University of Alberta

Vegetation Community Development of Reclaimed Oil Sands Wetlands

by

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A thesis submitted to the Faculty of Graduate Studies and Research in partial fulfillment of the requirements for the degree of
Master of Science

in

Water and Land Resources

Department of Renewable Resources

Edmonton, Alberta

Spring 2004

Your file Votre référence
ISBN: 0-612-96461-2
Our file Notre référence
ISBN: 0-612-96461-2

Abstract

I examined vegetation community development of reclaimed oil sands wetlands. Soil transfers from reference wetlands accelerated plant colonisation rates in a consolidated/ composite tailings (CT) wetland. Hydrologic regimes differed between reference and CT wetlands making it difficult to explain observed differences in species composition. CT was a hospitable environment for plants, indicated by rapid colonization of isolated CT plots in reference wetlands. Wetland soil seed banks were more similar in species-relative abundances than composition. CT subsoils reduced emergence from seed banks (species composition and relative abundance). Salinity (surface waters, subsoils), wetland isolation and northern climates may slow or alter species replacement sequences for Reclaimed wetland plant communities. Seed bank analyses overestimated species richness, compared to field observations of wetland vegetation communities, except for Newly Constructed wetlands. Adding a CT subsoil treatment increased the accuracy of the seed bank analyses to predict initial vegetation establishment in a Newly Constructed CT wetland.

Acknowledgements

This is the toughest section to write in this entire thesis. How can I put into words the appreciation that I feel for all of you who have helped me along the way? So first off, I want to say a big thanks to everyone! If I've failed to specifically mention your name, it's likely because I've run out of room.

I'd like to extend huge thanks (with a southern accent) to my supervisor, Lee Foote. I took Lee's grad level course, and became fascinated with Wetlands (yes, bugs and ooze too)! And after meeting his graduate students (Jon, Christine, Barb, Jordan and soon to be Robb), and getting to know this southerner a bit better, I decided to become a "Footian" and strapped on my chestwaders with pride. Lee supported and guided me throughout and inspired me to keep looking for the positive. When I wanted to switch gears and work on disturbance impacts on riverine wetlands, Lee kept me focused on the task at hand by saying, "with your CT wetland, you have no where to go but up, [success story] whereas in natural systems you'll likely be reporting negative changes [doom and gloom]." His less than politically correct humor, kept me smiling and laughing and guessing about what to expect next! On a personal level, I am very happy to have spent time with the Foote and Krogman family. Lee and Naomi have shown me how it is possible for two academics to have a life that includes two beautiful children, and varied interests beyond work.

I'd like to thank Suncor, Syncrude, Golder Associates, McMurray Resources and Technology, NSERC, Keyano College and the U of A for direct funding and in-kind support for this project. In particular, I'd like to thank Doug Kennedy, Bruce Anderson, Wayne Tedder, Leo Paquin, Amy Mullaly and various field techs from Suncor; Terry van Meer, Clara Qualizza, Martin Fung, Neil Rutley, Michael MacKinnon, Betty Fung, Allan Yeung and Joanne Hogg and various field staff from Syncrude; Steve Tuttle, Laura Mucklow, John Gulley, Farida Bishay, Patricia Marlowe, Mark Sherrington, Anthea Hastie and Veronica Chisholm and others from Golder Associates; Laura Smithies, Herman Techteimester, and others from MRRT, Marie, Terry and Sylvie Forrest from Keyano College, Suzanne Bayley (external committee member), and David Chanasyk (committee member) from the U of A . A big warm thanks to my field techs Sarah Hall and Scott Boorman from the U of A; they were the real troopers of this project ... you guys were invaluable... thanks so much! I'd like to acknowledge the expertise of Darcy Henderson, Jon Hornung, and Shane Patterson who kindly reviewed early drafts of chapters. I'd also like to rethank Michael MacKinnon, who not only served as a committee member, but mentored me in the lab, and provided all of the water and soil analyses for this project. I'd also like to acknowledge Anne Naeth who initially inspired me to pursue graduate studies in the field of reclamation- her commitment and zest for the field is amazing!

To current and graduated Footians (Jon, Christine, Barb, Jordan, Robb, Nadele, Kathryn, Steph), I really appreciated for your comments, jokes and above all, the friendships that I have developed with each of you. The older cohort, Jon, Christine, Barb and Jordan served as mentors to me and welcomed me to the group with open arms. Thanks guys!

To the Fort McMurray ladies and gents (Tara, Diane, Selma, Joanne, Carla, Christel, Amy, Sarah, Scott and Pat) thank-you for making my non-field time memorable. Tara- here's to running, revving and our rant sessions! Sarah—

remember: chocolate and popcorn are field necessities and as far as yoga goes, one day you'll see the light! Scott- cheers to the mullets and Chinese food. Lady Di- you're done up there for good! Pat, thank-you for continuously amazing and inspiring me with your plant, and fossil ID skills... you are an invaluable resource and a very good friend indeed ☺ .

I'd like to thank my family for their continued support and encouragement. Mom and Dad, thank-you for believing me and helping me achieve my goals. I'd like to thank Kyle for accompanying me to the mountains snowboarding and continuing to educate me in the realms of punk music, skateboarding and style. Ginny—thanks for our weekly phone conversations and blessing me with a beautiful and very headstrong niece.

My grad school experience would not have been the same without the friendships that I have maintained and created over this time. My non-gradschool friends (Kelly, Jen, Nichole, Shannon, Heather, Dennis and Brett): thanks for being there and reminding me that there is life beyond school. Jen and Nichole: here's to snowboarding in the mountains and crazy fun times! Kelly, thanks to you and Troy for reminding me that marriage isn't so bad (well maybe one day). My past roomies, Shannon, Heather and Dennis, you are like siblings to me. Thanks for keeping in touch. I'd like to thank my Death Race Team BEAR BAIT(Angela, Steve, Mario and Elise). we kicked ass! To my Friday beer mates and companions on various social outings (you know who you are)! My cohort (Ryan, Lane, Kathryn, Nadele, Robb, Zhili and Diane)—thanks for being there alongside me, during tough and fun times. A big hug to each of you. Kathryn, thanks for being my roommate and engaging in numerous chats, and laughs and reminding me to think about making ethical, social and environmentally responsible choices. Nadele, Sue, Elise, Christine, Shannon and Meg—here's to our mountain adventures, North Country Fair and afternoon tea, wine, movie sessions and snow golf. Ben—thanks for the sanity breaks from my thesis writing and inspiring me to continue to work at making a life, rather than making a living. Alastair and Joanne, thanks for the office chats, and Andrea thanks for the awesome dinners (chef extroardinare) ☺

And lastly, a special big thanks to two individuals who made my graduate experience a memorable one: Elise and Dave. "Princess Parker"—you've been such a wonderful friend in so many ways, from lending an ear and a shoulder, to our crazy treks up Healy Pass, the North Boundary Trail and the Death Race. Here's to many more such adventures! And let us not forget our obsession with Corb Lund.. awe Corb ☺ "Sk8er Dave"—you helped me believe in myself and inspired me to pursue my dreams. Your friendship, love and companionship meant a lot to me. You put up with my less than stellar birding skills (though I did catch you on a couple), folk/ hippie/ girly music and constant teasing. Thank-you.

And it wouldn't be a Natalie Original if I didn't tack on this line at the end: Where there is hope, there is life and dance like you do when nobody's watching. Smiles to all of you!!

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List of Terms and Definitions

The following is an alphabetical list of definitions for terms used in this document (as defined in the Glossary of Reclamation and Remediation Terms used in Alberta (Powter 2000)):

Bitumen: "The heavy viscous hydrocarbon associated with the Athabasca Oil Sands deposit s. It contains some mineral and sulphur contamination."

Consolidated/ composite tailings: "Composite (Syncrude) or Consolidated (Suncor) tailings are formed by injecting mature fine tailings from the tailings pond into the regular (whole) tailings sand stream, with a floculant such as gypsum. This mixture is sent to the tailings ponds to form a non-segregating soil mixture which will result in a trafficable surface in the reclaimed landscape."

Equivalent Land Capability: "The ability of the land to support various land-uses after reclamation is similar to the ability that existed prior to any activity being conducted on the land, but the ability to support individual land uses will not necessarily be equal after reclamation (regulatory definition)."

Macrophyte: "A member of the macroscopic plant life (larger than algae) especially of a body of water."

Marsh: "A class in the Canadian wetland classification system; a marsh is a mineral or a peat-filled wetland which is periodically inundated by standing or slowly moving water." Vegetation is typically dominated by a variety of emergent plants including: cattails, rushes, sedges, reeds and grasses, along with some floating and submersed aquatic plants in deeper areas.

Mature Fine Tailings: "A term used in the oil sands industry to refer to the material accumulating at the bottom of oil sands tailings ponds. It is a matrix of dispersed clays, fine materials, residual hydrocarbons, and various contaminants. Note that whole tailings includes tailings sand which settles rapidly and is used to form tailings dykes."

Muskeg: "A North American term frequently employed for peatland. The word is of Algonquin Indian origin and is applied in ordinary speech to natural and undisturbed areas covered more or less with *Sphagnum* mosses, tussocky sedges and an open growth of scrubby trees."

Natural Recovery: "Natural re-establishment of plants on disturbed lands. Relies on revegetation from the topsoil (seedbank) or invasion from adjacent lands."

Reclamation: "The process of reconverting disturbed land to its former or other productive uses."

Species Richness: The number of species (wetland plant species) found in a given area.

The following list of definitions is also used in this document (as defined by SMLO (2001)):

Gypsum: "A mineral composed of hydrated calcium sulphate. Natural gypsum is found in bedrock containing evaporate deposits. It is also a byproduct of industrial processes (flue gas desulphurisation)."

Oil sand: "A sand deposit containing bitumen in the pore space. Rich oil sand may contain up to 18 percent bitumen (weight basis)."

Overburden: "Overlying material that must be removed before ore can be mined."

Powter, C, B. (complier) 2000. Glossary of Reclamation and Remediation Terms Used in Alberta - 6th Edition. Alberta Environment, Environmental Sciences Division, Edmonton, Alberta. Report No. ESD/LM/00-3. 63 pp ISBN 0-7785-3342-3.

SCML. 2001. 2001 Closure and Reclamation Plan: Syncrude Mildred Lake Operation. Syncrude Canada Ltd. Edmonton, Alberta. 361pp.

Chapter 1: Research Rationale and Thesis Overview

1.1 Background to Oil Sands Wetland Reclamation Issues

Creating functional wetlands in the oil sands mining areas of Northeastern Alberta brings tremendous challenges. In the Northeastern Boreal Region of Alberta, oil sands mining continues to increase, yet basic understanding of the structure and function of reclaimed wetland systems, compared with pre-disturbed wetland systems remains unknown. As part of their operating approval agreements, oil sands leaseholders are required to return mined lands to equivalent or better land capability compared with pre-disturbance, and these reclaimed lands must be continuous with surrounding leases (Alberta Environment 1999; Oil Sands Wetlands Working Group 2000).

Wetlands comprise a large proportion of the pre-disturbance mining landscape in the Fort McMurray region and thus will remain important landscape features post-disturbance. "Reclamation will be guided and directed by existing policy and, legislation and planning initiatives including: the Report and Recommendations of the Oil Sands Mining End Land Use Committee, the Fort McMurray-Athabasca Integrated Resource Plan, the Recommended Wetland Policy for Alberta, the Environmental Protection and Enhancement Act (EPEA), the Water Act, and the Oil Sands Regional Development Strategy" (Oil Sands Wetland Working Group 2000). Post-mining land uses for reclaimed wetlands include: aboriginal traditional and recreational land uses, wildlife habitat, and surface water treatment (Golder Associates 2000).

1.2 Post-mining wetland types

Post-mining wetlands have been classified into five types: altered wetlands, opportunistic wetlands, constructed wetlands, vegetated watercourses, and littoral zones (Oil Sands Wetlands Working Group 2000). Altered wetlands occur in areas that were not mined, but are impacted by post-mining processes (e.g., dewatering, drainage, flooding). Opportunistic wetlands are unplanned landscape features, occurring in depressions or in dynamic drainage areas. Constructed wetlands are landform features that are planned to function as flood control, water treatment or habitat for wildlife populations. Vegetated watercourses will form in ditches and drainage features for transfer of water to and from wetlands, both on and offsite. Littoral zones will be present along the slopes of reclaimed end pit lakes and water bodies.

1.3 Consolidated/ composite tailings wetlands

The establishment of vegetation on consolidated/ composite tailings (CT) or process derived substrates is of particular interest to oil sands operations managers. The CT deposits are a gypsum-enriched (inorganic coagulant) saturated slurry of mature fine tails produced as a by-product of processed oil sands (MacKinnon et al 2001; Matthews et al. 2002). Both produced waters and CT are slightly to moderately saline, high in total dissolved solids, and contain elevated ionic concentrations of sulphates, chlorides, sodium, as well as naphthenic acids and trace elements. Backfilled mine cells hold large volumes of CT, and it is expected that wetlands will be constructed on a large proportion of CT deposits. These constructed wetlands are predicted to develop into shallow, highly productive water habitats, connected with the surface drainage for that area (Golder Associates 2000). Previous studies have indicated that many of the CT constituents can negatively affect terrestrial plant performance (plant recruitment, growth, development and reproduction) (Apostal 2002; Renault et al. 1998; 1999), but it is unknown how wetland plants will respond to the physical and chemical CT environment.

1.4 Natural recovery and CT wetlands

There are two models or approaches that are commonly applied for wetland restoration and creation projects: (1) self- organization/ self design and (2) ecosystem restoration/ designer wetland. The former relies on the "self- organising ability of ecosystems in which natural processes contribute to species introduction and selection" (Mitsch et al. 1998). An ecosystem will organize by "selecting for the assemblage of plants, microbes and animals that is best adapted for the existing conditions" (Mitsch et al. 1998). Alternatively, the ecosystem restoration/ designer wetland approach refers to the "specific introduction of specific organisms (often plants) whose survival becomes the measure of success of the restoration" (Mitsch et al. 1998). Current knowledge of wetland plant tolerances to the specialised chemical and physical conditions present in CT wetland is extremely limited; it is likely that the designer approach would be less effective and or sustainable than the self-organisation/ self design approach which relies on natural processes.

Under the self organisation/ self design approach, typical revegetation options for northern oils sands wetland reclamation projects include: (1) natural recovery using slurried salvaged soil (muskeg), and (2) seeding or transplanting of wetland plant species endemic to the region. Seeding and transplanting are currently not considered feasible methods to vegetate large-scale reclamation areas such as post-mining CT deposits. Transplanting would require extraordinary manual labour, and would not be feasible for all regions, except possibly wetland fringes, where the CT may safely support equipment or humans. Provincial legislation requires oil sands operators to use seed native to the Fort McMurray region, and presently, native seeds for wetland plant species are not available commercially. Natural recovery, relying on the seed bank and bud bank in the muskeg is by far the most cost-efficient method to vegetate large areas, though it is not known how effective this method will be in producing diverse, self-sustaining plant communities, within a long-term planning horizon of 10-15 years.

Establishment of suitable hydrology and substrate (soil, detritus and plants) is expected to be an essential basis for functional food webs and for ameliorating the deficiencies of recently processed CT subsoils. Soil characteristics, combined with hydrologic regimes, control the levels of available nutrients, which in turn will affect the floristic composition of wetland vegetation and soil microflora and fauna (Chambers et al. 1994). Muskeg capping, like wetland sediment transfers used in many wetland creation/ restoration projects, will provide sources of propagules, seeds, nutrients and serve as a substrate providing anchorage to emerging and colonizing wetland plant species (van der Valk and Pederson 1989). Soil transfers are expected to accelerate wetland plant community development, resulting in higher species richness and cover, than that of natural invasion processes (seed dispersal through wind, water and wildlife populations) (Mueller and van der Valk 2002).

The effectiveness of using natural recovery through soil transfers in wetland creation projects has been well documented throughout the United States and in parts of Canada. Vivian-smith and Handel (1996) examined the use of imported wetland soil in a freshwater reclamation project and found enhanced species richness and plant density, further supporting the use of imported seed and bud banks to enhance revegetation success. McKnight (1992) showed species richness to increase in created wetlands receiving salvaged soils, compared to the wetlands where the material was salvaged from. Increased species richness was attributed to existing seed banks at created wetland sites, and/ or physical disturbance of the salvaged soil, exposing seeds and facilitating germination (Smith and Kadlec 1983). In new CT wetlands receiving imported soils, it is likely that species richness would be less than, or equal to, that of wetlands where the soil was salvaged. This is because contributions of wind, water and waterfowl populations to seed bank development in CT wetlands would be minimal and likely limited to early colonizing weedy species at the outset.

To date, in the Northeastern Boreal Region of Alberta, informal pilot studies have examined natural recovery of CT wetlands using muskeg capping in conjunction with various techniques to establish plant communities, such as transplanting individual plants and vegetation islands (Golder Associates 2000; 2002), but none have examined natural recovery in isolation. Wetland plant species assemblages (species composition and relative abundance) occurring on muskeg capped CT wetlands have not been directly compared to species assemblages of the wetlands where the soil was salvaged. Furthermore, isolation and placement of CT materials in natural wetland settings are necessary to assist in understanding effects of CT and CT water characteristics on wetland plant species assemblages and vegetation community development. This information is needed to help determine the degree of inputs, management and monitoring necessary to produce a productive, diverse, and self-sustaining ecosystem that is continuous with the surrounding landscape.

1.5 Spatial patterns of wetland plants

The initial alteration of the boreal terrain by surficial oil sands mining is profound and of a magnitude (locally) comparable to volcanism or glaciation. Prior to the commencement of mining, vegetation is removed, and the upper soil horizons are stripped and stored or directly placed onto a new reclamation site (direct transfer). The topsoil (locally referred to as muskeg) is a mix of organic rich peat, muskeg and mineral overburden (Golder Associates 2000). The extreme alteration of the landscape provides an opportunity to examine near primary succession colonization rates of wetland plants.

Species-specific reproduction methods are one of the primary biological controls of spatial patterning of wetland plants (whether a species reproduces asexually (vegetative reproduction) or sexually (seed production) or both). Theoretically, sexually reproducing species should have a competitive advantage over asexual reproducers in their ability to disperse and colonize larger areas. For example, *Ruppia maritima,* a submersed species has been shown to produce large amounts of seed and it is postulated that its dispersal might explain how it is able to rapidly colonize some areas (Silberhorn et al. 1996). However, sexual reproducers do not always have the competitive advantage, because other controls in successful colonization include herbivory, seed persistence and viability in the seed banks (Mitchell et al. 1998). Hutchings and Russell (1989) found that the seed bank in an emergent salt marsh had limited persistence and mostly reflected seed production from the current year. In a calcareous fen, seed production for *Shoenus ferrugineus* and *Molina caerulea* was higher than residual seeds in the seed bank, indicating low survivability and high turnover rates of seeds in upper soil layers (Schopp-Guth et al.1994). Some emergent species, such as *Carex spp.*, produce seeds that undergo dormancy cycles in the seed bank (Schütz 1998).

Seeds of early seral species dominate seed banks in recently colonised plant communities (Bekker et al. 2000). In a near-primary successional environment such as a CT wetland, early seral species may have a competitive advantage over mid-late seral stage species. Early seral species often have broad niches, and are better able to adapt to a range of environmental conditions. Early seral species have preserved their genomes over time through adaptations that allow use of a larger portion of the resource spectrum in unpredictable and dynamic environments (Pickett and Bazazz 1978). This has been shown in prairie research where early successional annuals occupied larger proportions of a moisture gradient, overlapping more than what would be expected if the community was randomly organised. The germination and emergence of playa species have evolved to occur throughout the period of suitable environmental conditions (Haukos and Smith 2001). In contrast, Parrish and Bazazz (1982) demonstrated that late seral species have reduced niches that allow for reduced competition and enhanced survival in late successional communities.

In conjunction with species specific life history characteristics, a variety of physical and chemical factors control spatial patterns of wetland plants. Species composition and abundance of wetland plant species vary according to hydroperiod duration (Shay et al. 1999). Short-term dewatering (drawdown) followed by extended flooding events shifts vegetation structure (Baldwin et al. 2001; Schneider 1994; Gerritsen and Greening 1989). This information has been applied successfully by managers in establishing emergent vegetation in created wetlands. Soil transfers from donor wetland sites were subjected to drawdown regimes in early spring (McKnight 1992), because most emergent seeds germinate under dry or moist soil conditions, while submersed species germinate under flooded conditions (van der Valk and Pederson 1989).

The relationship between plant communities and hydrology is complex. In riverine ecosystems, germination rates of *Rumex* are controlled by flooding dynamics (Vosenek and Blom 1992); *Rumex acetosa* germinates in early autumn and is rarely found in flooded zones, while wetland species *Rumex crispa* and *palustris* germinate in late spring and often have multiple post-flood germination cohorts. Mud-flat annuals were more abundant than emergent species in soils taken from wetlands with short hydroperiods (Poiani and Johnson 1988). Welling et al. (1988) found that high soil moisture inhibited the germination of *Scholochloa festucacea* and *Carex atheroides*, while low moisture levels inhibited germination of *Typha spp*. and *Scirpus lacustris spp. validus*. van der Valk et al. (1999) found that recruitment of *Carex spp.* was highest under conditions of maximum soil moisture, at sites with highest soil organic matter, and where *Carex* seeds were the youngest/ freshest.

Sedimentation can also affect recruitment from the seed bank (Dittmar and Neely 1999; Giroux and Bédard 1995; Jurik et al. 1989; Rybicki and Carter 1986; Wang et al. 1994), and suppress and stimulate growth (Foote 1991; Spencer 1987; van der Valk et al. 1983; Wang et al. 1994). Dittmar and Neely (1999) found that large sediment loads lowered total seedling density, compared to low sediment loads, but responses by individual plant species were inconsistent.

High soil electrical conductivities can inhibit seed germination at low soil moisture levels (Galinato and van der Valk 1986; Lieffers and Shay 1981). Hackney et al. (1996) identified four distinct vegetation zones in a mesohaline tidal marsh that indicated distinct chemical and physical environments. Sherman et al. (1998) suggested that a "plant-soil microbial feedback" partially determined spatial patterning of mangrove forest vegetation communities and soil variables. The distribution of *Laguncularia racemosa* across a tidal gradient was related to concentrations of total phosphorous, dissolved organic carbon, sulphates in the soil porewater and soil pyrite concentrations.

Air temperature and stratification affect germination rates. Some species such as *Scirpus acutus* responded favourably to high air temperatures and long stratification periods (25 °C and 12 weeks, respectively) (Thullen and Eberts 1995). Air temperature strongly regulated germination of four species of cespitose *Carex* (*Carex arenaria, acutiformis, extensa, flacca*) (Schütz 1998); diurnal and mean seasonal air temperature fluctuations, as well as lowered air temperatures due to shading, influenced germination rates. Light conditions and high temperature regimes favored the germination of mudflat annuals (*Aster laurentianus, Atriplex patula* and *Chenopodium rubrum*) and perennial emergents (*Hordeum jubatum, Scholochloa festucacea, Phragmites australis* and *Typha glauca*) (Galinato and van der Valk 1986).

The Continuum Concept integrates the aforementioned biologic, physical and chemical factors that regulate spatial patterns of wetland plants (Mitsch and Gosselink 1986). The Continuum Concept describes spatial patterns or zonation of wetland plants as the result of "an environmental gradient of physical factors interacting with the biotic potential of a specific site" (Parrish and Bazzaz 1982). According to this theory, "the physical and chemical characteristics of each vegetative zone should represent a continuum, with each assemblage of plant species (zone) representing the portion of continuum where it (they) outcompetes other species or assemblages" (Parrish and Bazzaz 1982). Applying this theory to natural and created wetlands, we would expect patches or rings of wetland plant species based on the competitive tolerance to water depth, salinity, anoxia, etc.

Ultimately, spatial patterning of wetland plants in CT wetlands will depend on the adaptations of plant species to the physical and chemical characteristics of CT wetland deposits. Depending on the size of CT wetland deposits, it may or may not be possible to manage water levels to represent natural hydroperiods (i.e., manipulate weirs during precipitation events). Salt concentrations in soil and/ or surface waters could prohibit managers from mimicking complete drawdown regimes, because salt crusts could adversely affect recruitment, growth and survival of non salt tolerant vegetation. There is limited knowledge as to which species occurring in seed banks and bud banks will germinate under flooded conditions, versus those species that will germinate under only drawdown regimes and subsequently will need to be grown and transplanted on site. It is not understood whether the presence of CT as a secondary soil layer will influence wetland plant species recruitment from the seed bank, possibly resulting in reclaimed wetland plant communities that are dissimilar to those of natural or pre-mining wetlands. This information is needed to help resolve the degree of inputs, management and monitoring necessary to produce a productive, diverse, and self-sustaining ecosystem that is continuous with the surrounding landscape.

1.6 Tolerance of wetland plants to saline and anoxic conditions

Survival of wetland plants in anoxic conditions has been described as an endurance phenomenon (Crawford 1983), with atmospheric oxygen diffusing into the plant shoot and then the plant transporting it to the roots. The roots protect the plant against accumulation of phytotoxic ions such ferrous iron, manganese and sulphides by oxidizing the surrounding rhizosphere, depositing excess ions on the root tissue and eventually sloughing the material. Thibodeau and Nickerson (1986) showed that *Avicennia spp.* could oxidize its rhizosphere, thereby controlling concentrations of soluble sulphides in the soil porewater. Some wetland plant species have adapted to flooded conditions by changing or thickening their root forms. Other wetland plants have hollow stems that allow storage of air during saturated conditions. As carbon dioxide levels in the stem increase, aerial parts of the plant become enriched and subsequently the photosynthetic capacity of the plant is increased, resulting in increased oxygen transport to the roots. The simplified structure of wetland plants such as hollow stems, aerenchyma, adventitious roots and non-woody tissues also enhances the transport of oxygen to the roots and supports the diffusion of anaerobic respiration by-products such as acetaldehyde, ethanol and carbon dioxide.

Plants respond similarly to salinity levels, by concentrating, barring or diluting ions through selective ion absorption (Howes Keiffer and Ungar 1997; Sanderson et al. 1997). Elevated concentrations of salts in porewater results in increased water potentials; the plant must adapt by expending more energy to allow water absorption into its tissues (Sanderson et al. 1997). *Phragmites australis* initial establishment and invasion was found to be controlled primarily by anoxia and rhizome exposure (Bart and Hartman 2002). Clonal plants did not show reduced emergence rates under conditions of elevated salinity or sulphide concentration; however, overall plant performance was reduced, because survival, growth and biomass were reduced. In one study, *Spartina patens* showed limited gas exchange functioning and nitrogen uptake when salinity was 6 ppt and redox potential (Eh) was lowered to -115 mV (Bandyopadhyay et al. 1993). Although the occupation of wetland plants in anoxic and or saline zones is seen as an inefficient use of their energy resources, these specialised species are able to occupy a niche that is free from competition with terrestrial species.

Previous research in the Northeastern Boreal Region has focused on responses of high salinity oil sands tailings water to trees and shrubs (Apostal et al. 2002; Renault et al. 1999; Renault et al. 1998). It is not known what the response of wetland plants will be under these conditions. Understanding tolerances of wetland plants to anoxic and saline conditions of CT wetlands will help in predicting vegetation community development over time.

1.7 Assessing wetland vegetation community development

Reclamation advisory groups, industrial operators and stakeholders are interested in developing standard methods to evaluate wetland reclamation success. Of primary interest is (1) how similar are reclaimed and natural wetlands on the landscape over time and (2) are there indicator species or groups of species that can be used as evaluators of reclamation success? Various metrics have been developed over time to evaluate structure and function of reclaimed and created wetlands. These metrics have included bird use, (Brawley et al. 1998), soil development (Bishel-Machung et al. 1996; Nair et al. 2001; Stolt et al. 2000), benthic invertebrate communities (Leonhardt 2003); productivity, (McKenna 2003) and vegetation community development (Edwards and Proffitt 2003; Galatowitsch and van der Valk 1996; Leck 2003; Seabloom and van der Valk 2003). New research by Leonhardt (2003) has indicated that reclaimed wetlands, some of which are used in this study (oil sands process materials wetlands), differ in benthic invertebrate species composition, compared to natural wetlands of similar age. Mechanisms responsible for observed differences (water quality, water and sediment chemistry, macrofauna, flora, pollutants) are not well understood yet.

Macrophyte plant community development is commonly used as a metric to compare equivalency or functionality between created or restored wetlands and natural or pre-existing wetlands (Edwards and Proffitt 2003; Galatowitsch and van der Valk 1996; Seabloom and van der Valk 2003). Seed bank analyses, using the emergence method, have been used to assess vegetation history and predict vegetation community development over time (Haukos and Smith 1993; 2001; Rossell and Wells 1999; Poiani and Johnson 1988; van der Valk and Davis 1976; van der Valk et al. 1992; Welling et al. 1988; Wetzel et al. 2001). Soil is collected from wetlands and then placed in near optimal growing conditions (greenhouse environment), under different treatments; the seedlings that emerge are identified and counted. Treatments commonly tested, include flooding/ non-flooding and substrates.

The emergence method is not without its biases. Greenhouse conditions often do not accurately simulate light and temperature levels, water and nutrient regimes occurring in wetland settings (van der Valk et al.1992). Secondly, both physical disturbance of the soil during sample collection and sample storage conditions can result in breaking of dormancy cycles for some species, resulting in an under or overestimation of the potential of the seed bank to represent the floristic composition of a site (van der Valk et al. 1992).

The alternative to the emergence method involves manual separation of the seeds from the soil and then either identifying or germinating the seeds. Manual seed separation results in a potentially biased estimate of relative abundance and species composition, because not all seeds within a seed bank are viable (Hutchings and Russell 1989; van der Valk et al. 1992). Manual seed separation and identification is also very time consuming and requires specialised skills for accurate identification to the species taxonomic level. The latter results in an underestimate of the potential of a given seed bank to represent the vegetation composition of the site, because propagules, rhizomes, tubers and other vegetative parts of plants known to produce new plants are removed. The emergence method is by far most commonly used because of its simplicity and reliability to produce an estimate of relative abundance (Brown 1998; van der Valk et al. 1992).

van der Valk (1989) proposed that species composition at newly created wetlands could be predicted from the vegetation composition and environmental moisture regime of the wetland from which the soil was imported from. Haukos and Smith (1993; 1998) determined that this model became less reliable as environmental variability increased. As previously discussed, in a wetland setting, species assemblages are controlled by a variety of chemical, physical and biological factors. The hydrologic regime to which a seed bank is exposed will have great impact on species germination; submersed species will occur in permanently flooded areas and emergent species are typically found in seasonally flooded and moist environments, although once established, some emergent species can survive under permanently flooded conditions. Using the emergence method of seed bank analysis, an estimate of both submergent and emergent species composition is obtained by subjecting soils to flooded and moist soil treatments.

In addition, species assemblages differ according to the type of substrate /growing environment to which propagules are exposed to (Rybicki and Carter 1986). In the case of soil capped CT wetlands, emergence from the seed bank could differ compared to the wetland where the soil was salvaged from, possibly due to the physical and/or chemical characteristics of the underlying CT material (secondary soil layer). It is not presently understood whether the presence of CT as a secondary soil layer will influence germination and emergence of wetland plant species, possibly resulting in reclaimed wetland plant communities that are dissimilar to those of natural or pre-mining wetlands. This information is needed to help resolve the degree of inputs, management and monitoring necessary to produce a productive, stable, and self-sustaining ecosystem that is continuous with the surrounding landscape.

1.8 Thesis Overview

In Chapter 2, I examine vegetation development on a newly constructed CT wetland and the fates of soil transfers from natural and opportunistic wetlands. In addition, I examine colonization rates of CT transferred into a natural and opportunistic wetland. In Chapter 3, I assess seed bank development of various opportunistic, constructed, altered and natural wetlands within Suncor and Syncrude Leases. In Chapter 4, I evaluate the effectiveness of using seed bank trials as a metric to assess plant community development in reclaimed wetlands. In Chapter 5, I suggest techniques for future management and directions for future research.

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Chapter 2: Vegetating Consolidated/ Composite Tailings Wetlands Using Salvaged Wetland Soils, in the Oil Sands Mining Region of Alberta

2.1 Introduction

Creating functional and self-sustaining wetlands in oil sands mining areas of northeastern Alberta brings tremendous challenges. Wetland habitats exist along a transitional zone from terrestrial to deepwater aquatic habitats. Simple changes in hydrology can shift wetland conditions to support different life forms. Such is the case with wet post-mining sites. The initial alteration of the boreal terrain by surficial oil sands mining is profound and of a magnitude (locally) comparable to volcanism or glaciation. Prior to the commencement of mining, vegetation is removed, and topsoil (referred to as muskeg) is either is mixed with overburden and stockpiled for later use as a reclamation amendment, or directly placed onto a new reclamation site. Unlike natural processes, mining alterations to habitats are planned and within this planning are designs for establishing specific replacement habitats after mining. As leaseholders from the Province of Alberta, Suncor Inc. (Suncor) and Syncrude Canada Ltd. (Syncrude) are required to reclaim disturbed sites to end land-uses that have equivalent or better land capability compared with pre-disturbance conditions, and end land-uses must also be continuous with surrounding leases (Alberta Environment 1999). Wetlands existing in the planned mining area are mostly bogs and fens, with some marshes occurring around streams and beaver impoundments (Vitt et al. 1998).

In 1999, Suncor constructed a series of wetlands to demonstrate the potential for establishing plant communities on Consolidated/ Composite tailings (CT) deposits (Golder Associates Ltd. 2000). The CT deposits are a gypsum-enriched (inorganic coagulant) saturated slurry of mature fine tails produced as a by-product of processed oil sands (MacKinnon et al 2001; Matthews et al. 2002). The texture of the CT in the Suncor constructed wetlands varies slightly from that of Syncrude's CT deposits; there is more clay in Suncor's constructed CT wetlands (Mackinnon, Personal Communication 2004). Both produced waters and CT are slightly to moderately saline, high in total dissolved solids, and contain elevated ionic concentrations of sulphates, chlorides, sodium, as well as naphthenic acids and trace elements. Backfilled mine cells hold large volumes of CT, and it is expected that wetlands will be constructed on a large proportion of CT deposits. These constructed wetlands are predicted to develop into shallow, highly productive water habitats, connected with the surface drainage for that area (Golder Associates 2000). Previous studies have indicated that many of the CT constituents can negatively affect terrestrial plant performance (plant recruitment, growth, development and reproduction), (Apostal 2002; Renault et al. 1998; 1999), but it is unknown how wetland plants will respond to the physical and chemical CT environment.

There are two models or approaches that are commonly applied for wetland restoration and creation projects: (1) self-organization/ self-design and (2) ecosystem restoration/ designer wetland. The former relies on the "self-organising ability of ecosystems in which natural processes contribute to species introduction and selection" (Mitsch et al. 1998). An ecosystem will organize by "selecting for the assemblage of plants, microbes and animals that is best adapted for the existing conditions" (Mitsch et al 1998). Alternatively, the ecosystem restoration/ designer wetland approach refers to introducing plants and measuring restoration success based on their survival and expansion (Mitsch et al. 1998). Because current knowledge of plant tolerances to chemical and physical conditions present in the CT wetland is extremely limited, it is likely that the designer approach would be less cost-effective and/ or sustainable, compared to the self-organisation/ self design approach which relies on natural selection processes.

Under the self organisation/ self design approach, typical options for establishing wetland vegetation include: (1) natural recovery using slurried salvaged wetland soils, and (2) seeding or transplanting wetland plant species endemic to the region. Seeding and transplanting are presently not considered feasible methods for vegetating large-scale reclamation projects, such as post-mining CT deposits. Transplanting would require significant manual labour requirements, and would only be feasible along wetland fringes, because fresh CT has a very low specific gravity and will not support humans or even tracked equipment. Legislation requires oil sands operators to use seed native to the Fort McMurray region, and native seeds for wetland plant species are not available commercially. Natural recovery provides an adaptive management opportunity to seek out plant species that thrive in the CT wetland conditions, with the assurance that they are indigenous to the Northeastern Boreal Region. Although natural recovery appears to be the most cost-efficient method to vegetate large areas, it is not known how effective this method will be in producing diverse, productive, and self-sustaining plant communities, within the planning horizon of 10-15 years.

Establishment of suitable hydrology and substrate (soil, detritus and plants) is expected to be an essential basis for a functional food web and for ameliorating the deficiencies of recently processed CT subsoils. Soil and water chemical and physical characteristics control the levels of available nutrients, which will affect the floristic composition of wetland vegetation and soil microflora and fauna (Chambers et al. 1994). Muskeg capping, like salvaged wetland soils used in many creation/ restoration projects, will provide sources of propagules, seeds, nutrients and serve as a substrate providing anchorage to emerging and colonizing wetland plant species (van der Valk and Pederson 1989). Salvaging wetland soils from mined areas, and then directly placing the material onto a reclamation site (referred to as soil transfers) will likely accelerate wetland plant community development, resulting in a wetland with higher species richness and cover, than what would be expected from natural invasion processes (seed dispersal through wind and wildlife movement) (Mueller and van der Valk 2002).

The effectiveness of natural recovery through soil transfers to enhance species richness and plant densities at created wetlands has been well documented throughout the United States and in parts of Canada (McKnight 1992; Smith and Kadlec 1983; Vivian-Smith and Handel 1996). To date, in the oil sands mining region of Alberta, pilot studies have examined natural recovery of CT wetlands using muskeg capping in conjunction with other techniques, such as transplanting individual plants and vegetation islands (Golder Associates 2000; 2002), but none have compared establishing CT vegetation communities against those of salvaged or reference wetlands, typical of the region. Furthermore, isolation and placement of CT materials in natural wetland settings is necessary to assist in understanding effects of CT and CT water characteristics on wetland plant species assemblages and vegetation community development. This information is needed to help resolve the degree of inputs, management and monitoring necessary to produce a productive, diverse, and self-sustaining ecosystem that is continuous with the surrounding landscape.

The purpose of this study was to assess the effectiveness of natural recovery through the inputs of wetland soils from reference wetlands to vegetate CT wetlands. Specifically I addressed the following questions:

(1) Whether chemical and physical conditions of CT wetlands would limit plant emergence, growth and survival from the seed banks of reference wetland soils?

a. I compared species richness, aerial percent cover and similarity of the vegetation between three soil treatments: control (bare CT plots), and two reference wetland soil types (Natural and Opportunistic wetland soils) transferred into the CT wetland.

b. I compared species richness, aerial percent cover and similarity of the vegetation between two reference wetland soil types (Natural and Opportunistic wetland soils) transferred into the CT wetland, against plots in the reference wetlands (Natural and Opportunistic wetlands).

(2) Whether or not local plant species would naturally colonise CT materials that had been placed under natural wetland conditions?

a. I compared species richness, percent aerial and similarity of the vegetation between CT plots transferred into two types of reference wetlands (Natural and Opportunistic) against control plots (bare CT) in the CT wetland.

2.1 Methods

2.1.1 Experimental Design

The study was designed as a reciprocal or switchback experiment (MarAnon and Bartolome 1993), wherein experimental soil plots were laid out in reference wetlands (one Natural and one Opportunistic marsh). These field plots were randomly paired with plots in a constructed CT wetland (Figure 2-1). The Opportunistic wetland was not classified as a natural wetland, because it formed "opportunistically" in an area of altered drainage patterns adjacent to mining operations. The Opportunistic wetland formed on a sodic soil and was included in the study design, because it was hypothesized that plant species occurring in this wetland might be more tolerant to salinity in pore waters, resulting in enhanced survival in the CT wetland.

I used a completely randomized block design to lay out soil plots in each of the three wetlands (Figure 2-2). Three 20 m transects (separated by 15 m) were spaced across both reference wetlands to mimic boardwalks that had been constructed in the CT wetland. Each side of the boardwalk/ transect was treated as one block, for a total of 6 blocks in each wetland. In the CT wetland, five replicates of each of the three soil treatments (Opportunistic and Natural wetland soils, and CT (control)) were randomly assigned to each block. The exact same design was employed in the reference wetlands, except there were only two soil treatments: reference soil and CT plots (i.e., Opportunistic soil was not switched into the Natural wetland and vice versa because such a comparison was considered trivial). Soil buckets (plots) (as described below) were separated by 0.5 m along each transect/ boardwalk. A total of 210 plots were established: 60 plots in each of the Natural and Opportunistic wetlands and 90 plots in the CT wetland.

Prior to sediment removal, vegetation stems were clipped to the soil surface. To prevent soil washing into other plots or out of the experimental wetlands, all samples were placed in 10 L buckets. Each bucket was filled with the corresponding substrate (eg., CT in the CT wetland) and then the reference wetland soil was placed, surface-oriented, in the top 20 cm of the buckets. Holes (1 cm diameter) were drilled into the sides (4 columns) of the buckets, plus one hole was drilled in the bottom. The holes were then covered with a commercial landscape fabric to allow for movement of water and ions into the soils, while preventing soil loss from the buckets. Surface soils from the CT wetland were likewise placed on the excavation location in the Natural and Opportunistic wetlands. Transplanting occurred during June and early July 2002, after the upper 20 cm of soil thawed.

2.1.2 Vegetation, water and soil sampling

Vegetation assessments were conducted in mid August 2002 and 2003. Each plot was visually assessed for aerial percent cover by species. All vegetation surveys were conducted by the same person to reduce observer bias and variability. Difficult species were brought to the lab and identified using Flora of Alberta (Moss 1994) in consultation with botanists from Golder Associates Ltd. and the University of Alberta. In August 2003, 3 buckets per treatment were randomly removed from all 3 wetlands and a qualitative root assessment was conducted. Presence of roots and depth of rooting were noted.

Three surficial water samples were collected monthly (June – August 2002 and 2003) at permanent sampling locations (transects 1, 3, 5) in each of the three treatment wetlands and analysed at the Syncrude Research Centre for major ions, trace metals, and naphthenic acids. Water samples were filtered (0.45 um milipore) prior to analyses. Metals (Al, B, Ba, Ca, Cd, Cl, Co, Cr, Cu, F, Fe,Li, K, Mg, Mn, Mo, Ni, Pb, Sb, Se, Si, Sr, Ti, V, Zn, Zr) were analysed using an Inductively Coupled Argon Plasma Atomic Emission Spectrometer (ICAP-AES). Anions (PO_4^{3-}, NO_3^-, NO_2^-, SO_4^{2-}, CO_3^{2-}, HCO_3^-) were analysed by Ion Chromatography. Electrical conductivity, pH, and temperature were measured in the field using a portable YSI 63 multi-meter at the time of water sampling. In August 2003, water samples were also sent to the Limnology Lab at the University of Alberta for analysis of total nitrogen and phosphorous. Because there were no surficial waters present in the Natural wetland in August 2003, water samples were taken at random points on an adjacent stream that floods the study area during high water levels.

In August 2002 and 2003, soil samples were taken at two depth increments (0-10 and 10-20 cm) both within (bucket samples) and outside (reference samples) the buckets, using a 5 cm diameter, plastic coring tube and/ or a 6 cc syringe. Two bucket samples per treatment were randomly chosen in transects 1, 2 and 6 at each of the three wetlands. Soil samples from outside the buckets were taken at permanent sampling locations (transects 1, 3, 5). Using a 2 time extraction with distilled water procedure (sample: water ratio 1:1), soil samples were centrifuged, filtered (0.45 um milipore). Leached porewaters were analysed for major ions, trace metals and naphthenic acids. Particle size distribution and total oil, water, solids content were also analysed on a subset of the original samples. Particle size distribution was determined using a Coultier LS130 laser diffraction particle analyzer (Syncrude Canada Ltd. 1995). Percent oil, water and solids content was determined by refluxing tolulene in a soxhlet extraction apparatus (Syncrude Canada Ltd. 1995). All soil preparation and analyses were done by technicians at the Syncrude Research Center.

Surficial and pore waters were classified with Piper plots using Hydrochem Software (Rockware 1997). Cations (Mg^{2+}, Ca^{2+}, Na^+, K^+) and anions (HCO_3^-, Cl^-, SO_4^{2-}) are expressed in trilinear diagrams in terms of their relative percentages of the individual ion levels (meq/ L) of total cations and anions, respectively.

2.1.3 Statistical analyses

All statistical analyses were done using the SPSS 14 statistical program (SPSS Inc. 1999). To control for Type I errors due to the large sample size, a more conservative alpha level of 0.01 was chosen, compared to alpha level of 0.05 commonly used in ecological studies. Before prediction-testing analyses were conducted, all data were tested for normality using histograms and for homogeneity of variance using box plots and the Levense Statistic (Conover 1980). Consequently, nonparametric statistics were used, because the data violated assumptions of homoscedasticity and normality. Vegetation data were rank transformed and two-way nonparametric repeated measures general linear models were used to examine whether differences in percent cover and species richness were significant between soil treatments (Conover and Iman 1981). Mean sums of squares were then adjusted to calculate an H value and then compared against the critical H values of the Chi Square distribution to determine significance. *Post hoc* multiple comparisons were made using Dunnett's C test (Zar 1999).

Sorensen's Qualitative Index was used to assess the similarity of species composition for soil transfers and reference plots in the CT, Opportunistic and Natural Wetlands over time (August 2002 and 2003 sampling periods) (Sorensen 1949 as cited in Magurran 1988). Sorensen's Qualitative Index (IS) is defined as:

IS (qualitative) = 2c / (a +b),

- where a is the sum of all species (i.e., species richness) for one treatment (wetland soil),
- b is the sum of all species for the second treatment (wetland soil),
- and c is the number of species common to both treatments (wetland soil).

Sorensen's Qualitative Index is "Qualitative", because the presence or absence of a species is only considered in the similarity formula; the contribution of that species to the total abundance of individuals within the seed bank is not accounted for. Sorensen's Qualitative Index has been regarded as a useful index of similarity between species lists (Magurran 1988). A value of 1 would indicate that the treatments were identical, while a value of 0 would indicate that the wetland groups had no species in common.

2.3 Results

2.3.1 Vegetation Performance

Photos of soil transfer plots in the CT wetland and corresponding reference plots are shown in Appendix A-1.

2.3.1.1 Soil transfer plots in the CT wetland

Soil transfers from reference wetlands significantly increased both percent cover and species richness in the CT wetland, compared to what was seen on bare CT plots ($p< 0.0001$, Table 2-4). Vegetation cover for Opportunistic and Natural wetland soil transfer plots were significantly greater than that of CT plots (Opportunistic and CT: mean difference = 41.55, Natural and CT: mean difference = 38.94, ά = 0.01, Table 2-4). Differences in percent cover between Opportunistic and Natural wetland soil transfer plots in the CT wetland were not significant. Opportunistic wetland soil transfer plots supported the most species, followed by Natural wetland soil transfer plots and CT soil plots, respectively (Opportunistic and CT: mean difference = 45.03, Natural and CT: mean difference = 26.97, Opportunistic and Natural: mean difference= 18.07, ά = 0.01, Table 2-4).

After one growing season (August 2002), the CT plots remained bare, except for minor colonization of *Chara spp.* (Tables 2-1, 2-2). Both the Natural and Opportunistic wetland soil transfer plots showed good cover by a few species (Mean percent cover, Opportunistic: 49, Natural: 52, Table 2-2). Natural wetland soil transfer plots were dominated by moss with some scant individuals of *Typha latifolia, Myriophyllum exalbescens* and *Polygonum spp.*. Opportunistic wetland soil transfer plots were dominated by *Chara spp.* and *Potamogeton spp.,* with some *Typha latifolia.*

Overall the similarity of the vegetation in terms of species composition increased between August 2002 (year 1) and August 2003 (year 2) (Table 2-3) for all soil treatments in the CT wetland. In August 2002, *Scirpus validus* and *Typha latifiolia* colonised some of the Natural, and Opportunistic wetland soil transfer plots and CT soil plots. From year 1 to year 2, mean percent cover increased substantially for Opportunistic wetland soil transfer plots, while CT soil plots showed only a slight increase and Natural wetland soil transfer plots remained relatively unchanged (Table 2-2). After year 2, *Myriophyllum exalbescens* disappeared from both

Opportunistic and Natural transfer soil plots. Overall, moss cover decreased in the Natural wetland soil transfer plots; a layer of iron oxides or a biofilm was noted on the surface of the moss. In addition, some of the Natural wetland soil transfer plots either sank or were inundated with CT.

Visual qualitative rooting assessments revealed that plant roots were not constrained to the upper layers of the reference wetland soil transfers in the CT wetland. Most plant roots extended past 20 cm for all treatments, well into the CT subsoil layer.

2.3.1.2 Soil transfer plots in the CT wetland vs. soil plots in the reference wetlands

2.3.1.2.1 Natural wetland soil transfers vs. reference plots

Live vegetation cover and species richness of the Natural wetland reference soil plots were significantly higher than soil transfer plots in the CT wetland ($p < 0.0001$, Table 2-5). Increases in species richness over time were greater for Natural wetland reference soil plots than soil transfer plots in the CT wetland ($p < 0.0001$, Table 2-5). Species composition of Natural wetland reference soil plots were not similar to that of soil transfer plots in the CT wetland (Tables 2-1, 2-2); overall similarity decreased between August 2002 (year 1) and August 2003 (year 2) (Table 2-3). In August 2002, water levels dropped in the Natural wetland, because of beaver dam failures upstream (Figure 2-3). Hence the vegetation community of the Natural wetland reference plots shifted from an aquatic to a wet meadow type (Table 2-1). In 2002, Natural wetland reference soil plots were dominated by moss, *Polygonum spp*., and *Lemna minor*. In 2003, wet meadow species emerged from the seed bank and dominated the vegetation cover *(Carex spp., Agrostis scabra, Aleopecurus spp., Glyceria grandis* and other *Poeaceae spp*). *Epilobium cilatum, Potentilla gracilus* and *palustris, Utrica diocia, Ranunculus acicularis*, and other forb species also emerged from the Natural wetland soil seed bank. Species assemblages of the Natural wetland soil transfer plots in the CT wetland are discussed previously in Section 2.3.1.1 (Table 2-1).

2.3.1.2.2 Opportunistic wetland soil transfers vs. reference plots

Aerial percent cover of Opportunistic wetland soil transfer plots in the CT wetland and Opportunistic reference wetland soil plots were not significantly different between wetlands, blocks and years. However, species richness was significantly different between Opportunistic wetland soil transfers in the CT wetland and Opportunistic reference wetland soil plots; both wetland and wetland x block were significant ($p < 0.0001$, Table 2-6). On average, soil transfer plots in the CT wetland had more species than reference plots. In year one, Opportunistic wetland soil transfer plots in the CT wetland and Opportunistic reference wetland soil plots shared some of the same species, including *Potamogeton spp.*, and *Chara spp.* (Tables 2-1, 2-3). Overall the species composition-similarity of the soil plots increased between August 2002 (year 1) and August 2003 (year 2) (Table 2-3). Species assemblages of the Opportunistic wetland soil transfer plots in the CT wetland are previously discussed in Section 2.3.1.1 (Table 2-1).

2.3.1.3 Isolated CT plots in reference wetlands

Vegetation cover and species richness for CT transfer plots in the Opportunistic and Natural wetlands were significantly greater than that of CT plots in the CT wetland (PERCENT COVER: Opportunistic and CT mean difference = 25.78, Natural and CT mean difference = 47.40; SPECIES RICHNESS: Opportunistic and CT mean difference = 21.89, Natural and CT mean difference = 42.3, $\acute{\alpha} = 0.01$, Table 2-7). CT transfer plots in the Natural wetland had more vegetative cover and species compared to CT transfer plots in the Opportunistic wetland (PERCENT COVER mean difference = 25.77, SPECIES RICHNESS mean difference = 20.42, $\acute{\alpha}=0.01$, Table 2-2). Increases in species richness for the CT transfer plots in the Natural wetland were much higher than what was observed for the CT transfer plots in the Opportunistic wetland and reference plots in the CT wetland (wetland x year: $0.01 < p < 0.001$).

After one growing season (August 2002), CT plots in the CT wetland remained bare, except for minor colonization of *Chara spp.* (Table 2-1, 2-2). CT transfer plots in the Opportunistic wetland were sparsely covered with *Chara spp.* and *Potamogeton spp.,* with some *Scirpus validus and Myriophyllum exalbescens* (mean percent cover: 8, Tables 2-1, 2-2). CT transfer plots in the Natural wetland were dominated by moss with some scant individuals of *Carex spp.* and *Lemna minor* (mean percent cover: 45, Tables 2-1, 2-2).

Overall the species composition-similarity of the vegetation between CT transfer plots in both reference wetlands, compared to reference plots in the CT wetland showed little change between August 2002 (year 1) and August 2003 (year 2) (Table 2-3). In August 2003, *Scirpus validus* and *Typha latifiolia* colonised some of the reference CT plots in the CT wetland. Mean percent cover increased substantially for CT plots in the Opportunistic and Natural wetlands, while CT plots in the CT wetland showed only marginal increases (Table 2-2). *Eleocharis acicularis,* and *Potamogeton pectinatus* colonised the CT transfer plots in the Opportunistic wetland (Table 2-1) CT transfer plots in the Natural wetland were colonised by a number of emergent, wet meadow and forb species such as: *Agrostis scabra, Aleopecuris spp., Epilobium ciliatum, Polygonum spp., Poeaceae spp., Potentilla gracilius, Typha latifolia, Urtica diocia,* and *Vaccinium spp.* (Table 2-1).

A salt layer was found on most of the drier CT transfer plots in the Natural wetland in August 2003. Visual health assessments revealed that the plants were showing signs of stress in the form of chlorosis (browning and yellowing of stems), and necrosis of moss spp..

2.3.2 Water and Soil Characterization

2.3.2.1 Surface waters

Water quality and depth measurements revealed differences in electrical conductivity, pH, and water depth between wetlands (Appendix B-1). Surface waters of the Opportunistic and CT wetlands were slightly basic, with a pH of ~8.5, whereas the pH of Natural wetland surface waters were near neutral. Based on the electrical conductivity, surface waters in the Natural wetland are classified as freshwater (296 uS/cm, September 2002), whereas CT wetland surface waters were slightly saline (2143 and 1987 uS/cm, September 2002 and August 2003, respectively). Electrical conductivity levels in the Opportunistic surface waters were intermediate between the other two wetlands (conductivity= 945 and 540 uS/cm, September 2002 and August 2003, respectively). Water levels in the CT wetland remained relatively stable between years at ~ 20 cm (Figure 2-3). In the Opportunistic wetland, water levels increased between years, from ~10 cm in 2002, to ~20 cm in 2003. In the Natural wetland, early season water levels of ~55 cm dropped to ~5 cm by August

2002. Water levels did not recover in 2003 and by August 2003 there was no standing surface water in the study area.

Hydrochemical facies of surface waters differed between sites. Natural wetland surface waters were dominated by $Ca(HCO_3)_2$, whereas the Opportunistic and CT wetlands were dominated mainly by $NaHCO_3$ and Na_2SO_4 (Appendix C-1, C-2). Surficial water chemistry was variable between sampling rounds and years for the CT and Opportunistic wetlands. Higher levels of Ca^{2+} were found in the CT surface waters in early season samples (June 2002, July 2003). In 2002, surface waters of the Opportunistic wetland were dominated by Na_2SO_4, whereas in 2003, $NaHCO_3$ dominated. A significant drawdown occurred during the 2002 season, but water levels recovered in 2003 (Figure 2-3). Compared to the Natural Wetland, both the Opportunistic and CT wetlands had elevated levels of Na^+, Mg^{2+}, Cl^-, $HCO3^-$, SO_4^{2-}, B, and S. Trace amounts of Ba and NH_4^+ were detected in the CT surface waters.

2.3.2.2 Porewaters

2.3.2.2.1 CT wetland plots

Leached porewaters of soil samples taken from plots in the CT wetland were similar in terms of hydrochemical facies between treatments, years and depths (Appendices C-3, C-4). Porewaters were dominated by $NaHCO_3$, and to a lesser degree Na_2SO_4. Trace amounts of Ti, V, Zr, and Cr were detected in CT plots and CT reference soils, but not in Opportunistic and Natural wetland soil transfer plots (Appendix B-2).

2.3.2.2.2 Soil transfer vs. reference soil plots

Porewaters of samples taken inside Opportunistic wetland reference soil plots nd samples taken outside the plots (control areas) were dominated by $Ca(HCO_3)_2$ and $Mg(HCO_3)_2$ with some $CaSO_4$ and $MgSO_4$ (Appendices C-5, C-6). These trends were seen for both sampling years (2002, 2003) and depths intervals (0-10 and 10-20 cm). Na^{2+} and K^+ porewater concentrations were higher in samples taken from Opportunistic soil transfer plots in the CT wetland, than samples taken from Opportunistic reference soil plots and control areas, whereas Ca^{2+} and SO_4^{2-} porewater concentrations were lower (Appendix B-3).

Porewaters of samples taken inside Natural wetland reference soil plots and control areas were dominated by $Ca(HCO_3)_2$ and to a lesser degree $Mg(HCO_3)_2$ and $NaHCO_3$ (Appendices C-5, C-6). These trends were seen for both sampling years (2002, 2003) and depth intervals (0-10 and 10-20 cm). Na^{2+}, K^+, Mg^{2+} porewater concentrations were higher in samples taken from Natural soil transfer plots in the CT wetland, than samples taken from Natural wetland reference soil plots and control areas (Appendix B-4).

Trace amounts of B, Ba, Cr, F, Ti, V, and Zr were detected in both Opportunistic and Natural soil transfer plots in the CT wetland, but not in samples taken from reference soil plots and control areas in the Opportunistic and Natural wetlands.

2.3.2.2.3 CT Transfers

Porewaters of samples taken from CT transfer soil plots in the Natural wetland were dominated by $NaHCO_3$, $Ca(HCO_3)_2$ and $Mg(HCO_3)_2$ and to a lesser degree Na_2SO_4, $MgSO_4$, and $CaSO_4$, whereas samples taken from CT transfer soil plots in the Opportunistic wetland and reference plots in the CT wetland were dominated mainly by $NaHCO_3$ with some $Ca(HCO_3)_2$ and $Mg(HCO_3)_2$ (Appendices C-7, C-8). Ca^{2+} and SO_4^{2} concentrations in the porewaters of samples taken from CT transfer soil plots in the Natural wetland in 2003 were noticeably higher, while Mg^{2+} concentrations and overall conductivity were lower compared to samples taken from the CT reference plots and control areas in the CT wetland.

2.3.2.3 Soil Physical Properties

Percent water, solid and oil content and particle size distribution of soil transfers and reference soils in all wetlands are outlined in Appendix B-6.

2.3.2.3.1 CT wetland plots

Soil textures at both 0-10 and 10-20 cm depth intervals were similar for samples taken from CT soil plots and outside the plots (control areas) with approximately 20% < 2.8 μm (clays and fines), 20% >2.8 and <22 μm (silt) and 60% >22 μm (sand and coarse fragments). All contained approximately 30% water, 67% solids and 3% oil at the 0-10 cm depth interval and, 23% water, 74% solids and 2% oil at the 10-20 cm depth interval.

Samples taken from Opportunistic transfer soil plots contained more clay than CT soils, with approximately 46% and 31% <2.8 μm, at the 0-10 and 10-20 cm depth intervals, respectively. Samples taken from the Opportunistic transfer soil plots in the CT wetland contained less percent oil (1%) than CT soil plots.

Soil textures for the Natural wetland soil transfers were similar at both 0-10 and 10-20 cm depth intervals, with approximately 16% < 2.8 μm, 25% >2.8 and <22 μm and 59% >22 μm fractions. Because of the highly organic nature of this soil, percentages of sand, silt and clay can not be determined with accuracy. Samples taken from the Natural wetland soil transfer plots in the CT wetland contained approximately 60% water, 40% solids and <1% oil content at both sampling depth intervals.

2.3.2.3.3 Soil plots in the Reference Wetlands

At the 0-10 cm sampling depth interval, samples taken from Opportunistic reference soil plots contained approximately 44% <2.8 μm, 36% >2.8 and <22 μm and 20% >22 μm, whereas samples taken outside the buckets (control areas) contained approximately 66% <2.8 μm, 34% >2.8 and <22 μm and 0% >22 μm. At the 10-20 cm depth interval, samples taken from the reference soil plots and control areas were similar, with approximately 65% <2.8 μm, 35% >2.8 and <22 μm and 0% >22 μm. Samples taken from the Opportunistic reference soil plots and control areas contained approximately 30% water, 70% solids and trace oil at both the 0-10 and 10-20 cm depth intervals.

For the 0-10 cm depth interval, soil textures of the Natural wetland reference oil plots and control areas were similar, with approximately 15% <2.8 μm, 15% >2.8 and <22 μm, and 70% >22 μm. At the 10-20 cm depth interval, the samples taken from the control areas contained more sands and coarse fragments compared to the Natural wetland reference soil plots (66% and 39%, respectively). Samples taken from the Natural wetland reference soil plots and control areas contained approximately 30% water, 70% solids and trace oil for both the 0-10 and 10-20 cm depth intervals.

2.3.2.3.3 CT transfers

Soil textures of samples taken from CT transfer soil plots in the Opportunistic and Natural wetlands and samples taken from CT plots and control areas in the CT wetland were similar for the 0-10 cm depth interval, with approximately 20% <2.8 μm, 20% >2.8 and <22 μm and 60% >22 μm. For the 10-20 cm depth interval, samples taken from the CT transfer soil plots in the Natural and Opportunistic wetlands contained more clay (~30%), than *in situ* CT plots and control plots in the CT wetland, indicative of the CT layer mixing with the lower soil layer (corresponding subsoil in the reference wetlands) in the buckets. Samples taken from CT soil transfer plots in the Opportunistic wetland contained lower amounts of oil (~1%) than that of samples taken from *in situ* CT plots and control areas in the CT wetland, but were similar in terms of % water and solids. Samples taken from CT transfer plots in the Natural wetland had lower % water contents (~21%) at both depth intervals.

2.4 Discussion

2.4.1 Assessing the effectiveness of natural recovery in accelerating plant colonization in CT wetlands

The results presented in this study indicate that natural recovery is effective in accelerating plant community colonization under CT wetland conditions. After two growing seasons, soil transfer plots showed reasonable cover. Visual assessments of plant health and rooting depths did not indicate that physical and chemical conditions inhibited plant growth and survival for most species. One exception would be the disappearance of *Myriophyllum exalbescens* from the plant communities of both treatments in year two (August 2003). Based on the limits of this experiment, I can not fully explain the limited persistence of *Myriophyllum exalbescens* in the CT environment, but I speculate that water chemical factors such as turbidity and/ or salinity may have reduced its survivability. In July 2003, extremely turbid waters (less than 2 cm visual clarity), lasting approximately two weeks, were noted. Previous studies have indicated that turbidity and salinity can affect the survival of aquatic wetland plants (Howes Keiffer and Ungar 1997; Sand-Jensen et al. 2000). Continued monitoring and/or manual reintroduction would be required to determine whether *Myriophyllum exalbescens* will re-establish and persist under the CT wetland conditions. It should also be noted that not all species present in the CT plots in 2003 were from transferred soil seed banks. Most of the *Typha latifolia* and *Scirpus validius* likely colonised due to seed rain from areas adjacent to the plots.

It could be argued that two years of monitoring in this study is inadequate to predict the long-term sustainability of wetland plant communities on CT. Work by Golder Associates Ltd. (2002) has shown survival and expansion of *Typha latifolia* plots under similar CT wetland conditions. It remains to be seen whether other plant species that have established in the CT wetland plots will respond similarly. A wide range of tolerances to chemical and physical characteristics exists among the aquatic plant taxa (Deschenes and Serdoes 1984; Foote 1991), and responses of one aquatic plant species can not be used to predict growth and survival for other wetland plant species. However this work shows that the use of salvaged wetland soils does speed the development of a vegetation community, indicated by the survival of most species once established. The initial establishment of vegetative cover is an important step in the series benchmarks leading to recreating functional wetland ecosystems. Over time, these wetland species may alter the soil environment by increasing the redox potential through oxidizing the surrounding rhizosphere (Crawford 1983) and contributing organic matter through growth and die back (Mitsch and Gosselink 1986). This "soil conditioning" may allow constitute an expansion of the niche space available and allow for the establishment of other species in the system. Continued monitoring of these plots will allow more accurate predictions as to the possible sequences of species assemblages that these CT wetland plant communities are capable of over the longer term.

2.4.2 Examining CT wetland vegetation community similarity with reference wetlands

Although plant cover was similar between the Opportunistic soil transfer plots in the CT wetland and Opportunistic wetland reference soil plots, species composition of the CT wetland soil transfers differed from that of both reference wetlands. This is not entirely surprising or unusual, considering the variability of physical conditions in the reference wetlands over the course of the study period. Whereas physical and chemical factors remained relatively stable in the CT wetland, water depths fluctuated in both reference wetlands. Species composition and abundances of wetland plants are known to vary with hydroperiod (Shay et al. 1999). Short-term drawdowns, followed by extended flooding periods or the reverse pattern have been shown to shift vegetation structure (Baldwin et al. 2001, Gerritsen and Greening 1989; Shneider 1994). In year one of the study (2002), the Opportunistic wetland experienced a significant drawdown due to drought conditions in the Northeastern Boreal Forest Region. By the end of the season, emergent species such as *Sparganium angustifolium* and *Typha latifolia* had germinated and begun to establish. In year two (2003), many of these emergent individuals disappeared from the system as the water levels rose, and the system switched from an shallow emergent plant community to an aquatic submersed plant community. An opposite hydrologic shift occurred in the Natural wetland, wherein July 2002 a beaver dam cascade lowered water levels from deep aquatic to those typical of wet meadows. The vegetation responded accordingly, resulting in germination and establishment of various emergent and wet meadow plant species by August 2003. Therefore species composition of the vegetation in the CT wetland soil plots cannot be directly evaluated against that of the reference wetlands, without considering hydrologic differences between wetland sites.

Previous studies in saline and brackish marsh ecosystems, where plant communities are dominated by only one or a few species, have identified salinity as a major factor contributing to low species richness (Galinato and van der Valk 1986; Hammer and Heseltine 1988). Similarly, the CT wetland plant communities were very species poor, comprised of *Potamogeton spp., Chara spp., Typha latifolia,* and *Scirpus validus* and additionally moss in the Natural wetland transfers. Interestingly, Hammer and Heseltine (1988) found that only *Potamogeton pectinatus, Ruppia maritima* and *Ruppia occidentalis* are found in great abundance in high-salinity lakes. I speculate that salinity in surface and pore waters is another factor that is inhibiting the initial establishment of species from the seed and bud banks of the Natural and Opportunistic wetland transfers. More controlled laboratory and/or field experimentation, replicating physical conditions in the CT wetland, while isolating the effects of salinity in surface and pore waters, is needed to support this hypothesis.

Luong (1999), showed reduced growth of *Myriophyllum spicatum,* and *Potamogeton richardsonni* under saline surface waters (similar to those in the CT environment), whereas *Chara vulgaris* maintained similar growth patterns. I further speculate that over time, we may see a similar trend in the CT wetland vegetation communities, because of the elevated concentrations of Na^{2+} and SO_4^{2-} in the surface and porewaters. Over time, we may see stunted growth and possibly necrosis, as the plants accumulate Na^+ and SO_4^{2-} ions in their tissues, and are less able to bar, concentrate or dilute these ions in their tissues (Howes Keiffer and Ungar

1997). Continued monitoring of health and expansion of these plants along with plant tissue analyses is required to further investigate this dynamic.

Understanding and predicting structural and functional equivalencies of post-mining CT wetlands with natural wetlands typical of the Northeastern Boreal Region is important to oil sands mining reclamation practitioners looking to meet legislation requirements. Further examination of the effects of water depth and salinity on species composition and productivity will help in understanding the limits of the natural recovery approach to establish wetland plant communities over time.

2.4.3 Examining vegetation colonization patterns of isolated CT plots in reference wetlands

Understanding the effects of CT and CT water characteristics on wetland plant species assemblages is important to oil sands reclamation practitioners. Isolated CT transfers in reference wetlands showed much higher colonisation rates of wetland plants, compared to what was observed in the CT wetland reference plots (bare CT). Greater availability of seeds and propagules would be expected in older wetlands with well established plant communities and waterfowl populations.

Rapid colonisation and establishment of plants suggests that the physical and chemical characteristics of CT do not prohibit germination and establishment. However, in some of the drier Natural wetland CT transfer plots, a white powder (possibly salt) was present on the soil surface and the vegetation was showing visible signs of stress. This suggests that under unsaturated soil conditions, salts or other parameters in the CT transfer plots may be phytoxic.

Based on the results presented in this study, reclamation practitioners should prevent salt crusting and subsequent vegetation stress by managing flow rates in CT deposits to ensure shallow yet saturated soil conditions. Surficial water chemistry will likely control initial establishment and survival of wetland plant species under the natural recovery approach in CT wetlands. Further controlled experimentation, likely in a laboratory setting, using varying dilutions of CT waters with freshwater over CT capped with wetland soil is recommended. This information would enhance the current understanding of CT water characteristics and its effects on plant establishment and survival under the self-design/ organisation approach vegetating CT wetlands.

2.4.4 Summary

Soil transfers from reference wetlands significantly accelerated plant colonisation rates in CT wetland plots. The composition of wetland plant species in the CT wetland differed from reference wetlands. However, reference wetlands were not an appropriate benchmark to compare wetland species assemblages, because of the natural variability in water regimes within the reference wetlands, compared to the relatively static water regime in the CT wetland. Isolated CT transfers showed significantly higher colonisation rates of wetland plants compared to CT reference plots in the CT wetland. Further research is required to determine if chemical characteristics of CT waters will limit persistence of wetland plant species in CT wetlands.

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Table 2-1. Species composition of the soil transfers and reference plots in the CT, Natural and Opportunistic Wetlands (August 2002 and 2003)

	CT Wetland						Natural Wetland				Opportunistic Wetland			
Species	CT soil		Natural soil		Opportunistic soil		CT soil		Natural soil		CT soil		Opportunistic soil	
	2002	2003	2002	2003	2002	2003	2002	2003	2002	2003	2002	2003	2002	2003
Agrostis scabra								x		x				
Aleopecurus aequalis								x		x				
Carex spp.							x	x		x	x	x	x	x
Ceratophyllum demersum													x	
Chara spp.	x	x			x	x					x	x	x	x
moss spp.			x	x			x	x	x	x				
Eleocharis acicularis												x		
Epilobium ciliatum								x		x				
Glyceria borealis										x				
Lemna minor							x		x			x		x
Mitella nuda								x		x				
Myriophyllum exalbescens			x		x						x	x	x	x
Poeaceae family								x		x				
Polygonum spp.			x				x		x					
Potamogeton spp.					x						x			
Potamogeton pectinatus				x		x						x	x	x
Potamogeton pusilus				x		x						x		x
Potentilla gracilis								x		x				
Potentilla palustris								x		x				
Ranunculus acicularis										x				
Rorripa spp.								x		x				
Scirpus validus		x		x	x	x					x	x		x
Sparganium angustifolium					x								x	x
Typha latifolia		x	x	x	x	x		x		x				
Utricia diocia								x		x				
Vaccinium spp.								x		x				

Table 2-2. Averages (+/- SE) of percent cover and species richness of soil transfers and reference plots in the CT, Natural and Opportunistic wetlands (August 2002 and 2003).

	CT Wetland						Natural Wetland				Opportunistic Wetland			
Parameter	CT soil		Opportunistic soil		Natural soil		CT soil		Natural soil		CT soil		Opportunistic soil	
	2002	2003	2002	2003	2002	2003	2002	2003	2002	2002	2002	2003	2002	2003
Percent Cover	0.03 (0.33)	4.83 (2.39)	49.37 (5.68)	62.67 (5.99)	51.83 (7.64)	52.83 (7.87)	7.97 (2.49)	33.76 (4.67)	81.43 (6.38)	101.1 (5.91)	45.20 (7.46)	68.03 (5.75)	49.37 (5.38)	40.27 (3.80)
Species Richness	0	0.9 (0.2)	2.3 (0.1)	2.6 (0.2)	1.2 (0.1)	2 .0 (0.2)	1.0 (0.1)	3.8 (0.3)	1.2 (0.1)	5.0 (0.2)	0.7 (0.1)	2.0 (0.3)	1.5 (0.2)	2.0 (0.2)

Table 2-3. Sorensen's Similarity Coefficients for the CT wetland plots, Natural wetland soil plots in the CT and Natural Wetlands, Opportunistic wetland soil plots in the CT and Opportunistic wetlands, and CT plots in the CT, Opportunistic and Natural wetlands (August 2002 and 2003). Also included are Sorensen's Similarity Coefficients for the plots over time (August 2002: 2003).

	August 2002	August 2003
CT Wetland Plots		
CT (control) plots: Opportunistic wetland soil transfers	0.5	0.8
CT (control) plots: Natural wetland soil transfers	0	0.5
Opportunistic wetland soil transfers: Natural wetland soil transfers	0.4	0.8
CT (control) plots 2002: CT (control) plots 2003	0.5	
Opportunistic wetland soil transfers 2002: Opportunistic wetland soil transfers 2003	0.6	
Natural wetland soil transfers 2002: Natural wetland soil transfers 2003	0.7	
Natural wetland soil plots in the CT and Natural wetlands		
CT wetland Natural soil transfers: Natural wetland reference plots	0.3	0.2
Natural wetland reference plots 2002: Natural wetland reference plots 2003	0.1	
Opportunistic wetland soil plots in the CT and Opportunistic wetlands		
CT wetland Opportunistic soil transfers: Opportunistic wetland reference plots	0.4	0.5
Opportunistic wetland reference plots 2002: Opportunistic wetland reference plots 2003	0.5	
CT Transfers		
CT wetland (control) plots: Opportunistic wetland CT plots	0.4	0.4
CT wetland (control) plots: Natural wetland CT plots	0	0.1
Opportunistic wetland CT plots 2002: Opportunistic wetland CT plots 2003	0.8	
Natural wetland CT plots 2002: Natural wetland CT plots 2003	0.3	

Table 2-4. Results of a nonparametric two factor repeated measures GLM used to test for differences in percent cover and species richness between Years (August 2002 and 2003), Blocks (1-6) and Treatments (Natural, Opportunistic, CT wetland soils) in the CT wetland. Results of Dunnett's C Test to test for significant differences among treatments are also shown. Bolded effects are significant at ά=0.01

PERCENT COVER					
Source	*SS*	*df*	*H*	*Significance*	
Within Subjects Effects					
Year	0	1	0	0	
Year x Block	899.3	5	3.36	0.696	
Year x Treatment	574.4	2	2.15	0.385	
Year x Block x Treatment x	1216	10	4.54	0.938	
Error	21400	72			
Between Subject Effects					
Treatment	**1909**	**5**	**71.8**	**<0.000**	
Block	65008	2	1.80	0.238	
Treatment x Block	**7750**	**10**	**61.26**	**<0.000**	
Error	19785	72	7.30		
(I) Contrast	*(J) Contrast*	*Mean Difference (I-J)*	*Std. Error*	*99% Confidence Interval Lower Bound*	*Upper Bound*
Opportunistic	**CT**	**41.56**	**2.23**	**34.50**	**48.61**
Natural	**CT**	**38.94**	**3.68**	**27.33**	**50.55**
Opportunistic	Natural	2.62	3.92	9.76	14.99

SPECIES RICHNESS					
Source	*SS*	*df*	*H*	*Significance*	
Within Subjects Effects					
Year	0	1	0	0.1 <p< 0.2	
Year x Block	2818	5	7.30	0.05 <p< 0.1	
Year x Treatment	**1791**	**2**	**11.49**	**0.01 < p< 0.001**	
Year x Block x Treatment x	437.2	10	15.13	0.1 <p< 0.2	
Error	17021	72			
Between Subject Effects					
Treatment	**61632**	**2**	**63.19**	**<0.000**	
Block	2403	5	2.46	0.70 <p< 0.80	
Treatment x Block	3620	10	3.71	0.95 <p< 0.99	
Error	19248	72			
(I) Contrast	*(J) Contrast*	*Mean Difference (I-J)*	*Std. Error*	*99% Confidence Interval Lower Bound Bound*	*Upper*
Opportunistic	**CT**	**45.03**	**2.80**	**36.17**	**53.89**
Natural	**CT**	**26.97**	**3.18**	**16.92**	**37.01**
Opportunistic	**Natural**	**18.07**	**3.33**	**7.56**	**28.58**

Table 2-5. Results of a nonparametric two factor repeated measures GLM used to test for differences in percent cover and species richness between Years (August 2002 and 2003), Blocks (1-6) and Treatment wetlands (CT and Natural wetlands) for the Natural wetland soil transfers. Bolded effects are significant at ά=0.01.

PERCENT COVER				
Source	*SS*	*df*	*H*	*Significance*
Within Subjects Effects				
Year		1		
Year x Wetland	326.7	1	1.93	0.10 <p< 0.20
Year x Block	482.3	5	2.85	0.70 <p< 0.80
Year x Block x Treatment	1040.5	5	6.14	0.20 <p< 0.30
Error	8309	48		
Between Subject Effects				
Wetland	**5796**	**1**	**13.89**	**<0.000**
Block	1954	5	4.68	0.50 <p< 0.30
Treatment x Block	2559	5	6.13	0.30 <p< 0.20
Error	14307	48		
SPECIES RICHNESS				
Source	*SS*	*df*	*H*	*Significance*
Within Subjects Effects				
Year	0	1		
Year x Wetland	**6063**	**1**	**26.10**	**<0.000**
Year x Block	612.5	5	2.63	0.70 < p< 0.80
Year x Block x Wetland	1044	5	4.50	0.30 < p< 0.50
Error	6220	48		
Between Subject Effects				
Wetland	**5950**	**1**	**24.36**	**<0.000**
Block	1712	5	7.01	0.20 < p< 0.10
Treatment x Block	995	5	4.07	0.30 < p <0.50
Error	5754	48		

Table 2-6. Results for a nonparametric two factor repeated measures GLM used to test for differences in percent cover and species richness between Years (August 2002 and 2003), Blocks (1-6) and Treatments (CT, Opportunistic wetlands) for the Opportunistic soil transfers. Bolded effects are significant at ά=0.01.

		PERCENT COVER		
Source	*SS*	*df*	*H*	*Significance*
Within Subjects Effects				
Year	7.28 E-12	1		
Year x Wetland	1280	1	4.05	$0.02 < p < 0.05$
Year x Block	2578	5	8.15	$0.10 < p < 0.20$
Year x Block x Wetland	2469	5	7.81	$0.10 < p < 0.20$
Error	12643	48		
Between Subject Effects				
Wetland	1166	1	4.05	$0.02 < p < 0.05$
Block	3916	5	13.62	$0.01 < p < 0.02$
Wetland x Block	335.7	5	1.17	$0.90 < p < 0.95$
Error	11542	48		
		SPECIES RICHNESS		
Within Subjects Effects				
Year	0	1		
Year x Wetland	392.4	1	1.45	$0.20 < p < 0.30$
Year x Block	**5790**	**5**	**21.44**	**<0.000**
Year x Block x Wetland	1653	5	6.12	$0.30 < p < 0.50$
Error	8362	48		
Between Subject Effects				
Wetland	**4576**	**1**	**17.95**	**<0.000**
Block	467.3	5	1.83	$0.80 < p < 0.90$
Treatment x Block	256.8	5	1.00	$0.95 < p < 0.90$
Error	9734	48		

Table 2-7 Results for a nonparametric two factor repeated measures GLM used to test for differences in percent cover and species richness between Years (August 2002 and 2003), Blocks (1-6) and Treatments (Natural, Opportunistic, CT wetlands) for the CT soil transfers in the Opportunistic and Natural Wetlands and reference plots in the CT wetland. Results of Dunnett's C Test to test for significant differences among treatments are also shown. Bolded effects are significant at ά=0.01.

		PERCENT COVER			
Source	*SS*	*df*	*H*	*Significance*	
Within Subjects Effects					
Year	0	1			
Year x Wetland	516.9	2	2.89	$0.20 < p < 0.30$	
Year x Block	1577	5	8.81	$0.10 < p < 0.20$	
Year x Block xWetland	3953	10	22.09	$0.01 < p < 0.02$	
Error	10058	72			
Between Subject Effects					
Wetland	**67575**	**2**	**61.40**	**<0.000**	
Block	1238	5	1.12	$0.90 < p < 0.95$	
Wetland x Block	6359	10	5.78	$0.80 < p < 0.90$	
Error	22772	72			
(I) Contrast	*(J) Contrast*	*Mean Difference (I-J)*	*Std. Error*	*99% Confidence Interval*	
				Lower Bound	*Upper Bound*
Opportunistic	**CT/ Control**	**25.78**	**3.39**	**15.08**	**36.47**
Natural	**CT/ Control**	**47.40**	**2.95**	**38.08**	**56.71**
Natural	**Opportunistic**	**25.77**	**3.39**	**9.50**	**33.75**
		SPECIES RICHNESS			
Source	*SS*	*df*	*H*	*Significance*	
Within Subjects Effects					
Year	0	1			
Year x Wetland	**1618**	**1**	**6.70**	**$0.01 < p < 0.001$**	
Year x Block	1255	5	5.20	$0.30 < p < 0.50$	
Year x Block x Wetland	5275	10	21.86	$0.01 < p < 0.02$	
Error	13567	72			
Between Subject Effects					
Wetland	**53721**	**2**	**56.15**	**< 0.000**	
Block	1128	5	1.17	$0.90 < p < 0.95$	
Wetland x Block	6734	10	7.05	$0.70 < p < 0.80$	
Error	23571	72			
(I) Contrast	*(J) Contrast*	*Mean Difference (I-J)*	*Std. Error*	*99% Confidence Interval*	
				Lower Bound	*Upper Bound*
Opportunistic	**CT**	**21.89**	**3.66**	**10.32**	**33.46**
Natural	**CT**	**42.31**	**2.87**	**33.35**	**51.36**
Natural	**Opportunistic**	**20.42**	**3.81**	**8.38**	**32.45**

Figure 2-1. Aerial photo of the CT, Opportunistic (OP) and Natural (NAT) wetlands within the Syncrude and Suncor leases in 2002. Photo courtesy of Syncrude Canada Ltd.

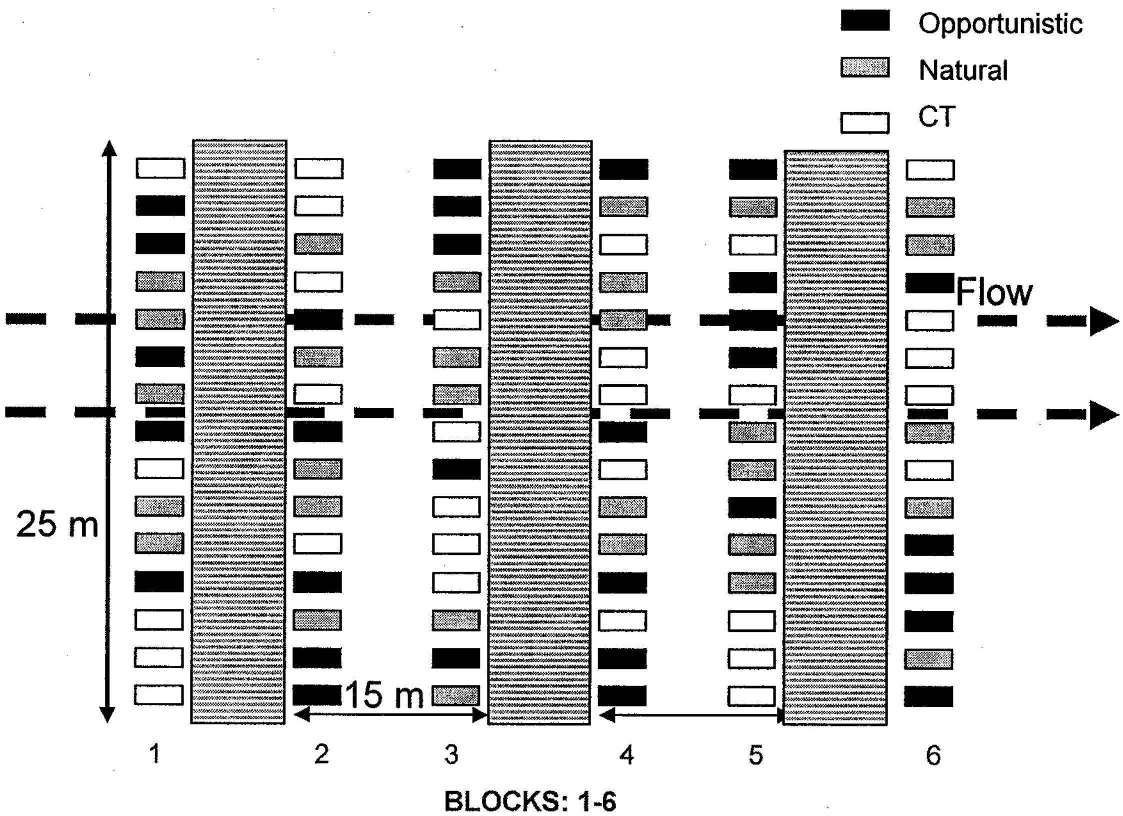

Figure 2-2. Experimental plot layout of the CT Wetland that included CT, Opportunistic and Natural wetland soil transfers.

Plot layouts were similar for both the Opportunistic and Natural wetlands, with the exception that each contained only two treatments (Opportunistic wetland: Opportunistic reference and CT soils; Natural wetland: Natural referfence and CT soils).

Water Depth in CT, Natural and Opportunistic Wetlands

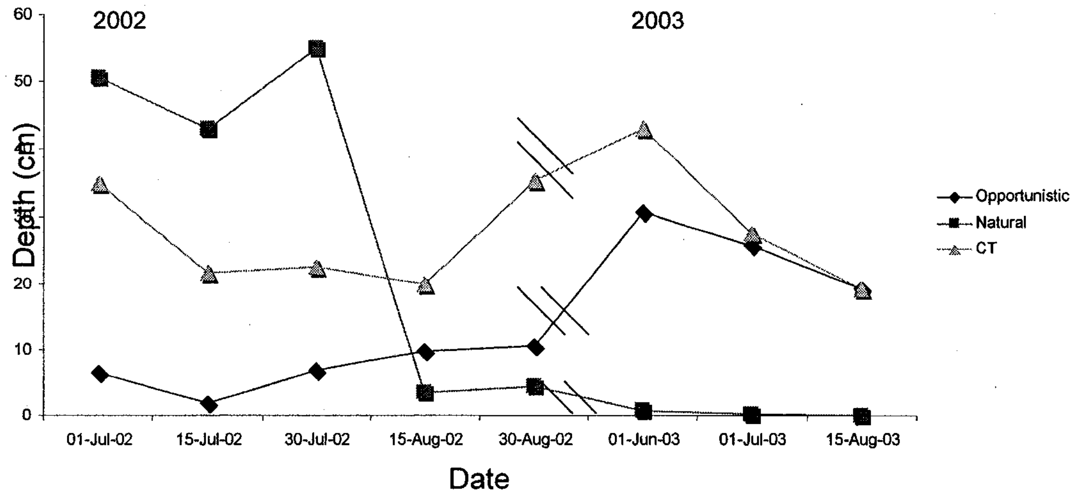

Figure 2-3. Mean water depth (cm) in each of the CT, Natural and Opportunistic wetlands over the course of the field seasons in 2002 and 2003.

Chapter 3: Assessing Vegetation Community Development in Reclaimed Northern Alberta Wetlands using Seed bank Analyses

3.1 Introduction

Oil Sands leaseholders are required to return mined lands to equivalent or better land capability compared with pre-disturbance, and these reclaimed lands must be continuous with surrounding leases (Oil Sands Wetlands Working Group 2000). Wetlands comprise a large percentage of the pre-disturbance landscape in the Fort McMurray region and thus will remain important landscape features post-disturbance. Post mining land uses for wetland occupied areas include: aboriginal traditional land uses, wildlife habitat and water treatment areas (Golder Associates 2000). In the past 20 years, various wetland reclamation projects have been undertaken, and techniques for vegetation establishment have focused on natural colonization, either through seed dispersal via waterfowl populations, wind, or from seed banks of salvaged soil material, though some plants were transplanted into select wetlands.

Consolidated/ composite tailings (CT) wetlands will be part of the post-mining landscape. Consolidated/ composite tailings are a gypsum-enriched (inorganic coagulant) saturated slurry of mature fine tails produced as a by-product of processed oil sands (MacKinnon et al 2001; Matthews et al. 2002)., that are held in a series of expansive holding ponds (an area of several km^2). Both produced waters and CT are slightly to moderately saline, high in total dissolved solids, and contain elevated ionic concentrations of sulphates, chlorides, sodium, as well as naphthenic acids and trace elements, compared to natural wetlands in the Fort McMurray region (Golder Associates 2000; 2002). CT deposits will likely be capped with a sand layer and thin soil layer (approximately 10- 20 cm of muskeg). Water levels will fluctuate with time, as the CT deposits begin to dewater. Based on known physical and chemical conditions present in CT deposits, and previous pilot studies by Golder Associates (2000; 2002), it was hypothesised that CT subsoils and water characteristics might inhibit species germination and emergence from imported soil seed banks, possibly resulting in reclaimed wetland plant communities that are dissimilar to those of natural or pre-mining wetlands.

Seed banks of macrophyte plant species are expected to differ between Natural, Reclaimed and Newly Constructed wetlands. Several studies have indicated significant differences in species richness and diversity between natural reference wetlands and restored or created wetlands (Galatowitsch and van der Valk 1996; Reinartz and Warne 1993; Seabloom and van der Valk 2003), and newly created and old ditch banks (Geertsema 1999). In all studies, differences in plant community compositions were attributed to lowered dispersal ability caused by increased distances connecting natural sites with restored sites. Dabbling ducks have been implicated as important contributors to wetland plant community development, at the local, regional and continental scales (Mueller and van der Valk 2002). Compared to Natural or older Reclaimed oil sands wetlands, isolated or wildlife-protected sites, such as Newly Constructed wetlands, may show differences in plant species richness, especially for species with limited persistence in seed banks. There is limited knowledge of specific reproduction strategies and corresponding moisture requirements for germination and emergence of aquatic emergent wetland plant species native to the Fort McMurray, Alberta region. It is also not understood whether the presence of CT as a secondary soil layer will influence wetland plant species emergence from the seed bank, possibly resulting in Reclaimed wetland plant communities that are dissimilar in both structure and function to Natural or pre-mining wetlands. This information is needed to help resolve the degree of inputs, management and monitoring necessary to produce a productive, diverse, and self-sustaining ecosystem that is continuous with the surrounding landscape.

The purpose of this study component was to assess seed bank similarity of Natural, Reclaimed and Newly Constructed wetlands located on Suncor and Syncrude oil sands leases. Furthermore, I examined the influence of CT subsoils on emergence of plants from seed banks. Specifically I addressed whether:

1) Emergence of wetland plant species from wetland soils was similar between Natural, Reclaimed and Newly Constructed wetlands in the Fort McMurray region.

a. I compared the similarity of seed banks for both species composition and relative abundance between: (1) Natural and Reclaimed wetlands,

(2) Natural and Newly Constructed wetlands, and (3) Reclaimed and Newly Constructed wetlands.

b. I used Principle Components Analysis to confirm observed differences in species composition and relative abundances of wetland plant species in seed banks of Natural, Reclaimed and Newly Constructed wetlands.

2) Composite/consolidated tailings as a secondary soil layer (subsoil) would affect emergence of wetland plant species.

a. I compared differences in emergence of wetland plant species (relative abundance and species composition) from wetland seed banks under two subsoil treatments (CT and potting soils).

3.2 Methods:

My approach was to determine the potential emergence of both submersed and emergent wetland plants from seed banks using a saturated and flooded water regime, and secondly under a CT subsoil treatment. The seed bank refers to "seeds, propagules, and other reproductive plant structures in the soil" (Poiani and Johnson 1988). All plants that emerged from the seed bank were counted and removed from the plots to prevent allelopathy or competition; hence survivability was not considered in this study.

3.2.1 Field Sampling

In late August 2002, 12 study wetlands, located on Syncrude and Suncor leases (Figure 3-1) were selected and categorized based on age, and disturbance regime (newly constructed, reclaimed and natural). Table 3-1 outlines the criteria used in selecting and categorizing each of the wetlands.

An initial reconnaissance survey was performed at each site (August 2002), which included mapping feasible sampling zones. Feasible sampling zones were the zones where most emergent and aquatic macrophyte plant species would be found (saturated soil, <20 cm of standing water). Five random subsampling points were selected within this zone. At each subsampling point, a 50 x 50 cm quadrat was placed. Vegetation was clipped to the soil surface, and the upper 5 cm of soil was removed using either a shovel or plastic PVC coring device. Samples were placed in labelled sealed plastic bags and then stored in a refrigerator temporarily until they were transported to the University. Once at the University, samples were stored outside in a shaded area, in large, sealed, opaque Rubbermaid™ bins for 5 months following similar procedures as Brown (1998). The soils were exposed to temperature regimes similar to field conditions; thus providing the conditions (temperature stratification) necessary to stimulate germination.

3.2.2 Experimental design and Seed bank analysis

The methods of the emergence study followed that of Brown (1998) and van der Valk et al. 1992), except that the samples were not sieved. Sieving removes many of the larger propagules, such as rhizomes or buds, which results in an underestimation of wetland plant species emergence from the soils, because many wetland plants regenerate vegetatively. In January 2003, the samples were thawed and the all of the subsamples taken at each site were composited. 250 mL of the composited material was placed on top of 500 mL of potting soil mix (Metro Mix 290-see Appendix D-1 for specifications), or CT material in 17 x 12 cm (6 cm deep) trays. Pans were randomly assigned to one of two depth treatments, either saturated soil or 4 cm of standing water. There were 5 replicates for each of the four treatments (flooded, saturated soil water regimes; CT subsoil and potting soil substrates), for a total of 20 pots per site (Figure 3-2).

A completely randomized design was not employed, because of the need to (1) separate flooded and non-flooded treatments, and (2) isolate CT subsoil pots, thereby reducing the potential for CT subsoil pots to contaminate potting subsoil pots. In the greenhouse, pots were stratified according to water treatment (flooded and saturated) (Figure 3-3) and flooded CT treatments. Four large trays were placed on the greenhouse bench to hold the pots (1 tray: CT substrate flooded, 1 tray: potting soil flooded and 2 trays: saturated soil pots), and the tray locations within the greenhouse were designated randomly. Pots were then randomly assigned a location within the trays. Two of the wetland sites contained CT materials (1 m CT and 4 m CT) and their flooded potting soil treatment pots were placed in separate trays at the end of the flooded potting soil treatment zone.

Pots were watered with tap water every 2 - 3 days to maintain either flooded or saturated conditions. Nutrients were not added, because it was thought that responses would likely be species-specific, and could therefore bias results. To prevent allelopathy or competition for light and nutrients, species were counted and removed once they could be identified. A variety of field guides were used for identification (MacKinnon et al. 1999; Moss 1992; Newmaster et al. 1997; Prescott 1969), and representatives of difficult taxa were allowed to grow until they reached the flowering life stage where identification was easier, and more accurate. Algae blooms were manually removed from the plots when they appeared. Greenhouse conditions were kept at 23°C with a 16-hour photoperiod. It was assumed that differences in light and air movement were insignificant, because of the small amount of space required for the seed bank trials in relation to the total size of the greenhouse chamber (12 m^2 vs. 200 m^2). The experiment ran from January 30th - April 30th, simulating the approximate length of the growing season in the Fort McMurray region.

3.2.3 Statistical analyses

All statistical tests were performed using SPSS version 11.5 (SPSS Inc. 1999). To assess the similarity of seed banks in terms of relative abundance and species composition, both Quantitative and Qualitative Sorensen's Indices were used (Sorensen 1949 as cited in Magurran 1988). Sorensen's Quantitative and Qualitative Indices are defined as:

> IS (quantitative)= 2c / (a +b), where a is the sum of all individuals (i.e., relative abundance) for one wetland group, b is the sum of all individuals for the second wetland group, and c is the sum of the lower value in either list:
> IS (qualitative) = 2c / (a +b), where a is the sum of all species (i.e., species richness) for one wetland group, b is the sum of all species s for the second wetland group, and c is the number of species common to both wetland groups.

Sorensen's Qualitative Index is "Qualitative", because the presence or absence of a species is only considered in the similarity formula; the contribution of that species to the total abundance of individuals within the seed bank is not accounted for. Sorensen's Indices, both quantitative and qualitative, were used because they are regarded as useful indices of similarity between species lists (Magurran 1982). The data obtained from the saturated and flooded water regime for the potting subsoil treatment were pooled for each site. Similarity coefficients were computed for: (1)

Natural vs. Reclaimed Wetlands, (2) Natural vs. Newly Constructed Wetlands, and (3) Reclaimed vs. Newly Constructed Wetlands. A total of 48 coefficients were computed ((4 wetland replicates x 4 wetland replicates) x 3 contrasts). A value of 1 indicated that the wetland groups were identical, while a value of 0 indicated that the wetland groups had no species or densities in common. Nonparametric Kruskall-Wallis ANOVAs were used to test for significant differences in similarity between wetland contrasts. Dunnett's C test was used for *post-hoc* multiple comparisons.

Principle Components Analysis (PCA) was used to reduce the potting subsoil treatment data set to fewer dimensions, thereby ordinating sites based on wetland plant species assemblages. CANOCO 4.0 was used to run the PCA. Again, the data obtained from the saturated and flooded water regimes were pooled for each site and used in the PCA. The PCA model was set to calculate distances between sites (inter-sample distances). Rare species were not down-weighted in this analysis, because rare species were removed from the data set prior to analysis as explained above. A species was considered rare if it was found in less than 3 sites. Species data were log transformed, centered and standardized prior to analysis. This allowed PCA to capture non-linear relationships and reduce the effect of dominant species on the ordination. The raw data matrix used in the PCA is shown in Table 3-2. Water chemistry data from Section 2.3.2 and previous research by Leonhardt (2003) were used to provide insight into observed differences in species composition and abundances for seed banks of Newly Constructed, Reclaimed and Natural wetlands (Table 3-3).

To determine whether emergence of plants from wetland soils was affected by the presence of CT as a secondary soil layer, the data obtained from the saturated and flooded water regime were pooled for the potting soil and the subsoil treatments for all sites. A Wilcoxon Signed Ranks paired nonparametric t-test was used to test for differences in species richness and relative abundance between CT and potting subsoil treatments (n=12) (Zar 1999). Principle Components Analysis was then used to group wetland sites based on their seed bank expression (species composition and abundances) for both CT and potting soil treatments, using the same procedures as described above. The raw data matrix used in the PCA is shown in Table 3-4.

3.3 Results

3.3.1 Similarity of seed bank composition and relative abundance between Natural, Reclaimed and Newly Constructed wetlands

Overall, seed banks of Natural, Reclaimed and Newly Constructed wetlands were more similar in relative abundances than species composition; Quantitative Sorensen's similarity coefficients computed higher than Qualitative Sorensen's similarity coefficients for (1) Natural: Reclaimed, (2) Natural: Newly Constructed and (3) Reclaimed: Newly Constructed wetland contrasts ((1): 0.62 vs. 0.39; (2): 0.21 vs. 0.14; (3) 0.37 vs. 0.21; Table 3-5). Both relative abundance and species composition were significantly different wetland seed bank contrasts (RELATIVE ABUNDANCE: $p= 0.001$, $df=2$, $N=3$; SPECIES COMPOSITION: $p< 0.0001$, $df=2$, $N=3$; Table 3-6). Contrasting seed bank similarities for Natural: Reclaimed Wetlands, compared to Natural: Newly Constructed Wetlands yielded significantly differences in both relative abundance and species composition (Table 3-6). Contrasting seed bank similarities for Natural: Reclaimed and Reclaimed: Newly Constructed wetlands yielded significant differences in species composition of wetland plants only (Table 3-6). Finally, there were no significant differences in wetland plant species composition or relative abundance for seed bank similarity contrasts of Newly Constructed: Natural and Newly Constructed: Reclaimed wetlands.

In the PCA ordination diagram, Natural wetlands (located in the upper left quadrant) are clearly distinguished from the other wetlands (Figure 3-4). The results indicate that seed banks of Natural wetlands are strongly associated with the presence of the following emergent macrophytes: *Chara spp., Typha latifolia, Carex spp., Sparganium angustifolium,* and *Juncus alpinus*. The seed bank of Peat Pond, a Newly Constructed wetland is similar to Bill's Lake, a Reclaimed wetland (Figure 3-4). These two wetlands are similar in that they both have high counts of *Poa palustris* and *Hordeum jubatum*, but Bill's Lake has a richer seed bank. Both 4 m CT and 1 m CT wetlands are grouped together (Figure 3-4). Seed banks of these Newly Constructed wetlands were dominated by *Juncus bufonius*, and *Scirpus validius* which are primary successional species. According to the PCA, neither Golden Pond nor Suncor Natural Wetland have seed banks that are similar to those of other sites. Suncor Natural Wetland is a Reclaimed site dominated by *Carex spp., Thalictrum venulosum* and *Juncus bufonius*. The seed bank for Golden Pond, a Newly Constructed wetland, was species poor, showing only one count each of *Typha latifolia* and *Calamagrostis inexpansa*. According to the PCA output, the first and second axes explained 28% and 24.6% of the variation in the data set, respectively (Table 3-7). The cumulative percent variance explained by all four axes is equal to 78.5%.

Surface water chemistry data indicated that several of Reclaimed and Newly Constructed wetlands had conductivity levels that were several orders of magnitude higher than what was observed in Natural wetlands (Figure 3-5). Surface waters of Reclaimed and Newly Constructed wetlands were typically dominated by Na_2SO_4 salts (Figure 3-6). Water chemical data is shown in Table 3-3.

3.3.2 Effect of CT subsoil on wetland plant species emergence

Consolidated/ composite tailings subsoil negatively affected wetland plant emergence from the seed bank soils (Figure 3-7). Both species richness and relative abundance were significantly lower for treatments receiving CT as a subsoil (SPECIES RICHNESS: $p= 0.009$, $n=12$; RELATIVE ABUNDANCE: $p= 0.019$, $n=12$; $\alpha= 0.05$). I observed the presence of salt crusting and visual signs of salt stress in many of the CT, saturated soil treatments.

The PCA ordination diagram confirmed differences in species abundance and richness between CT and potting subsoil treatments (Figure 3-8). The most notable sites where subsoil treatments did not overlap on the PCA ordination diagram (i.e., were not similar in number of plant taxa and the relative abundances of individual plant taxa) were: Shallow Wetland, S Pit, Natural Wetland, Bill's Pond, High Sulphate Wetland, and Golden Pond. Based on their absence in the CT subsoil treatment of the seed bank analyses, *Eleocharis palustris*, *Epilobium ciliatum*, and *Juncus spp.* (with the exception of *Juncus bufonius*) may be intolerant of the conditions associated with CT as a subsoil. According to the PCA output, the first and second axes explain 27%, and 53% of the variation in the data set, respectively (Table 3-8). The cumulative percent variance explained by all four axes is equal to 75%.

3.4 Discussion

3.4.1 Assessing seed bank similarity between natural, reclaimed and newly constructed wetlands

I found significant differences in similarity of both composition and relative abundance of species in the seed banks among the three wetland categories (Natural, Reclaimed and Newly Constructed). The seed banks were more similar in species relative abundances than species composition. Similar conclusions were drawn by Galatowitsch and van der Valk (1996) in their research examining vegetation communities and seed banks of restored and natural prairie wetlands. Seed banks of restored wetlands (≤ 3 years old, equivalent to Newly Constructed wetlands in this study) were less species rich and contained fewer seeds, compared to those of natural wetlands. In addition, species belonging to submersed aquatic, wet prairie and sedge meadow guilds, were not found in the seed banks of restored wetlands, but were found in the standing vegetation. In contrast, I found these guilds represented in the seed banks of Reclaimed wetlands, and absent from Newly Constructed wetlands.

There are many possible explanations of why reclaimed and restored wetlands do not structurally resemble plant communities of natural wetlands. The methods used by Galatowitsch and van der Valk (1996) may account for why some of the species in the submersed aquatic, wet prairie and sedge meadow guilds were not detected in the seed banks. In their emergence study, soils were sieved and tubers or other plant parts were removed, likely resulting in an underestimation of the potential of seed banks to represent vegetation communities. In contrast, in this study soils were not sieved, and I detected vegetatively reproducing species in the seed bank analyses (eg. *Typha latifolia, Carex aquatilis*). Connectivity with other wetlands is another factor that has been shown to increase the richness of wetland seed banks. Seed dispersal by surface waters, (Galatowitsch and van der Valk 1996; Leck 2003), wind, and waterfowl populations (Mueller and van der Valk 2002; Galatowitsch and van der valk 1996) contributes to wetland plant community development over the long-term. Many of the Reclaimed and Newly Constructed wetlands used in this study are located in small pockets of reclaimed lands that are isolated from larger water bodies or wetland complexes. However, field observations have shown increasing use of these wetlands by waterfowl populations. Therefore, it is likely waterfowl populations are responsible for the introduction of many of the submersed aquatic species found in the seed banks of Reclaimed wetlands.

Successional patterns of seed bank development in Reclaimed wetlands of oil sands mining regions and other wetland creation projects were expected to differ from those of restored prairie potholes that have undergone previous drainage and cultivation, because of differences in disturbance regimes. Drainage and subsequent cultivation of prairie pothole wetlands resulted in a loss of species from the seed banks through tillage displacement or dessication over time. Restoration of these systems typically involves restoring hydrology, and the longer the time since drainage, the more likely that remnant seeds in seed banks are no longer viable. Species with low dispersal ability (i.e., species that reproduce vegetatively and are not transported via wind or waterfowl) will likely be absent from the plant communities indefinitely (Seabloom and van der Valk 2003). The natural recovery approach adopted by oil sands reclamation practitioners is similar to most wetland creation projects in other regions where wetland soil seed banks are imported from similar wetland systems. This approach is advantageous in that it allows for many of the "less easily dispersed species" to be introduced to the wetland system. Assuming that connectivity with other wetland systems was similar between Reclaimed oil sands mining wetlands and restored prairie potholes, one might expect accelerated rates of succession with regards to seed bank and vegetation community development for reclaimed oil sands mining wetlands that receive imported seed banks.

The similarity indices and ordination diagrams indicate that replacement sequences for plant communities for Reclaimed oil sands wetlands may parallel those of Natural wetlands, in terms of seed bank species composition and abundances. Seed banks of Natural Wetlands were more similar to Reclaimed than Newly Constructed wetlands. However, most reclaimed oil sands mining wetlands will be affected by salinity, either through oil sands process material (OSPM), oil sands process waters (OSPW), saline overburden (subsoils), or seepage from tailings ponds. The 1 m CT and 4 m CT wetlands are affected by OSPM, Golden, Peat and Bill's pond were constructed on saline overburden, and High Sulphate wetlands, Suncor Natural wetland and S Pit are affected by OSPW (Table 3-1;

Figure 3-1). Previous studies in saline and brackish marsh ecosystems have identified salinity as a major factor contributing to low species richness (Galinato and van der Valk 1986; Hammer and Heseltine 1988). Interestingly, Hammer and Heseltine (1988) found that only *Potamogeton pectinatus, Ruppia maritima* and *Ruppia occidentalis* are found in great abundance in inland high salinity lakes. Based on this information, I speculate that salinity explains some of the observed differences in seed bank development between Natural, Reclaimed and Newly Constructed Wetlands. Further research isolating salinity and effects on wetland plant species recruitment is needed to verify or refute this hypothesis.

Time appears to be the biggest factor in the development of seed banks (Edwards and Proffitt 2003; Leck 2003; Reinartz and Warne 1993). Although waterfowl populations are likely helping to speed seed bank development in reclaimed oil sands wetlands, northern boreal ecosystems are characterised by short growing seasons and colder climates. Hence it is likely that seed bank and vegetation community development will take longer than what has been seen in warmer climates that experience longer growing seasons. Further research is needed to evaluate the role of time, climate, isolation from other wetland systems, and salinity on seed bank development in these reclaimed wetland systems.

3.4.2 Exploring the effects of CT subsoil on wetland plant species emergence

Oil sands process waters and materials are known to affect growth and viability of certain shrub and tree species (Renault et al. 1999; 1998). Based on previous pilot studies by Golder Associates (2000; 2002), it was hypothesised that CT subsoils might inhibit species germination and emergence from soil seed banks. CT subsoils significantly affected wetland plant species emergence from the seed bank. Both species composition and relative abundance were significantly lower for CT, compared to control treatments.

Based on observations of salt crusts and seedling stress in CT treatments, I speculate that lower emergence rates can be explained by the elevated concentrations of salts in the CT materials. Salt marsh vegetation is relatively low in species richness, often dominated by vegetative colonizers (Parker and Leck 1985). This is also reflected in the seed banks. It is likely that certain perennial species such as *Juncus* and other wet meadow species absent in the CT subsoil seed bank treatments are intolerant to slightly to moderately saline conditions. However, further research is needed to isolate the effects of salinity from other parameters such as naphthenic acids and other extractable hydrocarbons present in CT that might negatively impact seed bank emergence.

Based on the results of this study, I can not clearly predict how vegetation communities might develop on large scale CT deposits. It should be noted that the CT subsoil treatment used in this study reflects a worst-case scenario in the post-mining wetland CT landscape. CT deposits will be capped with a layer of sand (~1 m) and a thicker layer of soil than what was used in this experiment (Golder Associates 2000). Furthermore, a fresh water source, rather than CT waters was used to saturate and flood the seed bank treatments. CT waters are slightly to moderately saline (elevated concentrations of sulphate, chloride and sodium), with higher concentrations of total dissolved solids, naphthenic acids and metals, compared to natural wetlands in the Fort McMurray region. Future seed bank and vegetation field and laboratory research should attempt to simulate a range of post-mining conditions.

3.4.3 Summary

Overall, seed banks of Natural, Reclaimed, and Newly Constructed wetlands were more similar in species relative abundances than species composition. Seed banks of Natural wetlands were more similar to Reclaimed than Newly Constructed wetlands. CT subsoils significantly reduced wetland plant species emergence from seed banks in both species composition and relative abundance. The results indicate that species replacement sequences for plant communities of Reclaimed oil sands wetlands may parallel those of Natural wetlands. Further monitoring is needed to evaluate the role of salinity (surface waters and/or subsoils), isolation from other wetlands, and northern climates on seed bank development in Reclaimed oils sands wetland systems.

3.4.4 Study Limitations

Mitchell et al. (1998) noted many limitations that often plague seed bank experiments such as this one, including insufficient replication, and sample size.

Efforts were made to sample a sufficiently large area at each wetland, and thoroughly mix composite soils, but it is possible that some species present in either seed banks or vegetation may have been missed or failed to germinate. Nonetheless, this study provides a reasonable estimate of generic seed bank and vegetation species composition and abundances for the 12 sites examined.

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Table 3-1. Criteria used in selecting and categorizing wetlands used for seed bank analyses

Category	Wetlands	Characteristics
Newly Constructed	Peat Pond (PP) Golden Pond (GP) 1m CT (1CT) 4m CT (4CT)	<4 years old Vegetation establishment via ➢ natural colonisation (4CT) ➢ topsoil placement (PP, GP, 1CT) Located on a post-mining site or influenced by mining sites ➢ Affected by Oil Sands Process Water (OSPW) (1 CT, 4CT) ➢ Ground-water seepage from tailings ponds (1CT, 4CT) ➢ Formed on Oil Sands Process Materials (OSPM) (1CT; 4CT) ➢ Formed on saline overburden (PP, GP)
Reclaimed	Bill's Lake (BL) S Pit (SP) High Sulphate Wetland (HSW) Suncor Natural Wetland (NW)	Vegetation establishment by topsoil placement Located on a post-mining site or influenced by mining sites ➢ Affected by Oil Sands Process Water (OSPW) (NW) ➢ Ground-water seepage from tailings ponds (HSW, NW, SP) ➢ Formed on saline overburden (HSW, BL)
Natural	Highway 63 Wetland (HW63) Loon Lake (LL) McLean Creek (MC) Shallow Wetland (SW)	➢ Well established vegetation community ➢ Disturbance impacts minimal

Table 3-2. Raw data matrix used in the PCA to ordinate wetland sites against wetland plant species that emerged from seed banks under potting subsoil treatments.

(Newly Constructed wetlands= 1CT (1m CT), 4CT, (4m CT), GP (Golden Pond), PP (Peat Pond); Natural wetlands= HWY63 (Highway 63 wetland), LL (Loon Lake), MC (McLean Creek), SW (Shallow Wetland); Reclaimed wetlands= NW (Suncor Natural Wetlands), SP (S Pit), BP (Bill's Pond), HSW (High Sulphate Wetland))

Species	1CT	4CT	GP	PP	HY63	LL	MC	SW	NW	SP	BP	HSW
Care spp.	0	0	0	0	10	82	125	3	13	0	4	5
Pota pec.	0	0	0	0	9	1	0	1	0	4	15	0
Spar ang	0	0	0	0	11	11	0	43	0	1	11	1
Spar eur	0	0	0	0	5	0	0	0	0	0	6	2
Poa pal	0	0	0	21	17	0	44	27	0	0	84	0
Pota pus	0	0	0	0	11	5	0	37	0	0	30	0
Cala can	0	0	0	0	0	0	0	3	1	0	0	16
Cala ine	0	0	1	0	0	2	0	0	0	0	2	5
Junc buf	152	8	0	0	6	0	0	0	8	65	0	4
Junc spp.	0	0	0	0	17	6	3	0	0	0	0	28
Typh lat	0	0	1	0	10	19	5	40	4	0	40	18
Scir val	4	4	0	0	3	0	0	0	5	1	0	25
Char spp.	0	0	0	0	316	88	9	478	0	226	0	42
Hord jub	0	0	0	6	0	1	0	0	3	0	28	0
Thal ven	0	0	0	0	0	0	0	4	7	0	12	0
Desc cae	0	0	0	0	0	0	1	0	0	8	0	23

Table 3-3. Detailed water and sediment chemistry values for wetlands used in the seed bank analyses.

	1CT	4CT	GP	[1]PP	HW63	[1]LL	SW	MC	[1]NW	[1]SP	[1]BP	[1]HSW
Class[2]	NC	NC	NC	NC	Nat	Nat	Nat	Nat	Rec	Rec	Rec	Rec
Age in 2002	3	3	0	1	24	29	9	?	16	27	5	18
Sample Date	0208	0208	0306	0107	0107	0007	0208	0208	0107	0007	0107	0107
pH	8.12	8.17	7.25	7.01	8.3	9.1	9.38	7.22	8.39	10.1	7.86	7.92
Cond (us/ cm)	1580	1827	790	3750	675	452	776	189	1779	1480	768	1517
NH_3			BDL	BDL	BDL				0.30		BDL	BDL
$N(NH_4)$		5.1	BDL	BDL	BDL	<0.01	BDL	BDL	0.20		BDL	BDL
Na	363	397	52.8	684.0	72.60	56.90	138	5.6	470.0	321.0	87.10	248.0
K	11.9	15	6.6	11.50	BDL	2.30	BDL	BDL	14.90	3.70	9.85	13.60
Mg	28.2	26.1	27.8	114.0	38.40	16.20	24	7.85	14.70	34.40	31.00	69.90
Ca	42.3	47.3	88.5	125.0	41.60	25.40	20.7	29.1	24.70	8.83	73.00	97.70
F		BDL	0.6	BDL	BDL	BDL	BDL	BDL	2.30	BDL	0.15	BDL
Cl	61	70	9.7	80.00	94.00	51.00	35	24	65.00	220.0	6.20	BDL
SO_4	340	327	354	2800	33.00	70.00	197	3.1	210.00	290.0	240	880.0
CO_3	0.00	0.00	0	0.00	17.40		16.8	0	37.80		0.00	0.00
HCO_3	769.0	804	101	53.70	261.0	137.73	187	143	914.00	312.36	281.0	160.0
NO_2	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
NO_3	BDL	70	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
PO_4		BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Al	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
B	2.22	2.51	0.12	0.47	0.128	0.18	BDL	BDL	2.93	0.55	0.29	0.80
Ba	0.049	0.062	0.05	0.08	BDL	BDL	BDL	BDL	0.09	BDL	BDL	0.01
Cd	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Co	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Cr	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Cu	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Fe	BDL	BDL	BDL	1.33	BDL	BDL	BDL	1.09	0.31	BDL	BDL	0.23
Li	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Mn	BDL	BDL	BDL	2.13	BDL	BDL	BDL	BDL	0.04	BDL	BDL	0.10
Mo	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	0.28	BDL	BDL	BDL
Ni	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Pb	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
S	113	115	114	889.0	13.10		71.1	BDL	81.90		88.70	347.0
Sb	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Se	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Si	3.43	3.57	BDL	2.51	0.83	1.12	BDL	5.19	5.35	1.51	BDL	1.77
Sr	0.72	0.759	0.42	2.09	0.25	0.27	0.205	0.103	0.53	0.03	0.77	1.41
Ti	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	0.03	BDL	BDL	BDL
V	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Zn	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Zr	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Salinity				3.00	0.40	0.00			1.20	1.31	0.30	1.70
Sed. pH	7.54	8.04		8.30	8.30	7.40	7.36		8.10	7.90	9.60	7.90
Sed ORP				71	-179	-110			-174.0	-110	-107	-309
Sed Organic Content (%) LOI				9.05	6.20	1.81			5.30	4.59	4.55	18.5

[1]Data provided by Leonhardt (2003) [2]Class: NC= Newly Constructed, Nat= Natural, Rec= Reclaimed

Table 3-4. Raw data matrix used in the PCA to ordinate wetland sites against wetland plant species that emerged from seed banks under the seed bank treatments (CT and potting subsoils) against emergent and submergent wetland plant species.

(Newly Constructed wetlands= 1CT (1m CT), 4CT, (4m CT), GP (Golden Pond), PP (Peat Pond); Natural wetlands= HWY63 (Highway 63 wetland), LL (Loon Lake), MC (McLean Creek), SW (Shallow Wetland); Reclaimed wetlands= NW (Suncor Natural Wetlands), SP (S Pit), BP (Bill's Pond), HSW (High Sulphate Wetland))

Potting subsoil	Care spp.	Pota pec	Spar ang	Spar eur	Poa pal	Pota pus	Cala can	Cala ine	Junc buf	Junc spp.	Typ h lat	Scir val	Char spp.	Hord jub	Thal ven	Desc caes
1CT	0	0	0	0	0	0	0	0	152	0	0	4	0	0	0	0
4CT	0	0	0	0	0	0	0	0	8	0	0	4	0	0	0	0
GP	0	0	0	0	0	0	0	1	0	0	1	0	0	0	0	0
PP	0	0	0	0	21	0	0	0	0	0	0	0	0	6	0	0
HW63	10	9	11	5	17	11	0	0	6	17	10	3	316	0	0	0
LL	82	1	11	0	0	5	0	2	0	6	19	0	88	1	0	0
MC	125	0	0	0	44	0	0	0	0	3	5	0	9	0	0	1
SW	3	1	43	0	27	37	3	0	0	0	40	0	478	0	4	0
NW	13	0	0	0	0	0	1	0	8	0	4	5	0	3	7	0
SP	0	4	1	0	0	0	0	0	65	0	0	1	226	0	0	8
BP	4	15	11	6	84	30	0	2	0	0	40	0	0	28	12	0
HSW	5	0	1	2	0	0	16	5	4	28	18	25	42	0	0	23
CT subsoil																
1CT	0	0	0	0	0	0	0	0	85	0	1	4	0	0	0	0
4CT	0	0	0	0	0	0	0	0	5	0	0	5	0	0	0	0
GP	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PP	0	0	0	0	6	0	0	0	0	0	0	0	0	0	0	0
HW63	0	1	5	0	0	1	0	0	6	9	17	0	141	0	0	0
LL	23	0	3	0	0	4	0	0	0	1	10	0	62	0	0	0
MC	44	0	0	0	0	24	0	0	0	0	7	0	0	0	0	16
SW	0	0	36	7	0	35	0	0	5	0	45	0	388	0	0	0
NW	2	0	0	0	5	0	3	0	2	0	2	2	0	1	0	0
SP	0	1	0	0	0	3	0	0	8	5	0	0	304	0	0	0
BP	0	4	20	2	69	27	0	17	0	1	47	0	9	88	4	0
HSW	0	0	0	2	0	0	0	3	0	7	28	11	8	0	0	18

Table 3-5. Qualitative and quantitative Sorensen's similarity coefficients for seed banks of Natural, Reclaimed and Newly Constructed Wetlands

(I) Contrast	N	Minimum	Maximum	SI	Std. Error
Sorensen's quantitative coefficient					
Natural: Reclaimed	16	0.13	0.95	**0.62**	0.07
Natural: Newly Constructed	16	0.01	0.81	**0.21**	0.06
Reclaimed: Newly Constructed	16	0.01	0.88	**0.37**	0.08
Sorensen's qualitative coefficient					
Natural: Reclaimed	16	0.10	0.58	**0.39**	0.03
Natural: Newly Constructed	16	0.00	0.47	**0.14**	0.03
Reclaimed: Newly Constructed	16	0.00	0.50	**0.21**	0.04

Table 3-6. Results of multiple contrasts of computed similarity coefficients for seed banks of Natural, Reclaimed and Newly Constructed Wetlands using Dunnett's C test. Bolded contrasts are significant at ά=0.05.

(I) Contrast	(J) Contrast	Mean Difference (I-J)	Std. Error	95% Confidence Interval Lower Bound	Upper Bound
Sorensen's Quantitative (Kruskall Wallis: p= 0.001, df=2, N=3, ά=0.05)					
Natural: Reclaimed	**Natural: Newly Constructed**	**0.41**	**0.09**	**0.17**	**0.66**
Natural: Reclaimed	Reclaimed: Newly Constructed	0.26	0.11	0.02	0.54
Reclaimed: Newly Constructed	Natural: Newly Constructed	0.16	0.10	0.12	0.43
Sorensen's Qualitative (Kruskall Wallis: p<0.0001, df=2, N=3, ά=0.05)					
Natural: Reclaimed	**Natural: Newly Constructed**	**0.25**	**0.05**	**0.13**	**0.37**
Natural: Reclaimed	**Reclaimed: Newly Constructed**	**0.17**	**0.05**	**0.03**	**0.32**
Reclaimed: Newly Constructed	Natural: Newly Constructed	0.08	0.06	0.07	0.22

Table 3-7. Standardized PCA summary of wetland sites in wetland plant species-space

Axes	**1**	**2**	**3**	**4**	**Total Variance**
Eigenvalues	0.280	0.246	0.146	0.1141	1
Cumulative % variance (species)	28.0	52.6	67.2	78.5	
Sum of all unconstrained Eigenvalues=	1				

Table 3-8. Standardized PCA summary for wetland sites according to treatment (CT and potting subsoils) in wetland plant species-space

Axes	**1**	**2**	**3**	**4**	**Total Variance**
Eigenvalues	0.267	0.244	0.139	0.1011	1
Cumulative % variance (species)	26.7	52.6	67.2	75	
Sum of all unconstrained Eigenvalues=	1				

Figure 3-1. Aerial photo of the wetlands sampled for seed bank analysis.

(1= 1m CT, 2= 4m CT, 3= Bill's Lake, 4= Golden Pond 5= High Sulphate Wetland, 6= Highway 63 wetland, 7= Loon Lake, 8=McLean Creek, 9= Suncor Natural Wetland, 10= Peat Pond, 11= Shallow Wetland, 12= S pit). Photo courtesy of Syncrude Canada Ltd.

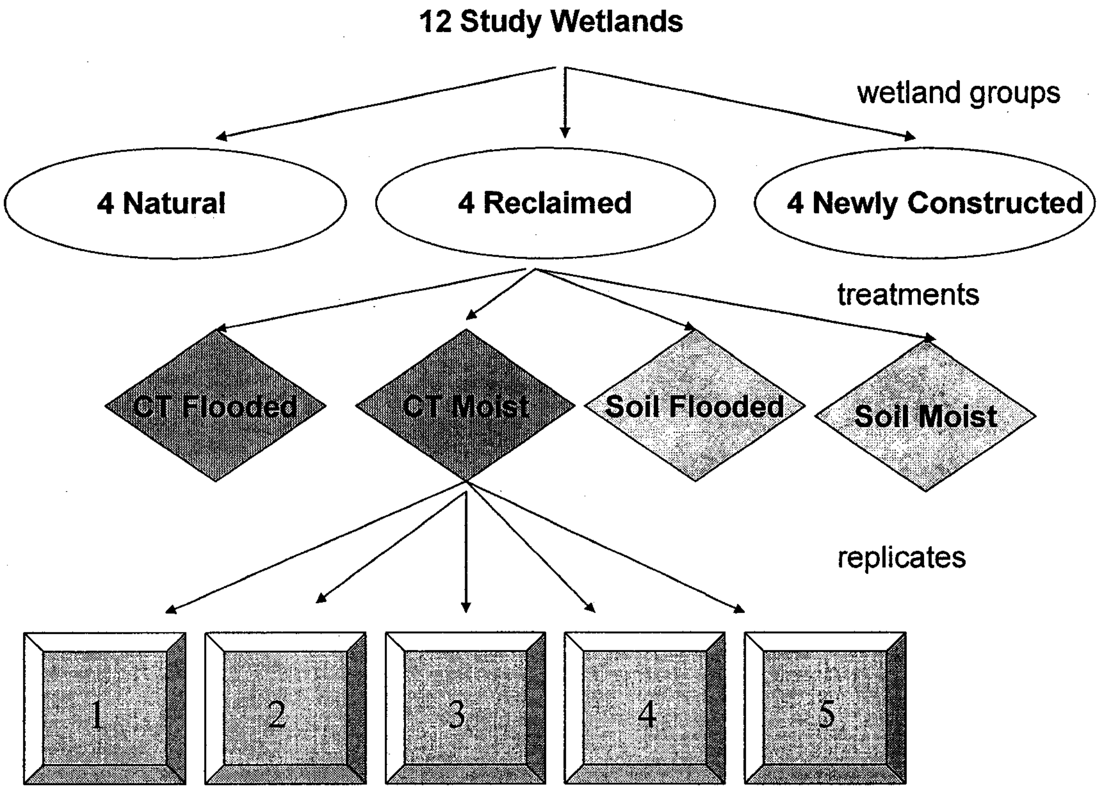

Figure 3-2. Schematic diagram showing the experimental design of the seed bank analyses.

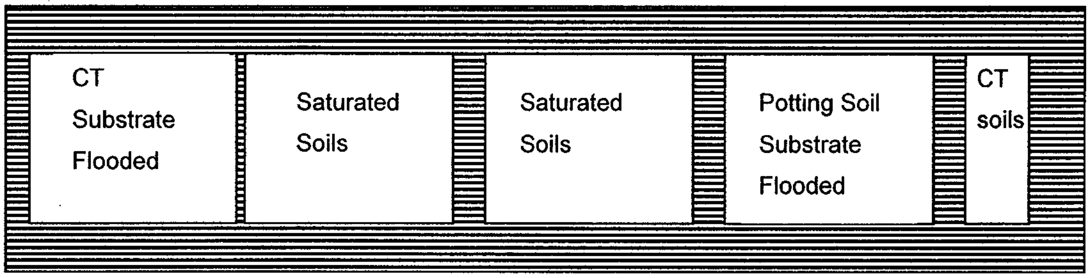

Figure 3-3. Schematic diagram of stratification of pots in the greenhouse according to substrates and water regime.

Each stratification was randomly assigned a position on the bench and pots were randomly assigned a designation within each stratification.

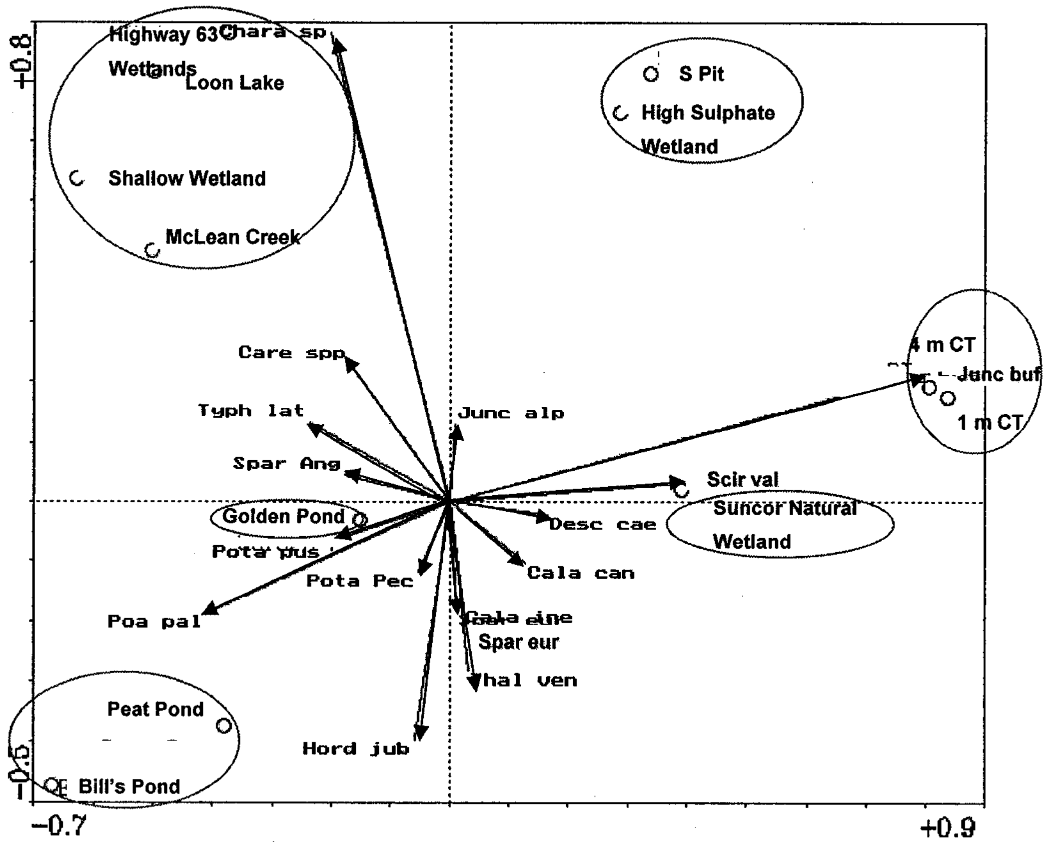

Figure 3-4. PCA biplot showing wetland sites ordinated in wetland plant species-space.

Data was logtransformed and species were centered and standardized prior to analysis. (*Newly Constructed wetlands*= 1m CT, 4m CT, Golden Pond, Peat Pond; *Natural wetlands*= HWY63, Loon Lake, McLean Creek, Shallow Wetland; *Reclaimed wetlands*= Natural Wetland (Suncor), S Pit, Bill's Pond, High Sulphate Wetland)

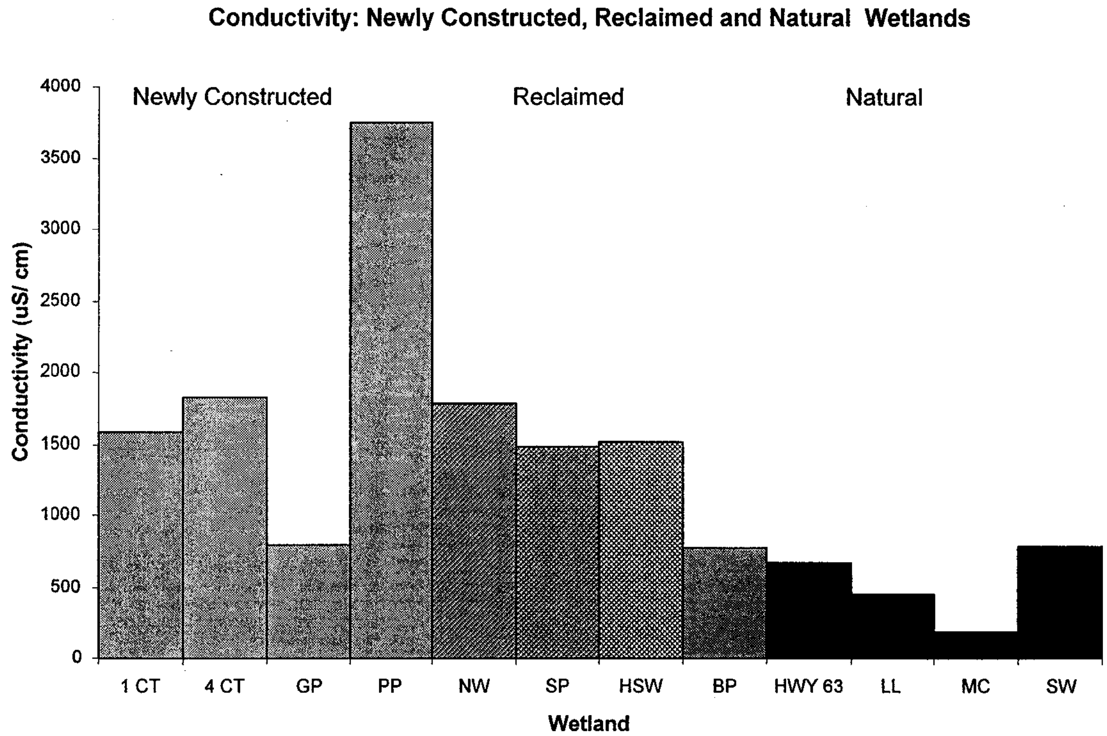

Figure 3-5. Conductivity of surface waters (uS/cm) in Newly Constructed, Reclaimed and Natural wetlands.

Data for PP, NW, SP, HSW, BP, HWY 63, LL was from Leonhardt (2003) (Appendix C-2). Standard error bars are not presented because values are based on single data points. (Newly Constructed wetlands= 1CT (1m CT), 4CT, (4m CT), GP (Golden Pond), PP (Peat Pond); Natural wetlands= HWY63 (Highway 63 wetland), LL (Loon Lake), MC (McLean Creek), SW (Shallow Wetland); Reclaimed wetlands= NW (Suncor Natural Wetlands), SP (S Pit), BP (Bill's Pond), HSW (High Sulphate Wetland))

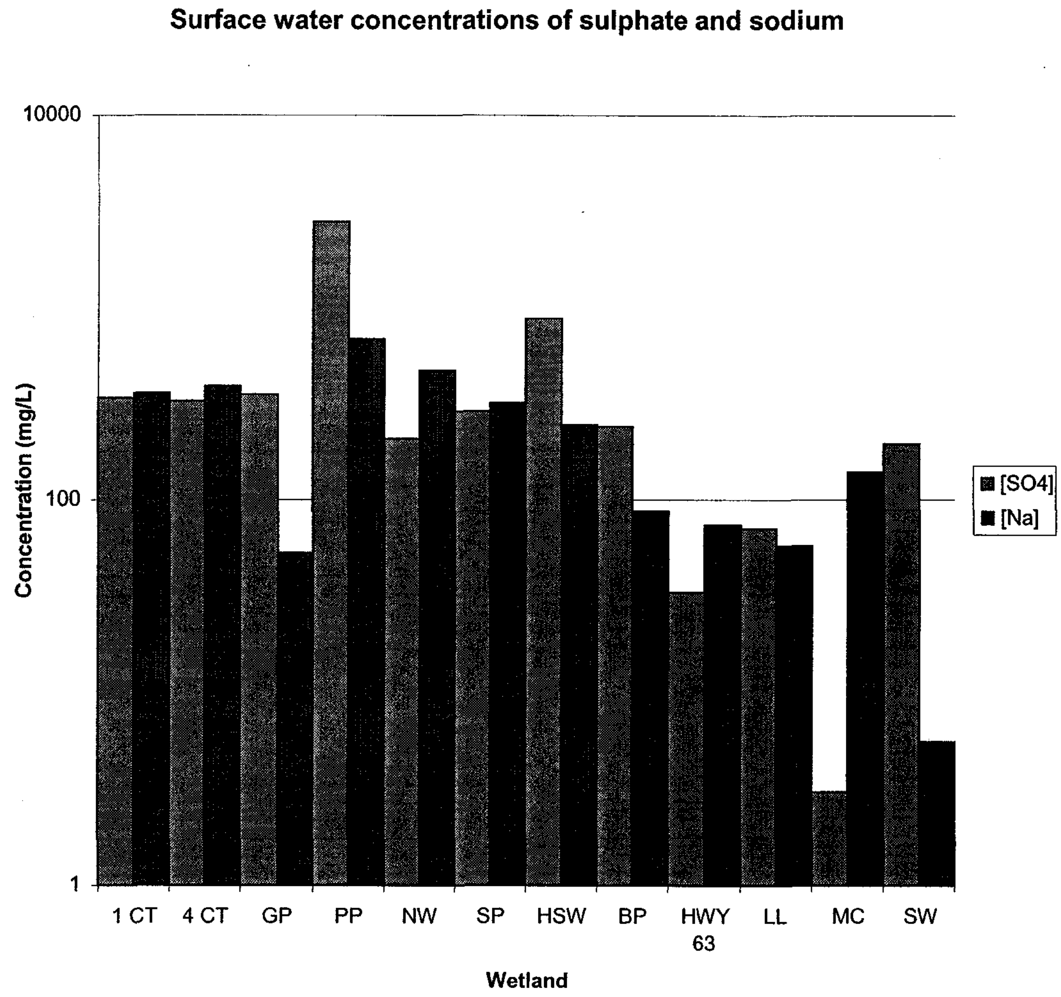

Figure 3-6. Sulphate and sodium concentrations (mg/ L) in surface waters.

Data for PP, NW, SP, HSW, BP, HWY 63, LL was from Leonhardt (2003). Values are plotted on a logarithmic scale. Standard error bards are not presented because values are based on a single data point. (Newly Constructed wetlands= 1CT (1m CT), 4CT, (4m CT), GP (Golden Pond), PP (Peat Pond); Natural wetlands= HWY63 (Highway 63 wetland), LL (Loon Lake), MC (McLean Creek), SW (Shallow Wetland); Reclaimed wetlands= NW (Suncor Natural Wetlands), SP (S Pit), BP (Bill's Pond), HSW (High Sulphate Wetland))

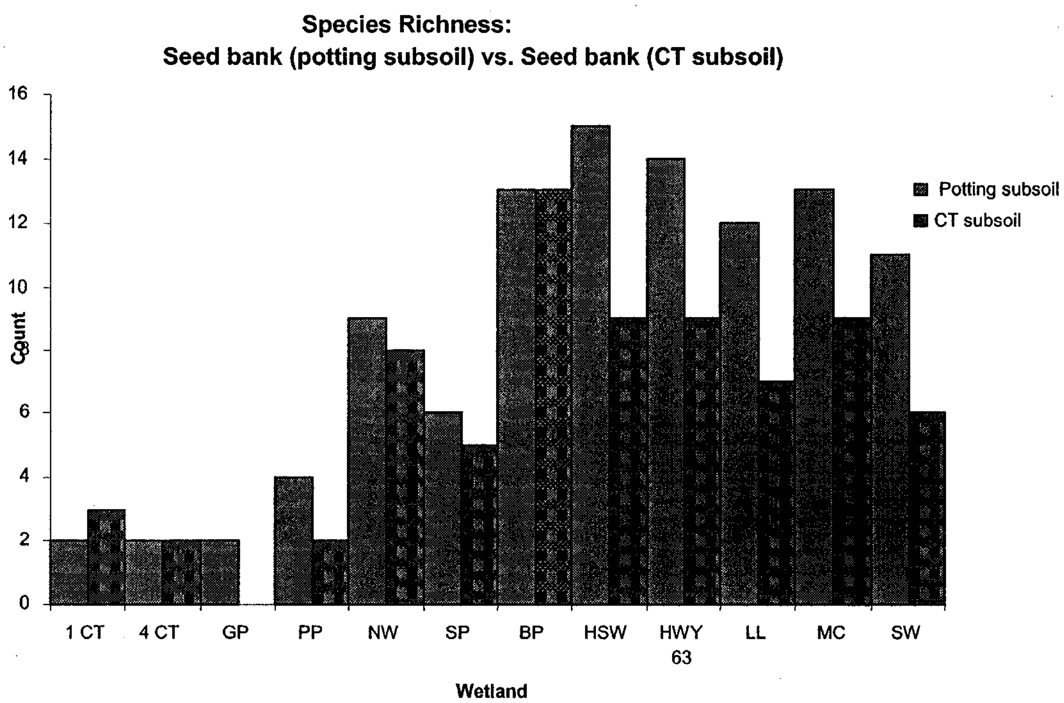

Figure 3-7. Species richness values for potting subsoil seed bank treatments compared to CT subsoil seed bank treatments.

(Newly Constructed wetlands= 1CT (1m CT), 4CT, (4m CT), GP (Golden Pond), PP (Peat Pond); Natural wetlands= HWY63 (Highway 63 wetland), LL (Loon Lake), MC (McLean Creek), SW (Shallow Wetland); Reclaimed wetlands= NW (Suncor Natural Wetlands), SP (S Pit), BP (Bill's Pond), HSW (High Sulphate Wetland))

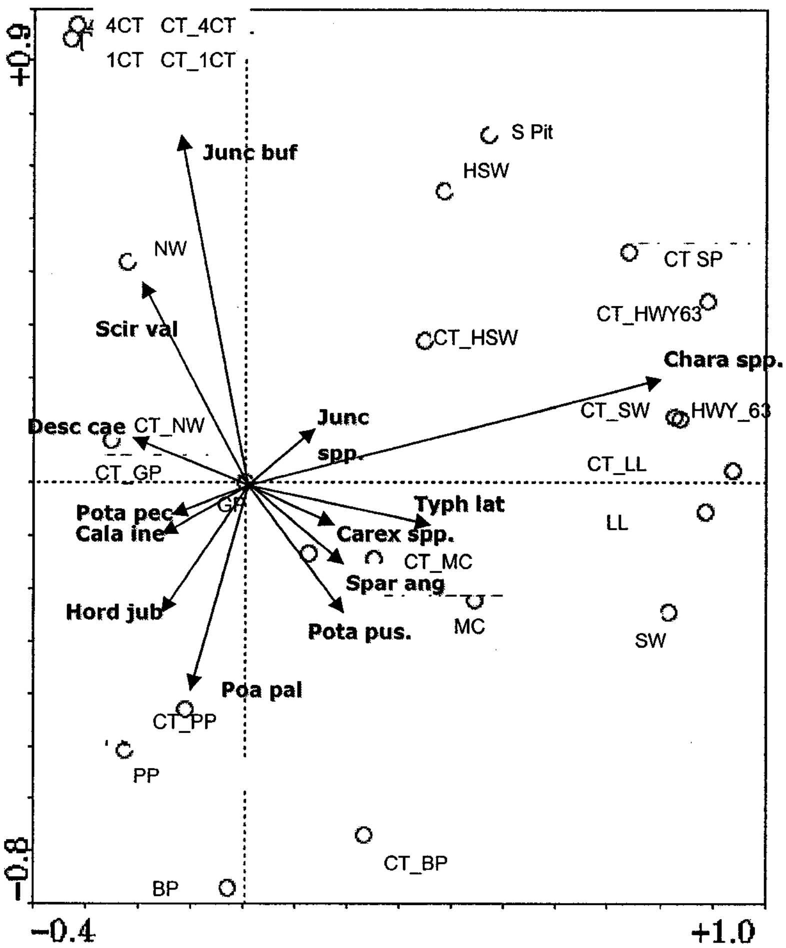

Figure 3-8. PCA biplot showing wetland sites, treatments (CT and potting subsoils) ordinated in wetland plant species-space.

Data was log-transformed and species were centered and standardized prior to analysis. Newly Constructed wetlands= 1CT (1m CT), 4CT, (4m CT), GP (Golden Pond), PP (Peat Pond); Natural wetlands= HWY63 (Highway 63 wetland), LL (Loon Lake), MC (McLean Creek), SW (Shallow Wetland); Reclaimed wetlands= NW (Suncor Natural Wetlands), SP (S Pit), BP (Bill's Pond), HSW (High Sulphate Wetland). CT treatments are indicated by CT_ Wetland Site.

Chapter 4: Evaluating the use of Seed bank Analyses to Assess Vegetation Community Development in Reclaimed Northern Alberta Wetlands

4.1 Introduction

Oil sands mining continues to increase in the Northeastern Boreal Region of Alberta, yet basic understanding of the structure and function of reclaimed wetland systems, compared with pre-disturbed wetland systems remains unknown. Reclamation advisory groups, industrial operators and stakeholders are interested in developing standard methods to evaluate wetland reclamation success. A variety of wetland types will be a part of the post-mining landscape, including wetlands formed on consolidated/ composite tailings (CT) deposits. The CT deposits are a gypsum-enriched (inorganic coagulant) saturated slurry of mature fine tails produced as a by-product of processed oil sands (MacKinnon et al 2001; Matthews et al. 2002). Both produced waters and CT are slightly to moderately saline, high in total dissolved solids, and contain elevated ionic concentrations of sulphates, chlorides, sodium, as well as naphthenic acids and trace elements. Backfilled mine cells hold large volumes of CT, and it is expected that wetlands will be constructed on a large proportion of CT deposits. These constructed wetlands are predicted to develop into shallow, highly productive water habitats, connected with the surface drainage for that area (Golder Associates 2000). Previous studies have indicated that many of the CT constituents can negatively affect terrestrial plant performance (plant recruitment, growth, development and reproduction), (Apostal 2002; Renault et al. 1998; 1999), but it is unknown how wetland plants will respond to the physical and chemical CT environment. Key questions include (1) how similar are reclaimed/created and natural wetlands on the landscape over time, and (2) are there indicator species or groups of species that can be used as evaluators of reclamation success?

Various metrics have been developed over time to evaluate structure and function of reclaimed/created wetlands. These metrics have included analysing bird use, (Brawley et al. 1998); soil development (Bishel-Machung et al. 1996; Nair et al. 2001; Stolt et al. 2000), benthic invertebrate communities (Leonhardt 2003); productivity (McKenna 2003) and vegetation community development (Galatowitsch and van der Valk 1996; Seabloom and van der Valk 2003). New research by

Leonhardt (2003) has indicated that reclaimed wetlands (built on oil sands process materials) differ in zoobenthic invertebrate species composition, compared to natural wetlands of similar age, but the mechanisms responsible for observed differences are not well understood.

Aquatic plant (macrophyte) community development is a commonly used benchmark to which equivalency or functionality is compared between created or restored wetlands and natural or pre-existing wetlands (Edwards and Proffitt 2003; Galatowitsch and van der Valk 1996; Leck 2003; Seabloom and van der Valk 2003). van der Valk and Pederson (1989) proposed that species composition at newly created wetlands could be predicted from the vegetation composition and environmental moisture regime of the site from which the soil was imported. It is unclear as to whether or not this theory holds true for all wetlands. Haukos and Smith (1993; 1998) determined that this model became less reliable as environmental variability increased. In a wetland setting, species assemblages are controlled by a variety of chemical, physical and biological factors, including: hydrologic regimes, (Baldwin et al. 2001; Gerritsen and Greening 1989; Poiani and Carter Johnson 1998; Schneider 1994; Shay 1999; van der Valk et al. 1999; van der Valk and Pederson 1989; Vosenek and Blom 1992; Welling et al. 1988), species- specific reproduction strategies (Spencer 1987), seed persistence (Hutchings and Russel 1989; Schopp-Guth et al.1994; Schutz 1998), inputs from the local seed rain (Schneider and Sharitz 1986; Titus 1991), depth of soil disturbance (McGee and Feller 1993), salinity in porewaters (Sanderson et al. 1997; Galinato and van der Valk 1986), and anoxia (Bart and Hartman 2002; Bandyopadhyay et al. 1993). Hence, predicting wetland plant community composition in created wetlands based on the standing vegetation of the site where the soil was imported from is a very difficult and somewhat questionable method.

Seed bank analysis, using the emergence method, has been used to assess vegetation history and predict vegetation community development over time (Haukos and Smith 1993; 2001; Leck 2003; Rossell and Wells 1999; Poiani and Johnson 1988; van der Valk and Davis 1976; van der Valk et al. 1992; Welling et al. 1988; Wetzel et al. 2001). Soil is collected from wetlands and placed under controlled growing conditions (greenhouse environment), under different treatments, and the emerging seedlings are identified and counted. Treatments commonly tested include water levels (flooding/ non-flooding) and substrates. The emergence method is not without its biases. Greenhouse conditions often do not accurately simulate light, temperature, water and nutrient regimes occurring in wetland settings. Secondly, dormancy cycles for some species can be broken either by the physical disturbance of the soil during sample collection and/or storage. The result can be an under or overestimation of the potential of the seed bank to accurately represent the floristic composition of a site. The alternative to the emergence method involves manual separation of the seeds from the soil and then either identifying or germinating the seeds. This results in a potentially biased estimate of relative abundance and species composition, because not all seeds within a seed bank are viable (Hutchings and Russell 1989; van der Valk et al. 1992). In addition, sieving removes propagules, rhizomes, tubers and other vegetative parts of plants known to produce new seedlings. Mechanical seed separation and identification is highly time consumptive and requires specialised skills for accurate identification to the species taxonomic level. The emergence method is by far most commonly used because of its simplicity and reliability to produce an estimate of relative species richness and abundance.

The purpose of this chapter was to examine the potential of using seed bank analyses to assess macrophyte community development in Reclaimed and Newly Constructed wetlands in the Fort McMurray oil sands mining area. To address this objective, I used data from a manipulative field study (Chapter 2) and a greenhouse study (Chapter 3). Specifically I addressed the following questions below. Regardless of factors that might influence the development of wetland vegetation communities and seed banks overtime (i.e., age, material that the wetland is constructed on (subsoil), surface and pore water chemistry):

(1) *Do seed bank analyses accurately reflect the standing vegetation at a given wetland site?*

(2) Can seed bank analyses and/ or vegetation surveys be used to predict the initial establishment of vegetation at a created wetland site?

To address the first question, I compared wetland plant species composition and richness values obtained from: controlled seed bank analyses, conducted in greenhouses (using a potting subsoil treatment and two water treatments: saturated and flooded), against vegetation field surveys conducted prior to sampling for the seed bank analyses. To address the second question, I compared this data against that of a manipulative study where the emergence of plant species from soil seed banks under natural wetland conditions was assessed (i.e., vegetation was clipped to the soil surface and soils were placed in buckets level with the soil surface in the Natural (McLean Creek) and Opportunistic (Shallow wetland) wetlands).

I further assessed the accuracy of seed bank analyses to reflect vegetation community development by accounting for subsoil (consolidated/ composite tailings (CT)) influences on emergence of wetland plant species. Specifically, I addressed the following questions:

(3) Does adding a CT subsoil treatment increase the accuracy of seed bank analyses to reflect the standing vegetation of a wetland constructed on a CT subsoil?

(4) Can seed bank analyses and/or vegetation surveys be used to predict the initial establishment of vegetation at a created CT wetland site?

To address the third question, I compared wetland plant species composition and richness values obtained from controlled seed bank analyses of two wetlands built on CT substrates (1 m CT and 4 m CT wetlands) using a CT subsoil treatment, against initial vegetation surveys conducted prior to soil sampling for the seed bank analyses. I addressed the fourth question by comparing wetland plant species composition and richness values obtained from vegetation surveys of two reference wetland soils (Natural wetland (McLean Creek wetland) and Opportunistic wetland (Shallow wetland) conducted prior to seed bank sampling, against (1) controlled seed bank analyses using a CT subsoil treatment and (2) a manipulative study where these soil seed banks were transferred into an experimental CT wetland (i.e., vegetation was clipped to the soil surface and soils were placed in buckets and transplanted (level with the soil surface) in the experimental CT wetland).

4.2 Methods

Figures 2-1 and 3-1 show the location of the wetlands used in the manipulative field experiment (soil transfers) and greenhouse experiment (and seed bank analyses), respectively. Methods used for obtaining species lists and richness values from the manipulative field experiment (soil transfers into the experimental CT and reference wetlands) are described in Section 2.1. Methods used for selecting wetlands, sampling soils for seed bank analyses, and obtaining species lists and richness values for seed bank analyses in the greenhouse under potting and CT subsoil treatments are described in Sections 3.2.1 and 3.2.2. At the time of soil sampling for the seed bank analyses, initial vegetation surveys were conducted at each of the five random soil sample locations, using a 50 x 50 cm quadrat. All species within the quadrat were counted and identified.

4.3 Results

Differences in species composition and overall species richness were revealed between potting and CT subsoil seed bank treatments, standing vegetation, and soils transferred into the CT wetland (Table 4-1). In all Reclaimed, Natural and one of the Newly Constructed wetlands (4 m CT), more species emerged from the potting subsoil seed bank treatment than what was predicted by the standing vegetation (Figure 4-1). For Newly Constructed wetlands, species richness of the potting subsoil seed bank treatment was either equal (Peat Pond) or less than what was expected by the standing vegetation (1 m CT and Golden Pond) (Figure 4-2). Species richness of the potting subsoil seed bank treatment was higher than species emergence from the soil transfers in the CT wetland and the standing vegetation prior to sampling (August 2002) (Figure 4-3).

When the CT subsoil treatment was included in the seed bank analyses, species richness was equal to what was predicted by the standing vegetation in the 1 m CT wetland, and greater than what was predicted by the standing vegetation in the 4 m CT wetland (Figure 4-4). Species richness of the seed banks for the CT subsoil treatment was greater than the species richness of the soils transferred into the CT wetland (Figure 4-5). However, species richness of the seed banks for the CT subsoil treatment, compared to the standing vegetation, differed between the reference wetland sites. The standing vegetation at McLean Creek (Natural wetland) was less species rich than the seed bank for the CT subsoil treatment, whereas in

Shallow Wetland (Opportunistic wetland), the standing vegetation was more species rich than expressed by the seed bank for the CT subsoil treatment.

4.4 Discussion

Development of standard methods to evaluate wetland reclamation success is a primary interest of oil sands operators, stakeholder groups and governments. I qualitatively assessed whether seed bank analyses could accurately predict reclaimed wetland communities in the Northeastern Boreal Region of Alberta. Differences in overall species composition and richness were found between seed bank treatments, standing vegetation and soil transfers into CT wetland and reference plots in reference wetlands.

Seed banks were more species-rich than standing vegetation, except in Newly Constructed wetlands where species richness was either equivalent or less than what was expected by the standing vegetation. Previous studies have also concluded that seed banks are more species-rich than standing vegetation (Geritsen and Greening 1989; Haukos and Smith 1992 and 1994; Rossell and Wells 1999; Wilson et al. 1993). Dominance by clonal species, or highly competitive species adapted to specific conditions in wetland settings, combined with local seed rain (Schneider and Sharitz 1986) prohibits the establishment of some species, even if seeds are persistent in the seed banks. Furthermore, specific wetland conditions are known to influence macrophyte establishment including: hydrologic regime, depth of soil disturbance (McGee and Feller 1993) anoxia and salinity (Hackney et al. 1996). The Continuum Concept integrates the aforementioned biologic, physical and chemical factors that regulate spatial patterns of wetland plants (Mitsch and Gosselink 1986). The Continuum Concept describes spatial patterns or zonation of wetland plants as the result of “an environmental gradient of physical factors interacting with the biotic potential of a specific site” (Parrish and Bazzaz 1982). The variability of chemical and physical factors within and among wetlands compared in this study is inherently high and as such it is not surprising that seed banks of Reclaimed and Natural wetlands did not accurately reflect the standing vegetation.

In contrast, research by Ungar and Woodell (1993) shows how seed bank analyses can be used to accurately predict vegetation communities that are dominated by annuals and non-clonal perennials in hyper saline systems. Similar trends were seen in this study where seed banks of Newly Constructed wetlands mirrored standing vegetation communities. Seeds of early seral species dominate seed banks in early successional communities (Bekker et al. 2000). In a near-primary successional environment such as a CT wetland, primary successional species (r-adapted species) may have a competitive advantage over mid- late seral stage species. Primary successional species often have broad niches, and are better able to adapt to a range of environmental conditions. Primary successional species have preserved their genomes over time through evolving to use a larger portion of the resource spectrum in unpredictable and dynamic environments (Pickett and Bazazz 1978). This has been shown in prairie research where early successional annuals occupied larger proportions of a moisture gradients, overlapping more than what would be expected if the community was randomly organised. The species that have established in the Newly Constructed wetlands are likely easily dispersed and tolerant to a broader range of salinity compared to those species found in older Reclaimed and Natural wetlands.

Previous studies have indicated that seed bank analyses can be used to accurately predict the initial establishment of species when hydrologic regimes are controlled for (van der Valk and Peterson 1989; van der Valk et al. 1992). In this study, the standing vegetation was a better predictor of species establishment in reference plots in the reference wetlands, than the potting subsoil seed bank treatments. The standing vegetation more closely represented the initial species establishment in the Shallow/ Opportunistic Wetland, whereas in the McLean Creek/ Natural wetland, initial species establishment was much lower than the standing vegetation. I attribute the differences between wetlands to water regimes. During most of the 2002 growing season, the water levels in McLean Creek/ Natural Wetland were high, indicative of a submersed aquatic system. The initial establishment of species reflected this, with aquatic moss species, *Polygonum spp.* and *Lemna minor* dominating the plots, whereas the standing vegetation was indicative of a sedge meadow community. However by August 2003, as previously discussed in Section 2.4.2, water levels receded due to an upstream beaver dam cascade, and the vegetation responded accordingly; wet meadow and other forb species established in the plots. In contrast, in 2002, Shallow/ Opportunistic wetland experienced a significant drawdown indicative of drought conditions within the Northeastern Boreal Region. Emergent species such as *Sparganium angustifolium* and *Typha latifolia* established in the plots, closely mirroring established plant communities of the Shallow/ Opportunistic wetland. Based on the results of this study and that of other researchers, manipulating water regimes in seed bank analyses should result in more accurate predictions of wetland vegetation community development.

In this study, seed bank analyses did not accurately reflect initial species establishment of wetland plant species in reference wetland soil plots transferred in the CT wetland. Species richness of both potting and CT subsoil seed bank treatments were greater than species richness of the soil transfers in the CT wetland. As explained previously in Section 3.4.2, greenhouse conditions were not reflective of field conditions in the CT wetland. Pots were watered with tap water, while field plots were exposed to CT surface and porewaters which contain elevated concentrations of salts, total dissolved solids, naphthenic acids and other extractable hydrocarbons. Although more species were present in the potting subsoil treatments than CT subsoil treatments, significantly fewer species were found in the field plots in the CT wetlands, indicating that initial establishment is likely inhibited by the characteristics of the CT waters. Based on the results of this study, I predict that including CT water as a factor in future seed bank analyses should result in more accurate predictions of seed bank and vegetation community development of CT wetlands.

Although seed bank analyses did not accurately reflect standing vegetation communities in this study, I would argue that seed bank analyses in conjunction with vegetation surveys should be used as metrics to evaluate reclamation success through time. Research by Leck (1993) demonstrated that the species composition of soil seed banks and vegetation became more similar over time. However, as demonstrated in this study, in addition to inter-species and intra-species competition, physical and chemical conditions do control the initial establishment of species. Therefore, physical and chemical conditions to which soil seed banks are exposed should mimic that of the wetland system in question. Seed bank analyses are a powerful tool that can assess the vegetation history of a site, and both seed bank and vegetation development can be compared to reference sites to gain information about the ecological direction of that system.

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Table 4-1. Species composition and overall species richness values obtained from soil seed bank analyses (potting and CT subsoil treatments), vegetation assessments and soil transfers (August 2002).

(Natural Sites: McLean Creek (MC), Shallow Wetland (SW), Loon Lake (LL), Highway 63 (63); reclaimed sites: Suncor Natural Wetlands (NW), S Pit (SP), Bill's Lake (BL), High Sulphate wetlands (HS); Newly Constructed: Peat Pond (PP), Golden Pond (GP), 1m CT (1CT), and 4m CT (4CT). Data for soil transfers is only available for MC, SW, and 4CT sites. Soils transferred into the CT wetland are indicated as: SW-CT and MC-CT.

Species	**Seed bank (potting)**	**Seed bank (CT)**	**Vegetation**	**Soil Transfers**
Aleopecurus aequalis	MC;	MC;	GP;	
Beckmannia syzigachne	BL; HS;	HS;	HS;	
Brassicaseae Family	BL; HS;	BL;		
Calamagrostis Canadensis	HS; SW;	NW;		
Calamagrostis inexpansa	BL; GP; HS; LL; NW;	BL; HS;		
Carex atheroides	63; LL	63; LL, NW;	MC; LL;	
Carex aquatilis			LL;	
Carex lasiocarpa			BL; LL;	
Carex utriculata	MC;	MC;	HS; LL, MC; NW;	
Carex viridula			63; LL;	
Carex spp.	63; BL; HS; LL; MC; NW; SW;	63; HS; LL; MC; NW; SW;	1CT; BL; HS; MC; NW; SW;	SW;
Ceratophyllum demersum				SW;
Chara spp.	63; HS; LL; MC; SP; SW;	63; BL; HS; LL; SP; SW;	SW;	SW; SW-CT; 4CT;
Deschampsia caespitosa	HS; SP; MC;	HS; MC;		
Moss spp.	MC;	MC;		MC; MC-CT
Eleocharis palustris	MC;		MC;	
Epilobium ciliatum	BL; PP;			
Equisetem fluvatile	PP;	PP;	63; BL; SW; MC;	
Hippipuris vulgaris	SW;	SW;	GP; SW;	
Hordeum jubatum	BL; LL; NW; PP;	BL; NW;	LL; PP;	
Juncus alpinus	63; HS; LL; MC;	63; HS; LL;		
Juncus balticus	NW;	63; SP;	63;	
Juncus bufonius	1CT; 4CT; 63; HS; NW; SP;	1CT; 4CT; 63; HS; NW; SP;	SP;	
Juncus nodosus	HS;			
Juncus vaseyi	63;			
Juncus spp.			63;	
Laportea canadensis	HS; MC;			
Lemna minor		MC;	MC;	MC;
Mitella nuda	NW;			
Myriophyllum exalbescens				SW; SW-CT; MC-CT
Nuphar pumila		MC;	SW;	

Species	Seed bank (potting)	Seed bank (CT)	Vegetation	Soil Transfers
Poa palustris	63; BL; MC; PP; SW;	BL; PP; NW		
Poeaceae family		BL;	PP;	
Polygonum spp.			SW; MC	MC-CT; MC;
Potamogeton pectinatus	BL; SP; 63; LL; SW;	63; BL; SP;	NW; SP;	SW; SW-CT; MC-CT
Potamogeton pusilus	63; BL; LL; SW;	63; BL; LL; MC; SW;		
Potentilla palustris	MC; SW;	MC;		
Pyrola elliptica	MC;			
Ranunculus spp.			PP; SW;	
Riccocarpos natans		MC		
Scirpus cyperinus			SW;	
Scirpus pungens	LL;			
Scirpus validus	1CT; 4CT; 63; HS; NW; SP;	1CT; 4CT; HS; NW;	1CT; NW; SP;	SW;
Sparganium angustifolium	63; BL; HS; LL; SP; SW;	63; BL; LL; SW;		SW; SW-CT
Sparganium eurycarpum	BL; HS; 63;	BL; HS; 63; SW		
Thalictrum venulosum	BL; NW; SW;	BL;		
Trifolium repens	LL,			
Triglochin maritimum	63;		1CT; 63; SP;	
Typha latifolia	63; BL; GP; HS; LL; MC; NW; SW;	1CT; 63; BL; HS; LL; MC; NW; SW;	63; BL; GP; HS; LL; MC; NW; PP; SW;	MC-CT
Species Richness[1]				
Wetland				
1 m CT	2	3	3	
4 m CT	2	2	0	0
Golden Pond	2	0	3	
Peat Pond	4	2	4	
Highway 63 Wetlands	14	9	6	
Loon Lake	12	7	7	
McLean Creek	13	9	8	4 (CT); 3 (MC)
Shallow Wetland	11	6	9	4 (CT); 7 (SW);
Natural Wetlands	9	8	5	
S Pit	6	5	4	
Bill's Pond	13	13	4	
High Sulphate Wetlands	15	9	4	

[1]Only emergent and submergent vascular species were included when tabulating overall species richness values.

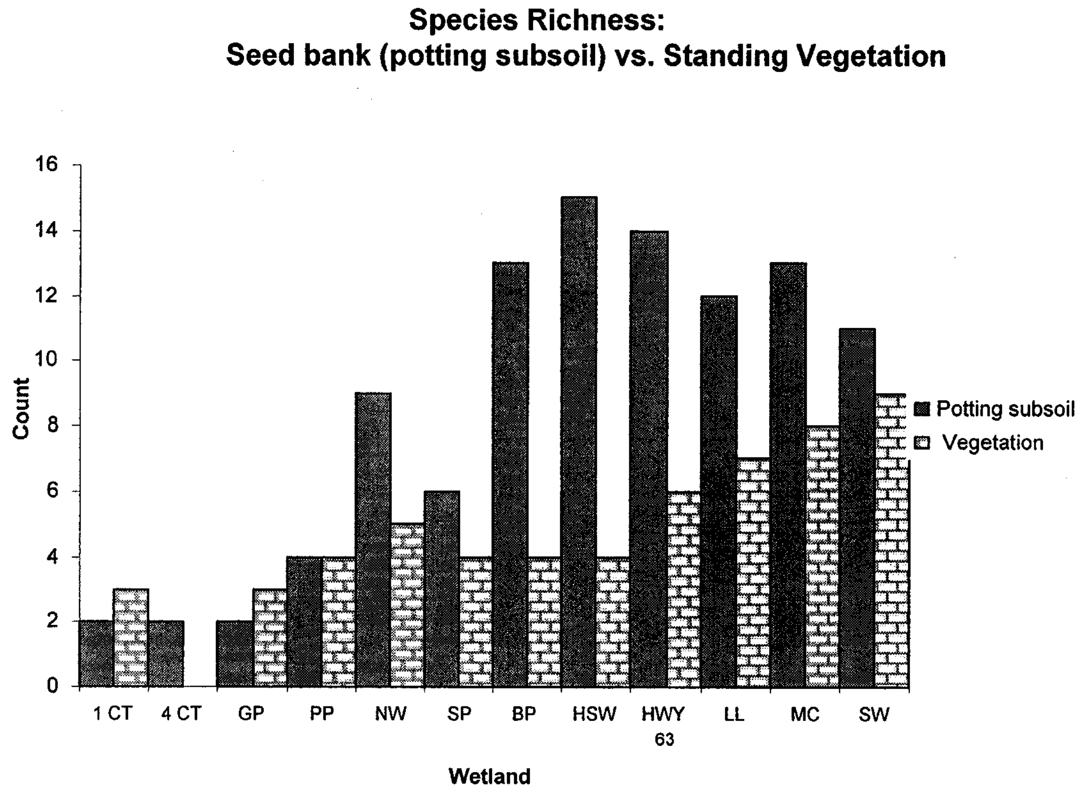

Figure 4-1. Species richness values obtained from seed bank analyses under the potting subsoil treatment compared to the standing vegetation observed at the wetlands prior to sampling (August 2002).

(Newly Constructed wetlands= 1CT (1m CT), 4CT, (4m CT), GP (Golden Pond), PP (Peat Pond); Natural wetlands= HWY63 (Highway 63 wetland), LL (Loon Lake), MC (McLean Creek), SW (Shallow Wetland); Reclaimed wetlands= NW (Suncor Natural Wetlands), SP (S Pit), BP (Bill's Pond), HSW (High Sulphate Wetland))

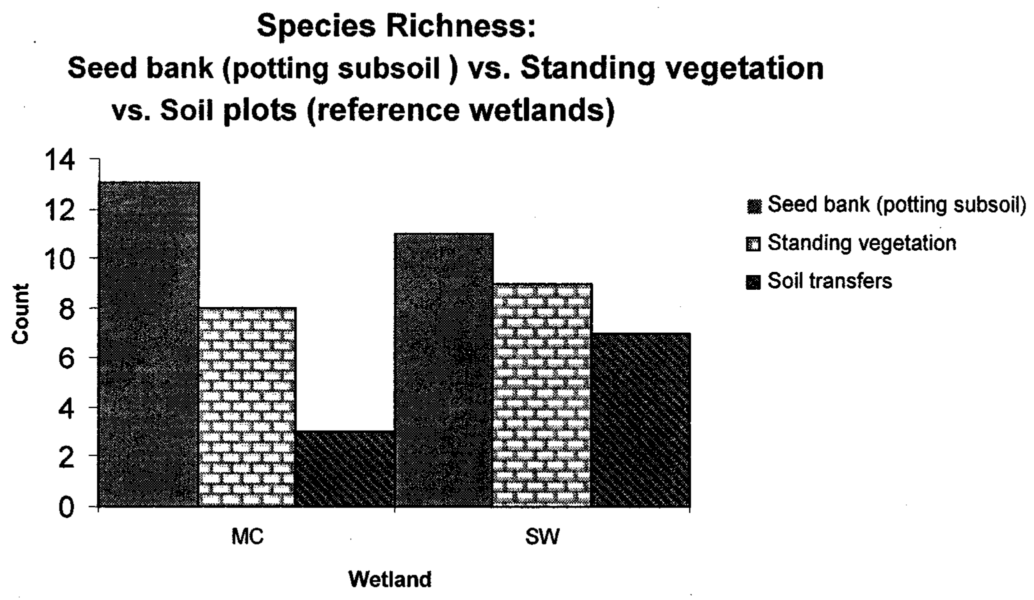

Figure 4-2. Species richness values obtained from (1) vegetation surveys conducted at reference wetlands prior to sampling (August 2002), (2) seed bank analyses under the potting subsoil treatment and, (3) assessing colonization of wetland plants in reference (control) soil plots in the reference wetlands (August 2002).

(MC= McLean Creek or Natural wetland soil transfers, SW= Shallow Wetland or Opportunistic Wetland soil transfers)

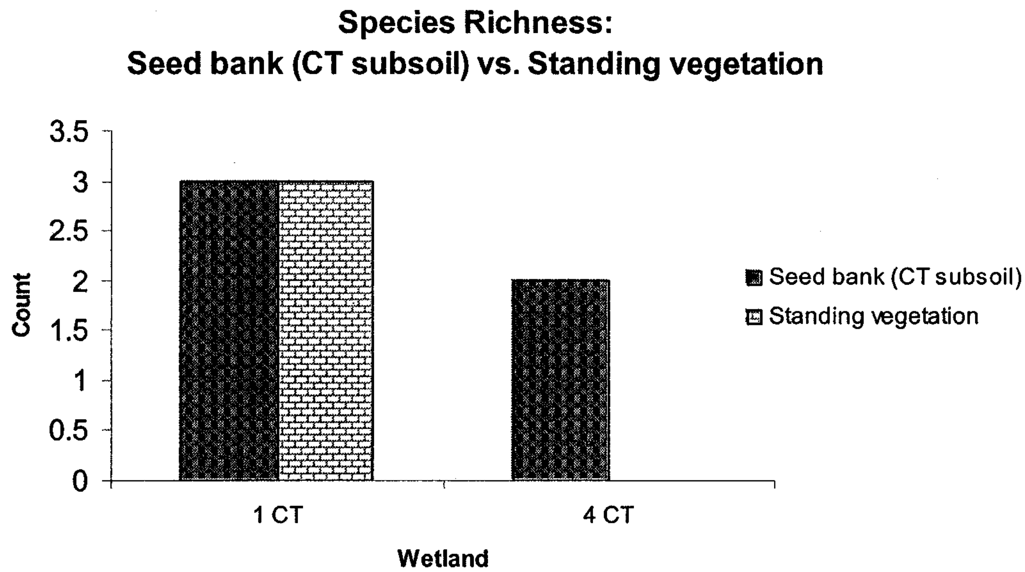

Figure 4-3. Species richness values obtained from (1) vegetation surveys of the CT wetlands prior to sampling (August 2002) and (2) soil seed bank analyses under the CT subsoil treatment.

(1CT= 1 m CT wetland, 4CT = 4 m CT wetland)

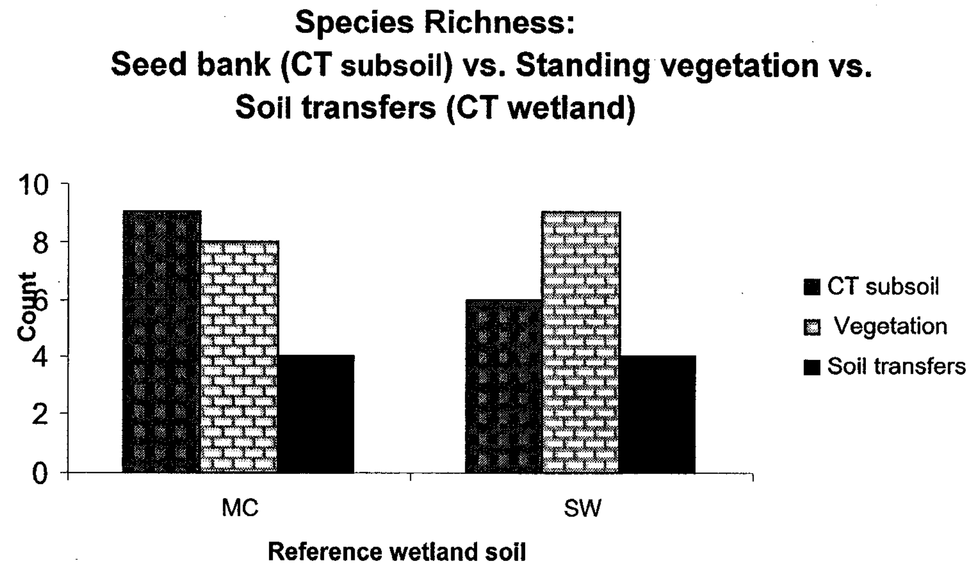

Figure 4-4. Species richness values obtained from (1) assessing the standing vegetation of reference wetlands prior to sampling, (2) analyzing soil seed banks of these reference wetlands under the CT subsoil treatment, and (3) assessing reference wetland soil plots transferred into the CT wetland (August 2002).

(MC= McLean Creek or Natural wetland soil transfers, SW= Shallow Wetland or Opportunistic Wetland soil transfers)

Chapter 5: Recommendations for Large Scale CT Wetland Reclamation

5.1 Recommendations for Management and Future Research

The following actions for oil sands wetland reclamation projects are recommended:

I. **Continue monitoring species composition and percent cover of plots in both CT and reference wetlands.** Two years of monitoring in this study is inadequate to predict the long-term sustainability of wetland plant communities on CT. Other studies have confirmed that short-term plant establishment data do not accurately represent longer term trends in plant community composition (Weiher et al. 1996). Continued monitoring of these plots will allow more accurate predictions as to the possible sequences of species assemblages that these CT wetland plant communities will demonstrate over the longer term.

II. **Continue monitoring health and expansion of plants in the CT and reference wetlands along with plant tissue analyses.** This information is required to further investigate whether CT wetland vegetation will show reduced growth over time. Because of the elevated concentrations of Na^{2+} and SO_4^{2-} in the surface and porewaters, over time, we may see stunted growth and possibly necrosis. If Na^+ and SO_4^{2-} ions accumulate in plant tissues, the plants will be less able to exclude, concentrate or dilute these ions in their tissues (Howes Keiffer and Ungar 1997).

III. **Initiate further controlled experimentation, likely in a laboratory setting, monitoring the emergence of wetland plants under varying dilutions of CT waters with freshwater over from CT subsoils capped with wetland soils.** This information would enhance the current understanding of CT water characteristics and its effects on plant establishment and survival under the self-design/ organisation approach to vegetating CT wetlands.

IV. **Initiate further studies that isolate the effects of salinity from other parameters such as naphthenic acids and other extractable hydrocarbons present in CT that might negatively impact wetland plant emergence, growth and survival.** Based on observations of salt crusts and seedling stress in CT treatments, I speculate that lowered emergence rates can be explained by the elevated concentrations of salts in the CT materials (Chapter 3). Salt marsh vegetation is relatively low in species richness, often dominated by vegetative colonizers (Parker and Leck 1985). This is also reflected in the seed banks. It is likely that certain perennial species such as *Juncus* and other wet meadow species absent from the CT subsoil seed bank treatments are intolerant of slightly to moderately saline conditions.

V. **Manage flow rates in CT deposits to ensure flooded soil conditions, thereby preventing salt crusting and subsequent vegetation stress.** In some of the drier Natural wetland CT transfer plots, salt crusts were present on the soil surface and the vegetation was showing visible signs of stress (Chapter 2).

VI. **Use seed bank analyses in conjunction with vegetation surveys as metrics to evaluate reclamation success through time.** Development of standard methods to evaluate wetland reclamation success is a primary interest of oil sands operators, stakeholder groups and governments. Research by Leck (2003) demonstrated that the species composition of seed banks and vegetation became more similar over time. However, as demonstrated in Chapters 2, 3, and 4, in addition to inter-species and intra-species competition, physical and chemical conditions do control the initial establishment of species. Therefore, physical and chemical conditions under which the seed banks are exposed to should mimic that of the wetland system in question. Seed bank analyses are a powerful tool that can assess the vegetation history of a site, and both seed bank and vegetation development can be compared to reference sites to gain information about the ecological direction of that system.

5.2 Literature Cited

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Appendix A

Appendix A-1. Photos of the transfer soil plots in the CT wetlands and corresponding soil plots in the reference wetlands

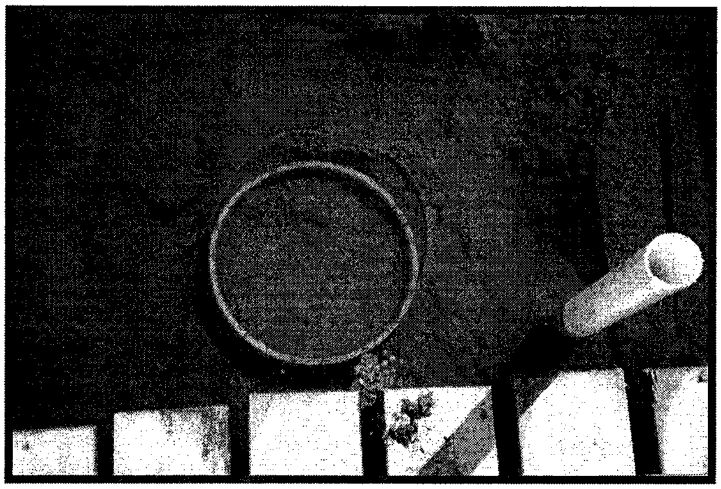


Photo 1. CT plot in the CT wetland (August 2002)

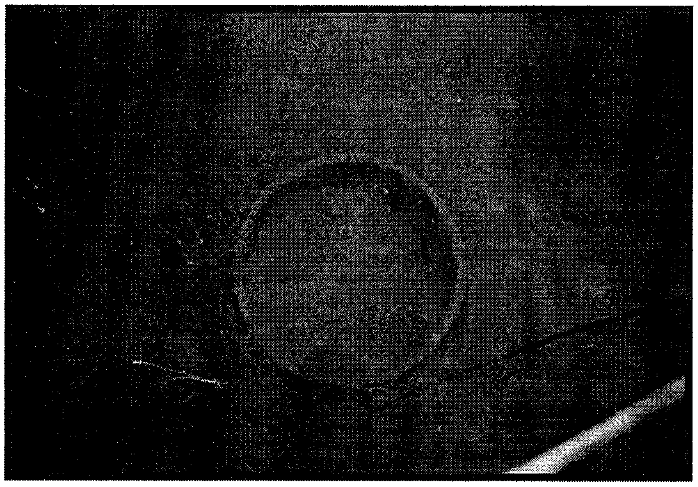


Photo 2. CT plot in the CT wetland (August 2003)

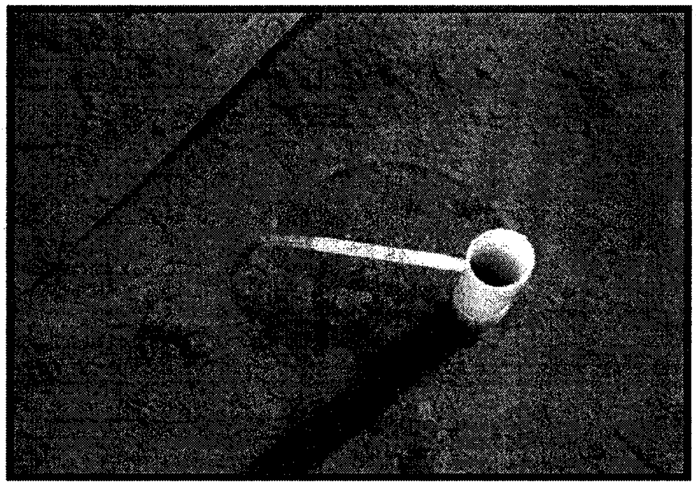


Photo 3. Opportunistic soil transfer plot in the CT wetland (August 2002)

Photo 4. Opportunistic soil transfer plot in the CT wetland (August 2003)

Photo 5. Natural wetland soil transfer plot in the CT wetland (August 2002)

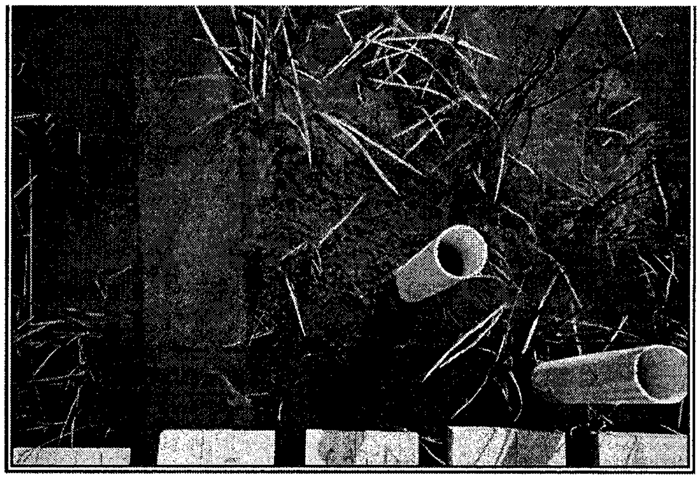


Photo 6. Natural wetland soil transfer plot in the CT wetland (August 2003)

Photo 7. Natural wetland reference soil plot (August 2002)

Photo 8. Natural wetland reference soil plot (August 2003)

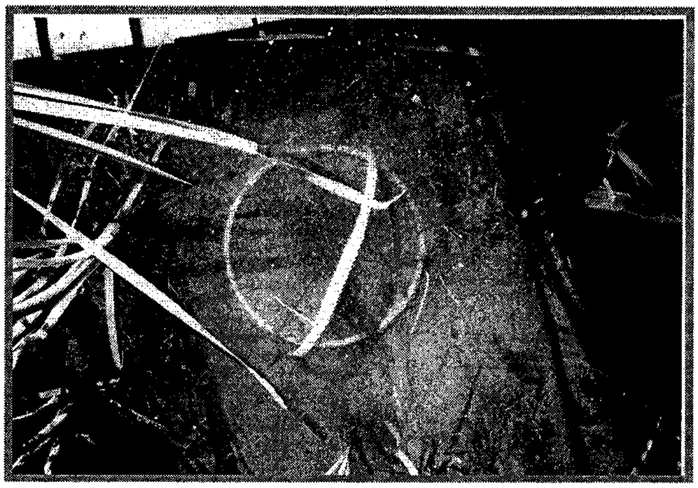


Photo 9. Opportunistic wetland reference soil plot (August 2002)

Photo 10. Opportunistic wetland reference soil plot (August 2003)

Photo 11. CT transfer plot in the Natural wetland (August 2002)

Photo 12. CT transfer plot in the Natural wetland (August 2003)

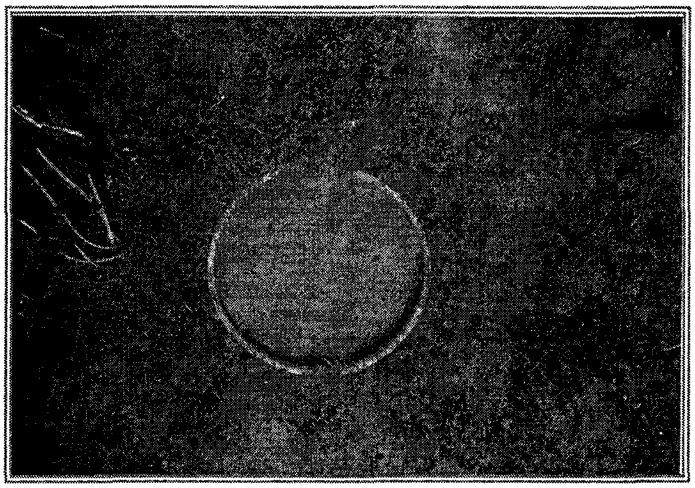


Photo 13. CT transfer plot in the Opportunistic wetland (August 2002)

Photo 14. CT transfer plot in the Opportunistic Wetland (August 2003)

Appendix B

Appendix B-1. Results of chemical analyses of surface waters in the CT, Opportunistic and Natural Wetlands (September 2002, August 2003). Field water quality measurements (salinity, conductivity, pH and temperature) are also reported. Reported values are means, followed by standard error of the mean.

	CT Wetland		Opportunistic Wetland		Natural Wetland	
(ppm)	2002	2003	2002	2003	2002	2003[1]
Ca^{2+}	46.3 ± 0.6	27.3 ± 2	48 ± 0.4	15.0 ± 0.4	34.9 ± 0.2	35.0 ± 2.6
Mg^{2+}	38.4 ± 0.2	23.0 ± 2	31.6 ± 0.2	35 ± 0.2	10.0 ± 0.0	10.0 ± 0.9
Na^{+}	357.7± 3.4	479.7 ± 4	147.7 ± 4.8	94.3 ± 1.5	6.3 ± 0.0	5 ± 0.2
NH_4^{+}	6.5	2.7 ± 0.6	BDL	BDL	BDL	BDL
K^{+}	11.9± 0.0	13.1 ± 0	5.6 ± 0.2	1.8 ± 1.7	BDL	0.1 ± 0
Cl^{-}	52 ± 0.6	86 ± 0	45 ± 0.6	35 ± 0.6	2.6 ± 0.0	1.2 ± 0.1
CO_3^{2-}	25.2 ± 13.9	BDL	BDL	BDL	BDL	BDL
SO_4^{2-}	400.3 ± 0.3	272.3 ± 2	311.7 ± 73.7	8 ± 1.2	BDL	7.3 ± 5
HCO_3^{-}	677.7 ± 39.4	969 ± 16	229.3 ± 43.5	349.3 ± 3.5	157.7± 39.4	180 ± 19.4
B	2.5 ± 0.0	2.6 ± 0.0	BDL	0.16 ± 0.01	BDL	0.05 ± 0.01
Ba	0.06 ± 0.00	BDL	BDL	BDL	BDL	BDL
S	114.7 ± 0.3	94.8± 0.1	70.9 ± 7.0	BDL	BDL	2.1 ± 2.1
Si	0.05 ± 0.03	3.0 ± 0.4	BDL	BDL	4.2 ± 1.0	1.7 ± 1.7
Sr	0.01 ± 0.01	0.54 ± 0.02	0.21 ± 0.00	0.16 ± 0.01	0.09 ± 0.02	0.11 ± 0
pH	8.47 ± 0.01	8.4 ± 0.0	9.6 ± 0.0	8.4 ± 0.1	7.1± 0.0	N/A
TN	N/A	2.93±0.34	N/A	2.66±0.33	N/A	1.61±0.25
TP	N/A	0.07±0.02	N/A	0.09±0.04	N/A	0.10±0.01
Salinity (ppt)	1.09 ± 0.00	1.1 ± 0.0	0.48 ± 0.08	0.3 ± 0.0	0.15 ± 0.01	N/A
Cond (µS/cm)	2143 ± 7	1987 ± 9	945 ± 9	540 ± 6	296 ± 17	N/A
Temp (°C)	22.4 ± 0.1	20 ± 0.2	21 ± 0.2	19 ± 0.1	17.2 ± 0.4	N/A
Depth (cm)	19.9 ± 0.5	19.3 ± 0.5	9.7± 0.7	19.3 ± 0.8	3.6 ± 0.8	0

[1]Surface water samples for the Natural wetland were taken at random locations adjacent to the plots, because there was no surface water in the study area in 2003. BDL= below detectable levels, N/A= not applicable (was not analysed for).

Appendix B-2. Porewater chemistry of leached soil samples taken from control areas and soil plots in the CT wetland in August 2002 and 2003.

Reported values are means (ppm), expressed as the mass of the specific ion in leachate (total amount of water added + % water content) followed by the standard error of the mean.

	Depth	Control Areas		CT Plots		Opportunistic Wetland Soil Transfer Plots		Natural Wetland Soil Transfer Plots	
ppm	(cm)	2002	2003	2002	2003	2002	2003	2002	2003
Ca^{2+}	0-10	166.9±9.8	117.8±8.4	194.2±30.1	127.6±14.6	158.3±17.2	158.3±12.7	136.9±15.8	177.3±25.9
	10-20	102.6±29.0	134.9±13.6	106.8±30.1	231.9±114	110.9±15.3	195.4±9.5	174.2±42.8	146.1±15.5
Mg^{2+}	0-10	76.6±11.7	126.4±32.5	373.7±116.5	164.5±47.1	107±47.5	107±11	71.8±7.7	165.1±27.5
	10-20	56.1±15.8	121.8±21.6	216±131.5	482±248	91.2±4.9	104.2±86	259.9±176.4	119.2±14.1
Na^{+}	0-10	708±73.4	669.1±84.2	868.8±119.3	954±93.5	717.8±71.2	718±63.1	768.5±79	958.6±51.9
	10-20	603.4±52.2	727.7±57.3	822.1±111.5	101.4±276.2	634.8±113.5	1022±15	781.2±12.6	936.3±15.4
K^{+}	0-10	93.8±18.2	204.3±97.2	1121±504	301.7±133.7	123.2±123.2	123.2±34.3	76.5±23.1	72.2±13.3
	10-20	79.1±15.5	205.9±60.7	530±332	1203±69	145±15.9	85±13	577.2±484.3	57.8±3.9
Cl^{-}	0-10	85.6±10.6	51.1±15.4	89.4±8.6	79.8±10.5	155.3±26.1	155.3±1.60	115.1±8.1	86.9±7.1
	10-20	64.1±6.8	52.8±7	68.7±4.6	84.7±18.5	75.8±3	105.2±8.1	87.5±19.5	89.1±5.4
SO_4^{2-}	0-10	89.4±17.1	120.2±94.8	108.6±41	113.5±45.3	13±8.9	13±23.5	13.5±4.5	9.9±3.8
	10-20	106.5±5.7	35.2±18.2	22.2±3.5	587.5±558.3	20.5±6.4	39.8±4.5	13.1±6.3	13.3±13.3
HCO_3^{-}	0-10	2661	4286	6258	3946	2840	3599	2725	3874
	10-20	2076.5	3147	4287	6825	2556	3738	4660	3455
Al^{3+}	0-10	126±34.9	605.7±559.3	5463±262	184.7±236.8	585.7±537	585.7±155	250±100.5	12.3±4.7
	10-20	69.4±38.4	160.7±271.5	812±248.1	5248.8±294	450.2±1.6	171.5±100.5	1155±995	13±9
B	0-10	6.6±0.7	7.8±0.7	17.7±11.1	9.9±1.8	5.5±2.8	5.8±0.5	5.2±1.1	6.6±0.6
	10-20	6.2±0.4	8.7±1.2	13.7±2.6	19.9±6.3	7.9±1.6	6.9±1.2	12.2±6.5	6.1±0.1
Ba	0-10	0.3±0	1.3±0.8	6.7±2.3	2.1±1.1	1.3±0.6	0.09±0.05	0.4±0.1	0.12±0.12
	10-20	0.2±0.1	1.3±0.5	3.8±2.8	8.5±3.4	0.99±0.23	0.39±0.39	4.5±4.1	0.84±0.01
Cr	0-10	BDL	0.84±0.84	5.8±5.8	1.1±1.1	0.76±0.38	BDL	BDL	BDL±
	10-20	BDL	0.55±0.55	2.9±2.9	7.2±2.9	BDL	BDL	3.7±3.7	BDL
F	0-10	9.7±1	13.2±5.8	15.0±2.5	8.4±0.4	4±2	1.8±1.8	2.3±1.3	32.3±27.9
	10-20	8.3±0.3	8±1.7	16.7±4.9	16.1±6.7	12.4±6.7	3.8±3.8	8.3±3.5	BDL
Fe	0-10	29.6±7.8	162.6±120.9	1268±612	66.2±142.1	148.8±123.7	149±42.2	70.7±23.1	6.5±1.2
	10-20	16.2±9.1	155.8±61.1	552.3±372.2	1309±803	108.8±1.2	51.3±23.1	584.5±540.4	5.4±3.3
Mn	0-10	BDL	0.28±0.28	2.9±1.4	0.41±0.41	1.5±0.4	1.5±0.3	0.8±0.5	0.17±0.17
	10-20	BDL	BDL	1.3±1.3	3.0±2.6	0.22±0.22	0.5±0.5	2.1±1.7	0.11±0.11
S	0-10	28.5±15.5	42.4±42.4	68±24.2	49.9±20.4	BDL	BDL	BDL	BDL
	10-20	40.5±0.3	62.5±2.9	23.6±23.6	241.4±146.9	27.4±27.4	BDL	24.4±24.4	BDL
Si	0-10	227.1±60.6	1170±847	6497±2277	1904±1113	1216±837	1216±21	452.6±161.3	58±5.8
	10-20	130.2±64.4	1169±0	3889±2761	8328±4537	847±85.2	470.8±340	4314±3986	58.4±13.2
Sr	0-10	1.8±0.2	2.3±0.4	4.8±1	2.8±0.5	2.4±1.2	1.6±0.1	1.1±0.2	2.1±0.4
	10-20	1.6±0.3	2.6±0.3	3.1±1.4	5.4±1.9	2±0.1	2.2±0.5	3.7±2.1	1.7±0.1
Ti	0-10	1.3±0.6	14.2±11.6	61.5±16.1	22.4±12.9	19±9.5	0.2±0.2	3.7±1.4	BDL
	10-20	0.64±0.27	15.0±6.1	50.2±40.8	59.8±23.1	9.5±2.7	3.3±3.3	57.2±53.5	BDL
V	0-10	BDL	1.2±1.2	9.2±7	1.5±1.5	0.98±0.49	BDL	BDL	BDL
	10-20	BDL	BDL	4.8±2.2	10.6±4.5	BDL	BDL	4.9±4.9	BDL
Zr	0-10	BDL	1.3±1.3	8.7±3.2	1.9±1.4	1.2±0.6	BDL	0.24±0.24	BDL
	10-20	BDL	0.77±0.77	5.1±3.6	10.1±4.7	0.54±0.54	BDL	5.9±5.9	BDL
pH	0-10	7.9±0	7.8±0.1	8.4±0.1	7.8±0.1	7.9±3.9	7.7±0.1	7.7±0.1	7.5±0.1
	10-20	7.9±0.1	7.9±0.0	8.2±0.1	8±0.1	7.7±0.1	7.8±0.2	7.7±0.3	7.6±0
cond	0-10	635±37	520±67	425±40	604±32	409±204.5	727±82	7±0	7.5±0.1
	10-20	567±33	464±17	440±0.5	542±19	415.2±4.5	550±94	7±1	7.6±0

BDL = below detectable levels

Appendix B-3. Porewater chemistry for leached soil samples taken from control areas and reference plots in the Opportunistic wetland (August 2002 and 2003).

Reported values are means (ppm), expressed as the mass of the specific ion in leachate (total amount of water added + % water content) followed by the standard error of the mean. BDL= below detectable levels

		Control Areas		Opportunistic Wetland Reference Plots	
(ppm)	Depth	2002	2003	2002	2003
Ca^{2+}	0-10 cm	401.4±101.1	303.4±4	299.4±33.5	321.8±40.4
	10-20 cm	476.2±126.5	404.5±57.7	758.5±168.4	302.2±55
Mg^{2+}	0-10 cm	141.8±35.9	110.2±12.4	127.1±13.3	128.7±22.2
	10-20 cm	181.9±70.8	144.8±27.2	274.8±56	10.5±16.1
Na^{+}	0-10 cm	481.9±11.7	248.2±34.8	453.3±41.8	461.1±92
	10-20 cm	478±115.4	514.1±55.4	552.4±11.2	362.3±20.3
K^{+}	0-10 cm	12.6±12.6	20.5±20.5	13.7±13.7	BDL
	10-20 cm	BDL	BDL	BDL	BDL
Cl^{-}	0-10 cm	98.6±8.3	72.5±47	196±56.8	91.8±15.8
	10- 20 cm	105.2±25.2	91.3±1.2	111.9±26.4	112±14.1
SO_4^{2-}	0-10 cm	704.9±546.4	328.5±225.5	26.9±8.9	86.5±63.8
	10-20 cm	841±689	479.8±27.3	2360±1131	264.1±265.1
HCO_3^{-}	0-10 cm	2178	1633	2412	2590
	10-20 cm	2396	2566	2165	1900
Al^{3+}	0 -10 cm	2.1±2.1	BDL	9.9±3.7	BDL
	10- 20 cm	BDL	7.5±3.8	BDL	4.5±4.5
B	0-10 cm	BDL	BDL	BDL	BDL
	10-20 cm	BDL	BDL	BDL	BDL
Ba	0-10 cm	BDL	0.13±0.13	0.13±0.08	BDL
	10-20 cm	BDL	0.17±0.17	BDL	BDL
Cr	0-10 cm	BDL	BDL	BDL	BDL
	10-20 cm	BDL	BDL	BDL	BDL
F	0-10 cm	BDL	2.3±2.3	4.8±0.8	BDL
	10-20 cm	BDL	18.4±18.4	BDL	BDL
Fe	0-10 cm	0.69±0.68	16±3.1	7.6±3.4	23±7.8
	10-20 cm	BDL	10.1±0.5	BDL	10.6±8.2
Mn	0-10 cm	6.9±2.2	8±3.2	2.1±0.4	3.7±0.5
	10-20 cm	7.5±2.4	6.6±1.5	3.5±2.5	3.1±0.2
S	0-10 cm	238.2±183.6	58.6±58.6	BDL	25.6±25.6
	10-20 cm	252.1±222.9	BDL	818.4±546.1	99.2±99.2
Si	0-10 cm	51.4±9.6	42.7±7.2	101.3±22.8	76.7±15.7
	10-20 cm	68.2±16.7	110±22.8	42.7±8.3	71.6±20.6
Sr	0-10 cm	2.2±0.1	1.7±0	1.9±0.1	2±0.4
	10-20 cm	2.4±0.7	2.2±0.6	3.5±1	1.6±0.1
Ti	0-10 cm	BDL	BDL	BDL	BDL
	10-20 cm	BDL	BDL	BDL	BDL
V	0-10 cm	BDL	BDL	BDL	BDL
	10-20 cm	BDL	BDL	BDL	BDL
Zr	0-10 cm	BDL	BDL	BDL	BDL
	10-20 cm	BDL	BDL	BDL	BDL
pH	0-10 cm	7.3±0.1	7.2±0.3	7.7±0.1	7.4±0.1
	10-20 cm	7.3±0.1	7.3±0.3	7.4±0.1	7.4±0.2
cond (uS/cm)	0-10 cm	724±144	620±51	498±62	443±24
	10-20 cm	668±87	469±46	645±88	390±70

Appendix B-4 Porewater chemistry of leached soil samples taken from control areas reference soil plots in the Natural wetland (August 2002, 2003).

Reported values are means (ppm), expressed as the mass of the specific ion in leachate (total amount of water added + % water content) followed by the standard error of the mean.

		Control Areas		Natural Wetland Reference Soil Plot	
(ppm)	Depth	2002	2003	2002	2003
Ca^{2+}	0-10 cm	92.1±18.1	30.6±6.8	110.3±23.1	186.8±23.8
	10-20 cm	80.1±8.7	211.2±55.2	190.6±58	188.5±24
Mg^{2+}	0-10 cm	25.5±3.1	9.3±1.5	29.9±5.9	50.2±6
	10-20 cm	40.9±15.3	55.8±12.5	51.1±17	64±13.4
Na^{+}	0-10 cm	26.7±5.1	20.8±5.6	34.4±5.4	28.3±8.9
	10-20 cm	18.4±4.6	43.7±11.7	35.6±14	29.3±10.1
K^{+}	0-10 cm	13.6±13.6	BDL	39.9±19.6	28.4±10
	10-20 cm	31.9±31.9	BDL	BDL	41.4±12.3
Cl^{-}	0-10 cm	26.1±6.2	10.7±2.7	36±13.7	15.7±2.7
	10- 20 cm	9.7±2.4	16.5±2	14.6±1.8	12.8±1.0
SO_4^{2-}	0-10 cm	3.1±3.1	8.6±1.9	2±1.3	10.9±4.9
	10-20 cm	BDL	10.1±4.6	13.2±13.2	15.7±8.4
HCO_3^{-}	0-10 cm	453.8	166.6	577.5	903.4
	10-20 cm	533.9	1003	893.7	1001
Al	0-10 cm	31.1±14.5	24.3±8	4.4±4.4	BDL
	10-20 cm	230±155.8	35.6±27.9	14.6±5.5	38.4±193.8
B	0-10 cm	BDL	BDL	BDL	BDL
	10-20 cm	BDL	BDL	BDL	BDL
Ba	0-10 cm	BDL	BDL	BDL	0.03±0.03
	10-20 cm	0.5±0.7	BDL	BDL	0.60±0.60
Cr	0-10 cm	BDL	BDL	BDL	BDL
	10-20 cm	0.22±0.22	BDL	BDL	BDL
F	0-10 cm	BDL	BDL	BDL	BDL
	10-20 cm	BDL	BDL	BDL	BDL
Fe	0-10 cm	15.5±5.8	10.2±3.1	3±1.7	7.9±2.4
	10-20 cm	105.7±72.1	16.3±11.6	43±35.9	145.8±123
Mn	0-10 cm	0.23±0.23	0.05±0.05	0.8±0.6	2.5±0.7
	10-20 cm	0.60±0.09	0.74±0.74	2.4±1.3	2.9±0.9
S	0-10 cm	BDL	BDL	BDL	BDL
	10-20 cm	BDL	BDL	BDL	BDL
Si	0-10 cm	135.8±319.2	66±8	48.8±4.7	49.8±11.7
	10-20 cm	484.4±8	140±54.2	70.1±7	411.4±362.6
Sr	0-10 cm	0.46±0.05	0.64±0.08	0.5±0.1	0.64±0.08
	10-20 cm	0.76±0.22	0.90±0.18	0.8±0.2	0.90±0.18
Ti	0-10 cm	BDL	BDL	BDL	3.2±3.2
	10-20 cm	BDL	3.2±3.2	BDL	BDL
V	0-10 cm	BDL	BDL	BDL	BDL
	10-20 cm	BDL	BD	BDL	BDL
Zr	0-10 cm	BDL	BDL	BDL	BDL
	10-20 cm	0.26±0.26	BDL	BDL	BDL
pH	0-10 cm	6.7±0	7.3±0.1	7.3±0.1	7±0.1
	10-20 cm	6.8±0.1	6.9±0.2	7.3±0	7.1±2.1
cond (us/cm)	0-10 cm	166.7±13	84±32	359±80	325±39
	10-20 cm	85±6	117±12	187±1	241±155

BDL= below detectable levels

Appendix B-5. Porewater chemistry of leached soil samples taken from CT transfer plots in the Natural and Opportunistic wetlands (August 2002, 2003)

Reported values are means (ppm), expressed as the mass of the specific ion in leachate (total amount of water added + % water content) followed by the standard error of the mean..

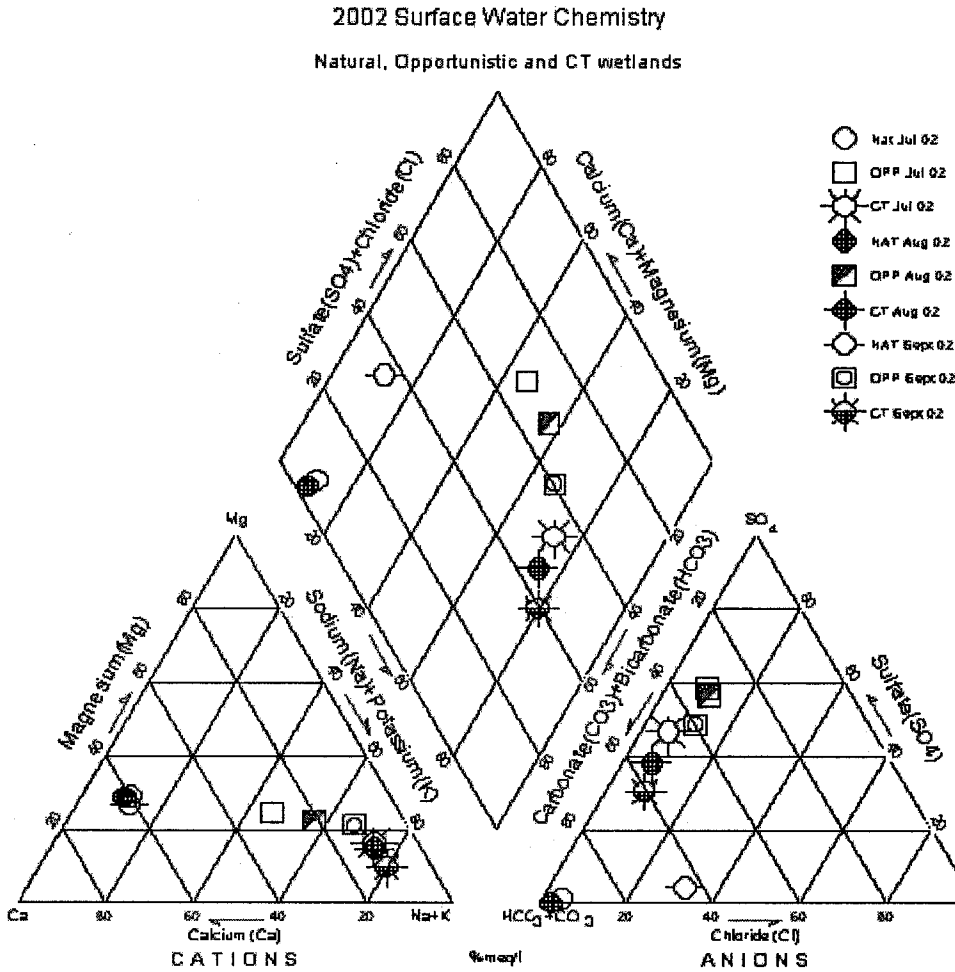
		Opportunistic		Natural	
(ppm)	Depth (cm)	2002	2003	2002	2003
Ca^{2+}	0-10	170.8±28.2	216.4±31.8	146.5±18.3	320.2±44
	10-20	268.2±109	161.4±27.9	165.3±7	306±1.9
Mg^{2+}	0-10	136.7±37.8	123.1±11.5	90.8±9.6	125.8±18.7
	10-20	139.7±34.3	72.6±12.8	79.5±13.9	116.9±8.9
Na^{+}	0-10	687.6±113.1	584.5±84.6	415.1±84.6	201±27.7
	10-20	629±29.1	517.9±88.8	391.7±68.6	276.2±11.5
K^{+}	0-10	197.5±96.3	97.7±21.7	125.3±23.1	146.5±38.8
	10-20	78.8±78.8	45±7.8	38.8±17.4	71.8±20.8
Cl^{-}	0-10	148.7±41.2	100.4±12.3	29.4±4.6	12.3±3.3
	10-20	118.8±7.4	93.3±13	29.1±1.8	15.3±0.6
SO_4^{2-}	0-10	267.4±187.2	144.2±118.1	12.9±3.1	515.1±52
	10-20	548.4±451.4	14.1±14.1	8.3±2.6	478.1±78.3
HCO_3^{-}	0-10	2752	2633	2138	1576
	10-20	2418	2127	1949	1738
Al	0-10	739.4±564.5	113±65.7	242.5±93.7	68.8±21.2
	10-20	217.6±186.6	43.7±3.6	62.7±5.9	38.5±5.6
B	0-10	8.3±1.4	6.1±1	6.6±0.9	5.1±0.9
	10-20	6±0.3	4±0.9	5.3±1.4	4.8±0.4
Ba	0-10	1.5±0.9	0.5±0.2	0.8±0.2	0.2±0.2
	10-20	0.48±0.48	BDL	0.3±0.3	0.1±0.1
Cr	0-10	0.78±0.78	BDL	BDL	BDL
	10-20	BDL	BDL	BDL	BDL
F	0-10	10.8±1.9	5.5±2.1	7±1	2.1±1.5
	10-20	6±0.7	BDL	2.3±2.3	2.4±2.4
Fe	0-10	153.5±113.4	27.8±15.4	54.4±20.8	15.9±4.7
	10-20	50.3±41.2	19.5±2.6	32.6±4.4	10.8±0
Mn	0-10	0.29±0.29	0.29±0.2	BDL	BDL
	10-20	0.26±0.26	0.89±0.2	0.24±0.24	0.38±0.38
S	0-10	107.1±63.8	48.6±48.6	BDL	192.3±3.8
	10-20	181.7±181.7	BDL	BDL	180±30.8
Si	0-10	1145±806	226.6±114.2	437.1±153.8	133.3±15.8
	10-20	399.3±308.8	136±18.6	135.6±35.2	93.1±11
Sr	0-10	3±0.6	2.8±0.4	2.4±0.2	3.5±0.6
	10-20	2.9±0.3	1.6±0.3	1.8±0.4	2.8±0.5
Ti	0-10	17.1±12.9	1±0.7	4.4±2.4	0.7±0.5
	10-20	6.6±6	0.22±0.22	2.3±0.1	0.30±0.30
V	0-10	1.2±1.2	BDL	BDL	BDL
	10-20	BDL	BDL	BDL	BDL
Zr	0-10	1.3±1.3	BDL	BDL	BDL
	10-20	BDL	BDL	BDL	BDL
pH	0-10	7.9±0.1	7.7±0.1	7.5±0.1	7.5±0.1
	10-20	7.4±0	7.6±0	7.4±0	7.5±0.1
cond (uS/cm)	0-10	463±36	447±4	408±30	341±19
	10-20	439±5	387±25	375±62	392±1

BDL= below detectable levels

Appendix B-6. Mean percent water, solids and oil content and particle size distribution for samples taken from soil plots, and control areas in the CT, Opportunistic and Natural Wetlands (2002).

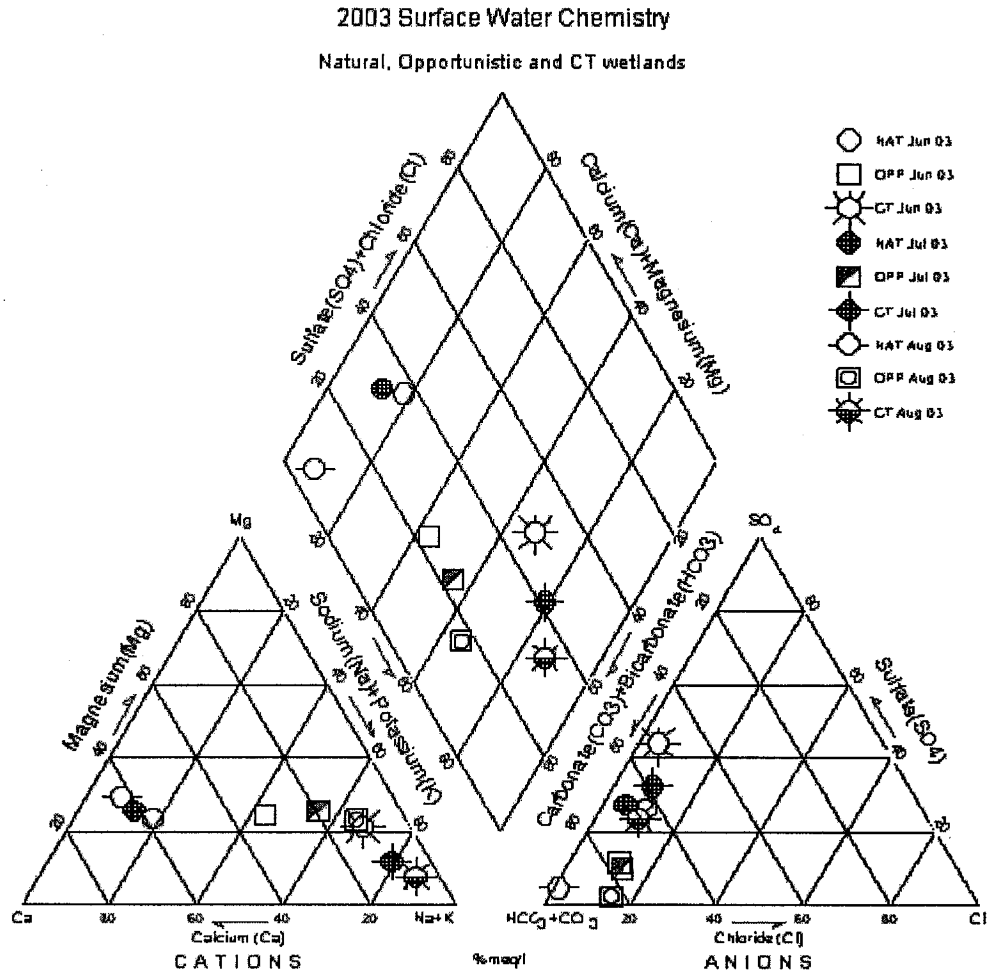
Location	n	Depth	% Water	% Solids	% Oil	Particle Size Distribution (um)						
						<1	2.8	5.5	11	22	44	125
CT Wetland Plots												
CT	3	0-10	30.4	67.3	2.8	0	18.0	29.1	40.5	49.3	55.5	69.8
	2	10-20	23.2	74.3	2.4	0	17.2	28.8	40.6	49.4	55.7	69.9
Opportunistic	3	0-10	37.5	62.5	0.7	0	45.9	61.7	75.5	86.1	94.2	100
	2	10-20	25.2	74.0	0.8	0	30.6	43.1	53.9	62.5	70.9	81.8
Natural	3	0-10	63.2	37.1	0.5	0	15.8	23.7	32.0	40.2	48.6	65.8
	2	10-20	57.4	41.8	1.2	0	16.8	25.3	34.5	43.3	51.8	70.2
Control Areas	1	0-10	30.0	66.3	3.0	0	21.4	35.0	48.9	59.0	65.6	78.5
	1	10-20	23.0	74.3	2.4	0	14.4	25.1	36.5	44.9	51.3	66.0
Opportunistic Wetland Plots												
CT transfers	3	0-10	26.6	71.0	2.6	0	25.4	42.6	60.0	73.1	82.3	98.2
	2	10-20	25.0	73.9	1.3	0	29.9	45.0	58.9	69.4	77.8	87.2
Reference Soil Plots	3	0-10	27.1	73.9	0.1	0	44.1	58.6	70.4	80.1	89.0	100
	2	10-20	27.3	72.5	0.1	0	62.1	78.9	94.1	99.8	100	100
Control Areas	1	0-10	31.4	68.3	0.1	0	66.2	81.8	94.1	100	100	100
	1	10-20	27.7	71.7	0.1	0	66.7	82.9	95.3	100	100	100
Natural Wetland Plots												
CT transfers	3	0-10	22.8	74.7	3.0	0	20.0	33.0	46.5	57.2	65.6	77.9
	2	10-20	20.7	77.4	1.9	0	29.1	45.6	60.6	71.2	78.9	88
Reference Soil Plots	3	0-10	63.0	37.4	0.1	0	13.7	20.4	27.6	35.4	44.6	66.1
	2	10-20	41.3	59.0	0.1	0	28.7	41.1	52.5	62.6	73.4	87.1
Control Areas	1	0-10	47.2	53.6	0.1	0	9.6	14.9	21.6	29.5	39.0	58.9
	1	10-20	16.1	84.5	0	0	17.	25.1	33.8	43.2	53.2	72.3

Appendix C



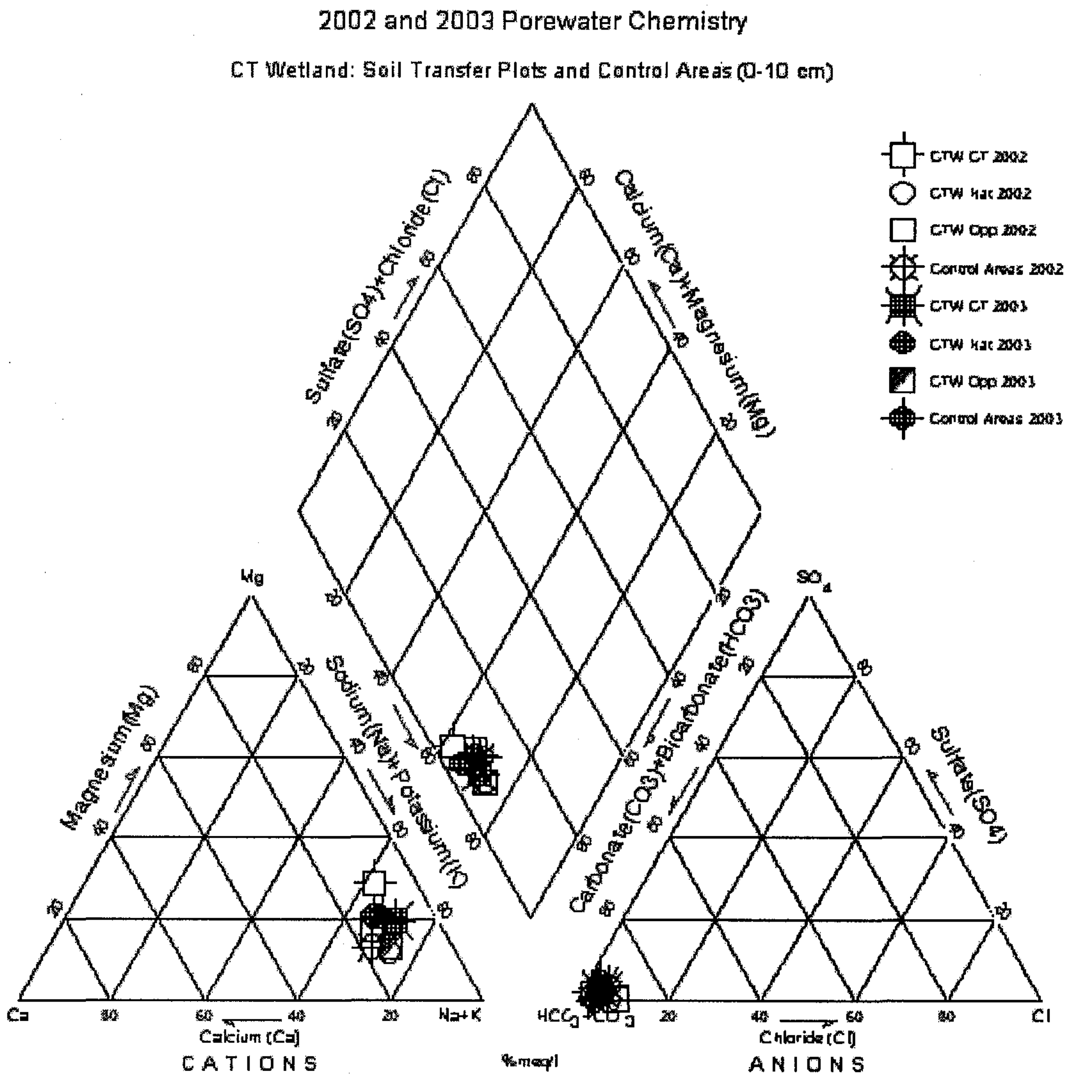
Appendix C-1. Piper plot of surface water chemistry for samples taken in July, August and September 2002 in the Natural, CT and Opportunistic wetlands.

(Nat= Natural wetland, OPP= Opportunistic wetland, CT= CT wetland)



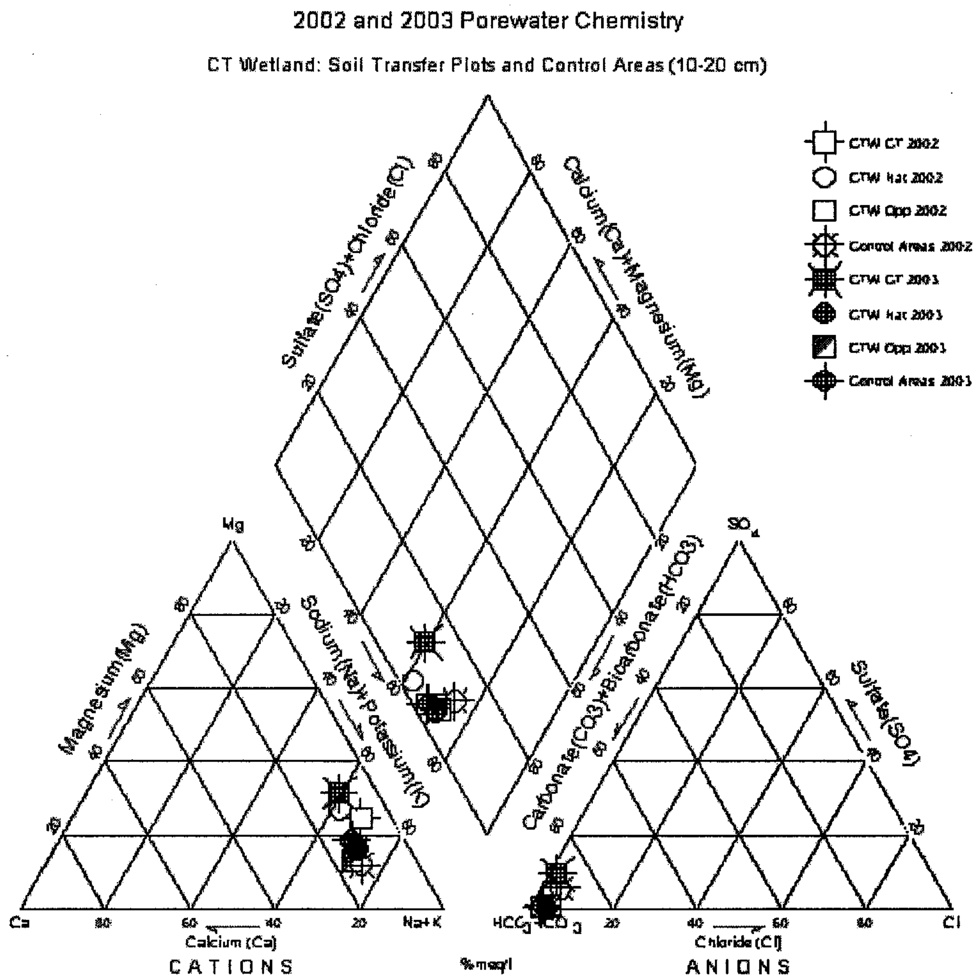
Appendix C-2. Piper plot of surface water chemistry for samples taken in June, July and August 2003 in the Natural, CT and Opportunistic wetlands.

(Nat= Natural wetland, OPP= Opportunistic wetland, CT= CT wetland).



Appendix C-3. Piper diagram of leached porewaters from soil samples taken at the 0-10 cm depth interval in 2002 and 2003 from Opportunistic and Natural soil transfers and reference plots in the CT wetland.

(CTW Nat= Natural soil plots in the CT wetland, CTW Opp= Opportunistic soil plots in the CT wetland, CTW CT = CT plots in the CT wetland, and CTW Ref= Reference samples taken outside the buckets in the CT wetland)

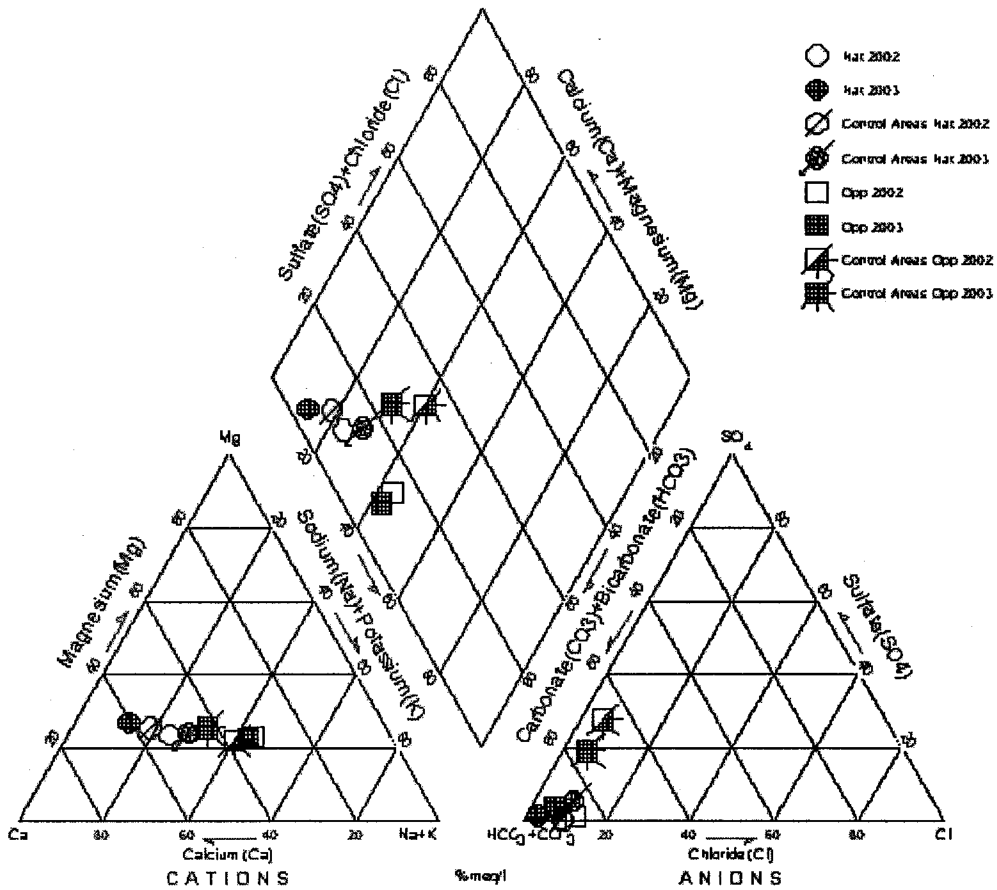


Appendix C-4. Piper diagram of leached porewaters from soil samples taken at the 10-20 cm depth interval in 2002 and 2003 from Opportunistic and Natural soil transfer plots and control areas in the CT wetland.

(CTW Nat= Natural soil plots in the CT wetland, CTW Opp= Opportunistic soil plots in the CT wetland, CTW CT = CT plots in the CT wetland, and CTW Ref= Reference samples taken outside the buckets in the CT wetland)

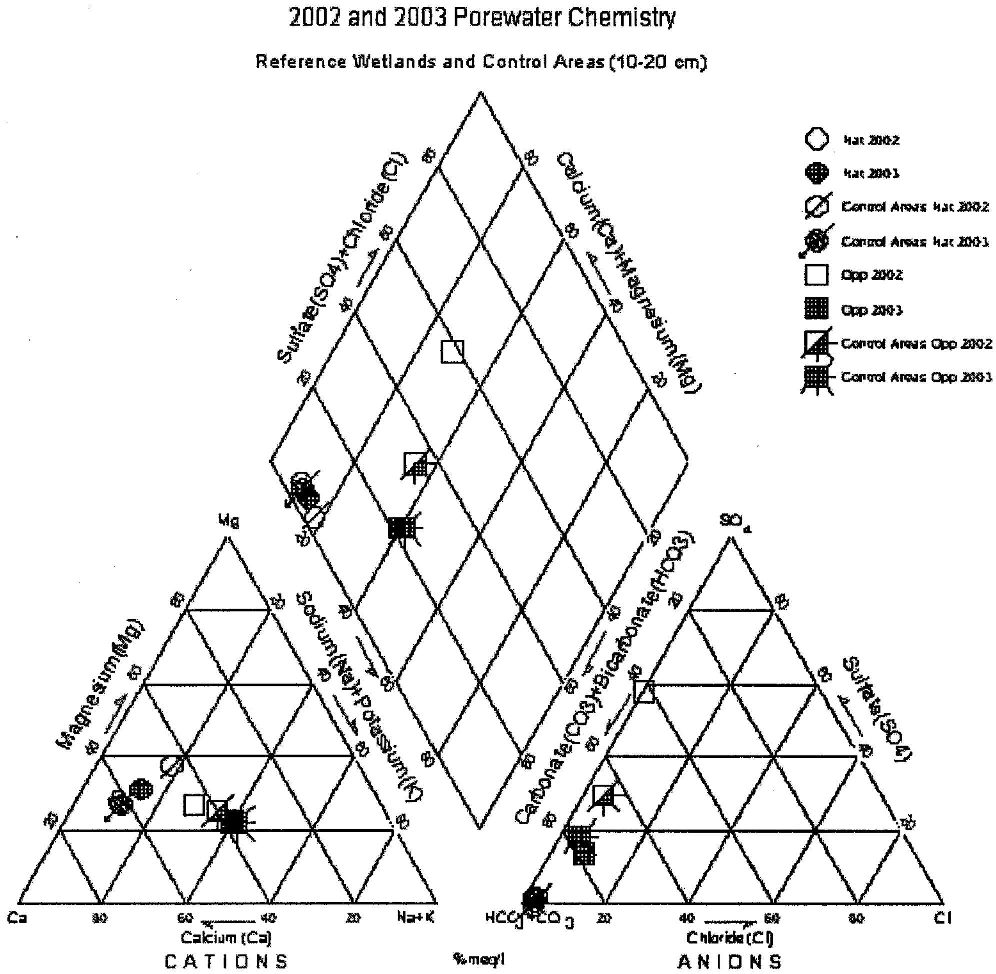
2002 and 2003 Porewater Chemistry

Reference Wetlands and Control Areas (0-10 cm)



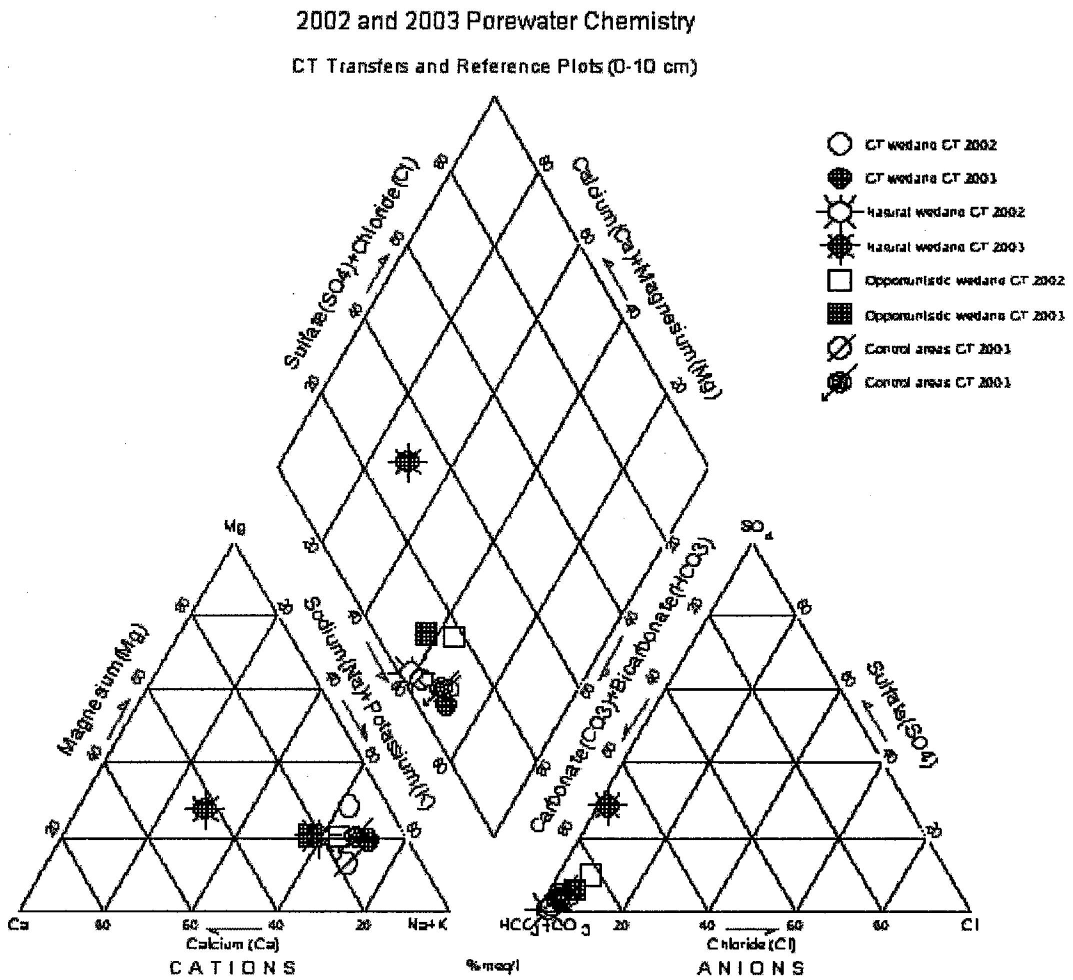
Appendix C-5. Piper diagram of leached porewaters from soil samples taken at the 0-10 cm depth interval in 2002 and 2003 from reference soil plots and control areas in the Opportunistic and Natural wetlands.

(Nat= Reference soil plots in the Natural wetland, Opp= Reference soil plots in the Opportunistic wetland, Control areas Opp= samples taken outside the plot areas in the Opportunistic wetland, Control areas Nat= samples taken outside the plot areas in the Natural Wetland)



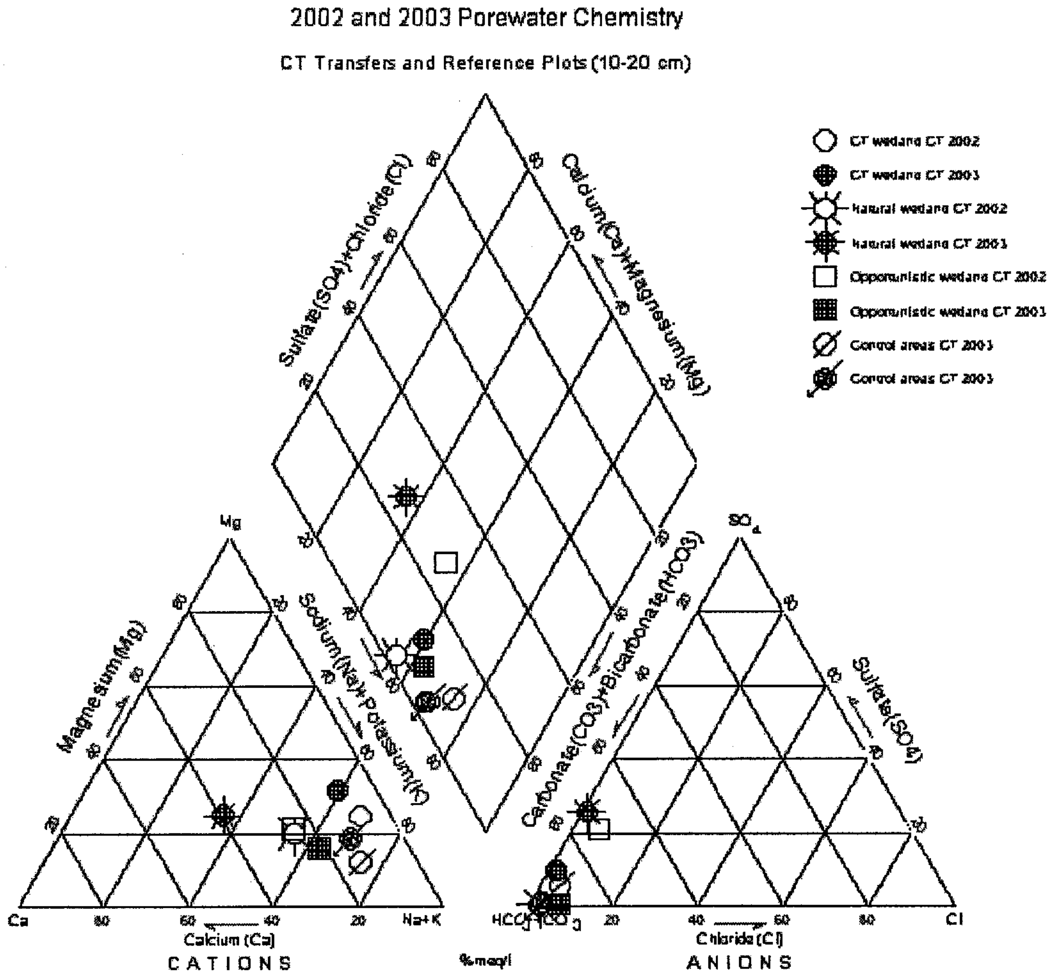
Appendix C-6. Piper diagram of leached porewaters from soil samples taken at the 10-20 cm depth interval in 2002 and 2003 from reference soil plots and control areas in the Opportunistic and Natural wetlands.

(Nat= Reference soil plots in the Natural wetland, Opp= Reference soil plots in the Opportunistic wetland, Control Areas Opp= samples taken outside the soil plots in the Opportunistic wetland, Control Areas Nat= samples taken outside the soil plots in the Natural Wetland)



Appendix C-7. Piper diagram of leached porewaters from soil samples taken at 0-10 cm depths in 2002 and 2003 from CT transfers and reference plots in the Natural, Opportunistic and CT wetlands.

(CTW= CT plots in the CT wetland, OppW CT= CT transfers in the Opportunistic wetland, NatW CT= CT transfers in the Natural wetland, Ref CT= reference samples taken outside the buckets in the CT wetland).



Appendix C-8. Piper diagram of leached porewaters from soil samples taken at the 10- 20 cm depth interval in 2002 and 2003 from CT transfers and reference plots in the Natural, Opportunistic and CT wetlands.

(CTW= CT plots in the CT wetland, OppW CT= CT transfers in the Opportunistic wetland, NatW CT= CT transfers in the Natural wetland, Ref CT= reference samples taken outside the buckets in the CT wetland

Appendix D

Appendix D-1. Metro-Mix 290 Specifications (Terra Lite 2000):

Vermiculite (Cas # 1318-00-9)
Water (Cas # 7732-18-5)
Bark & Related Material
Sphagnum Peat Moss
May contain Quartz (Cas # 14808-60-7)
Gypsum (Cas # 10101-14-4)
Perlite (Cas # 73763-70-3)
Calcium Carbonate (Cas # 1317-65-3)