
T.latifolia	0.588	**-0.809**	0.086	0.87
P.palustris	0.586	**0.811**	0.429	**>0.01**
S.canadensis	0.485	**0.874**	0.287	0.18
T.repens	0.439	**0.898**	0.273	0.24
G.lepidota	0.428	**0.904**	0.484	**>0.01**
T.pratense	0.315	**0.949**	0.362	0.067
E.arvense	0.304	**0.953**	0.625	**0.001**
L.minor	0.254	**-0.967**	0.354	0.061
S.suave	0.174	**-0.985**	0.396	**0.03**
C.palustris	0.075	**-0.997**	0.287	0.20
E.repens	-0.047	**0.999**	0.552	**0.001**
L.corniculatus	-0.059	**0.998**	0.584	**0.001**
M.sativa	-0.074	**0.997**	0.427	**0.02**
M.officinalis	-0.155	**0.988**	0.421	**0.01**
S.galericulata	-0.373	**-0.928**	0.311	0.15
P.palustris	-0.401	**0.916**	0.509	**0.001**
F.vesca	-0.439	**0.899**	0.465	**0.01**
C.stricta	-0.495	**-0.869**	0.171	0.70
S.arvensis	-0.498	**0.867**	0.796	**0.001**
H.vulgaris	-0.576	**-0.817**	0.267	0.25
M.alba	-0.633	**0.774**	0.357	0.08
C.utriculata	-0.677	**-0.736**	0.805	**0.001**
R.acicularis	**-0.755**	-0.655	0.290	0.20
E.palustre	**-0.848**	0.530	0.086	0.87
C.angustifolium	**-0.975**	0.224	0.719	**0.001**

Table 2- 8: List of indicator species within each zone (i.e. SAVZ, EZ and WMZ) of CT, CU, CP and natural marshes.

Zones	Indicator species	Marsh types	Specificity[*]	Fidelity[¥]	p-value
SAVZ	*Mosses*	Natural	0.8	0.5	0.03
	Ceratophyllum demersum	CU+Natural	1.0	0.8	<0.01
	Utricularia macrorhiza	CU+Natural	1.0	0.5	<0.01
	Myriophyllum sibiricum	CU+CP+ Natural	1.0	0.7	0.02
EZ	*Schoenoplectus tabernaemontani*	CT	0.9	0.4	0.04
WMZ	*Lotus corniculatus*	CT	0.6	0.6	0.04
	Calamagrostis canadensis	CU+Natural	1.0	0.4	0.04
	Galium trifidum	CP+Natural	0.9	0.6	<0.01
	Scutellaria galericulata	CP+Natural	0.9	0.5	<0.01

[*] Probability that the surveyed marsh belongs to the target marsh type given the fact that the species has been found (Dufrene and Legendre, 1997).

[¥] Probability of finding the species in marshes belonging to the marsh type (Dufrene and Legendre, 1997).

Mosses were a good indicator of natural marshes (Table 8). If *Mosses* were observed in a surveyed marsh, there was a high probability (i.e. Specificity=0.8) that the marsh surveyed was natural. However, *Mosses* were not found in all natural marshes (Fidelity=0.5).The species *C. demersum* and *U. macrorhiza* were both identified as indicator species for natural and CU marshes. Results suggested that both *C. demersum* and *U. macrorhiza* were good indicators of natural and CU marshes because they occured in marshes belonging to these types only (i.e., Specificity= 1.0), although not all marshes belonging to natural and CUclasses included these species (i.e., Fidelity = 0.8 and 0.5 respectively). *M. sibiricum* was identified as a good indicator species of CU, CP and natural marshes because they occur in marshes belonging to these types only (i.e., Specificity= 1.0), although not all marshes belonging to CU, CP and natural included this species (i.e., Fidelity = 0.7).

Although some differences exist, many species of the EZ were shared among the four different types of marshes (Figure 7, Table 7). *Typha latifolia* was a species commonly found in the EZ of many created marshes (regardless of their amendment) and in few natural marshes (Figure7, Table 7). As it was the case for the SAVZ*, Chara spp*. and *R. cirrhosa* tended to be associated with the EZ of most CT marshes. These two submersed aquatic species grew in the water underneath the foliage of the emergent macrophytes. Although few natural marshes shared species with CT marshes (e.g. *Chara spp*., *T. latifolia, Carex aquatilis*) the majority of the natural marshes presented a dissimilar vegetation composition (e.g. *C. utricularia*, *L.minor, M. sibiricum*) from CT marshes. The vegetation composition in the EZ of CU marshes varied among marshes. However, most plant species found in the EZ of CU were also found in natural marshes (e.g.*C. aquatilis*, and *C.utriculata)*. Submersed species were also found in the EZ of natural marshes. Hence, *C. demersum*, *M. sibiricum* and *U. macrorhiza* were associated with the EZ of most natural marshes. CP marshes were dispersed in the ordination space. While three of the CP marshes had *T. latifolia* in their EZ, one CP marsh EZ was dominated by *C. aquatilis*.

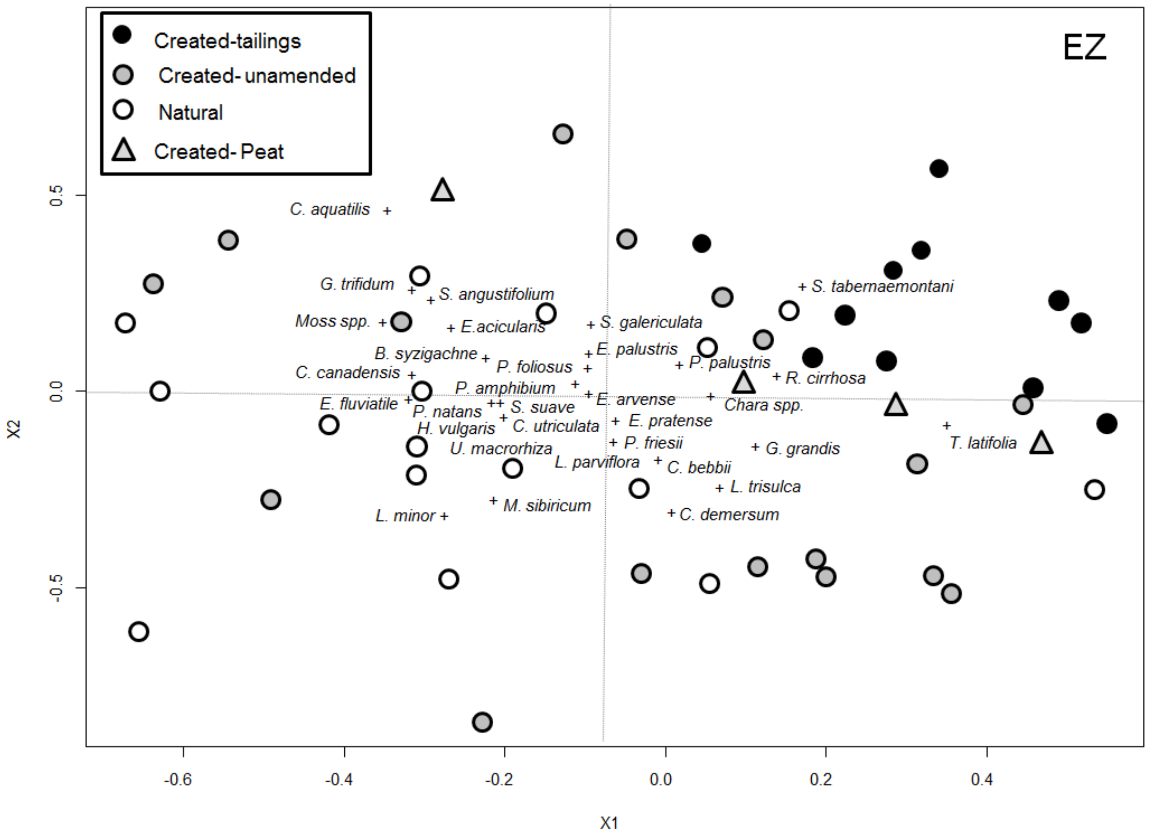

Figure 2- 7: NMDS for the EZ where the four marsh types are ordinated in the species spaces. Each point represented a marsh.

The species *Schoenoplectus tabernaemontani* was identified as a good indicator species for the EZ of CT only (Table 8). If *S. tabernaemontani* was identified in a surveyed marsh, there was a high probability (i.e. Specificity=0.9) that the marsh was a CT marsh. However, the results revealed that the probability of finding the *S. tabernaemontani* in CT marshes is low (Fidelity=0.4).

Upland species such as *Melilotus sp.*, *Sonchus arvensis.* And *Equisetum arvense* were common in the WMZ of most CT marshes and some CP and CU marshes (Figure 8, Table 7). *C. aquatilis* was found in the WMZ of almost all marshes sampled but varied in abundance from one marsh to another. *C. aquatilis* cover was dominant in most natural marshes where other species of *Carex* such as *C. artherodes* and *C. utriculata* were also present but less abundant. *C. aquatilis* and *C. utricularia* were found in the WMZ of CU and CP marshes. In the WMZ of CT, *C. aquatilis* was the only species of *Carex* found. Species such as *Calamagrostis canadensis* and *P. palustris,* were common in most natural marshes and in few CU marshes. *Scutellaria galericulata*, *Galium trifidum, Poa palustris*, and *Calla palustris* are examples of abundant plants species in the WMZ of natural marshes that are uncommon in other types of marshes, although CP and natural marshes shared the presence of *S. galericulata.*

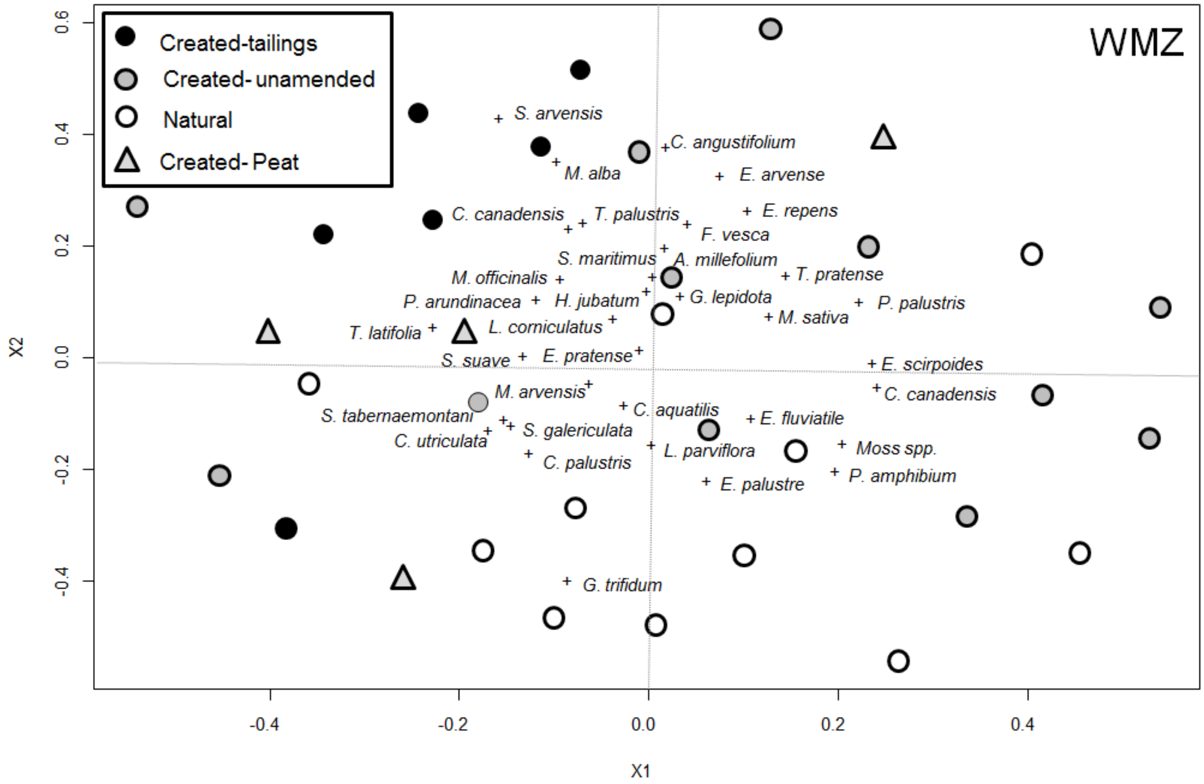

Figure 2- 8: NMDS for the WMZ where the four marsh types are ordinated in the species spaces. Each point represented a marsh.

For the WMZ, *L. corniculatus* was a good indicator species for CT marshes (Specificity=0.6, Fidelity=0.6) (Table 8). The species *C. Canadensis* was a good indicator for CU and natural marshes. Although the probability to find this species in surveyed marshes was relatively low (Fidelity=0.4) *C. Canadensis* was only found in these two types of marshes (Specificity=1.0). The species *G. trifidum* (Specificity=0.9, Fidelity=0.6) and *S. galericulata* (Specificity=0.9, Fidelity=0.5) were almost always found in CP and natural marshes.

The MRPP results for vegetation composition of the SAVZ revealed that CT had a significantly different vegetation composition then CU and natural marshes (with chance-corrected within-group agreement, A= 0.05 and p<0.001) (Table 9). The same results were obtained for the EZ where CT had significantly different vegetation composition from both CU and natural (A=0.07 and p<0.00001) whereas CU and natural are not significantly different. The WMZ of CT and CU marshes were composed of similar species whereas, the MRPP results suggest that both CT and CU marshes were composed of different species than the one characterising natural marshes (A= 0.06 and p=0.001).

Table 2- 9: MRPP results comparing vegetation composition with pair-wise comparisons. A= chance-corrected within-group agreement.

	CT-CU	CT-N	CU-N	A	p-value
SAVZ	0.009	0.001	0.247	0.055651	**>0.001**
EZ	< 0.0001	< 0.0001	0.214	0.066764	**>0.001**
WMZ	0.168	0.002	0.012	0.055651	**0.001**

2.5. Discussion

My objectives were to quantify and compare 1) the abiotic conditions and 2) the vegetation richness and composition of created marshes (i.e. CT, CU, and CP) and natural marshes of the Fort-McMurray region. I demonstrated that CT, CU and natural marshes under study were each characterised by distinct environmental conditions. I have showed that because of their distinct environmental conditions, created marshes maintain different filters on the plant species pool of the region. Generally, when compared to CT marshes, natural and CU marshes had the tendency to be more similar in term of water chemistry. However, differences in sediment chemistry was mostly attributable to the origin (created versus natural) of the marshes. In addition, my results showed that marsh types differed in species richness and composition. When compared to created marshes, the plant species richness of natural marshes tended to be higher in all three zones. Compare to natural and CU marshes, the SAVZ of CT marshes had significantly lower plant richness and vegetation composition that differ. Thus, it was assumed that the addition of OSPW and/or OSPM significantly reduced species richness in the SAVZ and led to a vegetation composition atypical of natural marshes of the region. The EZ and WMZ of CT and CU shared some similarity in term of species richness and composition suggesting that, in addition to the type of amendment, the origin of the marshes influences the vegetation. Thus, the abiotic differences observed among marshes of different origin were hypothesized to influence the vegetation diversity and composition of the EZ and WMZ. Interestingly, CP marshes environmental conditions, and vegetation richness and composition tended to be more similar to CT marshes than other types of marshes.

My study was the first to characterise and compare environmental conditions and vegetation community patterns simultaneously in the three zones (SAVZ, EZ and WMZ) in the three types of created-marsh (CT, CU, and CP). The role of PM additions for reclamation has been a highly contentious and important consideration here to fore (Kovalenko et al. 2012). This study is the first to give insight on the effect of PM amendment on the sediment and water chemistry, and vegetation composition of created marshes. My study provided reclamation tools by identifying important abiotic variables influencing vegetation composition and by providing reclamation benchmarks by which created marshes can be compared and reclamation practices guided and adjusted.

2.5.1. Created marshes proposed atypical abiotic conditions

My study revealed that abiotic characteristics of the created marshes often fall outside the range of conditions observed in natural marshes of the Fort-McMurray region. The relatively flat basin (average basin slope of 3.5^{o}) of natural marshes was characterised by relatively shallow water which gradually decreased in depth from an average of 57 cm in the SAVZ to an average of 4 cm in the WMZ. Natural marshes have wide vegetative zones (average SAVZ length= 66 m, average EZ length= 8 m, and average WMZ length= 9 m) usually interspersed with numerous and deeper beaver-excavated channels that extended out of the marshes. These results supported the description of natural marshes provided by Raab and Bayley (2012).

My results corroborate with the ones of Raab and Bayley (2012) and suggest that constructed WMZ (regardless of their type of amendment) had significantly lower sediment moisture content and lacked the organic-rich hydric soil conditions observed in natural marshes of the region. Hence, when compared to most natural marshes of the studied region, created marshes basin morphology provides atypical conditions for vegetation growth and community development. Interestingly, some wetland associated vegetation (e.g. *C. aqautilis*) did persist and colonise these anomalous conditions; a testament to the reclaiming power of marsh emergent plants. Most created marshes of the Fort-McMurray region, regardless of their types of amendment, tended to have steep basin slopes (average basin slope of 10.2^{o}) almost three times steeper than in natural marshes. These steep slopes compress the vegetative transition from the SAVZ to the WMZ to within couple of meters. The SAVZ of created marshes was characterised by a water depth of approximately 2 m and the SAVZ width (i.e. average of 3 m) was half the width of an average natural marsh SAVZ. In the case of the EZ and WMZ of created marshes, the width of each zone was approximately 1/3 the width of those in natural marshes (EZ and WMZ width of created marshes are on average 2 m and 3 m respectively). Although the size (area) of the created marshes was smaller than natural marshes (a fourteen fold difference on average), the SAVZ water depth can approach 2 m deep within two meters of the marsh shore. Water depth decreased from an average of 0.6m in the SAVZ to 0 m in the WMZ with an average distance of 9 m. In natural marshes with gradual and gentle basin slope, water level fluctuation can flood or drawdown a wide area around the marsh. Hence, I postulate that the steep basin slope of most created marshes reduces the spatial effect of water level fluctuation.

As expected, water conductivity/salinity was higher in CT than in CU and natural marshes (i.e. salinity was five times higher and conductivity more than 4 times higher). Surprisingly, three out of four of the CP marshes I studied fell within the same range of water salinity (i.e. between 0.6 to 0.8 ppt) and conductivity (i.e. between 1621 and 2108 mg/L) as the CT marshes. Although separated by an average of 2 km, these three CP marshes were all situated in the same geographic area (i.e. south of Syncrude Canada Ltd. on both sides of Highway 63). The study of aerial photo archives reveals that three out of four of these marshes were reclaimed on top of a refilled open-mined pit. The four marshes were built on top of a sandy material that may have been of marine origin. Indeed, most of the bitumen resources in the Fort McMurray region are contained in fluvial, estuarine and marginal marine sediments which contain elevated soluble salt levels (Hein et al. 2006). Through bitumen extraction, this sediment was exposed to the surface, removed and re-incorporated into the reconstructed landscape (Harris 2007). Furthermore, the PM amended to created marshes is usually stored as on-site stockpiles for an undetermined amount of time before being used (Harris 2007). Thus, the PM is exposed to weather and other conditions for many years before being used. The dewatering of the stockpiles over time due to evaporation concentrate the amount of salt found in the PM. I propose that the elevated salinity observed in CP marshes was attributable to the use of marine sediment and the PM amendment.

As noted by Trites and Bayley (2009), Slama (2010), and Rooney and Bayley (2012) water dissolved oxygen (DO) in CT marshes tended to be slightly higher than in other created and natural marshes. Water DO varies daily (i.e. is lowest before dawn and peaks in mid-afternoon) and is influenced by a wide variety of factors such as temperature, water salinity, photosynthesis, respiration, nitrification and chemical oxidation (Dauer et al. 1992; Mitch and Gosselink 2011). Water temperature was not significantly different among marshes and hence cannot explain the DO differences observed. Oxygen concentration is usually negatively correlated with salinity and one would expect to observe this relationship in CT marshes. The amount of DO, however, is not solely attributable to its solubility in water. Water DO is also positively correlated to water transparency (i.e. light) and photosynthesis (Willis and Ester 2004; Spalding and Ester 2007). However, Rooney (2011) observed a wide range of water transparency among CT marshes and did not found a significant difference among marsh types. Thus the transparency of the water and its effect on plant photosynthesis cannot support the observed difference in DO. Decomposition and respiration consume DO in water and sediment (Stratford et al. 2004). Trites and Bayley (2009) demonstrated that decomposition rate in marsh affected by OSPW and/or OSPM were not found different from natural marshes. Barr (2009) and Cooper (2004) have found that when exposed to OSPM/OSPW plant colonisation rate and zoobenthic community development were impeded. As explained above, photosynthesis and respiration both influence DO (i.e. photosynthesis increases DO, respiration decreases DO). Furthermore, Gardner (2010) found that the sediment of CT marshes had less microbial activity (microbes consume DO) but more chemical oxidation (use of DO) than CU and natural marshes. Hence, it remains difficult to associate the slightly higher water DO measured in CT marshes to a specific cause.

2.5.2. Composition of the SAVZ

Regardless of the marsh type, the SAVZ was the least diverse of all three zones. My results supported the ones of Titres and Bayley (2009) and Rooney and Bayley (2011), and suggested that the vegetation composition in the SAVZ of reference natural marshes was characterised by a relatively low number of plant species (i.e. mean richness was approximately equal to 5 species) where Mosses, *C. demersum*, *M. sibiricum*, *U. macrorhiza*, and *Chara spp.* for example may coexist. The vegetation composition difference observed in the SAVZ was mainly related to the type of amendment (CT, CU and CP) rather than due to the origin (created versus natural) of the marshes. Marshes that received OSPM/OSPW and PM had significantly higher water salinity. The SAVZ of created marshes that had higher water salinity tended to be characterised by lower plant richness and vegetation composition that differed from CU and natural marshes. In the majority of the CT and CP marshes, *Chara spp*. or *Ruppia cirrhosa* were dominant and found in monospecific stand. C*hara spp*. and *R. cirrhosa* tolerance to salinity (Verhoeven 1979; James et al. 2003) may explain their dominance over less salt-tolerant plants species (e.g. *U. macrorhiza*) in the SAVZ of CT marshes. My results corroborate the ones of Rooney and Bayley (2011) who found that *Chara spp*. were associated with oil sands created marshes that were characterised by alkaline, deep, and low nutrients waters. The addition of PM to created marshes does not seem to positively influence their submersed vegetation richness and composition. Barr (2009) also found that the addition of PM to created marshes did not improve the performance of either invertebrate community or living submersed aquatic vegetation.

In four CT marshes and one CP marshes, the SAVZ sampled did not contain any vegetation. Rooney and Bayley (2011) hypothesised that, in addition to the plant-incompatible chemistry and environment offer by CT water and sediment, the lack of vegetation may be due to geographic isolation from potential colonisation sources. On the three CT marshes sampled in my experiments that lacked vegetation, nearby marshes containing potential sources of submerged and floating vegetation seeds and propagules were located less than 100 meters away. The study of aerial photograph archives confirms that these nearby marshes had been present for as long as 15 years in some cases. The common presence and transfer by wind, muskrats and water birds would seem to have distributed propagules. Hence for the SAVZ, I believe that geographic isolation may reduce the dispersal of propagule in some instances, but I agree with Rooney and Bayley (2011) who denoted that the low water and sediment nutrient content, and the high salinity level in CT may prevent the development of submersed vegetation. In a prototype experiment conducted in 2011 (unpublished data) I have demonstrated that when PM was provided as sediment, *M. sibiricum* biomass was not significantly different (after two growing seasons) in OSPW or freshwater treatments. In the same experiment, *C. demersum,* a submersed plant species not producing roots, did not survive in any of the water treatments. These results support the finding of Rooney and Bayley (2011) that the low nutrient availability in water can be limiting to submerged plants species that lack a root system and thus the ability to uptake nutrients from the sediment. One CP marsh did not contain any submerged vegetation but this observation was associated mainly with the low water depth in this marsh which was less than 2 cm deep due to a recent drawdown.

The richness and composition of the SAVZ of CU marshes was similar to the one of natural marshes. Hence, *C. demersum*, *M. sibiricum*, and *U. macrorhiza* were also good indicator species of CU marshes. Unlike CT marshes, CU marshes were also characterised by the presence of floating plant species such as *Lemna minor* and *Potamogeton sp.* However, unlike natural marshes, CU marshes were characterised by a higher frequency of occurrence and a higher abundance of *Chara spp.*

CP high similarity with CT marshes' SAVZ vegetation composition illustrates the sensitivity of submersed aquatic vegetation to water quality (i.e. salinity). However, the effect of salinity on the biota subjected to OSPW and/or OSPM has been shown difficult to dissociate from the effect of NAs, a chemical also present in OSPW and OSPM that influences plant function in that laboratory (Kamaluddin and Zwiazek 2002). In my study, CT and two CP marshes had similar salinities but were different in terms of NA content (Daly and Ciborowski 2008). Because the two types of marshes exhibited very similar vegetation composition, physical condition and water salinity levels, I postulate that NAs, when compared to water salinity, may play only a minor role in shaping vegetation community composition in the created marshes of the Fort McMurray region. However, two out of four of the CP marshes contained high NAs (Daly and Ciborowski 2008) which may also explain the variation of vegetation composition among CP marshes. The effect of NAs on plant community composition remains to be tested.

Thus, although the origin of a marsh (natural versus created) seems to influence the composition of the SAVZ vegetation community, my study suggested that water quality, such as salinity and nutrients, certainly play a major role in shaping the plant community richness and composition in the SAVZ.

2.5.3. Composition of the EZ and WMZ

Although the species richness of the EZ of CT and CU marshes was significantly lower than in the natural marshes of the region, my results suggested that neither the sediment nor the water amendment were the main factors shaping vegetation compositions in the EZ and WMZ of marshes. Indeed, the EZ vegetation composition tended to be similar in the three types of created marshes but diverged from the community observed in natural marshes. Instead, the results suggested that the marshes' origins (i.e. created versus natural) and physical conditions (i.e. basin slope, water level, zone width) are the primary drivers of community composition in these two zones. The morphology of the EZ characterising the created marshes differs from natural marshes. The steep basin slopes lead to reduced EZ width which is typically approximately one meter wide. Regardless of created marsh amendment types, *S. tabernaemontani* and *T. latifolia* were found more frequently and in higher abundance in the EZ of created marshes than in natural marshes. In other studies, *T.latifolia* abundance has been associated with steep marsh basin slopes resulting in restricted littoral zones with stable water level (Rumble et al. 1985; Zampella and Laidig 2003; Wilcox 2011).

My results were different from the one of Raab and Bayley (2012) who found no difference between the vegetation composition of EZ and WMZ. Raab and Bayley (2012) have selected and assessed natural marshes different from the ones selected for my study. While the natural marshes selected for this study were all located in the Fort McMurray regioin, their study assessed were located across the Boreal Plain region of Alberta and Saskatchewan. My results suggested that although the vegetation composition of the EZ and WMZ of natural marshes tend to be similar, the vegetation composition of EZ and WMZ of created marshes tended to be dissimilar. This can be explained by the abrupt sediment moisture gradient caused by the steep basin slopes which transition from deep water levels with saturated sediment in which emergent plants (i.e. *Carex sp*. *T. latifolia*, *S. tabernaemontani*) grow to a drier wet-meadow substrates dominated mostly by non-obligate marsh plants (e.g. *Lotus corniculatus*, *Sonchus sp, Melilotus sp*.). In natural marsh EZ vegetation composition was often very similar to the WMZ with only minor differences in species occurrence. This phenomenon may be associated with the gentle basin slope, the shallow water level gradual transition from the emergent to the WMZ. Hence, both the EZ and WMZ of every surveyed natural marsh were dominated by *Carex sp.* The differences between the two vegetative zones results from the less abundant species growing among the *Carex sp.*. For instance, in the WMZ of natural marshes, species such as *S. galericulata*, *G. trifidium*, *Mentha arvensis*, and grasses (e.g. *Poa palustris*, *C. canadensis*) grow in water levels near zero (but in water-saturated soil) on the outside edge of the WMZ. Those species are replaced in wetter conditions (i.e. closer to the SAVZ) by species such as *Polygonum amphibium*, *Hippuris vulgaris*, and *Arum trifidium*. Thus, the WMZ vegetation composition differences do not represent the *a priori* marsh soil and water treatment categorisation. Instead, the differences reflect the origin (created or natural) and the associated physical conditions (basin slope, water level, zone width). Sediment and water chemistry may not influence vegetation composition as strongly as the physical conditions; hence, their influence on emergent and wet-meadow vegetation is not apparent in my results.

It has been speculated that the young age of created marshes relative to natural reference marshes may partly explain their lower biodiversity. However, Rooney (2011) found no significant relationship between the age of created marshes and vegetation richness.

2.5.4. Peat-mineral mix enrichment in created marshes

The positive effect of PM amendment in newly restored marshes has been mostly demonstrated on soil properties and ecosystem functions but remained to be determined for ecosystem plant diversity and composition (but see Cooper 2004) (Sutton-Grier et al. 2009; Ballantine et al. 2012 but see Kovalenko et al. 2013). In my study, conclusions regarding the benefit of amending marshes with PM are limited due to the small number of marshes in the post-mined landscape and the wide environmental heterogeneity characterising this group. Although CP marsh substrates were amended with PM, their vegetation composition shared more similarity with created marshes than with natural marshes of the Fort McMurray region. However, the presence in CP marshes of species (e.g. *M.sibiricum* in the SAVZ, S. *galericulata*, and *G. trifidium* in the WMZ) typical of natural marshes but absent in CT and or CU marshes suggest that the amendment of PM to the WMZ of created marshes may have some positive effects. My result support Cooper (2004) and suggested that the higher water salinity may overpower the potential beneficial effect of PM in the SAVZ.

Another point needs to be raised on the potential effect of PM amendment in created marshes of the oil sands post-mined landscape. In some cases, created marshes that were amended with PM received from 30 to 100 cm thick of PM. The benefits (e.g. water retention, temperature regulation, increase nutrients availability) of amending created marshes with that thickness of PM remains to be determined. Furthermore, amending marshes with PM is environmentally and economically expensive. PM is in limited in abundance and expensive due to costs of excavation, transportation, storage and re-localisation for reclamation purposes. In addition, PM is environmentally costly because once it is excavated, PM is subjected to a period of rapid decomposition (oxidation) which leads to CO_2 release in the atmosphere (Bruland and Richardson, 2006) and a loss of material volume for use. Thus, the next step to improve the knowledge on the role of PM amendment in newly created marshes may be to test the effect of a larger gradient of PM depth/amount to determine the optimal amount of PM needed to optimise vegetation richness and diversity.

In addition, the provenance of the PM may also be an important factor influencing vegetation composition. The PM used for restoration purposes in the oil sands of Alberta is typically harvested from on site stockpiles (Harris 2007). The stockpiles result from the storage of a mixture of various types of top soil removed during surface mining and may be stored and exposed to all sort of conditions for many years before being used (Harris 2007). The composition of the stockpile is unknown and the quantity of PM, the quantity, quality and nature of seeds or propagules that can be introduced when amending created marshes is also unknown.

2.5.5. Vegetation composition and functions

My results support the ones of Kovalenko et al. (2012), and suggested that created marshes will not provide processes and functions equivalent to the one characterising the natural marshes of the region. Differences in vegetation richness and composition, as observed in the studied created marshes, have been proved to result into different ecosystem functions (e.g. Tilman 1997; Ibekwe 2007; Ruiz-Rueda 2009). For example, in the SAVZ of the CT marshes *Chara spp.* cover the sediment with a dense and thick monospecific mat, which extend only a few cm into the water column. In natural marshes, although *Chara spp*. may be present, the water column was occupied from bottom to top by additional species such as *C. demersum*, *M.sibiricum* and *U. macrorhiza*. These differences in species composition between created and natural marshes are believed to confer to aquatic organisms, such as insects, a very different environment (St-Pierre and Kovalenko 2014). While natural marshes offer a complex structure that may increase and diversify potential for niches to other organisms, CT marshes offer less vertical heterogeneity and structural complexity. Differences in submerged vegetation composition may also result in differences in productivity, biochemical cycles, and the availability of shelter and food source for fish and waterfowl (Carpenter and Lodge 1986; Engelhardt and Ritchie 2002**;** Matthew and Sharfstein 2009). Submerged aquatic vegetation functions are diverse. Submerged aquatic vegetation reduces marsh water vertical temperature gradient or magnify it if the

SAVZ beds are dense (Dale and Gillespie 1977), reduce water flux (Weiler 1978), act as a sieve retaining coarse particulate detritus, and decompose organic matter to a greater extent than in unvegetated reaches of a water body (Dawson 1980). Based on my results, I can assume that marshes with lower richness and composition, such as CT marshes, will confer compositional and structural characteristics to the reclaimed landscape, and hence functions different from the ones characterising natural marshes of the region.

Although *T. latifolia* was present in some of the natural marshes surveyed, its presence in the EZ of some created was more frequent and more abundant (i.e. greater percentage cover). The dominance of *T. latifolia* in the EZ of some created marshes produces a canopy structure that contrasts drastically with the EZ of most natural marshes, which are dominated by *Carex spp.* Although both species produce vertical leaves, *T. latifolia* stand height was much higher than *Carex spp.* stands. Relationships between canopy structure and ecosystems functions and processes have been investigated for a long time (Monsi et al. 1973; Hirtreiter and Potts 2012). In some instances where a plant produces a dense canopy, its structure may act to reinforce its dominance through the more efficient or more complete uptake of nutrients or light resources (Zedler and Kercher 2004). This ability to uptake more light through a denser canopy in turn influence patterns of photosynthetic capacity and nitrogen partitioning within the canopy (Hirose and Werger *1994;* Hirtreiter and Potts 2012). Hence, the vegetation composition differences between created and natural marshes may result in community function differences.

The absence of conditions favoring the establishment of obligate marsh plants in the WMZ of many created marshes also raises some concerns relating to marsh functions. The WMZ is the area in a marsh characterised by successive periods of aerobic and anaerobic conditions due to periodic drawdown and flooding events (Wisheu and Keddy 1992). The aerobic conditions and high nutrient content of the WMZ of marshes coupled with the periodical anaerobic conditions reducing decomposition result in the accumulation of organic matter. Extensive WMZ are also known to provide buffer zones protecting waters and increase its quality by sequestrating pollutants and reducing erosion. Thus, the absence of an extensive WMZ in most created marshes may influence marsh functions and their related services.

2.6. Study Limitations

Although the comparison of the sediment chemistry among created and natural marshes provide conclusions similar to those of Raab and Bayley (2012), my results present are somewhat limited due to the low degree of replication and the potential for spatial autocorrelation. The CT marshes sampled were located near to one another, and the samples may not be independent (i.e. may pseudoreplicate) of each other.

Some of the created marshes included in my study were designed to mimic the process of end-pit lakes rather than marshlands. Consequently, these systems are not meaningful models of current reclamation practice. However, I included them in my study to increase the sample size of CT marshes assessed and to better understand the influence of OSPM/OSPW and effects of basin morphometry on vegetation richness and composition.

The four CP marshes present a wide range of conditions (Daly and Ciborowski 2008) and conclusions about the effect of PM on vegetation are thus limited. Two CP marshes had elevated levels of NAs due to groundwater upwelling (Daly and Ciborowski 2008). One of them had high salinity and NAs due to upwelling of water through a berm that bounded a tailings pond. The other marsh was situated on top of a lean oil sands deposit, and received NAs as well as saline groundwater. The two other CP marshes did not contain NAs, and their sediment and water salinity were derived from the sodic overburden layer used to reclaim the landscape in which they are located (Daly and Ciborowski 2008). The effect of NAs on plant community composition remains to be tested.

Studies comparing the vegetation composition between created and natural marshes are comparing relatively young created marshes to older natural marshes. Although there is currently no other alternative to compare vegetation composition among marsh types, our conclusions propose some bias because we are comparing primary succession using models of secondary succession.

2.7. Recommendations

Precise reclamation objectives need to be set for marshes constructed by the oil sands of Alberta. My findings are consistent with those of Cooper (2004), Trites and Bayley (2009), Slama (2010), Rooney and Bayley (2011), and Raab and Bayley (2013), and suggest that vegetation composition in created marshes is impaired and that there is a pressing need to set precise benchmarks to determine acceptable reclamation endpoints. Thus, the next step is to identify valued and necessary functions and services that must be provided by the reclamation efforts.

The direct or indirect addition of OSPM/OSPW to created marshes should be avoided when possible. The low nutrient content, the high salinity and possibly the NA content of these materials, negatively influenced vegetation richness and composition. The presence of OSPM/OSPW in created marshes will likely lead to a vegetation composition atypical of the natural marshes of the region.

The addition of a large amount of saline PM in the SAVZ of created marshes is not recommended. The high salinity content of PM may be detrimental to SAV richness and composition. In addition, there was no clear evidence that PM increased the nutrient content of the sediment. Thus, if the objective is to reclaim a vegetation composition as similar as possible as the one of the natural marshes of the region, I recommend that CU marshes are the best viable reclamation option.

Created marsh basin morphometry should be similar to that of natural marshes of the Fort McMurray region. Marsh water level fluctuation in space and time has been identified as the primary condition shaping vegetation composition, structure and functions (Wisheu and Keddy 1992; Mitsch and Gosselink 2011). Hence, constructed marsh basin slope should be gentle (i.e. near zero degrees) to maximise the extent of the zones' width. My results corroborate those of Raab and Bayley (2012), and suggest that the lack of hydric conditions in the WMZ may result in lower abundance of marsh obligate species which are replaced by non-obligate and sometimes invasive plant species. Gentle basin slopes and extensive zone width can consequently be subjected to hydrological processes similar to those observed in natural marshes. Ensuring the establishment of hydrological processes similar to natural marshes of the region would enhance the conditions favorable to desirable marsh vegetation and their associated functions and services. However, no model quantifying the hydrological processes (i.e. amplitude and frequency of water level fluctuation) of natural marshes of the Fort McMurray region is available. Such a model could provide benchmarks against which created marshes hydrological processes could be compared. The objective of Chapter Four is to quantify and compare hydrological processes of created and natural marshes.

The effect of PM amendment on the WMZ remains unclear. Species typical of the natural marshes but absent in every CT and CU marshes (*G.trifidum, S. galericulata*) were present in two CP marshes. However, the number of CP marshes was limited and further investigation is needed to corroborate my results.

Planting desirable species early in the reclamation process may enhance the vegetation composition of created marshes; although more research have to be done to understand the role of biotic processes (i.e. competition, facilitation, priority effect, etc.) in the development of marsh vegetation.

To provide further insights and guidance to the reclamation efforts on the oil sands of Alberta, research that investigates the role of stochasticity, and spatial and temporal factors (e.g. marsh isolation, connection etc.) in shaping marshes vegetation communities is needed. By identifying which filters are influencing vegetation composition at different spatial and temporal scales, reclamation efforts will better manipulate these variables and reach the desired targets.

Comparing relatively young created marshes to older natural marshes has limitations because we are comparing primary succession using models of secondary succession. The origin and early development of natural marshes of the Fort McMurray region are poorly understood. To better compare the primary succession of created marshes, future research must aim to understand the early development of natural marshes in this region.

2.8. Conclusions

I have demonstrated that the sediment and water chemistry, as well as the physical characteristics of created marshes influenced the vegetation richness and composition of the SAVZ, EZ and WMZ. By identifying which and how abiotic conditions influence plant community composition, I gave insights on the potential landscape that may emerge from the reclamation efforts under specific reclamation conditions. I have provided reclamation benchmarks and recommendations to improve the success of marsh reclamation in the oil sands post-mined landscape of Alberta.

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Appendix B

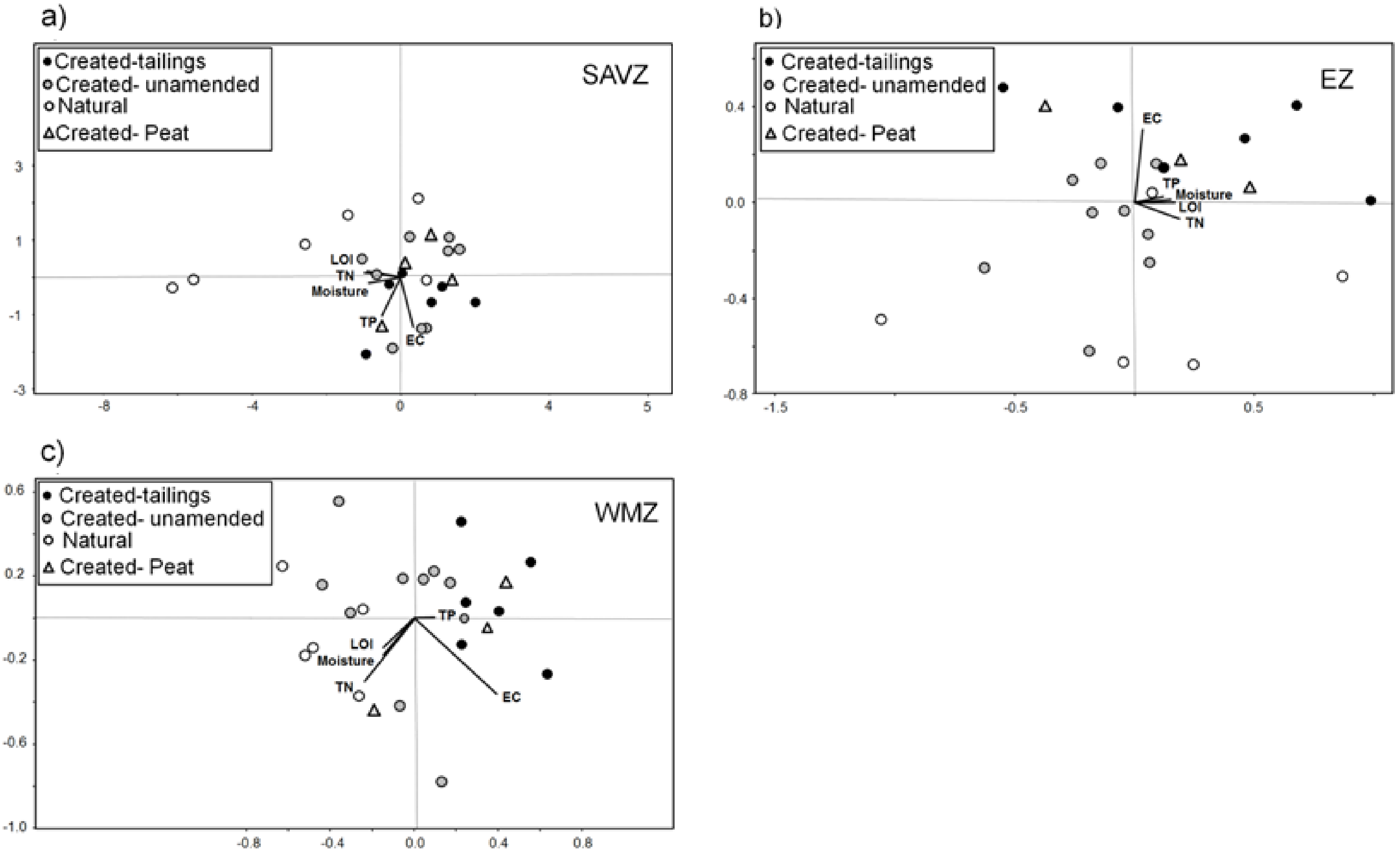

Figure B-1: Ordinations of the three marsh types based on their sediment variables measured at the zone level where **a)** is SAVZ, **b)** EZ, and **c)** WMZ.

Table B-1: Summary for each of the three PCA analyses using the sediment chemistry, averaged at the zone level.

	Variance explained (%)		
Zone level	PC1	PC2	Cumulative
Sediment chemistry			
SAVZ	48.8	32.6	81.4
EZ	64.4	23.4	87.8
WMZ	48.4	37.9	86.2

Table B-2: Summary for each of the three PCA analyses using the sediment chemistry, averaged at the zone level.

Sediment variables (zone level)		
SAVZ	**PC1**	**PC2**
EC	0.13	0.96
TP	-0.43	-0.08
Moisture	-0.50	0.1
LOI	-0.51	0.23
TN	-0.53	-0.01
EZ	**PC1**	**PC2**
EC	0.08	-0.98
TP	-0.42	-0.17
TN	-0.52	0.01
Moisture	-0.50	0.06
LOI	-0.54	-0.08
WMZ	**PC1**	**PC2**
EC	0.1	-0.91
TP	-0.39	-0.4
TN	-0.52	0.04
Moisture	-0.53	0.04
LOI	-0.54	0.04

Table B-3: Full list of plant species identified in created (created-peat (CP), created-tailings (CT), created-unamended (CU)) and natural (N) marshes. Certain species were not identified beyond genus-level. The USDA indicator status was provided for each species(X= present, 0=absent, OBL= obligate wetland, FACW= facultative wetland, FAC= facultative, FACU= facultative upland, UPL= obligate upland, TBD= to be determined, NA= non-applicable).

Species names	USDA indicator status	CP	CT	CU	N
Achillea millefolium	FACU	0	X	X	0
Achillea sibirica	UPL	0	0	X	0
Algea spp	FAC	0	X	X	0
Alisma plantago aquatica	OBL	0	0	X	0
Beckmannia syzigachne	OBL	0	0	X	X
Betula spp	NA	0	X	0	0
Bidens cernua	OBL	0	0	X	X
Brassica spp	NA	0	0	0	X
Bromus ciliatis	FACW	0	0	X	0
Calamagrostis canadensis	FACW	0	0	X	X
Calamagrostis stricta	FACW	0	0	0	X
Calla palustris	OBL	0	0	0	X
Caltha natans	OBL	0	0	0	X
Carex aquatilis	OBL	X	X	X	X
Carex bebbii	OBL	0	0	X	X
Carex utriculata	OBL	X	0	X	X
Carex aquatilis	OBL	X	0	0	0
Carex atherodes	OBL	0	0	X	X
Carex bebbii	OBL	0	0	X	X
Carex crawfordii	FAC	0	0	X	X
Carex diandra	OBL	0	0	X	X
Carex lanuginosa	OBL	0	0	X	X
Carex pseudo-cyperus	OBL	0	0	X	X
Carex sartwellii	FACW	0	0	X	0
Carex utriculata	OBL	0	0	X	X
Castilleja raupii	FAC	0	X	0	0
Ceratophyllum demersum	OBL	X	X	X	X
Chamerion angustifolium	FAC	X	X	X	X
chara spp	OBL	X	X	X	X
Chenopodiacea spp.	FAC	0	0	X	0
Chenopodium album	FAC	0	0	X	0
Chenopodium rubrum	OBL	0	0	X	0

Cicuta bulbifera	OBL	0	0	X	X
Cirsium arvense	FACU	0	0	X	0
Conyza canadensis	FAC	0	X	X	0
Corallorhiza trifida	FACW	0	X	X	0
Cornus canadensis	FACW	0	0	X	0
Deschampsia caespitosa	FACW	0	0	X	0
Eleocharis acicularis	OBL	X	X	X	X
Eleocharis palustris	OBL	0	0	X	X
Elymus repens	FACU	0	X	X	0
Epilobium palustre	OBL	0	0	X	0
Equisetum arvense	FAC	X	X	X	X
Equisetum fluviatile	OBL	0	0	X	X
Equisetum palustre	FAC	0	0	X	0
Equisetum pratense	FAC	0	0	X	X
Equisetum scirpoides	FAC	0	0	X	X
Eurybia conspicua	FAC	X	0	X	0
Fragaria vesca	NI	0	X	X	X
Galium trifidum	FACW	X	0	X	X
Gaultheria hispidula	FACW	0	0	X	X
Geum aleppicum	FAC	0	0	0	X
Glyceria grandis	NI	0	0	X	X
Glyceria striata	OBL	0	0	X	X
Glycyrrhiza lepidota	FACU	0	X	X	X
Hieracium umbellatum	NA	0	0	0	0
Hippuris vulgaris	OBL	X	0	X	X
Hordeum jubatum	FAC	X	X	0	0
Impatiens capensis	FACW	0	0	0	X
Juncus articus	OBL	0	0	X	X
Juncus balticus	OBL	0	0	X	X
Juncus nodosus	OBL	X	0	X	0
Ledum groenlandicum	OBL	0	0	X	0
Lemna minor	OBL	0	0	X	X
Lemna trisulca	OBL	0	0	X	X
Lemna turionifera	OBL	0	0	0	X
Lilium philadelphicum	FACU	0	X	0	0
Lotus corniculatus	FAC	0	X	X	0
Luzula palustris	FAC	0	0	0	X
Luzula parviflora	FAC	0	0	X	X
Lycopus asper	OBL	0	0	0	X
Matricaria spp.	NA	0	X	0	0
Medicago sativa	UPL	X	X	X	X
Melilotus alba	FACU	X	X	X	0
Melilotus officinalis	FACU	X	X	0	0

Mentha arvensis	FACW	X	0	X	X
Moss spp	NA	0	0	X	X
Myriophyllum sibiricum	OBL	X	X	X	X
Myriophyllum verticillatum	OBL	0	0	0	X
Nuphar lutea	OBL	0	0	0	X
Panicum capillare	FAC	0	0	0	X
Parnassia palustris	OBL	0	X	X	0
Petasites sagittatus	FACW	0	0	X	X
Phalaris arundinacea	FACW	0	0	X	X
Phleum pratense	FACU	0	0	0	0
Phragmites australis	FACW	0	0	0	0
Picea spp	NA	0	X	0	X
Platanthera hyperborea	TBD	0	0	X	X
Poa palustris	FAC	X	X	X	X
Polygonum amphibium	OBL	0	0	X	X
polygonum lapathifolium	FACW	0	0	X	X
Populus balsamifera	FACW	0	0	0	X
Populus tremuloides	FACU	0	0	0	X
potamogeton foliosus	OBL	0	0	X	X
Potamogeton friesii	OBL	0	0	0	X
Potamogeton natans	OBL	0	0	X	X
Potamogeton nodosus	OBL	0	0	0	X
Potamogeton pectinatus	OBL	0	X	X	X
Potamogeton pusillus	OBL	0	0	0	X
Potamogeton richardsonii	OBL	0	0	X	X
Potamogeton zosteriformis	OBL	0	0	0	X
Potentilla arguta	FAC	0	0	X	X
Potentilla palustris	OBL	0	0	0	X
Ranunculus abortivus	FACW	0	0	0	X
Ranunculus gmelinii	OBL	0	0	X	X
Ranunculus hyperboreus	FACW	0	0	0	X
Rhinanthus borealis	FACU	0	0	X	X
Rhinanthus minor	FACU	X	0	0	0
Rorippa islandica	OBL	0	0	X	0
Rosa acicularis	FACU	0	0	X	X
Rubus chamaemorus	FACU	0	0	0	X
Rubus idaeus	FACU	0	0	0	X
Rumex occidentalis	OBL	X	0	0	0
Ruppia cirrhosa	OBL	0	X	X	0
Sagittaria cuneata	OBL	0	0	X	0
Salix spp	NA	0	X	X	X
Schoenoplectus tabernaemontani	OBL	X	X	X	0
Scirpus cyperinus	OBL	0	X	X	X

Scirpus microcarpus	OBL	0	0	X	0
Scirpus paludosus	OBL	0	0	X	X
Scirpus pungens	OBL	0	0	X	0
Scirpus spp. (validus + acutus)	OBL	0	0	X	0
Scutellaria galericulata	OBL	X	0	X	X
Sium suave	OBL	X	0	X	X
Solidago canadensis	FACU	0	0	X	0
Sonchus arvensis	FAC	X	X	X	X
Sonchus uliginosus	FAC	0	0	X	0
Sparganium angustifolium	OBL	0	0	X	X
Spirodela polyrhiza	OBL	0	0	0	X
Stellaria longipes	FACW	0	0	0	X
Stuckenia pectinata	OBL	X	X	X	0
Symphyotrichum ciliolatum	UPL	0	X	X	X
Trifolium hybridicum	FAC	0	0	X	0
Trifolium pratense	FAC	0	X	X	X
Trifolium repens	FAC	0	X	X	0
Triglochin palustris	OBL	X	X	X	0
Typha latifolia	OBL	X	X	X	X
Urtica dioica	FAC	0	0	0	X
Utricularia macrorhiza	OBL	0	0	X	X
Utricularia minor	OBL	0	0	0	X
Vaccinuum oxycoccus	OBL	0	0	X	0
Vicia americana	FAC	0	0	X	0
Zannichellia palustris	OBL	0	0	X	X

CHAPTER 3

Title: Do peat amendments to oil sands wet sediments affect *Carex aquatilis* biomass for reclamation success?[12]

3.1. Introduction

In the Fort McMurray region of northern Alberta, Canada, the surface mineable area available to the oil sands industry is approximately 4800 km^2 (Government of Alberta, 2013). Currently 1670 km^2 of the mineable surface has either been mined or approved for development within the next decades (Government of Alberta, 2013). The oil sands region of Alberta is located in the north-west part of the boreal region of Canada (see Brandt (2009) for an overview of the North American boreal zone). The un-disturbed land of this region is formed by a mosaic of uplands (varying from aspen-dominated deciduous forest to spruce–fir–pine-dominated conifer forests) and wetlands communities (bogs, fens, marshes) adapted to long, cool winter and short, cold summers (Vitt and Bhatti 2012). Of the mineable landscape, 63% originally supported wooded fen vegetation and 3% supported marsh vegetation (Raine et al., 2002; Rooney et al., 2012). Despite the low abundance of marshes relative to fens and bogs in this area, a marsh community has been identified as the most achievable endpoint for the reclamation of wetlands under the challenging conditions of the post-mined landscape (e.g. mineral substrate, elevated water and sediment salinity, residual petroleum) (Purdy et al. 2005).

Studies of plant primary succession of sand extraction pits in Siberia, (Koronatova and Milyaeva 2011) and insect occupancy of quarries in Central Europe (Heneberg et al. 2013) showed distinct succession patterns and rapid natural contributions to on-site species richness. They make a convincing argument that post-mining sites respond to

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catenal, abiotic, and annual weather patterns to supplement regional species diversity. Similar processes are likely at work in the oil sands post-mined landscape, albeit at lower levels due to the continuously increasing distance to natural source areas and the post-mined physical and chemical conditions. Goals of maximising diversity are valuable but secondary to rapidity of establishing the stabilising influence of plant cover in the post-mined landscape. Although there are missed opportunities in the boreal forest to allow primary succession in mine-disturbed areas to serve as hotspots of plant and insect diversity (Kareiva and Marvier 2003; Koronatova and Milyaeva 2011; Heneberg et al. 2013), Alberta's reclamation policy and guiding regulations require as rapid a return to representative regional condition for wetlands as possible (Harris 2007; Government of Alberta, 2013). While abundant open area remains for natural colonisation, government policy in oil sands encourages technical reclamation to expedite soil stabilisation, carbon accumulation and functions as similar regional mature wetlands as possible (Harris 2007).

In the wet-meadow zone of natural marshes in this region, *Carex sp.* often represents more than 70% of the total macrophyte aboveground biomass (Roy and Foote, unpublished data). A sedge-community dominated by *Carex aquatilis* was identified by Raab and Bayley (2013) as the optimal target community that constituted successfully reclaimed marshes. In addition to providing plant composition and function in the oil sands mined landscape that are similar to local undisturbed marshes, *C. aquatilis* is a promising species for reclamation due to its ability to spontaneously colonise (Prach et al. 2011) during the early development of certain oil sands created marshes (Roy and Foote unpublished data) and to tolerate pollution (Mollard et al. 2012). The sedge-community of natural marshes is dominated by the cover and biomass of *C. aquatilis, C. utriculata and C. atherodes* and *a* sub-community composed of *Scutellaria galericulata*, *Polygonum amphibium,* and *Galium trifidum* for example (Raab and Bayley, 2013; Roy and Foote, unpublished data). In oil sands created marshes where a sedge-community is present, the cover is dominated by only one species of *Carex* (i.e. *C. aquatilis*) and a sub-community composed *Achillea sibirica,* and *Melilotus* spp. for example (Raab and Bayley, 2013; Roy and Foote, unpublished data). In certain oil sands marshes, *C. aquatilis* occurs in very low abundance or is absent and the wet-meadow community is dominated by species such as *Typha latifolia* that are atypical of natural marshes (Roy and Foote, unpublished data). Thus, if early plant cover in the wet-meadows of oil sands created marshes is desired, the early planting of *C. aquatilis* appears to be a possible solution to dispersal limitation and to pre-empt the establishment of less desirable plant species (Raab and Bayley 2013). In certain oil sands marshes *C. aquatilis* percent cover was similar to natural marshes. However, *C. aquatilis* aboveground biomass was significantly lower (Raab and Bayley 2013). This difference was tentatively attributed to the toxic content of oil sands reclamation materials, the low nutrient availability, and reduced organic matter content of newly created marsh sediments (Trites and Bayley, 2009; Giesy et al., 2010; Rooney and Bayley, 2011). Vitt and Bhatti (2012) proposed that that species performance in created sites of the boreal region must be based on and equivalent to natural benchmarks (Vitt and Bhatti 2012).

At the individual plant level, *C. aquatilis* growth and performance were reduced in oil sands marshes compared to natural references (Mollard et al., 2012). The chemistry and structural characteristics of material produced by the oil sands extraction process (tailings sediments and water) appeared to reduce plant growth and physiology (Mollard et al., 2012), thus slowing the rate of reclamation (Purdy et al., 2005; Mollard et al., 2012). Emergent macrophytes such as *C. aquatilis* constituted a major fraction of organic matter production in freshwater marshes and the decomposition of plant litter by microbial processes fuels the *in situ* energy flow and nutrient cycling (Malcom, 1990, Kuehn et al., 2000). Mining activities have resulted in massive loss of wetlands and stored carbon (Rooney et al. 2012). Carbon sequestration through peat accumulation, sediment deposition and plant biomass is a key function provided by North American wetlands (Bridgham et al. 2006). Wetlands' ability to sequester carbon has repercussions for the regional and global carbon dynamic and provides crucial services to society in the context of climate change (Zedler and Kercher 2005; Bridgham et al. 2006). Ensuring that *C. aquatilis* growing in the oil sands of Alberta has elevated biomass and consequently, produced litter at rates similar to natural marshes of the region is one step toward successfully reclaiming wetland processes and functions (Purdy et al., 2005; Johnson and Myanishi, 2008).

Wetland soils and sediments are critical and challenging components to reclaim (Bruland and Richardson, 2006). Organic matter amendment in newly restored marshes improves soil properties and ecosystem functions. The deliberate addition of organic matter in created wetlands has been the focus of numerous studies (e.g. Bailey et al., 2007; Sutton-Grier et al., 2009; Ballantine et al., 2012). However, its positive effects on community structure (i.e. richness and/or composition) and on macrophyte growth and physiology remain controversial (see Handa and Jefferies, 2000; O'Brien and Zedler, 2006). In oil sands created marshes, the PM that contains three to five times the amount of organic carbon found in consolidated-tailings (CT) and tailings-sand (TS) has been used to increase soil aeration, water retention, root penetration and microbial habitat (Brady and Weil, 2008) (Table 1). The amendment of created marshes with PM was also intended to buffer soil from strong pH fluctuations and remove contaminants from water by adsorption, sequestration and denitrification processes (Tsutsuki and Ponnamperuma, 1987; Craft et al., 1988; Hogan et al., 2004; Sutton-Grier et al., 2009; Harrison-Kirka et al., 2013). Saline marshes created using tailings sediments and freshwater then capped with PM and *Carex sp.* were observed to produce twice as much aboveground biomass (124.4 g/m^2, SD=43.5) as similar created marshes not capped with PM (Roy and Foote, unpublished data). However, the limited number (n=4) of these pilot marshes precluded strong conclusions about the benefits of organic matter amendment on plant biomass.

Table 3- 1: Chemistry of consolidated-tailings (CT), tailings-sand (TS) and peat-mineral mix (PM) from which mixtures (i.e. CTPM and TSPM) were obtained. The sediments were sampled in the field while transferring the sediments in pots. Data were analysed by the Natural Resources Analytical Laboratory of the University of Alberta (Alberta, Canada). (TP=total phosphorus, NPOC= non-purgeable organic carbon, DC=dissolved carbon, DN= dissolve nitrogen, DOC=dissolved organic carbon, TC= total carbon, TN= total nitrogen, EC=electric conductivity, SM=sediment moisture).

Sediment Types	**Fe^{2+} mg/kg**	**Mg^{2+} mg/kg**	**Na^{+} mg/kg**	**Cl^{-} mg/kg**	**TP mg/L**	**NPOC mg/L**	**DC mg/L**
TS	1.3	30.4	814.9	1.1	0.0	27.8	44.5

CT	1.7	335.6	995.2	14.4	0.4	47.7	69.0
PM	2.4	480.3	4018.8	78.7	0.6	66.1	114.1
	DN	**DOC**	**TC**	**TN**	**EC**	**SM**	
	mg/L	**mg/kg**	**mg/kg**	**mg/kg**	**us/cm**	**%**	
TS	0.6	37.2	59.4	0.8	306.2	25.8	
CT	0.9	67.2	97.3	1.3	395.0	31.8	
PM	3.4	180.3	311.3	9.4	598.1	90.7	

The oil sands created marshes are generally constructed using a variety of unusual sediment formulations resulting from the extraction and transformation of bitumen. Oil sands marsh creation occurs on the newly exposed marine–shale overburden (Purdy et al., 2005). Some of these oil sands created marshes are capped with processed materials produced during the bitumen extraction process such as CT, TS and oil sands processed water (OSPW). The CT is composed of sand, clay, and gypsum while TS is mainly composed of sand (Harris 2007). Both, CT and TS contain moderate to elevated salinity levels, residual bitumen and associated hydrocarbon contaminants (Harris 2007). Both CT and TS are low in organic matter. In addition to its chemistry, the density of CT has been hypothesised to reduce plant root penetration (Rooney and Bayley, 2011). Notably, OSPW is recycled many times through the extraction process, whereby it accumulates high levels of salts, NAs, and ions including ammonia (NH_4), chloride (Cl), boron (B), and copper (Cu) (MacKinnon et al. 2005; Giesy et al. 2010). The physical properties and chemistry of CT, TS and OSPW are likely to influence marsh vegetation and its functions (Crowe et al., 2002; Kamaluddin and Zwiazek, 2002; Kovalenko et al. 2013).

The PM was obtained during the top-soil removal preceding surface mining but its use in oil sands created marshes is expensive. In addition to its excavation, storage (for up to ten years) and transportation, stored PM is subject to a period of rapid decomposition (oxidation) which leads to CO_2 release into the atmosphere (Bruland and Richardson, 2006). Furthermore, Sutton-Grier (2009) found the richness of wetland species decreased with an increase of organic matter, suggesting the possibility for non-generalizable or non-linear effects of organic matter amendment in created wetlands. Determining if PM enhances macrophyte functions in oil sands created marshes is thus important to guide successful reclamation practices, validate key reclamation practice costs, while minimizing the related environmental costs. The post-mining oil sands landscape represents an ideal environment to test the effects of organic matter amendment on marsh vegetation functions in newly created wetlands.

The overall goal of this study was to determine if the addition of PM to oil sands sediments would enhance *C. aquatilis* biomass and metabolism. We studied the introduction of *C. aquatilis* and its short-term growth under the various post-mined conditions. My three objectives were to: 1) characterise the plant response to two types of water (i.e. freshwater and OSPW) and three types of reclamation sediment (i.e. CT, TS and PM), 2) experimentally test the effects on plants resulting from PM addition to CT and TS, and 3) understand the effects of oil sands process water (OSPW) on *C. aquatilis*.

3.2. Materials and Methods

3.2.1. Research Site

This field study was conducted in the Fort McMurray region of northeastern Alberta, Canada on *Suncor Energy Inc.* property (56°58'49.70"N and 111°30'22.45"W). Six parallel research trenches constructed in 1995 were used to conduct the experiment[13]. Trenches were oriented in a north-south direction. Each trench was approximately 40 m long and 5 m wide and a distance of approximately 12m separated them. During their construction, the trenches were capped with 10cm of overburden on top of a synthetic membrane to prevent leakage. As has been done for many other wetlands created in the post-mined landscape, overburden was used as reclamation subsoil. Overburden is a natural soil that is salvaged below the organic surface soils (i.e. PM and upland surface soils) in advance of mining and either stockpiled or directly placed onto an area for reclamation. From 2005 to 2012, the trenches were used for research purposes and water levels and contents were kept constant by the repeated

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[13] Experiments conducted in the trenches are subjected to pseudo-replication and spatial-autocorrelation. Although this issue could have been alleviated by using conducting the experiment in a multitude of marshes dispersed across the mine, mine access limitation and safety rules limited us to a single location.

addition of OSPW in trenches 2, 4 and 6 and freshwater in trenches 1, 3 and 5. Based on measurements made in created and natural marshes of the Fort McMurray region, trench water levels were allowed to fluctuate during the summer within *a priori* set water level range (i.e. minimum of 2 cm and a maximum of 10 cm deep) that would favor *C. aquatilis* growth. The trenches were periodically flushed or supplemented with their respective water treatment to account for water lost through evapotranspiration and to adjust solute concentrations. The OSPW used for the experiment was collected from a tailing pond water catchment basin, while freshwater was collected from an onsite constructed lake. The six trenches were assumed to be affected by similar environmental and climatic conditions characteristic of the Fort McMurray region. Mean summer and winter daily temperature averages of the region are 13.5◦C and -13.2◦C, respectively (Strong and Laggat, 1992).

3.2.2. Experimental design, data collection and statistical analyses

The PM addition was tested on CT and TS. The growth experiments were conducted under two factors being 1) sediment types (CT, CT and PM mixture (CTP), TS, TS and PM mixture (TSPM), and PM) and 2) water types (freshwater *versus* OSPW). The CTPM and the TSPM were composed of a mixture of 1/3 of processed material and 2/3 PM. The experiment was replicated three times (i.e. in three trenches). Thus, twelve treatments (i.e. six sediment combinations subjected to two water types) were tested. *A priori* power analyses were performed to calculate the total sample size needed to test the sediments × waters two-factor interaction at α level 0.05 and indicated a minimum power of 0.6. To increase the power of my analysis, CT and CTPM pots were compared to *a priori* selected PM pots (PM1) while TS and TSPM pots were compared to a different set of PM pots (PM2). Three unplanted control pots per sediment type were randomly placed in each trench to test for unanticipated plant invasion.

C. aquatilis plants were collected in early June 2010 from a natural marsh of the Fort McMurray region (56˚30’50, 31”N and 111˚16’17, 47”W). Plants collected had to be healthy (i.e. green leaves that did not show signs of stress), approximately 10 cm high and with a healthy root system (i.e. at least three roots that did not show sign of stress).

All plants collected from the natural marsh were randomly transplanted into one gallon pots (i.e. 15.9 cm in diameter and height) and the pots were randomly assigned to a position in the trenches during the same day. To ensure that the conditions (e.g. water level) were optimal for the transplanted *C. aquatilis*, the pots were located in the same zone of the trenches where naturally occurring *C. aquatilis* was growing.

Each week during the summers of 2010 and 2011, trench water chemistry including temperature, pH, oxidative-reduction potential (ORP), salinity, dissolved oxygen (DO), and specific conductance (adjusted for temperature and hydrogen ions) were monitored using a Handheld Oxygen, Conductivity, Salinity, and Temperature System (YSI85)(Table 2). Once each summer, naphthenic acids, anions and cation levels were sampled by Suncor Energy Inc. and analysed by Syncrude Canada Ltd. Environmental lab. Bicarbonates and carbonates were analyzed by alkalinity titration, chlorides, sulfates and ammonium were analyzed by ion chromatography, while sodium, magnesium, calcium, boron, manganese, silicon, sulfur, and strontium were analyzed using an inductively coupled plasma optical emission spectrometry.

Table 3- 2: Water chemistry components averaged (n=3) and their associated ±95% CI. * indicates significant differences between water treatments (F= freshwater, OSPW= oil sands process water)[1].

	F		OSPW	
	Mean	CI	Mean	CI
Temperature (◦C)	18.0	1.1	17.8	0.9
Salinity (ppt)*	0.2	0.0	0.7	0.0
Redox potential (mV)	95.5	10.1	85.0	17.4
pH	7.5	0.6	8.0	0.2
Dissolved oxygen (mg/L)	3.9	1.2	3.3	0.8
Naphthenic acid (mg/L)*	3.0	0.0	22.7	3.5
Carbonate (CO_3^{2-}) (ppm)*	0.0	0.0	21.4	14.8
Bicarbonate (HCO_3^-) (ppm)*	239.0	51.1	598.3	85.4
Chloride (Cl^-) (ppm)*	62.0	8.5	176.7	77.1
Sulfate (SO_4^{2-})*	90.1	3.4	174.0	58.2

Sodium (Na^+) (ppm)*	64.4	6.2	380.3	46.6
Magnesium (Mg^{2+}) (ppm)*	24.1	2.2	20.2	2.4
Calcium (Ca^{2+}) (ppm)*	59.8	20.0	36.4	1.5
Ammonium (NH4+) (ppm)	0.3	0.1	0.3	0.0
Boron (B) (ppm)*	0.2	0.1	1.8	0.4
Manganese (Mn) (ppm)*	24.1	1.1	20.2	1.2
Silicon (Si) (ppm)	0.4	0.2	0.9	0.5
Sulfur (S) (ppm)*	32.6	1.3	65.1	20.1
Strontium (Sr) (ppm)	0.3	0.1	0.4	0.1

[1] To determine which water chemistry components were significantly different between water types, a PCA coupled with t-tests ($\alpha=0.05$, $n=3$) was performed. The PCA was used to express the covariation of the water chemistry components (17) as a smaller number of composite variables (2). Prior to the PCA analysis, data were log transformed and standardized. The trench scores in the reduced space were then used to perform t-tests.

To test *C. aquatilis* stress level under each treatment, chlorophyll *a* fluorescence was used as a proxy for physiological performance. In August 2011, fluorescence measurements were carried out on *C. aquatilis* dark-adapted leaves in five pots of each soil and water treatment. The five measurements were then averaged for each replicate. Chlorophyll *a* fluorescence transients were measured with a Hansatech Pocket PEA (Hansatech Instruments Ltd., King’s Lynn, Norfolk, UK). The definitions and derivations of the fluorescence transient parameters are summarized in table 3 (for further details see Strasser et al., 2000).

Table 3- 3: Derivation of the parameters directly obtained from the recorded fluorescence transients.

Parameters	**Derivation**
$F_{0:}$ fluorescence at 50μs; assumed to be initial	F_0
F_m: maximal fluorescence intensity	F_m
F_v : variable fluorescence	$F_v=F_m-F_0$

$M_{0:}$ slope at the origin of normalized fluorescence rise	dV/dt_0=(f300 ms-F_0/(F_m-F_0)
ABS/RC: calculated absorption per reaction center[1]	(M_0/Vj)/(1-F_0/F_m)
F_v/F_m : maximum quantum yield of primary photochemistry	1-F_0/F_m
PI: Performance Index	(V_j/M_0)(F_v/F_m)(F_v/F_0)(ET_0/(dQ_A/dt_0)
ET_0/$TR_{0:}$ Probability that a trapped exciton moves an electron further than Q_A	1-V_j
ET_0/ABS: Probability that a absorbed photon moves an electron further than Q_A	(1-F_0/F_m)(1-V_j)

[1] Vj: After Christen et al. (2007) and Strasser et al. (2000). Vj = (F2 ms − F0)/(FM − F0).

At the end of the 2011 growing season each plant belowground biomass (roots and rhizomes) and aboveground biomass (culms and leaves) were harvested and individually stored in paper bags. The samples were brought to the lab and dried in the oven at 70◦C to a constant weight before they were weighed for biomass.

3.3. Statistical analysis

3.3.1. Data screening

Survival, biomass and chlorophyll *a* fluorescence data were inspected for outliers, normality, and homogeneity of variance prior to statistical analyses. Data with non-normal distributions were log transformed and data expressed as ratios were *arcsin* transformed.

3.3.2. C. aquatilis survival, biomass and physiology

*S*urvival (presence/absence) was first averaged in each of the eight pots of each sediment treatment of each trench. Then, survival was averaged for the three replicated treatments (i.e. trench). Biomass of surviving plants was collected in all eight pots in each treatment (i.e. CT, CTPM, PM1, TS, TSPM and PM2). Thus, the average biomass calculated per replicate represented the biomass produced by the remaining alive plants. Belowground mass ratio and aboveground biomass ratio were defined as the ratio of belowground biomass and aboveground biomass, respectively, to the sum of below and aboveground biomass.

Two-way ANOVAs (α=0.05, n=3) were performed to test whether 1) sediment and water treatments had an overall effect on *C. aquatilis* survival/biomass/fluorescence, and 2) the effect of sediment was the same under the two types of water (interaction of treatments). Where the F-ratio for the sediment or water treatment main effect was significant ($p< 0.05$), pairwise comparisons to isolate differences were performed using Tukey tests (α=0.05, n=3). CT was only statistically compared to CTPM and PM1 while TS was always and only statistically compared to TSPM and PM2.

3.3. Results

3.3.1. C. aquatilis survival

Of the CT sediment types (CT, CTPM and PM1), only sediment types influenced *C. aquatilis* survival (Figure 1, Table 4). *C. aquatilis* survival was significantly higher in freshwater and OSPW when growing in PM (PM1) than in either CT or CTPM. No sediment or water effects were found on *C. aquatilis* survival in TS sediment types (TS, TSPM and PM2). No overall effect was found from the interaction of sediment and water treatments for both CT and TS sediment types.

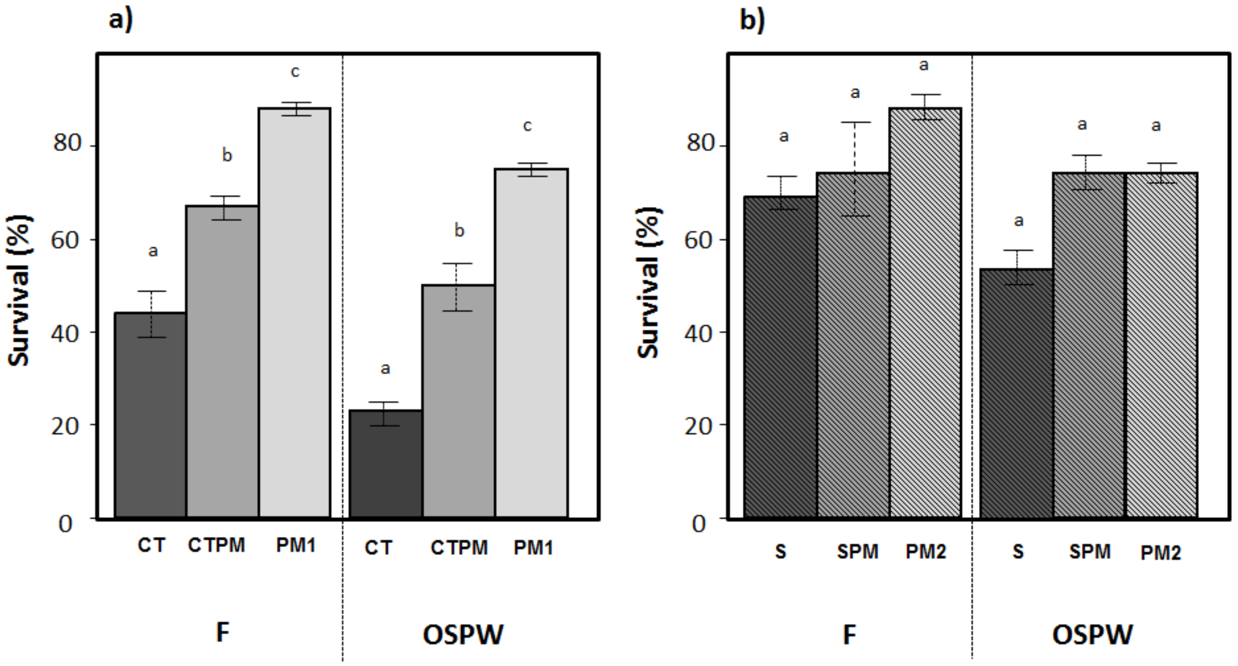

Figure 3- 1: *C. aquatilis* survival in the two factors experiment (i.e. sediment and water types): the bars represent the averaged percentages (n=3) of *C. aquatilis* survival in each treatment and their associated ±CI (95%). Letters indicate significant differences (ANOVA, alpha=0.05, n=3) among sediment treatments following pairwise comparisons using Tukey test. a) CT was statistically compared to CTPM and PM1 while b) TS was statistically compared to TSPM and PM2. Water types had no significant effect on *C. aquatilis* survival. (CT= consolidated-tailings, CTPM= consolidated-tailings and PM mixture, PM= peat-mineral-mix, TS= tailings-sand, TSPM= tailing-sand and PM mixture, F= freshwater, OSPW= oil sands process water).

Table 3- 4: Summary of the two-way ANOVAs (alpha=0.05, n=3) for *C. aquatilis* survival, below and aboveground biomass, and aboveground to belowground biomass ratio in the two factors experiment (i.e. sediment and water types) (CT= consolidated-tailings, CTPM= consolidated-tailings and PM mixture, PM= peat-mineral-mix, TS= tailings-sand, TSPM= tailing-sand and PM mixture).

		Sediments (S)		Waters (W)		S x W	
Sediments compared	Indicators	F value	Pr (>F)	F value	Pr (>F)	F Value	Pr (>F)
CT, CTPM, PM1	Survival	15.40	**<0.01**	5.95	0.09	0.06	0.94

TS, TSPM, PM2	Survival	1.81	0.21	1.71	0.22	0.05	0.96
CT, CTPM, PM1	Belowground Biomass	12.73	**<0.01**	19.44	**<0.01**	2.05	0.27
TS, TSPM, PM2	Belowground Biomass	14.92	**<0.01**	13.26	**<0.01**	1.05	0.38
CT, CTPM, PM1	Aboveground Biomass	4.67	**0.03**	2.01	0.18	2.58	0.12
TS, TSPM, PM2	Aboveground Biomass	27.76	**<0.01**	4.31	0.06	1.80	0.21
CT, CTPM, PM1	Aboveground:Belowground	1.63	0.24	8.29	**0.01**	0.76	0.49
TS, TSPM, PM2	Aboveground:Belowground	1.40	0.30	9.57	**0.01**	0.50	0.62

3.3.2. C. aquatilis biomass

In CT sediment types, sediment had a significant effect on both below and aboveground biomass ($p<0.05$)(Figure 2, Table 4). In OSPW plants had significantly higher biomass in CTPM and PM1 than in CT ($p<0.05$). Plants growing in the CT/OSPW treatment had significantly lower below and aboveground biomass when compared to other treatments ($p<0.05$). Despite the finding that plants growing in fresh water had a tendency to have higher biomass in CTPM and PM1 than in CT, the differences were not significant. No significant differences in below and aboveground biomass were found between CTPM and PM1 in either fresh water or OSPW. No interaction effect was found between sediment and water treatments in CT sediment types.

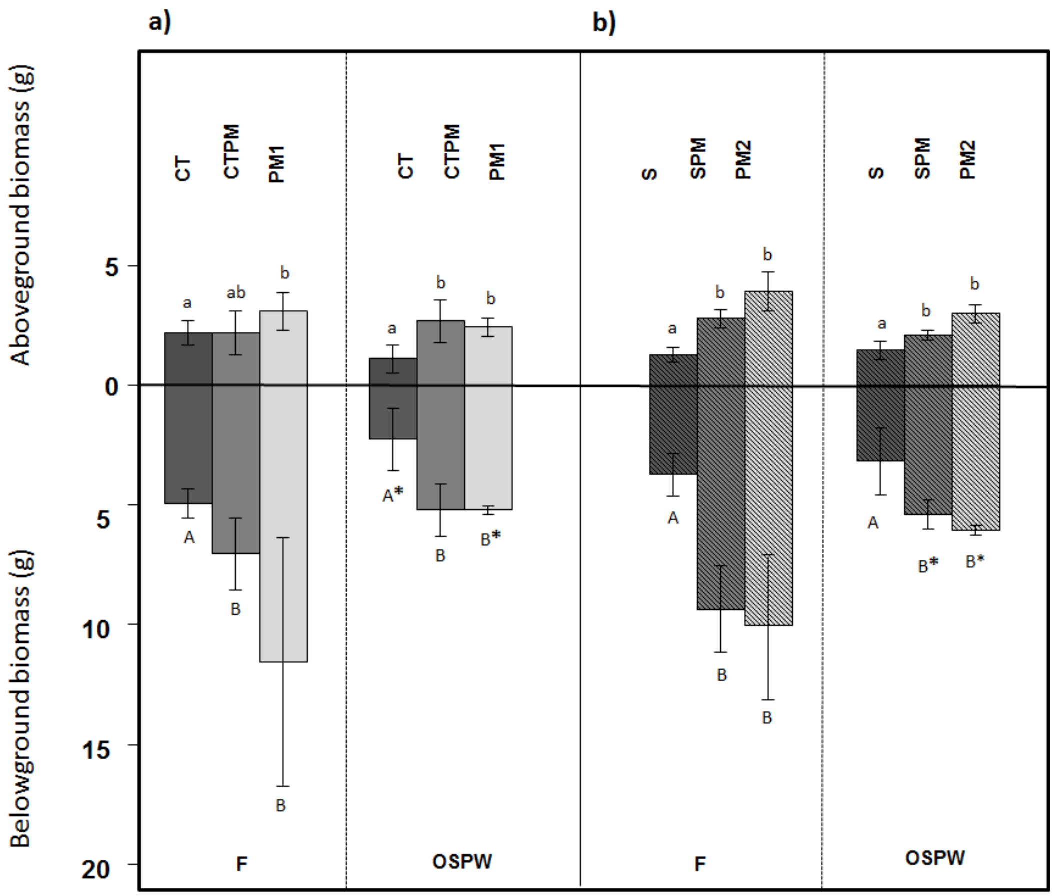

Figure 3- 2: *C. aquatilis* below and aboveground biomass allocation under a two factors experiment (i.e. sediment and water types). The bars represent the averaged biomass values (n=3) in each treatment and their associated ±CI (95%). Letters indicate significant differences (ANOVA, alpha=0.05, n=3) among sediment treatments following pairwise comparisons using Tukey test. * is used to indicate the significant influence of water types on *C. aquatilis* belowground biomass. a) CT was statistically compared to CTPM and PM1 while b) TS was statistically compared to TSPM and PM2. Non-capitalized letters represent significant differences among aboveground biomass and capitalized letters represent significant differences among belowground biomass. (CT= consolidated-tailings, CTPM= consolidated-tailings and PM mixture, PM= peat-mineral-mix, TS= tailings-sand, TSPM= tailing-sand and PM mixture, F= freshwater, OSPW= oil sands process water).

In TS sediment types, sediment types had a significant effect on below and aboveground biomass ($p<0.05$). Water types also had a significant effect on the belowground biomass of *C. aquatilis* ($p<0.05$). In fresh water, plant below and aboveground biomass in TS was significantly lower than biomass in TSPM and in PM2 ($p<0.05$). In OSPW, plant aboveground biomass was significantly higher in PM2 than in both TS and TSPM ($p<0.05$). No interaction effect was found between sediment and water treatments in TS sediment types.

The above to belowground biomass ratio was found to be significantly affected by water types ($p<0.05$) in both CT and TS sediment types. Although plant aboveground biomass allocation was not different among treatments, plant belowground biomass allocation was significantly lower in OSPW ($p<0.05$ data not shown). On average, the above to belowground biomass ratio was 0.45 for OSPW and 0.30 for freshwater. Plant sediment types and the interaction of sediment and water had no significant effect on the above to belowground ratio.

3.3.3. C. aquatilis chlorophyll a fluorescence

Chlorophyll *a* fluorescence statistical analyses indicate significant differences in basic fluorescence parameters among sediment and water treatments (Figure 3, Table 5). For plants growing in CT sediment types, sediment had a significant effect on initial fluorescence (F_0) and the slope at the origin of normalized fluorescence rise (M_0) ($p<0.05$). For plants growing in TS sediment types, F_0, M_0, and ABS/RC were significantly influenced by sediment types ($p<0.05$).

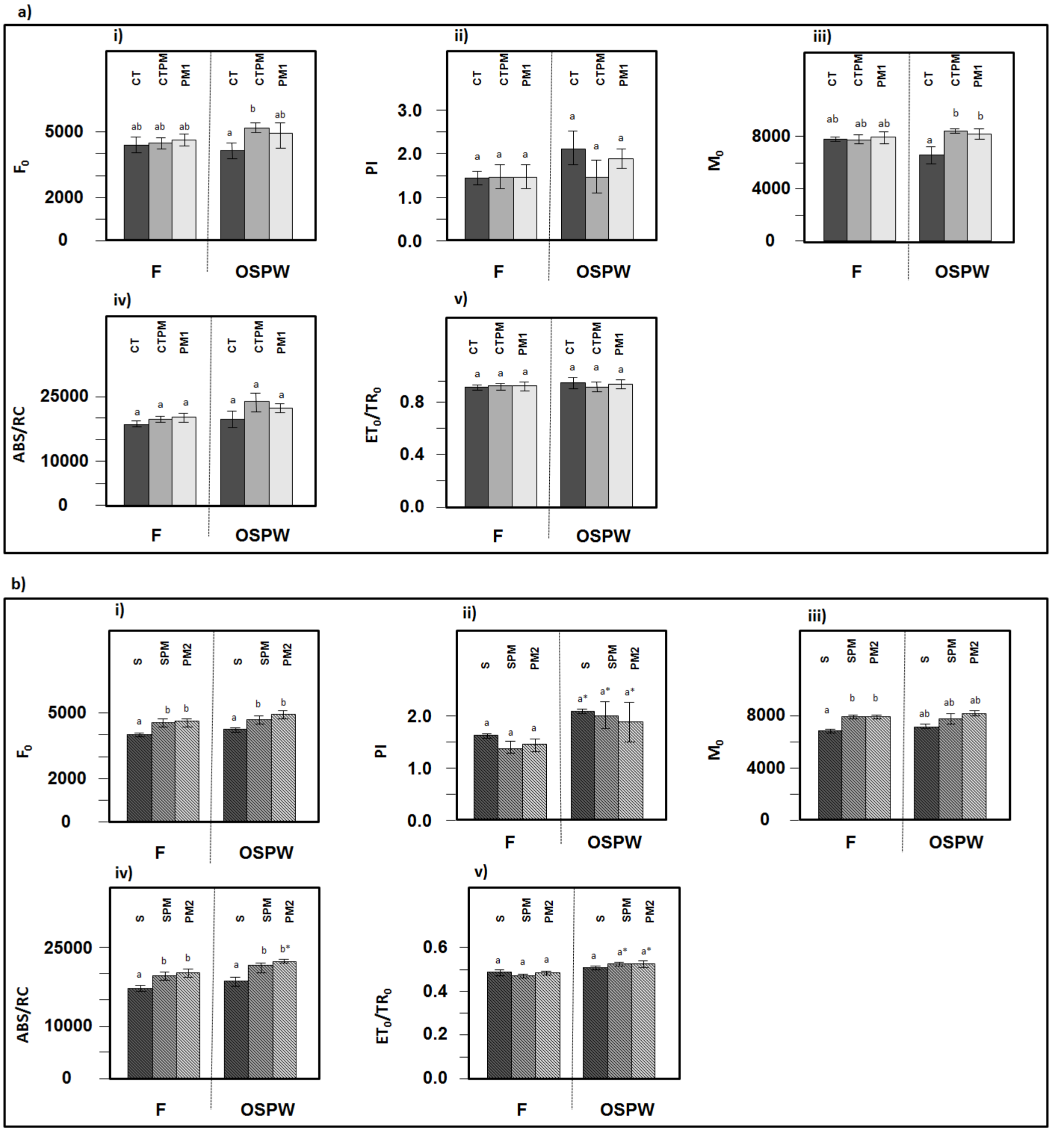

Figure 3- 3: Derivative chlorophyll *a* fluorescence parameters presenting statistically significant differences ($p<0.05$). The values represent the average ±CI (95%). Measurements were taken on *C. aquatilis* growing in different sediments and water treatments: **i)** initial fluorescence , **ii)** Performance Index, **iii)** slope at the origin of normalized fluorescence rise, **iv)** light absorption per reaction center, **v)** probability that a trapped exciton moves an electron further than QA. a) CT was statistically compared to CTPM and PM1 while b) TS was statistically compared to TSPM and PM2. Letters indicate significant differences among sediment treatments following pairwise comparisons using Tukey test. (*) is used to indicate the significant influence of water types on *C. aquatilis* derivative chlorophyll *a* fluorescence parameters. (CT= consolidated-tailings, CTPM= consolidated-tailings and PM mixture, PM= peat-mineral-mix, TS= tailings-sand, TSPM= tailing-sand and PM mixture, F= freshwater, OSPW= oil sands process water).

Table 3- 5: Summary of the two-way ANOVA (alpha=0.05, n=3) results for the analyzed chlorophyll *a* fluorescence parameters of *C. aquatilis* submitted to three sediments and two water types (CT= consolidated-tailings sediment type, TS= tailings-sand sediment type).

Parameters	**CT Sediments (S)**		**Waters (W)**		**SxW**		**TS Sediments (S)**		**Waters (W)**		**SxW**	
	F value	Pr (>F)	F value	Pr (>F)	F value	Pr (>F)	F value	Pr (>F)	F value	Pr (>F)	F value	Pr (>F)
F_0	4.17	**0.04**	1.98	0.20	2.02	0.20	4.17	**0.04**	1.98	0.18	2.02	0.17
PI	0.96	0.41	3.00	**0.01**	1.02	0.39	0.57	0.58	7.09	**0.02**	0.20	0.82
M_0	7.98	**0.01**	0.01	0.94	7.76	**0.01**	9.15	**0.01**	0.82	0.38	0.44	0.65
ABS/RC	0.70	0.51	0.45	0.52	0.68	0.52	7.54	**0.01**	5.50	**0.04**	0.13	0.88
ET_0/TR_0	0.49	0.62	0.66	0.43	1.13	0.35	0.12	0.88	6.08	**0.03**	0.48	0.63

Overall, basic fluorescence parameters tended to be higher in OSPW than in Freshwater. Water type had a significant effect on plant performance index (PI), a rough indicator of plant vitality (Strasser 2000) in both CT and TS sediment types ($p<0.05$). In CT sediment types, *C. aquatilis* growing in OSPW had a significantly higher PI when compared to plants growing in fresh water. In addition, water types had a significant effect on ABS/RC, and ET_0/TR_0 for plants growing in TS sediment types ($p<0.05$).
Discussion

3.4. Discussion

My study was performed to compare *C. aquatilis* biomass and performance under different sediment and water amendments used by oil sands companies. I have demonstrated that PM amendment to oil sands sediments (CT and TS) significantly increased *C. aquatilis* survival and its ability to accumulate biomass and consequently, its storage of carbon. My results indicate that despite the improvement of oil sands sediments quality by the addition of PM, OSPW still restricts *C. aquatilis* biomass. I have confirmed that despite oil sands sediments and waters serving as stressors on *C.*

aquatilis, its tolerance to oil sands materials makes it a good candidate for the reclamation to a sedge-dominated community in the wet-meadow zones of the post-mined landscape. Even with reduced growth, its survival and gradual accumulation of biomass represents an important ecological contribution to an otherwise low productivity site.

3.4.1. C. aquatilis is a good plant choice for reclaiming oil sands marshes

My results provide additional support for speculation of Raab and Bayley (2013) that *C. aquatilis* was a good candidate to reclaim the wet-meadow zones of oil sands created marshes. Although its biomass accumulation was significantly affected in the presence of OSPW, tailings and uncapped CT, *C. aquatilis* has shown an ability to tolerate post-mining conditions affected by CT, TS and OSPW. Despite the fact that most transplanted *C. aquatilis* shoots in the CT/OSPW treatment did not survived, approximately 20% of the transplanted shoots established, grew and spread over the two growing seasons. Thus, even in created marshes amended with CT/OSPW, some *C. aquatilis* are expected to survive and increase the likelihood of establishing a sedge-community. If planting *C. aquatilis* is a strategy used to restore CT/OSPW wetlands, changing the transplanting strategy or introducing a higher density of *C. aquatilis* may be necessary to achieve acceptable plant stands and density. Thus, based on *C. aquatilis* ability to survive even in CT, TS and OSPW treatments, establishing a sedge-community in oil sands created marshes is conceivable.

3.4.2. Sediment and water treatments influence C. aquatilis survival and biomass

Although planting *C. aquatilis* in CT and TS sediments to create vegetated marshes is possible, reestablishing its ability to store carbon that is equivalent to those of natural sedge-communities appears unlikely in the short term. The significantly lower initial survival rate of plants in CT/OSPW indicates that the limiting factor may be the establishment niche of young or stressed plants that have not acclimated to the conditions. The meager below and aboveground biomass of the surviving *C. aquatilis* in CT and TS treatments (in freshwater and OSPW conditions) illustrates the stress exerted by the chemical and physical components of these sediments on plant growth.

Compared to PM, both CT and TS contained lower levels of nutrients (carbon, nitrogen and phosphorus) and lower soil moisture content that may thereby explain the compromised below and aboveground biomass measured in CT and TS treatments compared to PM treatments. Trites and Bayley (2009) and Rooney and Bayley (2011) speculated that these sediment characteristics influenced plant community structure in created oil sands marshes. In addition to the sub-optimal conditions of the CT and TS, my analyses demonstrate that OSPW has high salinity content, high levels of Na^+, Cl^-, and naphthenic acids that may exert an additional and significant stress on *C. aquatilis* survival and functions. Although shown to be toxic to *Populus tremuloides* and *Arabidopsis thaliana* (Kamaluddin and Zwiazek, 2002; Pouliot et al., 2012; Leishan et al., 2013), the negative effect of NAs on emergent macrophytes with inundated roots remains to be demonstrated. Furthemore, little is known about the chemical, physical and biological properties of interstitial water in oil sands sediments. The interstitial water may be the key determinant of plant water and nutrient uptake (Young, 1998). This interstitial water can be compromised by freshwater interacting with the oil sands sediments, such as CT, or it can be compromised by the otherwise benign sediments being salinized by the high conductivity of surface water such as OSPW. Logically, CT/OSPW should have both detractors unless CT's detrimental effects on plants decrease over time (e.g. precipitation flushes CT salts) and unless OSPW provides nutrients that may compensate for poor initial conditions.

3.4.2.1 PM amendments favor C. aquatilis growth in oil sands processed material

Sediment and water types influenced *C. aquatilis* below and aboveground biomass. Hence, materials used for reclamation will influence *C. aquatilis* biomass and if the results can be extrapolated to the stand-level, the restoration of sedge-community functions. In marshes created with CT and TS sediments, *C. aquatilis* biomass was expected to be at its lowest. The use of PM or the mixture of PM with both CT and TS significantly increased below and aboveground biomass by presumably providing sediment with better chemical and physical conditions (i.e. increased access to nutrients, decreased bulk density for better root penetration and increased water-retention capability). Interestingly, no significant difference in below and aboveground biomass was observed between CTPM and PM1 or TSPM and PM2 (with the exception of plants growing in CT sediment types and freshwater). This result has important implications for oil sands reclamation practices. Indeed, most created oil sands marshes capped with PM have received at least a 50 cm layer of this peat-based sediment. My results suggest that instead of capping created wetlands with PM, a mixture of PM and oil sand sediments (CTPM and STPM) may be sufficient to optimise marsh sedge-communities biomass (but not survival). Mixing PM with CT and TS represents a 33% reduction of PM used and significant financial savings and environmental improvement over current reclamation practices.

3.4.2.2. OSPW significantly reduced C. aquatilis belowground biomass

The analyses of belowground biomass suggest that the choices of water in the construction of wetlands can significantly influence *C. aquatilis* belowground biomass but interestingly, not its aboveground biomass. The analyses of the above to belowground biomass ratio support these results and indicate that *C. aquatilis* belowground biomass allocation will be reduced in OSPW. Salinity and NAs content of OSPW directly influence plant physiological function by altering the uptake and transport of water and nutrients from the soil (Kamaluddin and Zwiazek, 2002). Plant roots subjected to OSPW exhibit significant physiological changes including cell death in the plant root epidermis and change in the chemistry of parenchyma cells in the root pith (Armstrong et al. 2008). These changes may also result from the indirect effect of naphthenic acids on bacterial communities beneficial to plant functions (Armstrong et al., 2008).

In a striking reversal of my predictions, chlorophyll *a* fluorescence data indicated less favorable habitat conditions in some treatments that involved freshwater or peat amendments than those that involved oil sands by-products. Likewise, plants transplanted into pots filled with tailings sands showed a lower F_0 than plants growing in peat-amended substrates or in peat. F_0 is an indication of photoinhibition and can be high due to photoprotective processes at the level of the light harvesting antenna (Maxwell and Johnson, 2000). Moreover, *C. aquatilis* in OSPW showed healthier photochemical activity (higher ET_0/TR_0, and PI) than plants growing in freshwater treatments. These results may seem counterintuitive; however, chlorophyll a fluorescence is sensitive to leaves nutritional status (Huang et at., 2004), suggesting oil sands processed waters positively affect physiological performance, possibly caused by introduced nitrogen compounds in OSPW and CT.

These results reinforce the findings of Mollard et al. (2012) showing that despite growing in tailings-polluted water, *C. aquatilis* can maintain its physiological performance. On the other hand, results indicate that, due to the higher element content of the oil sands by-products, plants subjected to industrial water and substrates may access better nutritional conditions than plants growing on natural substrates and freshwater, an effect that has been shown in other wetland species (Bendell-Young et al., 2000). The role of the above-mentioned better photochemistry activity in leaves of plants affected by industrial by-products on *C. aquatilis* survival and performance is still unclear as can be seen by a lack of positive responses at the whole plant level.

3.4.3. Reclamation of sedge-community functions at the landscape level

The use of PM amendments can significantly enhance *C. aquatilis* biomass. From a conservative estimation, amending CT and TS sediments with PM can increase the annual biomass production by *C. aquatilis* by a factor of 1.5. If these differences are maintained at the community level, such an increase of biomass becomes a noteworthy landscape improvement. On average, the wet-meadow zone of natural marshes found in the Fort-McMurray region represent approximately 60% of the total marsh area (the submersed aquatic vegetative zone is 40%) (Roy and Foote, unpublished data). If one quarter (i.e. 10330 km^2) of the total mineable surface was returned to shallow open water and marsh wetlands (Rooney and Bayley, 2011; Purdy et al., 2005), approximately 6200 km^2 of this area would return to wet-meadows supporting sedge-community dominated by *C. aquatilis*. Based on my estimations (Roy and Foote, unpublished data) and those of Raab and Bayley (2013), natural sedge-communities produce 120 to 475 g/m^2 of aboveground dry biomass annually. If the reclaimed *C. aquatilis*-community can produce as much biomass as the sedge-community of natural marshes of Fort-McMurray, the total dry aboveground biomass produced annually would range between 744 and 2945 metric tons across the 6200 km^2 reclaimed. Based on my results, *C. aquatilis* produces approximately 4.7 times more biomass belowground than aboveground. Hence, the combined below and aboveground dry biomass annual contribution for the area to be returned to sedge-community would range between 3496 and 13841 metric tons. Thus, the amendment of PM to CT and TS could make significant revegetation contributions at the landscape scale.

3.5. Study limitations

This experiement is subject to pseudo-replication and spatial auto-correlation, which could have been avoided by conducting the experiment in marshes scattered across the mine. However, due to safety rules imposed by the oil sands industry, the experiment was restricted to the trenches.

Although I have demonstrated that PM amendment to oil sands sediments (CT and TS) significantly increased transplanted *C. aquatilis* survival and its ability to accumulate biomass, in field applications PM is placed in a layer on top of the other sediments. By mixing the PM into the sediments I likely had different effects on the overall soil structure. Furthermore, the operational challenges of mixing PM with CT and or TS might, at the landscape scale, may be significant.

Although my results demonstrate that PM increases C.aquatilis survival and biomass accumulation, the advantages of PM at the community level remain to be tested (see Chapter One). For example, the amendment of PM to a created marsh can introduce plant propagules that may reduce the presence, abundance and/or performance of *C. aquatilis.*

3.6. Recommendations

I demonstrated that in CTPM, SPM and PM, the introduction of *C. aquatilis* shoots is a technically viable revegetation technique with moderate to high levels of first year survival and very good subsequent establishment and spread on suitable sediment types. Determining if the manual introduction of *C. aquatilis* shoots provides advantages over introducing its seeds remain to be tested.

I recommend that *C. aquatilis* be introduced early after marsh creation. Under the right conditions (i.e. sediment and water chemistry, hydrology, etc.) planting *C. aquatilis* may increase the plant cover of unvegetated sediment and/or prevent the establishment of less desirable (i.e. invasive, non-native, weedy) plant species observed in many self-designed oil sands wetlands (Raab and Bayley, 2013). This hypothesis remains to be tested.

3.7. Conclusions

Restoring ecosystem structure is more challenging than restoring its functions (Temperton, 2004). Although *Carex atherodes* and *C. utricularia* tend to dominate the natural marshes of the Fort McMurray region, my results demonstrate that with the amendment of PM, *C. aquatilis* is a good candidate to reclaim oil sands marsh functions (plant carbon storage). To improve my knowledge of the role of PM amendment in newly created wetlands vegetation, responses across a longer gradient (depth and proportion) of soil amendment should be tested (Sutton-Grier et al., 2009; Ballantine et al., 2012).

The massive challenge of revegetating the post-mined landscape which contains varying levels of contaminants, or natural sediments, and sometimes compromised surface water quality calls for creative combinations of organic amendments (PM) and wise choices of durable, adaptable, late succession plants that can survive, expand and serve as a base of an organic carbon detrital system. My research confirms that *C. aquatilis* planted in sites supplemented with locally derived peat-based soils is one of the best and most ecologically fitting combinations yet found for widespread wetland reclamation.

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CHAPTER 4

Title: The use of remote sensing photography to compare the area of submerged aquatic vegetation zone of natural and created marshes in northern Alberta (Canada).

4.1. Introduction

Wetlands are located at the interface of land and water and due to their connection with both ecosystems, wetlands are dynamic in every aspect (Keough et al. 1999). Wetland water level variation is an overriding factor governing wetland structure and function (Watt et al. 2007; Raulings et al. 2010). The water level variation results in the expansion and contraction of wetland boundaries (van der Valk 2005). Patterns of biotic succession and diversity, productivity, and habitat for insects, fish, birds and mammals respond strongly to change of water availability caused by water level variation (Zhang and Mitsch 2005). Thus, understanding the effect of water variation in wetlands is fundamental to effectively restoring or reclaiming ecosystem function and processes (Zhang and Mitsch 2005; Euliss et al. 2008).

The oil sands industry of Alberta is required to reclaim a vast mined landscape to equivalent land capability (Foote 2012). The created ecosystem’s capability is thus required to be equivalent to conditions preceding land disturbance (Harris 2007). Vitt and Bhatti (2012) have proposed that to enhance the success of reclamation in the Boreal region created ecosystem function must be based and on the one of natural analogue ecosystems. Reclaiming wetland hydrology is the primary factor in restoring wetland structure and function (Euliss et al. 2008). Because wetlands are dynamic entities, defining reference conditions has led to considerable debate over standards and endpoints for comparison (Wilson et al. 2013). Quantifying the long-term water level variation of natural marshes in this region and understanding the main factors that drive these patterns provides managers and regulators with a range of benchmarks and tools from which the success of created marshes can be assessed. No estimate of the variation of SAVZ area between year exisits for the natural marshes of the Fort McMurray region. Consequently, the changes in created marsh zone area lack comparative levels and benchmarks, and consequently oil sands companies do not have tools from which the success of created marshes can be compared. We used remotely sensed surrogate measurements to estimate this variation.I examined a decade of water level variation and resulting zone area fluctuation in created and natural marshes of the Fort McMurray region. Archival aerial photography allowed me to follow the same marshes over time. My main questions were:

1) What is the range of SAVZ area variation characterising the natural marshes of the Fort McMurray region? Hood and Bayley (2008) studied the relationship between the area of SAVZ and weather variables in marshes of the mixed-wood region (east-central Alberta, Canada) bwetween 1948 to 2002. They observed that weather patterns can cause the SAVZ of natural marshes to increase almost fivefold following period of higher precipitation and lower temperature. Similar patterns were anticipated for the natural marshes of the Fort McMurray region.

2) Are hydrological patterns of created marshes similar to those of natural marshes? Many created marshes present in the oil sands post-mined landscape have a bowl-shaped basin morphometry rather than the characteristic pan-shaped morphometry of most natural marshes of the region (see Chapter Two fore more details). It was thus expected that, compared to natural marshes, the bole-shaped basin morphometry of created marshes constrained the expression of water level variation resulting in a narrower range of SAVZ and VZ area variation over time.

3) Do weather factors and marsh basin morphometry influence zone area fluctuation in created and natural marshes? Based on findings of Hood and Bayley (2008) and Gray and Landine (1988, 1985) it was expected that the area of SAVZ and VZ of natural and created marshes varies according to weather patterns. For example, a period of higher precipitation (e.g. snow, rain), and/or lower temperature (reduce evapotranspiration) was expected to result in an increase of water in marshes (Hood and Bayley 2008). This increase of water availability was expected to result in a greater SAVZ area and a reduced VZ area in both natural and created marshes. Based on Brooks (2002) who observed that the presence of surface water in a vernal pool was positively correlated with its volume, it was hypothesised that marshes with higher surface area:volume ratio would have a more variable SAVZ.

4.2. Data acquisition, development and statistical analyses

4.2.1. Selected sites

A total of four natural and seven created marshes were selected for comparison based on access and complete annual records for the years 1967, 1989, 1990, 1998, 2001, 2005 and 2008. To provide as wide a range of reclamation benchmarks possible and to include a potentially greater range of weather conditions, natural marshes were studied for a longer period of time (i.e. 1967, 1989, 1990, 1998, 2001, 2005, 2008) than created marshes (i.e. 1998, 2001, 2005, 2008). Direct comparisons between natural and created marshes were however made based on the yearly data common to both natural and created including: 1998 , 2001, 2005, 2008.

The natural marshes represented the typical marsh-type wetlands found in the boreal plain ecozone. The natural and created marshes selected for the study were situated in the Fort-McMurray region (560 58’ 25.90” N, 1110 34’ 43.63” W) in Alberta, Canada. The Fort-McMurray region is located in the north-west part of the boreal region (see Brandt (2009) for an overview of the North American boreal region). The un-disturbed land of this region is formed by a mosaic of uplands (varying from aspen-dominated deciduous forest to spruce–fir–pine-dominated conifer forests) and wetlands communities (bogs, fens, larch and spruce swamps and marshes) adapted to long, cool winters and short, cold summers (Vitt and Bhatti 2012). The distances between the selected marshes was maximized (within marsh types) to reduce chances of spatial-autocorrelation and pseudo-replication (Table 1The natural marsh selected for this study were characterised by: 1) a natural origin, 2) no evidence of anthropogenic disturbance, 3) evidence of beaver activities (i.e. presence of lodges, dams or cleared vegetation), and 4) having two vegetative zones; SAVZ, and an emergent-wet-meadow (hereafter named vegetative zone (VZ)). The created marshes selected were characterised by 1) an anthropogenic origin, 2) an amendment with oil sands process water (OSPW), oil sands process material (OSPM) and/or peat-mineral-mix (PM), 3) a location in the post-mined landscape and 4) the presence of two vegetative zones (i.e.

SAVZ, and VZ) (see Appendix C for more details about each selected marsh). Although natural marsh ages were unknown they were assumed to be >1000 years old (Raab and Bayley 2012) and representative of the post-glacial landforms. Aerial photographs confirmed that all natural marshes studied were present in 1957. Natural and created marshes were assumed to have been affected by the same climatic conditions over time.

Table 4- 1: Natural and created marshes selected for the study with their location. N=natural and C=created marshes.

Marsh	**Location**	
(Names)	**Latitude**	**Longitude**
N1 (Broken Wing)	56°58'08.55"N	111°41'12.28"W
N2 (Dam wetland)	56°56'36.89"N	111°38'49.55"W
N3 (Jumping Dog)	56°56'19.08"N	111°39'41.85"W
N4 (Southwest Sands Beaver)	56°59'10.03"N	111°42'48.76"W
C1 (MFT-South)	56°59'32.83"N	111°31'55.59"W
C2 (Jan's Pond)	56°59'27.63"N	111°32'20.07"W
C3 (Natural wetland)	56°58'50.10"N	111°30'38.56"W
C4 (Test Pond 10)	57° 5'01.60"N	111°41'27.06"W
C5 (Test Pond 7)	57° 5'04.92"N	111°41'35.86"W
C6 (Mike's Pond)	57° 6'41.29"N	111°40'52.65"W
C7 (Peat Pond)	56°59'37.82"N	111°37'24.74"W

4.2.2. Aerial photo analyses

Aerial photos were provided by Syncrude Canada Ltd. and by Alberta Environment and Sustainable Resource Development (ESRD) Reference Library. The photos selected were taken in the months of July and August when standing vegetation was near peak standing crop. Each photo was scanned at a minimum resolution of 600 dpi (dots per inch) as greyscale images.Each aerial photo was geo-referenced using an orthorectified satellite image from the year 2007. The scale of the aerial photos was of 1:20 000, and orthorectified satellite images had a resolution (pixel size) of 1m^2 or 2 m^2. Using ArcGIS (v.10 ESRI), each marsh was divided into an SAVZ and VZ polygon. Using ArcGIS, areas were calculated for each polygon (i.e. zone) delineated. Each marsh was visited several times in 2008 and 2009 to validate the accuracy of measurements obtained using ArcGIS. Although it was hypothesised that VZ would vary negatively and in proportion with SAVZ area (e.g. if SAVZ area increased by 20%, VZ area would decrease by approximately 20%), our results do not clearly demonstrate this trend. This may be explained by the lack of precision and consistency when delineating VZ polygons using ArcGIS. Indeed, the VZ outside boundary was considered as ending where sedges and grasses were replaced and dominated by shrubs or trees. However, due to the photo resolution, determining when shrubs and trees were the dominant cover may have led to a lack of precision among marshes and years.

Prior to analysis, data were inspected for normality, skewness, kurtosis and outliers. Q-Q plots were used to visualize data distribution and variances and Shapiro-Wilk normality test was also performed on the data. The relationship (linear versus non-linear) between independent variables and dependent variables were examined using plots of residuals. Multicollinearity among variables was tested prior to analysis. When collinearity existed among variables, the variable the most strongly correlated with the dependent variables was selected and included in the models.

To compare the relative contribution of the SAVZ and VZ to total marsh area at each site for a given year, the SAVZ and VZ areas were transformed into proportions. For example, if the total marsh area for a given year was of 1000 m^2, and both the SAVZ and VZ areas were 500 m^2, the proportion occupied by the SAVZ and the VZ that year was of 50% each. To provide an estimate of the average proportion that occupies the SAV and VZ in each marsh type (i.e. created versus natural), SAVZ and VZ were averaged by marsh type for a given year and for each year under study.

The SAVZ and VZ area varied greatly among marshes regardless of their origin (i.e. natural or created). To facilitate the comparison among marshes, the variation in the area of each individual marsh was expressed relative to pre-determined earliest measurements (i.e. baseline year: both 1967 and 1998 for most natural marshes and 1998 for most created marshes). Thus, area variation over time was expressed as a ratio of each site by year compared to its earliest area measurement. A value >1 indicated an increase in area whereas a value <1 represent a decrease in area.

4.2.3. Variables influencing marsh SAVZ area

To examine factors influencing the area of SAVZ over time, a series of independent variables were selected based on data availability and previous literature including Devito et al. (2005) and Hood and Bayley (2008). Hence, regional weather (e.g. temperature and precipitation) and ratios of marsh surface area: volume were selected as candidate factors (Table 4.2). Temperature and precipitation data were obtained from Environment Canada (climate station ID: 3062693, position: 56°39'00.00" N, 111°13'00.00" W). Four climatic intervals were selected. Based on Hood and Bayley (2008), the influence of lagging climate data by three months, one, two and three years was examined for effects on marsh SAVZ area. To avoid including extreme weather events that may have occurred after the aerial photos were taken (i.e. after August or September depending on aerial photos), climate variables were calculated as *x* months preceding to the month the aerial photos were taken (Hood and Bayley, 2008). For example, if an aerial photo was taken in September 2008, annual climate values were calculated from September 2007 to August 2008. The same method was applied to represent the effect of shorter and longer climatic intervals (i.e. three months, two and three years) on marsh area.

A stepwise multiple regression[14] analysis was performed to help select the minimal adequate model that explained the greatest amount of variation of the SAVZ area. The response variable (SAVZ area) was calculated by averaging the SAVZ area of marshes for each year studied. Before analysis, data were inspected for normality and homoscedasticity and transformed when necessary. Possible collinearity between the selected independent variables was verified prior to analysis. When collinearity was found among variables, the variable with the strongest relationship with SAVZ area

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[14] Both backward and forward stepwise multiple regressions were tested and the results obtained with the two methods were not different.

variation was selected. The multicolinearity among the independent variables of total snow, total precipitation and total rain led to the exclusion of total precipitation and total rain from the multiple regression analysis. The multicolinearity between mean maximum temperature and mean temperature led to the exclusion of mean temperature. To determine if natural and created marsh SAVZ areas responded differently to the variables, two multiple regression analyses were run, one including only natural marshes and another including only created marshes.

To determine the relationship between SAVZ area and marsh ratio surface to volume, a simple regression analysis was performed. A surface area to volume ratio was estimated for each marsh. Using aerial photos from 2008[15], the total surface area of each marsh was estimated from the polygons measured with ArcGIS. Marsh average depth was calculated from averaging depths measured in the field in each marsh[16]. The volume of each marsh was approximated by multiplying its surface area by its average SAVZ depth.

Table 4- 2: A series of independent variables potentially influencing SAVZ area were selected and used to perform the multiple linear regression analysis.

	Time lag			
Independent Variables	3 months	1 year	2 years	3 years
Mean Maximum Temperature	X	X	X	X
Mean Minimum Temperature	X	X	X	X
Mean Temperature	X	X	X	X
Rainfall	X	X	X	X
Snowfall		X	X	X
Total Precipitation		X	X	X

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[15] The year 2008 was used because the water depth was also measured in the field in 2008.

[16] In the field, each marsh was visualy divided in four quadrants, within each quadrant, one or two transects were randomly positioned. Each transect crossed perpendicularly each zone (i.e. SAVZ and VZ). Along each transect and in each zone, water depth was measured. In total 18 locations (i.e. 6 in the SAVZ ans 12 in the VZ) within a marsh was assessed for water depth.

4.3. Results

4.3.1. Marsh area variation over time

In natural marshes the contribution of VZ area to total marsh area was greater than the contribution of the SAVZ area (Figure 1) for each year assessed. Between 1967 and 2008 the average contribution of the SAVZ area to the total marsh area was of 31% (n=4, SD=3.6) with a range from 13% (n= 4, SD=2.5) in 2001 to 40% (n=4, SD=12.8) in 1998. Thus over time the VZ occupied an average 69% (n=4, SD=3.6) of the total marsh area. The VZ lowest contribution was measured in 1998 and averaged 60% (n=4, SD=12.8) and the highest value was recordedin 2001 at 87% (n= 4, SD=2.5). Unlike natural marshes, created marshes' total area was dominated by an extensive SAVZ in all years. Between 1998 and 2008, the average contribution of the SAVZ aera to the total marsh area was of 74% (n=7, SD=1.4) varying from 71% (n=7, SD=5.6) in 2001 to 77% (n=7, SD=8.2) in 1998. Thus, VZ area occupied on average, 26% (n=7, SD=1.4) of the total marsh area. The VZ lowest contribution was measured in 1998 with 23% (n=6, SD=5.6). The VZ highest contribution was measured in 2001 at 29% (n= 7, SD=2.5). Thus created marshes had more open-water area (i.e. SAVZ) than did natural marshes.

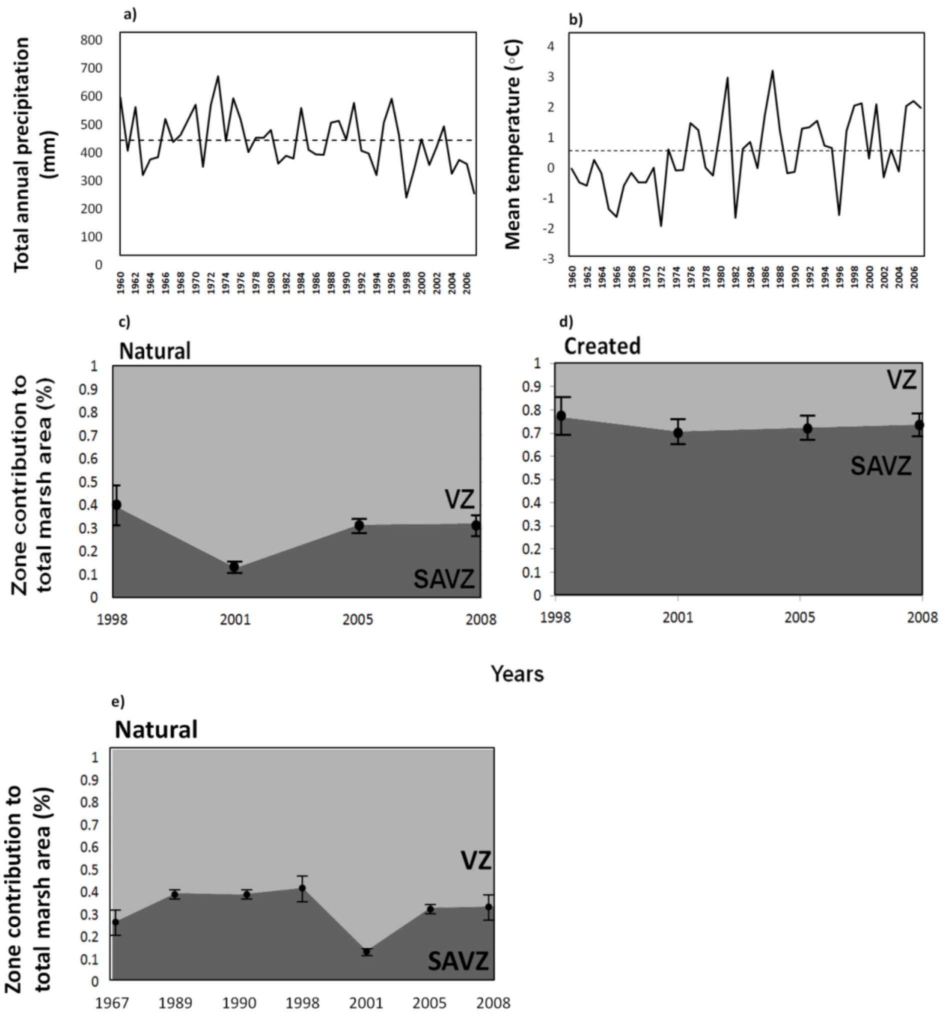

Figure 4- 1: a) Annual mean maximum temperature and b) total annual snow in Fort McMurray, AB (Canada), the contribution over time (1998 to 2008) of the SAVZ and VZ zone area to the total marsh area in c) natural (n=4) and d) created marshes (n=7), and the contribution over time (1967 to 2008) of the SAVZ and VZ zone area to the total marsh area in e) natural marshes (n=4). For each year, the average area (±SD) of the OW and VZ zone contributing to the total marsh area was calculated for each marsh type.

4.3.1.2. Variation of marsh zone area relative to earliest measurement year

The maximum areal increase in natural marsh SAVZ was in N2 in 2008 and showed a 6.3-fold increase over the1967 baseline measurement (Figure 2). The maximum areal decrease was measured in 2008 and was 0.9 times smaller than the 1967 baseline measurement.

The maximum increase in natural marshes VZ was in 2009 with 2.9 times baseline measurement. The maximum decrease was 0.5 in 2001. The VZ area of natural marshes tends to vary less over time than the SAVZ area. Although hypothesised that VZ would vary negatively and in proportion with SAVZ (e.g. if SAVZ area increased by 20%, VZ area would decrease by approximately 20%), our results do not clearly demonstrate this trend. This may be explained by the lack of accuracy and consistency when delineating VZ polygons using ArcGIS. Indeed, the VZ outside boundary ended where sedges and grasses were replaced and dominated by shrubs or trees. However, due to the photo resolution, determining when shrubs and trees were the dominant cover may have led to a lack of acuracy among marshes and years.

For created marshes, the maximum increase in SAVZ relative to their first measurement was of 1.2 fold in 2001 and 2005. The maximum decrease was 0.6 fold the baseline measurement and was measured for 2001, 2005, 2009. For the VZ of created wetlands, the most important decrease observed relative to 1998 was of 0.4 fold in 2008. The maximum increase of VZ area was noted in 2008 and reached 4.6 fold.

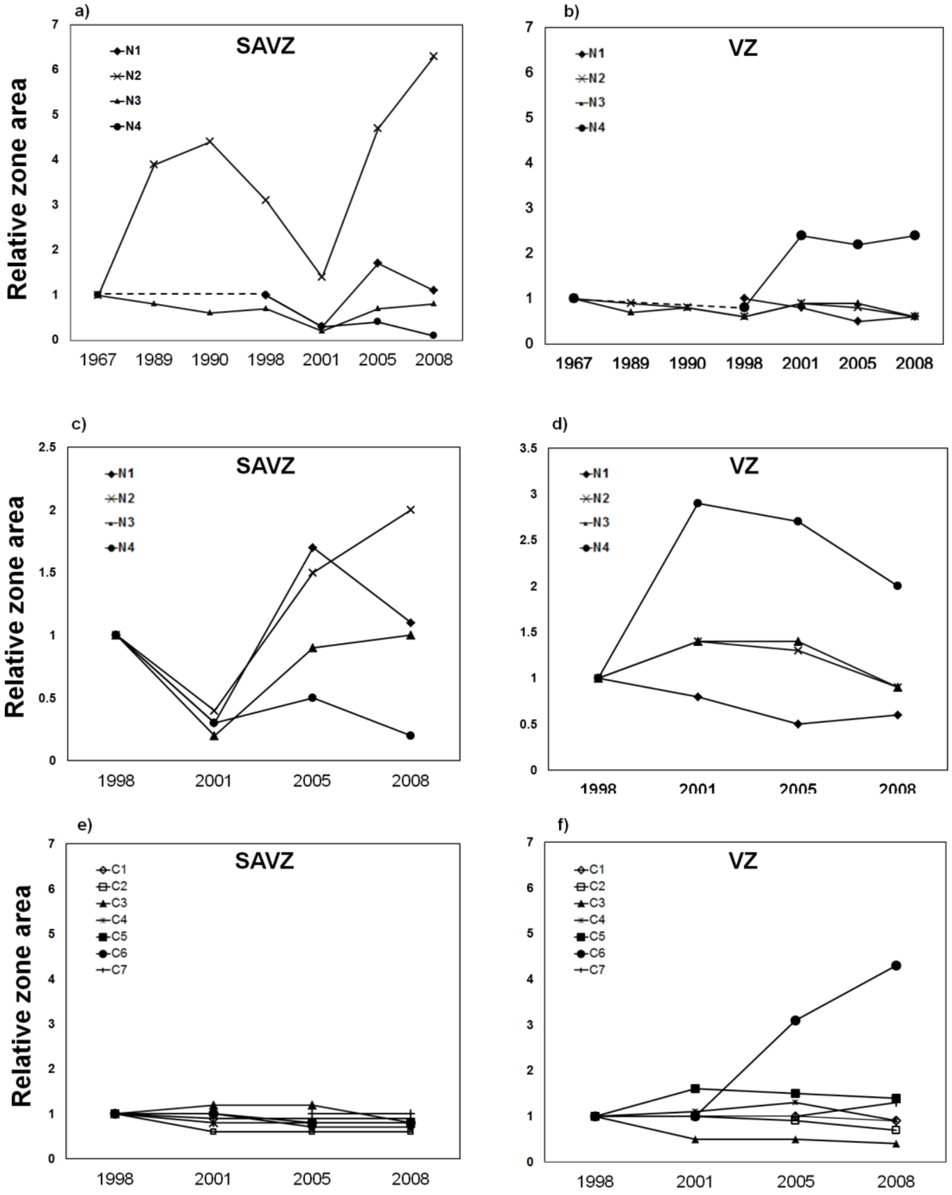

Figure 4- 2: Individual marsh area variation over time relative to their first measurement. A value >1 indicated an increase in area whereas a value<1 represent a decrease in area. A value of 1 means that area was constant. **a)** Natural marsh SAVZ area variation over time relative to 1967, **b)** natural marsh VZ area variation over time relative to 1967, **c)** Natural marsh SAVZ area variation over time relative to 1998, **d)** natural marsh VZ area variation over time relative to 1998, **e)** created marsh SAVZ area variation over time relative to 1998, **f)** created marsh VZ area variation over time relative to 1998.

4.3.2. Relationships of weather and surface to volume ratios with variation in SAVZ area

Multicolinearity was found among the independent variables total snow, total precipitation and total rain. Thus, total precipitation and total rain were excluded from the multiple regression analysis. Multicolinearity was found between mean maximum temperature and mean temperature. Thus, mean temperature was excluded from the analysis. The minimal adequate model that explained the greatest amount of variability for the SAVZ area included the variable annual mean maximum temperature and annual total snow (r^2=.0.81 p<0.01) (Table 4-3 Table 5)(Figure 3).

Table 4- 3: The most parsimonious model (i.e. lowest AICc) that explained the greatest amount of SAVZ area variation included both annual Mean Maximum Temperature (Y1MMT) and annual Total Snow (Y1TS).

Best model	**Variance explained** (%)	**p-value**	**AICc**	**Model**
Y1MMT+Y1TS[†]	81	0.01	130	y=18593.55 -2112.70(Y1MMT)+ 30.35 (Y1TS)

[†]=Standardized coefficient of variation Y1MMT=0.52, and Y1TS=0.46.

A significant relationship between the ratio surface area to volume of marshes and SAVZ area was found (R=0.8, n=11) suggesting that as the area to volume ratio increases, marsh SAVZ area variation increases (Figure 4). The averaged surface area ratio to volume for the studied natural marshes was of 4:1 (n=4), whereas it was of 2.5:1 (n=7) for created marshes. This surface to volume characterization confirmed field observations that created marshes were less pan-shaped and more bowl-shaped.

No relationship was found between the climate variables and the SAVZ area of created marshes.

Table 4- 4: The weather variables selected and tested to determine their relationship with SAVZ area variation in natural and created marshes. Values were obtained from simple linear regressions that included only one predictor variable. Variables are ordered using the analysis of natural marshes and from 1) the most to the least influential time lag category and 2) the highest to the lowest R-squared values.

	Natural Marshes		Created Marshes	
Independent variables	r^2	p-value	r^2	p-value
Annual				
Mean maximum temperature	**0.7**	**0.01**	0.0	0.96
Total snow	**0.7**	**0.02**	0.1	0.77
Mean temperature	**0.6**	**0.03**	0.0	0.97
Total precipitation	0.3	0.21	0.0	0.88
Total rain	0.1	0.62	0.0	0.84
Two years				
Mean maximum temperature	**0.6**	**0.03**	0.6	0.22
Total snow	**0.6**	**0.03**	0.0	0.94
Mean temperature	0.5	0.09	0.2	0.52
Total precipitation	0.2	0.31	0.1	0.78
Total rain	0	0.79	0.0	0.8
Three years				
Total snow	0.5	0.07	0.1	0.65
Mean maximum temperature	0.4	0.13	0.6	0.25
Mean temperature	0.3	0.21	0.4	0.39
Total precipitation	0.1	0.42	0.2	0.57
Total rain	0	0.68	0.2	0.53
Three months				
Total rain	0.2	0.34	0.0	0.96
Mean maximum temperature	0	0.76	0.2	0.75

Mean temperature	0	0.82	0.0	0.88

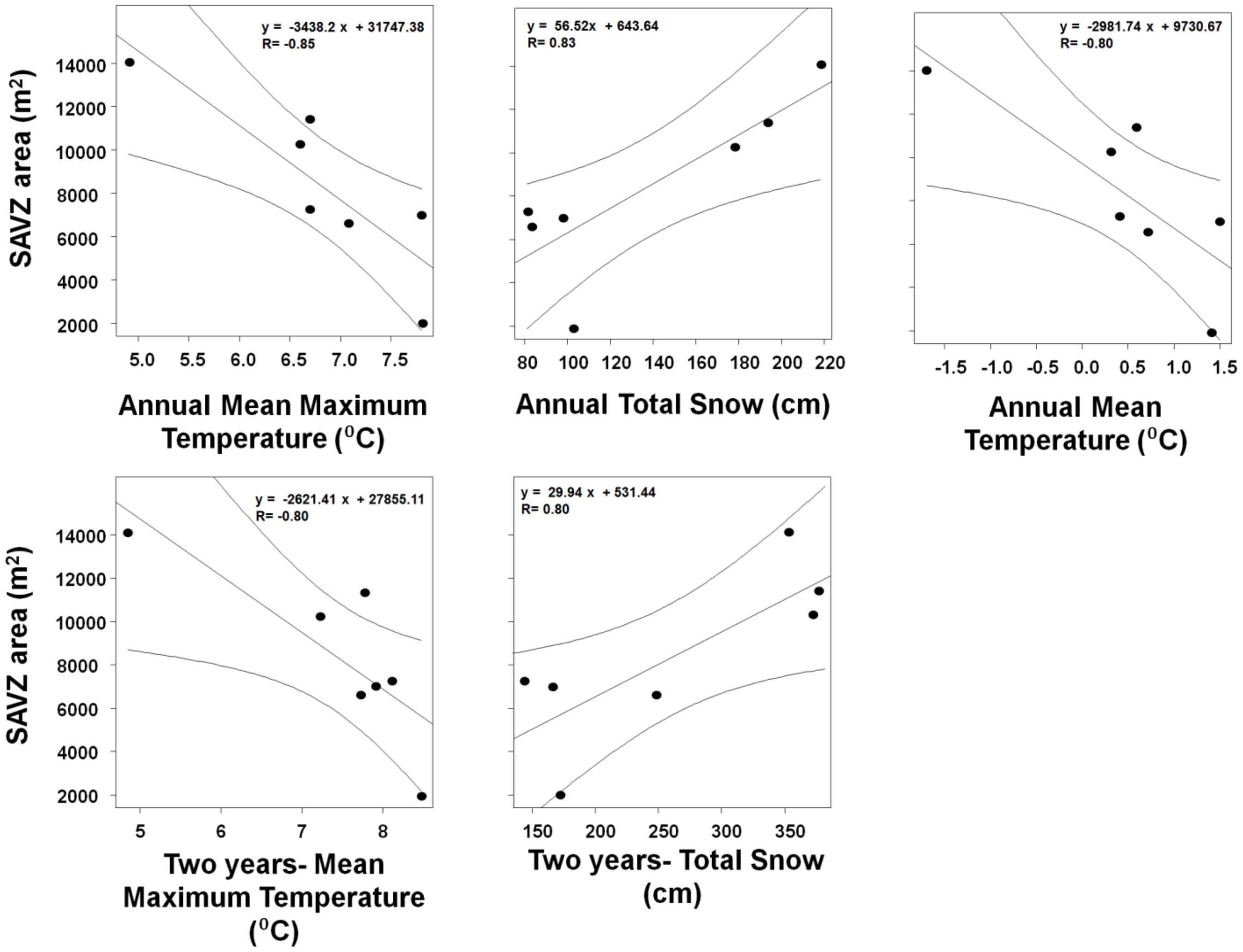

Figure 4- 3: Climatic variables having a significant relationship (R>0.5, p<0.05) with SAVZ area. The correlation coefficient (R) indicates the strength and direction of the relationship between variables.

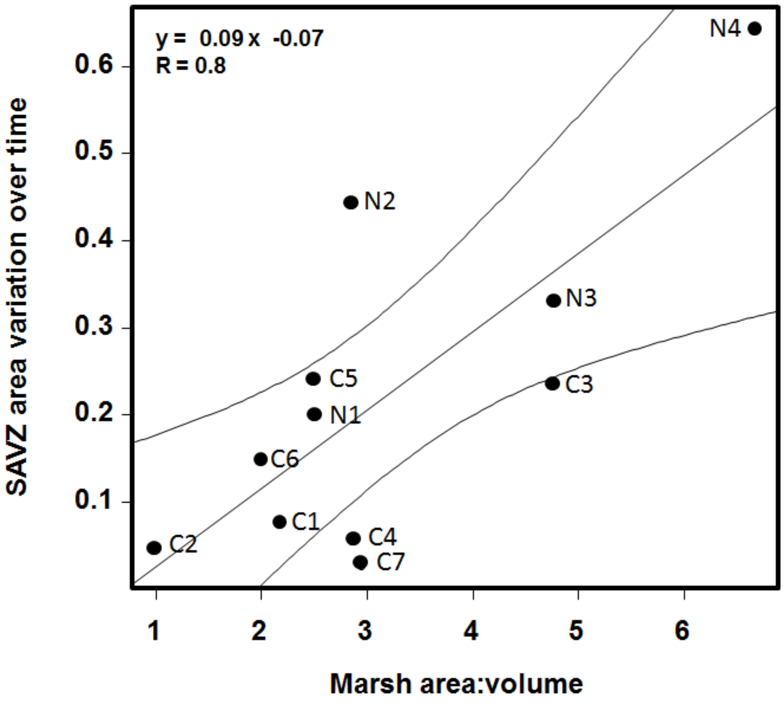

Figure 4- 4: Marshes with higher surface area to volume ratio have more variable SAVZ. Each data point represents one natural (N) or created (C) marsh. The curved lines represent the confidence interval about the estimated regression coefficient (CI=95%).

4.4. Discussion

I have demonstrated that the SAVZ of natural marshes of the Fort McMurray region vary over time following weather patterns. The area of the SAVZ of natural marshes tripled between the driest and warmest years and the coolest and wettest years. On average and over time, the SAVZ area of natural marshes occupied approximately one third of the total marsh area. In comparison, the area of the SAVZ of created marshes was more stable over time and on average occupied almost three-quarters of the total marsh area.

Most of the created marshes under study had a bowl-shaped basin morphometry rather than the pan-shaped morphometry characteristic of most natural marshes of the region. Thus, compared to natural marshes, the bowl-shaped basin morphometry of created marshes constrained the expression of water level variation resulting in a narrower range of SAVZ and VZ area variation over time. The “over-stabilisation” of created marsh zone area will affect their functions, thereby yielding outcomes that differ from those provided by natural marshes of the region. Different community outcomes also yield different evaluations of reclamation success. For example, stable SAVZ may reduce niche availability and hence species richness (Boers and Zedler, 2008). Stable SAVZ may also reduce and extend periods where soil is under anoxic conditions, thereby influencing decompositional processes and nutrient cycles (Wisheu and Keddy 1992). The stable SAVZ area of created marshes can be expected to benefit aquatic organisms such as invertebrates and ultimately fish while simultaneously disadvantaging organisms dependent on emergent plants such as birds that require cover, forage and nesting areas in emergent marshes. Other anticipated shortcomings of SAVZ expansion at the expense of VZ include poorer performance in shoreline stabilisation, rooting zone aeration, and durability to large swings in water chemistry.

Finally, I have demonstrated that weather factors and marsh basin morphometry influence zone area fluctuation. I identified variables that significantly influence marsh zone variation. Hence our results can provide guidance in the creation of marshes in the post-mined landscape and tools to predict SAVZ under different weather scenarios. Mean maximum temperature (annual and two-years lag), mean temperature (annual), total annual snow (annual and two-years lag) are significantly influenced natural marsh SAVZ area variation. Using the annual mean maximum temperature coupled with the annual total snow is however the simplest way to predict SAVZ area within natural marshes and provide reference for created marshes. In addition, marsh surface area to volume ratios were found to significantly influence the variation of SAVZ area. Our results support those of Brooks (2002) who observed that the presence of surface water in a vernal pool was positively correlated with its volume.

Marsh zonal areas tend to vary over time and between marshes regardless of type. Thus, zone area variation of an individual marsh is believed to also be attributable to their historical and local specificity.

4.4.1 The effect of extensive and stable SAVZ area in created marshes

The natural marshes assessed were permanent (i.e. had permanent water in SAVZ) (Stewart and Kantrud 1971). Similarly, created marshes also supported a permanent SAVZ. However, the amplitude of SAVZ area variation differed by marsh type. Hydrologic processes are the driving factors of biotic community structure, productivity and nutrient cycling within wetlands (Coops et al. 2003). We contend that the extensive and stable zonation of created marshes will support biotic and abiotic structure, functions and services that differ from natural marshes of the region. Differences in variation of SAVZ area between natural and created marshes typically lead to differences in biochemical processes and hence functions (Bridgham et al. 2006). Because biochemical processes are sensitive to soil redox conditions (Hefting et al. 2004) leading to specific nutrient availability and vegetation responses, water levels can drive plant community structure. Thus, the hydrologic processes characterising created marshes may results in an inability to reclaim marshes with capability equivalent to natural marshes.

Created marshes in the study region supported lower vegetation diversity and a different vegetation composition sometimes composed of less desirable dominant species such as *Typha latifolia* or upland plants species (Trites and Bayley, 2009; Rooney and Bayley 2009; Raab and Bayley, 2012). These differences in vegetation were mainly attributed to sediment moisture content and sediment quality and the isolation of marshes in the reclaimed landscape restricting early colonisation of less mobile propagules. Plant community differences may be attributed to reduced effect of water level fluctuation on SAVZ area in created marshes. Keddy and Reznicek (1986) showed that low water periods in the Great Lakes (Canada), allowed many plant species and vegetation types to regenerate from buried seeds in exposed shoreline areas, enhancing vegetation diversity. Furthermore, Casanova and Brock (2000) and Boers and Zedler (2008) showed that extended inundation periods led to reduced vegetation richness and favored the dominance of certain undesirable species such as *T. latifolia*. In this study, the dominant SAVZ area relative to the VZ in created marshes may also produce lower habitat quality for certain aquatic insect and birds. During marsh transitional stages, when water level increases and the ratio SAVZ to VZ approach 50:50, marshes reached their highest insect and bird productivity (Weller and Spatcher 1965). Thus, the stable and dominant SAVZ area in created marshes may decrease the spatial and temporal complexity of the whole biotic community structure and its related functions. Although the pattern seems to hold, the causal linkages of these hypotheses remain to be tested.

4.4.2. Natural marsh SAVZ vary according to regional and local factors

While natural and constructed marshes are assumed to be subject to similar continental and regional conditions (e.g. bedrock geology, annual precipitation, average temperature), each individual marsh, regardless of origin (i.e. natural or created), was affected by a range of different and sometime unique local conditions.

Natural marshes' zone areas tend to be correlated with temperature and total snow. Our results suggest that the natural marshes SAVZ variation is dominated by snowmelt rather than rain. My results are consistent with those of Gray and Landine's (1988, 1985) studies of prairies marshes in Saskatchewan showing that spring snowmelt accounted for approximately 80% of annual local surface runoff. Importantly, snowmelt runoff in early spring occurs as a result of frozen mineral soils and a relatively rapid water release from melting snowpack. Although on average the Fort McMurray region receives more rain than snow, it is assumed that most rainfall is consumed by evapotranspiration and has limited influence on SAVZ fluctuation. In addition, zones may be established in early spring in response to establishment water. Later in the season the summer rains may raise or lower the water level underneath the taller mature plants with little zone influence.

However, one important factor influencing marsh SAVZ area could not be included in this study due to a lack of data. Hood and Bayley (2008) found that beavers (*Castor canadensis* Kuhl) have very important influences on the creation and the maintenance of natural marshes of the mixed-wood boreal region of east-central Alberta, Canada. This scenario may be applicable to the marshes of the Fort McMurray region as all of the natural marshes assessed in our study had past or present evidence of beaver activities.

From 1998 to 2008 the SAVZ area of marsh N4 significantly decreased while the VZ doubled in area. This significant and sudden increase of VZ area was not observed in any other natural marshes studied. This isolated trend was assumed to be due to local rather than regional factors. Alteration of the surrounding landscape (i.e. alteration of the direction of water flow) and the departure of beavers from N4 may have resulted in a precipitous drop in water level and a commensurate reduction of the SAVZ area and an expansion of the VZ. This was verified in the field in 2009. Although no signs of current beaver activities were found, evidence of past beaver impoundment were present. Assuming that the earlier maintenance of the SAVZ area was mainly the result of beaver activities, the evacuation of beavers in N4 caused the SAVZ area to decrease of 65%. Although not measured directly in our study, beavers appear to play a crucial role in the creation and maintenance of natural marshes. Beavers are actively removed from many created wetlands on the mine site and in others insufficient shoreline forage exists to maintain colonies.

4.4.3. Factors influencing variation in created marsh SAVZ area

Although SAVZ areas of created and natural marshes were at their minima in 2001, no significant relationship was found between weather and created marshe SAVZ area variation. The typically steep slopes of created marshes make detection of water-level variation (shoreline change) difficult to measure using remote sensing methods even with large changes in stage. Hence, it is possible that the water-level in created marshes may have varied significantly with weather but this variation couldnot be detected using remote sensing method. Another possibility is that created marsh water–levels are relatively stable due to their modified water budgets that have high storage capacity and reduced evaporation/transpiration rates

4.4.3.1. Water budgets of created marshes

Most created marshes studied are hydrologically isolated. Hence, for these marshes, precipitation, surface (runoff) and subsurface inflows may be the most significant source of water income. Due to their hydrologic isolation in the landscape, marsh water-level varies independently from each other and their variation may reveal their unique local conditions such as the topographic setting, morphometry and the heterogeneity of surface material used during reclamation (e.g. overburden, sands, clay, peat). Despite their unique conditions, a few generalities are apparent. Compared to natural marshes, where vegetation cover is higher and the surrounding environment is dominated by forests, created marshes are generally younger and support less vegetation cover. Their periphery is usually dominated by grasses, shrubs and young trees yielding reduced evapotranspiration (Zhang et al. 2001). The lower evapotranspiration rate of the post-mined landscape and a preponderance of sealed soil surfaces may also increase water runoff into created marshes. For the same reasons, interception of precipitation by surrounding vegetation is assumed to be low. Interception by vegetation is known to vary with vegetation structure (e.g.leaf area, rooting strategy) and composition (Milly, 1997; Kergoat, 1998; Gertena et al. 2004). Rills formed from water runoff were observed along the steeper slopes of some unvegetated created marshes such as C2 and C6 and subsurface pipe-flow channels were evident. Furthermore, the created marshes studied had narrow VZ . Thus, due to their surroundings in early developmental stages and their abrupt basin morphometry, created marshes are expected to have elevated water inflows.The extensive VZ length of most natural marshes and their flat forested edges reduced the speed and the amount of lateral and vertical water inflow in marshes.

Evaporation rates of created marshes may be depressed as well due to low surface to volume ratios. The effects of atmospheric agents (e.g. wind, air temperature and humidity) that drive evaporation can be reduced though it is difficult to evaluate quantitatively due to the overriding effects of evapotranspiration, ground-water exchange, and climate for example (Brook and Hayashi, 2002).

4.5. Study Limitation

Hood and Bayley (2008) observed that beavers can mitigate the effect of climate on the area of the SAVZ in boreal wetlands. The natural marshes included in this study were characterised by the evidence of beaver activities (i.e. presence of lodges, dams or cleared vegetation). Hence, the effect of weather on the area of SAVZ may be different in constructed marshes inhabited by beavers.

Some of the created marshes included in my study were not constructed to be marshes but instead were designed to anticipate effects of biogeochemical processes expected to be found in end-pit lakes. These end-pit lakes are not realistic or meaningful models of current reclamation practice. They were however included in my study to increase the sample size of CT marshes assessed and to better understand the influence the basin morphometry on water level variation.

4.6. Recommendations

The geomorphometry of the reclamation landscape should be the initial focus of reclamation because it constrains the hydrology of wetlands (Euliss et al. 2008). The hydrology and its fluctuation over time and space constrains the abiotic and hence the biotic features of wetlands (and hence functions).

To avoid creating over-stabilised SAVZ area, future construction of marshes should ensure that created marshes are characterised by a low surface area to volume ratio, which means basin morphometries that are shallower and more pan-shaped than bowl-shaped. Although storage capacity may be compromised and surface variation increased, these are the very dynamics that drive important wetland functions such as detrital decomposition, off-gassing of sulphur compounds, shifting oxygen tensions that help break seed dormancy, and consolidation of sediments. Shallow inundation of larger areas may be accomplished with lower precipitation or runoff volumes too. Reclaiming the right hydrologic processes and related marsh zone area fluctuation within created marshes, will increase vegetation diversity, and enhance the whole biotic community structure and its related functions.

I suggest that to enhance the success of wetland reclamation in the post-mined landscape, reclamation should adopt a process-based approach. Euliss et al. (2008) and Wilcox (2012) suggested that to restore wetland function, the traditional structure-based approach (i.e. restoring wetland area, size, plant cover etc.) often used in restoration should be replaced by a process-based approach (i.e restoring processes such as nutrient cycling, hydrology etc.). However, processes are dynamic and vary according to local and regional conditions such as weather, climate, and/or biotic factors (e.g. beaver, human activities)(Euliss et al. 2008). The processes and related functions that must be reclaimed have to be defined based on reference analogues sites (Vitt and Bhatti 2012) that provide the selected and desirable processes and functions. Thus, reference analogue sites must be studied on a long enough period of time, and large enough scale (although relevant), to provide a range of reference conditions (i.e. range of processes and functions values under different (weather, climatic, biotic etc.) scenarios) that can be used as benchmarks to a given restoration project (Wilcox 2012).

Research that aims to determine the effects of water level change on salt concentrations in soil and/or surface waters must be conducted. My recommendations support those of Cooper (2004): creating saline marshes (resulting to their direct or indirect exposition to OSPM/OSPW) that mimicks the drawdown regimes of natural marshes may create conditions (e.g.salt crusts) that can adversely affect recruitment, growth and survival of non-salt tolerant vegetation.

Models that quantify the role of beavers in this region must be developed. The effects of their activities in the post-mined landscape may override the effect of weather (Hood and Bayley 2008) and may challenge predictions and expectations.

4.7. Conclusion

I used remotely sensed surrogate measurements to estimate the area variation of the SAVZ and VZ of natural and created marshes. I have quantified the long-term variation of the area of the SAVZ and VZ variation in natural marshes in this region and determined the main factors that drive these patterns. The models provide managers and regulators with a range of benchmarks and tools from which the success of created marshes can be assessed. I have further demonstrated that compared to the pan-shape basin morphometry of natural marshes of this region, the bowl-shaped basin morphology of the created marshes restrict the expression of water level variation. I suggested that to avoid creating over-stabilised SAVZ area, future construction of marshes should ensure that created marshes are characterised by a low surface area to volume ratio which means basin morphometries that are shallower and more pan-shaped than bowl shaped.

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CHAPTER 5

General discussion and conclusions

My thesis' main objective was to provide reclamation benchmarks and references by which created marshes of the Fort McMurray region could be compared and reclamation practices critically tested and adjusted. More precisely, I 1) used theories of plant community ecology to describe and contrast patterns of vegetation composition, functions and development in created and natural marshes and 2) identify factors that shaped the observed patterns. Below, I provide a thesis summary where I describe my contributions for each chapter.

In Chapter One I set the context for my four data chapters by providing an introduction and a synthesis of the major paradigms of plant community ecology where past and current accomplishments of this field of study were reviewed. I made a strong case for using natural or opportunistic reclamation settings to both apply and test established theories of plant community ecology. Using the oil sands of Alberta as the research context, I suggested avenues of investigation that contribute broadly to the fields of reclamation and community ecology. Through that synthesis, I underlined which of my chapters will contribute to community ecology, and how my results will contribute to the oil sands of Alberta reclamation.

The general objective of Chapter Two was to identify and describe environmental and plant assemblage patterns present in different types of created (i.e. created-tailings (CT), created-unamended (CU), and created-peat (CP)) and natural marshes of the Fort-McMurray region. I hypothesised that the *a priori* categorisation of marshes based on their origin (created or natural) and their amendment types would result in significant differences in environmental (chemical and physical) conditions and thus, growing conditions. Following the differences in growing conditions, marshes would be characterised by different species diversity and composition. Three main questions were thus investigated:

1) Are there patterns of environmental conditions among different marsh types?

I have demonstrated that CT, CU, CP and natural marshes under study were characterised by distinct environmental conditions. Some environmental differences (e.g. water salinity, and DO) were attributable to the types of amendments received, while other environmental differences (e.g. sediment organic content content and total nitrogen, basin slope, zone width) were attributable to marsh origin (i.e. constructed versus natural). In general, CT marsh environmental conditions were the most dissimilar to natural marshes. These environmental dissimilarities among marshes provide different abiotic filters (sensu van der Valk 1981) to biotic communities (Gotzenberger et al. 2012; HilleRisLambers et al. 2012). Using natural marsh environmental conditions as analogues (Vitt and Bhatti, 2012), I have demonstrated that created marshes maintain different filters on the plant species pool of the region. My second objective for Chapter Two was thus to describe vegetation community patterns in different types of marshes (i.e. CT, CU, CP, and natural) and secondly, to investigate the effects of amendments (i.e. OSPW/OSPM, natural water, no-amendment, and PM) on plant community composition. My two main questions were:

2) Are there generalisable patterns of vegetation richness and composition among different marsh types?

When compared to created marshes, the plant species richness of natural marshes tended to be higher in all three zones (i.e. submersed aquatic vegetative zone (SAVZ), emergent (EZ) and wet-meadow (WMZ)). Compare to natural and CU marshes, the SAVZ of CT marshes had significantly lower plant richness. Thus, it was assumed that the addition of OSPW and/or OSPM significantly reduced species richness in the SAVZ. In the EZ, natural marshes were characterised by higher vegetation richness than both CT and CU which presented similar species richness. Thus, in addition to the type of amendment, the origins of marshes influenced vegetation richness. I found that few species were shared between CT and natural marshes, whereas CT and CU had more species in common.

3) Do the chemical and physical conditions pre-determine plant diversity and composition patterns?

The two types of marshes having the most species in common were CU and natural marshes. When comparing ordination space of SAVZ vegetation composition, some proximity was apparent between CU and CT marshes. However, more proximity was seen between CU and natural marshes suggesting that the higher water and sediment salinity of CT marshes relative to CU and natural marshes may be important factors influencing vegetation composition. The EZ patterns of vegetation composition reflected both the chemistry (e.g. water and sediment salinity) and the physical conditions i.e. basin slope, zone width) of each marsh type. Differences in the wet-meadow vegetation composition among marshes was attributable mainly to the differences in physical conditions (i.e. water depth, soil LOI) rather than to water and sediment chemistry.

My results are consistent with the findings of Trites and Bayley (2009), Rooney and Bayley (2011) and Raab and Bayley (2013). I have shown that created marshes (amended and unamended) vegetation composition show dissimilarities with natural marshes of the region. The presence of tailings products (OSPW or OSPM) in created marshes seems, however, to cause the most important differences in vegetation richness and composition. The effect of peat amendment on vegetation composition was indefinite.

Chapter Two set the stage for my other chapters by identifying avenues for future research that would serve both community ecology and reclamation science. Chapter Two proposed insights on the effect of amendment on vegetation community composition but little was known about the influence of amendment at the plant level. Chapter Three has for objective to answer this lack of information.

In Chapter Two I have demonstrated that water and soil quality of the oil sands landscape are challenging components to reclaim. The deliberate addition of PM to improve soil quality in created marshes has been the focus of numerous studies (e.g. Bailey et al. 2007; Sutton-Grier et al. 2009; Ballantine et al. 2012). However, its positive effects on macrophyte growth and physiology remain controversial. In Chapter Three, I attempted to isolate the influence of OSPW, OSPM and PM amendment at the plant-level. Raab and Bayley (2013) proposed that a sedge community dominated by *Carex aquatilis* is a desirable late-succession community for the WMZ of oil sands-created marshes. I experimentally tested the response of *C. aquatilis* to amendments with PM on oil sand sediments (Consolidated-tailings (CT) and Tailings-sand (TS)). In a two factorial design experiment, I also tested the effects of OSPW on *C. aquatilis*. I assessed survival, below- and aboveground biomass, and physiological responses (chlorophyll *a* fluorescence). I had two main questions:

1) To what degree does the amendment of PM to CT and TS substrates enhance *C. aquatilis* functions?

The PM amendments to oil sands sediments significantly increased *C. aquatilis* survival as well as both below and aboveground biomass. Hence, I concluded that the amendment of CT and TS with PM is a viable and beneficial approach to improving *C. aquatilis* survival and biomass production.

2) To what degree does OSPW influence *C. aquatilis* functions?

The use of OSPW significantly reduced *C. aquatilis* belowground biomass and affected its physiological performance. Extrapolating these results to the community-level, it appears that sedge-communities growing in marshes amended with OSPW will produce sub-optimal amount of biomass.

Due to its tolerance and performance, I have verified that *C. aquatilis* was a good candidate for use in reclaiming the WMZ of oil sands-created marshes. Ultimately, amending CT and TS with PM may expedite the reclamation of the marsh to a *C. aquatilis*-community. These early findings have prompted additional on-going support by industrial partners for further tests of PM amendments.

Chapter Two and Three of my thesis explored and compared the patterns of vegetation diversity, composition and functions under different amendments. I provided reclamation benchmarks and references and suggested solutions to improve reclamation practices. However, the benchmarks provided do not take into account the temporal variability of marshes. Marshes are dynamic ecosystems in space and over time and understanding their range of variability is essential to provide a range of reclamation benchmarks (Zhang and Mitsch 2005). Chapter Four represents my attempt to quantify the temporal variability in created and natural marshes of the Fort McMurray region. Developing models of temporal variability in natural marshes for comparison to the development of created marshes is essential for understanding the success of reclamation.

In Chapter Two, Three I have demonstrated that despite vegetation diversity and composition being impaired in created marshes, certain functions, such as plant aboveground biomass, can be reclaimed under specific chemical and physical conditions. However, one main factor influencing marsh diversity, composition and functions remained unexplored. Marsh water-level varies continuously over time and is an overriding factor governing marsh structure and functions (Watt et al. 2007; Raulings et al. 2010). However, little was known about the amplitude of SAVZ-area variation characterising the natural marshes of the Fort McMurray region. Establishing hydrological processes in created marshes that are similar to the ones of natural marshes of the region may enhance the success of reclamation. The objective of Chapter Four was to visually compare zone-area variation over time in natural and created marshes of the Fort McMurray region.Three main questions were investigated:

1) What is the range of SAVZ area variation characterising the natural marshes of the Fort McMurray region?

In natural marshes, for each year assessed, the contribution of the VZ area to the total marsh area was greater than the contribution of the SAVZ. Between 1967 and 2008 the average contribution of the SAVZ area to the total marsh area was of 31% varying from 13% in warmer years to 40% in cooler years. This amplitude of zone area variation is hypothesised to shape in part the structure and functions of natural marshes. Defining SAVZ area variation of natural marshes of this region is important as it describes the ratios of water to vegetation in natural versus created marshes, hence, serving as a target for reclamation.

2)Are hydrological patterns of created marshes similar to those of natural marshes?

Unlike natural marshes the total areas of created marshes were dominated by an extensive SAVZ in all years examined. Between 1998 and 2008, the average contribution of the SAVZ to the total marsh area in created sites was of 74% (range = 71% to 77%). The amplitude of the zone area variation is different from that of natural marshes. Casanova and Brock (2000) have demonstrated that the duration and frequency of flooding can influence macrophytes richness and biomass. Their results suggested that plant richness and biomass was lowest under continuous flooding.

Hence, the water level stability of created marshes relative to natural marshes may translate into differences in community structure and functions. The stable SAVZ and VZ area in created marshes may thus explain in part the observed differences in species richness and composition observed in Chapter Two. This hypothesis remains to be tested.

3)Do weather factors and marsh basin morphometry influence zone area fluctuation in created and natural marshes?

Although I didn’t find any significant relationship between weather variables and created marsh SAVZ variation, I identified weather variables that significantly influence natural marsh SAVZ variation. My results suggest that mean maximum temperatures (annual and two years lag), total snow (annual and two years lag), and mean temperature (annual) were related to SAVZ area variation. However, using the annual mean maximum temperature and annual total snow was the simplest way to predict SAVZ area variation in natural marshes. Surface area to volume ratios were found to be significantly related to SAVZ area variation. To ensure the successful reclamation of created marsh structure, functions and services, SAVZ fluctuation over time should be influenced by similar abiotic factors as those influencing natural marshes of the region. Future created marshes should have surface area ratios of approximately 4:1. Creating suitable basin morphometry will help lead to hydrological patterns (from a plant’s rooted perspective) that are similar to those of natural marshes.

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Apendix-C

Table C-1: List of the marshes (used in Chapter Two and Four), and their location (CT=created-tailings, CU=created-unamended, N=natural, and CP=created-peat).

Wetlands	Location	
1 m CT	56°59'23.38"	111°31'50.99"
Beaver Lodge	56°30'53.40"	111°16'15.84"
Bill’s	56°59'56.37"	111°36'41.89"
Blueberry	56°031'9.02"	111°16'45.38"
Broken-Wing	56°58'08.55"	111°41'12.28"
CNRL Natural	57° 05'04.17"	111°41'32.75"
Crescent	56°54'02.01"	111°24'19.93"
Dam	56°56'36.89"	111°38'49.55"
Demo	57°04'55.84"	111°41'16.11"
Duck	56°59'14.86"	111°32'21.17"
Dyke 4	56° 59'23.38"	111°31'50.99"
Fireweed	57°04'57.35"	111°41'36.04"
Golden	56°59'50.11"	111°33'10.58"
Hammer-Stake	56°45'26.34"	111°35'36.53"
Hidden	56°45'37.25"	111°34'54.04"
High Sulfate	56°59'32.83"	111°31'55.59"
Intersection	57° 05'09.12"	111°41'40.42"
Jan’s pond	56°59'32.83"	111°31'55.59"
Tower Rd 2	56°75'78.81"	111°58'15.87"
Jumping Dog	56°56'19.08"	111°39'41.85"
MFT-North	56°59'31.76"	111°32'05.37"
MFT-South	56°59'31.76"	111°32'05.37"
Mike’s	57° 6'41.29"	111°40'52.65"
Mosquito	56°31'03.99"	111°16'25.71"
Moth Wetland	56°59'50.07"	111°37'27.23"
Muskeg	57°08'10.81"	111°36'05.32"
Tower Rd 1 (N1)	56°44'44.40"	111°29'40.54"
N20	56°44'41.48"	111°29'55.16"
Natural Suncor	56°59'37.82"	111°37'24.74"
North Beaver	57° 05'09.12"	111°41'40.42"
Peat Pond	56°99'38.66"	111°62'36.91"
Petro-Fireweed	57°02'07.26"	111°55'17.51"
Red-Jacket	56°56'27.94"	111°39'44.42"
Saria	56°56'42.30"	111°43'27.77"
Sheeva	57°08'19.81"	111°36'15.67"
South Beaver	57°05'09.12"	111°41'40.42"
Southwest Sands Beaver	56°59'01.03"	111°42'48.76"
S-Pit	57°05'59.21"	111°38'19.96"

Square	56°58'05.18"	111°41'07.78"
Table	56°30'56.18"	111°16'03.86"
Test Pond 1	57°05'04.85"	111°41'39.99"
Test Pond 10	57° 05'04.85"	111°41'39.99"
Test Pond 2	57° 05'04.85"	111°41'39.99"
Test Pond 3	57° 05'04.85"	111°41'39.99"
Test Pond 5	57° 05'04.85"	111°41'39.99"
Test Pond 7	57° 05'04.85"	111°41'39.99"
Test Pond 8	57° 05'04.85"	111°41'39.99"
Test Pond 9	57° 05'04.85"	111°41'39.99"
V-Notch	56°59'17.11"	111°31'57.28"

Table C-2: List of the marshes used in Chapter Two and Four, their type, water chemistry, and physical characteristics (CT=created-tailings, CU=created-unamended, N=natural, and CP=created-peat).

Marshes	Types	pH	DO	DO	Salinity	ORP	Conductivity	Water Depth	Basin Slope	Zone Width	Area
Names			(mg/L)	(%)	ppt	(mV)	(µS)	(cm)	(o)	(m)	(m2)
Jan’s	CT	7.9	3.1	0.4	1.1	91.0	2017.0	23.7	5.2	2.1	3700
MFT-North	CT	8.8	5.4	0.6	1.1	47.0	2030.0	38.0	17.3	1.3	10700
MFT-South	CT	8.6	5.9	0.7	0.9	73.0	1858.0	47.6	17.3	1.2	10700
Mike’s	CT	7.4	9.4	1.3	2.6	0.0	4991.0	32.4	23.0	1.5	50000
S-Pit	CT	9.7	6.8	0.8	0.4	3.0	868.0	10.6	2.5	3.7	36400
Test Pond 10	CT	7.7	9.4	1.2	1.6	-23.0	3081.0	16.3	30.4	0.0	53060
Test Pond 2	CT	7.2	7.9	0.9	0.4	61.5	769.0	24.2	10.0	2.1	15850
Test Pond 3	CT	7.2	9.3	1.1	0.4	19.0	717.0	24.2	23.4	2.1	8780
Test Pond 5	CT	9.0	6.2	0.7	1.4	74.0	2710.0	39.1	12.3	1.0	8780
Test Pond 7	CT	7.4	9.8	1.3	0.8	76.0	1574.0	26.9	20.1	1.4	8780
Test Pond 8	CT	7.7	9.4	1.2	1.6	-23.0	3081.0	15.2	10.0	3.4	53060
Test Pond 9	CT	7.8	6.8	0.9	1.0	45.0	2214.0	24.3	10.0	1.8	53060
1 m CT	CT	8.0	6.5	0.6	0.9	207.0	1700.0	14.5	4.0	3.8	4000
Demo	CT	9.8	8.0	0.9	0.9	69.0	1741.0	7.9	7.3	3.1	24600
Dyke 4	CT	7.9	6.5	0.6	1.0	143.0	2300.0	25.0	8.1	2.6	15850
V-Notch	CU	7.9	6.0	0.6	0.2	173.0	740.0	25.6	4.3	2.4	1204
Bill’s	CU	6.6	16.0	1.5	0.4	-169.0	748.0	30.1	15.0	1.7	5800
Blueberry	CU	7.2	2.4	0.3	0.2	129.0	342.5	42.8	9.4	2.0	4400
Crescent	CU	9.2	4.1	0.5	0.1	147.0	215.9	23.7	2.6	6.2	5100
Duck	CU	8.1	5.7	0.7	0.3	118.0	892.0	15.6	5.4	1.8	6700
Fireweed	CU	7.6	2.3	0.3	0.1	165.0	243.2	5.2	9.2	3.3	4392
Intersection	CU	7.4	3.7	0.4	0.3	107.0	560.0	46.7	8.9	2.2	880
Muskeg	CU	7.4	1.0	0.1	0.1	221.0	197.4	64.9	6.4	4.1	32200
North Beaver	CU	7.9	6.1	0.7	0.2	353.0	331.4	46.8	14.8	1.6	11424
Petro-Fireweed	CU	8.3	6.1	0.8	0.0	126.0	70.1	41.1	6.7	3.2	4392
Red-Jacket	CU	8.9	3.2	0.4	0.1	130.0	310.0	21.5	6.6	1.5	9300
Saria	CU	8.6	6.3	0.6	0.0	107.0	0.0	22.2	12.2	2.9	4400

South Beaver	CU	7.6	2.3	0.3	0.1	165.0	243.2	14.5	10.6	2.8	11424
Square	CU	8.9	2.5	0.3	0.1	111.0	294.7	23.1	9.1	1.8	2024
Table	CU	8.0	2.9	0.3	0.1	86.0	269.0	42.6	18.3	1.2	1500
Test Pond 1	CU	8.3	4.9	0.5	0.0	157.0	2.2	64.8	24.4	1.4	878
Beaver Lodge	N	7.3	4.4	0.4	0.1	135.0	273.8	39.4	0.6	8.7	50800
Broken-Wing	N	7.1	2.5	0.3	0.2	148.0	457.5	33.4	4.5	6.3	9300
CNRL Natural	N	8.2	7.3	0.8	0.0	120.0	1.3	42.9	3.4	4.5	21100
Dam	N	7.4	2.6	0.3	0.0	89.0	3.7	14.0	2.6	12.4	21100
Hammer-Stake	N	7.7	4.2	0.6	0.4	-147.0	888.0	15.0	5.0	7.4	78000
Hidden	N	7.5	4.0	0.4	0.0	126.0	95.5	31.7	4.5	8.9	53060
Tower Rd 2	N	7.9	2.4	0.3	0.2	205.0	335.8	37.1	1.0	7.8	127000
Jumping Dog	N	7.6	6.1	0.8	0.0	143.0	4.4	19.8	4.0	4.6	26050
Mosquito	N	7.1	3.0	0.3	0.2	133.0	420.5	30.6	1.4	8.3	26000
Tower Rd 1 (N1)	N	7.5	5.5	0.6	0.1	-21.0	166.3	31.7	2.5	9.7	21100
N20	N	9.1	6.1	0.1	0.2	130.0	225.5	23.8	4.5	9.3	33900
Sheeva	N	7.0	2.5	0.3	0.0	182.0	2.5	61.0	5.5	20.3	400000
Southwest Sands Beaver	N	8.3	7.0	0.8	0.0	158.0	3.5	6.3	5.7	6.0	53060
Moth Wetland	N	5.5	4.6	0.6	0.1	21.0	148.0	45.3	2.5	9.3	11233.0
Golden	CP	7.1	10.6	1.3	0.6	-25.0	1768.0	15.0	7.8	2.1	600
High Sulfate	CP	6.7	41.0	3.4	1.6	-120.0	2982.0	23.8	7.5	3.6	4000
Natural Suncor	CP	8.6	4.5	0.6	0.0	145.0	4.9	5.6	2.2	2.6	1160
Peat Pond	CP	7.0	9.7	1.1	0.7	-22.0	1621.0	39.0	16.2	1.5	750

Table C-3: List of the marshes used in Chapter Two and Four, and the chemistry of their sediment averaged at the marsh level. (LOI=lost on ignition, TP=total phosphorus, TN=total nitrogen, EC=electric conductivity).

Marshes	Moisture %	LOI (mg/L)	TP (mg/L)	TN (mg/L)	EC (mg/L)
Beaver Lodge	1329.9	37.8	1.7	27.8	78.0
Blueberry	73.8	5.1	0.6	3.0	224.6
CNRL Natural	211.2	11.0	0.5	8.4	117.9
Fireweed Syncrude	49.0	5.1	0.7	3.3	233.6
Golden	42.7	2.9	0.7	1.8	565.8
Hammer Stake Beaver	52.1	5.4	0.3	4.1	139.6
High Sulfate	73.9	10.6	0.5	7.1	1089.2
Intersection	44.8	5.2	0.6	2.9	488.6
Jan's Pond	78.9	8.7	1.0	4.1	1020.6
Jumping Dog	85.5	9.7	0.6	8.5	121.9
MFT-South	97.7	10.5	0.7	6.9	508.4
Mike’s	41.1	4.8	0.6	2.0	573.0
Mosquito	118.0	13.4	1.0	9.5	142.3
Tower Rd 1 (N1)	73.2	9.3	0.8	7.8	153.3
Natural Suncor	61.1	8.7	0.7	7.5	416.2
North Beaver	46.0	4.6	0.8	2.2	288.6
Peat Pond	50.7	4.9	0.8	2.9	518.4
Petro-Fireweed	59.2	5.9	0.8	4.1	82.8
Plover	28.0	3.1	0.6	1.9	38.0
Saria	72.2	6.3	0.9	4.6	206.5
South Beaver	44.8	4.6	0.9	3.0	279.8
Square Wetland	50.6	7.2	0.7	5.1	255.8
Southwest Sands Beaver	51.4	6.1	0.6	6.5	294.4
Table	64.4	5.1	0.5	2.4	94.2
Test Pond 1	51.5	4.5	0.6	5.2	263.5
Test Pond 10	34.9	5.0	0.6	2.0	292.2
Test Pond 3	31.2	4.1	0.7	2.9	451.5
Test Pond 7	126.8	6.7	0.7	3.0	463.9