Overcoming Northern Challenges

Proceedings of the 2013 Northern Latitudes Mining Reclamation Workshop and

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Northern Latitudes Mining Reclamation Workshop

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NORTHERN LATITUDES MINING RECLAMATION WORKSHOP

The Northern Latitudes Mining Reclamation Workshop is an international workshop on mining, land and urban reclamation and restoration methods. The objective of the workshop is to share information and experiences among governments, industry, consultants, Alaska Natives, northern First Nations and Inuit groups which undertake reclamation and restoration projects, or are involved in land management in the north or in comparable environments.

The first Workshop was held in Whitehorse, Yukon Territory, Canada in 2001 and it has been held every two years since, alternating between Canada and Alaska. The primary sponsors of the Workshop include the Yukon Geological Survey, Indian and Northern Affairs Canada, Natural Resources Canada, US Department of the Interior Bureau of Land Management, and the State of Alaska Department of Natural Resources.

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The CLRA/ACRSD is a non-profit organization incorporated in Canada with corresponding members throughout North America and other countries. The main objectives of CLRA/ACRSD are:

- To further knowledge and encourage investigation of problems and solutions in land reclamation.
- To provide opportunities for those interested in and concerned with land reclamation to meet and exchange information, ideas and experience.
- To incorporate the advances from research and practical experience into land reclamation planning and practice.
- To collect information relating to land reclamation and publish periodicals, books and leaflets which the Association may think desirable.
- To encourage education in the field of land reclamation.
- To provide awards for noteworthy achievements in the field of land reclamation.

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PAPERS

PERFORMANCE OF AN ENGINEERED COVER SYSTEM FOR A URANIUM MINE WASTE ROCK PILE IN NORTHERN SASKATCHEWAN AFTER SIX YEARS

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ABSTRACT

The Claude waste rock pile at Cluff Lake uranium mine in northern Saskatchewan's Athabasca basin contains ~7.2 million tonnes of waste rock, upon which an engineered enhanced store-and-release cover system was constructed. The primary design objectives of the cover system are to reduce percolation of meteoric waters into the waste rock pile, attenuate radiation emanating from stored waste to acceptable levels, and provide a growth medium for development of a sustainable vegetation cover. Instrumentation was installed during construction of the cover system to evaluate cover performance under site-specific climate conditions. Field data collected was input to water balances to quantify the volume of net percolation that occurred during the frost-free periods of 2007 to 2012. Based upon these water balances, the Claude waste rock pile cover system is performing as designed based on field monitoring data and observations collected since 2007.

Key Words: Field Performance Monitoring, Water Balance, Net Percolation

INTRODUCTION

Cluff Lake uranium mine, owned and operated by AREVA Resources Canada Inc. (AREVA), is located in northern Saskatchewan's Athabasca basin, approximately 75 km south of Lake Athabasca and 15 km east of the provincial border with Alberta. The mine operated from 1980 to 2002, and decommissioning work began in 2004 following an Environmental Assessment. The majority of decommissioning work was complete by the end of 2006. The project is now in the post-decommissioning and follow-up monitoring stage. AREVA is planning to demolish the last buildings on-site including the existing camp, airstrip facilities, and warehouse in 2013 and 2014. AREVA is awaiting regulatory approval to continue its environmental monitoring program through four site visits per year.

Decommissioning of Cluff Lake mine included placement of a multi-layer cover system over a waste rock pile (WRP) known as Claude WRP. The cover system was completed in 2006, and monitoring of its performance has been on-going since then. This paper reviews the design and construction of the cover system for reclamation of the Claude WRP, and focuses on hydrologic performance of the cover system based on six years of field monitoring data.

BACKGROUND

The Cluff Lake site is situated in a semi-arid environment; the mean annual precipitation and potential evaporation for the region is approximately 450 mm and 600 mm, respectively. Approximately 30% of the annual precipitation occurs as snow. Numerous lakes, swamps and rivers dominate the relatively flat topography of the region.

The Claude WRP was constructed between 1982 and 1989 and contains waste rock from the Claude pit. The pile is approximately 30 m high and covers an area of 26.4 ha to the south of the Claude pit (see Figure 1). It contains approximately 7.23 million tonnes of waste, with an estimated volume of approximately 4.1 Mm^3 based on a dry density of 1,750 kg/m^3.

Figure 1. Photo of the Claude waste rock pile in about 2003 prior to pile re-grading and cover system construction (Claude pit being backfilled in the foreground).

The pile was developed by end-dumping and contains well-developed traffic surfaces between lifts of dumped material. No attempt was made to segregate waste placed in the Claude WRP by chemical composition. The Claude WRP has shown high levels of uranium (200 mg/L) and nickel (43 mg/L) in piezometers around the toe of the pile (COGEMA 2001). This indicates that acid mine drainage is occurring and will continue to occur until the source is depleted. This finding was confirmed and quantified through a detailed waste rock characterization program completed in 1999. AREVA determined that the Claude WRP would be decommissioned in-place, meaning that an engineered cover system would be required for closure.

COVER SYSTEM DESIGN AND CONSTRUCTION

Cover system field trials (test plots) were constructed and instrumented in 2001 on the Claude WRP to examine the construction feasibility and hydrologic behaviour of the preferred cover system design alternative. One test plot (TP#1) was constructed on a relatively horizontal surface, while a second (TP#2) was constructed on a 4H:1V sloped surface. Both test covers had the same profile design, consisting of a 10 cm thick (nominal) reduced permeability layer (RPL) overlain by a 100 cm thick (nominal) layer of local silty-sand till. The RPL comprised weathered waste rock material that was compacted *in situ*; field compaction trials were completed in advance to determine the preferred techniques for RPL construction. Field data were collected and interpreted over a 5-year period including net percolation rates, *in situ* volumetric water content (automated and manual measurements), matric suction (negative pore-water pressure), and temperature of the cover and waste materials. The collected field data were used to aid in calibration of a soil-plant-atmosphere (SPA) numerical model as part of the cover system design process for full-scale WRP decommissioning.

Based on the success of cover system field trials, an enhanced store-and-release cover system was selected as the preferred design for closure of the Claude WRP. The final design included a 20 cm thick (nominal) layer of compacted waste rock overlain by 100 cm (nominal) of non-compacted silty-sand till with a grass and legume vegetation cover. The primary design objectives of the cover system are to:

1. reduce percolation of meteoric waters to attenuate peak concentrations for contaminants of concern in natural watercourses, to levels that can be assimilated without adverse effects to the aquatic ecosystem;
2. attenuate radiation emanating from stored waste to acceptable levels; and
3. provide a growth medium for development of a sustainable vegetation cover.

Decommissioning of the Claude WRP was completed between 2005 and 2006 and involved the following primary work activities:

- Re-contouring the side-slopes to a maximum slope angle of 4H:1V;
- Compacting the WRP surface to meet density specifications over a minimum depth of 0.2 m;
- Placing 1 m (nominal) of local silty-sand till material over the compacted waste rock surface;
- Constructing surface water drainage channels to handle the 24-hour, 100-year design storm event; and
- Applying revegetation seed and fertilizer mixture (a drill seeder as opposed to a hydroseeder was used to minimize erosion of grass seeds and maximize availability of fertilizer to the seeds).

Compaction of the waste rock surface was accomplished using a Caterpillar CS583 roller (AREVA 2007). Generally, two passes were required to meet the required minimum dry density of 95% of Standard Proctor Maximum Dry Density. Due to the unseasonably wet weather encountered during the waste rock compaction effort, moisture conditioning was not required to achieve the specified density. In general, the majority of the re-graded waste rock surface contained sufficient fine-textured materials to produce a relatively smooth surface (see Figure 2). Areas of the WRP that were visually determined to be too open-graded were re-graded to blend in additional fines (with either waste rock or till) and re-compacted. The estimated field saturated hydraulic conductivity of the compacted waste rock layer is 10^{-5} to 10^{-6} cm/s.

Figure 2. Photo of the re-graded Claude WRP being compacted in 2005 prior to till cover placement.

Instrumentation was installed in August 2006 to enable monitoring of the hydrologic performance of the Claude WRP cover system over time under site-specific climate conditions. Field data being collected on the cover system include precipitation, net radiation, runoff, and volumetric water content, matric suction, and temperature of the cover and upper waste rock materials. Figure 3 shows a layout of the instrumentation installed on the Claude WRP cover system. Monitoring stations were located at various slope positions and aspects due to potential differences in cover system performance at these different locations. Also, the monitoring system was automated to the extent possible to avoid missing collection of field response data during key times of the year (e.g., during spring snowmelt and storm events).

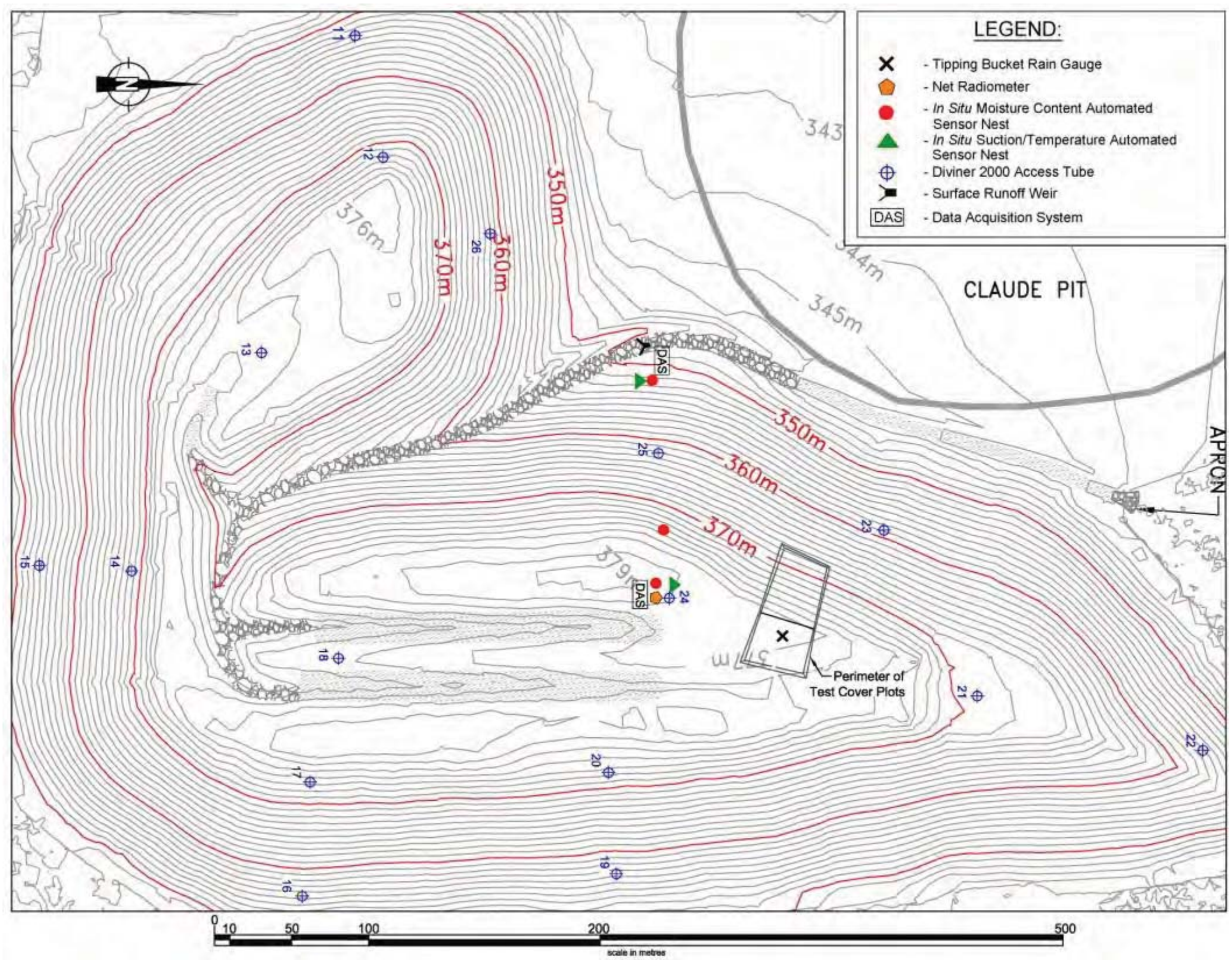

Figure 3. Performance monitoring instrumentation installed on the Claude WRP.

MONITORING PROGRAM RESULTS

Potential evaporation (PE) at the Claude WRP has been greater than rainfall for all years since the onset of monitoring (Table 1). PE estimates are based on the Penman (1948) method and climate data collected at the site, and represent a theoretical maximum evaporation of a free water surface. When PE is greater than rainfall, the capacity for the cover system to store precipitation and release it back to the atmosphere is greatly improved, thereby improving performance of the cover system by reducing net percolation rates.

Table 1. Annual rainfall accumulation and potential evaporation at the Claude WRP from 2007 to 2012.

	2007 (mm)	2008 (mm)	2009 (mm)	2010 (mm)	2011 (mm)	2012 (mm)
Rainfall	299	183	318	296	215	357
PE	533	581	503	550	596	536

Water storage data (Figure 4) have demonstrated that the water balance of the cover system during the year is largely controlled by storage of snowmelt/rainfall and subsequent release of water through evapotranspiration. There was an increase in storage following snowmelt and after rainfall events in the summer months, after which a slow decline occurred. The presence of vegetation improved the cycling of water storage by increasing evapotranspiration rates, further improving cover performance.

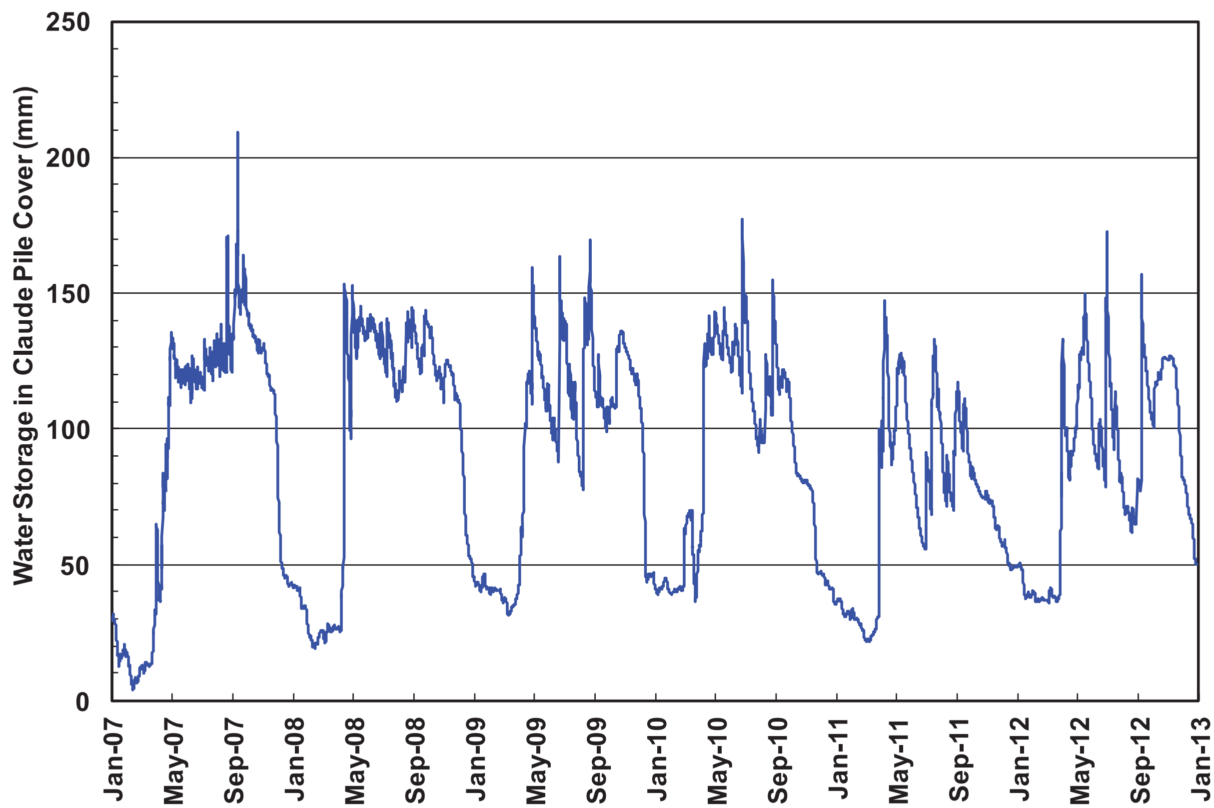

Figure 4. *Soil water storage in the till cover profile measured at the upslope monitoring station from 2007 to 2012 (low storage values during the winter seasons reflect frozen ground conditions).*

Thermal conductivity (TC) sensors are used to monitor temperatures in the Claude WRP cover and upper waste rock profiles at two different locations. Freeze-thaw cycles in the cover profile are important when interpreting runoff data and understanding when net percolation can be expected throughout the year. The timing and rate of freezing and thawing of the cover profile in the fall and spring depends on several factors including snow cover accumulation, ambient temperatures, and soil water contents. In general, the upper cover profile begins to freeze around the end of October, while complete thaw of the cover profile typically does not occur before mid- to late-May.

Performance of the cover system will evolve over time in response to site-specific physical, chemical, and biological processes (INAP 2003). Substantial growth of the various grass and legume species has occurred between August 2007 and June 2012 on the Claude pile cover system (Figure 5), with only minor observed erosion. A more mature vegetation cover will contribute to lower net percolation / seepage volumes through increased interception and transpiration rates.

(a) August 2007 – Plateau Station *(b) July 2008 – Plateau Station* *(c) September 2009 – Plateau Station* *(d) June 2012 – Plateau Station*

Figure 5. Photos illustrating evolution of vegetation on the Claude WRP cover system.

Field measurement data were used to determine net percolation rates through the cover system by using the water balance method:

$$PPT = R + AET + NP + \Delta S + LD \qquad \text{(Eq. 1)}$$

Where

- PPT is precipitation (rainfall plus snow-water equivalent),
- R is runoff,
- AET is actual evapotranspiration,
- NP is net percolation, and
- ΔS is change in moisture storage (all values in mm).

Water balances were estimated on a daily basis during the frost-free period, identified as approximately April 1st to October 31st. Lateral drainage (LD [mm]) is accounted for in sloping systems through an additional term on the right side of Eq. 1. Water balance fluxes at the top and sloping areas of the Claude WRP cover system since the onset of monitoring are given in Tables 2 and 3.

Table 2. Annual water balance fluxes for the plateau area of the Claude WRP for April to October.

Year	PPT (mm)	Water Balance Fluxes (mm and % of precipitation)			
		AET	Δ S	R	NP
2007	450	231 (51%)	34 (8%)	6 (1%)	179 (40%)
2008	272	297 (109%)	-96 (-35%)	6 (2%)	66 (24%)
2009	387	290 (75%)	31 (8%)	5 (1%)	61 (16%)
2010	358	303 (85%)	12 (3%)	2 (1%)	40 (11%)
2011	271	182 (67%)	9 (3%)	2 (1%)	58 (21%)
2012	430	317 (74%)	33 (8%)	5 (1%)	105 (24%)

Table 3. Annual water balance fluxes for sloping areas of the Claude WRP for April to October.

Year	PPT (mm)	Water Balance Fluxes (mm and % of precipitation)				
		AET	Δ S	R	LD	NP
2007	419	239 (57%)	17 (4%)	58 (14%)	0 (0%)	104 (25%)
2008	261	308 (118%)	-85 (-33%)	50 (19%)	-57 (-22%)	45 (17%)
2009	396	314 (79%)	15 (4%)	41 (10%)	0 (0%)	26 (7%)
2010	371	320 (86%)	4 (1%)	22 (6%)	-21 (-6%)	46 (12%)
2011	295	231 (78%)	3 (1%)	19 (6%)	25 (9%)	44 (15%)
2012	422*	310 (73%)*	51 (12%)*	24 (6%)*	0 (0%)	92 (22%)*

*Upslope data only.

As was observed when calculating the 2012 water balance, the timing of the precipitation can be of consequence to the amount of net percolation. If storm events occur in September and October, when evapotranspiration is no longer available to remove stored water from the cover system, greater than expected net percolation can occur. In addition, if the entire soil profile does not completely freeze, the water at the base of the cover system can continue to percolate into the underlying waste, further causing conditions for higher than expected net percolation.

In general, net percolation has decreased since construction of the cover system in 2006. This is likely due to an increase in vegetation cover causing an increase in evapotranspiration rates. However, variability in net percolation as a percentage of precipitation did occur during the six years of monitoring, which is to be expected in response to normal cycles in the local climate. General trends on the performance of the system cannot be inferred from a single year of monitoring data; it is only when examining net percolation over the long term and in the overall context of normal climate variability that trends in performance can be determined. Natural climatic variability is to be expected and was accounted for during the design of the Claude WRP cover system.

CONCLUSIONS

The Claude WRP cover system is a stable landform supporting the growth of productive native plant species and attenuating radiation emanating from stored waste rock to acceptable levels. Percolation of meteoric waters through the WRP has generally decreased since construction of the cover system in 2006. This case study serves to underline the importance of maintaining a long-term perspective when evaluating cover system performance in terms of reducing the net percolation of meteoric waters.

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INNOVATIVE CONCEPTS USED DURING THE REMEDIATION AND RECLAMATION PLANNING OF A SULPHUR HANDLING FACILITY

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ABSTRACT

Husky Oil Operations Limited's (Husky) Windfall bulk sulphur handling plant is 40 hectares in size and is located southeast of Fox Creek, Alberta. The Plant was opened in 1965 and operated for more than 40 years. As the source of sulphur to the Plant has declined with time, Husky has decided to decommission, abandon, remediate and reclaim the site.

Over its history, the site has contained: a gas well, a prilling facility, sulphur storage and handling areas, block storage areas, water retention and treatment ponds, a waste pit and several sulphur burial areas. Furthermore, aerial deposition, block fires and molten spills of sulphur have occurred around the site. The result of this history is a large volume of acidic, sulphur impacted material that must be handled in a responsible manner.

As Husky has moved through the regulatory process that is required to successfully reclaim this site, several lessons have been learned. A collaborative approach with the regulatory organizations has allowed Husky to develop innovative solutions. These solutions include a site specific risk assessment, stratified remediation, an on-site landfill, re-definition of waste, and location specific reclamation end-points.

This presentation will aim to convey Husky's most significant lessons learned throughout the regulatory assessment and planning processes.

FARO MINE COMPLEX REVEGETATION ACTIVITIES

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ABSTRACT

In preparation for remediation and closure of the Faro Mine Complex (FMC), significant work has been completed to develop and test revegetation methods at the mine site. Development and implementation of various revegetation field trials and strategies has been occurring since 2007 to gain knowledge on successful revegetation techniques at this challenging northern site. This paper focuses on revegetation efforts and subsequent monitoring at the Grum Overburden slope and Grum Sulphide Cell (GSC).

Grum Overburden site revegetation activities involved the implementation of trials covering approximately two hectares on soils to be used as a reclamation cover. Different grass seed mixes were applied with and without fertilizer, and woody species (alder, willow, and poplar) were planted in the seeded plots. Four years of monitoring have demonstrated that revegetation success was primarily dependant on fertilization and erosion protection was heavily influenced by site preparation method.

Revegetation at the GSC will provide ground cover for erosion protection and develop a long-term, self-sustaining system integrated with the mine surroundings. Building on results from Grum Overburden site, the 2012 revegetation prescriptions included hydroseeding, fertilization, planting of woody species, and testing of fertilization and hydration-paks. Early monitoring results from the summer of 2013 will provide insight on the success of the revegetation works at the GSC site.

Key Words: Fertilizer; Nurse and Native Seed; Surface Treatment; Teabags; Hydroseeding; Grum.

INTRODUCTION

Government of Yukon – Assessment and Abandoned Mines (YG-AAM) branch is responsible for planning for remediation and closure of the Faro Mine Complex (FMC), a large abandoned lead, zinc, silver and gold mine site in central Yukon. In preparation for closure, significant work has been completed to develop and test revegetation methods at the mine site. Establishing vegetation in FMC soils requires some initial care and maintenance to ensure success. The short growing season, nutrient poor soils and the physical properties of the soils all combine to make establishment of vegetation at the FMC challenging. Revegetation must advance quickly to achieve the short-term goal of reducing soil erosion but also must also consider of longer term goals such as allowing for natural succession trajectories.

Revegetation activities and trials have been conducted by EDI Environmental Dynamics Inc. (EDI) at two sites at the FMC: the Grum Overburden site and the Grum Sulphide Cell (GSC). Grum Overburden trials (~2 ha) were designed to identify the most effective combination of surface treatments and revegetation options to establish vegetation and mitigate erosion. The GSC revegetation activities (~26 ha) were conducted using knowledge gained from outcomes at the Grum Overburden trials. Additional variation/experimentation was included to further refine, evaluate and provide additional direction for future revegetation practices at the FMC.

This paper provides an overview of revegetation activities conducted and results and observations from the Grum Overburden trials and preliminary observations from GSC revegetation treatments implemented in 2012.

METHODOLOGY

Grum Overburden Trials

Grum Overburden re-vegetation trials established in 2009 include the following variables:

- 3 grass seed mixes (agronomic, native, and nurse and native; Table 1);
- 3 woody plant treatments (horizontal and vertical stakes and alder seedlings);
- some seeding of spruce and dwarf birch collected at site;
- 2 fertilizer treatments (unfertilized and fertilized [8-38-15], 400 kg/ha and 200 kg/ha for first year and second year, respectively); and
- 3 soil surface treatments (micro-rill, planar, and rough-and-loose).

Table 1. Summary of seed mix and application rates

Seed Mix and Application Rate		Seeds by Weight	
Agronomic	(40 kg/ha)	Red Fescue (Arctared)	15%
		Meadow Foxtail (Common)	11%
		Kentucky Bluegrass (Nugget)	5%
		Slender Wheatgrass (Adanac)	49%
		Alsike clover (Common)	20%
Native	(29 kg/ha)	Slender Wheatgrass	10%
		Northern (Rocky Mountain) Fescue	20%
		Glaucous Bluegrass	37%
		Tufted Hairgrass	33%
Nurse and Native	(33.5 kg/ha)	Slender Wheatgrass	14%
		Northern (Rocky Mountain) fescue	27%
		Glaucous Bluegrass	58%
		Barley	0.5%

The study randomized the seed mix; woody plant; fertilizer; and soil surface treatment variables as follows (Figure 1):

- Seed mixes within each surface treatment;
- Fertilizer treatments within each seed mix treatment;
- Woody plant treatments within fertilizer treatments.

Surface Treatment	Planar						Micro Rills						Rough-and-loose					
Fertilizer	x			x	x			x	x			x	x			x	x	
Distance from toe of slope / Seed Mix:	agronomic		nurse + native		native		native		agronomic		nurse + native		nurse + native		native		agronomic	
70 m																		
60 m	horiz. 7	alder 14	vert. 21	horiz. 28	vert. 35	alder 42	alder 49	vert. 56	alder 63	vert. 70	horiz. 77	alder 84	vert. 91	horiz. 98	alder 105	alder 112	horiz. 119	vert. 126
50 m	not planted 6	horiz 13	alder 20	vert. 27	alder 34	vert. 41	vert. 48	alder 55	vert. 62	horiz. 69	vert. 76	horiz. 83	alder 90	alder 97	horiz. 104	horiz. 111	alder 118	vert. 125
40 m	alder 5	vert. 12	horiz. 19	vert. 26	horiz. 33	horiz. 40	horiz. 47	horiz. 54	horiz. 61	alder 68	alder 75	vert. 82	horiz. 89	vert. 96	alder 103	vert. 110	vert. 117	alder 124
30 m	white spruce seed, alder & horiz.	white spruce seed, alder & horiz.	white spruce seed, alder & horiz.	white spruce seed, alder & horiz.	white spruce seed, alder & horiz.	white spruce seed, alder & horiz.	dwarf birch, alder, & horiz.	dwarf birch, alder, & horiz.	dwarf birch, alder, & horiz.	dwarf birch, alder, & horiz.	dwarf birch, alder, & horiz.	dwarf birch, alder, & horiz.	alder and horiz.	alder and horiz.	alder and horiz.	alder and horiz.	alder and horiz.	alder and horiz.
20 m	horiz. 3	alder 10	vert. 17	alder 24	vert. 31	horiz. 38	alder 45	horiz. 52	horiz. 59	vert. 66	alder 73	vert. 80	alder 87	vert. 94	horiz. 101	vert. 108	horiz. 115	alder 122
10 m	Alder 2	horiz. 9	alder 16	horiz. 23	horiz. 30	vert. 37	horiz. 44	vert. 51	vert. 58	alder 65	horiz. 72	alder 79	vert. 86	horiz. 93	vert. 100	alder 107	vert. 114	horiz. 121
Toe of slope = 0 m	vert. 1	vert. 8	horiz. 15	alder 22	Stakes 29	alder 36	vert. 43	alder 50	alder 57	horiz. 64	vert. 71	horiz. 78	horiz. 85	alder 92	vert. 99	horiz. 106	alder 113	horiz. 120
Distance along toe of slope	90m	75m	60m	45m	30m	15m	90m	75m	60m	45m	30m	15m	78m	55m	42m	39m	26m	13m

Figure 1. *Grum Overburden Dump trial layout. 3 surface treatments; 3 seed mixes. Half of each seeded section was fertilized. Woody species planted in blocks throughout the site (alder = alder seedlings; horiz = willow and poplar stakes planted horizontally; and vert. = willow and poplar stakes planted vertically).*

During monitoring from 2010 to 2012, percent vegetation cover was estimated visually in each plot. In 2011 and 2012, percent cover was estimated using a rectangular quadrat (0.5 m^2), six subsamples per plot, and calculating average percent cover.

All live stems of poplar, willow, alder, and other woody plants were also counted in each plot. Live stems are defined as any shoot, or cluster of shoots, emerging independently of other shoots.

<u>Grum Sulphide Cell Revegetation</u>

The Grum Overburden trials focused on grass mixes, physical surface treatments, fertilizer effects, and inter-planting woody and herbaceous species. Key treatments in the 2012 GSC revegetation program included the following treatments:

- Surface treatment similar to rough and loose, ripping across the slope
- Nurse and native seed mix (hydroseed)
- Fertilizer (included in hydroseeding)
- Planting woody species
    - Plugs
    - Horizontally staked willow and poplar cuttings
- Fertilizer teabags

YG-AAM was responsible for overseeing and implementing the soil surface preparation. Prior to planting, a D7 Caterpillar dozer using three rippers on the back excavated furrows of 10 – 30 cm depth perpendicular to the slope. Soil surface preparation was completed between July 9 and August 8, 2012.

The GSC was hydroseeded from August 11 – 13, 2013. A native seed mix was used with annual rye grass as a nurse crop. The seed mix consisted of the following grasses:

- 54.1% Slender Wheatgrass 19 kg
- 17.1% Northern (Rocky Mountain) Fescue 6 kg
- 0.5% Glaucous Bluegrass 0.185 kg
- 7.1% Tufted Hairgrass 2.5 kg
- 21.2% Annual Rye Grass 7.5 kg

Experience from hydroseeding on site indicated nurse crops, if applied properly, have immediate effective response that helps address short term revegetation goals. The native mix allows for better establishment over the long term. The seed was applied at a rate of 35 kg/ha over the 26 ha GSC. Additives to the hydroseed mix included fertilizer and mulch/tackifier mix (Hydrostraw® Guar Plus).

The key driver to revegetation cover at the Grum Overburden site was fertilizer. Fertilizer used at the Grum Overburden trial was 8-38-15; limited or no vegetation grew in areas with no fertilizer application. Nitrogen was found to be very low in all GSC soil nutrient analysis results and a higher nitrogen fertilizer composition (18-18-18; 400 kg/ha) was more appropriate for the site and for the establishment of native grasses.

Experimental plots were set up on the GSC to study further refine, evaluate, and provide additional direction for future revegetation practices at the FMC. The study was set up as a complete randomized design with six treatments and seven replicates, resulting in 42 experimental plots; each experimental plot measures 30 m x 50 m. Plots are marked out with wooden posts and plots labeled with tags on the northeast posts of the plots.

In preparation for general planting works at the FMC, locally collected dwarf birch (*Betula nana* L.) and alder (*Alnu*s sp.) seed, as well as cuttings from locally collected balsam poplar (*Populus balsamifera* L.) and willow (*Salix* sp.) were sent to Peel's Nurseries Ltd., Mission, BC, in March 2011 for propagation. The propagated plugs were shipped to FMC in August of 2012 and included the following: Alder (2,660), Birch (4,320), and Poplar and Willow (8,190). Planting of native woody vegetation plugs at the GSC is not only for long term erosion control and site stabilization, but also to initiate the restoration of the site to a more natural state, along local successional trajectories.

The propagated plugs were supplemented by the on-site collection of dormant poplar and willow stakes prior to planting horizontally at the GSC. Horizontal staking has been successful at surviving in the less than ideal soil conditions at the FMC and can produce multiple stems per cutting compared to the one stem a vertical stake produces. Horizontal staking also has the potential to assist in the reduction of erosion due to positioning the stake across the slope, slowing runoff moving down the slope.

Over 14 ha of the 26 ha GSC were planted with woody species in either experimental plots or general planting plots. One entire slope and areas outside of the plots were left as a control to compare revegetation establishment without the planting of woody vegetation; these areas were hydroseeded and fertilized consistent with all areas. Experimental treatments also included three teabag products intended to improve establishment of plugs in disturbed areas.

Specific experimental treatments, outlined in Figure 2, are as follows, with additional details on each in the sections below:

- Plugs only (P plots);
- Stakes only (S plots);
- Plugs and stakes (PS plots);
- Plugs with Fertilizer-pak (Chilcotin blend) teabag (Pf);
- Plugs with Hydration-pak teabag (Ph);
- Plugs with Genesis-pak teabag (Phf).

Plots not containing stakes were planted with 375 plugs: birch (116), alder (72), and balsam poplar/willow (187). Plots with plugs and stakes were planted with birch (58), alder (36), balsam poplar/willow plugs (94), and balsam poplar/willow stakes (~188). Stakes only plots were planted with approximately 400 stakes of a mix of balsam poplar and willow. The density of the plots with plugs (P plots) was 2,500 plugs per hectare and plots with stakes (S plots) was approximately the same, 2,500 stakes and/or plugs per hectare. Some mortality is expected.

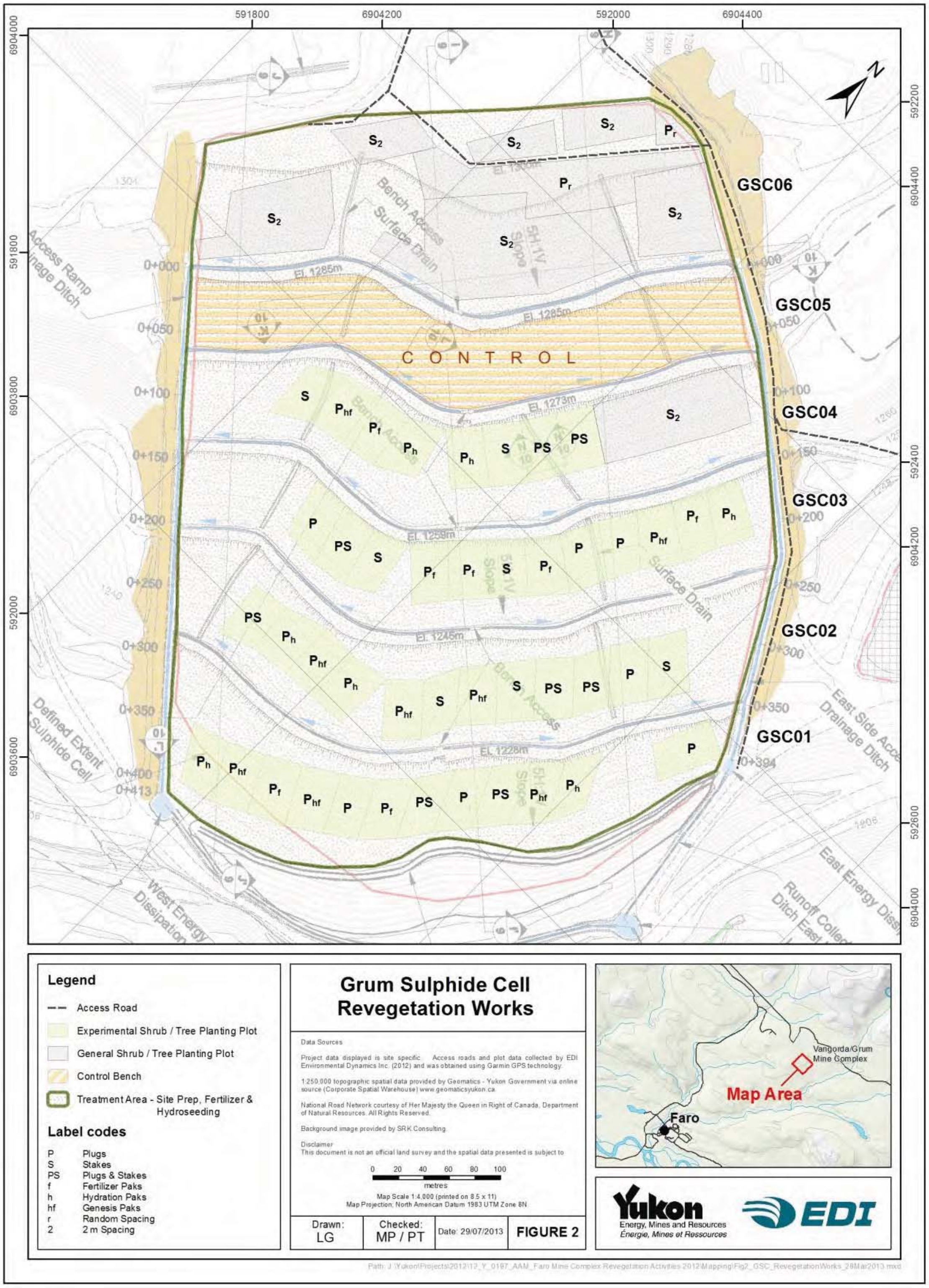

Figure 2. Grum Sulphide Cell Revegetation Works.

Plugs Only (P plots)
Woody vegetation plugs were planted without any amendments. The 'plugs only' treatment tests plug success and also acts as a control for the treatments involving teabags.

Staking (S Plots)
Balsam poplar and willow species stakes were collected at the FMC, near the GSC, within days of planting. Stakes were cut in September as plants were going into dormancy and stored in a ponded area east of the GSC. Stakes were bundled and soaked in water for one to two days prior to planting. Stakes were cut to ~45 cm in length and laid horizontally in shallow trenches that were dug perpendicular to the slope, and backfilled at approximately 10 to 20 cm below the ground surface.

Plugs and Staking (PS plots)
Stakes were planted alternately with plugs, both across and with the slope.

Fertilizer-Paks (Pf plots)
Chilcotin blend teabags (17-5-7) are used for disturbed planting sites with low levels of organic matter. One 10-gram teabag was placed in a separate hole, immediately upslope of the plug, about 5 cm below the soil surface. Placing the Chilcotin teabag in the separate hole prevents the fertilizer from burning the plug roots. The Chilcotin blend teabag releases the following ingredients over twelve months: total nitrogen 17%; available phosphate 5.0%; soluble potash 7.0%; magnesium 1.2%; sulfur 10.4%; humic acid (leonardite-derived) 4.5%; kelp extract (*Ascophyllum nodosum* (L.) Le Jolis) 4.0%; co-polymer of acrylamide 4.0% (intended to help retain moisture during periods of reduced soil moisture).

Hydration-paks (Ph plots)
Hydration-pak teabags (16-8-5) are similar to the Chilcotin-pak fertilizer, but they also contain more moisture-retaining polymer to assist seedlings with establishment during times of moisture stress. One 10-gram hydration-pak teabag was placed in the planting hole with the plug. Placing the hydration-pak in the same hole is possible because the moisture retaining polymer swells and protects the roots from the fertilizer. The hydration-pak includes the following ingredients: total nitrogen 16.0%; available phosphate 8.0%; soluble potash 5.0%; sulphur 6.6%; co-polymers of acrylamide 19.0%.

Genesis-pak (Phf plots)
One 40-gram Genesis-pak was placed in the planting hole and broken open prior to planting the plug. Genesis-paks release the following ingredients over twelve months: total nitrogen 8.0%; available phosphoric acid 5.0%; soluble potash 5.0%; sulfur 2.0%; boron 0.037%; humic acid (leonardite-derived) 7.5%; kelp meal (*Ascophyllum nodosum* (L.) Le Jolis) 5.0%; composted vegetative matter 30%.

General Shrub/Tree Planting Plots (S2 plots; Pr plots)
In addition to the experimental plots, larger plots were set up in areas to add woody plant material treatments on a larger scale. Poplar and willow stakes were buried at a spacing of 2.5 m (S2 plots). Approximately 3,350 plugs remained after all experimental plots were planted and were planted on the uppermost slope in a random arrangement (Pr plots). Some areas were planted at very high densities to simulate a more natural setting; other areas were planted with regular 2.5 m spacing.

RESULTS AND DISCUSSION

Grum Overburden Trials

Monitoring of Grum Overburden trials found that fertilizer application is necessary to establish herbaceous vegetation at the FMC. Unfertilized plots had minimal growth and all other treatments were secondary to the application of fertilizer. Results from 2012 indicated that all seed mixes and surface treatments resulted in equally successful vegetation coverage over time in the fertilized plots.

All seed mixes demonstrated increased coverage in the third year after seeding; however, if the objective is to have more immediate coverage, the appropriate seeding may need to contain more nurse crop seed; low amounts of barley (0.168 kg/ha) were applied at the Grum Overburden site. In 2010 and 2011, the native seed mix provided the least cover, suggesting that the nurse crop seed is beneficial for the success of native species establishment. In 2012, the native seed mix provided the same amount of vegetation cover as the other two mixes indicating the native species needed more time to establish. The ratio of nurse crop and native seed that will provide the best balance between immediate cover and sustainability is unclear; however, recent work at the GSC may provide additional insight.

Regarding woody plant establishment, it is unclear whether fertilizer application influenced success of willow/poplar stakes; however, alder seedlings showed a negative response to fertilizer application, likely due to competition with herbaceous vegetation. Results also showed consistently higher stem counts for horizontal staking in contrast to vertical staking.

Of all the surface treatments, the most effective at controlling erosion due to run-off was the rough-and-loose surface treatment, even in the areas with minimal vegetative cover. The rough-and-loose surface treatment at the Grum Overburden site created micro-sites providing areas for pooling water, ideal for germinating seeds. The micro-sites in the depressions were areas of dense grass growth to the point where it may have negatively affected woody plant growth.

The soils and the climate at the former Faro Mine site make revegetation challenging. The amount of vegetation cover established during these trials was not at a level that will meet the objectives of controlling erosion and stabilizing slopes alone. Nonetheless, the Grum Overburden trial results indicate that selection of proper site preparation methods can help address erosion issues at the site and thus provide more time for vegetation to establish. The rough-and-loose treatment with seeding, horizontal staking and fertilization is a method that appeared to achieve the goals of revegetation and controlling erosion. Additional work on seed mixes and fertilization ratios/rates may further improve overall revegetation cover.

Grum Sulphide Cell

Monitoring and data collection will be conducted at the GSC in July/August 2013 and as such preliminary results will be presented during the Northern Latitudes workshop.

Monitoring of the 2012 GSC revegetation activities will be conducted over multiple years to gather data to determine revegetation success, site stability, most effective methods, and if trial objectives have been met. The following activities will take place each year as components of the GSC monitoring program.

- Soil sampling;
- Photo point monitoring;
- Estimation of herbaceous vegetation percent cover;
- Determination of woody vegetation densities;
- Assessment of woody vegetation health and vigor.

In addition to the suggested monitoring activities, general observations of treatments and the site will be recorded, including any observations of natural recruitment of native and non-native species at the GSC.

BIOREMEDIATION IN NORTHERN CLIMATES: HOW CAN PETROLEUM HYDROCARBON BIODEGRADATION BE ASSESSED IN SOILS UNDER SEASONAL FREEZE-THAW CONDITIONS?

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ABSTRACT

Accurate assessment for bioremediation feasibility plays a critical role in developing cost-effective and efficient remediation strategies adapted to northern climates, where warm seasons are short and accessibility is limited. Using a pilot-scale bioremediation system, this study quantitatively assessed bioremediation feasibility under dynamic site temperatures, including summers and non-summers, at a northern contaminated site. A total of 15 first-order rate constants were derived from previous pilot-scale experiments carried out under dynamic or fixed average temperatures of a cold site. The biodegradation rate obtained from pilot-scale testing was two times slower than that of the microcosm-scale experiments, which have been widely used for laboratory-based assessments. Site temperature variability and seasonality at cold sites greatly influences biodegradation kinetics. Using pilot-scale experiments under site-representative freeze-thaw temperatures, the unfrozen water content, which is a requirement for microbial activity in freezing soils, was predicted based on a TEMP/W model. The simulated unfrozen water content was in excellent agreement with the measured unfrozen water content during soil freeze-thaw. The correlation between unfrozen water content and microbial CO_2 production was statistically significant. This study suggests that on-site bioremediation for cold site soils is feasible in summers and can be extended to seasonal freeze-thaw months.

Key Words: Bioremediation, Petroleum Hydrocarbons, Seasonal Soil Freeze-Thaw, Unfrozen Water Content, Pilot-Scale Experiment, Biodegradation Rate, Cold Climates.

INTRODUCTION

The remediation of contaminated sites in northern Canada is a pressing environmental concern. An estimated 2,600 contaminated sites exist in the three northern territories (Northwest Territories, Nunavut, and Yukon) and 182 of these are classified as high priority sites in terms of environmental and health risks associated with the contaminants (Federal Contaminated Sites Inventory 2013). Federal government liability has been estimated at $1.5 billion for northern contaminated sites, which typically include mine sites, drilling sites, abandoned military stations, oil spill sites, and non-managed disposal sites (Office of the Auditor General of Canada 2012). At these sites, petroleum hydrocarbons are some of the most frequently identified contaminants. Petroleum hydrocarbons, which are hydrophobic organic compounds, including toxic and carcinogenic hydrocarbon fractions, are persistent in northern cold soils and serve as long-term contaminant sources for inhabitants and unique northern ecosystems.

Bioremediation is the use of microorganisms to degrade or detoxify contaminants in the environment and has been considered a cost-effective and non-destructive remediation technology for petroleum hydrocarbon-contaminated soils in northern cold regions. Cold-adapted, indigenous, hydrocarbon-degrading bacteria are able to survive and remain metabolically active in petroleum hydrocarbon-contaminated soils and groundwater at low temperatures, and thus have been the focus of bioremediation efforts at contaminated sites in cold regions, including the Arctic and Antarctica (Snape et al. 2008).

However, since many of the contaminated sites in northern climates are located in remote regions and site accessibility is limited during seasonal freezing and thawing months, assessing the bioremediation feasibility for hydrocarbon-contaminated soils still largely relies on controlled, laboratory-based biodegradation experiments. The costs of risk management, remediation and monitoring are prohibitively increased by cold climate seasonality, associated logistics and the environmental sensitivity of frozen ground. An accurate assessment of contaminated soils is required for enhancing hydrocarbon biodegradation and for implementing cost-effective and successful bioremediation at remote northern sites. Understanding site-specific environmental conditions and rate-limiting factors such as site temperature dynamics and the scale of the bioremediation system should be considered during the design of laboratory experiments.

The majority of previous laboratory-based assessments about optimal soil and nutrient conditions for enhancing indigenous hydrocarbon-degrading bacteria have been generally performed at microcosm scales (3 to 1200 g). However, larger field-scale or on-site pilot-scale bioremediation systems (e.g., 3600 m^3) at cold sites are likely to be influenced by greater soil heterogeneity. Although microcosm-scale experiments are useful for indicating the bioremediation potential of contaminated soils, the system scale may significantly influence the rate and extent of hydrocarbon biodegradation (Davis et al. 2003).

Furthermore, previous bioremediation feasibility studies have been predominantly performed at fixed, low temperature incubations, with temperatures ranging from near 0 to +25°C, which is not representative of realistic, periodic and seasonal temperature regimes at northern sites. Therefore, this study will investigate the influence of system-scale and site-representative temperature dynamics on the rate and extent of hydrocarbon biodegradation in a cold site soils.

On the other hand, current remedial strategies and on-site activities, regardless of the type of remediation technology, have often been designed to be effective only during short summers (2 to 4 months per year) due to the colder climate, slower degradation rates and limited site accessibility (Chang et al. 2011a; Paudyn et al. 2008). On-site remediation systems are left dormant after short treatment seasons when seasonal freezing begins. However, Chang et al. (2011a) showed that hydrocarbon biodegradation did not cease at sub-zero temperatures during soil freezing and thawing at seasonal soil temperature changes, which potentially provides important insight for understanding biodegradation activity in semi-frozen and frozen contaminated soils and for developing bioremediation strategies that are feasible during seasonal freeze-thaw months. Changes in unfrozen water availability become a critical rate-limiting factor for maintaining hydrocarbon biodegradation in semi-frozen and frozen hydrocarbon-contaminated soils.

In this study, unfrozen water availability was predicted using a numerical model (TEMP/W) and compared to measured unfrozen water content in the soil of a pilot-scale tank subjected to seasonal freeze-thaw temperatures. The correlation between unfrozen water content and microbial CO_2 production was investigated.

In summary, the specific objectives of the present study are to (1) compare the rates and extents of hydrocarbon biodegradation for microcosm and pilot-scale experiments, (2) determine favorable nutrient dosages for enhancing microbial activity in contaminated soils, (3) compare the rates of hydrocarbon biodegradation under fixed and dynamic temperature regimes, including seasonal freeze-thaw conditions, and (4) simulate unfrozen water availability during seasonal freeze-thaw and assess its correlation with microbial activity. Field-aged petroleum hydrocarbon-contaminated soils were shipped from a Resolution Island site in Nunavut. Microcosm- and pilot-scale bioremediation experiments were conducted in a large-scale cold room in order to program site-representative temperature dynamics, including seasonal freeze-thaw temperature regimes.

MATERIALS AND METHODS

Contaminated Soils

Unsaturated petroleum hydrocarbon-contaminated soils were shipped from a Resolution Island (RI) site (61°30'N 65°00'W) in Nunavut. The soils were classified as sand (gravel: 27%; sand: 71%; silt and clays: 1%) and the initial pH of site soils was 4.3. In accordance with four carbon-number-based fractions (F1-F4) from the CWS PHC (Canada-Wide Standard for Petroleum Hydrocarbons in Soils), the majority of hydrocarbon contaminants in the soils were semi-volatile (F2: >C10-C16) and non-volatile (F3: >C16-C34). Volatile (F1: C6-C10) and heavier hydrocarbon fractions (F4: >C34) were negligible. An abundance of viable, aerobic hydrocarbon-degrading bacteria (2×10^2 to 1×10^5 CFU/g) were enumerated in the soils at incubation temperatures ranging from 4 to 25°C. Catabolic genes (*AlkB*, *ndoB*, *phnAc*, and *xylE*) encoding for petroleum hydrocarbon biodegradation were detected in the contaminated soils. The feasibility of aerobic biostimulation in field-aged hydrocarbon contaminated soils from the site was therefore indicated. Many of the results from physical, chemical and microbiological characterizations of the RI soils were reported in Chang et al. (2010).

Microcosm- and Pilot-Scale Bioremediation Experiments

Nutrient-amended and unamended (control) soils were prepared at both the microcosm and pilot scales. The weights of wet unsaturated contaminated soils used in the microcosm- and pilot-scale experiments were 0.5 and 300 Kg, respectively. The microcosm experiments were conducted using 1-L glass jars. For the scaled-up studies, the pilot-scale tanks (1 m long × 0.65 m wide × 0.3 m high) equipped with activated carbon and moisture traps to capture volatile hydrocarbons were placed in a large-scale cold room. The temperature programs for site temperature scenarios were constructed based on the 20-year climatic data available from Environment Canada.

The soil treatment included nutrient amendment and soil buffering. Water-based, 20:20:20 commercial fertilizer (20% total N: 20% P_2O_5: 20% K_2O) was prepared to make samples with 50, 100 and 250 milligrams of nitrogen per kilogram of wet soil in order to determine the favorable loading of nutrients for the site soils. The site soils were acidic and buffered by adding 2000 mg $CaCO_3$/Kg. Soil pore gas concentrations of carbon dioxide (CO_2) and oxygen (O_2) were monitored using a gas monitor

(ATX 620, Industrial Scientific Co.). The soil samples were collected at intervals of 20 days and CWS PHC analyses were adopted for petroleum hydrocarbon analyses (CCME 2001). The detailed analytical protocols are described elsewhere in Chang et al. (2011a). The 15 rate constants for hydrocarbon biodegradation were determined based on a first-order kinetic model. Of these, nine rate constants were derived from previous series of pilot-scale experiments (Chang et al. 2010, 2011a,b).

The change in unfrozen water availability was simulated using TEMP/W, which is a commercial numerical modeling software (GEO-SLOPE 2008). Unfrozen water content curves in response to soil freeze-thaw were generated for the input data. The measured data for unfrozen water content and CO_2 production during soil freeze-thaw were derived from Chang et al. (2011a). The simulated and measured unfrozen water contents were compared and the correlation between CO_2 production (i.e., microbial activity) and unfrozen water content was evaluated.

RESULTS AND DISCUSSION

Effects of the Scale of the System on Biodegradation Rates for Petroleum Hydrocarbons

The present study indicated a significant difference in the bioremediation performance between the microcosm- and pilot-scale experiments. It is notable that TPH biodegradation was enhanced more easily at the microcosm scale (500 g) than in the pilot-scale system (300 Kg) at 5.5°C. Using a first-order kinetic reaction model, the rate constants, *K*, were determined from TPH data for both systems. As shown in Figure 1, the $K_{microcosm}$ of 0.01 day^{-1} was two times higher than the $K_{pilot\text{-}scale}$ of 0.005 day^{-1}. The calculated half-life for TPH biodegradation was 44 to 68 days (95% confidence interval) at the microcosm scale and 99 to 202 days (95% confidence interval) at the pilot scale.

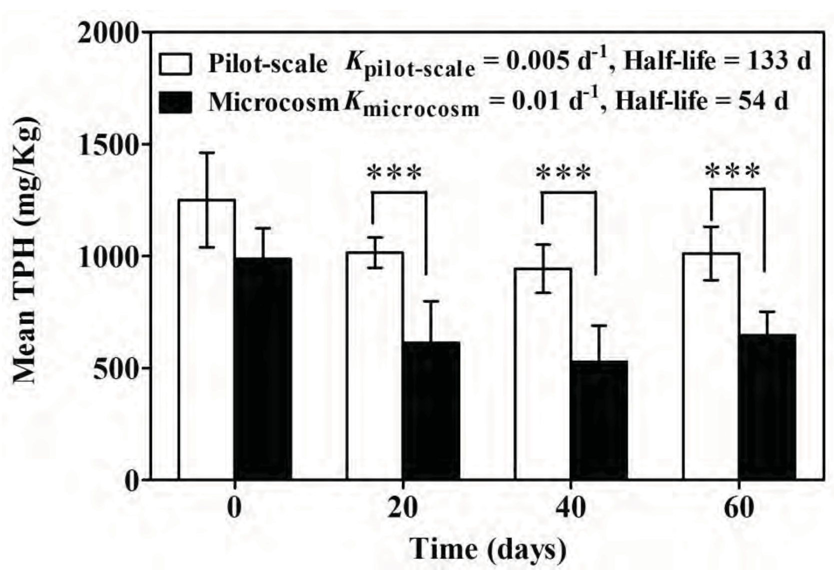

*Figure 1. The effects of system scale on the rate and extent of petroleum hydrocarbon biodegradation at 5.5°C. The standard deviation is presented as bars. ***: p-value < 0.0001 (obtained by two-way ANOVA). TPH: Total Petroleum Hydrocarbons.*

Approximately 40% of the initial TPH were removed at the microcosm scale. In the pilot-scale system, the percent removal of TPH was 21%. The two-way analysis of variance (ANOVA) confirmed that the effects of scale and time on TPH-biodegradation were statistically significant (the effects of both scale and time resulted in a p-value < 0.0001). In unamended soils (control), no significant TPH removal occurred in both the microcosm- and pilot-scale experiments.

The TPH chromatograms obtained from the GC/FID analyses (Gas Chromatography equipped with a Flame Ionization Detector) showed the degradation of a variety of petroleum hydrocarbon fractions, including the peak and hump areas of the two different systems. The hump area, which represents Unresolved Complex Mixtures (UCMs), potentially includes structurally complex hydrocarbons with branched and unsubstituted alkyl chains. At the microcosm scale, the UCM humps after the biotreatment were significantly smaller than those of the pilot scale. After the 60-day biotreatment, the final TPH concentrations in the microcosm- and pilot-scale experiments were 648±104 mg/Kg and 1012±119 mg/Kg, respectively. Based on the TPH-data analyses, it is necessary to consider the effects of scale when microcosm systems are used for predicting *in situ* rates and extents of petroleum hydrocarbon biodegradation in the RI landfarm soils. Although the RI site soils are relatively homogenous sandy soils in terms of particle distributions and soil compositions, the scale effect is most likely due to nutrient mass transfer limitations, buffering reagent amendments and the slower diffusive flux of oxygen through soil phases in larger systems. Considering the scale effect is therefore also important for scaling up from pilot to field scale, especially due to a potentially significant increase in the heterogeneity of site soils.

Soil Treatments for Enhancing Indigenous Microbial Activity in Contaminated Soils

A high dosage of nutrient supply to contaminated soils inhibits microbial activity and a low nutrient dosage delays the onset of microbial activity. The selection of nutrient concentrations and types is thus important for stimulating indigenous hydrocarbon-degrading bacteria in contaminated soils and for maximizing the effectiveness of soil bioremediation.

At the pilot-scale, all the nutrient amendments (50, 100, and 250 milligrams of nitrogen per kilogram of wet soil) stimulated aerobic microbial activity, which was estimated based on soil pore gas concentrations of CO_2 and O_2 (Figure 2). The nutrient amendments effectively increased CO_2 production and O_2 consumption, indicating enhanced microbial activity in petroleum hydrocarbon-contaminated soils at 5.5°C. Conversely, in untreated contaminated soils (control), no significant changes in microbial respiration activity over 60 days were indicated. The degree of microbial enhancement varied with nitrogen (N) concentration. Within the range of concentrations used (50 to 250 mg-N/Kg), a nitrogen dosage of 100 mg-N/Kg provided the most favorable conditions for microbial enhancement in the contaminated soils. Similarly, previous studies consistently indicated that moderate concentration of nitrogen (i.e., 100 mg-N/Kg) successfully stimulated hydrocarbon biodegradation in cold region soils (Braddock et al. 1997). In this study, high nitrogen concentrations (> 250 mg-N/Kg) inhibited biodegradation activity, which was likely due to toxic effects and osmotic stress on hydrocarbon-degrading bacteria caused by high nitrogen concentrations.

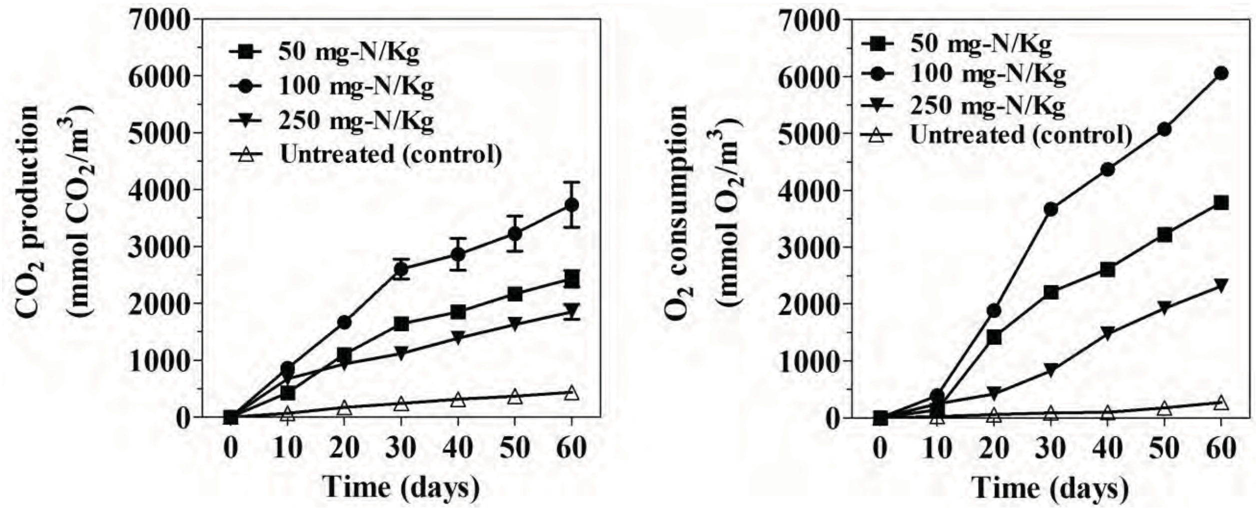

Figure 2. CO_2 production and O_2 consumption in the contaminated soils that received 0, 50, 100 and 250 mg-N/Kg in pilot-scale bioremediation systems.

<u>Impact of Site Temperature Dynamics on the Rate of Hydrocarbon Biodegradation</u>

A total of 15 first-order rate constants for petroleum hydrocarbon biodegradation in site soils were derived from the pilot-scale bioremediation assessments that were supplied with favorable nutrient dosages of 50 and 100 mg-N/Kg. The assessments were carried out under dynamic summer temperatures varying from +1 to 10°C (dynamic temperature regime), under fixed average temperatures of +5.5°C (fixed average temperature) and under extended seasonal freeze-thaw temperatures from -5 to +4°C (seasonal freeze-thaw).

Figure 3 demonstrates the distribution of rate constants for the three different temperature regimes. One-way ANOVA tests showed that the difference in the rate constant datasets was statistically significant (p-value < 0.0001). In addition, Bonferroni's test for the comparison of datasets indicated that the effects of summer temperature dynamics, when compared to a fixed average temperature, are also statistically significant. The mean rate constant for the realistic summer temperature dynamics was 0.017±0.004 day^{-1}, which is over three times higher than the mean rate constant of 0.0048±0.002 day^{-1} for the fixed average summer temperature. Under the seasonal freeze-thaw temperature regime, the mean rate constant was 0.0046±0.002 day^{-1}, which is comparable to that of the fixed temperature experiments. In particular, the rate constant for the biodegradation of semi-volatile hydrocarbons (C10-C16) was 0.006 day^{-1} in the seasonal freeze-thaw temperature regimes including subzero temperatures (half-life: 116 days; R^2 = 0.94, first-order kinetic model). Although the rate of hydrocarbon biodegradation under the seasonal freeze-thaw conditions was slower, hydrocarbon biodegradation activity did not cease during soil freeze-thaw at sub-zero temperatures ranging from -5 to 0 °C (Chang et al. 2011a).

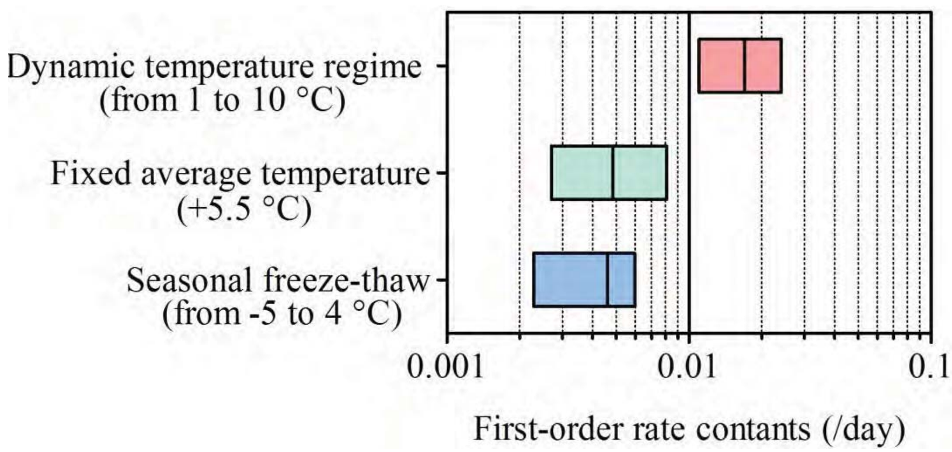

Figure 3. A total of 15 first-order rate constants were derived from a series of previous pilot-scale experiments under the three different temperature scenarios. The nutrient concentrations of 50 and 100 mg-N/Kg were supplied to the contaminated soils.

<u>Extended Microbial Activity as a Function of Unfrozen Water Content in Freezing and Thawing Soils</u>

The change in unfrozen water content in petroleum hydrocarbon-contaminated soils is a key factor for enhancing hydrocarbon biodegradation in semi-frozen and frozen soils (Siciliano et al. 2008). The onset and extent of microbial activity in relation to soil freezing and thawing were positively correlated to changes in unfrozen water content in contaminated soils (Chang et al. 2011a) and, thus, the prediction of unfrozen water content is important for developing bioremediation strategies extended to seasonal freeze-thaw conditions in cold-climate environments.

As shown in Figure 4, the time-dependent model based on a finite element method (TEMP/W model) for the prediction of unfrozen water provided a simulated quantity and distribution of unfrozen water in field-aged hydrocarbon contaminated soils in a pilot-scale soil tank subjected to site-representative seasonal freeze-thaw temperatures. Figure 4(A) shows a snapshot of the distribution of unfrozen water in semi-frozen soils during a soil phase change on Day 30. The simulated unfrozen water content was in excellent agreement with the measured unfrozen water content (Figure 4).

Characteristic curves for unfrozen water and other parameters such as the freezing-point depression (T_f) and the effective endpoint of unfrozen water (T_{ef}) are specific to different soils and their properties, which includes hydrocarbon contamination. The determination of the characteristic curve for unfrozen water for contaminated sandy soils played a critical role in accurately predicting unfrozen water content in unsaturated, hydrocarbon-contaminated soils. The simulated results based on conventional unfrozen water characteristics for uncontaminated sandy soils did not match the measured unfrozen water content in contaminated soils.

The unfrozen water in semi-frozen and frozen soils provides a nutrient reservoir for maintaining microbial metabolism and therefore hydrocarbon biodegradation can continue under the seasonal freeze-thaw conditions. For the hydrocarbon biodegradation assessment in the pilot-scale experiment, Chang et al.

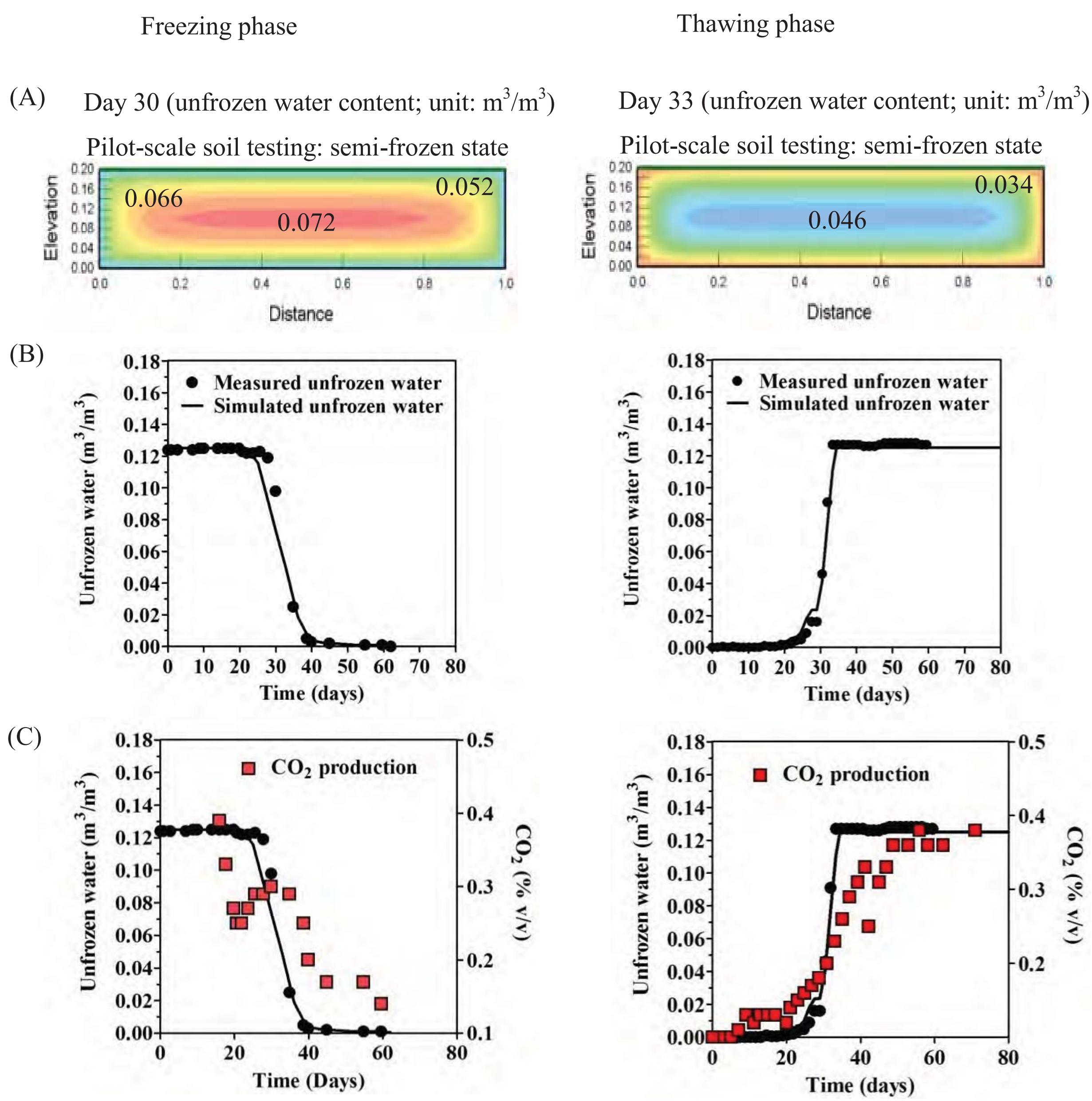

Figure 4. Results of unfrozen water content simulation using TEMP/W: (A) unfrozen water distribution in semi-frozen soils (B) comparison of simulated and measured unfrozen water content (C) correlation between unfrozen water content and microbial CO_2 production.

(2011a) reported that in the nutrient-amended hydrocarbon-contaminated soils, the total removals of semi-volatile (F2: C10-C16) and non-volatile (F3: C16-C34) hydrocarbons were 52% and 16%, respectively, during the soil freeze-thaw phase. Robust biomarker analyses (i.e., bicyclic sesquiterpenes) and ^{14}C-hexadecane mineralization results explicitly showed the progress of hydrocarbon biodegradation (not abiotic loss) in seasonally freezing and thawing contaminated soils. In the previous study (Chang et al. 2011a), increases in bacterial *alkB* genes (hydrocarbon degradation genes) and 16S rRNA gene copy numbers (total bacterial populations) were observed in the semi-frozen state of the soil in which ice and unfrozen water co-exist. Molecular-based microbial community analyses showed that hydrocarbon-degrading bacteria (*Corynebacterineae*- and *Alkanindiges*-related strains) that are adapted to the frozen and unfrozen states of the soils sequentially emerged in response to soil freezing and thawing and, thus, hydrocarbon biodegradation continued to progress in seasonal freeze-thaw temperature regimes. Therefore, time-dependent changes in unfrozen water content should be accurately assessed. This study represents an extension of the study by Chang et al. (2011a) to investigate the feasibility of predicting unfrozen water content in the RI contaminated soils.

There was a very strong correlation between unfrozen water availability and microbial respiration activity. In Table 1, the correlation analyses indicated a significant positive correlation between unfrozen water content and microbial activity (i.e., CO_2 production) in freezing and thawing hydrocarbon-contaminated soils. Correlation coefficients (Pearson r) during the freezing and thawing phases were 0.73 and 0.94, respectively. Therefore, the change in unfrozen water content in freezing and thawing biotreated contaminated soils can be an effective indicator of microbial activity under seasonal freeze-thaw conditions. This study suggests that the prediction of unfrozen water quantity and distribution in field-aged hydrocarbon-contaminated soils during the seasonal freeze-thaw period can be used for developing bioremediation strategies extended beyond conventional active treatment seasons in cold-climate environments, which are limited to short summers.

Table 1. Results of statistical correlation analyses for unfrozen water content and CO_2 production.

	Freezing phase	**Thawing phase**
Pearson r (correlation coefficient)	0.73	0.94
P value (two-tailed)	0.002	< 0.0001
Is the correlation significant? (α = 0.05)	Yes	Yes
R^2	0.53	0.88

CONCLUSIONS

The majority of laboratory assessments for bioremediation feasibility for hydrocarbon-contaminated soils in cold climates are based on microcosm-scale soil experiments at fixed average temperatures for cold sites. In this study, the rates and extents of hydrocarbon biodegradation in the scale-up bioremediation systems were significantly different from the microcosm-scale systems. The first-order rate constants obtained from the pilot-scale experiments were two times smaller than those obtained from the microcosm experiments. Compared to the pilot-scale experiments, biodegradation was also extended to higher molecular weight hydrocarbon fractions at the microcosm-scale. The microcosm experiments may overestimate the bioremediation performance that is achievable in the field. Inorganic, N-based nutrient supplies of 50 and 100 mg-N/Kg resulted in significant microbial enhancement in hydrocarbon-contaminated soils from the RI site. When comparing to conventional laboratory fixed temperature scenarios (averaged for summers), it becomes clear that the impacts of dynamic site temperature scenarios are important for estimating the achievable rate constants for petroleum hydrocarbon biodegradation in the field at the RI site.

In the pilot-scale experiment under the dynamic site temperature conditions, changes in simulated and measured unfrozen water content in freezing and thawing contaminated soils correlated significantly to microbial activity estimated based on CO_2 production. The quantity and distribution of unfrozen water during soil freeze-thaw can be predicted in hydrocarbon-contaminated soils using the TEMP/W model when characteristic curves for unfrozen water in hydrocarbon-contaminated soils are available. The predictive model for unfrozen water content can be an effective indicator for microbial activity and can be seen as a framework for developing a seasonality-based bioremediation strategy that is extended to freeze-thaw seasons, beyond the short, active summer treatment seasons at northern contaminated sites.

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FIELD ASSESSMENT OF SULPHIDE OXIDATION RATES IN COLD ENVIRONMENT: CASE STUDY OF RAGLAN MINE

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ABSTRACT

In this field study conducted on Raglan Mine's tailings storage facility (TSF), tailings oxidation rates were characterized with the oxygen consumption (OC) method. Surface tailings unfrozen volumetric water content and temperature were also measured simultaneously with OC tests. Oxygen fluxes between 30 and 550 $mol \cdot m^{-2} \cdot yr^{-1}$ were observed during summers 2011 and 2012 when tailings temperature and unfrozen volumetric water content varied between -0.1 and 12.8ºC and 0.09 and 0.23 respectively. These oxygen fluxes are non-negligible, but lower than those measured in the lab for the same tailings, which varied between 390 and 1070 $mol \cdot m^{-2} \cdot yr^{-1}$ at 21°C. Oxygen flux drops in October 2011 to values below 10 $mol \cdot m^{-2} \cdot yr^{-1}$ with unfrozen volumetric water contents less than 0.09 and temperatures of around -4°C. Results of this study show that, as expected for dry stack tailings, the degree of saturation at Raglan Mine's TSF (generally between 40% and 60%) is not sufficiently high to control oxidation reactions. Oxygen fluxes decrease with temperature and they are greatly reduced when tailings temperature reaches a value of -6°C, conditions observed for a period of 168 days of the year. This study also showed that Arrhenius' law can be used to approximate the effect of temperature on oxygen fluxes for activation energy values between 60 kJ/mol to 124 kJ/mol.

Key Words: Oxidation, Tailings, Oxygen Consumption, Temperature, Arctic.

INTRODUCTION

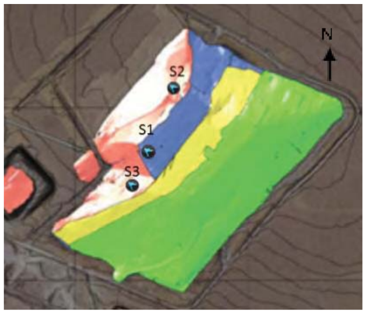
Interaction between sulphide minerals contained in tailings, water and oxygen can produce acid mine drainage (AMD), an environmental problem of the mining industry. It is generally accepted that molecular diffusion is the principal mechanism of oxygen migration in fine grained tailings (Collin and Rasmuson 1988). Modified Fick's second law can be used to describe oxygen diffusion through reactive tailings for a first-order reaction. Solving Fick's second law under steady state condition ($\partial C/\partial t = 0$) for homogeneous reactive tailings and specified boundary conditions gives the incoming oxygen flux (F_L) through a tailings surface (eq. 1) (Elberling et al. 1994; Mbonimpa et al. 2011). Oxygen fluxes have been used in field and laboratory studies to assess tailings oxygen consumption, in both temperate and Nordic conditions (Elberling 2001; Meldrum et al. 2001; Tibble and Nicholson 1997).

[1]

The flux of oxygen is controlled by the oxygen diffusion coefficient (D_e) and by the reaction rate coefficient (K_r). The value of the diffusion coefficient depends on the tailing's degree of saturation; maximal oxygen concentration through water-filled pores (C_w = 9.2 mg/L) is 30 times less than the oxygen concentration in air (C_a = 276.7 mg/L). A study by Ouangrawa (2007) shows that at saturation greater than 85%, sulphide oxidation rate is significantly reduced. Since oxidation reaction kinetics is affected by temperature, Arrhenius' law can be used to describe effect of temperature on tailings oxygen fluxes (Elberling 2001). Studies on tailings oxidation rates in Nordic environments have shown that oxygen consumption can take place when tailings temperature reaches values of -2°C (Meldrum et al. 2001) and -4°C (Elberling 2001). In this study, oxygen consumption (OC) tests were performed for a one year period at different locations on the Raglan Mine tailings storage facility (TSF) to characterize the *in situ* tailings oxidation rates. Measurements of tailings unfrozen volumetric water content and temperature were also performed simultaneously with the OC tests. The main objective of this study is to better understand the evolution of tailings oxygen consumption rate during a typical year and to relate it with two important factors of influence: temperature and volumetric water content.

STUDY SITE AND FIELD TESTS DESCRIPTION

Raglan Mine is located in Northern Quebec, Canada and has operated since 1997 to produce nickel and copper concentrate. The site is located in subarctic desert climate in a region of continuous permafrost. The climate is severe for both wind and weather conditions (mean annual air temperature of -10.3°C). The tailings deposition method used at the site is referred as dry stack tailings, which consists of filtering tailings coming from the milling process to a solid percentage around 85%, so tailings have a "cake" like consistency. Solid tailings are then transported by truck and stored at the tailings storage facility. In this study, tailings temperature, unfrozen volumetric water content and oxygen fluxes were measured at three different locations (S1, S2 and S3) on different configuration of the TSF surface (see Fig. 1-a). Temperature and unfrozen volumetric water content were recorded simultaneously with the 5TM sensor from Decagon Devices. A calibration curve was established in the laboratory to increase the accuracy in volumetric water content measurements in tailings (Coulombe 2012). Sensors were installed at a depth of 5, 15, and 25 cm under the tailings surface and measurements were taken every hour from July 5, 2011 to July 18, 2012. Tailings oxidation rate is characterized with the oxygen consumption test, which gives the oxygen flux (F_L) consumed by the tailings oxidation reaction. The OC test measured the decrease of oxygen concentration over time in a sealed chamber over tailings during a period of three to five hours. The test is performed in a 30 cm stainless steel cylinder of 15 cm diameter which is inserted in the tailings leaving an empty space "h" at the top of the cylinder ("h" in this study ranged between 10.2 cm to 13.8 cm). Oxygen concentration decrease was measured with the SO-110 sensor (Apogee Instruments) having an accuracy of ± 0.02% O_2. A schematic view of a typical monitoring station is show in Figure 1-b. Three OC setups were installed at each monitoring station and between 6 and 8 OC tests were performed in each cylinder from July 2011 to July 2012 at different times of the year. Tests are interpreted with the method described by Elberling et al. (1994), for a maximum decrease in oxygen concentration of 3%. With this method, slope of the plot of the logarithm relative change in oxygen concentration with time gives the parameter $(K_r D_e)^{0.5}$, that is inserted in eq. 1 to obtain the oxygen fluxes consumed by tailings.



a)

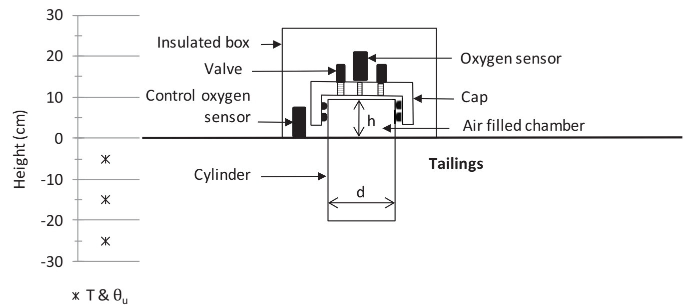


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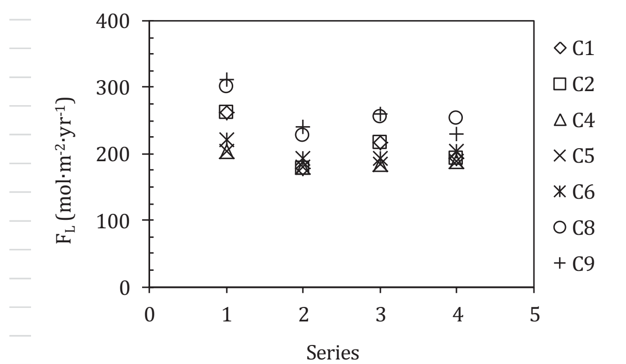
Figure 1 a) Station locations on the Raglan Mine TSF; b) Schematic view of the oxygen consumption test setup and instrumentation localization (5TM sensors measure temperature (T) and unfrozen volumetric water content (θ_u) simultaneously).

Variability of the OC test was studied on the Raglan Mine TSF. Four series of seven OC tests were performed in a three days period (from July 19 to 21, 2012) on a surface of 4 m^2 between stations S1 and S2 on the flat tailings surface. Tests were interpreted with the approach and the conditions described above (see Coulombe (2012) for details).

a)



b)

Figure 2. a) Setup used on TSF to assess the variability of OC tests; b) variability of oxygen fluxes measured with the OC test.

Results presented in Figure 2-b vary from 204 to 312 mol·m^{-2}·yr^{-1}, 178 to 241 mol·m^{-2}·yr^{-1}, 184 to 268 mol·m^{-2}·yr^{-1} and 190 to 254 mol·m^{-2}·yr^{-1} for the four test series (Figure 2-b), representing an oxygen flux variability between 25% and 35%. This variability is mainly due to tailings geochemical and unsaturated hydrogeological heterogeneities. Despite the variability observed with the oxygen consumption test, it is considered here as an appropriate tool to study the *in situ* oxidation rates of sulphide tailings.

MATERIAL CHARACTERIZATION

Tailings were sampled on the Raglan Mine's TSF at stations S1 to S3 and characterized in the laboratory for their physical, chemical and mineralogical properties (Table 1). Specific gravity measured with a helium pycnometer varied between 2.895 and 2.931. Tailings have a typical grain size distribution with a D_{10} (grain size at 10% passing) ranging from 0.0015 to 0.002 mm and 79% to 83% of the particles passing 80 μm. Carbon and sulfur analyses conducted with an induction furnace show that tailings sulfur content varies between 4.38 ± 0.5% and 4.95 ± 0.5%. The main sulphide mineral found by X-Ray diffraction (DRX) in these tailings samples is pyrrhotite with traces of pentlandite. The main gangue minerals are lizardite, chlorite, magnetite and hornblende. Tailings samples are considered acid generating because of their sulfide content between 10% and 20%wt and their low neutralization potential (absence of carbonate minerals).

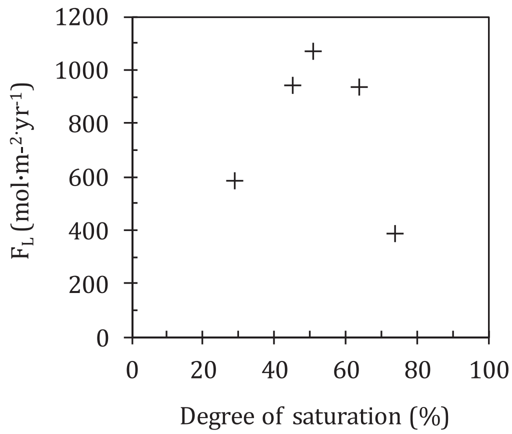
Physical	S1	S2	S3
Specific gravity	2.931	2.930	2.895
Grain size D_{10} (mm)	0.0015	0.0017	0.002
Grain size D_{60} (mm)	0.0229	0.0287	0.0323
C_U	14.8	16.4	16.5
Particles passing 80 μm (%)	83	79	80
S (wt%)	4.38	4.95	4.69
C (wt%)	0.294	0.264	0.185
Mineral content (%)			
Chlorite $(Mg, Al, Fe)_6(Si,Al)_4O_{10}(OH)_8$	12.8	9.2	12.2
Hornblende $Ca_2[Mg_4(Al,Fe)]Si_7AlO_{22}(OH)_2$	9.7	9.2	10.8
Lizardite $Mg_3Si_2O_5(OH)_4$	40.9	38.4	51.5
Magnetite Fe_3O_4	7.7	8.5	12.4
Pentlandite $(Fe,Ni)_9S_8$	0.9	1.4	0.4
Pyrrhotite $Fe(1\text{-}x)S$	10.4	13.2	12.1
Other	17.6	20.1	0.6

Table 1. Main physical, chemical and mineralogical properties of the tailings studied

Oxygen consumption tests on Raglan Mine's tailings were performed in the laboratory to determine oxygen consumption rates at ambient temperature (21°C) for different degrees of saturation. Five OC tests were performed on tailings in a column having a diameter of 14 cm and a height of 29 cm (see Figure 3-a) at porosity between 0.37 and 0.43. Tests were interpreted with the previously described approach of Elberling et al. (1994). Results of the oxygen consumption tests are shown in Figure 3-b. The maximum oxygen consumption rates obtained is 1070 $mol \cdot m^{-2} \cdot yr^{-1}$ at 51% saturation. When the degree of saturation reaches values close to 20% and to 80%, oxygen consumption decreased to values less than 580 $mol \cdot m^{-2} \cdot yr^{-1}$. These laboratory results are consistent with those presented in the literature that show an optimal sulphide oxidation for degrees of saturation between 40% and 60% (Bouzahzah 2013; Godbout

2012). This is explained mainly because gas diffusion in a porous media is negligible at high saturation degree (Aachib et al. 2004) and because not enough water is available to feed the oxidation reaction at low saturation (Godbout 2012).

a)



b)

Figure 3. Laboratory evaluation of tailings oxygen consumption a) column used for OC tests in laboratory; b) Oxygen fluxes measured in laboratory at different degree of saturation and at ambient temperature

MAIN RESULTS

Figure 4 shows the mean (from the three cylinders installed at each station), minimum and maximum oxygen fluxes (4-a), the hourly tailings temperature distribution at 5 cm under the tailings surface (4-b), and the hourly unfrozen volumetric water content distribution at 5 cm under the tailings surface (4-c) for stations S1 to S3. The focus in this study is mainly on sensors located at 5 cm under the tailings surface because tailings oxidation reaction takes place in the first centimetres (Godbout 2012). Oxygen fluxes measured on Raglan Mine's TSF from July 2011 to July 2012 vary between 0 and 550 $mol{\cdot}m^{-2}{\cdot}yr^{-1}$. Tailings' oxygen consumption annual tendency is similar for the three stations but at different intensities. Maximum oxygen fluxes are reached during the months of July (2011 and 2012) with maximum values of 240, 390 and 550 $mol{\cdot}m^{-2}{\cdot}yr^{-1}$ for S1, S2 and S3, respectively, when tailings temperature is higher. A maximum tailings temperature of 17.5°C, 12.5°C (incomplete data set) and 25.2°C for S1, S2 and S3 are reached during the months of June or July.

During the period between June and August 2011, the tailings unfrozen volumetric water content remained fairly stable at values ranging between 0.17 and 0.26, which corresponds to value of unfrozen degree of saturation between 40% and 60% (assuming a porosity of 0.43). It is important to remember that a degree of saturation greater than 85% is required to control oxygen diffusion in tailings and sulphide oxidation (Ouangrawa 2007). Tailings temperature reaches 0°C for the first time at mid-September 2011 and remains constant slightly above 0°C until the end of September 2011. Oxygen fluxes are still measurable the 26th of September 2011 when tailings temperature is close to 0°C, with values of approximately 30, 250 and 30 for S1, S2 and S3 respectively. Unfrozen water content varies between values of 0.17 to 0.21 (S1), 0.03 to 0.21 (S2) and 0.09 to 0.18 (S3) from mid-September 2011 to the end of September 2011, during the period where tailings temperature is close to 0°C. Oxygen fluxes reach values lower than 10 mol·m^{-2}·yr^{-1} by the end of October 2011, when tailings temperature and volumetric unfrozen water content values are -4.1, -4.2 and -4.5ºC and 0.01, 0.03 and 0.08 for S1, S2 and S3, respectively. During winter months, unfrozen water content values ranging between 0.01 and 0.05 at 5 cm were measured under the tailings surface. No OC tests were performed during the winter period because snow and ice accumulation was blocking access to the OC test cylinders. Unfrozen volumetric water content slowly started increasing at the end of April 2012, when tailings temperature was close to -8°C, and then increased rapidly when tailing temperature was close to 0°C (in June 2012) to reach values between 0.16 and 0.24 (40% < S_r < 56%), which is similar to data observed during summer 2011.

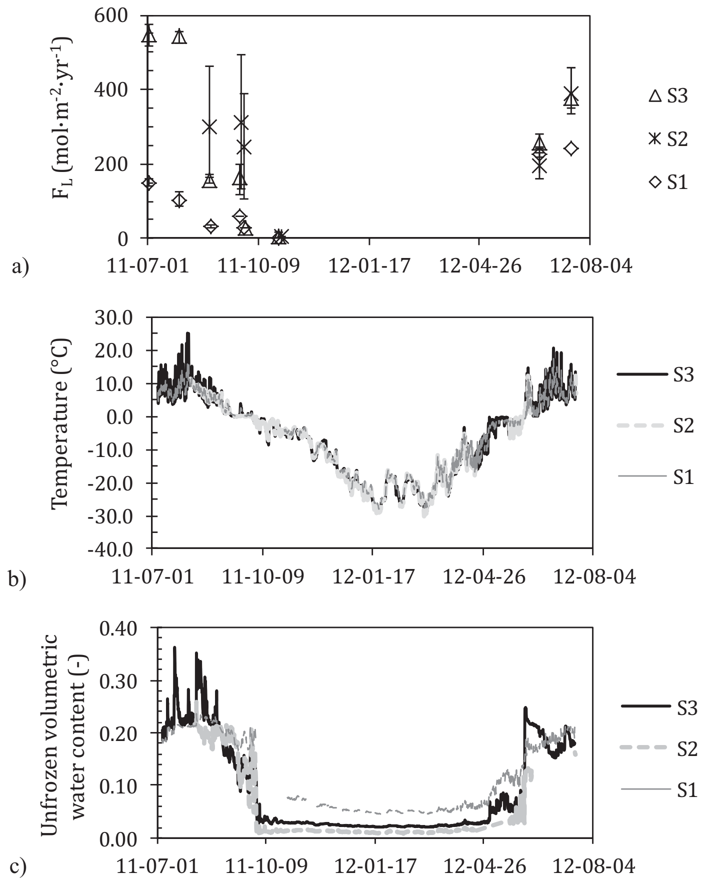


Figure 4. Main results obtained on the Raglan Mine TSF between July 5, 2011 and July 18, 2012; a) Mean, minimum and maximum oxygen fluxes measured; b) hourly tailings temperature distribution at 5 cm under surface; c) hourly tailings unfrozen volumetric water content distribution at 5 cm under the surface

Oxygen fluxes measured after the first winter (in June 2012) were similar to those observed in summer 2011 with values greater than 200 mol·m^{-2}·yr^{-1}. Figure 4 shows that oxygen flux appears to be closely linked with tailings temperature; when tailings temperature decreased, oxygen consumption decreased. Unfrozen volumetric water content (degree of saturation) also influenced tailings oxygen consumption.

The following section further examines the relationships between unfrozen volumetric water content and temperature on tailings oxygen consumption rates.

DISCUSSION

Effect of the Water Content on Oxygen Fluxes

The relationship between oxygen fluxes and tailings unfrozen volumetric water content are shown in Figure 5. Measurements of the oxygen consumption at 21°C performed in UQAT laboratory on tailings sampled on Raglan Mine's TSF are also presented for comparison. *In situ* oxygen fluxes were measured at rates between 30 and 550 mol·m^{-2}·yr^{-1}, for unfrozen volumetric water content values between 0.09 and 0.23, showing that tailings degree of saturation allows the oxidation reactions. Tailings temperature values during these OC tests varied between -0.1°C and 12.8°C. Figure 5 shows that the maximum oxygen flux measured in the field (F_L=550 mol·m^{-2}·yr^{-1}) is almost half the values of the maximum oxygen flux measured in the laboratory at 21°C (F_L=1070 mol·m^{-2}·yr^{-1}). These maximum oxygen flux values in the lab and in the field were observed for unfrozen volumetric water content values between 0.17 and 0.26, which corresponds to degree of saturation between 40% and 60%. Such degree of saturation corresponds to ideal conditions for tailings oxidation reactions to proceed (Bouzahzah 2013; Godbout 2012). At unfrozen volumetric content lower than 0.09, oxygen fluxes drop to values less than 10 mol·m^{-2}·yr^{-1}. Low unfrozen volumetric water content could explain (at least in part) the lower oxygen fluxes of less than 10 mol·m^{-2}·yr^{-1} in October (θ_u = 0.01, 0.03 and 0.08 for S1, S2 and S3). Those tests were performed when tailing temperature was under 0°C (about -4°C). The cold temperature is then another factor that could explain the low reaction rates observed.

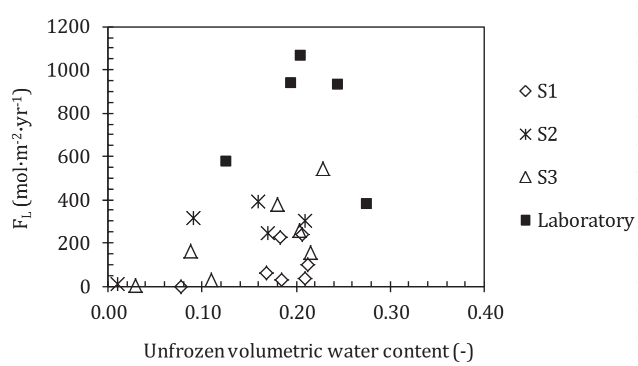


Figure 5. Mean oxygen fluxes measured at different unfrozen volumetric water contents in the fields for tailings temperature between -4.5 and 12.8°C and in the laboratory at 21°C

Temperature Effect on Oxygen Fluxes

The effect of tailings temperature on oxygen fluxes for the monitoring stations is presented in Figure 6-a. As expected, oxygen fluxes diminished with temperature. The highest oxygen fluxes (from 340 to 540 mol·m^{-2}·yr^{-1}) were measured at temperatures between 7.9 and 12.8°C. Oxygen fluxes that decreased gradually with temperature were still measurable at temperatures close to 0°C with values between 30 and 310 mol·m^{-2}·yr^{-1}. OC tests performed at tailings temperature less than -4°C (-4.1°C, -4.2°C and -4.5°C) yield values less than 10 mol·m^{-2}·yr^{-1}. Results suggest that sulphide oxidation reaction was almost stopped for tailings temperature somewhere between -1°C and -5°C. Assuming that a minimum temperature of -6°C is necessary to control sulphide oxidation reactions, tailings would be below this critical temperature value for about 168 days (46% of the year) at station S3 (similar results are observed for the other stations).

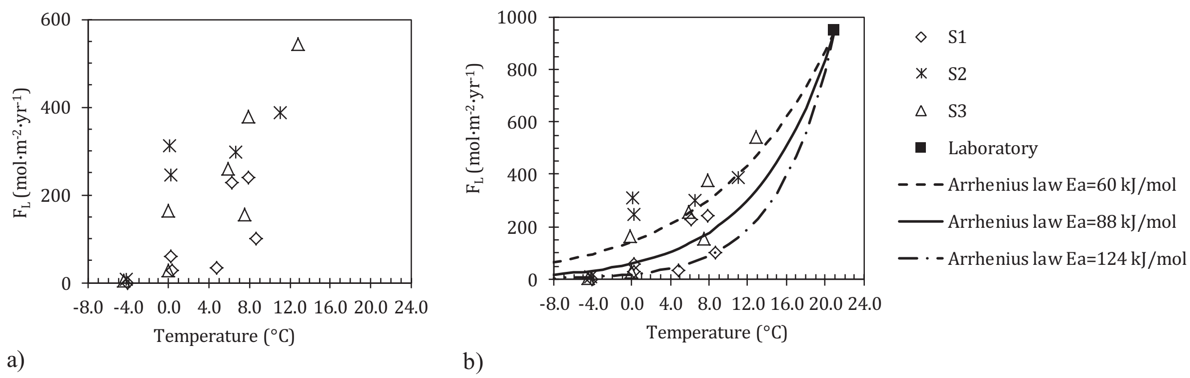


Figure 6. a) Effect of tailings temperature on oxygen fluxes; b) Effect of temperature on oxygen fluxes and prediction with Arrhenius law for different activation energies.

The Arrhenius equation is used to predict the effect of temperature on tailing oxygen fluxes measured at Raglan Mine's TSF (eq. 2), where K_1 and K_2 represent oxygen consumption at temperature T_1 and T_2, E_a is the activation energy and R is the gas constant (Figure 6-b). Three different activation energies have been used (60, 88 and 120 kJ/mol) corresponding to activation energy found in the literature for pyrrhotite (Janzen et al. 2000). A K_1 of 945 mol·m^{-2}·yr^{-1} measured in the laboratory at T_1 of 21ºC is used for the calculation.

— ——— [2]

Figure 6-b shows that Arrhenius' law accurately predicts field data for temperatures above 0°C with activation energy of 60 kJ/mol, considering that temperature is not the only parameter affecting oxygen fluxes. Results also show that Arrhenius' law seems to predict oxygen consumption rates at temperatures below 0°C more precisely with higher activation energy (between 80 to 124 kJ/mol). Nevertheless, more work is needed to better understand the fundamental aspects related to the influence of temperature on pyrrhotite oxidation.

<u>Spatial Variation of Pyrrhotite Content</u>

Figure 4-a demonstrates that oxygen fluxes can be quite different from one station to the other for an OC test performed the same day in similar conditions. As mentioned in previous sections, tailings degree of saturation is an important parameter controlling the oxygen diffusive flux and temperature affects the kinetics of sulphide oxidation reaction. However, tailings temperature and unfrozen volumetric water content distribution is generally similar from one station to the other (see Figure 4-b and 4-c). Hence, other factors must influence tailings oxygen consumption rates (Coulombe 2012). One of them is

pyrrhotite content. Indeed, a study has shown that the tailings reaction rate coefficient (K_r) varies linearly with relative sulphur content (Collin 1998). To verify this statement, an annual mean oxygen flux is calculated with OC tests results from August 26, 2011 to July 17, 2012 (n = 6) and related to pyrrhotite content (C_p) for station S1 (C_p = 10.4%), S2 (C_p = 13.2%) and S3 (C_p = 12.1%). Annual mean oxygen fluxes of 100, 240 and 170 mol·m^{-2}·yr^{-1} are calculated for stations S1, S2 and S3 respectively. A linear relation with a correlation coefficient of 0.9849 is observed between tailings sulphide content and annual oxygen fluxes (see Coulombe (2012) for more details). Therefore, tailings sulphide content could also be a cause of the variation observed in tailings oxygen consumption rates at Raglan Mine's TSF.

CONCLUSION

In this paper, the influences of temperature and unfrozen volumetric water content on tailings oxygen consumption were studied on Raglan Mine's TSF. Maximum oxygen fluxes of 550 and 1070 mol·m^{-2}·yr^{-1} were measured in the field and laboratory for unfrozen volumetric water content values between 0.17 and 0.26 (40 < S_r < 60%). The study showed that, as expected for dry stack tailings (Bussière 2007), the tailings saturation at Raglan Mine's TSF is not sufficiently high to control oxidation reactions. It is also shown, as expected, that oxygen fluxes decreased with temperature. Tailings oxygen consumption was greatly reduced when tailings temperature reached values below -6°C, conditions observed for approximately 50% of the year. It was confirmed that Arrhenius' law can be used to estimate the effects of temperature on measured oxygen fluxes with activation energy between 60 and 124 kJ/mol. This study also showed that significant differences are observed when assessing field oxygen fluxes at different locations on the same tailings stack due to mineralogical and unsaturated hydrogeological heterogeneities.

Results of this field study demonstrate that cold temperatures can be used to control AMD generation, but appropriate conditions must be reached. This study was completed as part of a larger effort to increase the understanding of Raglan Mine's tailings, their reactivity, and the factors that influence tailings reactivity in the current closure concept. The operation is currently in a pre-feasibility stage study using intermediate scale instrumented experimental cells testing different cover concepts and their effects on mitigating the factors that influence tailings reactivity in an effort to update the current closure concept.

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PHYSICAL-CHEMICAL TREATMENT WITH GEOTUBE® FILTRATION APPLIED TO UNDERGROUND GOLD MINE DEWATERING IN WINTER

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ABSTRACT

Vezza Mine, a former gold mine includes a three-compartment shaft with four underground levels down to a depth of 741 metres, This mine was never put in production since its construction around 1997. In 2010, North American Palladium bought the mine and had to dewater the underground infrastructure to restart mine exploitation. Mine water was highly charged with Total Suspended Solids (TSS), Iron (Fe), Zinc (Zn) and Copper (Cu) and it was not possible to discharge that water into the natural environment. First step was to pump and treat water to dewater mine galleries and second step was to treat mine water on a regular basis. ASDR installed a chemical conditioning unit coupled with a Geotube® bag filtration unit. Treated water reached and exceeds the regulatory discharge criteria of the Quebec Government Directive 019. System has been running since January 2011 and still produces the same water quality.

Key Words: Water Treatment, Mine, Geotube Filtration, Dewatering.

INTRODUCTION

Vezza is an advanced-stage exploration project situated 80 kilometres by paved road from NAP's Sleeping Giant gold mine. The Vezza gold deposit historically underwent extensive exploration of 85,000 metres of drilling, and substantial underground development. The project, which has power at site, includes a three-compartment shaft with four underground levels down to a depth of 741 metres, a hoist, and surface and pollution control infrastructure. An internal feasibility study was prepared by AEM in 1997, but the project was never put into production due to the absence of nearby milling facilities and relatively low gold prices at the time.

In 2010, North American Palladium bought Vezza mine (picture 1). To re-start mine exploitation, first step of the project was to dewater the underground galleries and main shaft. Mine water didn't comply with the Quebec Government regulation for mine water discharge in the natural environment. A water treatment program was required and the second challenge was to start pumping and treating water during the winter period.

Picture 1. Vezza Mine – Head Frame

SITE CONDITION AND TREATMENT GENERAL DESCRIPTION

North American Palladium Ltd. asked for a water treatment program to control Total Suspended Solids (TSS) level as well as Iron (Fe), Zinc (Zn) and Copper (Cu) level to comply with the Directive 019 levels (Quebec Ministry of Environment and Parks 2012) required by Quebec environmental authorities in case of mine dewatering and exploitation (Table 1).

Table 1. Directive 019 - Discharge Criteria under Quebec Regulation

PARAMETER	Monthly Average Concentration	Maximum Concentration
Arsenic	0,2 mg/L	0,4 mg/L
Copper	0,3 mg/L	0,6 mg/L
Iron	3 mg/L	6 mg/L
Nickel	0,5 mg/L	1 mg/L
Lead	0,2 mg/L	0,4 mg/L
Zinc (extractable)	0,5 mg/L	1 mg/L
Cyanide	1 mg/L	2 mg/L
Hydrocarbons (C10-C50)	-----------	2 mg/L
TSS	15 mg/L	30 mg/L

Pumping started in January 2011 and flow rate reached 100 m^3/h to 200 m^3/h for a 30 day period, 24 hours per day. Total pumped volume was about 72 000 m^3. As the objective was to maintain water level in the mine at a certain level, an average flow rate of 35 m^3/h was treated.

A physical-chemical treatment was used (picture 2), and is still in use, to comply with the level of contaminants authorized by Directive 019. This treatment is a process of coagulation-flocculation to capture dissolved ions and aggregate those ions into filterable flocs. Those flocs are filtered using geotextile filtration with a Geotube® bag. Once filtered, treated water is then stored in an existing lagoon and could be discharged or re-used in the mine exploitation process.

Picture 2. General view of treatment unit and Geotube® filtration

PHYSICAL-CHEMICAL TREATMENT

A 150 HP pump feeds a 4 inch underground pipe and above ground pipe to pump water from the galleries to the treatment unit. A 6 inch pipe is installed on the top of the treatment unit to feed the treatment unit.

As mine water contains TSS and heavy metals, in-line treatment is done in three chemical conditioning steps using three different additives, followed by a physical filtration (picture 3).

<u>Step1: pH adjustment</u>

pH is raised to 10,5 using sodium hydroxide addition. That step precipitates metals and unbalances electrical charges around the ions.

Picture 3. Water quality at different steps of chemical conditioning

<u>Step 2: Coagulation</u>

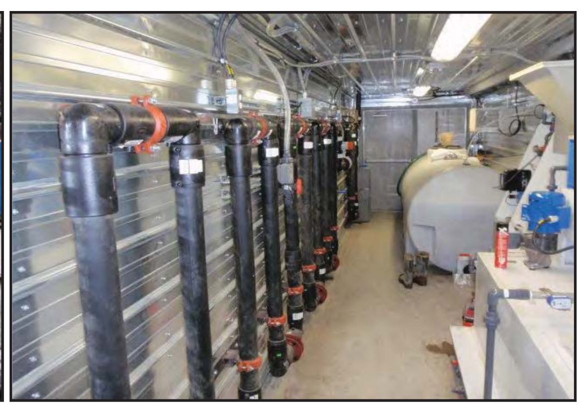
A coagulant (aluminum sulfate) is added to change electrical interaction between Fe and Zn ions that now are able to agglomerate creating a spin floc. At the same time, aluminum sulfate is reducing pH to an acceptable level for the last step of the chemical conditioning.

<u>Step 3: Flocculation</u>

A polymer is added to produce flocs composed of spin-flocs agglomerated by the polymer long carbon chain. Those flocs reach a size larger than 400 microns so they can be filtered using a Geotube® bag.

The whole process takes place in a treatment unit designed and provided by ASDR (picture 4 and picture 5).

Picture 4. ASDR treatment units on site



Picture 5. Inside the ASDR treatment unit

Physical separation using a geotextile tube

A Geotube® is made of GT500 fabric which is composed of high-tenacity polypropylene yarns, which are woven into a stable network such that the yarns retain their relative position (Table 2). GT500 is inert to biological degradation and resistant to naturally encountered chemicals, alkalis, and acids.

Table 2. Specifications of Tencate GT500 fabric

Mechanical Properties	Test Method	Unit	Minimum Average Roll Value	
			Machine Direction	Cross Direction
Wide Width Tensile Strength (at ultimate)	ASTM D 4595	kN/m (lbs/in)	70 (400)	96.3 (550)
Wide Width Tensile Elongation	ASTM D 4595	%	20 (max.)	20 (max.)
Factory Seam Strength	ASTM D 4884	kN/m (lbs/in)	70.1 (400)	
Apparent Opening Size (AOS)	ASTM D 4751	mm (U.S. Sieve #)	0.425 (40)	
Water Flow Rate	ASTM D 4491	$L/m/m^2$ (gpm/ft^2)	813 (20)	
Mass/Unit Area	ASTM D 5261	g/m^2 (oz/yd^2)	585 (17.3) (Typical Value)	
UV Resistance (% strength retained after 500 hrs)	ASTM D 4355	%	80	

Clear effluent water simply drains from the Geotube® container through the small pores in the specially engineered textile. This results in effective dewatering and efficient volume reduction of the contained materials. This volume reduction also allows for the repeated filling of the TenCate Geotube® container. Over 99% of solids are captured, and clear filtrate can be collected and recirculated through the system.

After the final cycle of filling and dewatering, the solids remain in the bag and continue to densify due to desiccation as residual water vapor escapes through the fabric. During winter and spring, repeated freeze-thaw cycles would increase solid-liquid separation and the solid content inside the bag. Volume reduction can be as high as 90%. When full, the Geotube® container and contents can be deposited at a landfill, remain on-site, or the solids can be removed and land-applied when appropriate. In the case of the Vezza project, Geotube® bag dewatered content is safely removed and disposed in the mine tailings facility.

Two Geotube® bags 60 ft circumference x 100 ft long were installed to filter mine water after chemical conditioning into the treatment unit (Picture 6 and 7). Those tubes are installed on an impermeable lay-down area to collect exfiltration water prior to gravity drainage to the storage lagoon of treated water.

Picture 6. A Geotube® bag on an impermeable membrane.

Picture 7. Treated water flowing from the Geotube® bag

Even at very low temperature, Geotube® bags do not freeze. Snow and ice on the top of bag act as an insulation layer protecting the bag from low temperature. Also, continuous water flow helps to keep enough energy to avoid freezing under that insulation layer.

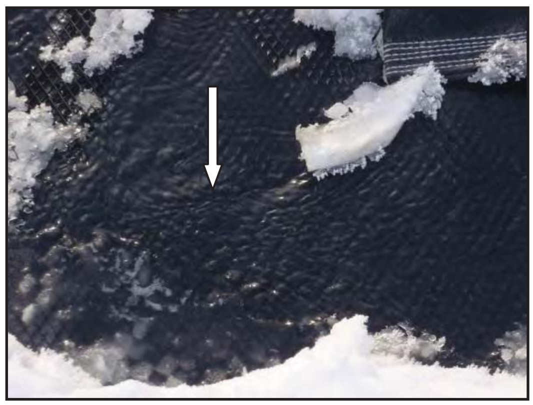
TREATMENT PERFORMANCES

Table 3 shows the results of mine water treatment using the ASDR treatment solution in 2010.

Table 3. Water analysis

Parameters	Raw water mg/L	Treated water mg/L	Directive 019 level mg/L
Copper (Cu)	0.061	0.02	0.300
Iron (Fe)	4.700	0.480	3.000
Zinc (Zn)	0.453	0.015	0.500
TSS	98	9	15
pH	8.05	8.11	6 to 9.5

Presently the treatment unit is operated by a team of Vezza Mine staff trained by ASDR. The treatment performance is still achieved after more than 3 years of operation (Picture 8).



Picture 8. Clean effluent from the geotextile filtration tube Geotube®

CONCLUSION

ASDR mine water physical-chemical treatment combined with Geotube® filtration is an efficient solution to reach environmental discharge objective even in Nordic conditions.

Vezza Mine utilized that solution for gallery mine water treatment and has now been applied daily for more than 3 years.

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MANAGEMENT OF CANADA'S URANIUM AND URANIUM MINING LEGACIES ON THE HISTORIC NORTHERN TRANSPORTATION ROUTE

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ABSTRACT

The Northern Transportation Route (NTR) was established in the 1930s to transport pitch blende ore 2,200 km from the Port Radium Mine in the Northwest Territories to Fort McMurray, Alberta. The ore was then shipped 3,000 km by rail to Port Hope, Ontario, where it was refined for its radium and uranium content. The corridor of lakes, rivers and roads that made up the NTR included a number of points where ore was transferred to other barges or trucks. Ore was occasionally spilled during these operations and, in some cases, subsequently distributed over larger areas as properties were modified.

Since 1991, the Low-Level Radioactive Waste Management Office (LLRWMO), working with communities and its consulting contractors, has characterized spill sites along the NTR where soils exhibit elevated concentrations of uranium, radium and/or arsenic. When feasible to consolidate contaminated material locally, it has been placed into Long Term Management Facilities to contain the materials over extended timelines. In those circumstances where local consolidation is not achievable, materials have been relocated to facilities outside of the region. The LLRWMO is continuing a program of consultation, technical evaluation and environmental assessment to develop management plans for the remaining ore-impacted sites on the NTR. This paper will highlight current activities and approaches for the responsible management of uranium and radium mining legacies.

Key Words: Uranium, Radium, Remediation, Port Radium, Mining, Fort McMurray.

INTRODUCTION

Since its establishment in 1982, the LLRWMO has remediated historic radioactive waste in multiple communities across Canada. Historic waste is defined for policy purposes as low-level radioactive waste (LLRW) that was inappropriately managed and for which the current owner cannot reasonably be held responsible. The LLRWMO is operated by Atomic Energy of Canada Limited (AECL) through a cost-recovery agreement with Natural Resources Canada (NRCan), the federal department that provides the funding, direction and priorities for the LLRWMO.

The LLRWMO and NRCan's activities related to the management of historic LLRW have been supported by long term relationships with several organizations in the consulting community, AMEC Environment & Infrastructure (AE&I) being one of these companies. AE&I has been supporting the LLRWMO with project management, civil engineering and earth science services related to LLRW management since 1992.

This paper was developed from a paper originally presented at the 2011 International Conference on Environmental Remediation and Radioactive Waste Management in Reims, France (Geddes et al. 2011).

HISTORY OF ORE HAULS AND CONTAMINATION ALONG THE NTR

The Northern Transportation Route (NTR) was the 2,200 km marine and portage route used, beginning in the 1930s, to haul pitch blende ore from the Port Radium Mine in Canada's Northwest Territories to the community of Waterways (today Fort McMurray) in the province of Alberta. The ore was then shipped approximately 3,000 km, by rail, to Port Hope, Ontario, in southern Canada on the shores of Lake Ontario, where it was refined, initially, for its radium content and, later, for uranium. At times, aircraft were used to transport ores from the nearby Sawmill Bay airstrip to the south. Figure 1 shows the location of the NTR within Canada.

Figure 1 The Northern Transportation Route

Impacted soils and ore spills have been found along the barge haul route, at Fort McMurray and at the receiving end in Port Hope. The original spills and contamination grew in volume as materials were moved and rehandled by the actions of man and the environment over the years. To put the volumes in perspective, roughly 1,500,000 m^3 are in the Port Hope area, 40,000 m^3 at Fort McMurray and perhaps 10,000 m^3 at sites along the NTR barge haul route.

NTR Communities Impacted

Contamination occurred at several sites along the NTR, typically as a result of accidental spillage of materials, primarily at the transfer points and portages where the ore was moved from one form of transportation to another. Ores were often distributed over larger areas as sites were subsequently modified and/or redeveloped. Beyond the contamination of lands along the NTR, there also was contamination of equipment including boats, barges, and aircraft, used for the haul. Such transportation equipment was used and stored at scattered locations in the north. Most of the equipment from these earlier times is now out of service and sites have been abandoned.

In the early 1990s, the LLRWMO investigated a number of uranium ore-contaminated sites along the NTR. These sites were centred in two areas: the Sahtu Region in the vicinity of Great Bear Lake, and the South Slave Region along the Slave River to the south. Contamination was also found at Fort McMurray – the terminus of the NTR. Figure 2 illustrates the NTR and the main locations of contamination along the barge haul portion of the NTR in the Sahtu and South Slave regions. A number of other locations along the NTR between these points have been investigated by the LLRWMO over the years and found to be uncontaminated.

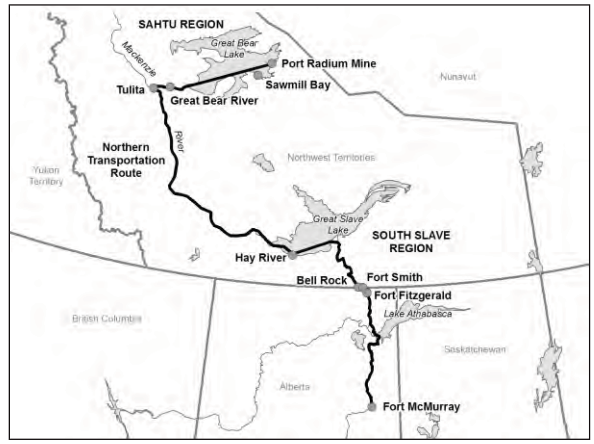

Figure 2: Sahtu and South Slave Sites on the NTR (Characterization and Remediation Sites)

Early Fact Finding and Discovery

In September 1991, consultants were retained by the LLRWMO to conduct a radiological assessment of suspected radioactive contamination at specific sites in Fort Franklin (now Délı̨ne) and Yellowknife, Northwest Territories. The sites were suspected to have been contaminated by uranium ore being transported from the Port Radium mine. The resulting discoveries of elevated radioactivity levels on a barge historically used for ore transport (the Radium Gilbert) and at a river landing prompted a review of the entire historic uranium ore transportation network in the Northwest Territories by the LLRWMO and its consultants. Through discussions with the Northern Transportation Company Limited (NTCL) in Edmonton and museum staff at Norman Wells and Fort Smith, the details of the historic uranium transportation system were established. Through a progressive series of discussions, open houses and meetings over time with many local individuals and groups in NTR communities, much local information and many clues were discovered revealing the history, events, and practices of the time and potential locations of interest for investigation.

In August and September 1992, radiological investigations were conducted for ten vessels, three former warehouse sites, two portages, seven dock/transfer sites, one outside ore storage area, and a number of steel barges used by the NTCL for the transportation of the uranium ore. No low-level radioactive contamination was found on any of these vessels or steel barges (although contamination on two other barges used in Hay River was identified later). However, contaminated building materials and/or soil were found at most of the dock/transfer, warehouse, and storage sites (Senes Consultants Limited 1994). Discrete pieces of uranium ore were also found. The LLRWMO continues to actively monitor these NTR sites as part of its management of the contaminated soils to ensure soils are safely managed, are not further distributed and do not impact the public or the environment.

ASSESSMENT AND REMEDIATION OF SITES TO DATE

Subsequent to the initial discovery of contamination along the NTR, the LLRWMO initiated its program of gamma radiation surveys at potential transfer points along the NTR. Coincident with these surveys, the LLRWMO removed and consolidated contaminated soil from certain properties in Tulita, Fort Smith, Hay River and Fort McMurray. The contaminated material, where consolidated locally, was placed in temporary storage mounds where annual inspections are conducted to demonstrate good management and a safe environment for local residents.

The approach to assessment and remediation of the NTR sites has generally included the steps illustrated in Figure 3 and described as follows:

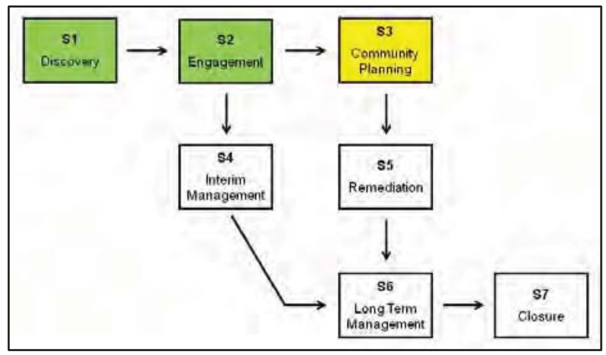

Figure 3: Typical Community Consultation and Remediation Process

- <u>Step 1 (Discovery)</u>: in which the presence of legacy ores is discovered via historical reviews and/or community inputs, and preliminary field investigations are undertaken to characterize the general scale of the impact and its associated consequences.
- <u>Step 2 (Engagement)</u>: in which initial contacts with the community are expanded and consultations regarding the nature, scale and significance of the impact continued.
- <u>Step 3 (Community Planning)</u>: during which the LLRWMO and the community identify and assess alternatives for mitigating and managing the impacts over both intermediate and long term timelines.
- <u>Step 4 (Interim Management)</u>: as soon as dialogues with the community are started (i.e., early Step 2), the preliminary characterization data (i.e., Step 1 outputs) are evaluated to determine if interim actions are required to mitigate near term risks to public health and safety. These interim actions might include access controls and/or material consolidation and temporary containment activities.
- <u>Step 5 (Remediation)</u>: in which the outcomes of planning lead to the identification of a long term management option consistent with the community's constraints and objectives, and the development and execution of a remediation program that puts that management option in place. This step includes the transition/conversion of any interim facilities into the long term management plan.
- <u>Step 6 (Long Term Management)</u>: during which the actions needed to validate the performance of the management system (including any containment/storage structures associated with it) are taken. In addition, the maintenance activities required to ensure the long term physical integrity of the system are applied.
- <u>Step 7 (Closure)</u>: in which the outcomes of the remediation program are shared and celebrated with the community and ongoing activities/actions associated with long term management communicated.

Generally, the approach is aimed at gaining control and putting in place management of the contaminated materials as soon as possible, reflecting the appropriate level of concern and response.

NTR Remediation Chronology

The process outlined in Figure 3 has been applied in full or in part to a range of sites and former facilities on the NTR. Key remediation milestones for activities completed along the NTR are summarized below.

1978/79 Investigation and remediation of COSMOS 954 satellite crash impacts leads to identification of historic uranium ore impacts related to the NTR in Fort Smith, NWT.

1992 First comprehensive survey of potential ore impacts on transfer points and other facilities/equipment related to the NTR conducted by the LLRWMO.
Remedial activities initiated in the Lower Town area of Fort McMurray, Alberta.

1993 Interim consolidation of materials in a dedicated local mound constructed at the Tulita airport (400 m^3).
Initial stage of Fort McMurray Long Term Management Facility (LTMF) developed.

1993-95 Remediation of eight sites in Fort McMurray completed (30,000 m^3).

1998 Impacted soils from Sawmill Bay and Hay River consolidated and shipped to LLRWMO storage facility in Chalk River, Ontario (15 m^3).
Former Fort Smith NTCL warehouse containing radioactive building components demolished and transferred to a dedicated cell at the municipal landfill site for storage.

1999 The most active materials from the Tulita storage mound are segregated and shipped to the LLRWMO's Chalk River, Ontario, storage facility.

2001 Additional soils added to the Tulita mound from local lands identified by residents (a former over winter storage site).
Removal of impacted soils from three properties in Fort Smith and placement in the dedicated cell at the municipal landfill site.

2003 Remediation of the final property in Fort McMurray completed along with final phase of LTMF development.
Decontamination of two barges used for ore transport in Hay River and the dismantling of a third in Délı̨ne.
Attenuation of a local ore accumulation in Fort Fitzgerald via the placement of a sand cover.

2005 The Canada-Délı̨ne Uranium Table (CDUT) releases a series of some 26 recommendations for actions at Port Radium including one to initiate remedial activity at the site.

2006 Contents of Tulita storage mound reconsolidated into some 755 bulk shipping bags.

2007-08 Remediation completed at the Port Radium site by the federal Department of Indian and Northern Affairs (INAC).

2008-09 Tulita mound materials shipped via Hay River to final disposition at a commercial facility in the United States (800 m^3).

2010 The final pocket of ore-impacted soil in Fort Smith (Peregrine Street) was excavated and consolidated with other LLRW materials at the dedicated municipal landfill cell.
First phase of remediation (housekeeping/barrel crushing) initiated at Sawmill Bay. A joint federal activity will be the outcome at Sawmill Bay with INAC and NRCan sponsoring their respective activities.

2011 Interim consolidation of impacted materials impeding roadway improvements in Fort Fitzgerald, Alberta.

Remedial activities at two of the key sites referenced in the above chronology are described in more detail in the following sections.

Port Radium

The Port Radium Site is the original source of radium in Canadian pitch blende. It is also a former uranium and silver mine located on a peninsula along the eastern shore of Great Bear Lake in the Northwest Territories, 450 km north of Yellowknife and 265 km east of Délįne within the Sahtu Dené and Métis traditional lands. The site was decommissioned in 1982 to the standards of the day. Due to more than 40 years of mining, silver, copper and uranium were present in soils and surface water at the immediate site. The site also had waste rock and tailings containing radionuclides. Small amounts of hydrocarbons and asbestos residue were also present at the site. Physical hazards, such as mine openings, were the most immediate safety issues on the property.

Remediation work was completed at the site in 2007/08 and included:

- improving drainage to reduce leaching of silver, copper and uranium into soils and surface water around the immediate site;
- reducing gamma radiation levels by covering waste rock and tailings;
- removing small amounts of hydrocarbons and asbestos residue;
- covering exposed waste materials or moving them to a landfill on-site; and
- closing mine openings.

Long term monitoring is an important element of the Port Radium Remediation Plan. The first four years of monitoring included inspections intended to confirm that the site remains in a stable condition and that the remediation solutions are working. In year five of the monitoring program, a more detailed study of the site looked at the health of fish in the Great Bear Lake area around Port Radium, as well as soil and lake sediments. Finally, a complete gamma survey of the entire Port Radium Site will be completed to make sure that the radiation covers are working as designed (Indian and Northern Affairs Canada 2009).

Fort McMurray, AB - The NTR Terminus

The Fort McMurray Historic Uranium Cleanup Project involved the removal of some 42,000 m^3 of soils contaminated with uranium ores and ore concentrates from nine properties in the City of Fort McMurray, Alberta. These soils were placed into long-term management in a dedicated, locally developed and secure facility. The project was executed over a 10-year time period, involved the participation of the local community at critical junctures, and restored 28 ha of land to productive use. The remedial work completed at Fort McMurray was the largest and most comprehensive program completed on the NTR to date, and is given proportionate emphasis in this paper (Geddes et al. 2005).

Program Development and Planning

A program of public consultation and engagement was initiated to identify possible solutions to the uranium ore contamination and to guide the execution of remedial plans. A Working Group comprised of the local municipality, regulators, health authorities and the LLRWMO was maintained as work was planned and executed. The LLRWMO and the Working Group developed a solution that involved removing the materials from the subject sites and placing them into long-term management in a dedicated cell at the local municipal landfill site.

Cleanup Criteria
The principal contaminants of concern for the Fort McMurray Cleanup Project were radium, arsenic and uranium. The cleanup criteria adopted by the Working Group for these parameters were 0.1 Bq/g radium and 30 µg/g for both arsenic and uranium.

These criteria were selected on the basis of unrestricted future use of the lands, including residential use. While residential development was unlikely for the lands in question over the near term, it was felt that adopting relatively conservative criteria was an appropriate way of minimizing long-term land use restrictions.

Site Remediation
The remediation plan provided for mildly contaminated soil, described as Category B material, to be placed in the management cell constructed at the municipal landfill. Materials considered low-level radioactive waste, or Category A material, were shipped to the LLRWMO storage facility at Chalk River Laboratories in Chalk River, Ontario. Between 1993 and 1995, a total of eight properties in the Lower Town area were remediated (Figure 4). Cleanup of the last remaining property in the Waterways area was planned and undertaken between 2000 and 2003.

Figure 4: Cleanup Operations in Lower Town

The contaminated materials excavated from the subject sites (Figure 5) were transferred into long-term management at a dedicated facility constructed at the Fort McMurray landfill site (Figure 6). The management facility is a secure mound structure equipped with engineered containment, cover and leachate management systems.

Figure 5: Contaminated Soil Identification and Excavation

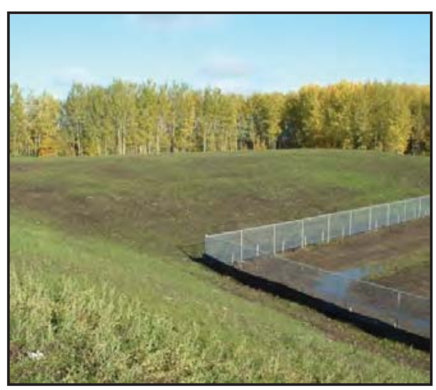

Figure 6: The Long Term Management Facility

The following features or protocols were applied as the LTMF was developed:

- Mound Design – the structure was developed with an engineered liner and cover, and a leachate collection and monitoring system;
- Contamination Control Procedures – a set of operating protocols designed to prevent the spread of both chemical and radioactive contamination and, as such, mitigate potential radiological impacts to the public, workers and the environment;
- Utilization of Dedicated Equipment – as part of the contamination control program, equipment was dedicated exclusively to site operations to ensure that all potentially contaminated items were maintained within a restricted area until such time as they had been monitored and decontaminated;
- Erosion Control – dust control protocols were applied to mitigate wind erosion and surface waters were directed and stored in ways that minimized sediment transport; and
- Monitoring – a comprehensive monitoring program was put in place to ensure that radiological exposures to workers and the public were maintained within acceptable limits.

Following completion of the LTMF, a program of long-term management, surveillance and monitoring was implemented to demonstrate that the facility is performing as designed and is in compliance with regulatory standards. This program was developed to fulfill the LLRWMO's obligations with respect to monitoring that are outlined in a legal agreement with the Regional Municipality of Wood Buffalo.

Consultation Program Overview

An effective process of engaging and informing the local community was always an integral component of the remedial program in Fort McMurray. A public participation program was implemented through the Working Group that guided the planning and implementation of the Project. In addition to the Working Group as key stakeholders, the consultation program extended into the community in several ways, including:

Figure 7: Open House Display

- local community consultation through one-on-one interviews and tracking of concerns and issues;
- information events (e.g., open houses) conducted in the community at appropriate project junctures (Figure 7); and
- media notices to advise the public about the project and its progress, and to provide details regarding pending information events.

The Result

The end result of the Historic Uranium Cleanup Project in Fort McMurray has been that nine properties covering about 28 ha have been made available for alternate uses. Many of these properties are in prime commercial locations, and as a consequence, have already been redeveloped into retail outlets. This rehabilitation was completed at minimal risk to the community and by devoting only 1.5 ha of non-productive land to the long-term management of contaminated materials.

ADVANCING THE NEXT PHASE OF REMEDIATION IN THE NORTH

The success of remedial programs to date on the NRT has created increased expectations in the remaining communities where contamination is still present. The Government of Canada continues to be committed to the remediation of this remaining contamination at sites in both the South Slave Region and in the Sahtu Region. Consideration of cleanup options for these areas had begun in 2007, prior to the completion of the Tulita cleanup, when NRCan and the LLRWMO convened a meeting of all government stakeholders in Yellowknife to discuss contamination issues and the process for moving forward. Since then, local communities have also come forward expressing interest and some urgency in advancing the cleanup. This has led to community meetings and the initiation of continuing dialogue with local leadership and others in both the Sahtu and the South Slave. Interest is rising and progress is being made.

CONCLUSIONS

For some 30 years, the LLRWMO and NRCan, along with their partners in other government agencies, and within the consulting community, have developed processes and precedents for managing the legacies of uranium and radium mining in Canada's north. This experience has demonstrated that the critical determinants of success are those efforts directed towards engaging the local communities to fully understand the locations, scale and significance of impacts, and in the development of plans for the long term management of those impacts. The Office's seven step process for working with communities from issue discovery to closure has provided a framework that has facilitated the effective mitigation of a large proportion of concerns generated by NTR legacies. Current activities underway with communities in the Sahtu and South Slave regions of northern Canada are applying these and related techniques towards moving the remaining NTR uranium and radium mining legacies in Canada's north to closure.

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PHYTOREMEDIATION OF PETROLEUM HYDROCARBON IMPACTED SOILS AT A REMOTE ABANDONED EXPLORATION WELLSITE IN THE SAHTU REGION, NORTHWEST TERRITORIES

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ABSTRACT

Sub-arctic remote locations pose a challenge for remediation of petroleum hydrocarbons. Selecting the best method that considers and balances each of the social, economic and ecological factors that are associated with the problems can be difficult. Cost and logistical difficulties with conventional remediation by hauling hydrocarbon impacted material off-site from an abandoned exploration well site in the Sahtu Region of the Northwest Territories (NT) led to the decision to remediate the soils on-site through the use of phytoremediation with plant growth-promoting rhizobacteria (PGPR). PGPR perform many functions in the soil, e.g., increased nutrient uptake, enhanced germination and tolerance to toxic contaminants, which can lead to enhanced plant growth and subsequent breakdown of hydrocarbons in the soil. The site was seeded in 2012 with an annual northern tolerant species mix and fertilized. Biochar, a carbon-rich by-product of pyrolysis, was added to select grids to determine if a soil enhancing amendment would have a positive impact on soil fertility and stability in an otherwise inhospitable growing medium. Results from 2012 show decreases in concentrations of benzene, toluene, ethylbenzene, xylene, F1 and F2 hydrocarbons. Remediation below guidelines is anticipated by the end of summer 2014.

Key Words: Plant Growth-Promoting Rhizobacteria, Biochar, *Delftia Acidovorans*, Well Site.

INTRODUCTION

Petroleum hydrocarbons (PHCs) are organic contaminants that can be persistent and are challenging to remediate to stringent environmental quality guidelines (EQG) using alternative techniques to conventional landfilling (Huang et al. 2005). Recently there has been extensive research on the use of phytoremediation utilizing plant growth-promoting rhizobacteria (PGPR) to enhance rooting zone biomass in plants; the PHCs are metabolized within the soil matrix. This approach allows the physical structure and biological properties to be preserved and soil fertility to be enhanced (Andreoni and Zaccheo, 2010). In addition, the use of phytoremediation at locations that are impractical to access by ground is an attractive remedial option to balance the economic, social and ecological factors of the project.

Cost and logistical difficulties with conventional PHC soil remediation (excavation, transport and landfill encapsulation off-site) led to a decision to remediate soils on-site by phytoremediation with PGPR at an abandoned exploration well site in the Sahtu Region of the Northwest Territories (NT). The abandoned well site is approximately 60 km south of Tulita, NT and is accessible only by air in the summer. In conjunction, as a secondary goal, this project is part of a research program to make PGPR a commercially viable and available product for consultants and industry to use on future reclamation and remediation projects. This paper presents the first year results of a phytoremediation program of PHCs at an abandoned well site in the Sahtu Region, NT.

FACTORS AFFECTING REMEDIAL OPTIONS

Various factors were considered when determining the best remedial method for the site. Ensuring that an environmentally sustainable approach was implemented that encompassed the future protection of traditional harvesting areas, and that could be completed in a reasonable amount of time, and cost were important in the design of the overall remedial approach. Stakeholder engagement occurred with the local communities of Norman Wells and Tulita and the Sahtu Land and Water Board (SLWB). Aboriginal Affairs and Northern Development Canada (AANDC) aided in the overall plan development to ensure that any community and stakeholder concerns were addressed upfront. Significant consideration was given to the remoteness and air-only accessibility of the well site which prohibited the removal of impacted soil. Additionally, sourcing suitable clean fill would be difficult since it involves quarrying (disturbing more land), beyond the boundaries of the well site which was not acceptable to the stakeholders. Therefore, on-site remediation of impacted soils was selected.

METHOD

Phytoremediation Scientific Principals

Plants release nutrients into the soil which can be utilized as energy sources by microorganisms and biomass produced associated with plant roots/rootlets promotes and sustains microbial activity in the rhizosphere. If hydrocarbons are present in the rhizosphere, microorganisms will metabolize the hydrocarbons for energy (Atlas 1981). Biodegradation occurs as the microorganisms metabolize and use the carbon in the hydrocarbon chains as an energy source for cell production and growth; the hydrocarbon bonds are degraded and broken down into less harmful substances. Although biodegradation can occur naturally, the challenge is maintaining the microbial abundance and biomass in the rhizosphere under nutrient-deficient conditions in the presence of the contaminants (Huang et al. 2005).

Optimizing the phytoremediation process has included developing techniques that protect the plant from the contaminants that can arrest photosynthesis, plant respiration and metabolism resulting in reduced biomass production. When plants are stressed, they produce ethylene, a phytohormone, which invokes a stress response, leading to a reduction in plant and root growth and biomass in the rhizosphere (Saraf et al. 2011), which then inhibits microbial activity and, ultimately, hydrocarbon degradation.

PGPR increases the bacterial activity of 1-amino-cyclopropane-1-carboxylic acid (ACC) deaminase (an enzyme that catalyzes the removal from the plant of ACC, the immediate precursor to ethylene) in the rhizosphere which regulates the plants' production of ethylene (Bhattacharyya and Jha 2012; Saraf et al. 2011). PGPR also synthesizes and introduces other phytohormones consisting of auxins, gibberellins, cytokinin, and abscisic acid which regulate the growth of the plant (Bhattacharyya and Jha 2012; Saraf et al. 2011).

The combination of increased ACC deaminase activity and the production of auxins by the PGPR increases the plants' tolerance to contaminants; promotes increased plant and root growth and biomass production in the rhizosphere; and promotes the release of plant enzymes (e.g., oxidases and hydrolases) that breakdown PHCs, and exudates that stimulate microbial growth (Andreoni and Zaccheo 2010). The combination of these processes leads to a greater proliferation in microbial activity and a very active rhizosphere, which increases degradation and metabolism of hydrocarbons by the microorganisms. The hydrocarbon molecules are metabolized by the microorganisms for energy use which creates end products of carbon dioxide (CO_2), cell mass and water (Andreoni and Zaccheo 2010). PGPR also provide added protection to the plant against cold, drought and phytotoxic compounds that would otherwise inhibit plant and biomass production.

Implementation Logistics and Site Preparation

All equipment and materials for the remedial program needed to be heli-portable. Equipment, materials and fuel to construct a suitable pad and excavate the soils were not available locally, so were required to be trucked to Fort Simpson, NT then barged down the Mackenzie River to the closest staging area. A medium-lift helicopter was used to transport equipment, materials and personnel to the site. Local labourers, equipment operators and wildlife monitors were hired to support the program.

In the absence of a suitable quantity of low permeability clay deposits, and to limit the amount of disturbance to surface soils and re-vegetated areas at the well site, the phytoremediation pad was constructed from imported material, i.e., timber and a 30 mil linear low-density polyethylene (LLDPE) liner. The LLDPE liner was laid directly on the ground surface to contain the PHC-impacted soil during treatment and prevent impacted soils from cross-contaminating the underlying ground. At the edge of the liner, timber supports were constructed and erected to a height of about 1 m above ground level. The liner was extended over the top of the timber supports to form a perimeter berm. The liner was secured to the rail supports with nails and staples.

To allow moisture and water to infiltrate and excess water to drain away from the soil, the pad was oriented to allow run-off water to collect in the lowest corner. Continuous monitoring of the run-off collection was not feasible due to remoteness of the location, therefore, two RainDrain™ filters (a carbon passive system designed to trap hydrocarbons while allowing water to pass and be released outside the berm) were installed through the berm liner at the lowest corner of the pad.

In late June 2012, PHC-impacted soils were excavated from the vicinity of well centre and transported to the phytoremediation pad, constructed only 15 m east of well centre to minimize soil transport distances as heli-portable equipment and wet ground conditions slowed the movement of large volumes of soil. The impacted soils were directly loaded into the pad. The excavation was advanced to the top of the permafrost (approximately 1.6 m below ground level) over an area of approximately 105 m^2 (11.1 m x 9.5 m). An estimated volume of 170 m^3 of PHC-impacted soil was removed from the source area. On completion of the excavation work, a wire-mesh panel fence was erected around the open excavation to prevent access by resident or transitory wildlife. The excavation remained open to return the native soils back to the original location upon successful remediation.

Once the impacted soil had been stockpiled on the pad, the soil was spread to achieve a maximum thickness of 50 cm. The soil thickness of up to 50 cm was designed to allow for optimum conditions for root penetration to the bottom of the impacted soils therefore decreasing the duration of remediation. During this procedure, large woody debris and rocks were removed from the soil.

The soil was mechanically rototilled to break down the soil to smaller conglomerates and substrates to improve the physical structure of the soil, and create voids for airflow (soil aeration), water movement, greater treatment penetration/contact and increase soil exposure to sunlight for photo-oxidation.

<u>Soil Amendments, Fertilization and Forage Species</u>
Biochar (or charcoal) a carbon rich by-product of pyrolysis, was added to select grids to determine if a soil enhancing amendment would have a positive impact on soil fertility and stability in an otherwise inhospitable growing medium. Biochar is a solid material obtained from the carbonization of biomass, such as plants. Biochar has a large number of micropores and a high surface area that enhances the habitat for micro-biota and their propagation (Maraseni 2010), thus making it attractive for PGPR colonization in the rhizosphere. This product was also selected since laboratory research and field applications have demonstrated that it can improve plant yields by increasing soil fertility; stabilizing soil structures; and reducing nutrient leaching, soil acidity, irrigation and fertilizer requirements (Maraseni 2010). Biochar is also considered carbon negative, that is, it can be retained in soil long-term, and can sequester CO_2, nitrous oxide (N_2O) and methane (CH_4) emissions from soils, thus further reducing greenhouse gas emissions (Maraseni 2010).

The fertilizer selected initially for use at the well site was a commercially available 18-24-12 since it provided a balanced mixture of nitrogen, phosphorus and potassium and the soil nutrient conditions were unknown. It has been demonstrated (through field application) that its use encourages plant establishment and promotes early root development and growth.

The forage species used were non-invasive, annual species that are hydrocarbon tolerant (helpful with the short growing season imposed by northern climates). The blend consisted of equal portions of annual ryegrass (*Lolium multiflorum)*, slender wheatgrass (*Elymus trachycaulus)*, creeping red fescue (*Festuca rubra)* and Canada wild ryegrass (*Elymus canadensis)*.

As the secondary goal was to demonstrate PGPR as a commercially viable and available product for reclamation and remediation, the impacted soil on the pad was divided into the following four sections:

- *Section 1*: to assess the effectiveness of PGPR on hydrocarbon degradation.
- *Section 2*: to assess the effectiveness of PGPR on hydrocarbon degradation and to determine if the addition of biochar would enhance the performance of PGPR and hydrocarbon degradation.
- *Section 3*: to assess if the addition of biochar leads to enhanced hydrocarbon degradation.
- *Section 4*: to be used as a control, to assess if no treatment has an effect on hydrocarbon degradation.

Plant Growth-promoting Rhizobacteria (PGPR) Species and Application
A commercially available PGPR product called BioBoost™, a non-pathogenic, non-genetically modified bacterium containing *Delftia acidovorans,* is being used for the project. It was selected, as it was the product of study for the partnership research company and it is one of several PGPR products that are currently being researched for commercial application for soil remediation projects. Laboratory and small scale field trials completed on canola crops have demonstrated that the application of BioBoost™ has led to a 5% increase in plant yields and associated biomass production in the rhizosphere and has a known ability to biodegrade hydrocarbons (BrettYoung 2012).

Following seed germination, BioBoost™ was applied using a backpack sprayer to the soils pre-treated with fertilizer and biochar (in Sections 2 and 3). Early seed germination and root development was required to provide a surface upon which the BioBoost™ could locate, attach and accumulate.

FIRST YEAR RESULTS

The phytoremediation pad was divided into a 12 point grid to ensure the same locations for samples could be maintained for comparison. In July 2012, 12 soil samples were collected SP01 to SP12 (Figure 1) to establish baseline hydrocarbon conditions. In October 2012, 12 samples were collected from SP01 to SP12 (Figure 1) to assess the performance and effectiveness of the phytoremediation over the first season.

Under the Canadian Council of Ministers of the Environment (CCME) environmental quality guidelines (EQG) there are four generic land use classifications (CCME 2006). Considering the remote location of the well site, and the applicable exposure receptors, the residential/parkland land use was selected for Tier 1 EQG as the initial remedial endpoint. This land use was selected over the more stringent agricultural land use due to the low future potential that the well site would be utilized for livestock tendering or crop production (CCME 2006).

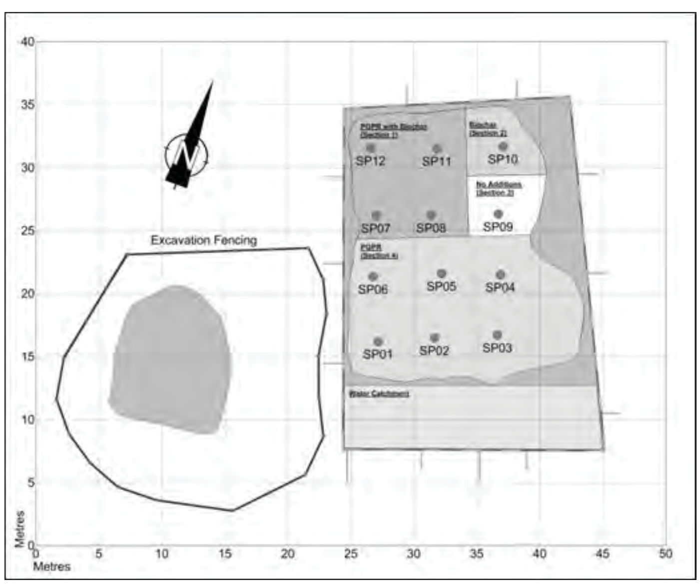

Figure 1. Excavation and Phytoremediation Pad Layout

Benzene, Toluene, Ethylbenzene, Xylenes (BTEX)

BTEX constituents that were reported above the EQGs in July had fallen below the EQGs in October at the same sampling points, with the exception of toluene in SP01 where the laboratory detection limit required adjustment above EQG due to moisture content. This suggests that the soils at these locations were remediated, likely by volatilization. Light end hydrocarbons with high vapour pressures have an increased tendency to be readily diffused from the soil matrix to the atmosphere. Volatilization was likely facilitated by soil movement and aeration from rototilling, or by the influence of the four treatment scenarios.

F2 (C_{10}-C_{16}) Petroleum Hydrocarbons

F2 PHCs were reported above the EQGs in SP01 to SP12 in July and, and with exception of SP11, in October, 2012. Although F2 PHC concentrations remained above the EQGs in October, there was a marked reduction in F2 PHC concentrations in SP01 to SP09. In SP11 this reduction led to a drop in F2 PHC concentrations to below the EQGs. The decrease in F2 PHCs for SP01 to SP09 ranged from 57% in SP07 to 95% in SP06 (Figure 2). Average F2 PHCs percentage reduction was approximately 75% over the 3 month remediation period. The reduction in F2 PHC concentrations is likely the result of a combination of factors, including some volatilization resulting from the movement and rototilling of the soil, soil homogenization, influences from the four treatment scenarios and phytoremediation.

The only exceptions to the downward trend in F2 PHC concentrations were in observed SP10 and SP12 which reported a 56% and 107% increase, respectively (Figure 2). It is possible that this variation reflects the natural heterogeneity of the soil between sampling events or the sample contains a higher abundance of biogenic organic compounds that may have biased the analytical results.

In terms of variations in the rate and effectiveness of phytoremediation associated with the four treatment sections, it appears that varying the treatment conditions had little effect on remediation performance. Consistent reductions of similar magnitude in F2 PHCs were recorded across the pad even in *Section 3* (SP09) which acted as the control location. At this stage in the remedial program, a similar pattern of reduction in the control (SP09) and the other three treatment sections suggest that physical handing, homogenizing and exposing the soil microorganism to the atmosphere (oxygen) for the three month period following preparation alone may have had more of an influence on F2 PHC reductions rather than the application of the various treatments. If the effects of the treatments are to be realized it is expected that the control will remain stable and the F2 PHC concentrations at the other treatment scenarios should decline.

With the addition of PGPR to SP12 in *Section 1* it is anticipated that F2 PHC concentrations should decline at this sampling location. For SP10, located in *Section 2* with biochar and the documented improvements to soil fertility, structure and acidity, there should be future F2 PHC reductions, albeit to a lesser rate and degree than those sections treated with PGPR (*Sections 1* and *4*).

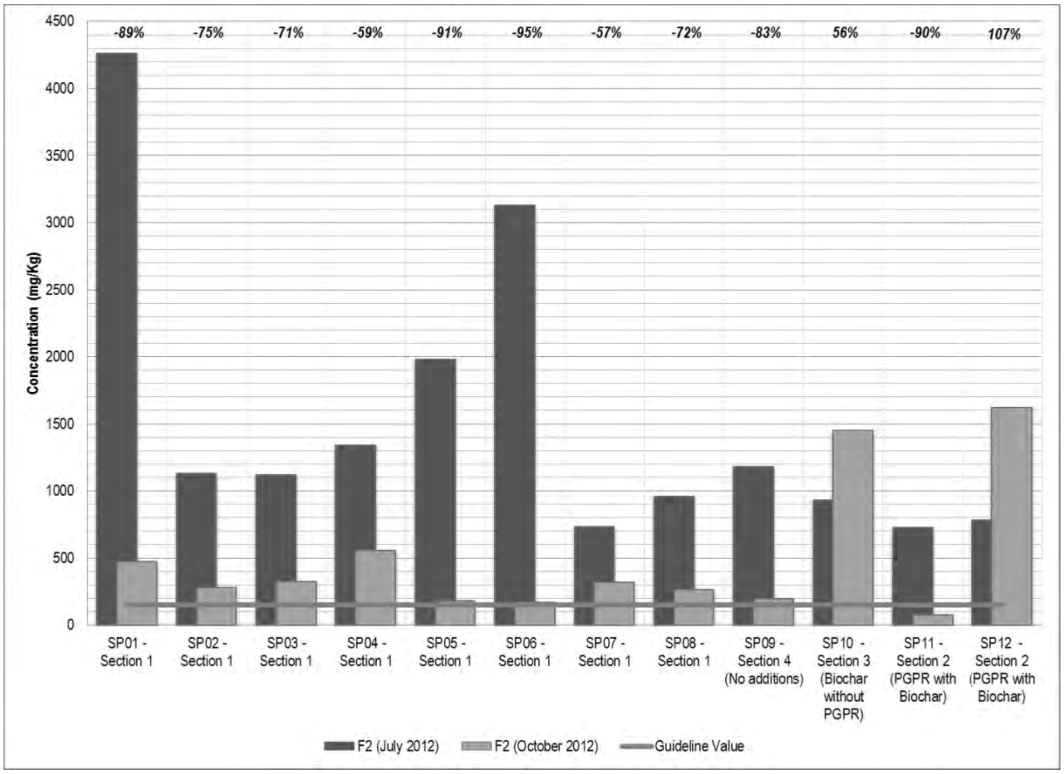

Figure 2. A comparison of F2 (C10 – C16) soil concentrations between July 2012 and October 2012

F3 (C_{16}-C_{34}) Petroleum Hydrocarbons

F3 PHC was reported above the EQGs in SP01 to SP04 in July 2012. Noticeably the F3 PHC concentrations increased between the July and October sampling events in SP02 and SP04 to SP12, with new exceedances of EQGs in SP05, SP08 and SP12. This percentage increase in F3 PHCs at these locations ranged from 1% in SP06 to 296% in SP12 (Figure 3). The average F3 PHC increases were about 83% over the 3 month remediation period.

Although the increase in F3 PHC concentrations may be attributed to the natural soil heterogeneity between sampling events, (a common issue in heavier end hydrocarbons which tend not to homogenize as well as lighter end hydrocarbons), it is more likely that the F3 increase is a result of biogenic interference between sampling events. The CCME method for F3 PHC analysis reports a single concentration for carbon bonds C_{16}-C_{34}. The reported concentration includes any type of carbon compound with 16 to 34 carbon bonds which can include both biogenic (e.g., plant alkanes, sterols, fatty acids and waxes) and petrogenic (e.g. aromatics, olefins and asphaltenes) compounds.

Previous site assessment findings have indicated that the soils being remediated are peaty and high in organic matter content (which is typical of the soils in the region). Further, with the use of PGPR and application of fertilizers and biochar there will likely be a greater abundance of biomass and plant material (biogenic) in the rhizosphere which will provide an advantage to the degradation of petrogenic compounds in the soil. However, it is hypothesized that due to the high amount of biogenic compounds in the soil, the biogenic compounds are creating a false positive result when analyzed for F3 PHCs.

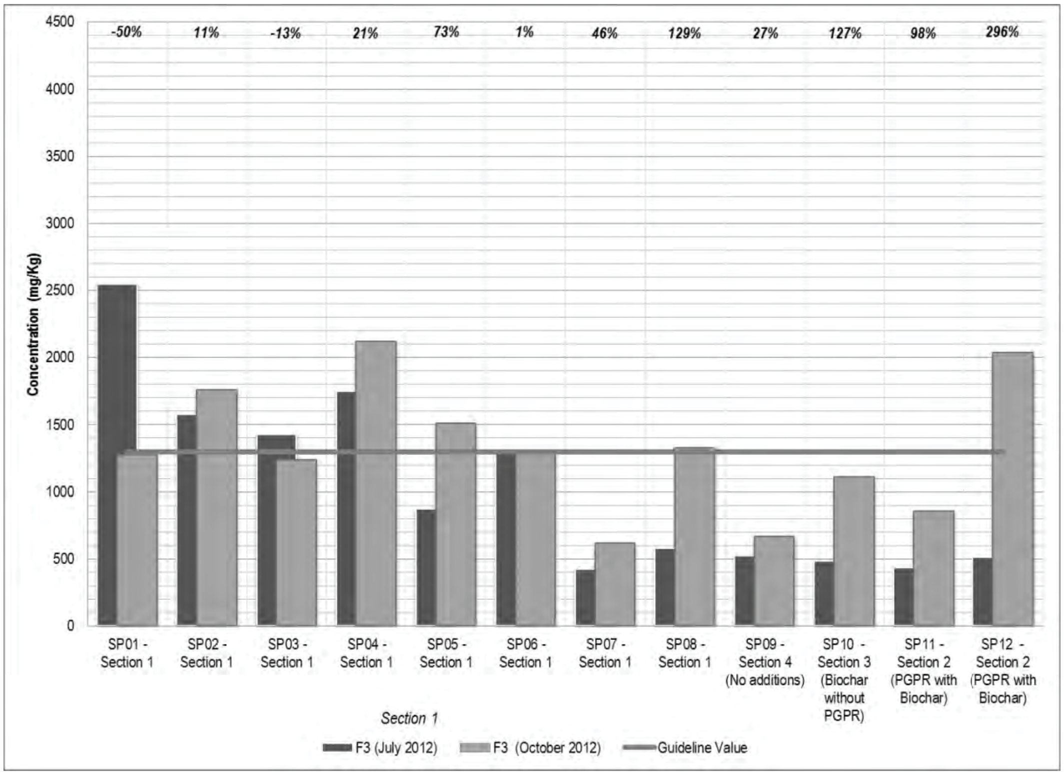

Figure 3. *A comparison of F3 (C16 – C32) soil concentrations between July 2012 and October 2012*

Supporting this theory is that biogenic content would have been lower at baseline conditions in July 2012, then with the addition of PGPR, fertilizer and biochar, there would have been a proliferation of microbial activity and biomass production in the soil which would have resulted in an increase in F3 PHCs in the October 2012 sampling event. This increase between sampling events is evident in the analytical data. Moreover, when looking at the locations where biochar was applied, the percentage increases in these sections (*Sections 1* and *2*), was higher than in sections that did not receive the biochar amendment (*Section 3* and *4*).

Since heavier end hydrocarbons are more complex (they have more carbon chains and structures than light end hydrocarbons) it is likely that the F3 PHC fractions will take longer to degrade (for their concentrations to decline). Also, unlike the lighter end hydrocarbons, F3 PHC have a lower vapour pressure and are less likely to volatilize so are less likely to respond to degradation through physical handing and will require microbial breakdown induced by the phytoremediation process. The only exceptions to the increasing trend in F3 PHCs were in SP01 and SP03 (*Section 1)* which reported a 50% and 13% decrease, respectively, in F3 PHC concentrations (Figure 2). It is possible that this reduction is a reflection of soil heterogeneity between sampling events, but may also reflect the effects of phytoremediation.

In terms of variations in the rate and effectiveness of phytoremediation associated with the four treatment sections, it appears that, with the possible exception of two sample points in *Section 4,* varying the treatment conditions had little effect on remediation performance between the four scenarios. Consistent increases of similar magnitude in F3 PHC were recorded across the pad.

At this stage in the remedial program, a similar pattern of increase in the control (SP09) and the other three treatment sections suggest that physical handling, homogenizing, treatments and exposing the soil microorganism to the atmosphere (oxygen) has had little to no positive effects on F3 PHC reductions. If the effects of the treatments are to be realized it is expected that the control will remain stable and the F3 PHC concentrations in the other treatment scenarios should decline varying.

With respect to the F3 PHC reductions in SP01 and SP03 in *Section 1*, although this may be heterogeneity between sample events it may also reflect the application of PGPR in this section. As mentioned above as the control sample in SP09 received no treatment and had an increase in F3 PHC concentrations compared to SP01 and SP03 (*Section 1*) which did receive treatment and registered a F3 PHC reduction, it could be that the addition of PGPR in *Section 1* is the factor at play in reducing F3 PHC concentrations at these two sample points.

CONCLUSIONS

In the implementation of the phytoremediation program at the well site, there were reductions from baseline BTEX and F2 PHC concentrations (above the EQGs in sample points in July, 2012), which were reported below the EQGs at the end of the first remediation season. The exception to this pattern was for F2 PHCs in SP10 and SP12 which reported an increase above the EQGs at the end of the first remedial season in October 2012. For F3 PHCs, with the exception of SP01 and SP03, all F3 PHC concentrations (above the EQGs at baseline) increased in concentration at the end of the first remediation season. For both F2 and F3 PHCs, there did not appear to be any overall significant variations in the effectiveness of the remediation between the four treatment sections. After the first season no improvements were directly attributed to the PGPR; however, it is assumed that degradation will be more evident in subsequent years.

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BIOENGINEERING TECHNIQUES FOR REVEGETATION OF RIPARIAN AREAS AT THE COLOMAC MINE, NWT

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ABSTRACT

Factors such as nutrient poor soils, harsh climate, remote locations, and high costs make revegetating disturbed areas in northern environments a challenge. We present a case study where novel bioengineering and project planning techniques were employed to revegetate and remediate riparian areas at Colomac Mine, an abandoned gold mine 220 km north of Yellowknife, NT. The revegetation plan focused on establishing pioneer species and facilitating natural recovery and succession. A 'rough and loose' technique was used to allow the soil to capture and retain moisture, trap windborne seed, promote easy root penetration and prevent erosion. Harvesting and planting of local willow cuttings, alder seeds, and sedge plugs ensured that the vegetation at these sites was adapted to local climate and soils. Multi-year monitoring was initiated which included vegetation counts and photographic documentation. Initial results have shown success rates of 60-100% plant survival on the majority of areas where bioengineering techniques were used. In contrast, poor revegetation success rates of 8-33% plant survival were experienced in areas where techniques were either used incorrectly or implemented too late in the season. The bioengineering techniques implemented at Colomac Mine provided a successful, cost effective, and local approach to revegetation in a northern environment.

Key Words: Revegetation, Riparian, Bioengineering, Remediation, Monitoring

INTRODUCTION

Mining operations result in residual chemical and physical impacts to the landscape. Often, the primary remedial objectives for mine site closure focus on the chemical and physical hazards, with minimal consideration to rehabilitation or restoration of the environment to a more natural state. However, integrating revegetation activity with remediation work has been garnering more recognition, as the overall sustainability of remediation projects is being recognized. In Canada's Northwest Territories (NT), many abandoned mine sites are under federal responsibility and require remediation. The high cost of remediation in remote locations and the harsh growing conditions make revegetating disturbed areas in northern environments a challenge.

Revegetation strategies for large disturbed areas have evolved to focus on the use of natural processes to initiate and speed up the recovery of natural plant succession. Soil bioengineering techniques for site preparation, and the use of locally collected pioneer species for revegetation are being used as a cost efficient approach to restore vegetation that is compatible with the surrounding habitat at disturbed mining sites (Polster 2011a). These approaches and techniques were used for revegetation along the shoreline and riparian areas during remediation at the Colomac Mine.

SITE DESCRIPTION

Colomac Mine (Colomac), a former gold mine located approximately 220 km north of Yellowknife, NT, Canada (64º 23' 42" N // 115º 07' 16" W), is a contaminated site under the custodial responsibility of Aboriginal Affairs and Northern Development Canada (AANDC). AANDC has managed the site through the Federal Contaminated Sites Action Plan (FCSAP) since 1999. The site is situated between Baton and Steeves Lakes and is surrounded by numerous small lakes. Access to the site is by air only. The primary infrastructure of the mine was located along the shoreline of Steeves Lake.

During operations, natural drainages and lakes on the mine site were in-filled with waste rock. Historic petroleum hydrocarbon releases adversely impacted the sediments along Steeves Lake shoreline and required remedial action. The remediation option selected involved the construction of a berm to contain the hydrocarbons and the capping of impacted sediments along 750 metres of the shoreline. The construction of this containment berm and cap impacted fish habitat in the shoreline area and required implementation of a revegetation plan for several areas, including the new Steeves Lake shoreline, the restored Truck Lake to Steeves Lake Shoreline, Riparian and Wetland Area (Truck Lake Channel), and the restored Dam 2 Drainage Riparian Area (Dam 2).

REVEGETATION TREATMENT AREAS

Steeves Lake Shoreline

The constructed shoreline along Steeves Lake is an engineered system of hydrocarbon containment and filtration covering approximately 750 m of the impacted shoreline. The width of the constructed shoreline varies according to the extent of sediment impact, creating a total area of 10,218 m^2 which was covered with a mixture of peat and silty sand. An armoured 1 m by 1 m by 750 m trench was constructed along the outer edge of the newly constructed shoreline. The trench was lined with landscape fabric and filled with peat and silty sand to provide a substrate suitable for revegetation (Figure 1). After construction, remnant alder (*Alnus sp.*), black spruce (*Picea mariana)*, white spruce (*Picea glauca)*, willow (*Salix* sp.), sedges (*Carex sp*.), Labrador tea (*Ledum groenlandicum*) and bog rosemary (*Andromeda polifolia*) remained along the original shoreline.

Truck Lake Channel

Truck Lake Channel was constructed to reconnect Truck Lake and Steeves Lake. The revegetation objective was to re-establish approximately 7,300 m^2 of riparian vegetation along the channel and the Truck Lake shoreline. It consisted of a 240 m long meandering constructed channel bordered by rocky terrace zones, which were covered with a light layer of peat and silty sand, and upland benches of original ground (Figure 1). A short braided stream and wetland catchment area of peat and silty sand was constructed at the Truck Lake outlet. Along the Truck Lake shoreline, waste rock fill was removed to expose the original shoreline which was then covered with peat and silty sand.

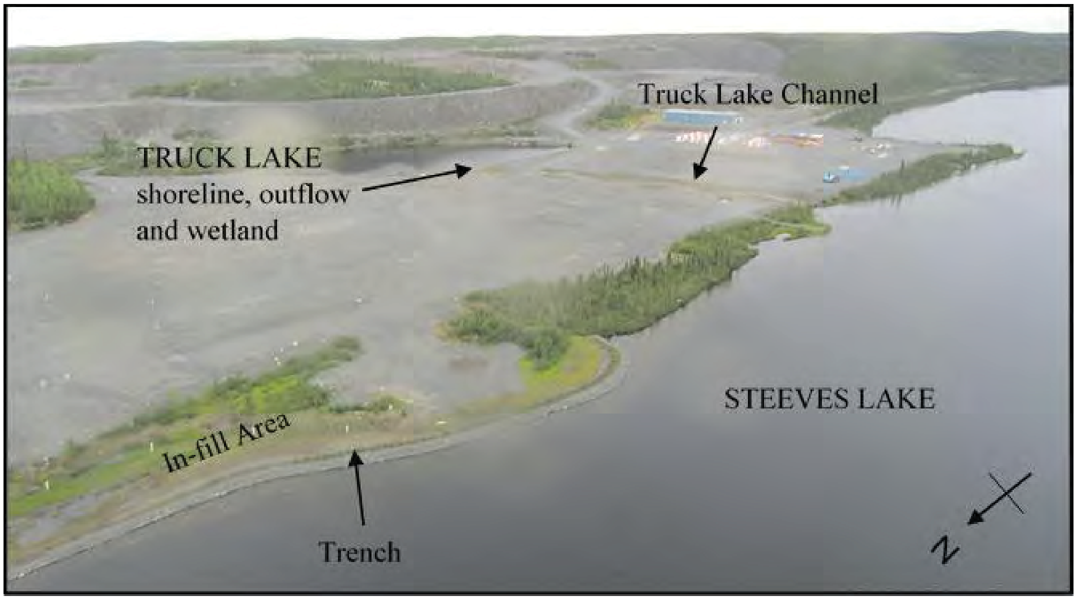

Figure 1. Arial view of the south end of Steeves Lake Shoreline and the new channel connecting Truck Lake to Steeves Lake. Dam 2 site is located 5.5 km NE from Truck Lake Channel outlet into Steeves Lake.

Dam 2 Drainage Riparian Area (Dam 2)
The Dam 2 channel flows from Tailings Lake to North Pond at the northern end of the site and was constructed as a 140 linear metre meandering channel with ten riffles.

REVEGETATION APPROACH

Identifying and emulating natural conditions at the site to be restored can assist in natural recovery solutions (Polster 2010). Understanding the naturally occurring pioneer and successional species in the surrounding environment becomes an important step in determining a suitable revegetation approach. Early in the planning phase, the need for expertise in revegetation strategies and local capacity were identified as limitations. An expert in revegetation, David Polster, was engaged and came to the site to provide recommendations and two days of hands-on training on revegetation approaches. Training ensured that all project personnel understood how to properly implement the revegetation plan and techniques.

Identification of limitations to the establishment of vegetation at the site was an important first step in determining the appropriate site preparation method (Clewell and Aronson 2007, Polster 2011a). Limitations to natural recovery at the Colomac revegetation sites included extensive areas of compacted waste rock with no soil, low precipitation, extreme cold temperatures, and a short growing season (Polster 2010). The areas to be revegetated consisted of waste rock aggregate covered to the extent possible with a peat and silty sand blend to re-establish the organic soils necessary for recovery of vegetation (Aboriginal Engineering 2010). Peat material for remediation and construction activity was salvaged from other areas on the site that had been disturbed.

Once the sites were prepared, the revegetation treatments for the riparian areas focused on the establishment of locally collected willows, alders and sedges. These plants develop extensive root systems that stabilize slopes by slowing and redirecting the flow of surface runoff, reduce sediment transportation and minimize the impact of raindrop erosion (Polster 2011b). Willows and alder are pioneering species adapted to growing in nutrient poor soil conditions. Sedge plugs were transplanted along the channel edges and wet areas to quickly establish riparian vegetation. Harvesting and planting of local willow cuttings, alder seeds and sedge plugs ensured that the new vegetation at these sites was adapted to local climate and soils (Polster 2011a).

The soil surface was mechanically disturbed to promote natural revegetation using the 'rough and loose' technique. The technique is implemented with an excavator which creates a checkerboard of small holes and hills, breaks up the substrate creating an environment which will capture and retain moisture, trap windborne seed, promote easy root penetration and prevent erosion (Poster 2011b). Two rough and loose trial plots were established, one at Steeves Lake Shoreline and the other at the Truck Lake Channel, to determine the effectiveness of this technique and the impact it would have on construction design and hydrocarbon containment in these areas.

REVEGETATION TREATMENT APPLICATIONS

Willows (*Salix sp.*) were used as they are a quickly established woody pioneer species and harvesting has little impact on the environment because only the stems are cut and replacement shoots grow quickly from the undisturbed crown roots. The cuttings are best harvested in the fall when they are dormant and their stored carbohydrates are at a maximum so they can benefit from the higher moisture availability during freshet (Polster 2011b). The harvested willows were trimmed and soaked in Steeves Lake for at least 6 days before planting. The willows were cut into 1.2 m lengths and planted using an excavator to pull back the substrate to a depth of 1.0 m and placing five cuttings into the hole (Gravel Bar Method). Only single cuttings were planted in the Steeves Lake 'rough and loose' trial plot and Truck Lake Channel wetland area. Once planted, the new roots and shoots grow from the auxiliary buds on the cuttings (Polster 2011b).

Alder (*Alnus sp.*), a nitrogen fixing plant, germinates easily from seed (Polster 2011b). Seeds from cones harvested in the fall were spread on the appointed sites. The seeds are easily collected, very light weight and stored refrigerated, making this a cost effective method to treat large areas (100 g of seed per hectare).

Sedge (*Carex sp*.) plugs have the benefit of carrying seeds and roots from many other plants which can germinate and increase wetland plant diversity. Sedge plugs were transplanted to re-establish wetland vegetation in low-lying wet areas and along the banks of the new channels. The plugs were harvested from a nearby wetland. Less than 10% of the plants at the donor sites were harvested to minimize impacts.

The shorelines, channel terraces and upland benches were lightly seeded by hand with native grass seed mix (15 kg of seed/ha): *Poa glauca, Festuca saximontana***,** *Festuca brevissima (Festuca ovina* ssp.

alaskana), Elymus alaskanus [ssp. *latiglumis*] and the quickly germinating annual rye, *Lolium multiflorum.*

Steeves Lake Shoreline

The shoreline trench was treated with five 1.2 m long willow cuttings every 1.5 to 4 m and hand spread alder seed. The in-fill area between the constructed berm and the old shoreline was lightly seeded with alder and native grass. In addition, sedge plugs were planted in low lying wet areas. A 100 m^2 rough and loose trial plot was located at the south end of the of the in-fill area (Figure 1).

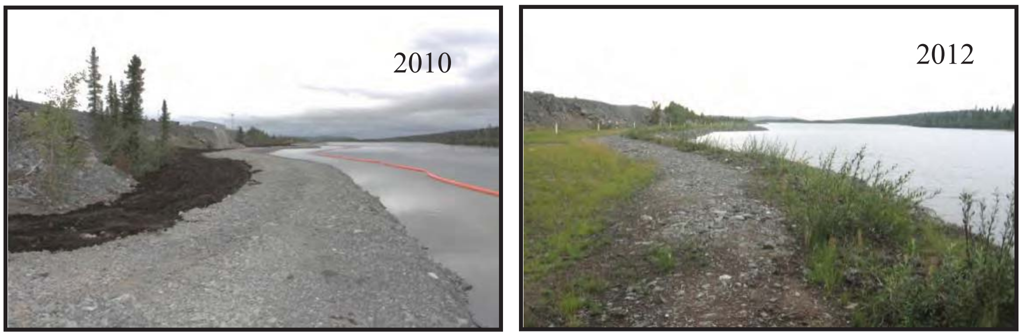

Figure 2. Steeves Lake shoreline before (2010) and after (2012) revegetation efforts.

Truck Lake Channel

The wetland catchment area was 'rough and loosed', however, on the channel terrace the waste rock material collapsed in on the excavated holes and the checkerboard pattern of depressions and mounds could not be maintained. Erosion control mat was laid along the Truck Lake Outflow banks and islands to prevent bank erosion during spring freshet (Figure 3). Willow cuttings were planted with an excavator along the Truck Lake shore and channel, sedges were planted in selected areas along the channel and the whole area was seeded with alder and native grass.

Figure 3. Top pictures, Truck Lake Channel before (2010) and after (2012) revegetation efforts. Lower pictures, Truck Lake Channel outflow area before (2010) and after (2012) revegetation efforts.

Dam 2 Drainage Riparian Area (Dam 2)
In 2010, construction of the Dam 2 drainage was finished and water flowed in the channel. Revegetation included planting 40 sedge plugs in the channel and sowing native grass and alder seed along the channel terraces and upland banks (Figure 4).

Figure 4. Dam 2 channel before (2010) and after (2012) revegetation efforts.

MONITORING

A monitoring plan was developed to evaluate the success of the revegetation effort over time. Monitoring was conducted over the first two seasons and subsequent monitoring will be done each year for five years and again in the 10th year after planting (AANDC 2013). The focus of short-term monitoring was on the success of the revegetation work itself in terms of the survival and growth of the target pioneering species and coverage. Future assessment will focus on assessing successional trajectories, ecosystem function and structure, to measure the change over time, and evaluate the overall sustainability of the revegetation effort in these disturbed areas. Photographs before re-vegetation and each successive year will provide a good visual record of plant growth.

RESULTS

Monitoring of plant survival and growth in 2012 showed that the majority of the re-vegetation areas, where the soil bioengineering techniques were used, met or exceeded expected success rates for survival and coverage. Assessments of the Steeves Lake Shoreline and Truck Lake Channel areas were completed in August of 2011 and 2012; however, the remote location and decommissioned road precluded access to the Dam 2 site in 2011. The short-term revegetation success was determined by the number of planted willow cuttings that sprouted shoots, the number of sedge plugs that survived the transplant and the percentage of each site covered by vegetation (Tables 1 and 2).

Planting Success
Willows were most successful along the Steeves Lake Shoreline Trench – 69% survival in 2011 and 60% survival in 2011, where the soils had been 'rough and loosed' and sufficient moisture was available (Table 1). The sedges were most successful in the Steeves Lake Shoreline In-fill Area and the Dam 2 Site

showing 100% survival where a wetland environment existed and were less successful in the Truck Lake Channel and Outflow showing 75% and 83% survival rates where water in the channel dried up in the summer resulting in the transplanted sedges dying or struggling (Table 1). The goal was to have 2 alder seedlings germinate per metre square across the site. In 2011, the alder seedlings were very small (< 1.0 cm). By 2012 these seedlings had grown to 1-10 cm. They were most successful in the areas that were 'rough and loosed' in the Steeves Lake Shoreline and the Truck Lake Outflow Areas. They were absent at the Truck Lake Shoreline where the soils were compacted. Native grass seed germinated at all sites and were most prolific where the soils were 'rough and loosened' and moist (Table 2).

Table 1. Survival rate of the willow cuttings and sedge plugs planted in 2010. Percent success was calculated based on the number of cuttings/plugs planted. 1 ND: no data, unable to access site in 2011 (FRC 2012).

Site	Zone	Plants	Plant Location	Number Planted 2010	Survival				Observations
					2011		2012		
					#	%	#	%	
STEEVES LAKE SHORELINE	Wetland	Sedge plugs	Wet areas	52	52	100	52	100	Growing vigorously and spreading.
	'Rough and Loose' Trial	Willow cuttings	Cuttings planted singly & gravel bar method	80	73	91	73	91	Willow cuttings not growing as well as those in the Trench.
	Trench	Willow cuttings	Planted every 1.5 - 4.0 m along the trench	1125	780	69	684	60	Growing well (shoots up to 2 m long). Some cuttings (2011) were destroyed during installation of drainage channels.
TRUCK LAKE CHANNEL	Shoreline	Willow cuttings	Gravel bar method	350	30	8	30	8	Low success could be due to being planted late in the season & not deep enough (Nov 2010).
	Outflow and Wetland Catchment	Willow cuttings	Gravel bar method along channel	85	28	33	28	33	No cause of low survival rate, those that survived were doing well in 2011 & 2012. Shoots grew from willow cuttings used to stake ECM.
		Sedge plugs	Channel edge	34	28	83	28	83	The sedge plugs that survived are doing well and spreading.
	Channel	Willow cuttings	Gravel bar method above channel	235	103	44	89	38	Cuttings planted well above water table were struggling.
		Sedge plugs	Channel edge	40	30	75	30	75	These plugs were struggling due to absence of water in the channel. Ten plugs were destroyed during channel modifications 2011.
DAM 2	Channel	Sedge plugs	Channel edge	20	ND[1]	ND[1]	20	100	All sedges were growing vigorously and spreading.

Vegetation Cover

Overall, vegetation cover (vertical projection of exposed leaf area onto the ground) was visually estimated for each site (Roberts-Pichette and Gillespie 1999). The 60% vegetation cover at the Steeves Lake Shoreline Trench was the highest (Table 2). Steeves Lake In-fill Area with 36% cover was the only site that had a large area under water (15%) where a wetland has developed in the Steeves Lake Shoreline. Water is collecting in the wetland area because the finished construction topography is lower than specified in the design. Thirty-five native plant species have germinated from windborne seed, increasing biodiversity. Vegetation cover in the Truck Lake Channel and Dam 2 ranged from 10% to 35%. The relatively low success is probably due to generally drier conditions and soils being more compacted relative to the 'rough and loose' sites.

Table 2. Percent vegetation cover was calculated by visual estimate (FRC 2012).

Site	Zone	Percent Cover			Observations
		Vegetation	Bare Ground	Water	
STEEVES LAKE SHORELINE	In-fill Area, Wetland & Rough and Loose Trial	36	49	15	Vegetation was growing well, there were some bare patches where equipment had torn up the site in 2011. Many species growing from windborne seeds.
	Trench	60	40	0	Many small seedlings germinated from windborne seed.
TRUCK LAKE CHANNEL	Shoreline	10	90	0	Very sparse vegetation cover, many seedlings, 9 species germinated from windborne seed, soil was dry with mud cracks
	Outflow	20	80	0	Many small seedlings germinated from windborne seed. This site was disturbed in 2011 The wetland was dry with mud cracks forming.
	Channel	35	65	0	Channel was dry, soil was damp under under rip rap, sedges were struggling. There was more moisture in 2012 than in 2011
DAM 2	Channel, terrace, upland bench	20	80	0	Plants growing on the terrace were smaller, less robust than those growing along the bank of the channel. Terrace dry rocky surface

DISCUSSION

Willow, alder, and sedges established within the revegetated areas and will likely continue to grow. The revegetation efforts at Steeves Lake Shoreline showed the best results. In particular, the areas that were prepared with peat and had the 'rough and loose' method applied demonstrated the highest initial success rates for plant survival and coverage, as well as increased biodiversity. The soil bioengineering techniques involving willow cutting treatments showed the most success when implemented correctly and in areas with adequate moisture and failed when techniques were used incorrectly and planting was attempted too late in the year.

Engineering design challenges, unexpected conditions, and implementation practices at Steeves Lake Shoreline and Truck Lake Channel affected the success of the revegetation effort in these areas. The unexpected ponding of water on the Steeves Lake Shoreline cap caused an extensive die-back of the pre-existing vegetation, but encouraged more wetland vegetation to establish than originally anticipated. At the Truck Lake Channel, the compacted soils, inadequate soil depth, poor nutrient conditions and the unexpected absence of water in the channel will continue to hinder the recovery of vegetation. The poor survival rate of the willow cuttings planted along the Truck Lake shoreline resulted when the revegetation plan was not followed and late planting (November 2010) with inadequate planting depths was attempted. Earlier consideration of revegetation options for this project may have permitted more suitable environments for plant germination and growth to be incorporated into the engineering design, in areas such as the Truck Lake Channel, and necessary adjustments made in the revegetation sites to enhance natural recovery.

The remote location of the Colomac Mine site and post-remediation condition presented logistical challenges for monitoring the revegetation effort. The cost for site visits was high and access to Dam 2 was limited because of decommissioned roads requiring all-terrain vehicles to conduct some monitoring activities. This resulted in less time available for monitoring than initially planned.

CONCLUSIONS AND RECOMMENDATIONS

The experience at Colomac demonstrated the ability to incorporate revegetation and natural recovery into remediation plans. The success of implementing these treatments requires: 1) revegetation objectives for the site to be identified; 2) consideration of revegetation early in project planning; 3) recognition of the importance of expertise and people trained in revegetation; 4) need for flexibility and adaptive management in implementing the revegetation plan; and 5) recognition of the sustainability of the revegetation effort. There needs to be a willingness to adapt standard remediation contract specifications to move away from traditional focus on seeding to include alternative approaches that use soil bioengineering techniques and natural process to assist in revegetation of the site.

Preparing a site using the ‘rough and loose’ technique facilitated successful revegetation. It was easy to incorporate during the remediation, as equipment and people were available on the site, and it helped natural revegetation occur. The identification and use of materials readily available, including salvaged peat and locally collected pioneer plants proved to be an efficient and cost effective method for revegetation of large areas at this remote site.

Also, to measure success and build on revegetation and restoration experience in the north, appropriate monitoring strategies need to be implemented. Monitoring of revegetation area recovery should be incorporated into both short and long-term monitoring programs. Short-term monitoring will assess performance in the treatment areas and identify any immediate issues that may require maintenance or corrective action. Long-term monitoring will evaluate sustainability of the restoration/revegetation focusing on overall plant health and establishment, as well as the change in species composition and structure over time. The bioengineering techniques implemented at Colomac provided a successful, cost effective, and local approach to revegetation in a northern environment.

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RESTORATION PLANNING AND APPLICATION OF ECOLOGICAL SUCCESSION PRINCIPLES: UNITED KENO HILL MINE CASE STUDY

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ABSTRACT

Restoration of post mining disturbed sites within the boreal sub-alpine ecological communities of the Yukon has varied levels of success (Sheroan et al. 2010; Stewart and Siciliano 2013). Employing ecological succession principles to further the science of restoration of northern ecosystems on post-mining sites will contribute to the restoration body of knowledge. It will allow us to gain a better understanding of how to restore northern boreal forest sites where climate and poor soil development pose unique challenges for restoration (Clark and Hutchinson 2005). The re-establishment of ecosystems to conditions that once supported community and traditional land uses are another challenge in this region (ERDC 2012). Understanding how ecosystems respond to restoration designs will inform regulators and practitioners and therefore contribute to establishment of best practices policies.

The goal of this paper is to demonstrate the use of ecological succession principles to increase restoration success and ecosystem functional sustainability in the north, coupled with applying the planning tool "SMART" (Doran 1981). To demonstrate the application of ecological succession principles, this paper will use the former United Keno Hill Mine (UKHM) site in the Yukon as an example. The goal for the former UKHM site is to establish communities containing pioneer species that, over time, will be self-propagating on boreal low to subalpine bio-climate units (EBA 2003). In this paper we will apply the "SMART" planning tool to describe how to define goals and how to successfully establish appropriate objectives and targets to support this goal.

Key Words: Boreal, Community, Monitoring, Phytotoxic, Sub-Alpine, Restoration Ecology, Mine Tailings, Waste Rock, Pioneer Species, Biological Soil Crust, SMART Planning Tool.

INTRODUCTION

The successful long-term restoration of disturbed sites in northern regions is a complex and difficult task (Clark and Hutchinson 2005). Post-mining landscapes are generally void of several or all of functioning ecosystem components such as soil flora and fauna (Cooke and Johnson 2002). Remnant soils are compacted and or phytotoxic, lacking a seed bank which inhibits or prevents the recovery of once thriving ecosystems (Polster 2009). Often these undertakings require numerous interventions such as slope engineering, soil toxicity treatments and other treatments over an extended timeframe. In addition to physical and geochemical challenges, establishing a self-perpetuating vegetative cover in northern climates is challenging. Poor soil nutrients, arid environments, cold temperatures and a general lack of information on successful restoration processes present challenges. The use of native vegetation that has adapted to these harsh physical, chemical and climatic conditions is therefore a practical option (Clark and Hutchinson 2004). The focus of the terrestrial restoration project at the former United Keno Hill (UKHM) property is to produce clear and meaningful goals, objectives and targets to reach a sustainable, functioning, successional ecological community on sites with varying degrees of disturbance. The “SMART” planning tool (Doran 1981) is a useful guide in the development of attainable and management project goals in concert with application of ecological succession principles as detailed below.

HISTORY AND PROJECT DESCRIPTION

The historic Keno Silver Mining District, located in central Yukon, produced over 214 million ounces of silver from 1914 to 1989 (Alexco 2013). Mining methods used to extract resources included open pits and underground workings. Ownership of the former UKHM property defaulted to the Canadian Government in 2004. Alexco Resource Group became the preferred purchaser of the assets in 2005 and entered into a unique cost sharing partnership with Canada to continue the care and maintenance of the property under the project specific company Elsa Reclamation and Development Company (ERDC). Continuing investigations to inform and support the closure selection process are in the final phase and selection of preferred closure options by the closure team is expected to take place early 2014.

The former UKHM property has an area roughly 250 km^2 out of which approximately 110 ha will require some degree of ecosystem restoration including the 90 ha valley tailings facility. Mining disturbances include three tailings deposits, 68 waste rock dumps, 19 open pits, multiple shafts, buildings and infrastructure. Waste rock dumps associated with adits contain some low grade ore and will require soil covers whereas waste rock dumps associated with pits are comprised mainly of country rock and are not considered phytotoxic (Alexco 2013). The UKHM terrestrial program focuses on creating suitable conditions for the establishment and perpetuation of early seral native plant species found in adjacent reference communities, including disturbed areas where soil and ecological memory has been removed, and on sites that have been ameliorated by the installation of soil covers (Alexco 2013).

The closure plan goal for the UKHM property is to establish a northern boreal sub-alpine community containing pioneer species that will be self-propagating. To reach this goal we have proposed the application of ecological succession principles. Primary succession will be facilitated by assessing, planting and monitoring native pioneer species onsite including the use, where appropriate, of biological soil crusts.

NATURAL SUCCESSION

Natural succession, a process of ecological change in which a series of natural communities are established and then replaced over time, from shade intolerant species eventually supplanted by shade tolerant species (Kimmins 1997; Tansley 1920), can be used as an indicator for appropriate species selection and design in restoration. The theory of natural succession asserts plants have an optimal range for growth and development (Polster 1991) and that over time, increases in biomass, primary production, respiration, and nutrient retention lead to diversity and changes in species composition and ultimately, increased structural complexity, or climax succession. Mechanisms that drive ecological succession include facilitation, tolerance, and inhibition and can be generalized into two stages.

1) Primary succession is the community formation process that begins on substrates that had never before supported any vegetation (Mueller-Dombois and Ellenberg 1974). Depending on the conditions, pioneer species can take many years to establish and therefore, selection of appropriate plants is critical for long term success of any reclamation program (Polster 2009). Generally, pioneer species trend from bryophytes, lichens and biological soil crusts to graminoids, herbs, dwarf shrubs, large shrubs, sapling trees to late seral tree species. Several pioneer plants fix nitrogen and create conditions and space for successional advancement. Decomposition and nutrient cycling enables soils to support the more complex plants that are eventually replaced by later seral coniferous tree species.

2) Secondary Succession originates only from a partial disturbance of an ecosystem (Mueller-Dombois and Ellenberg 1974). The general trends during secondary succession are similar to primary succession. Graminoids or forbs, such as fireweed (*Epilobium angustifolium*) may dominate the herb layer. Annual species, over time are replaced by perennial species, and then, depending on the ecological zone, by shrubs, an early seral forest, followed by mature boreal forest consisting of late seral coniferous tree species.

Understanding the process of ecological change is essential in planning for successful restoration as it mimics models from nature. The underlying idea in application of successional theories in restoration is to let nature do the work. Walker and del Moral (2003) suggest primary succession is integral to restoration planning as it allows for direct observation over time, employs comparative studies to later seral stages, enables linkages to long term processes, and provides tools for restoring anthropogenic and naturally disturbed ecosystems. Furthermore, ecological restoration is a manipulation of successional processes to meet realistic targets in restoring damaged landscapes (Walker et al. 2007).

Prior to initiating a restoration program, the following three necessary steps should be considered:

1) Determine the ecological zone, moisture regime and nutrient regime;
2) Confirm the site successional stage. If the site is in a state of primary succession the use of pioneer species appropriate for the region including nitrogen fixers is recommended. Adjacent or nearby permanent reference sites located on a similar slope and aspect will provide historical system information and species autecology (Walker et al. 2007) ensuring selected species are suitable for the site conditions. Sites that are advanced to the stage of secondary succession will benefit from using opportunistic species appropriate for the area. Reference sites should ideally contain the species of the desired ecological community; and
3) Finally, determine if there are any limiting factors that would prevent the establishment of vegetation (soil contaminants, compaction, erosion processes, biotic pressure, social components, or any other abiotic/biotic factors).

Following completion of the above three steps, goals, objectives, targets (and performance indicators) can be developed. These are discussed in detail below.

"SMART" PLANNING TOOL: APPLIED TO RESTORATION

The "SMART" tool has been adapted from a simple planning process and is related here in terms of Restoration Program development (as adapted by the University of Victoria course Restoration Ecology ASP503).

Table 1: "SMART" Terms Defined

Letter	Minor Term	Minor Terms	Planning Term
S	Specific	Significant, Stretching, Simple	Objective
M	Measureable	Meaningful, Motivational, Manageable	Target
A	Appropriate	Appropriate, Achievable, Agreed, Assignable, Actionable, Ambitious, Aligned, Aspirational	Objective
R	Realistic	Realistic, Resourced, Resonant	Objective/Target
T	Time-Bound	Time-oriented, Time framed, Timed, Time-based, Timeboxed, Timely, Time-Specific, Timetabled, Time limited, Trackable, Tangible	Target
E	Evaluate	Ethical, Excitable, Enjoyable, Engaging Ecological	-
R	Re-evaluate	Rewarded, Reassess, Revisit, Recordable, Rewarding, Reaching	-

The planning terms outlined as part of the "SMART" tool are Goals, Objectives and Targets. Goals are long-term, broad statement about what a program/project hopes to achieve. Objectives are the short-term, concrete, stepping stones towards achieving a goal. Objectives should be Specific, Appropriate, and Realistic; the S, A, and R in "SMART". Targets are Measurable, Realistic and can be achieved within a specified Time frame; the M, R, and T in "SMART". The term performance indicator will also be used, which is a measurable unit to help define the target. These terms and the "SMART" planning tool will be used to develop the restoration plan for the former UKHM site.

DEVELOPMENT OF A RESTORATION PLAN: UKHM CASE STUDY

The former UKHM site is located in the North Yukon Plateau Ecoregion. The landscape was formed by past glacial activity and soil is limited. Primary tree species along the mid and lower slopes are white spruce (*Picea glauca*), trembling aspen (*Populus tremuloides*), Alaskan birch (*Betula neoalaskana*), and the occasional balsam poplar (*Populus balsamifera*). The lowlands are vegetated by a matrix of scrub birch (*Betula glandulosa*), willow (*Salix sp*.) and Ericaceous shrubs with sparse to open black spruce cover. White spruce in a matrix of dwarf willow, birch, Ericaceous shrubs, and, occasionally, lodgepole pine forms extensive open forests, particularly in the northwestern portion of the ecoregion. Black spruce, scrub willow, birch, and mosses are found on poorly drained sites. Alpine fir and lodgepole pine occur in higher subalpine sections, whereas alpine vegetation consists of mountain avens, dwarf willow, birch, ericaceous shrubs, graminoid species and mosses (Smith et al. 2004).

Several areas requiring reclamation are devoid of topsoil and remain in a state of primary succession; however, limited ingress of pioneer species is evident in some locations and an inventory of species as candidates is presented in Table 2.

Table 2: Pioneering and early seral species identified at former UKHM property

Botanical Name	Common Name	Growth Form	Habitat	Soil Moisture	Soil Characteristics
Agrostis scabra	Ticklegrass	Graminoid	Low to mod elevations. Dry to wet disturbed areas, dry rocky slopes.	Sub-xeric to sub-hygric	Tolerant of acidic soils, drought and low nutrients and permafrost.
Calamagrostis canadensis	Bluejoint	Graminoid	Widespread in boreal to subarctic. Riparian, cool, moist forest communities.	Sub-mesic to sub-hygric	Moist to wet sites; tolerant of acidic and saline soils.
Carex aqualtilis	Water sedge	Herb	Wet sites.	Mesic to sub-hygric	Moist to wet.
Epilobium angustifolium	Fireweed	Herb	Low to subalpine elevations. Mesic open forests, burns.	Mesic	
Equisetum arvense	Common horsetail	Herb	Riparian, lake edges, marshes, fens, bogs, low to mid elevations.	Mesic/Sub-hydric	Moist to wet.
Lupinus arcticus	Arctic lupine	Herb	Range from lowland riverbanks to alpine, tundra.	Mesic	Tolerant of low nutrients, permafrost, nitrogen fixing.
Oxytropis campestris	Field Locoweed	Herb	Gravel bars, rocky outcrops, roadsides, dry and open woodland into alpine tundra.	Sub-mesic to xeric	Nitrogen fixer.
Alnus crispa	Alder	Shrub	Widespread.	Very Xeric/Mesic /Subhydric	Moist, gravelly to rocky, generally acidic pH 5.0 to 6.5. Nitrogen fixer.
Betula glandulosa	Scrub Birch	Shrub	Sub alpine, found in many *P. mariana* and white spruce *P. glauca* communities.		Moist, sandy, gravelly loam to organic soils. Tolerant of salinity and pH 3.1 to 6.5.
Dryas spp.	Avens	Shrub		Mesic	Moist, river bars, lowland to alpine. Nitrogen fixer.
Empetrum nigrum	Crowberry	Shrub	Subarctic, tundra, heathlands, swamps and bogs from sea level to alpine.	Very Xeric/Mesic /Subhydric	Acidic, moist, sandy to rocky soils, glacial till. Soil pH ranges from 2.5 to 7.7.
Salix alaxensis	Felt-leaf willow	Shrub	Widespread through riparian, boreal, tundra, subalpine to alpine.	Mesic	Moist silty or mineral soil. Wet meadows and thickets.
Salix glauca	Grey-leaved willow	Shrub	Swamps, fens, bogs, streambanks, dry to wet open forest, to alpine.	Sub-Mesic-sub-hygric	Gravely soils to bogs and fens.
Shepherdia canadensis	Soapberry, Buffaloberry	Shrub	Co-dominates numerous seral willow and mixed-shrub.	Very Xeric/Mesic /Hydric	Dry, calcareous. Nitrogen fixer.
Populus balsamifera	Balsam poplar	Tree	Widespread, prefers riparian areas of boreal forests.	Very Xeric/Mesic /Subhydric	Prefers moist areas, but will grow on dry sites.
Populus tremuloides	Trembling aspen	Tree	Widespread.	Very Xeric/Mesic /Subhydric	Variable, but prefers well drained loamy soil with high OM content.

To create conditions amenable to seed germination and plant growth, the disturbances or filters listed in Table 3 below must be ameliorated to improve the physical and chemical nature of the sites and prevent a state of arrested succession (Cooke and Johnson 2002).

Table 3. Limiting factors (disturbance elements) of the former UKHM Site

Disturbance Elements	Condition	Possible Solutions
Phytotoxic soil	Elevated level of zinc, cadmium etc. inhibiting plant growth	Soil covers; application of lime
Soil nutrients	Mineral soil; some biological soil crust	Seed, plugs of early seral pioneering plants including nitrogen fixers; addition of biochar/ soil amendments where warranted
Aeration	Compacted sites, smooth microtopography	Scarify where possible to create rough and loose conditions
Coarse substrate	Limited fines required for moisture retention and plant establishment	Identify where fines are located; addition of fines in some areas and pocket plant to create environmental resource patches
Moisture	Xeric, sub xeric to mesic sites	Species selection – outcome from ecosystem mapping program; soil amendments where possible

Establishment of goals are meant to ensure the long-term success of the terrestrial restoration program by developing a clear executional plan. The goal of the UKHM project, under agreement with Canada is to develop and implement an Existing State of Mine (ESM) Reclamation Plan; the goal of the terrestrial soil and vegetation program is to "establish a northern boreal sub-alpine community containing pioneering species that, over time, will be self-propagating". As stated previously, objectives are short-term concrete steps toward achieving a goal. There may be several objectives that support a single goal but should be clear and unambiguous. Using "SMART" terms, the following questions and steps are considered when formulating objectives and targets:

Specific Objectives

What is expected? It is expected that over time, vegetation will successfully establish on a variety of site specific locations. Tailings and waste rock piles containing elevated levels of zinc and other phytotoxic elements that are not able to support a thriving plant community will be covered and vegetated. In locations where amelioration or cover of substrates is not required, establishment of pioneering plant and biological soil crusts to attain an early seral plant community will undergo separate treatment in concert with closure team members. It is expected these sites may require scarifying to create 'rough and loose' soil conditions amenable to seed germination and growth (Polster 2011).

Why is it important? The need to plan and understand successional processes and implement this knowledge in northern restoration projects cannot be overemphasized. EDI Environmental Dynamics Inc. (2009) conducted a review of several revegetation projects and techniques in the Yukon and recommended additional research and monitoring of species other than grass, and root cutting techniques. They also suggested research and trial work on developing reliable and cost effective methods in the production of cuttings, transplants and stakes for reclamation projects in the north.

Who is involved? Where is it going to happen? Which attributes are important? The long term expectation is to continue to create capacity within the citizenship of the Nacho Nyack Dun (NND) whose traditional territory encompasses the Keno Hill Silver District and within the communities of Mayo and Keno City. The development of the ESM Plan and the completion of the regulatory and permitting process is scheduled to occur over the next 5 years with implementation of the final reclamation plan to begin in 2017. NND students and the community have been involved in several aspects of the terrestrial revegetation program including field work, ecosystem mapping and more recently, seed collection.

Appropriate Objectives

Existing infrastructure owned by local residents will be utilized to support a seed increaser program and native plant propagation trials. Enthusiasm about learning and contributing to the overall objective of restoring the sites is evident in the community and preliminary planning is underway to develop experiential and practical programming opportunities.

Realistic Objectives

Using pioneer plants for planning and selection is tailored to represent and support onsite conditions such as metal tolerance, drought resistant, nutrient poor, condition. Objectives such as using native species to attain the overall goal will set the project on the right trajectory. Realistic and successful restoration objectives are those include knowledge from the structural and functional characteristics of the natural ecosystem and from which measurable targets can be established (Cooke and Johnson 2002).

Targets are measureable, realistic and can be achieved within a specified time frame. Ruiz-Jaen and Aide (2005) recommend at least two variables within each of the three ecosystem attributes and two reference sites at a minimum are required to establish targets. The three ecosystem attributes that are measurable include diversity, vegetation structure and ecological processes. Establishing targets with these attributes will provide critical information about ecosystem resilience, nutrient cycling and succession (Ruiz-Jaen and Aide 2005).

Measureable

This target usually answers questions such as how much, how many, how to know when it has been accomplished and what are the units of measurement (Gonzales 2013). Defining measurable targets is critical for performance monitoring and implementation of adaptive management measures should monitoring results indicate a need to do so. Proposed measurable targets for the former UKHM property may include the evaluation of restoration success by comparing the trajectory of recovery of established variables through time with reference sites (Ruiz-Jaen and Aide 2005); and to establish a native pioneering community by Year 5, with a proposed target density that will be obtained from existing literature and adjacent recovering plant communities. (Cargill and Chaplin 1987; Government of Saskatchewan 2008; Greene et al. 1999; MWLAP 2002; Walker et al. 1986; Wilson et al. 1996).

Realistic

Similar to realistic objectives, targets must also be realistic. Assessing those targets can be done during the evaluation phase (below) which may trigger an adaptive management response depending on the outcome.

Time Bound

This target will usually answer questions such as when, what can be done 6 months from now, what can be done 6 weeks from now, what can be done today? (Gonzales 2013). The temporal scope of implementation and monitoring of the closure program at the former UKHM site is at minimum two decades. The time-bound targets of the ESM reclamation plan will be reflected in the terrestrial program.

Evaluate

Evaluation is a tool to measure the success of the project, in addition to providing a way to communicate and report the efficacy of the project (Gonzales 2013). Evaluation of established ecosystem attributes such as diversity, vegetation structure and ecological processes presented above will provide key information about the restorative progress of the site to the closure team and stakeholders.

Re-evaluate

This final term emphases the cumulative opportunity to learn, improve and develop when using "SMART" project management, particularly for long-term and related projects. An adaptive management cycle to advance knowledge will always incorporate re-evaluation (Gonzales 2013). An adaptive management plan is a tool that will inform the closure team what can and cannot be accomplished and must therefore have the flexibility to address results by adjusting goals or objectives if necessary (SER 2005).

CLOSURE AND SUMMARY

Through the use of ecological succession as a model, letting nature be our guide, and careful, thoughtful consideration while establishing goals, objectives and targets we can design and implement successful ecologically based restoration projects. The ecology of each restoration site is complex and dynamic; efforts will be made to guide restoration sites to an acceptable functioning state that will establish ecosystems that will provide continuing services over long periods of time and into the uncertain future of climate change. Additional attributes of success will be developed as the need arises during the planning and implementation phases of the project. Monitoring and related adaptive management are essential for continued learning. The "SMART" planning tool is recommended for development of adaptive management strategies, goals, objectives and targets within the monitoring program.

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PASSIVE TREATMENT OF MINE DRAINAGE WATERS: THE USE OF BIOCHARS AND WOOD PRODUCTS TO ENHANCE METAL REMOVAL EFFICIENCY

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ABSTRACT

Passive biological treatments have been proposed as a possible efficient and cost effective treatment method for metal bearing water discharged from mine sites after closure. Several biofilters are under study in Yukon and have produced variable, but promising results up to now. However, concerns are typically expressed around biological treatments and their suitability in northern, colder climates. Biofilters allow for metal removal using a variety of chemical, physical and biological mechanisms. If biological processes are affected by a cold climate to some extent, chemical processes are typically not affected by the temperature the same way and can be reliable in cold waters. This study focused on metal sorption and metal removal by chemical mechanisms and assessed the sorption capacity of biochar and wood products which could be later introduced in bioreactors to help with metal removal from mine-impacted cold waters.

Biochars allowed for more than 90% removal of Cd, Cu and Zn from a metal-bearing effluent along with 35 to 69% removal of arsenic. Wood products displayed good removal capacity as well, in the range of 51 to 94% for Cd, Cu and Zn. However, arsenic and selenium removal by wood products was limited; Se also showed minimal sorption on biochars and was in one case released during sorption testing. Metal leaching from the materials was observed to some extent, including Cu and Zn from poplar and spruce products. Amongst spruce products, the chips from the trunk proved to be slightly more efficient than the needles. Overall, biochars and wood products showed potential for use in water treatment for metal sequestration in combination with other mechanisms such as sulfide precipitation in sulfate-reducing bioreactors. Such materials could be collected or produced on remote mine sites and could help with mine remediation.

Key Words: Water Treatment, Metal Removal, Metal Sorption, Adsorption, Bioreactors, Bioremediation.

INTRODUCTION

Water management in the mining industry has become a priority focus in our world, which has been increasingly concerned with sustainable development. To lower their footprint on the environment, mines are constantly working toward limitation of contaminant discharge to the environment. Hard rock mines have to closely control the concentrations of metals according to federal and provincial/territorial regulations, which leads to the treatment of mine-affected waters, including run-off waters, drainage from tailings or waste rocks pads, process waters, etc. Water treatment is required during operation of the mines as well as after closure for the long term. Current water treatment technology development is focused on long-term passive treatments that require low operation and maintenance. Various passive treatments are available, including chemical adsorption and bioremediation (Johnson and Hallberg 2005). Along with metals, mine waters commonly contain high sulfate content that results from the breakdown of sulfide minerals (Akcil and Koldas 2006; Kalin et al. 2006). There is an increased interest in the use of Sulfate-Reducing Bacteria (SRB) to help with metal removal from mine waters (Dar et al. 2007; Genty 2011; Jong and Parry 2003; McCauley et al. 2009; Neculita et al. 2010; USEPA 2002). Under anaerobic conditions, SRB reduce sulfate (SO_4^{2-}) into sulfide (S^{2-}) using electrons from organic matter. Sulfides, in turn, precipitate metals. The solubility of metal sulfides being generally very limited, it consequently lowers the concentration of metals in the effluent and provides a stable metal precipitate (Blais et al. 2008). To accomplish sulfide production, SRB catalyze the oxidation of organic carbon from the surrounding organic matter (eq. 1), where CH_2O represents organic carbon (Waybrant et al. 1998):

$$SO_4^{2-} + 2CH_2O \rightarrow H_2S + 2HCO_3^{-} \text{ (eq. 1)}$$

Anaerobic bioreactors are being studied in the mining industry for effluent treatment. In these systems, the effluent passes through a biofilter (e.g., in trenches) filled with permeable solid support (gravel, sand) and substrate (organic matter). The substrate used to support SRB growth can be variable. Neculita and Zagury (2008) showed that cellulosic materials like wood waste have a beneficial effect on SRB treatment efficiency. Wood products (leaf mulch, wood chips, sawdust, wood compost, peat moss) mixed with sewage sludge or manure can be an excellent substrate for SRB and have been shown to achieve reduction of metals concentration in mine effluent (Neculita et al. 2010; Waybrant et al. 1998). Besides efficiency in the short-term, the ideal mixture of substrate should also last in the long term, i.e, not be too biodegrade and deplete before the end of the life time of the bioreactor, or to extend the useful life of the bioreactor before it would require replacement or refreshment of the media. Drury (2006) used a mathematical model and showed that organic matter with an older apparent age, less biodegradable, can sustain bioreactor efficiency for longer duration. Additionally, the residual organic products may improve the stability of the metal sulfide precipitate once the treatment system is closed.

In northern climates, when the temperature is low, concerns have been expressed about the efficiency of SRB to sustain a sufficient level of biological activity during winter time to maintain treatment efficiency (Nordin 2010). The objective of this study is to assess if a range of substrates can also help with metal removal using chemical mechanisms, which are generally not as temperature-dependent as biological mechanisms. Besides providing feed to SRB, solid substrates can also act as a metal adsorbent. Cellulosic materials like sawdust and wood chips are known for their metal adsorption capacity (Argun et al. 2008; Keng et al. 2013; O'Connell et al. 2008) due to reactive groups within the substrate. Wood is an abundant resource in remote mine sites in northern Canada and other northern climates, and wood chips could easily be included in bioreactors to help metal removal by providing biodegradable organic matter and metals site adsorption. This study looked at the adsorption capacity of Spruce (trunk and needles) and Poplar (trunk) chips, as both species are very common in Yukon mine sites. In addition, biochars made from wood products were also studied. Biochar is defined as a carbon-rich material produced by thermal decomposition of organic material under limited supply of oxygen at relatively low temperature (<700°C) (Lehmann and Joseph 2009). On-going projects look at the construction of mobile pyrolysis ovens (personal communications with K. Stewart, Yukon Research Centre, M. Garcia-Perez, University of

Washington). Hence biochars could be produced in remote locations, providing that the mine site has access roads. Due to the thermal decomposition, the remaining biochar is recalcitrant and is likely to persist in bioreactors on the long-term. Biochars are also capable of adsorbing metals on their surfaces and several biochars proved to have good potential for metal removal from effluent, although metal adsorption capacities can be very variable. Metal sorption by biochars depends largely on biochar characteristics, including feedstock, pyrolysis temperature, oxygen content, etc. Table 1 presents the adsorption capacity measured by various authors.

Table 4. Review of adsorption capacity of biochars from literature

Metal	**Adsorption capacity (mg/g)**	**Adsorption pH**	**Biochar feedstock**	**References**
Cd	1.5	5	Alamo switch grass	Regmi et al. 2012
Cu	4	5		
Cd	16.6	6	Pig manure	Kolodynska et al. 2012
Cu	6.3	5		
Pb	19.8	6		
Zn	4.2	5		
Pb	4.1	5	pinewood residues	Liu and Zhang 2009
Pb	2.4	5	rice husk residues	
Cr (VI)	3.0	2	Oak wood	Mohan et al. 2011
Cr (VI)	4.6	2	Oak bark	
Cu	0.04	5	peanut straw	Tong et al. 2011
Cu	0.09	5	canola straw	
Cu	12.5	5	corn straw	Chen et al. 2011
Zn	11	5	corn straw	
Cu	6.8	5	Hardwood	
Zn	4.5	5	Hardwood	
Cu	48.5	6	Salt-marsh plant	Li et al. 2013

Three different biochars, poplar wood chips, spruce wood chips and spruce branch mulch were studied as metal adsorbents in this study.

MATERIALS AND METHODS

Adsorbents Sampling and Preparation

Poplar and spruce trees were cut down in the Whitehorse region, Yukon Territory, Canada. Branches were removed from the tree before the trunks were were ground into chips using a log chipper (Bandit M65 XP, USA). Spruce branches were ground separately into mulch using the same equipment. Wood chips and mulch were used fresh, with less than a week of drying.

Three biochars were collected from different manufacturers. Biochar made from mixed spruce, pine, and fir was produced by Diacarbon Energy Inc. (Burnaby, BC, Canada) and named “BCD”. Biochar made of spruce, pine, fir, willow and poplar was produced by Zakus Farms ("BCZ"). The biochar collected from Titan (Saskatoon, SK, Canada) was made from Willow and fish bone meal ("BCT").

Adsorbents pH Measurements

Suspensions were made using 1:10 (w/w) ratio of biochar or wood with DI water. The pH of the suspension was measured at t=0 ($pH_{t=0}$) and after a week ($pH_{1\ week}$) at room temperature using a pH meter (Oakton pH5+, Vernon Hills, IL, USA) equipped with Ag/AgCl combination reference electrodes. pH calibration was done using certified pH 4, pH 7 and pH 10 standards (Fisher, catalogue number SB101-500, SB107-500 and B115-500).

Batch Adsorption Studies

Synthetic drainage effluent was produced using sulfate metal salts (As_2O_5, $CdSO_4•8/3H_2O$, $CuSO_4•5H_2O$, $FeSO_4•7H_2O$, SeO_2, $ZnSO_4•7H_2O$ and $NaSO_4•10H_2O$, all ACS reagents) dissolved in DI water at pH 6. Then 2, 4, 6, 8 and 10 g of adsorbent materials were mixed with 200 ml of synthetic drainage effluent in a 500ml baffled Erlen Meyer and shaken for 24 hours at room temperature to allow for metal sorption equilibrium. To assess metal leaching from the materials, 10g of adsorbent was mixed with DI in the same conditions. Supernatant was then filtered through 0.45μm porosity glass fiber filters (Cole Parmer, catalogue number RW03-04700) and stored for further analysis.

Analytical Techniques

Effluent pH was measured (Oakton pH5+, Vernon Hills, IL, USA with Ag/AgCl combination reference electrodes). Total Solids contents were measured according to APHA method 2540B. Biochar and wood products were partially digested using ACS grade nitric acid and hydrogen peroxide (method USEPA 3050b) to allow for determination of metal contents. Metal concentrations were measured using Inductively Coupled Plasma-Optical Emission Spectroscopy (ICP-OES), Vista-AX CCO, by Varian (Palo Alto, CA, USA). Quality controls were performed with certified multiple element standards from SCP Science (Lasalle, QC, Canada) to ensure conformity of the measurement apparatus. Limit of Quantification (LQ) was calculated as 10 times the standard deviation measured obtained after measurement of 10 blanks.

RESULTS AND DISCUSSION

Material Characteristics

The material studied includes three biochars (BCD, BCZ and BCT) and three wood products (poplar chips, spruce chips and spruce needles). Table 2 presents the results of characterization. As expected, biochars displayed alkaline pH, initially between 9 and 10 and reducing down to close to 8.5 after a week. On the other hand, the wood products were acidic, with the poplar chips producing the most acidic conditions after a week, at pH 3.84. Arsenic and selenium contents in all materials were low, under the quantification limits of 0.9 mg/kg for As and 2.1 mg/kg for Se, except the biochar made of willow and bone meal (BCT) at 6.3 ± 0.7 mg/L Se. Overall, the BCT material, made of willow and bone meal, displayed higher concentrations of metals, including significant amount of Cd, Zn, Fe and Na. In general, higher metal contents were measured in biochars than in wood products. Biochars were made by pyrolysis, which involved volume reduction and subsequent concentration of the metals in the residual product.

Table 5. Measured adsorbent characteristics (1:10 water suspension was used for pH measurements; HNO_3/H_2O_2 digestion was used for metal contents analysis)

Material	Total solids	$pH_{t=0}$	pH_{1week}	As	Cd	Cu	Fe	Na	Se	Zn
	%			mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg
BCT	81.0	9.84	8.27	<0.9	11.0	3.3	3755.8	15814.5	6.3	164.2
BCD	81.1	9.91	8.51	<0.9	0.8	26	1362.3	1281.0	<2.1	55.8
BCZ	96.5	9.23	8.47	<0.9	0.2	15.2	939.1	123.0	<2.1	60.0
Pop. chips	93.7	6.06	3.84	<0.9	0.2	30.8	32.6	53.3	<2.1	30.0
Sp. chips	93.6	5.94	5.5	<0.9	2.0	67.9	54.5	111.4	<2.1	27.0
Sp. needles	89.9	5.28	5.11	<0.9	2.4	22.9	110.1	124.8	<2.1	51.1

Metal Leaching

Wood products and biochars contained heavy metals to some extent. Mixing of the materials with DI water for 24 hours was completed to assess the potential for metal leaching. The results are presented in Figure 1 along with the concentration measured in the synthetic drainage water for comparison. Wood products leached out more metals than biochar products, even if metals contents were generally lower (Table 2). The metals contained in pyrolysis products may be tightly bound and less available for leaching. Poplar chips and spruce needle leached significant amount of copper and zinc in the first 24 hours of being submerged in water, with concentrations 0.13 and 0.12 mg Cu/L and 0.13 and 0.17 mg Zn/L respectively using 5% S/L ratio. No selenium leaching was observed (< LQ of 0.021 mg/L) and arsenic leaching was observed only for BCT (0.012 mg As/L, otherwise < LQ of 0.0093 mg/L). Hence, the use of natural materials such as wood, in bioreactors should be managed carefully, with special attention to metal leaching potential in the initial operation period.

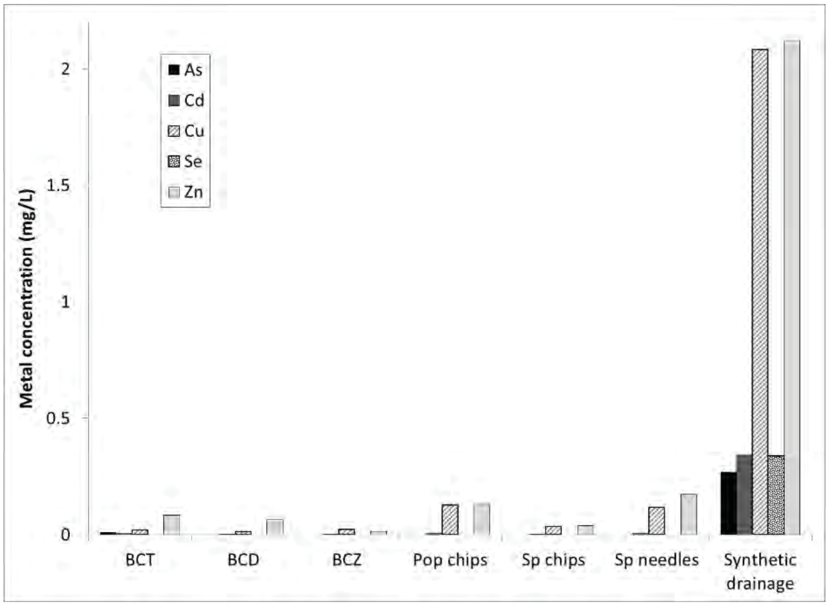

Figure 4. As, Cd, Cu, Se and Zn concentration observed after leaching from biochars BCT, BCD, BCZ, poplar chips, spruce chips and spruce needles and in the prepared synthetic drainage water (Solid to liquid ratio of 5% in DI water, 24 hours equilibrium, LQ are 0.009 mg/L for As)

<u>Metal Adsorption</u>

The primary objective of this study was to assess the sorption capacity of materials which can be collected or produced on-site in many northern and especially Yukon mines, to help with metal sequestration. Three biochars and three wood products were mixed with pH 6 synthetic drainage water containing metals commonly found in mine impacted waters, namely As, Cd, Cu, Se and Zn. Figure 2 presents the relative concentration (C/C_o) of the different metals remaining in the effluent after exposure to each of the adsorbents for 24 hours. Biochars results (BCT, BCD and BCZ) are displayed in Figure 2a, b and c. The profiles are very similar for Cd, Cu and Zn, with more than 90% removal on average using 1% to 5% S/L ratio. Arsenic removal was lower, with 54, 69 and 35% respectively using BCT, BCD and BCZ. The selenium profiles were somewhat surprising. The three biochars studied were not able to remove more than 30% of selenium and the Biochar BCZ actually released selenium to a significant extent. Although 10 g of BCZ leached less than 2 µg of Se in DI water, when mixed with synthetic drainage the same amount of biochar released more than 80 µg of Se. Selenium initially bound to the biochars may have been displaced through an exchange process during sorption of other metals with higher affinities like Fe, Na, Zn, Cu, Cd on the biochar. The mechanisms of selenium release should be further studied.

Divalent metal removal by wood products was reasonably effective. Although not as efficient as biochar, poplar, spruce chips and spruce needles were able to remove Cd up to 87, 94 and 84% respectively, Cu up to 76, 81 and 66% respectively and Zn up to 83, 88 and 51% respectively. The spruce chips obtained from trunks displayed higher ability than the needles for metal removal. This difference may be due to the different structure, porosity, and surface chemistry of the needles versus the trunk chips. Amongst the three wood products, Spruce chips gave the best results.

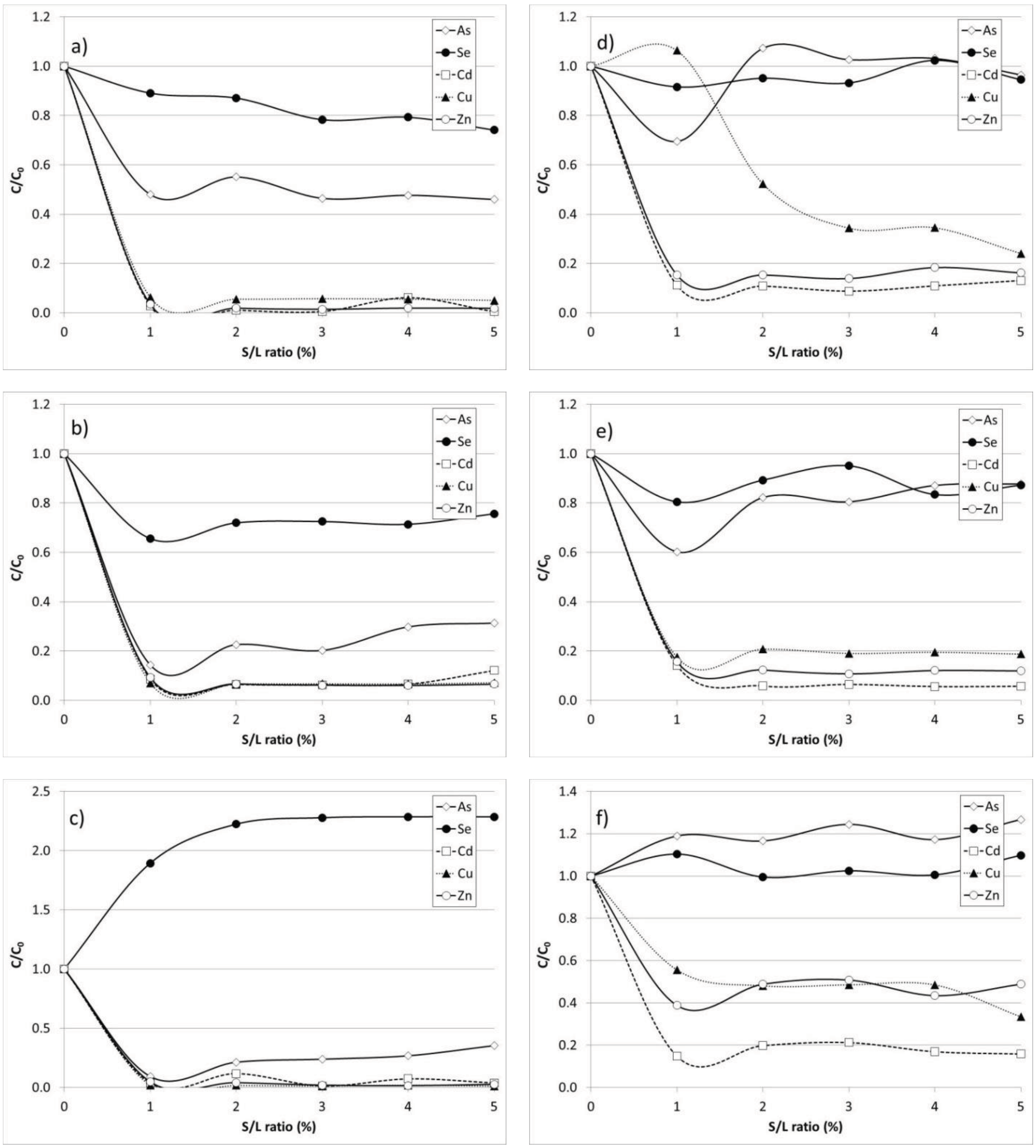

Figure 5. Relative concentrations (C/Co) of As, Se, Cd, Cu and Zn after 24 hours equilibrium with variable solid to liquid ratio using a) BCT, b) BCD, c) BCZ, d) poplar chips, e) spruce chips and f) spruce needles (initial concentrations of 0.27 mg As/L, 0.34 mg Cd/L, 2.08 mg Cu/L, 0.34 mg Se/L and 2.12 mg Zn/L).

CONCLUSION

Natural wood products and biochars were studied for metal removal from a synthetic drainage effluent. Biochars exhibited alkaline properties whereas wood products generated acidity however, both materials displayed good capacity for divalent metal removal like Cd, Cu and Zn although biochars were slightly more efficient than wood products. As and Se were less amenable to adsorptive removal than divalent metals but biochars helped to remove arsenic to some extent. Overall, this study gave evidence of the potential of wood products and biochars for water treatment and metal sequestration, providing that metal leaching from the material itself is controlled. Further study should investigate the effect of using such materials on sulfate-reducing bacteria growth and bioreactor efficiency, and these tests are planned using some of the same media as was tested in this study.

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TWIN SISTERS NATIVE PLANT NURSERY: INTEGRATING RESEARCH, TRAINING, AND OUTREACH FOR THE PROPAGATION OF NATIVE AND CULTURALLY SIGNIFICANT PLANT SPECIES IN NORTHEASTERN BRITISH COLUMBIA

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ABSTRACT

The newly incorporated Twin Sisters Native Plant Nursery, a joint venture of the Saulteau First Nations and West Moberly First Nations, was created to meet a growing demand for native plant species for use in reclamation in the Peace River region of British Columbia. Associated with this new native plant nursery is the development of a number of programs that support successful nursery operations as well as the use of native plants in reclamation work in the region. Collaborative research involving First Nations and industry partners aids in the identification of a wide range of culturally significant and ecologically appropriate native plant species for propagation. Training programs in native plant horticulture support the creation of a skilled and knowledgeable workforce. A proposed native plant reclamation guidebook will contribute to successful installation of native plant species at reclamation sites. In this paper we describe the creation of the Twin Sisters Native Plant Nursery and discuss how the associated research and training programs as well as the creation of reclamation guidelines support the overarching goal of the nursery to promote the use of native plant species in reclamation in Northeastern British Columbia.

Key Words: First Nations, Traditional Ecological Knowledge (TEK), Native Plant Propagation, Peace River Region, Reclamation.

INTRODUCTION

The Peace River region of Northeastern British Columbia has sustained a high degree of development from the mining and the oil and gas sectors over the last 30 years (Lee and Hanneman 2012). This development has brought economic benefits to the region but has also contributed to the disturbance of large tracts of land through the development of open pit coal mines, roads, cutlines and pipelines (Lee and Hanneman 2012; Nitschke 2008). Proposed developments in the region, including a number of new pipelines and mines, as well as the BC Hydro Site C Project, are currently in various stages of environmental assessment. Current reclamation practices have led to large fragmented areas being reclaimed using a range of agronomic plant species. In the short term, this type of reclamation is believed to set ecosystems on a different successional pathway than would naturally occur post-disturbance, making it unlikely for the natural establishment of a native plant-dominated community (Keefer 2010).

Both the Saulteau First Nations (SFN) and West Moberly First Nations (WMFN) have expressed concerns with current reclamation standards that create agronomic plant communities in locations formally host to culturally and ecologically productive landscapes. These First Nations have emphasized the need for the incorporation of native and culturally significant plants in reclamation activities in the region to better mitigate the impacts of industrial development[1]. At present there is an insufficient supply of genetically local native and culturally significant plants to meet industry demand for reclamation in the Peace River region and elsewhere in British Columbia. It is believed that an increased supply of native species appropriate for use in various environmental conditions that exist at reclamation sites will enhance the broad scale adoption of these plant species in reclamation practice and standards.

To address shortages in native and culturally significant plants that are appropriate for use in reclamation, a collaborative partnership – including Walter Energy Ltd., which operates three surface metallurgical coal mines in Northeastern British Columbia; Keefer Ecological Services (KES), a British Columbia-based consulting firm with expertise in reclamation and ethnobotany; and Saulteau and West Moberly First Nations – was formed to develop a native plant nursery in the Peace River region. In 2011 a suitable site for the proposed nursery was purchased in a location between the two partner First Nations communities. The economic viability of the proposed nursery was analyzed and supported by a KES business feasibility study (Meade et al. 2012). Nursery construction began in 2012 with the arrival of two 35' x 200' greenhouses and an operational business plan was produced in 2013 based on the results of the KES business feasibility study (TSNPN 2013). The nursery was incorporated as Twin Sisters Native Plant Nursery (TSNPN) in the summer of 2013. A pilot crop of native plants was propagated in 2013 in preparation for the initial commercial crops that will be produced in 2014.

While the creation of the Twin Sisters Native Plant Nursery represents a major achievement involving the creative energies of a multitude of private and First Nations interests, its existence alone does not guarantee the successful incorporation of native and culturally significant plant species into reclamation practice. A number of related and integrated programs are also required to support the financial success of TSNPN and by extension, produce a large volume of native and culturally significant plants for use in reclamation activities in the Peace River region. These programs include collaborative ecological and ethnobotanical research, the delivery of a native plant horticulture training program, and the development of a native plant species best practices reclamation guidebook. This integration of business, education and research supporting native plant production has application to other regions of British Columbia where the use of native and culturally significant plant species in reclamation may be hindered by an insufficient supply coupled with a need for more effective collaboration between First Nations and industry[2].

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[1] Culturally significant plant species are native plant species that hold importance with local peoples (usually First Nations) as food, medicine or technological resources.

[2] All of the programs we describe in this paper were initiated in 2010 or later and are in their initial stages of delivery or development. This paper, therefore, does not report on the results of these projects but rather provides a description of their development and current status.

PROJECT DESCRIPTIONS

Ethnobotanical Research

Ethnobotanical studies seek to document cultural uses of plants using culturally appropriate and scientifically valid methods, which often include extensive interviews with keepers of Traditional Ecological Knowledge (TEK) [3]. Ethnobotanical research that identifies native plant species with cultural significance is required for the successful incorporation of these plant species into ecological restoration activities.

Successful ethnobotanical research is a *collaborative* process implying that participating First Nations are involved in all stages of the research program; from the initial stages where research agendas are determined through to the dissemination of research results (Smith 1999). Collaboration in the context of ethnobotanical research also implies that First Nations partners retain the intellectual property rights for all data obtained through the program as well as the right to limit the dissemination of results concerning sensitive TEK (Ermine et al. 2004; Smith 1999). Ethnobotanical research should also be *current* and *local*, meaning that research must be site specific and involve knowledge holders as research partners. Reviews of existing literature may be a starting point in such research, but alone are insufficient for the identification of culturally significant plant species as TEK is an oral and dynamic form of knowledge that is substantially altered when committed to writing and translated to English (Simpson 1999; Smith 1999). Substantial First Nations involvement at all stages of the research program helps to ensure the culturally appropriate collection of data and accurate interpretation of research results.

The selection of culturally significant native plant species for propagation at the Twin Sisters Native Plant Nursery has been supported by ethnobotanical research initiated by Saulteau First Nations in collaboration with Walter Energy and Keefer Ecological Services. Initial ethnobotanical research conducted in 2012 and 2013 involved three collaborative meetings with Saulteau First Nations Elders and resulted in a preliminary list of culturally significant plant species. Data collected through this research program will be held by Saulteau First Nations in perpetuity in deference to the highly sensitive aspects of Traditional Ecological Knowledge. Elements of TEK emerging from this program that are less sensitive may enter the public domain through the production of an ethnobotanical handbook. For their contribution to the research, Walter Energy will be provided with a list of important cultural plant species for consideration for use in reclamation which will be annotated with cultural information about these plants, including their uses as food, medicine or technology. Saulteau First Nations Elders are supportive

[3] Traditional Ecological Knowledge (TEK), sometimes called Traditional Knowledge (TK), Traditional Ecological Knowledge and Wisdom (TEKW) or Indigenous Knowledge (IK), is a controversial term that is not easily defined (Simpson 1999). For the purpose of this paper we define TEK as "knowledge of ecological principles, such as succession and interrelatedness of all components of the environment; use of ecological indicators; adaptive strategies for monitoring, enhancing, and sustainably harvesting resources; effective systems of knowledge acquisition and transfer; respectful and interactive attitudes and philosophies; close identification with ancestral lands; and beliefs that recognize the power and spirituality of nature" (Turner et al. 2000, p. 1275).

of this program and funding has been secured for additional research to continue with the creation of a comprehensive list of Saulteau First Nation culturally significant species.

Ecological Research

Without detailed knowledge of the habitats that existed prior to disturbance at a reclamation site, native plant species with high potential for use in reclamation are identified through detailed surveys of nearby habitats that have recovered from disturbance without human assistance. Working with Keefer Ecological Services, Walter Energy has supported ecological research in the Peace River region to aid in the selection of appropriate native plants species for propagation at the Twin Sisters Native Plant Nursery. In 2011-2012 data were collected from a series of ecological plots in the areas surrounding Walter Energy mines. Plot work focussed on areas that had experienced natural or anthropogenic disturbances without subsequent reclamation and included landslides, old drill pads and roads, soil stockpiles, cutblocks, gravelly streambeds, and other disturbed locations. Species diversity, above-ground biomass, terrain information and soil data were collected to model the native plant species expected to thrive in the post-mine environment depending on ecological conditions present, and to guide prescription development, seed collection and associated plantings on sites. Further information on these methods can be found in a related study conducted in the East Kootenays in 2010 (Keefer et al. 2011).

Growing Our Futures: Native Plant Horticulture Training Program

Skilled nursery workers are needed for the efficient production of quality native plant seedlings. The Twin Sisters Native Plant Nursery business feasibility study found that a lack of qualified horticultural workers in the Peace River region would likely impede the initial success of the TSNPN nursery (Meade et al. 2012). This issue has been addressed through the development of the *Growing our Futures: Native Plant Horticulture Training Program*, which was developed as a collaborative effort involving Saulteau First Nations, West Moberly First Nations, Keefer Ecological Services (KES) and Royal Roads University Centre for Livelihoods and Ecology (RRU-CLE). Funding for this program was provided by the Investment Agriculture Foundation of British Columbia (IAF), the North East Native Advancing Society (NENAS) and Walter Energy.

Over the summer and autumn of 2013 an instructional team composed of plant ecologists and nursery professionals delivered the pilot training program to students from the SFN and WMFN communities. The program is practical and hands-on, providing trainees with the required knowledge and skills to enter directly into the native plant nursery workforce or to pursue related horticulture and reclamation employment or educational opportunities. Cultural teachings and Elder instruction were important elements of the training program. Student feedback from the program has been overwhelmingly positive and 11 students will continue their learning with a minimum seven months of work experience at Twin Sisters Native Plants through to spring of 2014.

The *Growing our Futures: Native Plant Horticulture Training Program* was also piloted in the summer and fall of 2013 in collaboration with the Ktunaxa First Nation and Tipi Mountain Native Plants (TMNP) with funding from the Aboriginal Training for Employment Program (ATEP) and the British Columbia Aboriginal Mining Training Association (BCAMTA). This program provided a much needed theoretical basis and additional skill development to a group of 11 trainees, the majority of whom are existing nursery workers at TMNP. Future training initiatives elsewhere in British Columbia and Canada are currently under discussion with interested First Nations and post-secondary institutions. These training programs will develop a skilled workforce who will be able to effectively work at native plant nurseries and manage the involved processes required for the production of large volumes of native plant seedlings to be made available to industry clients, while at the same time providing much needed economic opportunities for First Nations individuals within their home communities.

Land Reclamation Best Practices Field Guidebook

The outplanting success of native plant species in reclamation practice will have large implications on their broad-scale acceptance as a viable alternative to traditional agronomic reclamation species. To enhance the abilities of reclamation practitioners to effectively incorporate native plants with traditional significance into reclamation projects, a practical and standardized outreach tool in the form of a *Land Reclamation Best Practices Field Guidebook* has been proposed. The first volume of this guidebook will be produced for the Peace River region by Keefer Ecological Services (KES), Encana Corporation and Royal Roads University Centre for Livelihoods and Ecology (RRU-CLE) with the support of West Moberly First Nations (WMFN), Doig River First Nation (DRFN) and Prophet River First Nation (PRFN). The guidebook will identify native plant species with broad ecological amplitudes, well known propagation requirements, certain wildlife attributes and species of cultural importance. Plant species identified in this way will be grouped by British Columbia Biogeoclimatic Unit (BEC unit) allowing practitioners to use the estimated BEC unit of a reclamation site to predict the native plant species appropriate for reclamation at that site.[4] Ethnobotanical research with WMFN, DRFN and PRFN will support the inclusion of First Nations cultural values alongside propagation information and techniques for incorporating these species in reclamation work. The format of the *Land Reclamation Best Practices Field Guidebook* will follow that of the BEC system of guidebooks in order to enhance its familiarity, utility and to encourage its widespread use.

CONCLUSIONS

Saulteau First Nations, West Moberly First Nations and Walter Energy Ltd. share a common goal of returning impacted lands to functioning native plant ecosystems in the Peace River region of Northeastern British Columbia. The use of culturally significant plant species, such as food or medicine plants, in reclamation meets the needs of traditional land users through the restoration of biologically and ethnobotanically diverse landscapes. At the same time, the incorporation of native and culturally significant species into reclamation practice aids industry in meeting commitments toward socially and ecologically sound reclamation. The Twin Sisters Native Plant Nursery represents a sound business opportunity for Saulteau First Nations and West Moberly First Nations while enhancing these communities' ability to participate in industrial development; employ community members, provide a new local tool for ecological restoration, and meet the market demand for native plants. The success of the Twin Sisters Native Plant Nursery, however, depends on an integrated business approach that includes

[4] Biogeoclimatic ecosystem classification (BEC) is a system of ecological classification widely used in British Columbia with units of classification resulting from a synthesis of vegetation, climate, and soil data (Pojar et al. 1987).

extensive and collaborative ethnobotanical and ecological research, culturally relevant native plant nursery training and the development of a *Land Reclamation Best Practices Field Guidebook* for the effective use of native plant species in reclamation. The ultimate goal of this innovative approach of integrating business development, research, training and outreach is to promote the expanded use of native and culturally significant plant species in reclamation in the Peace River region of Northeastern British Columbia and beyond.

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TOOLS FOR ARCTIC REVEGETATION: WHAT'S IN YOUR TOOLBOX?

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ABSTRACT

Revegetation in arctic climates is a challenge for many reasons. There are two approaches to arctic revegetation: natural regeneration and active reclamation. Natural regeneration is an inexpensive option that can provide a diversity of locally adapted species. This has been shown to be effective on smaller disturbances at the De Beers Snap Lake Mine. However, natural regeneration can be quite slow and will not work as well on large disturbances where seeds and spores have to travel a long way to populate disturbed areas. Intervention using active reclamation techniques may help accelerate establishment and maturation of reclaimed sites. Determining when and how to intervene can be challenging and can affect the results of reclamation efforts. Erosion, costs, accessibility, diversity, stress factors, size of disturbed area, and rate of succession must be considered and, in some cases, a combination of solutions may be required for specific areas or for a whole site.

Key Words: Revegetation, Natural Regeneration, Active Reclamation, Arctic, Seeding, Ecological Intervention.

INTRODUCTION

Natural resource exploration (mining and oil) in the Canadian North has intensified in the last 20 years, leading to severe environmental disturbances that will need to be reclaimed in the future. Studies of the patterns of disturbance and natural revegetation in various regions have contributed to the body of knowledge on arctic ecosystem recovery. Different approaches to enhance revegetation and ecosystem regeneration (active reclamation) have been described in several studies and reviews (Adams and Lamoureux 2005; Baasch et al. 2012; Drozdowski et al. 2012; Firlotte and Staniforth 1995; Forbes and Jefferies 1999; Jorgenson and Joyce 1994). The numerous examples of natural regeneration and spontaneous succession occurring in disturbed sites suggest that active reclamation may not always be the best answer to regenerate ecosystems after disturbance (Holl and Aide 2011; Prach and Hobbs 2008).

Revegetation programs, in general, have common challenges such as shortage of commercially available native seed, a lack of understanding of propagation protocols (lichens, mosses and vascular plants), a shortage of facilities to propagate native species, and timely and cost effective protocols to determine quality, viability, and vigour of stored seed and propagules. In arctic climates, revegetation is even more challenging due to unique development constraints, including low air and soil temperatures; short growing season; permafrost; irregular surfaces and moisture regimes; limited access to site in the warm, summer season; and the slow growth rate of arctic species (Adams and Lamoureux 2005; Drozdowski et al. 2012; Forbes and Jefferies 1999).

Revegetation is a complex term, often broadly lumped with the terms *restoration*, *re-seeding*, *reclamation*, *land rehabilitation*, and *erosion control*; although related, these terms differ in purpose and definition. For the purposes of this paper, we will use the following definitions (Figure 1).

RE-VEGETATION: replacing or allowing for the establishment of vegetation on a disturbed site

RECLAMATION: returning disturbed land to a useful purpose, in this case a functioning native ecosystem, using active of passive techniques

NATURAL REGENERATION: allowing the establishment of and ingress of vegetation with no active intervention

ACTIVE RECLAMATION: re-vegetation by seeding or planting

SUCCESSION: changes in plant communities over time in both floristic and functional characteristics

INTERVENTION: active reclamation events used to enhance or accelerate natural regeneration

Figure 1. Key Definitions

There are two schools of thought in arctic revegetation methods: natural regeneration and active reclamation. These two extremes each have benefits and drawbacks. Natural regeneration is an inexpensive option that can provide a diversity of locally adapted species; however, it can be quite slow and will not work as well on large disturbances where seeds and spores have to travel a long way to populate disturbed areas. Active reclamation programs provide immediate erosion control and can allow for planting of more mature individuals; however, this method can be expensive and time consuming and there are technical gaps, which could result in lower species diversity than the undisturbed environment. Determining when to intervene with active reclamation techniques be challenging and can affect the results of reclamation efforts. Site ecology, erosion risks, costs, propagation knowledge, accessibility, diversity, stress factors, size of disturbed area, and rate of succession must be considered and, in some cases, a combination of solutions may be required for specific areas or for a whole site.

In this paper, we will discuss some of the issues and suggest potential approaches to maximize reclamation success in the Arctic. We will focus on a case study on the DeBeers Snap Lake Mine site, where natural revegetation has been shown to be successful on small disturbances over a period of several years in the arctic region. We will review the data and discuss what other approaches could be applied to enhance natural regeneration and/or reclamation success in the Arctic.

NATURAL REGENERATION

As mentioned previously, there are generally two major methods which can be used to establish vegetation on disturbed sites; natural regeneration and active reclamation. While the easiest and cheapest method would seem to be to allow natural succession to occur and heal the system, for reclamation projects the success rate and the time needed for a stable system to develop may be unacceptable in the regulatory sense. In addition, stakeholder perceptions of this method tend to be negative. In the following sections we will discuss these methods, their pros and cons, and their rationale for use.

Succession has been defined in various ways, but in this instance it means the changes in plant communities over time in both floristic and functional characteristics. Function as used here, is the collective intraspecific and interspecific interactions of the biota within the ecosystem. Revegetation of disturbed sites involves more than the simple replacement of vegetation. For mining sites where disturbances include the removal of all characteristics of the ecosystem (i.e., vegetation, soil, and topography), the simple replacement of vegetation may not restore function to the systems and reclamation efforts could fail. In early reclamation efforts, vegetation was seen as a means to an end, such as soil stability; what is "green" is recovered. These efforts often utilized standard agricultural or urban revegetation methods in the context of strip mines and pipeline corridors (Adams and Lamoureux 2005; Forbes and Jefferies 1999). While many of these methods were successful at reestablishing vegetation to disturbed areas, these early attempts often resulted in the persistence of nonindigenous species and little establishment of native vegetation (Densmore and Holmes 1987; Johnson 1981). For example, Kentucky bluegrass (*Poa pratensis* var. nugget) and red fescue (*Festuca rubra* var. arctared) were seeded on drilling pads in the Alaska tundra in hopes of stabilizing soils and to act as a surrogate for native species (Younkin and Martens 1987). After 12 years both species persisted, and due to both species having extremely dense root mats and litter layers, less than 15% of the total cover was attributed to native species. The presence of vegetation had been restored, but the function of the ecosystem was not fully developed leaving a system lacking in diversity and ecosystem value and with decreased opportunity for native species establishment. Conversely, when Younkin and Martens (1987) investigated disturbed plots where natural regeneration was allowed to take place, there was an 80% cover of native species after 12 years.

Natural regeneration and active reclamation are "intrinsically linked" since both are mechanisms for recovery and a path to more established or mature ecosystems (Walker et al. 2007). However, natural regeneration relies on an active seed bank (buried seed communities) or seed rain (influx of seed) and an adequate substrate for seedling development. Gartner et al. (1983) found that one of the major factors governing natural revegetation in disturbed tussock tundra was the presence of a viable seed bank and the presence of some organic soils. In areas where long soil stockpiling reduces seed viability or the organic layer is not intact after disturbance, seed rain may be the only alternative for vegetation establishment using natural regeneration. This can be problematic when considering the patch size (i.e., scale) of the disturbance; larger disturbances could have reduced revegetation potential than smaller patches due to limited dispersal of propagules (Forbes et al. 2001). The severity of the disturbance can also impact the ability of the system to recover. The prevalence of mineral-rich soils at most disturbed sites can limit both the reestablishment of species, which originally occurred at sites and species vigor (Gartner et al. 1983). This can impact the time required for natural regeneration to result in a stable ecosystem. Natural revegetation after a disturbance is slow in the arctic. In areas with heavy soil erosion potential, natural regeneration would not provide adequate protection within the first few years of disturbance (Adams and Lamoureux 2005).

The success of natural regeneration on mine sites in the tundra is not well known, since most mines are required to develop and maintain active reclamation efforts at mine closure. However, Kershaw and Kershaw (1987) were able to find 80 un-reclaimed borrow pits in the tundra of northwestern Canada that had been allowed to be revegetated naturally. Disturbances of various ages between 5 to 35 years were observed, and in all cases sites were colonized by native and some non-native species. In general, they found 433 taxa and although most of the successful colonizing species were herbaceous, several woody species (mainly *Salix* spp.) were successful in terms of cover and number of sites established.

A study of natural regeneration of disturbances in the artic is in its early stages at De Beers Snap Lake mine where reclamation plots have been left to regenerate since 2002 with periodic monitoring. The preliminary results of this study are discussed below.

SNAP LAKE MINE

A current study that has been initiated is at the Snap Lake Mine, a diamond mine owned and operated by De Beers Canada Inc. (De Beers) and located approximately 220 kilometres northeast of Yellowknife, Northwest Territories, Canada. Portions of this mine have been removed from operations and have been used in a reclamation monitoring program. Two types of monitoring plots were established; a set of natural plots (control plots) to give a benchmark for recovery, and later a set of plots within disturbed locations (reclamation plots) to monitor vegetation changes to the sites without active intervention. Reclamation monitoring plots were established when a location was released from mine production. Most of the current reclamation plots were last disturbed in the summer of 2002 and include a gravel quarry and a decommissioned camp site. These sites have been allowed to naturally revegetate, and to date (summer 2013) no assisted reclamation has been applied to these plots. Both control and reclamation plots were surveyed in 2004, 2008 and 2013 for vegetation type, vigor and cover. The 2013 data will not be presented in this report.

Figure 2 shows the species richness (i.e., number of species) of vegetation found in both the control and reclamation plots two and four years after disturbance. Only two ecological land classifications (ELC) are found in both the reclamation and control plots; the Tussock Hummock and Heath/Boulder communities. The plots show that, for species richness, the sites display an impressive rate of recovery. It is assumed that all vegetation present on reclamation plots is either from existing soil seed banks or seed rain from the surrounding vegetation communities. The species composition of reclamation plots contained no non-native species and a high proportion of shrubs and moss species compared to grasses. This is similar to the findings of Kershaw and Kershaw (1987). However, when the average percent cover of vegetation within the reclamation plots is considered it is apparent that the vegetation cover is still much less than the control plots (Figure 3). While this cover may not be enough to ward off erosion in areas with steeper slopes, it does indicate that natural regeneration in the tundra can be a valuable tool.

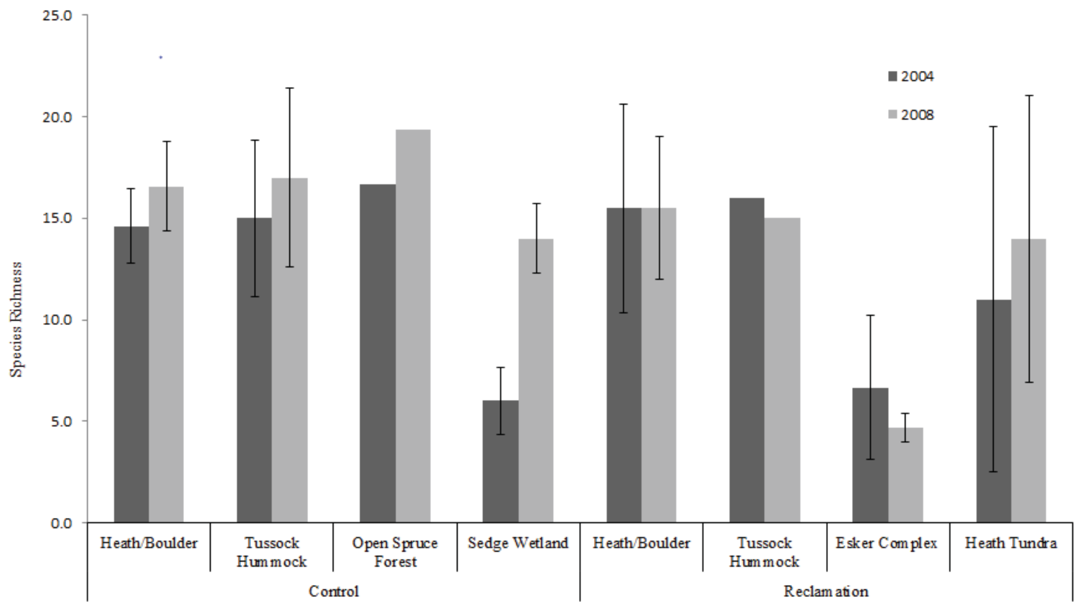

Figure 2. Species Richness (i.e., number of species) For Ecological Land Classifications in Control and Reclamation Plots for the Snap Lake Mine Site

As more sections of the mine are released from production, additional reclamation plots will be added to the reclamation monitoring program. The mine is expected to remain active for at least 20 years and reached full production in 2008. While these plots were not originally designed to investigate natural regeneration, the potential to add more plots and monitor natural change to this disturbance is valuable. However, can natural succession do enough to recover these sites, especially considering that future areas of reclamation are likely to be larger and more intensively disturbed?

THE DECISION PROCESS

Unfortunately, there is not one solution for all revegetation programs in all situations, and in most cases a combination of methods is likely required. The ultimate choice in reclamation approach will depend on the ecology of the site, the type of disturbance and the goals of the reclamation program. There is a spectrum of active intervention in reclamation programs, which is influenced by a large number of factors. Figure 4 shows several of the factors that need to be taken into account when choosing how to revegetate and when to intervene.

Invasives

A site with a large population of invasive or non-native species will be less prone to natural regeneration as there will be increased competition from non-native species reducing the likelihood of functional ecosystems developing without intervention. Interventions may include mechanical or chemical weed control or, in appropriate situations, planting of a "nurse crop" consisting of an annual surrogate, which may help limit invasive establishment. Caution should be taken using this approach as in certain situations surrogates can limit ecosystem development.

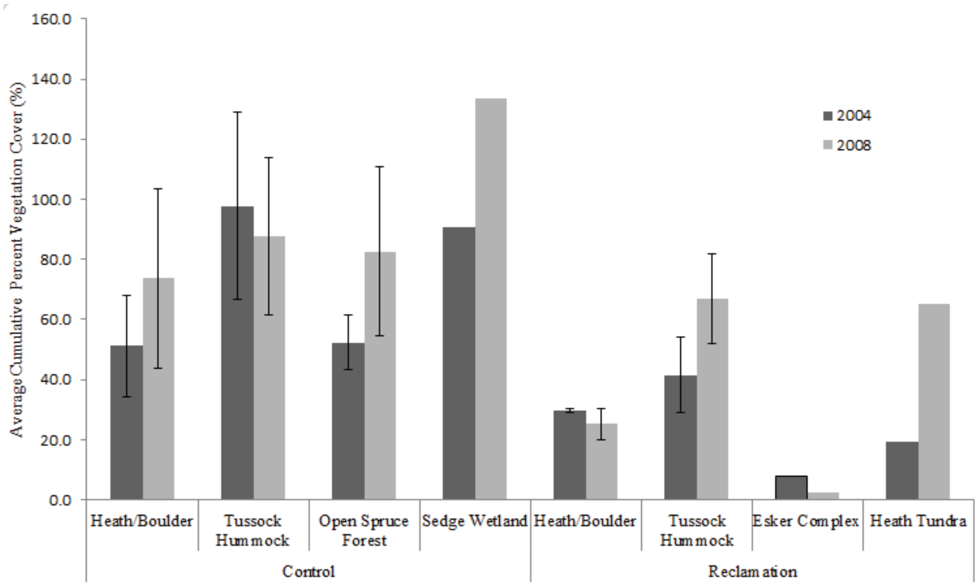

Figure 3. Average Cumulative Percent Vegetation Cover for Ecological Land Classifications in Control and Reclamation Plots for the Snap Lake Mine Site

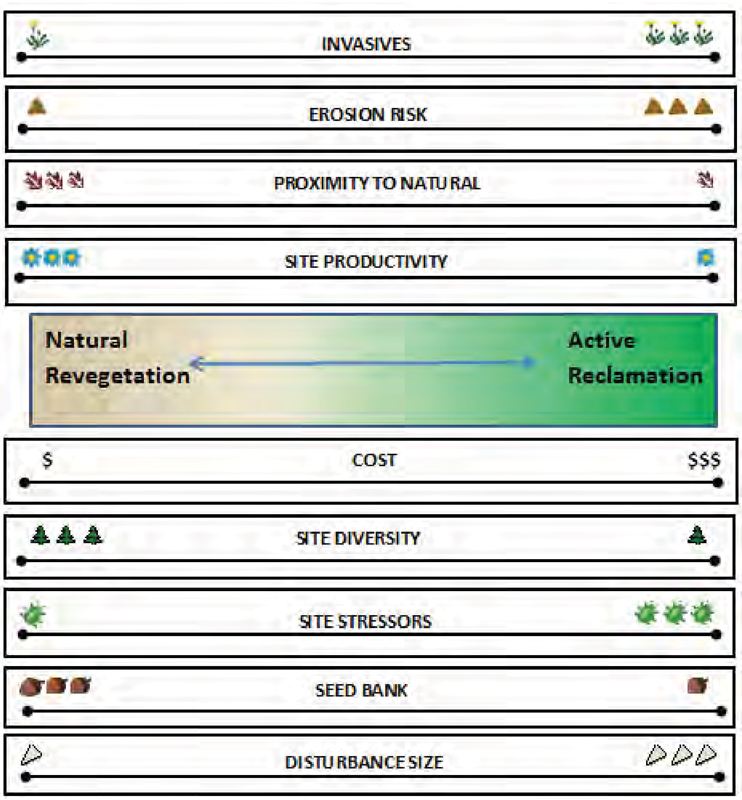

Figure 4. Decision Factors in Choosing Revegetation Methods

Erosion Risk

Natural regeneration can be a slow process especially in arctic climates. In areas with high erosion risk (i.e., steep slopes or loose soils) waiting for vegetation to establish naturally will not provide sufficient protection to conserve soil and maintain landform stability. In these situations, interventions such as planting of plugs, spreading seed, hydroseeding or installation of mechanical controls (e.g., silt fences or geotextile) may help maintain soil integrity while allowing for native species to ingress.

Proximity to Natural or Undisturbed Ecosystems

Ingress of native species through seed rain and vegetative propagation is more likely to occur in a site that is adjacent to an undisturbed ecosystem. This is particularly important in sites that do not have an active seed bank due to long periods of disturbance, impacted soil or spreading of soil that has been stockpiled for long periods of time

(Cooper et al. 2004). Reclamation areas that are surrounded by other disturbances may require intervention to begin vegetation establishment.

Site Productivity

Highly productive sites are more likely to regenerate naturally. High nutrient levels, appropriate moisture conditions and ideal landscape position mean that these types of sites will require little to no intervention to establish a diverse cover of native species in a relatively short period of time. Caution should be taken to monitor these sites regularly as they will also be susceptible to the ingress of non-native or invasive species.

Site Diversity

Generally speaking, moderate sites are the most diverse ecosystems. In these types of situations natural regeneration is preferred as it is more likely to allow for the establishment of a similar diversity of species after reclamation as was found before disturbance. It would be difficult and costly to achieve this level of diversity using only active reclamation; however, a combination of methods may produce excellent results on these types of sites. Arctic ecosystems are a bit of an exception to this rule because they have relatively low diversity, but because the species found in these environments are uniquely adapted to the short growing season and cold winters, they seem to respond well to natural regeneration. This may be partially due to the fact that most arctic species have little to no dormancy, making seeds ready to germinate as soon as they encounter the opportunity (Densmore 1992).

Cost

This is often a factor in choosing reclamation approaches and is usually offset by the time available to achieve reclamation goals. If natural regeneration is possible and several years are available to allow the ecosystems to develop, this method will have very low costs. However, if a site is left to naturally regenerate in the wrong conditions, invasive species, erosion issues and lack of plant establishment can simply defer costs to a later date and extend timelines even further.

Site Stressors

A site that is highly stressed due to disease, insects, contamination or other factors will need some level of intervention to either remediate the stressors or to establish communities that are immune to the specific conditions present on each site. Highly stressed sites can be costly to reclaim and often involve intensive monitoring and ongoing mitigation until the site reaches a self-sustaining state. These are often the situations where novel communities are established as native ecosystems are not suitably adapted to the conditions. Natural regeneration is not preferred in these situations.

Seed Bank Viability

The seed bank in stockpiled soil maintains its viability for a maximum of 15 months (Mackenzie and Naeth 2009). This period may be longer in colder climates, but there is still a relatively short time where stockpiled soil can be placed back and still contribute a viable seed and propagule bank. Seeds banks may also be compromised in areas where soil is compressed, eroded away or contaminated. In these cases natural regeneration may only be possible if there is an adjacent undisturbed ecosystem that can contribute seed rain or vegetative propagation to help the establishment of native vegetation. For large disturbances with a depleted seed bank, some form of intervention is likely required.

Disturbance Size

Large disturbances are more difficult to reclaim due to increased erosion risks and further distance from seed and propagule sources.

Vegetation Propagation Knowledge

Certain species do not propagate well in a greenhouse setting, others are difficult to grow outside their natural habitat (Hagen 2002). The target species may determine if active revegetation is possible at all. In some cases collection, storage and propagation of the seed of target species is not financially or physiologically feasible. The more you know about the species you are trying to grow, the more information you can glean about how reclamation is most likely to be successful.

Intervention

Intervention in reclamation may include any number of activities such as: planting woody species; planting surrogate crops; installing erosion control; fertilization; seeding; recontouring; topsoil building or rehabilitation; thinning; weed control; underplanting; and adding biodiversity and wildlife enhancing features.

Most intervention occurs at the initial reclamation stage through site contouring, soil placement, planting or seeding. Other interventions such as weed control and replanting of unvegetated areas are commonly applied. However, many other potential enhancements are available such as planting understory species when the appropriate successional stage is reached; thinning the overstorey species to accelerate succession; leaving rock piles, brush piles or other refugia on site for wildlife habitat use; and establishment of microsites to increase diversity (Jorgenson and Joyce 1994).

When to use Non-native Species

Direct seeding with either native or non-native species (non-invasive) has been shown to help reduce erosion, reduce dust, retain soil moisture and stabilize ecosystem processes (Adams and Lamoureux 2005; Densmore 1992; Firlotte and Staniforth 1995; Rausch and Kershaw 2007). In areas where topsoil disturbance has reduced the viability of the natural seed bank or where disturbance is so great that surrounding vegetation cannot adequately provide propagules, direct seeding may be needed. The use of indigenous species over introduced species would seem to be the logical choice when it comes to reclamation. Native species have evolved to survive and grow in the local environment while non-native species may not be able to thrive in these conditions. However, native species are generally perennials, not adapted to large soil disturbances and often re-vegetate at a slower rate than non-native species more adapted to early successional conditions (Adams and Lamoureux 2005; Reynolds and Tenhunen 1996). In addition, the lack of commercially available native seed sources limits the practical use of many native species (Rausch and Kershaw 2007).

The assumption that non-native species will make a good surrogate for native species has been used successfully in some reclamation efforts. Chapin and Chapin (1980) successfully used non-native grasses

(*Phalaris arundinacea*, *Poa pratensis*, *Lolium perenne*, *Festuca rubra*, *Phleum pratense* and *Alopecurus pratensis*) in an attempt to recover disturbed tundra. These non-native species established within one growing season, densities dropped off considerably by the third year and were virtually eliminated after five years. During this time native cottongrass (*Eriophorum vaginatum*), Bigelow's sedge (*Carex bigelowii*) and other native species had increased in abundance within disturbed areas. Similar results have been reported by other studies (Densmore and Holmes 1987; Johnson 1981; Webber and Ives 1978).

On the other hand, where non-natives remained in the environment and reduced native species establishment, a number of studies have found a detrimental effect caused by the establishment of non-native species (Cargill and Chapin 1987; Densmore 1992; Forbes and Jefferies 1999; Younkin and Martens 1987). Use of non-native seed species should be done with caution and may be effective in some areas. If time is not of concern, then seeding with native species would be the best solution for recovering disturbance; however in areas with high erosion potential surrogate non-native species may be needed.

OTHER CONSIDERATIONS

Laboratory bench-scale studies could be used to model and optimize approaches to revegetation efforts in the arctic. Studies can be conducted year round, as opposed to a limited time in the field and experimental conditions can be controlled to closely simulate natural soil, temperature, photoperiod and moisture conditions likely encountered by plant species in arctic sites.

There are many factors that may influence the success of reclamation strategies. The quality and viability of seed for native species have a critical impact on the success of revegetation. Although reproduction by seeds is not common in arctic environments, relatively few species have dormant seeds, which allows for germination whenever conditions permit (Bell and Bliss 1980). Seed viability testing can provide an accurate estimate of the potential germination success in the field, which may help determine the condition of the seed bank or the viability of seeds for propagation. Rapid laboratory based seed viability assessment methods such as electrical conductance and tetrazolium (TZ) testing (Miller 2010) can be performed in 2 to 3 days rather than the weeks or months required for full germination studies. Although the TZ test and other rapid screening methods do not account for germinability, factoring in the bulk weight of the seed will provide a measure of the amount of seed required to achieve a desired application rate and potential emergence rate for a given species of seed in the field.

The TZ test has been effectively used to test the viability of several arctic species and methods can be adapted for use with other species as well. Species with established testing protocols include: black spruce (*Picea mariana);* tamarack (*Larix laricina*); white spruce (*Picea glauca*); birch species (*Betula* sp.); small bog cranberry (*Oxycoccus microcarpus*); cranberry, blueberry, bilberry (*Vaccinium* sp.); sedges (*Carex* sp.); and reed grass (*Calamagrostis* sp.). The TZ test is not useful for determining the viability of groundcover species such as lichens and mosses (bryophytes).

CONCLUSIONS

In certain environments, such as the Arctic where species are specialized and conditions are extreme, natural regeneration may be a preferred option. However, innovative approaches are required to enhance and accelerate reclamation through interventions at various stages of development. By taking into consideration site characteristics and the goals of reclamation, specialized approaches can be developed to successfully reclaim disturbed sites in a variety of climates and conditions.

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GALENA HILL, YUKON, ECOSYSTEM MAPPING PROJECT

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ABSTRACT

The Galena Hill Ecosystem Map (GHEM) was initially developed to provide information about existing plant communities and their growth conditions to guide upcoming reclamation efforts at the historical silver mining area around Keno, Yukon. Disturbed areas and soil covers on mine wastes will need to be vegetated to reduce soil erosion, enable evapotranspiration and eventually integrate with the surrounding landscape. The GHEM project used the guidelines develop by the Yukon's Ecological and Landscape Classification (ELC) working group (ELC Working Group 2011).

Key Words: Ecosystem, Ecozones, Ecoregions, Bioclimate Region, Bioclimate Zone, Bioclimate Subzone, Ecosites, Polygons, Reclamation, Restoration, Revegetation.

INTRODUCTION

Elsa Reclamation and Development Company Ltd. (ERDC), a unit of Alexco Resource Corporation (Alexco), is responsible under a funding agreement with the Governments of Canada and Yukon, for the care and maintenance and the eventual closure of the former United Keno Hill Mine (UKHM) site (Yukon Government 2009). Reclamation planning and implementation has been ongoing since the former UKHM site was transferred to Alexco in 2007. Numerous investigative projects have been initiated to assess the extent and degree of remediation required to stabilize and reduce past mining impacts. The ecosystem mapping project is part of this investigative program. Its main purpose is to inventory the vegetative communities and growth conditions that currently exist in the Galena Hill to inform restoration planning, installation and subsequent monitoring. The intended objectives of the GHEM project are:

- A means to integrate abiotic and biotic ecosystem components that can be presented on one map;
- Develop a record of current vegetation communities and ecological site conditions that can be used as a framework for monitoring ecosystem response to changes;
- A means to locate areas of disturbance, sources for reclamation materials, different successional stages and sensitive areas;
- Provide in situ templates for revegetation efforts;
- Identify possible locations for seed collection and plant stock that match the environmental conditions of revegetation areas; and,
- Use of ecosystem plots that are established during the ground truthing phase as references during closure and reclamation phase for natural vegetation succession, nutrient cycling, and soil elemental profiles.

The first section of this paper will briefly describe the hierarchical framework of ecosystem mapping in the Yukon. Then the general process used in the development of the GHEM. The latter half of this paper is a discussion on the challenges and benefits the GHEM projects and finally recommendations for further studies.

HIERARCHY OF LANDSCAPE ECOLOGICAL CLASSIFICATION

Ecosystem: "An observable unit of the landscape with relatively uniform vegetation (a plant community) occurring on relatively uniform soil conditions" (ELC Working Group 2011).

The main premise of the Yukon Ecological and Landscape Classification (ELC) system is that climate is the foundational environmental factor that influences the type of ecosystems found in the territory. The ELC system begins at a broad spatial level and then as the scale increases more detailed information regarding climate, terrain, soil and vegetation, can be integrated until localized ecosystems can be recognized and classified (RIC 1998a). Over thirty years of research has gone into developing a Yukon focused ecosystem classification system and a formalized approach is still being synthesized (Lipovsky and McKenna 2005). The ecosystem mapping project of Galena Hill drew upon the main concepts that are currently recommended by the ELC. However it must be recognized that information currently available is limited as the Yukon Interior Plateau Ecoregion has only recently been classified to Bioclimatic Zone level. The regional classification hierarchy is briefly described below, and is a work in progress (ELC Working Group 2011).

Bioclimate Region

Bioclimate regions represent areas of broad, relatively homogeneous climatic conditions (Grods and McKenna 2006). The location and orientation of major mountain ranges and plateaus, interacting with territorial-scale weather patterns, create distinct regional climates throughout Yukon. Bioclimate regions generally correspond to Yukon ecoregions (Smith et al. 2004), with a few exceptions. There are ten recognized Bioclimate regions identified within the Yukon Territory, but these are considered provisional as research is still ongoing. The Galena Hill study area is within the northern portion of the Yukon Interior Plateau Bioclimate Region.

Bioclimate Zone

The bioclimatic zones are broad areas of similar regional climate that are characterized by distinctive plant communities and their distribution on the landscape. Bioclimate zones result primarily from changes in elevation and/or latitude. Within each bioclimate region, a bioclimate zone has a characteristic range in elevation and corresponding temperature and precipitation conditions. In mountainous areas, bioclimate zone boundaries are visible as relatively abrupt changes in general vegetation communities along an elevation gradient. In lower elevations or rolling terrain, bioclimate zone boundaries may be subtle and transitional (ELC Working Group 2011).

There are seven provisional general bioclimate zones currently recognized in Yukon; Alpine (ALP), Sub-alpine (SUB), High Boreal (BOH) and Low Boreal (BOL). The Wooded Taiga (TAW), Taiga Shrub (TAS) and Tundra (TUN) are bioclimatic zones that replace BOL and BOH, respectively, in more northern Bioclimatic Regions, and are not of concern in the Galena Hill area. The Galena Hill study area occupies two bioclimatic zones BOH and SUB. Adjacent areas that are within the former UKHM site also have the ALP bioclimatic zone; these areas have not yet been delineated nor interpreted. Table 1 defines the bioclimatic zones found in the Yukon Interior Plateau – North and the percentage of each zone that is represented on the GHEM.

Table 1. Bioclimatic Zones and Definitions

Bioclimatic Zone (elevation range)	Percentage of Total Area	Definition
Low Boreal (200 m to 500 m)	0%	Forested valleys and lower slopes composed of white/black spruce and aspen, moderately developed shrub layer. Non-forested areas include: wetlands, riparian, exposed soil/rock and anthropogenic structures. BOL did not occur within the Galena Hill study area.
High Boreal (500 m to 1100 m)	36.4 km^2 70.9%	The boreal highland forested areas are a mix of subalpine fir and White Spruce with a lichen and moss understory on the majority of the slopes. Late seral areas have Alaskan birch and tall willows as the dominant tree cover. Upper elevation forests are subalpine fir dominant with moderate to well-developed shrub layer. Non-forested areas include: wetlands, riparian, avalanche tracks, exposed soil/rock and anthropogenic structures.
Subalpine (1100 m to 1450 m)	14.9 km^2 29.1%	Open to sparse forest canopy cover, main trees species is sub-alpine fir. A well-developed shrub layer composed mainly of scrub birch and willow replaced forest cover with only a few widely scattered Sub-alpine fir.
Alpine (1450 m+)	0%	Alpine communities include dwarf ericaceous shrubs, dwarf birch (*Betula* sp.), willow (*Salix* spp.), grass/sedges (*Gramineae*), lichen, and bare bedrock at elevations above the tree line on Galena Hill only the very highest portion of its ridgeline was in this bioclimatic zone, less than 1 hectare, so was not delineated out.

At higher latitudes the boundaries of these bioclimatic zones decrease in elevation as annual temperatures are lower, soil development and nutrient cycling is also slower. The Keno area is near the 64° latitude mark. The treeline is at approximately 1300 m on northern aspects and 1360 m on southern aspects. Most of the study area is located on the northwest side of Galena Hill. The study area was restricted between 700 m to 1400 m elevation range.

Bioclimate Subzone

Bioclimate subzones have characteristic vegetation communities reflective of each bioclimatic zone; ALP, SUB, etc. but are in different regions influenced by different climates, for example, the plant communities that grow in the Kluane and Ruby Range Bioclimatic Region will be different than the plant communities in the Interior Plateau Bioclimatic region (ELC Working Group 2011). The Interior Plateau Ecoregion had not been subdivided into Bioclimatic subzones at the time this report was written.

Ecosites

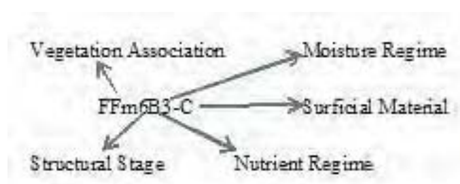
Within a Bioclimatic Subzone, ecosites are organized along landscape position, where certain plant associations occur at predictable locations based on slope, aspect, surficial material, and nutrient and moisture regimes (ELC Working Group 2011). The reference ecosite best reflects the climate of that specific Bioclimatic subzone. Meaning the reference ecosite would be in neutral landscape position that drains water at an equal rate at which it receives precipitation, usually on a moderate slope. The nutrient content of the soil is average and the aspect of the slope would be orientated East or West so solar exposure would be moderate (ELC Working Group 2011). The ecosites have not been formally established for the GHEM. The plant associations found during the GHEM ground truthing phase are situational based, just to verify polygon interpretations.

Ecosites are the most detailed division of ecosystem classification and used at a local scale. Ecosites are defined based on moisture and nutrient availability and landscape position. For example a ridge would shed water faster than it would collect water, so this landscape position would be considered dry and nutrient poor. The other ecosites within the same Bioclimatic Subzone are compared to the reference site according to the differences in moisture and nutrient availability and landscape position. Lower slopes would be moister, richer sites with vegetation association that has plants that require more water for growth as opposed to higher or more exposed sites that would host different plants that are drought resistant. Ecosites have characteristic vegetation associations that are described based on their mature or relatively stable successional phase (ELC Working Group 2011). The GHEM is a first step in defining the ecosites based on topographic position; more work will be needed before ecosite classification for this bioclimatic subzone is achieved.

Ecosystem Polygon Labeling

Each bioclimatic zone can be further delineated into vegetation polygons, which can be further divided into ecosystems that are based on vegetation associations, variations in moisture/nutrient regimes, and surficial materials. Each ecosystem is identified on the ecosystem map using four characteristic components (ELC Working Group 2011):

1) Vegetation association (vegetation)
2) Moisture and nutrient regime of site (soil)
3) Slope/aspect influences (climate)
4) Surficial material (terrain)



When two ecosystem codes are needed, deciles are put in front of each ecosystem units to indicate the percentage of each identified ecosystem that is present in the polygon.

70% Fir-Feathermoss-Mature Stand-Poor Nutrient Level-Submesic-Colluvial

7FFm6B3-C/3EsWi3aC5-F

30% Shrub birch-Willow Short Shrub-Average Nutrient Level-Subhygric on Alluvial

RESULTS

The Ecosystem Map of Galena Hill (draft) is the main outcome of the 2012 ecosystem investigative program. It presents the spatial relationship of the local ecosystems within the study area. Each polygon conveys information regarding vegetation association(s), structural stage, nutrient and moisture regime and surficial material. The different colour hues are used based on leading species of the different vegetation associations.

The 51.3 km^2 area that was mapped included the Galena Hill section of the former UKHM property; the BOH Bioclimatic Zone represented 70.9% which is equivalent to 36.4 km^2. The SUB portion at 29.1% covered 14.9 km^2. There were 156 polygons delineated, interpreted and assigned an ecosystem(s) code on the Galena Hill Ecosystem map.

Information provided from aerial interpretation, plot data sheets, field notes and photographs resulted in 42 different vegetation associations identified (see Table 2). The vegetation associations are tentative as they are based on a limited number of ecosystem plots (36) completed during the ground truth phase.

Surficial materials placement was based on the 1998 Surficial Geology map prepared by J. Bond and aerial interpretation done during polygon delineation. Ground surveys also provided local scale confirmation of underlying parent material from trenches and road cuts.

Nutrient codes are letters A to E, where A is very nutrient poor and E is very rich. Moisture regime codes are numbers 0 to 8, where 0 is very xeric, 4 is mesic and 8 is hydric (water is at or above soil level).

In Table 2 the plant associations, found in the BOH or SUB, are expressed in codes and the ecosystem where they were found is described.

Table 2. Codes and Descriptions of Plant Associations and Ecosystem Codes within GHEM Bioclimatic Zones

Plant Association	Ecosystem Code	Description of Ecosystem
SUB-ALPINE (1100 m to 1450 m)		
Heather-Lichen	HLi	Colluvial - Dwarf shrub communities, heather (*Cassiope tetragona* and Phyllodoce), crowberry (*Empetrum nigrum*), lingonberry (*Vaccinium vitis-idea*) and lichen, a few grasses. Exposed well drained soils. Upper Sub-alpine, exposed rocky area, A(B)2-3.
Shrub Birch-Willow Feathermoss	EsWiFm	Morainal - Moist upper mountain gentle slope, variable aspects, solifluction lobes may be present ground hummocky, often extensive coverage. B3-5
Fir-Sw-Shrubs	FSwSh	Lower Sub-Alpine, mature trees, high diversity of shrub species, often on glacial fluvial deposits or colluvial. B3-5.
Fir-Sw-Feathermoss	FSwFm	Upper/middle slopes, well drained on colluvial medium to coarse soil texture. Varied aspects. Lower Sub-alpine. B4.

Plant Association	Ecosystem Code	Description of Ecosystem
Fir-Shrubs-Feathermoss	FShFm	Open canopy mature fir, higher variety of shrub species, mainly on colluvial over morainal, medium soil texture. Moderate slopes.
Fir/shrub birch-willow	FEsWi	Variable aspects, dry to moist sites F>10% 5-3/B. Fir coverage decreases as elevation increases. Morainal or colluvial.
Fir-Alaskan birch-Feathermoss	FEnFm	Young Forest, morainal and colluvial over morainal. Regeneration after slumping or anthropogenic disturbance.
Fir-Feathermoss-Lichen	FFmLi	Moderate to steep slopes on colluvial. Shallow soils. Few shrubs open to sparse trees. B4-3.
Fir/Feathermoss	FFm	Morainal - Open to dense forests on mountain slopes various aspects. 5-3/B, also in High Boreal.
Carex-dwarf willow	WiCx	Shallow depressions or flat surfaces along cool aspects in upper Sub-alpine. Populated with dwarf willow species like: *Salix arctica*, *S. reticulate*, *S. pulchra* and *S. barratianna*. Other plants encountered were *Festuca altaica*, *Deschampsia cespitosa*, *Carex* sp. on morainal C5-7.
High Boreal (500 m to 1100 m)		
Aspen-Kinnikinnick	AAu	Glaciofluvial - Open/Dense Aspen with variable low shrubs, *Rosa acicularis*, forbs and grasses 4-3/C.
Aspen-Willow	AWi	Moderate to steep slopes open canopies on glacial fluvial
Aspen-Sw-Rose grasses	ASw	Glaciofluvial - well drained, steep slopes, south facing sides of river corridors, B3-2.
Alder-Willow	AlWi	Along riparian edges or old disturbances occasional flooding. Alluvial deposits sand, gravel, cobbles. C4-6.
Alder-Balsam-Popular-Willow	AlBWi	Fluvial - on flood plains deposit frequent flooding sand and gravel. C-6.
Balsam poplar - Willow	BWi	Fluvial - Floodplains, islands and older channels.
Balsam poplar-Shrubs-Forbs	BShFb	Natural regeneration of disturbed areas, where soils have been stripped. Gravel and some sand left, variety of shrubs; alder, willows, roses and Balsam poplar. On cut terraces where water can collect. B4-2.
Sedge-Cotton grass	CxEr	Organic - in fen areas 7-5/B tussocks and open water present, 8-5/B.
Calamagrostis-Sedge	CaCx	Morainal/organic-edges of streams and lakes 7-6/D.
Alaskan Birch-Sw	EnSw	Morainal - Cool, moist N facing slopes, hilltops and terraces. 5-3/B-C.
Alaskan birch-forbs	EnFb	Closed birch canopy, reduced shrub growth, forbs and mosses, gentle slopes North facing slopes C4-5.
Shrub birch-willow-Feathermoss-Sphagnum	EsWiFmSp	Organic - in lowlands with poor drainage, community. 6-5/B.
Lichen-Mosses	LiMo	On colluvial, primary succession. Lichens various forms. Mosses include *Polytrichum*, *Dicranum*, *Racomitrium* and others.

Plant Association	Ecosystem Code	Description of Ecosystem
Sb-Labrador Tea-Sphagnum	SbLeSp	Level to depression, organic, nutrient poor bog, B/ 5-7.
Sb-Sw-Feathermoss	SbSwFm	Morainal, with an organic veneer. All aspects, thick organic forest floor (~30cm) over till, *Ledum* common, over permafrost. B/6-4.
Sb-Shrub mix-Feathermoss	SbShFm	Mid to lower north facing slopes, thick moss carpet on morainal or colluvial over morainal, several species of shrubs, often associate with permafrost.
Sb-Ledum-Feathermoss	SbLeFm	Lower slopes and lowlands on organics. Open to sparse Sb canopy. Often complexed with EsWiFm.
Mixed Shrubs-Forbs	ShFb	Regeneration sites that have exposed soils and less moss cover can also be found under mixed or deciduous tree cover.
Shrubs-Feathermoss-Lichen	ShFmLi	Upper slopes, well drained, thick moss cover with lichen, sun exposed, some forbs, course textured soils, colluvial. B/C3-2.
Sw-Alsakan birch-Feathermoss	SwEnFm	Mixed forest on colluvial or glacial fluvial, previous disturbance.
Sw-Alaskan birch	SwEn	Well drained coarse soils, often found on knolls and colluvial. Shrubs rose, willow, bearberry, Kinnikinnick, graminoids and forbs. C3-4.
Sw-Willow-Crowberry	SwWiEm	Significant slope, cool aspect, deep medium textured soils.
Sw-Feathermoss	SwFm	Upland open to close forest, moderate to well drained slopes variable aspects. B4-3.
Sw- Balsam popular-equisetum	SwBEq	Alluvial - subject to infrequent flooding.
Sw-Balsam popular	SwB	Along waterways lower slopes and lowlands. Glacial fluvial and fluvial-shrub understory if open canopy.
Sw-Willow-Scrub Birch	SwEsWi	Upland forest, gentle slopes, deep medium textured soils. B3-5.
Sw-Scrub Birch-Cladina or Sw-Lichen	SwLi	Significant slope, warm aspect, shallow soils. Xeric to subxeric.
Sw-Alder-Equisetum	SwAlEq	Glaciofluvial - low terraces lower slopes or between channels, infrequent flooding, gentle slopes or flat. 6-4/C.
Willow and Equisetum	WiEq	Fluvial - older floodplains subjected to occasional flooding, 5/C.
Willow-Alder-Equisetum	WiAlEq	Riparian edges, occasional flooding, grasses often present.
Willow-Graminoids	WiGr	Along edges of lakes, ponds and slow flowing streams, alluvial. C 6-8.
Willow-Sedge	WiCx	Along edges of ponds, slow flowing streams or standing water in depressions. C 6-8.

The prevalent vegetative community in the BOH is White spruce-Subalpine Fir-Shrubs-Feathermoss (SwFShFm) and in the SUB it is the Shrub birch-Willow-Feathermoss (EsWiFm). The EsWiFm association is ubiquitous, occurring in both bioclimatic zones, and across the entire elevation range (700 m to 1400 m asl).

During the ground truthing phase several ecosystem plots were permanently established for future monitoring. The plots below were also selected to be used as references or possible seed sources for restoration sites.

Table 3. Selected Ecosystem Plots for Cover Trials and Source

Plot Number	Reasons	Ecosystem Unit(s)	GPS UTM Coordinates
SG004	Pioneering shrubs and forbs growing in very coarse substrate on disturbed land near old adit. Similar conditions to areas needing revegetation. HBOL.	4ShFb3bB2-R: 3LiMo1aA2-R	0482241E, 7088515N
FCW12	Established mature to old growth on northern aspect. Common vegetation association in study area. Background soil mineralization profile. BOH.	SbShFm6B5-Gf	0475958E, 7076246N
CCW9	Shrub dominant ecosystem at 1365 m elevation. Numerous disturbances nearby that are at different stages of natural revegetation, good comparison for SUB revegetation attempts. Possible Reference Ecosite.	EsWiFm3aC4-M	0481339E, 7087938N
FCW3	Exposed colluvial with primary succession of lichen and moss. Adjacent is a submesic low shrub successional stage. Shallow soils and poor nutrient levels.	5ShFm3aB3/5LiMo1bA1	0477593E, 7086423N
NCW14	Typical of road edge tall shrub stage regeneration. Source of shrub cuttings and seed collection for Balsam poplar, willow and alder.	3SwFm5B4C/ 7AlBWi3bB5-C	0478515E, 7089691N
FCW9	Edge of Husky waste pile, natural regeneration in Subhygric-hygric. Low shrubs and graminoids. Potential source of seeds for moister revegetation sites.	SwEsWi5B5/CaCx2bC6	0473916E, 7085984N

The table below lists the plant species that were frequently encountered on naturally regenerating areas, recovering from mining disturbances. Comments regarding how certain plants can be used in restoration are included.

Table 4. Possible Candidate Native Plants for Revegetation Efforts

Species	Comments
Willow *Salix alaxensis, S. pulchra, S. planifolia, S. arbusuloides*	Only use in wet locations and only use the species listed. Willow is the main plant used for staking of live/dormant cuttings and for bioengineered structures.
Poplar *Populus balsamifera*	Only use in wet/moist locations; easily established through staking of live/dormant cuttings.
Shrub birch *Betula glandulosa, Betula nana*	Only plant seedlings grown in a nursery from locally-collected seed. Tolerant of acid soils. Use on moist sites with good organic content.
Alder *Alnus crispa, Alnus tenuifolia*	Only plant seedlings grown in a nursery from locally-collected seed. Tolerant of slightly acidic soils and low nutrients. Use on moist/wet sites that are not alkaline.
Raspberry *Rubus idaeus*	Mesic – sub mesic sites.
Rose *Rosa acicularis*	Mesic – sub mesic sites.
Yellow locoweed *Oxytropis campestris*	Tolerant of drought and low nutrients. Low growing bunches. A nitrogen-fixing forb that is found in dry, granular disturbed areas.

Species	Comments
Showy locoweed *Oxytropis splendens*	Tolerant to drought and low nutrients. Low growing bunches. A nitrogen-fixing forb that is found in dry, sandy disturbed areas.
Bear root *Hedysarum alpinum*	Tolerant to alkaline soils, drought and low nutrients. A nitrogen-fixing forb that grows in a variety of alkaline sediments in disturbed areas at low to mid elevation.
Arctic lupine *Lupinus arcticus*	Tolerant to low nutrients, permafrost, limited drought. A low-growing, nitrogen-fixing forb. Grows mostly on moist soils in disturbed areas ranging from lowland riverbanks to alpine and tundra, also along roadsides.
Yarrow *Achillea millefolium*	Tolerant to alkaline soils, drought, low nutrients. Mostly found in areas with well-drained but poorly developed soil. Produces tiny seeds in abundance.
Wild Rhubarb *Polygonum alaskanum*	Possibility for drier clayey soils. May overgrow other candidate species. Check literature.
Northern rough fescue *Festuca altaica*	Tolerant to low nutrients, drought, high elevation and permafrost. Medium size bunchgrass with low seed yield, but spreads by rhizomes. Widespread, grows in open woods, alpine grasslands, tundra, at all elevations.
Northern bluegrass *Poa alpigena/Poa pratensis*	Tolerant to mildly acidic soils, low nutrients and permafrost. Grows in sandy areas along lakeshores and moist meadows.
Ticklegrass *Agrostis scabra*	Pioneering on disturbed gravelly sites such as roadsides and disturbed areas.
Purple reed-grass *Calamagrostis purpurascens*	Moist woods, meadows, wetlands, lakeshores and clearings; widespread across boreal region.
Slender wheatgrass *Agropyron trachycaulum*	Native of gravelly and river shores, cliffs and talus slopes.
Tufted Hair-grass *Deschampsia caespitosa*	It has a high tolerance of metal-contaminated soils and grows not only in nutrient-rich, poorly drained habitats, but also in well-drained, nutrient-poor soils.

DISCUSSION

The past anthropogenic disturbances have made it difficult to clearly define homogenous ecosystems, as the landscape is mix of different structural stages that occur in close proximity to each other, yet are not large enough to be separated into different polygons. Larger polygons have a higher degree of diversity as a larger area is likely to have more variation in microtopography which influences site growth conditions. The resolution and accuracy of the 1:40,000 aerial photographs were questionable for the detail required at a 1:12,500 scale map. There were areas difficult to decipher due to shadow cast, photo distortion and an inconsistency with the grey scale. Also, gradual change in slope and vegetation makes it difficult to determine a defining line between vegetative types. For these reasons there is inherent error in the placement of polygon boundaries, the polygon as presented on the ecosystem map should be considered at best approximates. Further ground surveys and ecosystem plots are needed to improve the accuracy of the GHEM.

Most of the study area is situated on North and North-easterly aspects, so the vegetative associations listed in Table 2 are reflective of cooler growth conditions. The aerial photography interpretation shows higher diversity of plant communities and geomorphology on the southern aspects. In future investigations, plots can be established on these southern aspects to complete the inventory of vegetation associations found in the local area. There is a wide spectrum of plant associations, one factor is the variation in topography, and the other factor is the numerous disturbances in the area from mining and human habitation over the last 100 years. The Galena Hill Ecosystem Map produce at this stage still requires more input and refinement and should be considered as a draft.

RECOMMENDATIONS

The following recommendations are meant to be incorporated into any future ecosystem mapping endeavours to increase the accuracy and usefulness of this product for reclamation and revegetation in the former UKHM site.

- More ecosystem plots need to be established in polygon types not yet visited, to ensure accuracy of ecosystem labeling;
- More ground plots, transects and visual checks needed to achieve accurate placement of polygon boundaries;
- Aerial photography will need to be updated preferably after the main terrestrial reclamation projects have been completed;
- Control plots that match the growth parameters (aspect, SMR, SNR, surficial geology) of the cover trials need to be established when test sites have been selected;
- Disturbed areas consisted of old mine works, main roads, gravel pits and urban development. These are identified on the ecosystem map as red coloured polygons. A few of these areas are worth further investigation as they are in primary and/or early secondary succession that provide templates for revegetation efforts and have plant species that are pioneers and heavy metal tolerant;
- An active weed monitoring/management program needs to be in place to prevent invasive species encroachment. Several invasive plant species have already been observed in the area. Areas of weed infestation can be shown on the GHEM and monitored;
- Data check and map review needs to be done in coordination with ELC coordinator and technical working group, as ecomapping standards and plant associations are evolving;
- Determine the usefulness of the GHEM in guiding and monitoring the progress of restoration projects as according to nine attributes as laid out in the SER Primer.

CONCLUSION

By integrating the aerial photo interpretation and vegetation survey information, an ecosystem map was produced. The map is the stratification of the landscape into polygons according to a combination of ecological features, primarily climate, terrain, soil, and vegetation.

The Galena Hill Ecosystem map is at a scale of 1:12,500, with 156 separate ecosystem polygons. It is meant to provide guidelines for reclamation and revegetation projects so these areas can eventually integrate into the surrounding landscape and be ecologically functioning. Polygons identified as in early succession can be seed resources for pioneering native plants and references for monitoring the trajectory of reclaimed areas. Certain vegetation associations are better suited for growing on shallow soils over colluvial surficial material, similar to engineered covers. These vegetation communities can be found easily on the ecosystem map and used as a guide. The ecosystem map can also be used as a land management tool and should be viewed in conjunction with planning, e.g., placement of roads, trenches, waste rock, tailings or any other activity that will involve the disturbance of natural areas.

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REMOTE SENSING IN VEGETATION MONITORING: MORE THAN JUST A PRETTY PICTURE

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ABSTRACT

Monitoring of reclaimed sites is a complex, interdisciplinary undertaking, especially in large, disturbed areas with difficult access. In that context, remote sensing is a unique and valuable tool that provides a synoptic view of an entire reclamation program and its progress over time, extending the more detailed but sparsely distributed *in situ* monitoring. Using remote sensing data, we are creating reclamation maps that provide easily understood information about a site's vegetation history, and whether or not it has reached and maintained biomass above the permit threshold for self-sustaining status. These maps are produced at various scales, are Geographic Information Systems (GIS) compatible, and often provide data for remote, inaccessible locations or for locations where historical data are missing. Reclamation maps are designed to help decision-makers focus remediation efforts on specific locations most needing it, rather than making unnecessary and potentially costly wholesale changes to entire sites. The maps are useful, not only in reclamation and multidisciplinary studies, but also in public demonstration of industry's successful reclamation practices. As climate in the north continues to change, and resource exploration and extraction activities increase, mapping the changing landscape becomes key for the conservation and sustainable management of resources. In this paper, we present two examples of long-term remote sensing monitoring at reclaimed mine sites in British Columbia and one of long-term vegetation changes in the Northwest Territories. Other applications of remote sensing are also discussed, such as the generation of habitat maps for wetland monitoring at reclaimed tailings ponds in support of wildlife habitat or biodiversity studies. Examples of field applications of remote sensing will be presented in a companion poster entitled "Practical field uses of remote sensing."

Key Words: Mine Reclamation, NDVI, Change Detection, Vegetation Trends, Habitat Mapping.

INTRODUCTION

Reclamation is a central part of the decommissioning of mining operations, and requires monitoring of re-vegetated areas to assess the progress toward the end land use objectives, as well as to determine whether individual sites have achieved a self-sustaining state. Traditional *in situ* monitoring programs provide very detailed information on selected sampling sites, but ground-based surveying of entire reclaimed areas and hard-to-reach sites can prove to be a difficult, many times impossible, task.

Remote sensing, whether from aircraft or satellites, is a practical tool that provides continuous maps of vegetation changes over time for entire mine sites, supplementing the more detailed but less synoptic ground biological surveys. It provides a means to focus remediation efforts on specific locations that need it most, instead of making wholesale changes to entire sites (Borstad et al. 2005; Martínez et al. 2012; Richards et al. 2003), and is also a valuable tool for monitoring and quantifying a wide array of biophysical changes attributed to climate change in northern ecoregions (Borstad et al. 2008; Laidler et al. 2008).

METHODS

Remote Sensing Data

The high spatial resolution, together with the flexibility of spectral configuration and acquisition timing characteristic of airborne systems, make them ideal for reclamation monitoring (Borstad et al. 2005). Recently satellites have seen improvements in spatial resolution in particular that make them a less expensive alternative; however this reduction in cost comes at a significantly increased risk, since satellite surveys are more affected by cloud than those from aircraft that can be more readily adapted to avoid or operate under cloud (Borstad et al. 2005).

In this paper we present examples of data obtained with the Compact Airborne Spectrographic Imager (CASI) configured to acquire imagery in 9 spectral bands at 2.5 m spatial resolution (Borstad et al. 2005; Brown et al. 2006; Martínez et al. 2011; Richards et al. 2003), as well as from satellites such as Landsat, and QuickBird2 (QB2) and WorldView-2 (WV2) (Digital Globe 2012). The free data from the Landsat series, launched in 1972 and extended with the successful launch of Landsat 8 in 2013, provides consistent global coverage (GLOVIS 2013). WV2 and QB2, launched in 2009 and 2011 respectively, provide higher spatial and spectral resolution than the Landsat series; however they are not free, and must be specifically tasked to acquire imagery over any particular target. Their short historical catalogue makes them less useful for long-term analyses, but their higher spatial resolution makes them better suited to study smaller areas in more detail.

Land Vegetation Indices

For land vegetation studies we use two vegetation indices derived from reflectance data for both airborne and satellite imagery (Table 1). The Normalized Difference Vegetation Index (NDVI) is commonly used in remote sensing as an index that provides a measure of green vegetative cover or biomass (Peñuelas and Filella 1998; Tucker 1979). Our 'Normalized Yellow Index' (NYI) provides an index of desiccation and/or senescence (Borstad Associates 2006).

Table 6. Calculation of vegetation indices and interpretation.

Index	Formula	Interpretation
NDVI	(Infrared – Red)/(Infrared + Red)	< 0 unvegetated, > 0 vegetated (1 maximum)
NYI	(Green – Red) / (Green + Red)	< 0 dry or senescent, > 0 green sparse to dense (1 maximum)

NDVI is a measure of plant chlorophyll. Red reflectance (near 660 nm) from healthy green vegetation is low because of light absorption by photosynthetic pigments, mainly chlorophylls, whereas a plant's spongy mesophyll leaf structure creates considerable reflectance in the near infrared region of the spectrum. NDVI can be calibrated to apparent biomass, as shown in Figure 1, but NDVI plateaus at high biomass values (Figure 6), and it can also be affected by disease, flowering, and water availability (Carter and Knapp 2001; Wang et al. 2001; Yin and Williams 1997). We developed the Normalized Yellow Index (NYI) to assist differentiating the low NDVI values caused by chlorophyll degradation from those caused by moderate or sparsely vegetated green areas (Borstad Associates 2006). As plants lose chlorophyll, red reflectance (near 660 nm) values increase, thus NYI compares spectral bands in the green (560 nm) and the red wavelengths, having positive values for dense green vegetation, and negative values for yellowish vegetation.

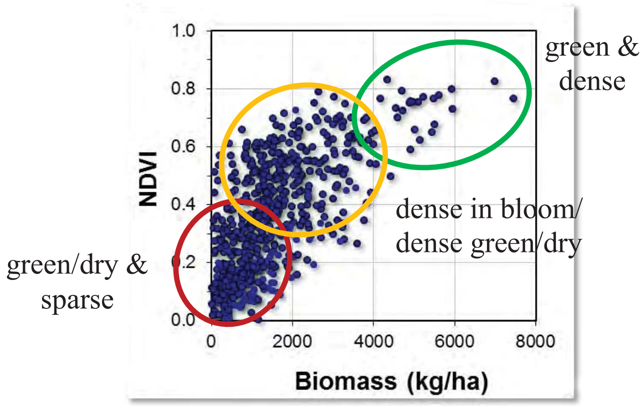

Figure 6. NDVI calibration to apparent biomass.

<u>Multi-temporal Analysis</u>

To assess vegetation changes over time, we build time series datasets using the annual NDVI images, and classify them using an unsupervised algorithm (ISODATA), which groups similar NDVI histories into up to 255 statistical classes. Pixels with similar trends are then grouped manually (Borstad et al. 2009; Martinez et al. 2012) to create maps of reclamation status (see Figure 2). As described above, NYI data are used to assist to interpret low NDVI values, as are photographs and visual observations when available.

Another way that we use NDVI time series is to assess long-term trends in vegetation, such as those associated with climate change. Changes of this nature are evaluated using regressions of NDVI versus year (see Figure 3). The images of r^2 and slope allow us to map the rate and direction of change and its level of statistical significance. Maps of regression slope clearly delimit regions of positive and negative change as well as their magnitude, facilitating interpretation of the underlying factors, as well as permitting the calculation of such statistics as area of habitat loss. It is important in any vegetation monitoring program to understand the long-term context within which reclamation efforts are conducted.

<u>Mapping Aquatic Habitat</u>

Different habitat types can be characterized from remote sensing imagery based on the optical properties of the vegetation types present (communities or species), as well as the non-biological components. These optical properties need to be considered when selecting the remote sensing data type, processes and techniques. Aquatic habitat mapping presents a special case, as the vegetation has unique spectral properties due to the modulating effect of water. For this reason airborne sensors, with their flexible spectral configurations, are often used for aquatic applications. Specific bandsets can be used to extract information for aquatic vegetation and water quality parameters such as turbidity, and phytoplankton blooms.

The techniques used for aquatic mapping differ somewhat from terrestrial applications because of the unique spectral properties of the vegetation. For example, instead of NDVI used for terrestrial vegetation, for aquatic vegetation we use an index referred to as Shifted Red Peak Height (SRPH). Like NDVI, this index measures the height of the near infrared reflectance near 700 nm that occurs in spectra of all multicellular plants. In aquatic environments, where longer wavelengths are strongly attenuated by water, this near infrared "plateau" becomes a "red peak" when vegetation is covered by water, thus providing an unambiguous indication of aquatic vegetation (see Figure 4A2).

EXAMPLES

Long-Term Changes at Reclaimed Mine Sites

We have been monitoring reclaimed areas at a large copper mine using aerial remote sensing since 2001. The time series has permitted a variety of analyses that are being used in the management of the reclamation program. The trends in annual NDVI (Figure 2) and apparent biomass demonstrate the history of the reclamation at each site, and more importantly, clearly delimit the extents of areas that have reached a biomass above 1500 kg/ha ('Rapid growth'; green in the reclamation map), the accepted reclamation threshold in this case. The analyses also show which areas considered successfully reclaimed are subject to desiccation (shown in yellow) and those requiring additional intervention ('Limited cover' shown in red).

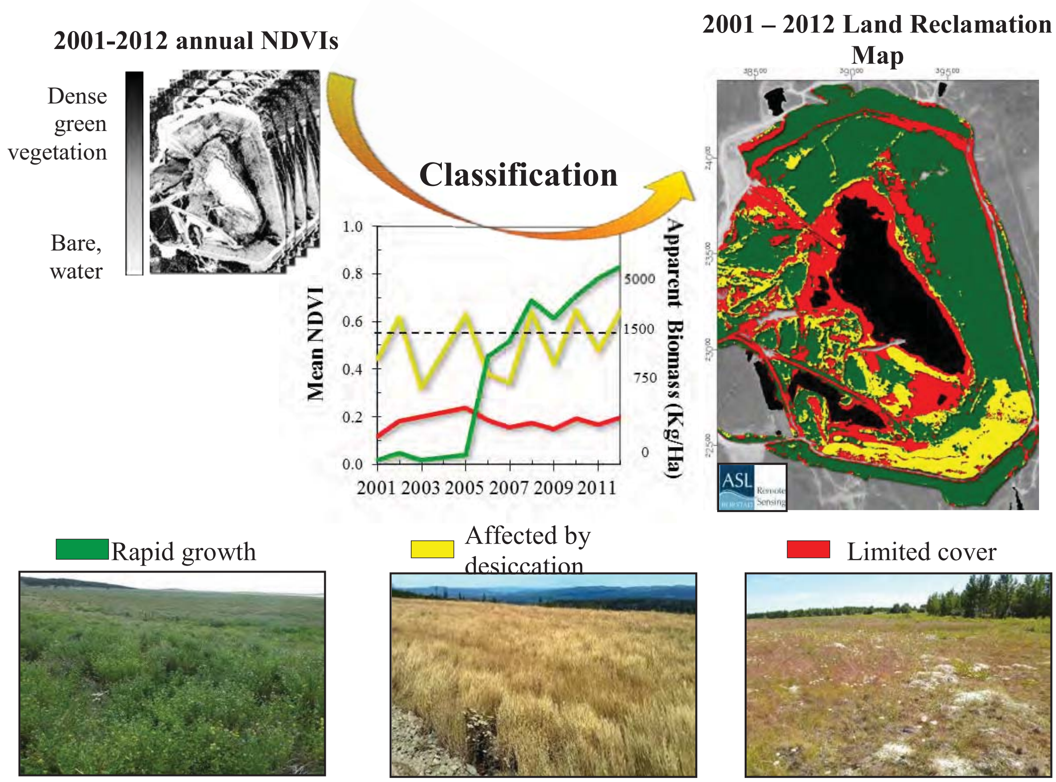

Figure 7. Reclamation map and 2001-2012 vegetation trends at a reclaimed mine site.

Vegetation Trends at Anderson River Delta, NWT

The primary objectives of this study (Figure 3A) were to quantify reports of habitat loss in the Anderson River Delta and identify areas that had been impacted. Analysis of four large regions of interest (ROIs) showed that the largest reductions in vegetation cover occurred in the Outer Islands between 1972 and 2003 (Figure 3B), with an NDVI decline of 38% during the study period, corresponding primarily to areas most heavily used by nesting Lesser Snow Geese and Black Brant (Armstrong 1995; Kerbes et al. 1999).

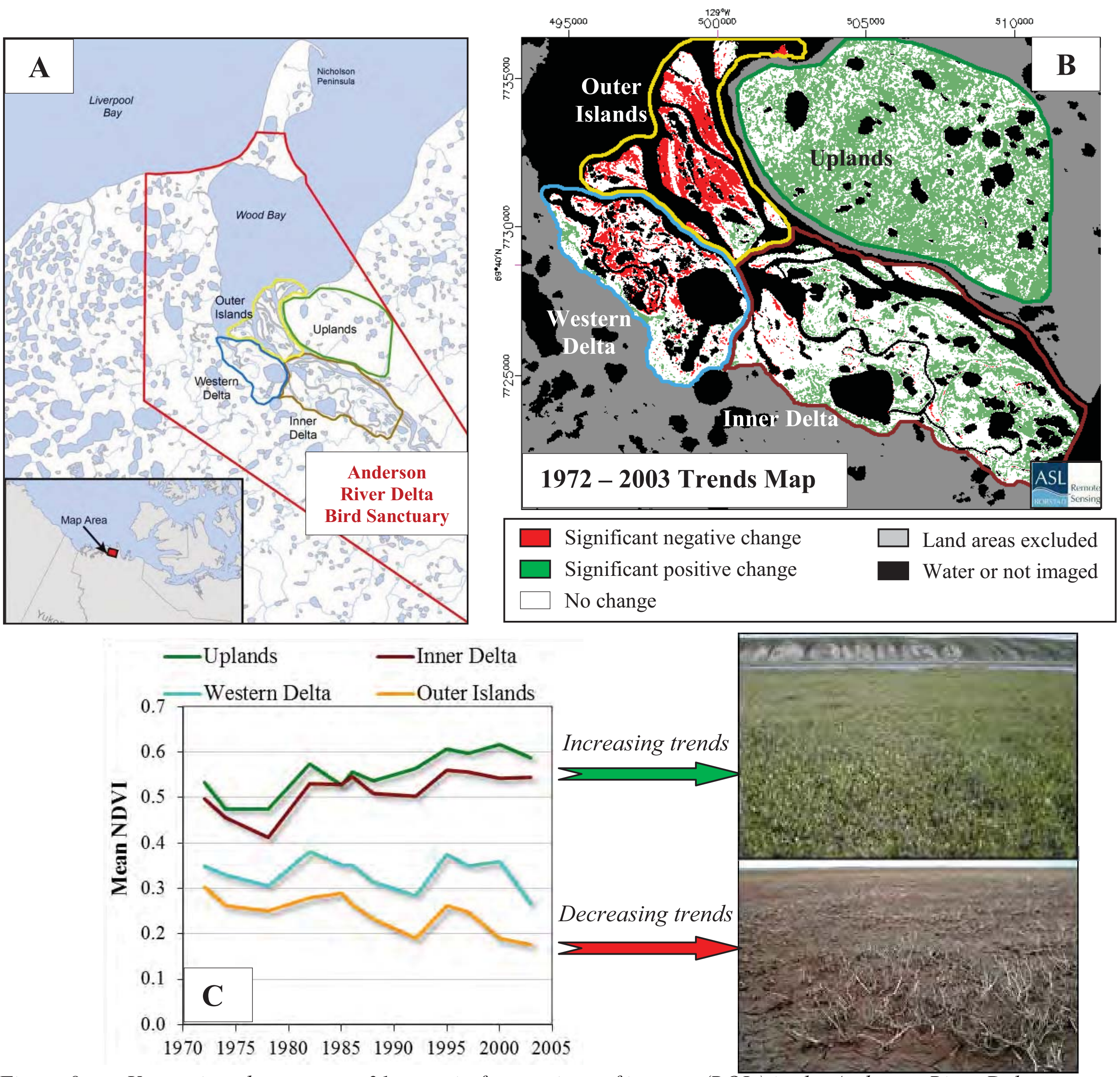

Figure 8. Vegetation changes over 31 years in four regions of interest (ROIs) in the Anderson River Delta.

The Western Delta ROI showed a 12% decline in vegetation over the study period, while further inland, the Inner Delta showed little vegetation loss. The Uplands had a statistically significant increase in NDVI over time, which agreed well with results from other studies in North America and Eurasia for the same time period (Jia et al. 2003; Jiang et al. 2011; Verbyla 2008; Walker et al. 2012). The changes seen in the satellite time series were confirmed by qualitative reports (Armstrong 1995; Barry 1967).

Mapping Wetland Habitat

Tailings ponds can be converted to functional wetlands once water levels are stabilized, but they are often treacherous to traverse. This severely limits ground-based sampling, and presents a hurdle to clear understanding of the ecology and evolution of the reclamation. For such water bodies, remote sensing provides an alternative means of detecting and monitoring submerged vegetation. The pond shown in Figure 4A has very soft tailings, and had developed channels where streams of bubbles created holes 4 to 10 m in diameter and several metres deep (photo shown in Figure 4A1, taken on the northwest shore), preventing access even from small boats. The airborne imagery (Figure 4A) revealed large amounts of submerged aquatic vegetation in the center of the pond. The discovery of *Ruppia maritima,* was a complete surprise to the limnologists (Borstad et al. 2005). As well as detecting *Ruppia*, the maps of aquatic plant distribution helped to quantify the success of the aquatic reclamation program. Twelve milfoil and aquatic buttercup plant sandwiches had been installed 1996, and in seven years the plants had expanded to cover 13 hectares in low density weed beds, with occasional high density milfoil patches (Larrat Aquatic 2003).

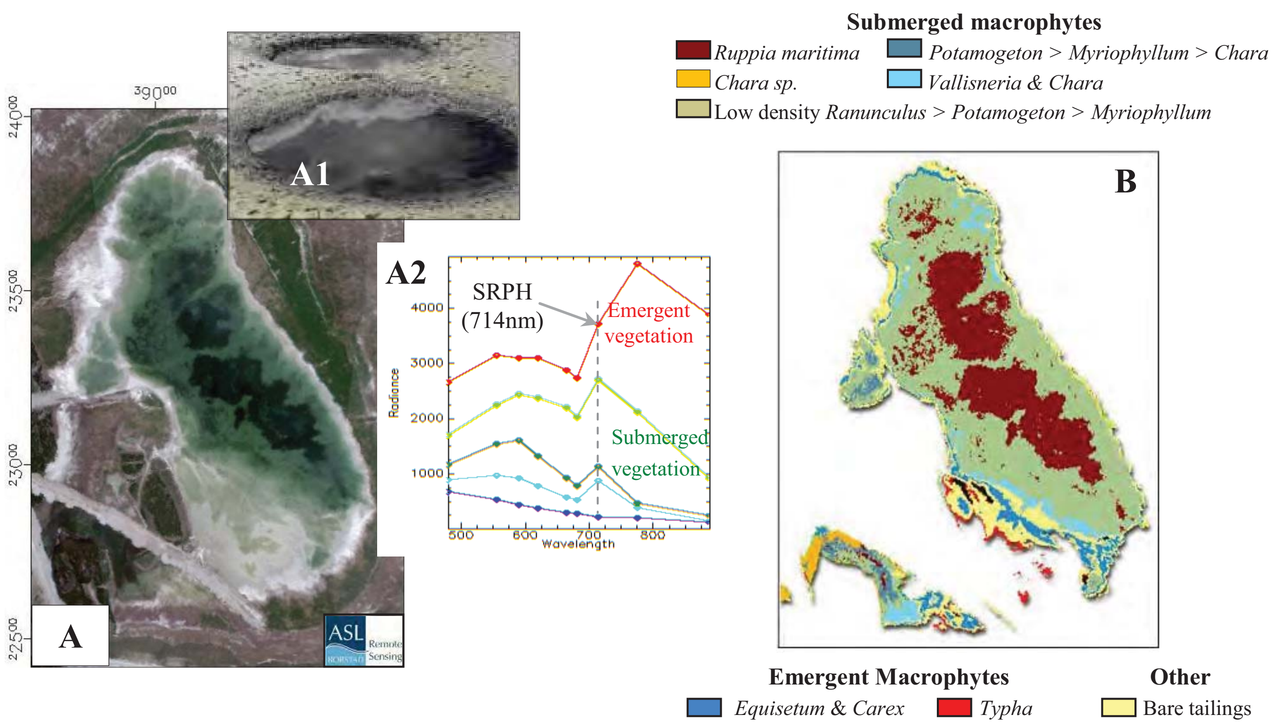

Figure 9. Aquatic plant distribution in a reclaimed tailings pond derived from multispectral CASI imagery

CONCLUSIONS

Remote sensing maps are more than just pretty pictures; they are valuable tools for monitoring and quantifying biophysical changes (Borstad et al. 2008; Lantuit and Polland 2008; Verbyla 2008; Walker et al. 2005). Aerial remote sensing surveys can be optimized using specific spectral bands to acquire information for vegetation changes in a variety of habitats. Long-term series imagery from aerial or satellite sensors support many kinds of land and water quality monitoring studies, and help to understand where and what changes have taken place, and to quantify the rate of change. Recent cost reductions of remote sensing data have also made this tool more accessible to non-profit groups and other small, independent interests.

ACKOWLEDGEMENTS

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KEY FACTORS IN DEVELOPING AND IMPLEMENTING A SUCCESSFUL MINE RECLAMATION PLAN – DENISON SITES 20 YEARS AFTER CLOSURE

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ABSTRACT

Developing and implementing a successful reclamation plan is dependent on a number of key factors, including defining clear objectives, assessing available options to meet those objectives, gathering adequate and accurate data for evaluating options, revising the selected reclamation plan during implementation if required, and developing a focused and integrated monitoring network to track performance and effectiveness of the plan over time. Successful reclamation can be defined as ensuring effective protection of health, safety, environment and community, with measures of effectiveness driven by regulatory requirements, corporate policies and stakeholder expectations. A successful reclamation plan must also be flexible, adaptive and cost effective in both the short and long term. This paper illustrates how these key factors were and continue to be essential in the success of the reclamation plans designed and implemented at the Denison Mines Inc. uranium mine sites in Elliot Lake, Ontario, over the past 20 years.

Key Words: Closure, Decommissioning, Rehabilitation, Uranium, Elliot Lake.

INTRODUCTION

Mine closure planning and implementation of reclamation measures have evolved over the past twenty to thirty years as "planning for closure" has transitioned from a concept in the early 1980s (Culver et al. 1982) to standard practice in modern mine development (McKenna 2011). As closure and reclamation considerations have become routine aspects of mine development from the earliest stages of the mining life cycle, mines that were previously developed in the absence of closure plan legislation have also designed closure plans for current operations, as well as planned expansions, with many initiating progressive reclamation while still in operation. Historic mines that developed, operated and closed or were abandoned prior to the legislated requirement for closure planning face greater challenges in designing and implementing successful reclamation plans as generally no measures were taken to minimize the operational footprint and prevent and/or mitigate impacts to surrounding land, water and air. Often these sites have limited funds available for closure plan design and implementation of reclamation activities, as they are no longer generating profits for the mining company or the site has been abandoned and reclamation efforts have become the responsibility of the government.

Regardless of the stage of the mining cycle during which the closure planning process is initiated, there are key factors to consider in designing and implementing a successful reclamation plan. These include defining clear reclamation objectives, assessing available options to meet those objectives, gathering adequate and accurate data for evaluating options, having a flexible and adaptive plan that can be revised during implementation if required, and developing a focused and integrated monitoring network to track performance and effectiveness of the plan over time. The stage of the mining cycle during which this process is initiated will; however, determine the reclamation options that are available, as well as the time frame and cost of achieving success. Measurements of successful mine reclamation are also different for mines developed before and after reclamation legislation was adopted; therefore there is a need to develop site-specific performance indicators based on closure plan objectives established through collaboration with regulators, local community stakeholders and affected First Nations. This will allow for a balance of expectations and viewpoints on what can and cannot be achieved during the stages of mine closure. Although the closure plan objectives and performance indicators may be specific to each mine site, there is general consensus that the overall goals of mine reclamation are to restore mine sites and affected areas to self-sustaining ecosystems that are compatible with a healthy environment and human activities.

HISTORY OF DENISON SITES

The discovery of uranium in the region of Elliot Lake, Ontario during the post-World War II era led to the development of 12 mines between the years 1955 and 1958 (Figure 1), three of which, Denison, Stanrock and Can-Met, are owned by Denison Mines Inc. Most of the uranium was produced for contracts with the United States Atomic Energy Commission during the late 1950s and early 1960s; however, following the cancellation of these contracts, most mines in the area closed. Denison mines continued to operate, supplying uranium to electrical power generating utilities in Japan and Ontario from the early 1970s to the early 1990s. By 1996, due to diminishing ore grades and high production costs, all the Elliot Lake uranium mines had ceased operations and begun the decommissioning process, which included design and implementation of closure plans for each of the facilities.

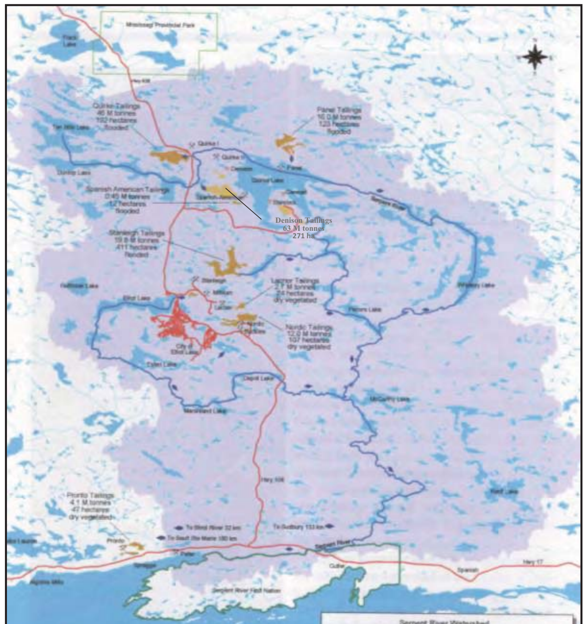

Figure 1. Locations of Former Mine Sites in the Serpent River Watershed

Denison Property

Denison Mine, located 16 km north of the City of Elliot Lake, consisted of a conventional mill and underground mine that operated between 1957 and 1992. Over this time, Denison generated a total of 63 million tonnes of uranium tailings that was deposited into two bedrock lined basins, which are now referred to as tailings management areas (TMA-1 and TMA-2).

Stanrock and Can-Met Properties

The Stanrock property, located 21 km northeast of the City of Elliot Lake, was also developed in the 1950s and consisted of a mine, mill and TMA. The underground mine used conventional milling to recover uranium from 1958 to 1964. From 1964 to 1970 and from the late 1970s to 1983, a mine water recovery operation was used to extract uranium from water that was allowed to flood the underground workings. During the latter period, the mine was rehabilitated and a small amount of ore was hoisted to surface and hauled to the Denison mill for processing.

The Can-Met property is located adjacent to the Stanrock property and was operated (both a mine and mill) from 1957 until 1960. Tailings from both the Stanrock and Can-Met facilities were discharged into the natural basin of a small lake located immediately south of both mines. This became the Stanrock tailings basin (TMA-3), which contains approximately 5.7 million tonnes of tailings.

PLANNING AND DESIGN OF CLOSURE PLANS

Prior to closure, the mines operated in accordance with stringent licensing requirements by the Atomic Energy Control Board of Canada (AECB), now known as the Canadian Nuclear Safety Commission (CNSC), in consultation with other Federal and Provincial regulatory agencies. When closure of the mines was announced in the early 1990's, the Canadian regulatory framework governing mine closures was still in the process of being finalized, and the decommissioning process for the uranium mines was the largest privately-owned, multi-mine closure program seen up to that time in Canada (Payne et al. 2004). Over the course of three years, extensive Environmental Hearings were conducted to evaluate the proposed closure plans and potential environmental and socio-economic impacts. The approvals process was facilitated by the establishment of a Joint Review Group (JRG) comprised of more than 15 Federal and Provincial regulatory agencies with the CNSC as the lead regulatory agency. This collaborative effort allowed for coherent responses and decisions on all issues during the hearing process.

Both companies (Denison Mines Inc. and Rio Algom Ltd.) involved in the decommissioning process shared common overarching objectives for the reclamation of all sites, which were to protect public health and safety, minimize long-term environmental impacts from the decommissioned facilities and ensure the decommissioned sites would not prejudice the survival of affected communities after the mine closures. Fortunately, since the 1970s, the physical, chemical and radioactive characteristics of the Elliot Lake tailings had been studied by both companies in conjunction with government agencies such as the Canada Centre for Mineral and Energy Technology (CANMET) and the Mine Environmental Neutral Drainage Program (MEND), as well as various university research groups. This work provided the companies with a significant database of information that was essential in planning, design and implementation of the closure plans (Payne et al. 2004).

The Elliot Lake uranium deposits are classified as low grade (< 0.1% U_3O_8). Although the radioactive characteristics of the tailings are a concern, detailed radiological modeling conducted as part of the National Uranium Tailings Program has confirmed that exposures and risks to the public from the Denison facilities are very low (Denison Mines Ltd. 1995). However, in addition to radioactive elements, the tailings contain an average pyrite content of 6%. When pyrite is exposed to air it reacts with oxygen to produce sulphuric acid, which in turn mobilizes metals and radioactive isotopes having a detrimental effect on receiving waters if not treated. All of the water from the TMAs drains into a network of rivers and lakes that eventually discharge into Lake Huron. Acid generation was deemed to be the most significant issue for consideration during the development of decommissioning options, although long-term physical stability of the sites and achieving radiation doses as low as reasonably achievable were also important (Denison Mines Ltd. 1995).

A number of closure options were considered and evaluated against the established design criteria and objectives prior to selecting the final reclamation plans for each site (Table 1). Site specific probabilistic risk assessments were also completed to examine the likelihood and consequences of failure resulting from discrete disruptive events such as earthquakes, floods or droughts having return periods of up to 1 in 10,000 years. In addition, the risk assessments considered the expected costs of environmental releases resulting from discrete events together with the cost of on-going care and maintenance (Belore et al. 1999; Kam et al. 1999a,b; Welch et al. 1996). The ability of each option to meet the specified design objectives and criteria for each TMA was assessed on the basis of the option's potential environmental, economic, and social impacts. The evaluation also took into account the guidelines for environmental assessment laid out by the Federal Environmental Assessment Review Organization (FEARO) Panel and the recommendations made at public hearings. Following thorough evaluation of all the options and consideration of the risk assessment results, the Panel concluded that the final approved reclamation plans were designed and constructed with the best available technology to address the problem of perpetual containment (CEAA 1996). The decommissioning plans that were selected and implemented at the Denison and Stanrock sites are described in the following sections.

Table 1. Options Considered for Decommissioning of TMAs (Denison Mines Ltd. 1995)

Option	**Description**
Base Case	• long-term collection and treatment of contaminated seepage and run-off water
Water Cover	• use of natural precipitation and run-off water to flood the exposed tailings • water retaining perimeter dams required
Soil Cover	• use of non-acid generating soil material to cover the exposed tailings and encourage a raise in the water table elevation
Complete Removal	• hydraulic relocation of tailings • underground disposal • deep lake (Quirke Lake) disposal
In Situ (Stanrock only)	• a modification to the base case • construct new, low permeability dams to establish and maintain an elevated water table near the tailings surface • similar advantages to a water cover

IMPLEMENTATION OF CLOSURE PLANS

Denison Property

A water cover was selected as the most suitable option for decommissioning of TMA-1 and TMA-2 at the Denison site (Figure 2) based on the following reasons:

- elimination or near elimination of acid generation,
- elimination of airborne releases (i.e., dusting),
- geography of the area is well-suited to flooding,
- risk of structural failure is very remote,

- flooded tailings would result in no requirement for new lands,
- limited burden on future generations,
- basins are returned to a use very similar to what they were like pre-mine development,
- radiation dose to receptors will be much less than anticipated limits, and
- water cover is an effective barrier to intrusion by man.

The decommissioning of the Denison site was completed in 1996 (SENES 1998) using the water cover option at both TMA-1 and TMA-2; however, the process of implementation was unique to each TMA. Constructing large dams would not have been economical in TMA-2 nor sustainable within a small watershed. As a result, tailings were removed from TMA-2 to lower the elevation and meet the requirements of a water cover. The tailings were relocated to two areas (TMA-1 and the underground workings) using hydraulic monitors and slurry pumping. The appearances of TMA-1 and TMA-2 before and after decommissioning are illustrated by pairs of photographs (Figure 3).

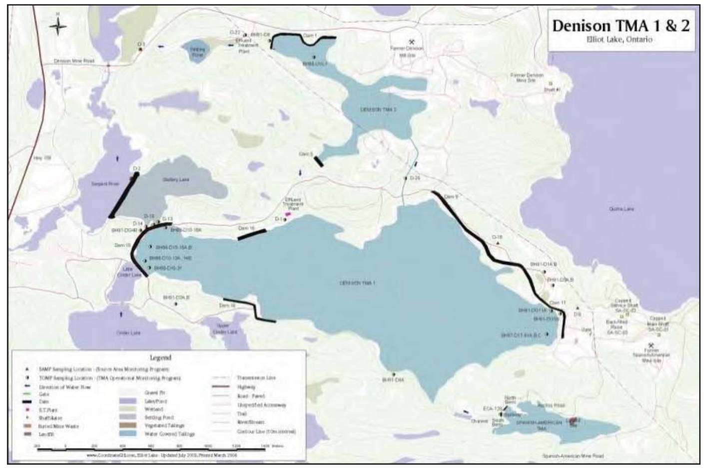

Figure 2. Denison Site Map (TMA-1 and TMA-2)

All surface facilities at the Denison site were demolished and the site was revegetated, with the exception of the Executive Lodge, the two effluent treatment plants, and roadways required for long-term access. Prior to demolition, equipment was removed for sale when possible, and scrap steel was removed for recycling. Asbestos was removed and placed into the Denison landfill located at the northeast end of TMA-2, which was subsequently graded, capped with a till cover and revegetated. All other hazardous materials, such as laboratory chemicals, were disposed of by a firm licensed to undertake that work.

Stanrock Property

The "*In Situ* Management Plan" was chosen as the preferred option for decommissioning the Stanrock TMA (Figure 4). The reasons for selecting this option included:

- the majority of the acid generating tailings would be stored below the water table,
- airborne dust and radon releases would be reduced,
- surface water quality would be within accepted loadings, no new areas were required,
- the basin would be returned to conditions similar to those of pre-mine development,
- cost effective option that addressed all concerns within the existing TMA, and
- the future risks of major system failures were minimal.

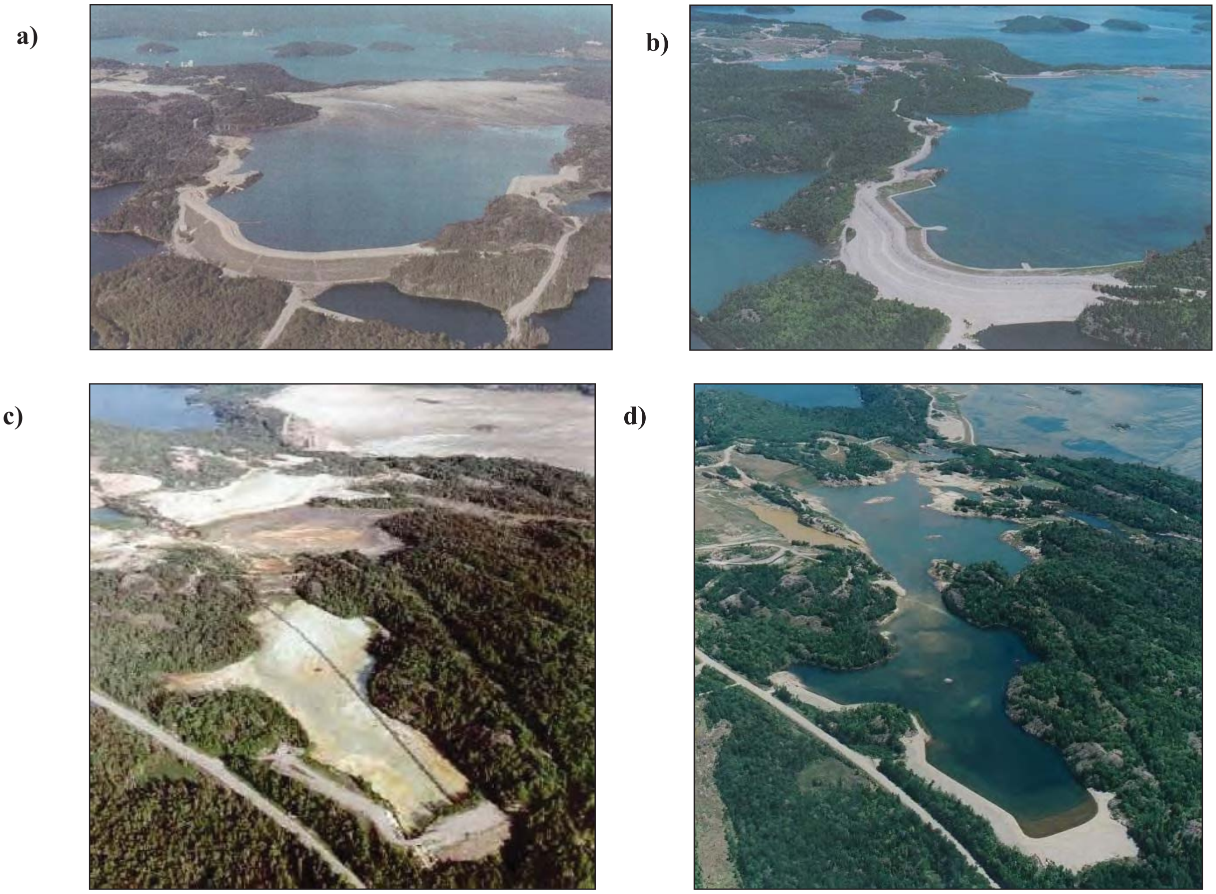

Figure 3. *Denison Before and After Decommissioning: TMA-1 in (a) 1992 and (b) 1998; TMA-2 in (c) 1992 and (d) 1998*

The decommissioning of the Stanrock and Can-Met sites were completed between 1992 and 2000 (Ludgate et al. 2005). New containment dams were constructed downstream of the existing structures to raise the water table. The dams incorporated a water retaining core of compacted till with a bedrock foundation beneath the dam core that was grouted to minimize seepage. Filter drains were provided to prevent internal erosion and a build-up of porewater against the dam. Although there is a small headpond, water is generally not impounded in the TMA, but drains from the surface and passes through a spillway in Dam A which flows, along with seepage collected from all of the dams, to a treatment plant southwest of the plant. Although building the dams raised the water table, it could not be raised high enough to completely cover all of the tailings, as anticipated in the planning process (SENES 1997). Unoxidized tailings above the water table still react to form additional

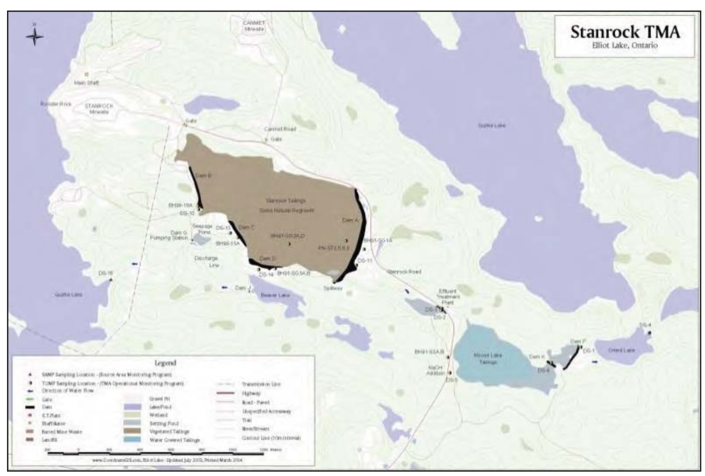

Figure 4. *Stanrock Site Map*

acid, which is flushed out along with the acid already present. Modelling has shown that it may take up to 50 years to deplete the entire contained acid inventory.

The demolition of the Stanrock surface facilities was carried out in 1992 and 1993 (SENES 1997). Facilities that were demolished included: the mill complex, head frames, mine offices, workshops, warehouse, mine hoist rooms, pumphouses and sewage plant. The demolition excluded some power lines, an electrical substation, the Effluent Treatment Plant and the Dam G pumps, trestle and pipeline, as these services are required for ongoing treatment of acid water. Prior to demolition, equipment and material were sold for reuse or recycling when possible and all hazardous materials were removed and disposed of appropriately. The underground crushing and grinding plant, located 200 ft below the surface, was used as a disposal site for solid waste and was completely filled with demolition material.

LONG-TERM MONITORING, CARE AND MAINTENANCE

General Care and Maintenance Programs

Each of the TMAs were upgraded to ensure that they meet current standards for the protection of health, safety and the environment. There are a number of programs in place to ensure the continued safe operation, care and maintenance of these decommissioned sites. The general programs include: Site Security, Radiation Protection Programs; Health and Safety Programs; Inspection Programs; Tailings Management Operating Programs; Monitoring Programs; Reporting Programs; and Emergency and Contingency Response Programs.

In addition to routine monitoring of all site infrastructure by care and maintenance staff, annual inspections are also completed by professional engineers in order to verify that the facility is performing as designed; confirm that routine care and maintenance activities are being completed as required; and identify any potential areas of concern that may require remedial action. To date, there have been no issues at any of the structures and the TMAs and basins are operating as designed.

Environmental Monitoring Programs

Denison has undertaken extensive environmental monitoring programs since the early 1970s. At the time of closure, each mine had its own environmental monitoring program conducted under an operating licence from the AECB (now CNSC) or a Certificate of Approval (now Environmental Compliance Approval) from the Ontario Ministry of Environment (MOE). In 1997, both Denison Mines Inc. and Rio Algom Ltd. reviewed their existing monitoring requirements in terms of their relevance to current environmental data and predictions of changing conditions associated with decommissioning as outlined in the Environmental Impact Statement (Denison Mines Ltd. 1995). The evaluation resulted in the development of a focused and integrated monitoring network to track the performance and effectiveness of the reclamation plans over time. The integrated strategy is subdivided into four programs that supersede the monitoring requirements listed in the licenses and Certificates of Approval for the TMAs. The monitoring programs include:

- Serpent River Watershed Monitoring Program (SRWMP) – replaced the various mine-specific receiving environment monitoring programs with one comprehensive program focused on water and sediment quality, benthic invertebrate and fish communities, as well as radiation and metal doses to humans and wildlife (Beak 1999; Minnow and Beak 2001a, Minnow 2002a).

- In-Basin Monitoring Program (IBMP) – companion program to the SRWMP to assess the health risks to biota potentially feeding at each of the aquatic and vegetated management areas (Beak 1999; Minnow and Beak 2001b, Minnow 2002a)
- Source Area Monitoring Program (SAMP) – monitors the nature and quantities of contaminant releases to the watershed (Minnow 2002b)
- TMA Operational Monitoring Program (TOMP) – generates the data used to track TMA performance and supports decisions regarding the management and discharge compliance of the TMAs (Minnow 2002c)

Each program was designed to directly complement the other three programs in terms of monitoring locations, parameters, and sampling frequency, and thus ensure that the overall monitoring framework is comprehensive and interpretable. These programs are objective-driven and allow for modifications to be made over time in response to changes in the conditions at the sites.

Source Areas

Data collected under TOMP and SAMP demonstrate the effectiveness of decommissioning activities at both the Denison and Stanrock sites. Since decommissioning, there have been substantial reductions in the amounts of reagents used at the effluent treatment plants or directly within the TMAs. For example, at the Denison site, the liming of the surficial tailings and the use of water cover has resulted in circum-neutral pH in both TMA-1 and TMA-2. As a result, the need for caustic treatment has been modest and generally only required to treat acidic rain and meltwater. (Figure 5). Decommissioning activities were also effective in reducing the concentrations of heavy metals, and sulphates in the TMAs, as indicated by the trend of decreasing concentrations present in pre-treated water collected from TMA-1 between 1992 and 2012 (Figure 6). However, with the reduction in sulphate concentrations, there has been a slight increase in radium concentrations. This observation is not surprising based on initial modeling and additional studies have shown increased mobility and release of radium in pyritic uranium tailings when sulphate ion mobility control has been depleted for both on-land and underwater disposal scenarios (Davé 1999a,b; Davé et al. 2002). The decommissioning of TMA-1 and TMA-2 has been successful in addressing the primary concern of acid generation and has resulted in improved water quality; and on-

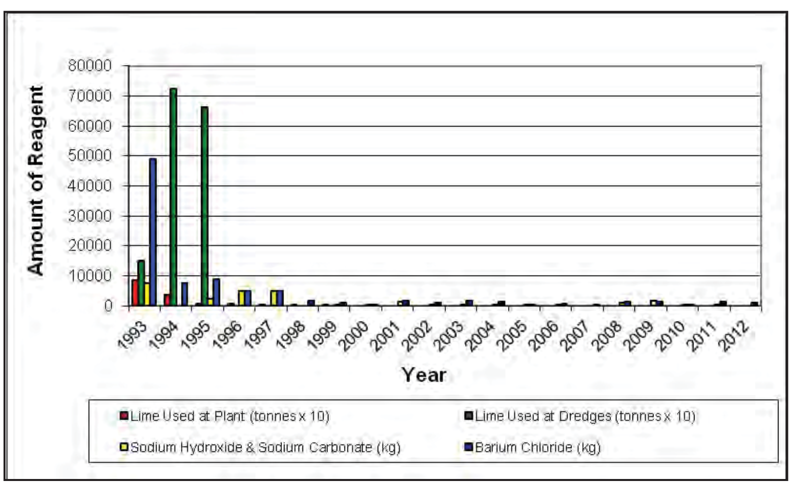

Figure 5. Annual Reagent Consumption at Denison TMA-1

going and future management of the TMAs is focused on the possibility of increased radium mobility with the depletion of sulphate over time within the basins (Ramsay et al. 2013).

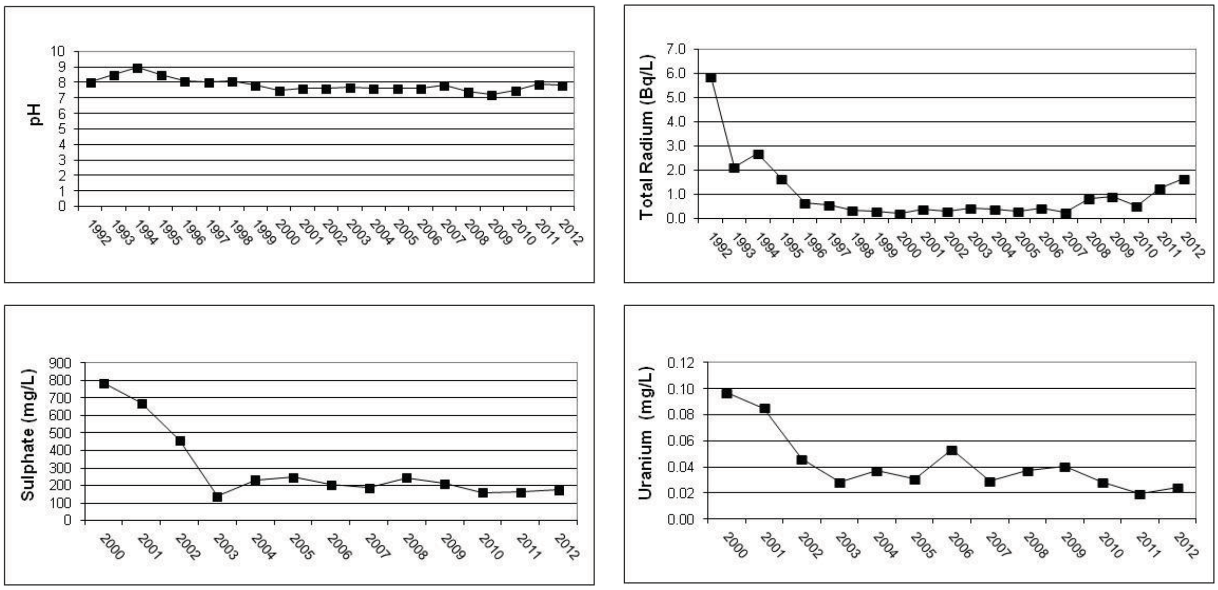

Figure 6. Annual Average Water Quality in TMA-1 Prior to Treatment

At Stanrock TMA-3 there have also been great improvements in the reagent requirements for water treatment as result of decommissioning activities. Prior to the construction of the new dams, the TMA surface run-off and the combined seepages required large quantities of lime to neutralize the water before it was released. Since dam construction and decommissioning, the consumption of neutralizing agents and barium chloride has decreased substantially (Figure 7) due to decreased tailings water acidity and radium concentrations in the influent water.

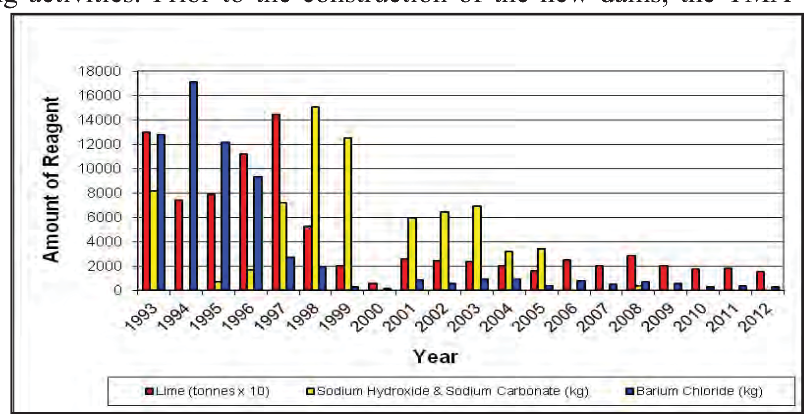

Figure 7. Annual Reagent Consumption at TMA-3

<u>Receiving Waters</u>

The SRWMP is conducted on a five year cycle with three cycles of the program having been completed to date (1999, 2004 and 2009). Water quality in receiving waters is better than criteria established for the protection of aquatic life and in most cases, the concentrations of “mine indicator” parameters have improved or remained consistent over time. Sediment quality is below upper limits established for the protection of aquatic life, but remains above lower limits and background concentrations. Benthic invertebrate density and community structure in exposure lakes are different from those in reference lakes, but the magnitude and number of differences are decreasing over time indicating a gradual recovery. Public radiation doses are below Health Canada Guidelines of 0.3 mSv/y (Minnow 2011).

Public Information Program

Denison Mines Inc., in association with Rio Algom Ltd., has developed an extensive Public Information Program to ensure effective communication of site activities with local residents. A more intensive program was deemed necessary because of the changing demographics of the community. Elliot Lake is no longer a mining town, but it has become a leading provider of Retirement Living services. At mine closure, both Denison and Rio Algom were instrumental in establishing the Elliot Lake Retirement Living Program. A significant part of the population is now elderly and from outside of the Elliot Lake area.

The public information program includes; annual site tours (in conjunction with the annual Elliot Lake Uranium Heritage Days held in early July), public presentations, public awareness meetings, development of an information web site, newsletters and the release of reports. An Elliot Lake Rehabilitation Information Web Site link (www.denisonenvironmental.com) was developed and provides both historical information and current monitoring results. Bi-annual presentations are made to City Council and public meetings are held periodically to inform the public of important events. An annual newsletter is distributed through the local paper and is also available on the web site. All significant reports are provided to the City and are available to the public at the local library.

CONCLUSIONS

Despite the significant challenges faced by Denison Mines Inc. when the mines ceased operations in the early 1990s, the decommissioning process in Elliot Lake has been successful. Reclamation objectives were established through collaboration with regulators, local communities, and affected First Nations. Options available to meet the objectives were determined by drawing upon an extensive database of information that was established in conjunction with government and university research groups. Reclamation options were evaluated against design criteria and objectives, and risk assessments were completed with the final reclamation plans approved by the FEARO Panel (CEAA 1996). During implementation of the reclamation plans, a focused and integrated monitoring network was developed, which allows tracking of the performance and effectiveness of the plans over time. This monitoring network also allows for adaptive plans to be created and implemented if required to ensure the objectives of the reclamation plans continue to be met.

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BIOCHAR APPLICATION FOR REVEGETATION PURPOSES IN NORTHERN SASKATCHEWAN

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ABSTRACT

Our research was focused on biochar application for revegetation purposes under northern Saskatchewan conditions. The Gunnar Mine Site, located on the northern shore of the Athabasca Lake, was used as a case study to test the effectiveness of biochar as a soil amendment. Greenhouse and field trials were run to study the effect of biochar and peat application on the growth and establishment of native plant species.

The greenhouse trials showed that both peat and biochar had a positive effect on plant growth, but different plant species had individual responses to each organic amendment. The field trials showed that peat promotes vegetation cover establishment better than biochar. Nevertheless, biochar also showed a positive effect on vegetation recovery through both establishment of seeded plants and self-establishment of natural invaders (plant species not seeded during the experiment). It was also determined that different plant species have a preference for organic amendment. In general, both peat and biochar can be used to promote plant establishment and growth, but biochar's effect on plant growth can vary widely depending on its properties.

Key Words: Exotic Species, Native Species, Organic Amendment, Peat, Mine Remediation.

INTRODUCTION

The establishment of vegetation cover on disturbed mine sites is one of the prime tasks of mine closure to protect the soil surface from wind and water erosion, restore wildlife habitats, and create opportunities for sustainable development of local Aboriginal communities. Properties of vegetation growth media is a most significant factor defining revegetation success. In northern environments, the fertile soil layer (topsoil) of uplands is very thin, with low organic matter and nutrients. It also can be easily destroyed or lost in during mining activities. Soil organic amendments and mineral fertilizers are, therefore, usually applied to improve topsoil properties and increase effectiveness of revegetation activities. Transportation of organic media in remote northern areas is very expensive because of their low density, but local harvesting for organic materials (e.g. peat) destroys natural habitats (e.g. wetlands). As a result, there is an emerging need for alternative organic soil amendments.

Biochar is a solid material obtained from the carbonisation of biomass through pyrolysis (Lehman and Joseph 2009). Addition of biochar to the soil can improve both its chemical and physical properties (Lehman and Joseph 2009; Verheijen et al. 2009). It has also been shown that biochar application creates favorable conditions for soil microbiota, promotes plant growth, and increases plant resistance to disease (Biederman and Harpole 2012; Elad et al. 2012; Verheijen et al. 2009). Therefore, this type of organic amendment can be beneficial for site restoration purposes. Potentially, biochar can be produced on-site or in nearby communities from local feedstock (e.g. organic wastes), which makes its attractive substitute for conventional organic amendments (Roberts et al. 2009). On the other hand, most biochar research is focused on its effect on cultivated crops and few research studies have considered its impact on native plant species (Adams et al. 2013; Elad et al. 2012; Sovu et al. 2013). Thus, there is a gap in understanding as to whether biochar can be used in ecological restoration and which trades off can be associated with its application.

The purpose of our research was to test effectiveness of biochar as a soil amendment for mine site restoration in northern Saskatchewan. The Gunnar Mine Site, located on the northern shore of the Athabasca Lake, was selected as a case study for the research, since one of the project tasks is to establish self-sustaining vegetation on the engineered cover that will be installed on the Gunnar tailings areas (SRC 2013). The cover material is to be taken from the local airstrip and/or neighboring areas. The proposed borrow material is coarse sand with gravel inclusions and relatively low content of organic matter (less than 0.1%), and has a limited capacity to support plant establishment and growth. As a result, application of organic amendments and mineral fertilizer is necessary to enhance its properties as a growth medium. During 2011 and 2012 two organic amendments (i.e. peat and biochar) and mineral fertilizer were tested under greenhouse and field conditions to study the response of native plant species to the soil treatments.

METHODS

Greenhouse Trials

The greenhouse trials comprised growing four plant species (i.e., Slender Wheatgrass (*Elymus trachycaulus* (Link) Gould ex Shinners), Rocky Mountain Fescue (*Festuca saximontana* Rydb.), American Vetch (*Vicia americana* Muhl. ex Willd.), Common Yarrow (*Achillea millefolium* L.)) in pots containing combinations of the borrow material with mineral fertilizer and two organic amendments (peat and biochar). The experiment had a completely randomized design with five replicates of each soil mixture and plant species.

Borrow material for the trials was collected from the borrow area at the Gunnar airstrip. The borrow material was sampled from the depth below 20cm to exclude top soil with its seed bank from the experiment. The borrow material was poor in organic carbon, nitrogen, and plant available phosphorus and potassium. It was poor in silt and clay, and composed mostly of coarse sand with a high proportion of gravel and big stones. Prior to the trial start-up, the borrow material was sieved through 1 cm sieves to remove the stones.

Sphagnum peat and willow dust biochar were used as organic amendments for the greenhouse trials. Both peat and biochar were purchased from commercial suppliers. Both organic amendments had low contents of plant available nitrogen, phosphorus, potassium, and sulfur. The organic matter content was higher in peat compared to biochar (93% vs. 76%). The water holding capacity of the peat was 509%, while that of the biochar was 454%. The application rate of organic amendments was targeted to achieve 2% of the organic matter in the soil mixture, so application rates for peat and biochar were 80 t/ha and 95 t/ha, respectively.

Borrow material and organic amendments were mixed by hand and used to fill 2 L pots (18 cm in diameter). All the pots were placed in the enclosed greenhouse in random order. The greenhouse conditions were adjusted to the Gunnar average monthly temperature during the growing season (i.e., 20ºC average air temperature during the 16 hours of light and a 10ºC average air temperature during the 8 hours of darkness).

Seeds for the greenhouse trials were obtained from commercial seed suppliers. Burton and Burton's (2003) recommendations on growing selected plant species were used as a basis for seeding rates and seeding depth, as follows:

- Slender Wheat Grass – 6 pure live seeds (PLS) per pot at the depth of 1.5 cm
- Rocky Mountain Fescue – 22 pure live seeds (PLS) per pot at the depth of 1 cm
- American Vetch – 4 pure live seeds (PLS) per pot at the depth of 1 cm
- Common Yarrow – 11 pure live seeds (PLS) per pot on the soil surface

Before seeding, all pots were excessively watered to imitate spring snowmelt conditions. Fertilizer was applied to the corresponding pots after seeding. Saskatchewan Forage Council (1998) recommendations on slender wheatgrass cultivation were used as a basis for fertilizer rates, which were 45 N kg/ha, 84 P_2O_5 kg /ha, and 112 K_2O kg/ha for soils with poor nutrient content.

The trial time period was 12 weeks, which is close to the growing season at Gunnar. During the trial period, the pots were rotated weekly to avoid the edge effect, and were watered every third day at a rate imitating the Gunnar average monthly precipitation that varied from 38 mm (week 1 to 4) to 53 mm (week 5 to 12). During the trials, the seedling number in every pot was measured weekly. On the third week of the trials, it was noticed that direct sunlight might overheat the soil mixtures with biochar because of its black colour, impeding seed germination and growth. To avoid such undesirable effects, the greenhouse shades were closed. No other changes in temperature or the water regime were made. At the end of the experiment, the aboveground biomass from each pot was harvested, dried, and weighed.

The experimental data were further processed and analyzed to quantify the following indices: plant establishment rate, seedling emergence rate, seedling survival rate, and aboveground biomass dry weight.

All data were tested for normality using the Shapiro-Wilk test. If data did not fit a normal distribution, the Kruskal-Wallis test, followed by the Conover-Iman test, was used to assess statistical differences in response of the investigated indices to the soil treatments. If data were normally distributed, analysis of variance (ANOVA), followed by the Tukey's HSD test (honestly significant difference), was applied. XLSTAT was used to run the above statistical tests for all data groups. The significance level for all tests was 0.05.

<u>Field Trials</u>

The field trials comprised the sowing of a native species seed mix on different combinations of the borrow material, two soil organic amendments (three rates), and mineral fertilizer (two rates). The experiment had a factorial design with 4 replicates of each combination of borrow material with organic amendment or/and mineral fertilizer.

The study area is located within the Taiga Shield Ecozone and the Tazin Lake Upland Ecoregion. The trials were set up on the abandoned side of the Gunnar airstrip in the middle of June 2012. Before the trial set up, the research area was cleared of vegetation. Due to high compaction of the airstrip material and high content (up to 50% by volume) of big stones in it, we constructed 7 wooden bottomless boxes (frames). Each box was 0.3 m x 4 m x 6 m and was divided into twelve 1.5 m x 1.5 m cells. The boxes were half-buried below the soil surface. One box was filled with the pure borrow material and six boxes were filled with mixture of borrow material with two organic amendments at three different rates. Boxes for the soil mixtures were assigned on a random basis.

Borrow material for the trials was collected from a borrow area at Gunnar. The borrow material was sampled at the depth below 20 cm to exclude top soil with the seed bank from the experiment. The borrow material was poor in organic carbon, nitrogen, plant available phosphorus, and potassium. It was composed mostly of sand with a high inclusion of gravel and big stones and poor of silt and clay. Prior to the trial start-up the borrow material was screened through 5 cm steel mesh to exclude large stones.

Sphagnum peat and pine chunky biochar were used as organic amendments for the field trials. Both peat and biochar were purchased from commercial suppliers. Both types of organic amendments had low contents of plant available nitrogen, phosphorus, potassium, and sulfur. Organic matter content was higher in peat and lower in biochar (94% vs. 78%). Water holding capacities of the peat and biochar were 523% and 68%, respectively. The application rate of organic amendments was targeted to achieve 2%, 4% and 6% of organic matter in the soil mixture, so application rates for organic amendments were 80, 160, and 240 t/ha of peat (hereafter, peat rates referred as "peat at low, medium, or high rate") and 90, 190, and 280 t/ha of biochar (hereafter, biochar rates referred to as "biochar at low, medium, or high rate").

After the boxes were filled with soil treatments, as described above, native plants were seeded by hand broadcasting on 1 m^2 plots placed in the centre of the box cells. The seed mixture comprised eight grasses, five forbs, and one shrub. Its composition in percentage of pure life seeds by weight was as follows:

- Rocky Mountain Fescue (*Festuca saximontana* Rydb.) – 20%
- American Vetch (*Vicia americana* Muhl. ex Willd.) – 20%
- Streambank Wheatgrass (*Elymus lanceolatus ssp. riparius*) – 10%
- Slender Wheatgrass (*Elymus trachycaulus* (Link) Gould ex Shinners) – 10%
- Violet Wheatgrass (*Elymus violaceus* (Hornem.) Feilberg) – 10%
- Tufted Hairgrass (*Deschampsia caespitosa* (L.) P. Beauv.) – 7%
- Rough Hair Grass (*Agrostis scabra* Willd.) – 7%
- Canada Buffaloberry (*Shepherdia canadensis* (L.) Nutt) – 6%
- Canadian Milkvetch (*Astragalus Canadensis* L.) – 4%
- Marsh Reed Grass (*Calamagrostis canadensis* (Michx.) P. Beauv.) – 3%
- White Bluegrass (*Poa glauca* Vahl) – 1%
- Alpine Milkvetch (*Astragalus alpinus* L.) – 1%
- Prairie Crocus (*Anemone patens* L.) – 1%

- Fireweed (*Chamerion angustifolium* (L.) Holub) – 0.1%

The seeding rate was 2000 PLS/m^2 or 15.6 PLS kg/ha.

After seeding, the mineral fertilizer was applied by the hand broadcasting. Fertilizer rates were designed in line with the lowest and highest agronomic rates recommended by the Saskatchewan Forage Council for slender wheatgrass cultivation (SFC 1998). The low fertilizer rate was 22 N kg/ha, 56 P_2O_5 kg /ha, 56 K_2O kg/ha, and 10 S kg /ha. The high fertilizer rate was 45 N kg/ha, 84 P_2O_5 kg /ha, 112 K_2O kg/ha, and 20 S kg/ha. Plots for fertilizer application were assigned within each box on a random basis.

A vegetation survey of the trial plots was carried out two months after seeding. For each sampling quadrat, the vegetation cover was assessed using the modified Daubenmire method (Bayiley and Poulton 1968). As vegetation on the plots was presented by both seeded plants and natural invaders, the effectiveness of soil treatments was assessed on the basis of their impact on total vegetation cover, seeded plant cover and cover of dominant invaders (i.e., rough cinquefoil and strawberry blite). The Kruskal-Wallis test, followed by the Conover-Iman procedure, was used to assess the statistical significance of the response of the investigated indices to the soil treatments. XLSTAT was used to run the above statistical tests for all data groups. The significance level for all tests was 0.05.

RESULTS

<u>Greenhouse trials</u>
Figure 1 shows the aboveground biomass dry weight (ABDW), seedling emergence rate (SER), seedling survival rate (SSR), and plant establishment rate (PER) for each species on the tested soil mixtures.

Mineral fertilizer addition to the borrow material promoted slender wheatgrass growth, increasing ABDW by a factor of 1.6 (from 62 to 110 mg per pot; $p = 0.036$), but had no effect on the growth of other plant species (p varied from 0.065 to 0.258, depending on the species). This treatment also fostered seedling survival of American vetch, increasing SSR by a factor of 1.7 (from 35 to 59%; $p = 0.048$), yet its impact on seedling emergence was not strong enough ($p = 0.432$) to provide a statistically significant overall positive effect on the plant establishment ($p = 0.843$). There was no significant effect of fertilizer application on the plant establishment, seedling emergence, and seedling survival of the other three plant species (p varied from 0.144 to 0.977, depending on the index and plant species).

Addition of peat to the borrow material fostered growth of all four plant species, increasing ABDW by a factor of 3 for slender wheatgrass (from 62 to 209 mg per pot; $p<0.001$), 6 for rocky mountain fescue (from 62 to 373 mg per pot; $p<0.001$), 14 for American vetch (from 8 to 111 mg per pot; $p<0.001$), and 73 for common yarrow (from 2 to 131 mg per pot; $p<0.0001$). This treatment had an overall positive effect on establishment of rocky mountain fescue and common yarrow, increasing SER by a factor of 1.2 ($p = 0.001$) for rocky mountain fescue and 2 ($p = 0.001$) for common yarrow. SSR increased by a factor of 1.3 ($p<0.001$) for rocky mountain fescue and 8 ($p<0.001$) for common yarrow, while PER increased by a factor of 1.6 ($p<0.001$) for rocky mountain fescue and 19 ($p<0.001$) for common yarrow. The favorable effect of peat on American vetch resulted in an increase in the SSR by a factor of 3 ($p<0.001$) and PER by a factor of 4 (p = 0.001). There was no significant effect of peat application on slender wheatgrass seedling emergence (p = 0.602), seedling survival (p = 0.532), or plant establishment (p = 0.974).

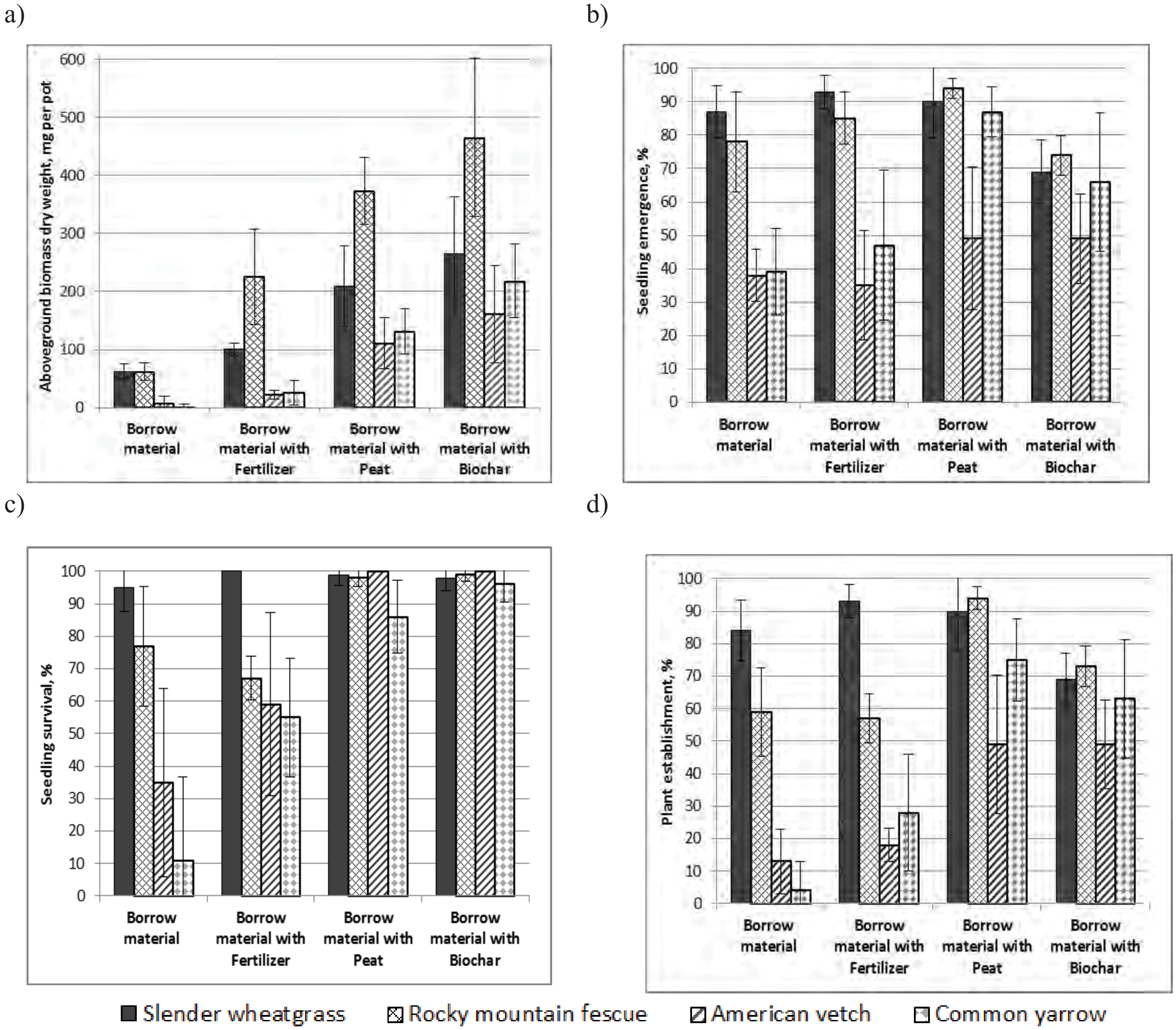

Figure 1. Effect of mineral fertilizer, biochar, and peat on the aboveground biomass dry weight (a), seedling emergence (b), seedling survival (c), and plant establishment (d) of the plant species tested during the greenhouse trials. Error bars indicate standard deviation.

Biochar addition to the borrow material fostered the growth of all four plant species, increasing ABDW by a factor of 4 for slender wheatgrass (from 62 to 265 mg per pot; p<0.001), 8 for rocky mountain fescue (from 62 to 465 mg per pot; p<0.001), 20 for American vetch (from 8 to 161 mg per pot; p<0.001), and 121 for common yarrow (from 2 to 218 mg per pot; p<0.001). This treatment also promoted common yarrow seedling emergence and seedling survival, increasing SER and SSR by factors of 1.7 (from 39% to 66%; p = 0.037) and 9 (from 11 to 96; p<0.001), respectively, which resulted in an increase of PER by a factor of 16 (from 4% to 63%; p<0.001). The favorable effect of biochar on mountain fescue and American vetch resulted in increases in SSR by factors of 1.3 (from 77 to 99%; p<0.001) and 3 (from

35% to 100%; $p<0.001$), respectively, for these species, which resulted in an increase of PER by a factor of 1.2 (from 59% to 73%; $p = 0.013$) for rocky mountain fescue and 4 (from 13% to 49%; $p=0.001$) for American vetch. Biochar application, however, impeded slender wheatgrass seedling emergence, decreasing SER by a factor of 1.3 (from 87% to 69%; $p = 0.006$), yet it had no pronounced effect on the seedling survival ($p = 0.532$) or plant establishment ($p = 0.107$).

In general, a favorable effect of organic amendments on the investigated plants was more pronounced than the effect of mineral fertilizer, except for the following cases:

- Peat and mineral fertilizer had similar effects on slender wheatgrass seedling emergence, seedling survival, and plant establishment (p = 0.498, 0.454, and 0.974, respectively) and American vetch seedling emergence ($p = 0.229$)
- Biochar and mineral fertilizer had similar effects on seedling emergence of American vetch and common yarrow (p = 0.229 and 0.149, respectively) and seedling survival of slender wheatgrass ($p = 0.393$).

Comparing the effects of peat and biochar on the tested plant species, we obtained the following results:

- Both amendments had similar effects on the establishment of American vetch and common yarrow (p = 1 and 0.136, respectively) and the growth of slender wheatgrass, rocky mountain fescue and American vetch (p = 0.657, 0.288, and 0.165, respectively)
- Peat addition to the borrow material resulted in better establishment of rocky mountain fescue in comparison with the biochar addition (94% on peat vs. 73% on biochar; $p = 0.001$)
- Biochar addition to the borrow material resulted in higher ABDW of common yarrow than addition of peat (218 mg per pot on biochar compared to 131 mg per pot on peat; $p = 0.014$)
- Biochar addition to the borrow material had a negative effect on the SER of slender wheatgrass, while peat addition did not affect this index (87% on borrow material compared to 90% on peat and 71% on biochar; $p = 0.002$ for peat compared to biochar).

<u>Field Trials</u>

Two months after the trial start-up, vegetation was observed on all the plots. A total of 30 vascular plant species were found within the overall research area (all plots together; Table 1). Of this total, 14 species were seeded during the trial startup and 16 species were natural invaders that were incorporated into the plots with the borrow material or were transported from nearby areas by wind.

Table 1. List of plant species observed during the field trials.

Seeded species	**Invading species**
Agrostis scabra Willd. – Rough Hair Grass (native) *Astragalus alp*inus L. – Alpine milkvetch (native)	*Achillea millefolium* L. – Common yarrow (native)
Astragalus canadensis L. – Canada milkvetch (native)	*Arabis hirsuta* (L.) Scop. – Hirsute rock cress (native)

Seeded species	Invading species
Brassica napus L. – Canola (exotic, seeded by accident)	*Arabis holboellii* Hornem – Reflexed rock cress (native)
Calamagrostis canadensis (Michx.) P. Beauv. – Marsh Reed Grass (native)	*Artemisia campestris* L. – Sagewort wormwood (native)
Chamerion angustifolium (L.) Holub – Fireweed (native)	*Chenopodium album* L. – Lamb's quarter (exotic)
Deschampsia cespitosa (L.) P. Beauv. – Tufted hairgrass (native)	*Chenopodium capitatum* (L.) Ambrosi – Strawberry blite (native)
Elymus trachycaulus (Link) Gould ex Shinners – Slender Wheatgrass (native)	*Crepis tectorum* L. – Annual hawksbeard (exotic)
Elymus lanceolatus ssp. riparius – Streambank Wheatgrass (native)	*Geranium bicknellii* Britton – Bicknell's geranium (native)
Elymus violaceus (Hornem.) Feilberg – Violet Wheatgrass (native)	*Matricaria matricarioides* L. – Pineapple weed (exotic)
Festuca saximontana Rydb. – Rocky Mountain Fescue (native)	*Plantago major* L. – Common plantain (exotic)
	Polygonum aviculare L. – Prostrate knotweed (native)
Poa glauca Vahl – White Bluegrass (native)	
Shepherdia canadensis (L.) Nutt – Canada buffaloberry (native)	*Potentilla bimundorum* Soják – Staghorn cinquefoil (native)
Vicia americana Muhl. ex Willd. – American vetch (native)	*Potentilla norvegica* L. – Rough cinquefoil (native)
	Rorippa palustris (L.) Besser – Bog yellowcress (native)
	Salix spp. – Willow (native)
	Taraxacum officinale F.H. Wigg. – Common dandelion (exotic)

Figure 2 shows the total vegetation cover (TVC), seeded plant cover (SPC), rough cinquefoil cover (RCC) and strawberry blite cover (SBC) on the tested soil mixtures.

Plant establishment on the borrow material without any amendments (control) was very poor (TVC = 2%, SPC = 0.5%, RCC = 0.4%, and SBC = 0.3% in average). Organic amendments alone had very low or no impact on plant establishment (the average increment of TVC did not exceed 6% for biochar and 4% for peat). Fertilizer alone applied at the high rate promoted plant establishment to a larger extent than organic amendments (the average increment of TVC was 16%).

In comparison with the control, peat/fertilizer combinations had the most positive impact on all TVC, SPC, and RCC. All three indexes were the highest when:

- peat at the low rate was combined with fertilizer at the high rate (TVC = 33%, $p<0.001$; SPC = 10%, $p<0.001$; RCC = 15%, $p<0.001$)
- peat at the medium rate was combined with fertilizer at the high rate (TVC = 39%, $p<0.001$; SPC = 10%, $p<0.001$; RCC = 15%, $p<0.001$)

- peat at the high rate was combined with fertilizer at the low rate (TVC = 44%, p<0.001; SPC = 8%, p<0.001; RCC = 44%, p<0.001) and the high rate (TVC = 28%, p<0.001; SPC = 10%, p<0.001; RCC = 25%, p<0.001).

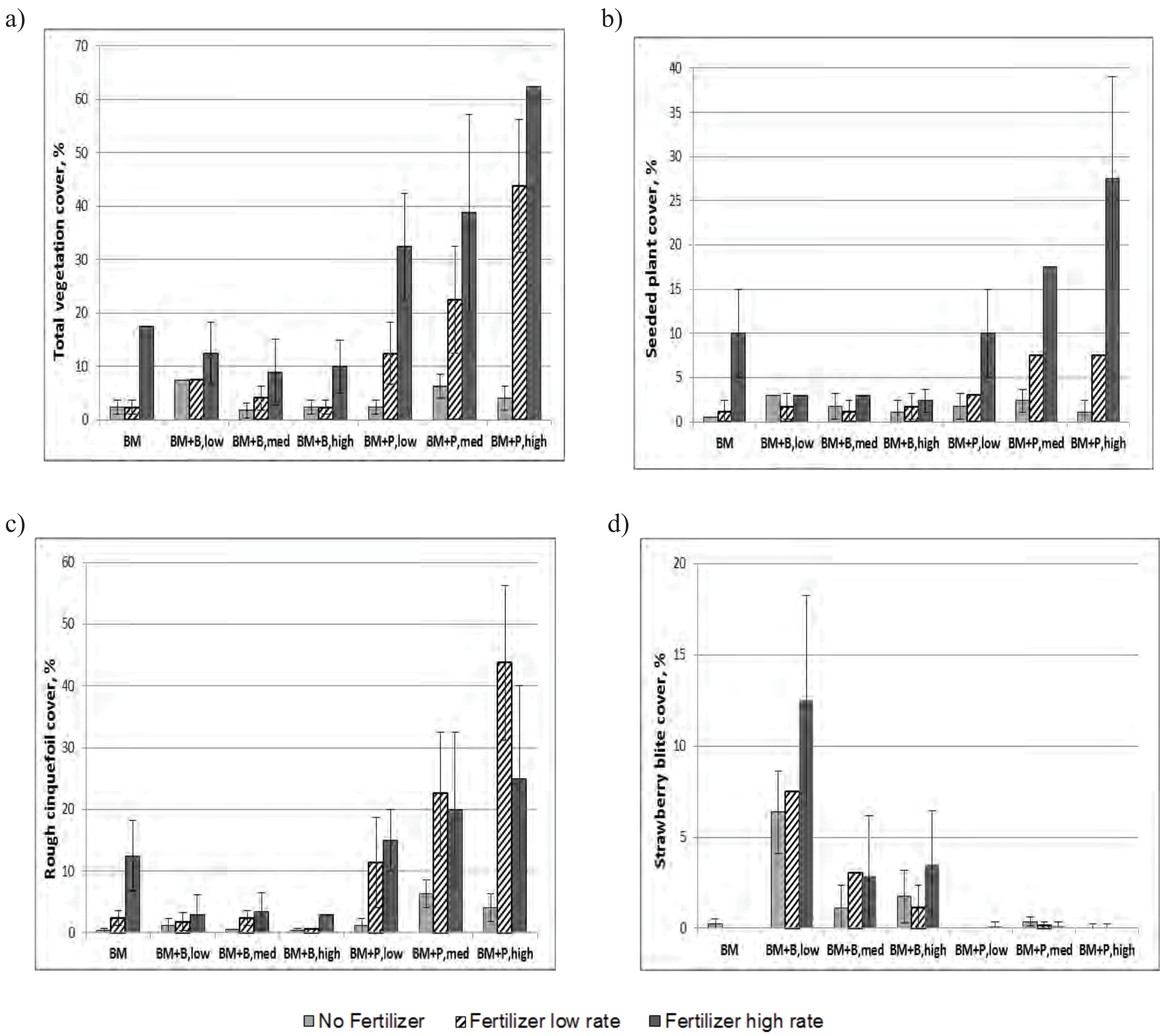

Figure 2. Effect of mineral fertilizer, biochar, and peat on the total vegetation cover (a), seeded plant cover (b), rough cinquefoil cover (c), and strawberry blite cover (d), at field trials. Error bars indicate standard deviation (absence of error bar means that the standard deviation is zero). BM – borrow material; B, low/med/high – biochar added at low/medium/high rate; P, low/med/high – peat added at low/medium/high rate.

There was no statistically significant difference for TVC, SPC, and RCC data for the above treatments (p varied from 0.059 to 0.761).

All biochar/fertilizer treatments increased SBC, while peat application did not affect this parameter. SBC varied from 1% on the plots with biochar at the high rate and fertilizer at the low rate to 13% on the plots with biochar at the low rate and fertilizer at high rate, which is significantly higher compared to control plots ($p = 0.001$ and $p<0.001$, respectively). Biochar/fertilizer combinations also promoted TVC (up to 13%), SPC (up to 3%), and RCC (up to 4%), but these effects were significantly lower than effects from the above peat/fertilizer treatments ($p <0.001$ in all cases).

Interestingly, when biochar was applied alone, the increase of its rate from low to high resulted in a significant decrease of TVC (from 6% to 2%, $p<0.001$), SPC (from 3 to 1%, $p = 0.006$) and SCC (from 6% to 2%, p = XX). When biochar was applied with fertilizer at the low rate, the same trend was observed, only for TVC. The latter decreased from 8 to 2% when the biochar rate was increased from low to high ($p<0.001$). There was no significant difference between the indexes when biochar at different rates was applied with fertilizer at the high rate (p varied from 0.092 to 1.000).

DISCUSSION AND CONCLUSION

While the greenhouse trials demonstrated that biochar is a good substitute for peat as a soil amendment, the field trials showed that peat promotes plant establishment and growth to a larger extent than biochar. The contradictory outcome from our research can be explained by the variability of biochar, which is not a standardized material; as a result, its properties vary depending on feedstock and the pyrolysis process used for the biochar production (Biederman and Harpole 2012; Lehman and Joseph 2009; Verheijen et al. 2009). In our case, the willow dust biochar that was used for the greenhouse trials had a water holding capacity similar to the peat, while the water holding capacity of the pine chunky biochar used for the field trials was eight times lower than that of the peat. Therefore, borrow material mixed with biochar during the greenhouse trials had a higher capability to retain water and nutrients compared to borrow material mixed with the same amount rate of biochar used for the field trials. This demonstrates that water holding capacity is likely an important factor in the benefit gained from a given soil amendment.

Our research also showed that peat and biochar had different effects on the establishment and growth of different plant species. The greenhouse trials showed that establishment of rocky mountain fescue was promoted by peat application to the larger extent than by biochar, but growth of common yarrow was fostered by biochar application to the larger extent than by peat. The slender wheatgrass establishment was impeded by biochar application. As a result of the field trials, biochar had a positive effect on the growth of strawberry blite, while peat promoted growth of rough cinquefoil. A better understanding of these finding would require a separate study including a literature review and specially designed experiments.

In case of the field trials, the effect of the organic amendment on the plant community composition can be explained by species competition traits, as follows. Peat has higher ability to hold and retain water and nutrients than biochar; therefore, its presence in the borrow material was more favorable for the those plants normally found in wetter areas, such as rough cinquefoil, tufted hairgrass, marsh reed grass or streambank wheatgrass. Faster development of these plants under favorable conditions made them stronger competitors for the resources, e.g. they could develop faster than other species, thereby impeding the development of the latter. Addition of biochar to the borrow material also improved its properties, but to a lower extent than peat. For the treatments with biochar, those plant species adapted for moist conditions were less competitive than other species. Thus, biochar addition to the borrow material created better conditions for ruderal species, such as strawberry blite, which is known as a pioneer species on disturbed areas relatively depleted of water and nutrients, but is not a strong competitor under more favorable conditions.

It should be noted that higher rates of biochar application had a negative impact on both vegetation establishment and development. This phenomenon is in line with the results of other researchers who suggested an idea of "biochar loading capacity" (Verheijen et al. 2009). Biochar loading capacity (BLC) is the maximum amount of biochar that can be added to a soil without compromising its other properties and, therefore, impeding plant growth. BLC can vary from a few tens to a few hundreds of tonnes per hectare, depending on soil properties, biochar properties and plant species. In our case, BLC is likely to be within the interval of 90 to 190 tonnes of biochar per hectare and increases in proportion to fertilizer rates.

However, unlike peat, biochar can be produced locally, which may save cost on the transportation of soil amendments to remote sites, create local job opportunities, provide opportunities for organic waste recycling, and also reduce the anthropogenic footprint (since no wetlands will be destroyed due to peat harvesting). Biochar is also a very stable material, since decomposition takes decades, so it can serve as a carbon sink addressing global warming issues (Biederman and Harpole 2012; Montanarella and Lugato 2013; Verheijen et al. 2009). Thus, biochar application can provide a more sustainable approach to land reclamation.

In conclusion, although both biochar and peat treatments showed a significant positive effect on plant establishment and growth, peat appears to be a more suitable organic amendment for revegetation projects. On the other hand, biochar application can assist in achievement of other sustainable remediation goals, such as carbon sequestration or waste management enhancement. Therefore, assessment of biochar and peat effectiveness under sustainable remediation framework is a focus of our next research.

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NATURAL PROCESSES: AN EFFECTIVE MODEL FOR MINE RECLAMATION

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ABSTRACT

Restoration programs based on the use of natural processes can reduce the costs of restoration while providing self-sustaining restored ecosystems that re-integrate with the local recovery trajectories. Natural processes have been restoring disturbance sites (Walker 2012) since the advent of terrestrial vegetation over 450 million years ago. By following how these processes operate these recovery processes can be harnessed for the reclamation of mining disturbances. The first step in developing a restoration program that uses natural processes is to identify the factors or filters that are preventing or constraining natural recovery. Polster (2009) listed eight filters that are common on many mine sites. Compaction, erosion and steep slopes are three of the most common filters at large mines. How do natural processes solve these problems? Using a model based on observation of these natural solutions to common problems, restoration systems for drastically disturbed sites can be developed. Flattening slopes and preparing the site using the rough and loose technique (Polster 2009) addresses these issues. Pioneering species such as willows, poplars and alder can be used to initiate the natural successional processes that will restore the site (Polster 1989). The addition of physical structures such as large woody debris and rock piles can aid in the return of the functions that were associated with these structures (e.g., habitat). The use of natural processes and the re-establishment of ecological functions associated with these processes can greatly benefit the recovery of degraded sites at little or no cost to industry. Examples are provided from the author's experience.

Key Words: Vegetation Succession, Natural Processes, Recovery, Restoration, Filters, Rough and Loose.

INTRODUCTION

The complexity of ecosystems makes it difficult to understand how they operate (Gonzales 2008). The study of this complexity on drastically disturbed sites such as mines and other major disturbances has only recently drawn the attention of scientists (Walker 2012). However, by looking at the ecology of naturally disturbed sites such as landslides (Walker and Shiels 2013) or talus slopes (Polster and Bell 1980) and the methods and conditions that natural systems use to establish vegetation on these sites, we can gain insights into how we can use these same processes to establish vegetation on sites we disturb. Restoration is defined by the Society for Ecological Restoration as the process of assisting the recovery of an ecosystem that has been degraded, damaged or destroyed (SERI 2004). In this paper I use the words reclamation and restoration interchangeably since reclamation seeks to re-establish productive, self-sustaining ecosystems on lands disturbed by mining. Effective reclamation therefore is ecological restoration. Public acceptance of large scars on the land caused by mining is waning. Providing effective restoration of mining disturbances is essential for the maintenance of social license.

Determining the factors that are preventing the recovery of the site is the first step in designing an effective restoration program for a disturbed mine site. The word 'filters' is used to describe the features that prevent recovery (Clewell and Aronson 2013). This term implies that the limiting factor will allow some species to occur but not others, much as a filter allows some things to pass but not others. The edge of a compacted roadway where only a few weedy species occur is an example. Compaction of the roadway coupled with lack of nutrients, possibly de-icing materials (salt and sand) from the road and the disturbance of road grading may all contribute to preventing non-weedy species from occurring. Filters can be abiotic (non-living) such as compaction and steep slopes, or biotic (living) such as weeds or a lack of propagules or excessive herbivory. The concept of ecological filters within the context of mine reclamation is described in greater detail below.

Natural systems have evolved a variety of ways of addressing the filters that prevent vegetation establishment. For instance, glacially compacted tills underlie many areas of Canada. These tills are as compacted as mine haul roads, and yet productive forests have established on them in the ten thousand years since glaciation. How does this happen? What can be done to mine haul roads so that it does not take ten thousand years for regrowth? Natural strategies for addressing filters are discussed further below.

The species that colonize disturbed sites provide functions and processes that assist the recovery of disturbed sites (Polster 1989). What are these functions and processes and how can we use them to assist in the recovery of sites disturbed by mining? These aspects of restoration of disturbed sites are discussed below.

FILTERS TO RECOVERY

Polster (2009) listed eight abiotic filters common in industry. **Steep slopes** are one of the most common filters on mine waste rock dumps. The continual movement of angle-of-repose (37^o) slope surfaces prevents the establishment of plant seedlings (Polster and Bell 1980). The lower portions of waste rock dump slopes and natural talus slopes are composed of coarse rock fragments with few fine textured materials to hold plant-available moisture and nutrients. The coarse rock at many mine sites creates an **adverse texture** filter. A lack of available plant nutrients is common at many industrial sites so **nutrient status** is another common filter. In some cases **adverse chemical properties** such as acid rock drainage or high salinity levels restrict the growth of plants. Dark substrates create **soil temperature extremes** that can limit plant growth. **Compaction, adverse micro-climatic conditions** and **excessive erosion** are other common filters at many sites.

Biotic filters can prevent recovery as well (Polster 2011). H**erbivory** (Green 1982), **competition** (Temperton et al. 2004), **propagule availability** (Temperton et al. 2004), **phytotoxic exudates** (GOERT 2011), **facilitation** and **species interactions** (Temperton et al. 2004) are all biotic filters. These filters may operate independently or they may combine with abiotic filters to create complex filters. For instance excessive erosion (abiotic filter) can be a problem on bare slopes. Seeding with an agronomic grass and legume cover has been the standard approach to deal with this problem. However, creating a dense stand of grasses and legumes can create habitat for small mammals (Green 1982) that then causes excessive herbivory and competition (Polster 2010). Care must be taken so that solving one problem does not create others. The following paragraphs present solutions to common filters that can be used at mines in British Columbia. These solutions are based on the strategies that are found in natural systems for addressing these filters. Solutions for mining problems can be found by observing how natural systems solve common filters.

STRATEGIES FOR ADDRESSING FILTERS

Steep, angle-of-repose rock slopes are composed of fine textured materials at the top of the slope grading into progressively coarser textured materials down the slope until the coarsest materials (often large boulders) are found at the bottom of the slope. These patterns occur on natural talus slopes (Polster and Bell 1980) as well as on mine waste dump slopes (Milligan 1978) and help explain why these slopes are inherently stable. Moisture that collects in the materials can freely drain out the bottom of the slope. How do natural systems address these problems? At the top of steep slopes the slope and fine textured materials results in erosion, moving the fine textured material down the slope. Plants colonize the fine textured substrates and eventually erosion is controlled. Although this is a slow process and one that is governed by chance events, there are a variety of soil bioengineering methods that can be used on steep slopes to address the slope / erosion issues (Polster 1999). These may be considered too expensive to use in a mining context; for example, the option of re-sloping large waste rock dumps can involve extensive machine time and be far more costly in the long run.

The coarse textured materials in the middle of a natural angle-of-repose slope are slowly revegetated by fine textured soils that wash from the slopes above through erosion or through the accumulation of organic matter that collects in the interstitial spaces between the rocks (Polster and Bell 1980). These natural processes are very slow. Soil bioengineering can be used to initiate (assist) the recovery processes on coarse textured substrates through a technique called pocket planting (Polster 2008). This treatment applies fine textured soils brought in to fill the voids between the rocks and create pockets of vegetation that then help the recovery processes on the remainder of the slope. The coarse materials at the bottom of the slope recover by the collection of organic material without contributions from above. Pocket planting can be used in coarse rock sites to expedite this process. Resloping waste rock dumps with heavy equipment moves the fine textured materials from the top of the slope down, covering the coarse materials lower on the slope. Wrap-around waste dumps can significantly reduce the costs of resloping (Milligan 1978).

Filters such as compaction, dark substrates, erosion and a lack of micro-sites can all be addressed through the use of a technique called 'rough and loose' (Polster 2011). Rough and loose surface configurations can be achieved by using a large excavator to open holes on the slope, dumping the material that is generated from the holes in mounds between the holes. The excavator, using a digging bucket (not clean-up), takes a large bucket full of soil and places it to the left of the hole that was just opened; half a bucket width from the hole so it is half in and half out of the hole. A second hole is then excavated half a bucket width to the right of the first hole. Material from this hole is then placed between the first and second holes. A third hole is now opened half a bucket width to the right of the second hole, with the excavated soil placed between the second and third holes. Care should be taken when excavating the holes to shatter the material between the holes as the hole is dug. The process of making holes and dumping soil is continued until the reasonable operating swing of the excavator is reached. The excavator then backs up the width of a hole and repeats this process, being sure to line up the holes in the new row with the space between the holes (mounds) on the previous row.

As the name implies, making sites rough and loose addresses compaction by breaking up the surface down to approximately one metre (depending on excavator size). This allows the soil to absorb moisture rather than flowing on the surface and therefore prevents erosion. North and south exposures are created so the dark substrates associated with coal mines can be ameliorated by planting on the north slopes. Conversely where cool temperatures limit plant growth such as in northern areas the south-facing slopes can be used. Roots freely penetrate the loose substrates allowing access to moisture and nutrients. Rough and loose substrates have an abundance of micro-sites where seeds can lodge and seedlings can grow. Rough and loose treatments can be applied to the upper covering on covers designed to control acid rock drainage or other adverse chemistry. The 'sponge' cover system (O'Kane et al. 2001) allows a forest to be developed on top of a cover that seals reactive wastes. Using the rough and loose treatment on this cover provides excellent growth of forest species thus enhancing transpiration and the effectiveness of the cover.

Poor nutrient status is a common filter, especially if mine wastes are compared to agricultural soils. However, when compared to natural substrates on disturbed sites such as might occur following a landslide or talus slope or on a river gravel bar, the nutrient status of mining wastes is equivalent. How do natural systems address nutrient deficiencies on these sites? Pioneering species colonize natural disturbances (Walker and del Moral 2003). Many of these such as alder are associated with nitrogen fixing organisms (Binkley et al 1982). Red Alder is the most common pioneering species in coastal British Columbia. Alder is often found colonizing forest landslides (Straker 1996) and is an important contributor to the nitrogen balance in forest ecosystems (Peterson et al 1996). Sitka Alder is an important species in Interior locations (Sanborn et al. 1997). In both cases, alder contributes nitrogen to the local recovering ecosystems. Other pioneering species such as Balsam Poplar have been implicated in the enhancement of nitrogen status of recovering ecosystems (Peterson et al. 1992). Lichens also fix nitrogen and contribute to the nitrogen status of forest ecosystems, especially in the north (Henriksson and Simu 1971).

Downed woody debris is an important source of nutrients in recovering forests. In addition, woody debris provides an important function in the control of erosion. Woody debris also provides habitat for a variety of plants and animals. Red Huckleberry, an important forest species in coastal British Columbia is often found growing on rotting logs and old stumps. Birds may play an important role in distributing this species as they perch on the debris. Similarly, woody debris forms important habitat for many small mammals, reptiles, amphibians and invertebrates. Including woody debris as piles and/or single pieces either standing or on the ground can contribute immensely to the creation of habitat and the cycling of nutrients on restoration sites. In addition, the cost of woody debris placement can be far less than the cost of chipping or burning.

Leaf litter is an important contributor to ecosystem health. The litter of Red Alder contributes substantial amounts of nitrogen to ecosystems where it occurs. Leaf litter can also protect bare soils from raindrop erosion. Leaf litter provides habitat for a variety of organisms important in the nutrient cycling processes of ecosystems. Leaf litter adds carbon to the soil providing an important carbon sequestration role. In some cases invertebrates need leaf litter to complete stages in their life cycles. There are opportunities to bring the spores and propagules of important soil organisms (e.g., mycorrhizal fungi) from forests to restoration sites by collecting leaf litter from the adjacent forest and scattering it on the restoration sites.

Establishing species that will provide structure for the developing ecosystem can expedite the recovery processes. The winter branches of Red Alder catch the spores of Swordferns as well as supporting the perching of frugivorous birds and soon the species they eat such as Salmonberry, start to show up in the understory (Polster 2010). Similarly, providing habitat for squirrels and chipmunks encourages the growth of mycorrhizal fungi as these small mammals collect the fruits of the fungi and cache them in various places in the forest. Understanding how these processes operate allows simple measures to be implemented during the restoration of the disturbed area that will build on the simple treatments that have been applied (e.g., making soils rough and loose, planting pioneering woody species and scattering woody debris). In some cases, supplying nest boxes for key species can bridge the gap between the open mining disturbance and when the pioneering species reach a level of maturity to provide the habitat.

Propagule availability can be an important element in ecosystem establishment. On large disturbed sites seeds of many species may not reach appropriate locations (Walker 2012). However, many pioneering species have developed effective means of distributing over large distances. The fluffy seeds of Balsam Poplar can be seen floating around at certain times in the spring. Similarly, the seeds of Sitka Alder can be found on the first winter snow in the fall. Although these seeds may travel long distances, in situations where parent plants are not available near the disturbed site or where the distances are too great, the lack of seeds of pioneering plants may be the limiting factor in the establishment of these species. Collection of the seeds of pioneering species and the application of these on disturbed sites can help overcome this filter. In some cases, animals can move the seeds of plants onto reclaimed areas. Providing perching sites or denning sites such as woody debris piles or rock piles can assist in this process.

Herbivory can be an important filter preventing recovery of some species. Small mammal (Green 1982) populations can explode under the cover of grasses and legumes that have been traditionally been used for reclamation. Similarly, populations of ungulates (deer and elk specifically) have responded positively to the extensive areas of grass and legume seeding at many mines. This has resulted in excessive herbivory and changes in the recovering ecosystems. Competition is another factor that can limit recovery. Careful study over sixteen years at the Island Copper Mine have identified that dense stands of seeded grasses and legumes can compete with planted woody species for moisture during periods of dry weather (Polster 2010). In some cases, seeded grass and legume species facilitate the establishment and growth of non-native weedy species (Polster 2010). This further complicates the establishment of productive, self-sustaining ecosystems as species such as Scotch Broom can inhibit tree growth by exuding a phytotoxic material. In some cases, specific species interactions such as between a plant and a pollinator can limit vegetation establishment. Wind pollinated pioneering species avoid this issue.

Incorporating biodiversity enhancements into the restoration of drastically disturbed sites can greatly improve the restoration work that is undertaken often at little or no cost. Resilience (Holling 1973) is built on redundancy. Providing a suite of nitrogen fixing species from trees and shrubs such as Red Alder and Sitka Alder to a diversity of shrub species including Soopolallie, Ceanothus and Wolf Willow (*Elaeagnus commutata*) as appropriate to the site being treated, that can all provide a similar ecological function, will ensure the restored site is prepared for future uncertainty. Similarly, the creation of rough and loose surface configurations ensures a level of topographic heterogeneity (Larkin et al. 2008) that will enhance diversity since it creates a variety of different habitats.

CONCLUSIONS

Natural recovery processes have been restoring disturbed ecosystems for millions of years. Following how these natural processes operate and the functions that various components provide can greatly enhance restoration operations and significantly reduce costs. Avoid spending money on seeding a cover of grasses and legumes that has been shown to limit recovery. Instead, seek to integrate natural processes into the restoration of disturbed sites and allow these natural processes to 'pay' for the recovery. Finding the filters that are preventing recovery is the initial step in providing a site that restores itself. In some cases, all that is needed is to create the appropriate surface conditions (e.g., rough and loose), add some woody debris and rocks as structure and apply some forest litter to re-establish nutrient cycling pathways. In other cases, planting of pioneering species will be required to provide the recovery functions associated with nitrogen fixation and the creation of ecological structure.

Social license is required to open new mines and effective reclamation of disturbances is essential to build and maintain social license. Expectations for the reclamation of new mines are significantly greater than in the past. The days of lakes being used as tailings disposal areas or that creek valleys could be used for the deposition of waste rock, exiting the rock drain as selenium contaminated stream, are past. Similarly, the thought that hyper-abundant ungulates wading in belly deep alfalfa can serve as a surrogate for biodiversity has been debunked (Martin et al. 2011). Restoring mining disturbances to enhance biodiversity is the future of mining. Natural processes and functions can provide that pathway.

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IMPLEMENTATION OF CONTAMINATED WATER MANAGEMENT SYSTEM UPGRADES TO ALLOW FOR DEWATERING OF TWO OPEN PITS AT THE VANGORDA PLATEAU, FARO MINE COMPLEX, YUKON

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ABSTRACT

The Vangorda Plateau at the Faro Mine Complex contains two open pits that, starting in 2013, will both require dewatering. Since mine abandonment in 1998, water levels in Vangorda pit have been actively maintained below a maximum recommended elevation, whereas Grum pit has been filling at a rate of approximately 3 m per year due to average annual inputs of 400,000 m^3. In late 2011, Grum pit levels reached a threshold elevation requiring dewatering to begin in 2013 to prevent exceeding the maximum recommended elevation. Continued filling of Grum pit could allow contaminated pit water to enter groundwater in as little as two years, and reduced pit capacity may prevent its use for contingency surface water storage during extreme flood events. Water from the pits cannot be discharged directly to the environment because it is noncompliant with respect to metals. Zinc (Zn) is one of the main contaminants of concern, with a site discharge limit of 0.5 mg/L. Vangorda pit water samples have contained up to 235 mg/L Zn (at pH 3 to 5), and Grum pit water samples have contained up to 6 mg/L Zn (at pH 7.5 to 9).

The Vangorda Water Treatment Plant (WTP) was constructed in 1992 to treat acidic and metal-contaminated water from Vangorda pit utilizing a low-density sludge lime treatment process. Contaminated water from Vangorda pit is pumped uphill to the WTP through a 24" HDPE pipeline. Treated water is directed to a polishing pond where sludge is settled, and compliant effluent is decanted for release to the environment. The pond's maximum sludge storage capacity is reached after treatment of 350,000 to 500,000 m^3 of Vangorda pit water over a four week summer period. Sludge removal is a laborious process that occurs in mid-winter when frozen sludge is excavated and hauled by truck to a nearby overburden dump. Thus, the current system has no excess sludge management capacity available to allow for treatment of an additional 400,000 m^3 of water that will need to be removed annually from Grum pit. To ensure a sustainable water management solution for 2013 and beyond, a pump and 8" HDPE pipeline will be installed to transfer water from Grum pit to Vangorda pit, and a dredging system will be installed over the polishing pond so that sludge removal can occur intermittently during the treatment season. Sludge will be pumped out of the pond and flow by gravity down the 24" influent pipeline to Vangorda pit for subaqueous disposal. These upgrades will allow for seasonal treatment of 750,000 to 900,000 m^3 of contaminated water, and will provide excess treatment capacity should extreme storm or flood events necessitate emergency pit dewatering. This paper reviews the upgrade requirements and compares the possible alternatives based on their cost, time required to implement, and site-specific operational parameters.

INTRODUCTION

This is a case study of a contaminated water management strategy that was developed for mine infrastructure on the Vangorda Plateau at the Faro Mine Complex. The Vangorda Plateau is situated 6 km upstream of the Town of Faro in the Vangorda Creek regional catchment. The importance of Vangorda Creek as a water resource is exemplified by its role in providing recharge to the town's municipal water supply, and in providing fish habitat between its confluence with the Pelly River and Vangorda Creek falls, which are situated on the upstream margin of the town. Thus, protecting Vangorda Creek from mine impacted water is crucial for protecting the environment, and human health & safety.

BACKGROUND

The Faro Mine Complex (FMC) is an abandoned lead-zinc-silver mine located in mountainous, subarctic Yukon, Canada, at 62°N latitude in a cold climate. The FMC developed around three open pits that are geographically divided into two areas connected by a 14 km haul road (Figure 1). Mining of the Faro pit began in 1968, whereas mining of the Grum and Vangorda pits started in the early 1990s. Operations ceased in 1998 due to owner bankruptcy. The FMC is now managed by the Government of Yukon, with funding provided by the Government of Canada. The site is currently in a state of care and maintenance while engineering and design work is undertaken to support remediation plans.

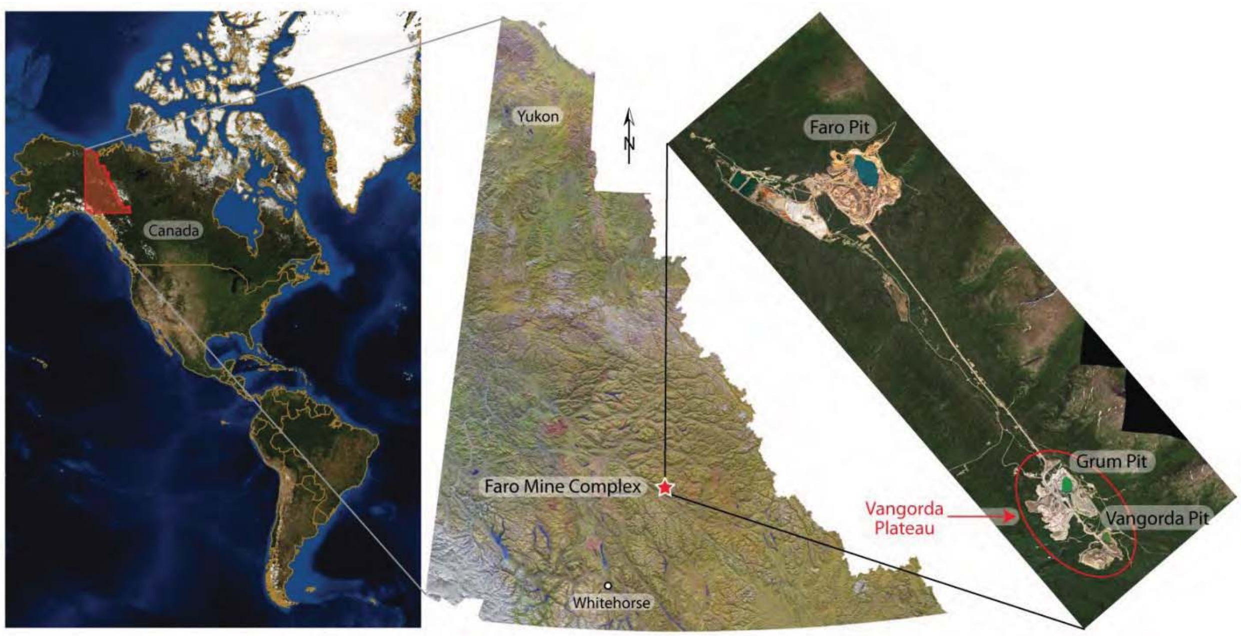

Figure 1. *Location of the Vangorda Plateau at the Faro Mine Complex*

The main ongoing challenge at the FMC is water management. The goal is to prevent contaminated water from entering the receiving environment, and preventing fresh water from mixing with contaminated water. A key component of the contaminated water management strategy is seasonal dewatering and lime-treatment of acidic and/or metal-contaminated water from pits and ponds. This creates ample storage capacity in water containment areas to safely manage natural inputs of atmospheric precipitation and snowmelt, thus preventing noncompliant discharges. Annually, effective water management is most crucial during freshet and occasional extreme precipitation events. It is imperative that excess storage capacity also be maintained to protect against low probability, large magnitude floods and other geohazards that may only occur on a decadal scale or longer.

A subarctic continental climate, where daytime highs remain below freezing for 40% of the year, necessitates that all water management activities be conducted in the short summer season when pits and ponds are ice-free. Mean annual air temperature is approximately -5°C with a range of mean monthly temperatures from -30°C in January to 20°C in July, however, pit lake surfaces commonly remain ice-covered well into June. Despite the region receiving only moderate amounts of precipitation (mean of 300 mm, up to 400 mm in a wet year), rapid snowmelt in May or June regularly creates conditions where excess amounts of surface water must be managed in a short period of time, commonly when variably thawed ground creates difficult working conditions.

VANGORDA PLATEAU SITE CONDITIONS

Major mine infrastructure features on the Vangorda Plateau that must be considered in a contaminated water management strategy are Vangorda pit (Figure 2A), Grum pit (Figure 2B), the Vangorda Water Treatment Plant (Figure 2C), and the Water Treatment Plant (WTP) polishing pond (Figure 2C). The WTP (1307 metres above sea level (masl)) and polishing pond (1301 masl) are at a higher elevation than Grum pit (maximum recommended elevation 1213.4 masl) and Vangorda pit (maximum recommended elevation 1091.8 masl). In addition to inputs of clean surface water that cannot be diverted, the pits are used as repositories for other sources of contaminated water prior to being processed at the WTP (Figures 3 and 4). In a normal treatment season, Grum and Vangorda pits are respectively anticipated to contribute 400,000 m^3 and 350,000 to 500,000 m^3 of water to the WTP.

With up to 6 mg/L zinc (at pH 7.5 to 9), Grum pit water is less contaminated than Vangorda pit water (up to 235 mg/L Zn at pH 3 to 5). Water in both pits, however, is above the discharge limit of 0.5 mg/L zinc, necessitating treatment prior to release. The WTP was constructed in 1992 to treat Vangorda pit water, whereas a passive microbiological treatment program was tested on Grum pit. The passive treatment program was discontinued in 2012 when the dewatering trigger elevation was reached and Grum pit water was still noncompliant.

RISK MITIGATION

Pit dewatering and treatment reduces the risk of noncompliant discharges of contaminated water to the receiving environment. If pit water elevations are allowed to rise unchecked, the pits may cease to be groundwater sinks, thus allowing contaminated pit water to enter groundwater, particularly at the bedrock-overburden contact. At Grum pit, a portion of the north pit wall is mainly formed of unstable till that progressively slumps and fails. A catastrophic failure of this material could create a large wave capable of overtopping the pit, thereby releasing contaminated water and potentially creating a health and safety risk to individuals working downstream.

Figure 2. Major mine infrastructure at the Vangorda Plateau. A) Vangorda pit looking southeast. B) Grum pit looking southwest. C) Polishing pond at the Vangorda Water Treatment Plant. D) Terminus of the Vangorda Creek Diversion. Vangorda pit visible in upper right.

To allow for open pit excavations, diversions were constructed during mine development in order to route creeks around the pits and prevent pit flooding. During implementation of the final closure plan, larger, more robust diversions will be constructed in new locations farther away from the pit walls to reduce the probability of failure. Ample storage capacity must be maintained in pits to accommodate flood inflows in the event of diversion failure. The North East Interceptor Ditch prevents a small creek from entering Grum pit, whereas the Vangorda Creek Diversion (VCD) prevents much larger flows from entering Vangorda pit. The Vangorda deposit was the first ore body discovered at the FMC, due to it being incised by, and exposed along Vangorda Creek. The VCD was constructed in 1991 to route Vangorda Creek around the northern edge of the pit (Figures 2D and 3). The VCD was sized for a 1-in-100 year flood event (10 m^3/s) and had a design life of 10 to 15 years. It is already 10 years past its intended lifespan and is estimated to have passed three 1-in-100 year flood events since 1998. A complete failure of the VCD near the start of a 1-in-1000 year flood event with sustained flows could cause Vangorda pit to fill and overtop to Vangorda Creek in 7 days, whereas a 1-in-100 year event could causing filling in 16 days. Dilution of Vangorda pit water by flood waters would be unlikely to prevent a noncompliant discharge. The VCD will eventually be reconstructed in a new location to ensure long-term environmental protection. Until a new diversion is constructed, however, the contaminated water management strategy must allow for enough Vangorda pit dewatering capacity to accommodate flows that could result from diversion failure.

Figure 3. Aerial image of Vangorda Plateau indicating locations of mine components. Important mine components considered in the development of the contaminated water management strategy are circled in red.

DESIGN CONSIDERATIONS

Updating the contaminated water management strategy for the Vangorda Plateau for 2013 was driven by two main factors: 1) Grum Pit dewatering (and associated treatment) must commence in 2013, and 2) the WTP could not treat the additional water from Grum Pit because of limited sludge storage capacity in the polishing pond. Based on these considerations, options were developed for Grum Pit dewatering (Table 1) and treatment sludge management (Table 2).

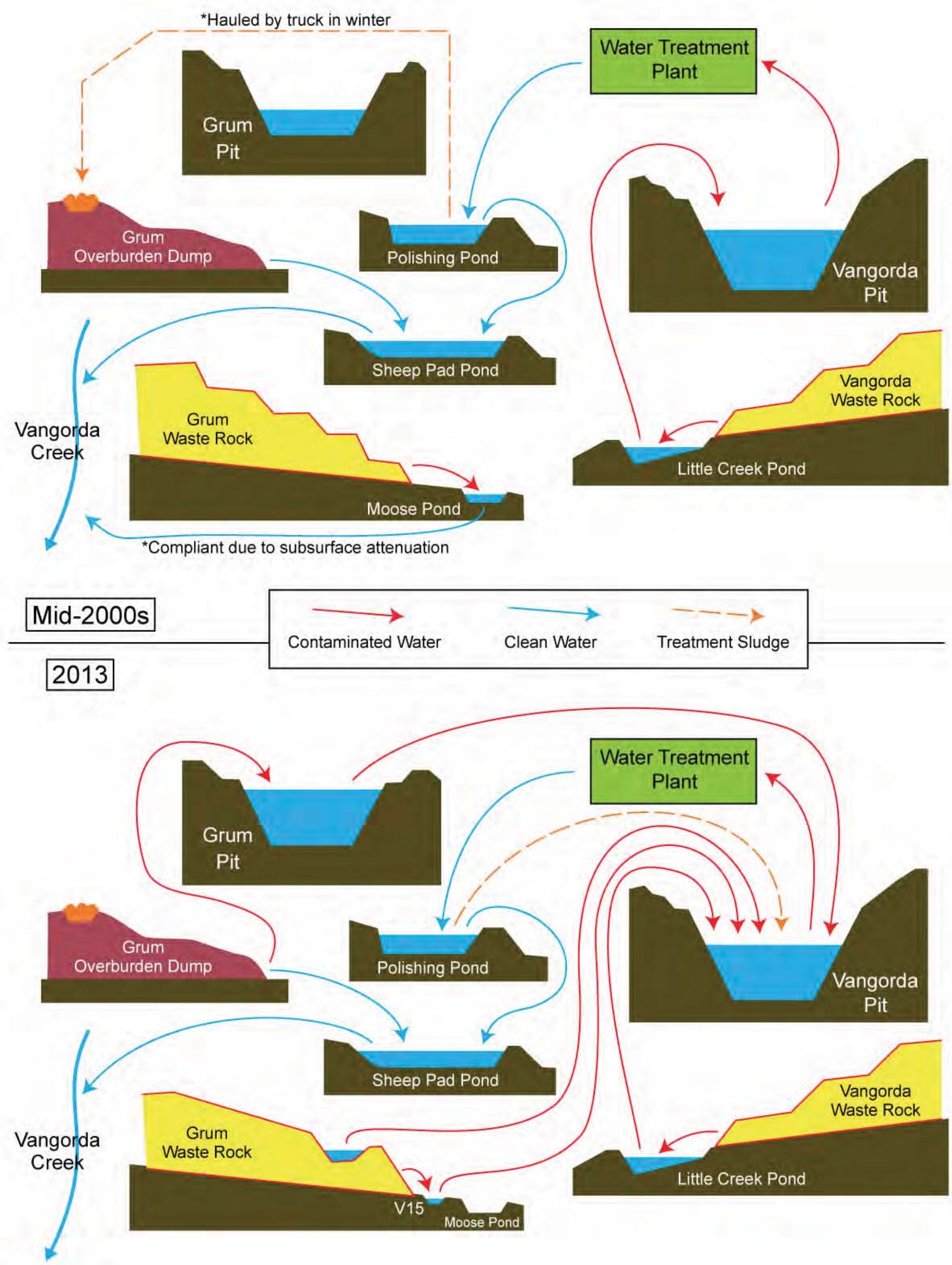

Figure 4. Schematic showing Vangorda Plateau contaminated water management strategy before the upgrades implemented in 2013 (top) and after (bottom).

Table 1. Grum pit dewatering options

Option	Description	Advantages	Disadvantages
1	Pump directly to WTP	1) Slightly shorter pipeline distance 2) Stainless steel pump not required (cost savings)	1) High pressure pipeline required 2) High head pump required 3) Increased lime consumption due to lower metal load in Grum pit water 4) Must configure WTP for two geochemically distinct influent sources 5) Difficult pipeline routing
2	Pump to Vangorda pit	1) Single influent source to WTP simplifies treatment process 2) Expensive stainless steel pump not required 3) Cheaper, lower head pump than Option 1 4) Most of the flow path is downhill, thus high pressure pipeline only required for first 800 m to crest of Grum pit, then low pressure pipeline to Vangorda pit	1) Slightly longer pipeline distance 2) Increased electricity consumption and component/pump wear due to double handling of Grum pit water
3	Pump Vangorda pit to Grum pit, Grum pit to WTP	1) Single influent source to WTP 2) Allows Vangorda pit water to be pumped to Grum pit for emergency storage 3) Compatible with long-term closure option (in 10+ years) to treat all water on the Faro side of the property; allows for staging of Vangorda Plateau contaminated water in Grum pit	1) Expensive high head stainless steel pump required in Grum pit 2) High pressure pipeline required from Grum pit to WTP 3) Requires expensive and cumbersome reconfiguration and re-routing of large-diameter pipeline currently in use 4) Acceleration of water quality degradation in Grum pit due to input of acidic, highly metal-contaminated Vangorda pit water to circum-neutral pH Grum pit water

Table 2. Treatment sludge management options

Option	Description	Advantages	Disadvantages
1 A	Excavate and haul sludge to Grum Overburden Dump	1) Short haul distance 2) No design, procurement, construction, installation or commissioning of equipment required	1) Sludge cannot be removed from polishing pond in summer 2) Contaminant-containing sludge placed in a cell excavated on clean borrow material that will be used for waste cover system construction 3) Limited storage capacity in current sludge cell, thus new cell would need to be excavated 4) Labour-intensive and time-consuming to bulk sludge in winter with excavator to promote freezing, then haul nearly 1000 dump truck loads 5) Sludge must be consolidated after dewatering in summer to maximize storage capacity in sludge cell
1 B	Excavate and haul sludge to Grum pit slot cut	1) Abundant long-term storage capacity 2) No sludge storage on clean borrow source	1) Sludge cannot be removed from polishing pond in summer 2) Labour-intensive and expensive to bulk sludge in winter with excavator to promote freezing, then haul nearly 1000 dump truck loads 3) Extensive and expensive preparation work required to dewater Grum slot cut and construct truck access 4) Slightly longer haul distance
1 C	Excavate, haul and place sludge on ice-covered pit lake (assume Grum pit due to close proximity to WTP)	1) Abundant long-term storage capacity 2) No sludge storage on clean borrow source 3) Rapid implementation possible (less preparation than Option 1B) 4) Sludge settles to pit bottom upon melting of pit lake ice cover	1) Sludge cannot be removed from polishing pond in summer 2) Annual design, construction and monitoring of pit lake ice road 3) Increased risk to worker health and safety by driving heavy equipment on ice-covered pit lake 4) Labour-intensive and expensive to bulk sludge in winter with excavator to promote freezing, then haul nearly 1000 dump truck loads 5) Slightly longer haul distance

Option	Description	Advantages	Disadvantages
2 A	Dredge pond + pump sludge to Grum pit	1) Sludge removal can be completed in summer to increase treatment capacity and extend treatment season 2) Once installed, system is less labour-intensive than excavating and hauling 3) Abundant long-term storage capacity 4) No sludge storage on clean borrow source	1) Design, procurement, construction, installation and commissioning of dredging equipment required 2) Design, procurement and installation of sludge pipeline to pit required
2 B	Dredge pond + pump sludge to Vangorda pit	1) Sludge removal can be completed in summer to increase treatment capacity and extend treatment season 2) Once installed, system is less labour-intensive than excavating and hauling 3) With minor modifications, can use existing influent pipeline for directing sludge to Vangorda Pit 4) Abundant long-term storage capacity 5) No sludge storage on clean borrow source	1) Design, procurement, construction, installation and commissioning of dredging equipment required 2) WTP must be stopped for up to two weeks to allow sludge to be passed through the influent pipeline
3	Upgrade WTP to High Density Sludge from Low Density Sludge System	1) Installation of a thickener eliminates the need to remove sludge from the polishing pond 2) Provides robust, long-term solution (10+ years)	1) Most expensive option 2) Cannot be implemented in time for 2013 treatment season 3) New pipeline required to Grum pit for sludge disposal

OPTION SELECTION

For the Grum pit dewatering project, Option 2 was selected as the best solution because it can be implemented rapidly, has the lowest cost, and is the easiest to integrate with the current contaminated water management systems. For the treatment sludge management project, Option 2 B was selected as the best solution because it could be implemented quickly at a reasonable cost while enabling treatment sludge to be removed from the polishing pond during the treatment season.

DISCUSSION AND CONCLUSIONS

At the time of writing, the Grum pit dewatering system was complete and had just begun to transfer water to Vangorda pit. The polishing pond dredge system was on site, but had not yet been utilized for transferring sludge to Vangorda pit because maximum sludge storage capacity in the pond had not yet been reached. The effectiveness of the completed Vangorda Plateau contaminated water management system upgrades will be evaluated at the end of the 2013 water treatment season in October.

DEFINING DISTURBANCE AND RECOVERY: THE INFLUENCE OF LANDSCAPE SPECIFIC ECOLOGICAL RESPONSES TO OIL AND GAS LINEAR DISTURBANCES IN NORTH YUKON

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ABSTRACT

Across northern Canada evidence of oil and gas seismic exploration remains from the 1950's to current day. While many of these linear features are still visible, others can no longer be seen. Research carried out by Yukon Government, Energy Mines and Resources looked at the status of historical oil and gas exploration disturbances and recovery. A key question was the definition of the words "disturbance" and "recovery". What do we use for criteria in the determination of whether a site is disturbed or not, and when do we consider that a site has recovered? What variables are important to measure? A key tool that was considered was a comparison with natural disturbance regimes in the study area. Does natural recovery depend on the disturbance conditions under which the lines were constructed or is the limiting factor, or filter, the dominant ecological process in the area in which the exploration was carried out. Our findings indicate that disturbance factors include changes to permafrost, soil structure and mycorrhizal dynamics in combination with propagule removal, community shifts to graminoid dominance and nutrient availability. The dominant ecological processes are permafrost, available nutrients and temperature.

Key Words: Seismic, Succession, Permafrost, Nutrients, Active Layer, Compaction.

DESCRIPTION OF PROJECT

A Seismic Line Disturbance and Recovery Research Study was initiated in 2006 by Yukon Government, Energy Mines and Resources (EMR). North Yukon field work was carried out in the Eagle Plains area in 2006 and 2007 and in the Peel Plateau area in 2007. Further field work was carried out in the south-east Yukon and south-west NWT in 2008. This paper focuses on the results of the north Yukon research.

A number of assumptions and questions were on the table when we started this work and we wanted to:

- Assess the current status of historical linear feature recovery in the two current Yukon planning areas (North Yukon and Peel Watershed) and SE Yukon
- Understand if we could use remote imagery to understand and monitor the status of linear disturbance or recovery
- Understand the variation and dynamics of recovery in terms of natural succession
- Understand and support cumulative effects management as it relates to exploration activities, particularly in reference to the use of a thresholds approach to land management
- Determine the appropriateness of certain mitigation measures and best practices (e.g., line width, the re-use of linear features, snow depth, etc.) to provide advice to operators about appropriate mitigation and monitoring strategies or end of project life expectations.

A barrier to answering these questions was the lack of a definition for "Disturbance" and "Recovery" in this context. For some the perception of disturbance (i.e., being able to see a line from the air) is enough to call it a disturbance. For others the presence of woody vegetation at least 1.5 m high (North Yukon Land Use Plan 2009; NYLUP) defines recovery. Without clearly defined terms we cannot "grow" disturbances on and off the landscape.

METHODOLOGY

This study considered climate, hydrology, site chemistry, soil material, biotic factors, fire history and physical factors. Sites were pre-selected from a number of years of aerial survey and detailed analysis of photos and air photos. Information on age and history of linear disturbance using a variety of sources including air photos, National Energy Board data records, topographical maps and satellite images was collected. Unmapped features found on air photos and satellite images were hand plotted and aged.

Criteria for site selection included securing information from a full range of historical seismic line ages. Selection was also made on the basis of securing data from a range of landscape positions as well as mineral soil dominant vs. peat/organic dominant landscapes. Some bias was introduced in the field where pre-selected treed study sites were not helicopter accessible.

In the field, information was collected on the presence and depth of permafrost; active layer depth; soil structure, moisture and texture; nutrient availability; depth and composition of peat; vegetation abundance and composition broken down into floristic communities; treed community, tree structure and age; presence of fire; and landscape position (aspect and orientation to sun and prevailing winds, potential for snowfall accumulation due to geography and topography, etc.).

Analysis of the vegetation data was conducted post-field using a tabular sorting method that groups study plots with similar species composition together into groups which are then used to define community types. Soil texturing and tree ring aging was also done post-field in the EMR geoscience and forestry labs.

Physical characteristics of disturbance noted included feature type (winter road, trail, seismic line, well site, airstrip, camp, staging area), width and depth of disturbance, presence and height of windrows and age of disturbance. Other observations were made on a site by site basis and included evidence of human activity such as camps and snowmobile trails. Wildlife observations were also made but were casual in nature and included presence of birds and mammals, habitat use (e.g., vole tunnels, nests), defined travel corridors, discarded antlers, evidence of predation or herbivory and scat. Scat was analyzed post-field by the Yukon Department of Environment carnivore biologist.

STUDY AREA DESCRIPTION

Our study area was in the Eagle Plains, British Mountains and Peel River Plateau ecoregions (Smith 2004). Permafrost is ubiquitous at this latitude and only high rock and gravel ridges and water bodies are permafrost free. The geophysical histories of these three ecoregions set the stage for the present vegetation communities found across the landscape and the study location offered a unique opportunity to consider linear disturbance response on both the beringian and non-beringian sides of the British Mountains.

All control sites in the study area consisted of an open black spruce-Labrador tea forest in different moisture phases. The primary community type was *Betula-Rubus* representing a richer phase of the black spruce-Labrador tea forests followed by *Betula-Polytrichum* representing a drier phase of the black Spruce-Labrador tea forests and influenced by elevation (Polster 2009).

Fire is the dominant disturbance regime in this area along with rapid landscape change resulting from permafrost slumping. The black spruce-Labrador tea forest common to this area recovers primarily through the regeneration of shrubby species such as Labrador tea, lingonberry and cloudberry from underground root systems that are not damaged (Polster 2009). Where a recent fire (2004 and 2005) overlapped our study area, fire recovery processes dominate and is reflected in the vegetation.

Active-layer thickness was very homogeneous averaging 45 cm in undisturbed (anthropogenically or naturally) sites (Table 1). Burned areas such as site 2007-09 responded to fire with increased active layer thickness. This is consistent with long term active-layer records that have been collected in the NWT and north Yukon (Mackay 1995). Mackay's data indicate that fire disturbance and subsequent ecological recovery had a dominant effect on active-layer thicknesses.

Table 1. Active layer depth

Sample ID	Feature type	Vegetation community	Depth to permafrost
07-001e/5	well	8	0.575
07-006	road/burn	8	0.8
07-007	winter road	8	0.9
07-009	seismic line/burn	8	0.6
07-009a	control/burn	8	0.6
07-010b	winter road	8	0.65
07-014	road	8	0.9
07-017	road	8	1.1
07-018	well	7	0.9
07-019	staging site	7	0.725
07-020	staging area	7	0.6
07-021	staging area/road	7	0.5
07-024	trail	3	0.8
07-026	trail	5	0.6
07-031	well	8	Refusal at 30
07-035	well	8	0.6

EVOLVING TIMES AND EVOLVING TECHNOLOGY

Exploration was conducted as early as 1944 and until the early 70's this exploration took place in the summer months. In the Eagle Plain and Peel Plateau areas surface exploration commenced in the mid-1950s, with 92% of the seismic lines shot before 1975 (Hannigan 2001). The presence of tracked and/or wheeled vehicles in this vulnerable summer landscape and the application of such methods as stripping the active layer and organic overburden down to permafrost or mineral soil to facilitate summer travel resulted in extensive damage to vegetation and permafrost.

There was little to no regulation of land use activities until 1972 when regulations under the *Territorial Lands Act* came into force and were applied to this region. Until then companies only reported to the National Energy Board in Calgary regarding the successful acquisition of seismic data but there was no requirement to report on the physical disturbance resulting from the seismic lines, access trails and camps.

Seismic survey methods of the day were based on trigonometry and triangulation methods and relied on long line-of-sight lines to delineate the seismic program. These methods were used until the development of the global navigation satellite system in the 1980s allowed for geo-spatial positioning instead.

Line-of-sight methods usually involved heavy equipment that operated with the blade down to strip down to permafrost thereby destroying the insulation on the line and leaving large windrows on either side of the disturbance. In forested areas, cutlines were traditionally cleared using a bulldozer to pull up roots, stumps and woody materials which were then pushed into windrows. The remoteness of this area necessitated “cat trains” and mobile camps to support the crews on the land – with this equipment often running parallel to the seismic programs.

Seismic lines, winter trails and winter roads were all established along with well sites, camps, staging areas and airstrips. Single-pass winter trails were used for convoys of tracked heavy-axled vehicles. Natural frost penetration in the ground and snow was relied upon to support their weight and there was no prior surface preparation. With the surface organic layer remaining intact, the active layer on winter trails was generally not affected (Kemper and Macdonald 2009a, b).

In contrast, winter roads were built for repeat winter traffic by heavy-axled wheeled vehicles, including haul trucks. All woody vegetation was cleared, and each winter, a load-bearing road bed was prepared with packed snow or ice. Winter roads caused mechanical damage to vegetation (Adam and Hernandez 1977; Bliss and Wein 1972; Felix and Raynolds 1989a), and if the depth of frozen soil was insufficient, they disturbed or compacted the surface organic layers (Bliss and Wein 1972; Felix and Raynolds 1989b) which led to increased depth of the active layer (Bliss and Wein 1972; Forbes et al. 2001; Haag and Bliss 1974; Hernandez 1973). The different kinds of disturbance are reflected in the status of these features.

Since the 1990’s, evolving footprint minimization approaches have been developed and are now utilized in the seismic industry. Low Impact Survey (LIS) techniques have been introduced to minimize disturbance – in particular these techniques narrowed seismic line widths, reduced the loss of merchantable forest, and minimized disturbance of the soil and ground cover (CAGC 2011). Low impact seismic access lines are now being cut as narrow as 1.75 m, smaller “enviro” drills are capable of traversing narrow lines to drill shot holes and the use of geo-positioning technologies allows operators to establish a meandering seismic trail between shot points which reduces the line-of-sight (CAPP 2004).

All this is to say that while the understanding of the dynamics behind historical seismic disturbance and defining what we mean by disturbance and recovery is important from a land management, cumulative effects and prescriptive mitigation perspective, the severity of early disturbances should not be a reflection on current practices. However, the intensity of activity in the north Yukon, while currently low, has the potential to grow. The complex geology and anticipated high exploration risks associated with all exploration plays in the Eagle Plains area suggest that considerable amounts of new seismic data and more exploration wells may be required to properly evaluate the region’s hydrocarbon potential

(Hannigan 2001). Therefore lessons learned from the past will help the industry move forward and continue to develop and implement strategies to reduce the environmental impact of their operations.

RESULTS

Disturbance Type and Response

Our findings broke disturbance down into three types as follows:

Type 1 disturbance

- No or very limited disturbance such as a meandering snowmobile track
- Hand-cut lines
- Travel over frozen wetlands
- Blade kept high with understory and ground surface protected by sufficient snow cover and frozen ground
- Disturbance dynamics mimicked natural environment (e.g., line width same width as natural tree spacing)
- Natural disturbances had re-set the successional trajectory

No recovery is required to "grow" Type 1 disturbance conditions off the landscape. No permafrost or soil response was noted and floristic communities and forest structure were the same both on and off the line. Where natural disturbance such as fire overlapped with oil and gas disturbance the natural successional processes were re-set and the seismic line disappeared. This was evidenced by identical disturbance recovery processes being observed both on and off line. In addition, as per the NYLUP these sites do not facilitate travel or access by wildlife or people.

Approximately one third of our study plots fell into this category. As a note, this finding disagrees with similar studies in the NWT (Kemper 2006). This is likely due to the very poor historical seismic data record (Smith and Groenewegen 2008) in the NWT. Without accurate map-based records of historical activity, researchers would not be able to find historical features that had either grown off the landscape or that had not actually disturbed the landscape at all thus investigating only lines that were still visible. As well, given the distinct difference in disturbance response to different kinds of oil and gas features in our study area, I suggest that there may not have been a clear differentiation between what was a road, trail and or seismic line in some studies.

Type 2 disturbance

- Removal of trees only on the line with limited disturbance to shrubs and understory
- Low level disturbance with the some surface disturbance to the duff but the organic horizon left largely intact and propagules, in particular suckering roots, unaffected
- Some crushing of vegetation and slight (<30 cm) depression along line

Under these conditions locally consistent early pioneering seral species were established and sites were moving along the appropriate successional recovery trajectory. Approximately one third of our study plots fell into this category. Disturbance recovery was compared with natural disturbance recovery and the same processes were in play as for natural disturbance recovery. Recovery timing was consistent with the level of disturbance. The NYLUP suggests that human-caused surface disturbance is considered recovered when it no longer facilitates travel or access by wildlife and people. All the historical airstrips fell into the Type 2 category as did most of the seismic lines and winter trails. Winter roads, well sites, staging areas and camps fell into Type 3 disturbance.

Type 3 disturbance

- Permafrost exposed
- Active layer removed or thinned
- Mineral soil stripped and exposed
- Mixed soil horizons
- Agronomic grass seeded
- Erosion and thaw failures
- Significant rutting or compaction leading to subsidence (>30 cm) of ground
- Scalping of cottongrass (*Eriophorum*) or sedge (*Carex* spp.) tussocks

Photo 1 and 2. Study Plot 2007-17. Calamagrostis successional stagnation and normal successional processes alongside in windrows

Response to disturbance Type 3 was significant, outside of the range of natural disturbance recovery and often modified to a locally inconsistent successional trajectory. Stripping and compaction resulted in or contributed to cool moist conditions, elevated water table, active layer deepening and graminoid successional stagnation (primarily *Calamagrostis*). Seeded agronomic species such as Kentucky bluegrass also caused successional stagnation. Both types of graminoid succession/stagnation response were remarkably resilient to change – with some disturbances over 50 years old while less disturbed same-age sites alongside them were recovering appropriately under a natural successional recovery curve (Photo 1 and 2). Horizon mixing would also tend to shift the community into a disturbance dynamic rather than a recovery dynamic.

Retrogressive succession or the setting back in time of disturbance occurred when peatlands (forested and non-forested) were stripped down to permafrost creating long linear fen like wetlands (Photo 3). Peatlands offer a particularly interesting response to disturbance. When disturbed, peatland becomes altered to present fresh physiographic conditions, its materials undergo change and it becomes endowed with a new set of thermal, hydrological, structural and chemical characteristics (Radforth 1977). Often these changes are irreversible. Natural disturbance (e.g., fire) overlain on Type 3 disturbance was not able to re-set the disturbance (Photo 4).

Photo 3. Retrogressive successional response.
Photo 4. Linear fen disturbance cannot be re-set by fire

The NYLUP suggests that in forested areas, a feature can be considered recovered when it contains woody vegetation (trees and shrubs) approximately 1.5 m in height. This may need to be re-considered as some Type 3 disturbed sites have recovered to this height or greater – but have done so in a modified manner presenting vegetated conditions anomalous to the region.

DISCUSSION

In this region recovery is reflective of slow rates of colonization as well as a low resource base that is associated with low temperatures, a short growing season and slow rates of decomposition and nutrient turnover (Bliss and Matveyeva 1992; Nadelhoffer et al. 1992). Dominant factors behind response to disturbance are discussed below.

Permafrost

The landscape and permafrost are very closely related and the degree of initial disturbance is an important control on the extent of permafrost thaw and thus the overall recovery of the linear disturbance. The presence of permafrost greatly increases the complexity of ecological responses to disturbance, due to feedbacks between soil topography, hydrology, and ground ice. Initial minor thaw settlement caused by disturbance can lead to water impoundment, decreased albedo, and increased heat flux, which in turn can cause more thaw settlement (Lawson 1986).

If a disturbance causes substantial changes to edaphic conditions (e.g., thermal regime, permafrost, hydrology, nutrient cycling) this will, in turn, influence community re-development (Chapin and Shaver 1981). The recovery of the pre-disturbance permafrost condition is considered a necessary pre-requisite for return of plant communities to their pre-disturbance state (Lawson 1986; Shirazi et al. 1998; Walker et al. 1987; Walker and Walker 1991). In all Type 3 disturbances in our study, permafrost degradation was implicated (Table 1).

Soil Structure and Mycorrhizal Propagules

Recovery may be impaired by the removal of propagules during soil stripping and horizon mixing. The removal of propagules, especially suckering roots, reduces a site's ability to revegetate naturally (Osko and Glasgow 2010).

Most forest plants are dependent on mycorrhizal fungi for establishment and productivity (Janos 1980) and mycorrhizal propagules are the primary mode of forest regeneration (Brundrett et al. 1996a). Sources of mycorrhizal propagules include old roots and spores (Brundrett and Kendrick 1988). Light, temperature and moisture are all factors known to influence mycorrhizal colonization and stripping and mixing soil disrupts old root systems and mycelia networks (Stottlemyer et al. 2009). Existing vegetation plays an important role as refugia for the mycorrhizal fungi and direct and indirect damage to propagules or changes in soil chemistry (Buchholz and Gallagher 1982; Klopatek et al. 1988) changes the mycorrhizal dynamics (Janos 1980).

If old roots and mycelia are the major sources by which mycorrhizal colonization is initiated, it is likely that the pre-disturbance vegetation community determines the type and amount of mycorrhizal fungi available to regenerating plants (Stottlemyer et al. 2008). Mycorrhizal fungi therefor have the potential to influence the trajectory of vegetation succession after a disturbance. On disturbance Type 3 sites where the successional trajectory has deviated to a new community type this is likely a factor.

The Roles of Graminoids in Influencing Vegetation Succession

The shift to a *Calamagrostis* community type was defined by *Calamagrostis canadensis*. This community was found in areas where clearing and grading had removed the organic surface horizons and left fine textured, moisture-retaining mineral soil that supports dense stands of *Calamagrostis*, limiting establishment of other species, hence the very low average number of species in this community type.

Only one of our study sites had been re-seeded (2007-34). This site was the largest overall disturbance covering a number of hectares and consisted of a well site, camp, staging site and airstrip as well as seismic lines. Kentucky bluegrass had been planted at some point in the 1970's and the site was still grass dominated. Agronomic grasses as well as natural grasses and sedges will act to prevent seedling establishment by pioneering species, thus creating a successionally stagnant environment.

This finding is supported by the work of Forbes (1999) who looked at the introduction of surrogate non-native species (graminoids) to hasten the establishment of plant cover. Osko and Glasgow (2010) found that at Alberta well sites the removal of propagules, especially suckering roots, promotes the dominance of the site by grass. Forbes and Jefferies (1999) work on revegetation of arctic sites also found that the prevalence of mineral rich soils at disturbed sites can limit the re-establishment of species which originally occurred at sites and promote the dominance of grasses.

Interestingly, grasses are the least dependent on mycorrhizal colonization of all mycorrhizal plants (Janos 1980) so able to establish much more independently than other species.

Compaction and Density

A marked difference of density (compaction) relative to undisturbed areas was observed in a number of our plots, none of which were seismic lines. Three road sites were particularly interesting. Site 2007-17 had the most intense historical activity and greatest extent of disturbance and included a well site, airstrip, seismic lines and multiple roads. This access road site was stripped to mineral soil and highly compacted. Permafrost depth was beyond the reach of our 1.1 m rod, contrasting with the average 0.45 m depths found at control plots, seismic lines and airstrips. The floristic conditions at site 2007-17 had shifted from an older mesic black spruce-Labrador tea forest to a *Calamagrostis*-based community. This road was also oriented ENE which would have contributed to the cool conditions. Combining increased soil density

Photo 5. Site 2007-17 with normal succession on airstrip. Photo 6. Site 2007- 24 Accelerated succession due to compaction

with increased clay content and poor soil structure (Osko and Glasgow 2010) can perpetuate cooler, moister conditions that prevent colonization by the adjacent forest and encourage colonization by grasses.

Adjacent to the road, the windrowed disturbance, the seismic lines and the airstrip were recovered (Photo 5) floristically back to a black spruce-Labrador tea forest although structural recovery (tree height) was not yet complete.

Sites 2007-24 and 2007-26 also exhibited a response to compaction (Photo 6). These were both winter trails on the Peel Plateau side of our study area. These sites were not disturbed in the conventional sense and no stripping or other soil or direct vegetation disturbance had been carried out. The sites were however slightly depressed from the weight of equipment that had traveled over them and some crushing of vegetation had occurred. A noticeable change in soil density and compaction was observed on the lines. The result of this disturbance was an increase in permafrost depth and an accelerated willow regrowth caused, presumably, by the combination of a nutrient flush from the damaged vegetation and some warming caused by the thickening of the active layer. There was no shift in community at site 2007-26 and a small shift at site 2007-24.

Nutrient Availability

The supply of available nitrogen appears to be strongly limiting for plant growth in this region (Bliss and Wein 1971). The ecosystem has, however, evolved an efficient means of recycling nitrogen with available nitrogen values much higher in the organic horizon than in the mineral B or C horizons. Any disturbance resulting in a modification of this layer will result in the disruption of the nitrogen cycling system and have a consequent effect on plant growth and production. In an environment in which nutrients are already limited, total levels of soil phosphorous, nitrogen and potassium may be lowered substantially with the removal of the organic layer (Cargill and Chapin 1987; Mitchell and McKendric 1975).

Nutrients, already low, are also bound in the organic material. Sufficient disturbance will often cause a nutrient flush followed by an accelerated growth of shrubs and woody species such as willow, alder and birch. So called "green-belts" along winter vehicle trails are commonly reported in the literature and increases in primary productivity have been reported in several tundra disturbance studies (Chapin and Shaver 1981; Hernandez 1973; Vavrek et al. 1999). Accelerated nutrient cycling is also thought to be associated with the compression of standing dead or living plant material into contact with the soil decomposer community (Abele et al. 1984; Rickard and Brown 1974), along with increased soil temperature (Chapin and Shaver 1981). Together, these factors increase decomposition, hence improving nutrient availability. Site 2007-24 and 2007-26 are good examples of this response.

Photo 7. Delayed recovery of spruce trees on a circa 1970 line due to insufficient disturbance to initiate a recovery response

Kemper (2006) found that vegetation composition and structure on seismic lines differs from control sites despite no persistent differences in organic layer depth or depth to permafrost. He proposed that this could reflect successional re-development following changes in soil conditions and nutrient availability arising from the disturbance. While this was true of structure and abundance in our study it was not true of composition on seismic lines, though it clearly was on areas of higher disturbance such as winter roads, well sites and staging areas.

Interestingly there were a number of sites in which there was a delayed recovery (photo 7), perhaps because, in this nutrient limited community, there was insufficient disturbance to kick start a recovery process and therefor an insufficient release of nutrients to allow successional processes to initiate.

<u>Winter Roads, Trails and Well Sites</u>

Winter roads are used to access remote communities and resource development camps in the north, yet little is known of their ability to recover after abandonment. Campbell and Bergeron (2012) evaluated the natural recovery of winter roads abandoned within 7 years on peatlands in the Hudson Bay Lowland and found a significantly thinner active layer, lower species richness and changes in species composition. Kemper suggested that although the visual signature of seismic lines was still apparent 18 to 33 years post disturbance, some may have been ice roads associated with exploration activity, which though not well studied, typically have more severe impacts than seismic lines (National Research Council 2003). Our study found an almost 100% correlation between Type 3 disturbance and these types of features (Table 1). The only exception to this was when severe fire had burned through the area (plot 2007-09).

Photo 8 and 9. Fire has re-set the successional processes on these seismic lines

<u>Fire versus Seismic Line</u>

An indication that underlying disturbance factors were not at play in the recovery of some seismic line disturbance were sites where fire had re-set the successional processes on the line to match that of the adjacent burned forest. In one case (plot 2007-09, Photo 8) severe fire damaged the site to the extent that the resulting vegetation both on the line and in the surrounding environment reflected the same early pioneering species. Permafrost depth was equally affected on and off the line. A significant number of other fire- affected sites that were observed from the air exhibited characteristics that indicating that underlying disturbance had been re-set (Photo 9). Not all of these sites could be visited due to burned tree snags preventing helicopter access and so permafrost depth at these sites is unknown.

In these cases clearing of the seismic line would have been done in such a way that the surface soil and vegetation was protected. With the exception of spruce trees either removed or burned there was little difference between the burned seismic line and the adjacent burned forest.

CONCLUSIONS

It is difficult to identify any one factor as responsible for the impairment of growth on these disturbances. Both the nature of the disturbance and the physical and environmental factors at play are very complex and it can be difficult to nail down a single limiting factor or filter to the recovery of these disturbances.

The two factors that appear most significant in terms of measuring disturbance are floristic community type and depth of permafrost or thickness of active layer and each is strongly correlated to the other. Our study, though focused on seismic disturbance, looked at a variety of oil and gas disturbances – well sites, staging areas, airstrips, roads, trails and seismic lines. There was a 100% correlation between type of disturbance and intensity of disturbance with winter roads, well sites and staging areas consistently shifting the floristic community into a disturbance community (not a successional community) and creating significant deepening of the active layer and depth to permafrost.

Our study found limited effects of seismic lines and airstrips on the soil organic layer and density or depth of permafrost 30 to 60 years after disturbance. However we found that historical seismic exploration has led to an increased cover of deciduous shrubs and reduced cover of mosses and lichens. The greater cover of deciduous shrubs on seismic lines can be attributed to their ability to take advantage of short-term increases in nutrient availability; however such effects are likely long-lasting given the high inertia of tundra plant communities. Overall, our results agree with those of Forbes et al. (2001) in that seismic lines had fairly similar floristics to the reference tundra, but the appearance of the vegetation was often quite different because of increases in cover of taller shrubs and pioneering trees such as willow, alder and birch. Comparing the disturbance response to natural disturbance response in the region we can assume that the timing of recovery will be the same. As such we fully expect to see a completion of this successional recovery response back to normal site characteristics as already found at a number of our sites (Photo 10).

Photo 10. Site 2007-1 1970 Airstrip with almost complete recovery
Photo 11. Winter road type 3 disturbance

This is not the case for the highly disturbed winter roads (Photo 11), well sites, camps and staging areas where we found an overall significant difference in vascular plant community composition, soil density, permafrost depth and active layer thickness.

We know that pre-disturbance environmental information is valuable for post-disturbance comparative purposes. The baseline for defining recovery or restoration of disturbance is the surrounding or control conditions – including natural disturbance regimes. These baseline or pre-disturbance conditions however can be even more valuable for prescriptive purposes. For example, knowing the depth of the active layer and water table in combination with soil texture could help predict how susceptible a site might be to post-disturbance domination by *Calamagrostis*, thereby enabling the prescription of preventative measures such as minimization of stripping. Presence, extent and volume of ground ice are important determinants of disturbance response (Lawson 1986; Walker and Walker 1991) and therefore prescriptive mitigation.

Walker et al. (1987) related the magnitude (and reversibility) of impacts on substrates to alternate states of post-disturbance recovery including: (1) complete recovery with succession, where substrates are not permanently changed and the pre-disturbance plant community returns; (2) positive functional recovery, where substrates are either moderately or severely changed and post-disturbance vegetation is dissimilar to the undisturbed condition but is more productive; and (3) negative functional recovery, where the post-disturbance community is dissimilar to, and less productive than, the pre-disturbance type. Our study findings of three types of disturbance and recovery agree with this.

The findings illustrate that the greater the intensity of disturbance the greater the deviation from the natural successional pathway to recovery. Highly disturbed Type 3 sites have a modified response that includes pioneering species such as grasses and sedges. This is significantly different than sites responding to lower level disturbance. Type 2 disturbance can recover via the same processes as do natural disturbances such as fires and rapid landscape change. Given the high inertia of recovering northern plant communities this becomes something of a societal or land management decision. Do we want to permit disturbance that requires the same timescale as a natural disturbance to fully recover? Are we willing to push the perceptual norms by calling a visible site that is appropriately recovering but not quite there yet a recovered site? Structural recovery may require many decades as the growth of trees is very slow in this region. This may mean that the line remains visible for as much as 50 to 70 years even though the site has recovered ecologically and it is called recovered.

The NYLUP has established a disturbance threshold approach for land management in the planning region. In so doing they have recommended that, as human-caused surface disturbances, including linear features, recover through natural re-vegetation or active reclamation, they are subtracted from the total amount of disturbed area. We propose that Type 1 disturbances can be removed from our accounting of disturbance in this region and that Type 2 disturbances can be "grown" off the landscape as appropriate to their age and stage of successional response. Type 3 disturbance will require intervention in order to recover and will otherwise remain on the landscape indeterminately.

REFERENCES - an extensive list of reference is available on request.

RESULTS OF VEGETATION SURVEY AS A PART OF NEUTRALIZING LIME SLUDGE VALORIZATION ASSESSMENT

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ABSTRACT

The use of the lime neutralization process is a common technique to treat acid mine drainage (AMD) generated by sulphidic mine wastes. The AMD-contaminated water collected on the mine site is neutralized by addition of air, lime and flocculant. The sludge generated by the treatment is stored in specifically designed storage ponds. A multidisciplinary project is being conducted to investigate if sludge may be valorized as a material for mine site reclamation. The Doyon-Westwood mine site, operated by IAMGOLD, has two sludge ponds which were colonized by vegetation shortly after sludge deposition. One objective of the study was to investigate metal bioaccumulation risk by plants, to evaluate the sludge's suitability as vegetation growth medium. Ten functional groups of plants were found on the sludge ponds. Arborescent species were among the most frequent vascular plants on the sludge ponds. Chemical analyses of tissues from three arborescent species (leaves, branches, wood and roots) have indicated that the trees growing on the sludge were not affected by the deficit of major nutrients. The metal concentrations in the plant tissues sampled on the sludge ponds fit in the range of those from a control site. Chemical analyses of plant tissues did not reveal any bioaccumulation. The sludge shows a significant potential as a vegetation growth medium.

Key Words: Sludge Valorization, Bioaccumulation, Vegetation Succession, Mine Site Reclamation, Trace Elements.

INTRODUCTION

Lime neutralization is an efficient and worldwide technique used to treat acid mine drainage (AMD) generated by sulphidic mine wastes (MEND 2013). The AMD-contaminated water collected on the mine site is treated by lime addition, which increases alkalinity and promotes precipitation of metal ions as hydroxides. The precipitated metals form a sludge, which also contains significant amounts of gypsum and unreacted lime. This sludge is generally stored in ponds for dewatering and permanent disposal. According to Zinck and Aubé (2000), approximately 7 million cubic metres of sludge are produced annually in Canada. For example, the Doyon-Westwood mine site, operated by IAMGOLD in the Abitibi (Quebec) region, has two closed sludge ponds. The observation of natural vegetation succession on the sludge ponds inspired a research project, which consists of evaluating the possibility of using the sludge as a material for mine site rehabilitation. No studies of the vegetation succession on the sludge and sludge effects on living organisms, particularly focused on the bioaccumulation risks, have been conducted previously. The potential use of sludge as a reclamation material highlights the need to investigate these questions. A multidisciplinary team of researchers from URSTM – Unité de recherche et de services en technologie minérale (UQAT), which includes specialists in the fields of geochemistry, geotechnique, hydrogeology and biology, is involved in this project.

For the projected usage of sludge as rehabilitation material, it was necessary to determine if sludge as a growth medium has an effect on plant status and contaminant bioaccumulation. Indeed, if the vegetation starts growing naturally on the sludge a few years after deposition in the ponds, this indicates that sludge may be a suitable substrate for plant growth. Also, it was observed that the sludge ponds are actively used by local fauna. However, sludge contains considerable concentrations of chemical elements (e.g., As, Co, Cd, Cu, Ni and Zn), that might be harmful for living organisms (Adriano 2001).

The principal objective of the part of the project herein presented focused on the evaluation of the influence of the sludge on the spontaneous vegetation development. The specific objectives were (1) to survey vegetation on the sludge storage ponds (SSP), focusing on arborescent species; and (2) to evaluate the risk of metal bioaccumulation by the vegetation due to the elements present in the sludge that represent a risk for the organisms.

STUDY AREA

IAMGOLD is one of the leading gold mining companies in the Abitibi region; the Doyon-Westwood Mine is a gold producing operation located 40 km east of Rouyn-Noranda, Quebec, Canada (Figure 1). It has been in operation since 1978, and is now the property of IAMGOLD Corporation. Because the ore and wastes (waste rocks and tailings) contain sulphides with minimal neutralising potential, acid mine drainage is produced by the waste rock piles, tailings impoundments, underground mine workings, and site infrastructure (roads) built with AMD-generating rocks.

Figure 1. Study area location (Demers et al., 2010).

All the acidic water is collected and treated using standard high-density sludge process, which includes neutralization of acidic water by lime addition, thereby precipitating metal ions as hydroxides. In the past, AMD neutralization sludge was deposited in two, now closed, dedicated storage ponds. The sludge ponds, filled during the last decade, contain nearly 1 million cubic metres of sludge with calcium (18%), iron (9%) and sulfur (7%) as main components, mostly as gypsum and ettringite (Bouda et al. 2012).

METHODS

To study the influence of the precipitated sludge on the spontaneous vegetation development, the chemical properties of the sludge, a survey of the vegetation and an evaluation of element concentrations in plant tissues were conducted on the northern sludge pond of Doyon-Westwood mine site. The study focused on one of the two closed ponds, i.e., the north SSP.

Chemical Properties of the Sludge in SSP

The chemical properties of the sludge sampled in the north SSP were analysed by ICP-AES, atomic absorption. Concentrations of 60 elements were measured (12 replicates). In this paper, we focus on the concentrations of macroelements (Ca, K, Mg) and of hazardous elements for the organisms in the case of bioaccumulation: As, Cu, Co, Zn. Concentrations provided by MDDEP (Ministère du Développement Durable, de l'Environnement et des Parcs 2013) and Biological Test Method (BTM) (2007) in the forest soils are considered as a reference level (Table 1).

Vegetation Survey

At the end of the fall season in 2011 and 2012, a vegetation survey was conducted. Based on visual observation, two ecotopes (lake and forest) were distinguished on the SSP. The forest ecotope is characterized by dense arborescent vegetation, while the lake ecotope includes three depressions filled with water during periods of excessive precipitation. Also, the lake ecotope is distinguished by characteristic vegetation consisting mainly of cyperaceous species and willows.

Eight transects of 45 m were delimited systematically across the study site. Five transects crossed the forest ecotope. In the lake ecotope three transects were started at the water border along the radius pointing to the centres of each lake. Study plots of 1.3 m in diameter were established along the transects at 5 m intervals. Within these circular plots, floristic composition was identified (Marie-Victorin 1964); species absence (0) or presence (1) was noted. Vegetation composition was assessed by a cluster sampling method (Hoshmand 2006). The occurrence per ecotope was evaluated as the percentage of plots where the species were found (n = 40). Mean occurrence per transect was calculated, and then mean occurrence per zone was assessed. All species documented on site were classified into functional groups (Bloom and Mallik 2006). The species of the same functional group in the boreal zone (the major functional groups are: herbs, graminoids, trees and shrubs, ericaceous and cyperaceous species) share ecological characteristics and play an equivalent role in the community. Also they are characterised by similar growth forms and general life history strategies. Hence, the litter properties, i.e., nutrient concentrations, decomposition rate, etc., of the species belonging to the same functional group are similar (Wardle et al. 1997).

Arborescent species represent a particular interest in this project because they stay for a long period on the site in addition they produce considerable biomass and are often consumed by large herbivores (e.g., moose). Thus, they need to be investigated more meticulously to evaluate the magnitude of heavy metal bioaccumulation in this functional group. To determine the age of the arborescent species establishment on the SSP, these taxa were sampled (discs) in three replicates per taxon at three locations along the SSP. The discs were sanded, and then age was determined by year ring counting.

The characteristics of arborescent species (with basal diameter greater than 1 cm) were documented according to the following variables: height, basal diameter, live status and species abundance per square metre. Based on the results of vegetation survey, three dominant arborescent taxa were identified: balsam poplar, paper birch and willow.

Element Concentrations in Plant Tissue

To evaluate the risk of bioaccumulation, plant tissues of the three tree species were analysed. Plant material was sampled in three replicates in the two ecotopes and in a control site that was selected 10 km away from the SSP. Four plant fractions were sampled: leaves, previous year branches, stems and roots. Concentrations of major nutrients (K, Ca, Mg, P) were measured in leaves and branches, and Ca and Mg in all four fractions. Furthermore, the elements representing a potential risk for the organisms (As, Co, Cu, and Zn) were analysed.

A two-way analysis of variance (ANOVA) with a general linear model (GLM) procedure (Legendre and Legendre 1998) was used to determine the influence of the ecotope and species on the concentrations of Ca, Mg, As, Co, Cu and Zn (SAS 1999). The objective of this analysis is to ensure that there is no significant interaction between ecotope and species. Preliminary tests confirmed that the data respond to the normality requirements. In the analyses, the location of sampling was randomized. The analyses were applied for each fraction separately. Afterwards, Tukey's multiple comparison tests were used to assess the difference in element concentrations in plant tissue between the two ecotopes and control locations for each species and fraction separately.

RESULTS AND DISCUSSION

Chemical Properties of Sludge

A characteristic feature of the SSP is the absence of soil stratigraphy and the absence of the forest floor, hence very low organic matter content. Concerning the major nutrients, Table 1 shows that the total concentration of macroelements (Ca, K, Mg) in the sludge is dramatically low compared to typical forest soil (approximately 80, 2000 and 100 times lower than in the reference soil, respectively). Although these are total concentrations and plant available concentrations do not necessarily correlate with total concentrations, the results allow concluding that the sludge has an oligotrophic status.

Arsenic is one of the elements representing a bioaccumulation risk. The analyses of the sludge show that the concentration of As is below the detection level of the analytical method. Low As concentrations in the sludge allow eliminating the toxic effect of this element for the site studied. Other elements related to the bioaccumulation risk, Co, Cu, and Zn, are present in high concentrations in the sludge compared to the reference soil (by MDDEP and BTM).

Table 1. *Total element concentrations in SSP of Doyon site (ppm)*

Element	Reference concentrations: criteria of MDDEP and BTM	SSP Doyon average (minimum-maximum) concentrations
Ca	963	11.98 (16.60 - 3.43)
K	250	0.14 (0.60 - 0.01)
Mg	192	1.82 (2.24 - 1.50)
As	5	<0.01
Co	20	130.63 (185.00 - 82.30)
Cu	50	598.83 (1030.00 - 297.00)
Zn	120	205.08 (321.00 - 114.00)

Co and Cu concentrations are more than six and ten times higher, respectively, compared to the reference soil, and Zn concentration is twice the reference. These concentrations allow anticipating high plant available forms of these elements, which may lead to their accumulation in plant tissue and high possibility to be translocated along the food chain.

Vegetation Survey

Species belonging to ten functional groups are found on Doyon SSP: arborescent species (trees and shrubs), lycopodiaceae, herbs, graminoids, cyperaceous species, equisetaceae, bryophytes, lichens and mushrooms (occurrence was not documented) (Figure 2). Among these groups, the bryophytes and cyperaceous are the most frequent species on the SSP. Lichens and lyciopodiaceae were found only in the forest ecotope, while thypaceae is limited to the lake ecotope. Cyperaceae is four times more abundant in the lake ecotope. Occurrence of other groups is rather equal in both ecotopes.

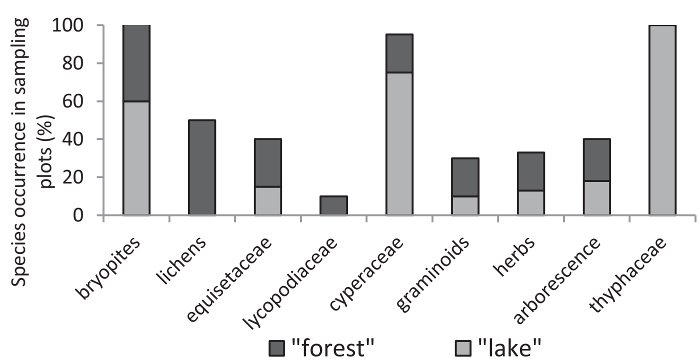

Figure 2. Species occurrence by major functional groups in two ecotopes (forest and lake) on the Doyon north SSP.

The results suggest that, regardless of the oligotrophic edaphic conditions, primary vegetation succession is occurring on the SSP. It is known that primary succession starts with pioneer non-vascular plants (lichens and bryophytes) followed by gramineas and perennials, finally arborescent species appear (Johnson 1992). The succession observed on Doyon SSP is different because it started by pioneer arborescent species, while the understory layer is poorly developed.

Characteristics of Arborescent Species

Ring counting of tree discs showed that the oldest tree specimen is a 25 year-old willow (Table 2). Considering that the sludge was deposited less than 15 years ago, this age indicates that trees were already present on the future SSP basin, and sludge deposition has not inhibited their growth.

Table. 2 Age of four arborescent species on the Doyon north SSP

Tree species	Average tree age (minimum-maximum),years
Willows	21.6 (18-25)
Balsam poplar	18.6 (16-21)
Trembling aspen	17.6 (16-20)
Paper birch	14.3 (12-16)

In the methodology section we reported that willows are the most abundant arborescent species in the lake ecotope, while in the forest ecotope high arborescent diversity is documented. Balsam fir, tamarack, paper birch, trembling aspen, balsam poplar, black spruce, jack pine and willows are found on the SSP. The study of tree and shrub occurrence showed that trembling aspen, paper birch, balsam poplar and willows are the most abundant species in the forest ecotope. The same species, except paper birch, are the most abundant species in the lake ecotope (Figure 3).

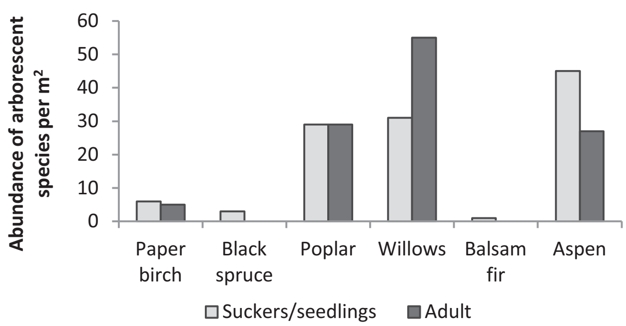

Figure 3. Age structure of arborescent species on the Doyon SSP

The analyses of arborescent species structure (Figure 3) showed that the abundance of arborescent seedlings and suckers is equal to the abundance of adult trees on the SSP. This suggests that arborescents successfully propagate on the site.

Element Concentrations in Plant Tissue

The analyses of variance suggest that there is no significant interaction between plant species and sampling location (ecotope and control), while the difference in terms of concentrations for As, Co, Cu and Zn is due to the ecotope and species (Table 3).

Significant differences in terms of element concentrations are observed in the foliar tissue compared to the branches, wood and roots. Generally, roots accumulate higher trace-element concentrations than other fractions. Surprisingly, plant metal concentrations in the control tissues are in most cases similar or even higher compared to the ones observed on the Doyon SSP. This difference may be explained by high As and metal concentrations in the bedrock of the Cadillac fracture, where the mine site is located.

*Table 3. Analyses of variance performed on main chemical element concentrations in three ecotopes for three species; statistically significant differences (p <**0.05**) are indicated with boldface italic fonts.*

			Ca		Mg		As		Co		Cu		Zn	
Fraction	Source	*DF**	*F*	*P*	*F*	*p*	*F*	*p*	*F*	*p*	*F*	*p*	*F*	*p*
Leaf	Overall model	8	4,33	***0,01***	2,39	***0,05***	1,57	0,19	3,28	***0,01***	3,87	***0,01***	4,68	***0,01***
	Error	20												
	Ecotope	2	5,38	***0,01***	7,38	***0,04***	3,06	0,07	4,19	***0,03***	6,19	***0,01***	9,75	***0,01***
	Species	2	9,88	***0,01***	1,75	0,20	1,40	0,27	4,39	***0,01***	4,57	***0,02***	7,06	***0,01***
	Ecotope*species	4	1,02	0.42	0,21	0,93	0,90	0,48	2,01	0,13	1,22	0,06	0,95	0,46
Branch	Overall model	8	4,28	***0,01***	4,04	***0,01***	1,17	0,36	0,63	0,74	2,20	0,07	2,43	***0,05***
	Error	20												
	Ecotope	2	5,77	***0,01***	4,63	***0,02***	4,32	***0,03***	1,06	0,36	2,18	0,14	5,71	***0,01***
	Species	2	8,48	***0,01***	10,85	***0,01***	0,03	0,97	1,05	0,37	4,34	***0,02***	0,40	0,68
	Ecotope*species	4	1,43	0.26	0,34	0,84	0,17	0,95	0,21	0,93	1,15	0,36	1,81	0,17
Wood	Overall model	8	12,48	***0,01***	3,26	***0,02***	0.66	0,72	0,46	0,87	0,52	0,83	18,89	***0,01***
	Error	20												
	Ecotope	2	15,90	***0,01***	1,47	0,25	0,15	0,86	1,41	0,27	1,37	0,28	31,23	***0,01***
	Species	2	27,60	***0,01***	10,17	***0,01***	0,68	0,52	0,15	0,86	0,42	0,66	16,82	***0,01***
	Ecotope*species	4	1,22	0,06	0,70	0,60	0,91	0,48	0,14	0,96	0,15	0,96	1,25	0,06
Root	Overall model	8	3,28	***0,02***	1,12	0,40	2,50	***0,05***	0,82	0,59	0.95	0,50	6.99	***0,01***
	Error	20												
	Ecotope	2	11,08	***0,01***	3,49	0,05	0,34	0,71	0,86	0,44	1,67	0,22	15,43	***0,01***
	Species	2	1,52	0,25	0,66	0,53	4,67	***0,02***	1,30	0,30	0,52	0,60	5,13	***0,01***
	Ecotope*species	4	0,26	0,90	0,17	0,95	2,49	0,08	0,56	0,70	0,81	0,53	2,28	0,13

*DF: Degree of freedom

Figure 4 presents the average element concentrations (ppm) measured in plant tissues of arborescent species in the two ecotopes in the SSP Doyon site and the control site. P and K were measured only in branches and leaves. Willows accumulate higher macroelement and As, Co, Cu and Zn concentrations (Figure 4). This effect is expected since it is well known that most willows are metal hyperaccumulators (Kuzovkina et al. 2004). Due to that phenomenon, many willows are used as metal phytoextractors on contaminated sites.

More details on element accumulation by the three arborescent species are provided in Table 3.

Paper Birch

There is a difference in element accumulation by birch on the SSP, although the SSP has poor macronutrient content (Table 3). In birch tissues, macronutrient concentrations were generally higher in the lake ecotope compared to the control site and the forest ecotope (Figure 4), while in the forest ecotope As concentrations in the wood and roots were three times higher than the lake and control site.

The analyses showed metal accumulation in the wood and roots of birches, while metal concentrations in branches and leaves were similar in the three locations. Concentrations of Co, Cu and Zn in wood and roots were more than two times higher in the control than in the lake and forest ecotopes. Interestingly, Zn concentrations in the birch roots in the control were the highest among the three species in all locations.

Birch is classified as a pioneer species (Gallagher et al. 2011); it possesses plasticity and tolerance to a wide range of environmental conditions, however according to the literature birches are not considered a species with outstanding tolerance to soil contamination. The results of our study suggest that the SSP is a suitable habitat for this species and the risk of bioaccumulation by birches is minimal.

Balsam Poplar

Poplar had similar trends of macronutrient accumulation as birch (Figure 4). Major nutrients in poplar, except Ca, were twice or threefold lower in all fractions in the control compared to the forest and lake ecotopes. Concentration of As in poplar tissues did not differ significantly among locations. Poplar roots in the control contained the highest concentration of Co, which were more than five times higher than the concentrations in other fractions in the forest and lake ecotopes. The highest concentrations of Cu were found in poplar roots in the lake ecotope. Concentrations of Zn showed similar sequence between fractions: wood > roots > leaves = branches in all locations.

Willows

Phosphorus concentrations in the tissue of willows did not differ among locations. Potassium concentrations were highest in the control, while the highest Ca and Mg concentrations were measured in the lake ecotope. In willows, the roots are the major As accumulators compared to other fractions. Willow roots in the control contained concentrations of As twice those in other locations. Cu root concentrations in the lake ecotope were ten times higher than the control site and twice higher than the forest ecotope. A similar trend was observed in poplar. Possibly more Cu is present in plant available form in the lake

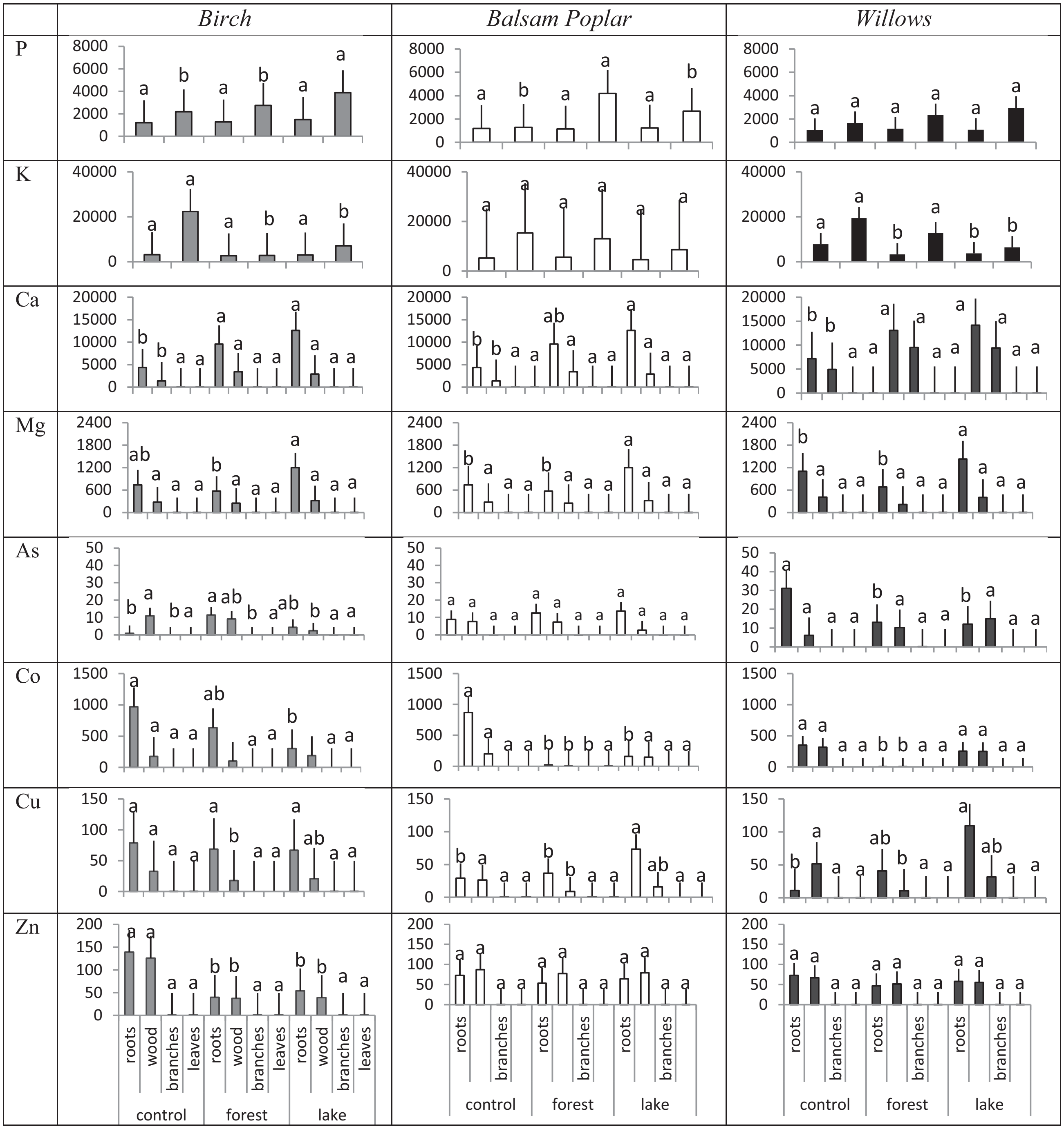

Figure 4. Average element concentrations (ppm) in plant tissue of arborescent species in two ecotopes of the SSP Doyon and in the control site. Tukey's test of element concentrations of the same plant fraction among three sampling locations. Columns with same letter are not significantly different according to the test. Bars indicate standard deviation.

compared to the other two locations, and the fact that willows and poplars are hyperaccumulators intensifies their capacity to accumulate Cu in this habitat. Cu concentrations in other fractions were similar and near the detection limit.

Concentrations of Zn and Co in different fractions of willows had similar distribution in different locations. It is worth noting that willow roots and wood contained equal concentrations of Co and Zn, which might be explained by the high translocation capacity of these elements within plants (Adriano 2001).

Balsam poplar and willows have many ecological similarities: they are fast growing and adapt to a wide range of ecological conditions, and can tolerate excessive metal concentrations in the soil. Therefore they are often used in phytostabilisation projects focused on phytoextraction (Kuzovkina et al. 2004) or contaminated site revegetation. The fact that As and other metal concentrations in the tissue of poplar and willows on the SSP do not surpass those of the control allow concluding that these species do not represent bioaccumulation risks by higher levels of the food chain.

CONCLUSIONS

In this investigation, a vegetation survey was performed on a sludge pond at the Doyon mine site. The bioaccumulation of contaminants by different fractions of three main arborescent species encountered has been investigated. Chemical analyses of plant tissues did not show any abnormal element accumulation compared to samples collected off the site, which indicate a possibility of using the sludge as a revegetation substrate for mine site reclamation. However, further detailed investigations, including analyses of plant available element concentrations and of other plant functional groups (e.g., lichens and perennial species) remain necessary to confirm the suitability of sludge as growth medium.

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ESTABLISHMENT OF NATIVE BOREAL PLANT SPECIES ON RECLAIMED OIL SANDS MINING DISTURBANCES

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ABSTRACT

Current revegetation strategies for disturbed oil sands sites include values such as biodiversity and sustainability. As a result, a wide range of species are sought for inclusion in revegetation programs. A trial was initiated in 2007 to evaluate emergence of 41 boreal vascular and provide definitive strategies for establishment.

Emergence of 24 species was observed by the second year. Many species responded differently depending on site characteristics while others established consistently over the three locations. We documented two species that emerge in higher percentages from whole fruit than from cleaned seed. We have also determined a number of species that benefit from either spring or fall sowing.

Direct sowing information, seed metrics and other culturally information on all 41 species was gathered to create 'Propagation and Establishment Profiles'. These 2 to 5 page fact sheets will be publically available through the Oil Sands Research and Information Network (OSRIN).

Key Words: Boreal, Emergence, Oil Sands.

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INTRODUCTION

Revegetation of disturbed mined oil sands sites has, historically, focused primarily on forest productivity. More recently, the focus of revegetation has widened to include values attributed to other end land uses, such as wildlife habitat, aboriginal land uses, recreation, biodiversity and sustainability.

A comprehensive field trial was initiated under ERRG in 2007 to encompass a large number of boreal species and provide more definitive strategies for establishment. The objectives were:

1. To determine the effect of sowing season and propagule type on emergence and establishment of up to 40 native boreal plant species under field conditions,
2. To provide site specific information regarding the establishment of individual species, and

3. To fill knowledge gaps in the literature used by practitioners to efficiently and effectively grow and establish locally harvested native boreal plant species (i.e., phenology, propagule harvest and cleaning methods, germination and pre–treatment requirements, seed and fruit metrics).

This field trial is ongoing and all results presented are preliminary.

PROJECT METHODOLOGY

Seeds are harvested from areas within 100 km of the actively mined oil sands of northeastern Alberta. Seeds are cleaned by hand, with methods documented (Table 1). Seeds are stored dry until the following fall and spring, at which time they are direct–seeded in field trials. Fruit (for species that produce fleshy fruit or berries) are also harvested and frozen until being sown the following fall and spring.

Seed weights are recorded and/or calculated for each species (Table 2). Sub-samples of 100, 500 or 1000 cleaned seeds (depending on size) are weighed and averaged. Fruit metrics are recorded for accessions with fleshy fruits: number of seeds/fruit (average), average number of fruit per litre, weight of fruit per litre. From this, the average number of seeds per kilogram of fruit is calculated.

Germination of individual species is tested using standard methods (Table 3). Germination trials are conducted on fresh seeds (within six to eight months of harvest) and again in the following year (12 to 18 months after harvest). Four samples of 25 seeds from each accession are counted and placed into petri dishes on Whatman #1 filter paper. Plates are placed in a refrigerator at 2 to 4°C for cold stratification, the length of which was determined based on literature and previous experience. Plates removed from stratification are randomized with control plates placed at room temperature (19 to 23°C) in ambient room light. Germinants are counted twice weekly for at least four weeks. Germination is considered complete when the radicle emerges.

All information regarding the methodologies (cleaning, germination, emergence) are used to populate Propagation and Establishment profiles. These fact sheets are intended for use by all levels of industry: growers, reclamation operators, harvesters. These profiles will be published by OSRIN (Oil Sands Research and Information Network) and should be available publicly by the end of 2013.

The three trial locations selected (Table 1) represent a diversity of environments in which to test emergence and establishment.

Target boreal plant species are divided into two broad categories – those that have dry seeds and those that produce a fleshy fruit (e.g., drupes or berries). The primary treatment for the dry-seeded species is sowing season (i.e., fall vs. spring). Whereas, there are two primary treatments for fleshy fruited species: propagule type (seed vs. fruit) and sowing season (fall vs. spring). Each category is replicated four times at each of three sites over two sowing years.

Table 1. *Experimental site locations and descriptions*

Site	Location	Description of Reclamation
Suncor – Steepbank Sand Pit Area	N 56° 53' 56.8" W 111° 24' 7.27"	Reclaimed 2003 to d1 Ecosite; overburden; soil depth 28 cm – directly placed peat/mineral; fertilized in 2003 with 23.5-25-8 (NPK) at a rate of 200 kg/ha. Although planted in 2004, the site was ploughed in 2007 to provide a relatively open site for this experiment.
Syncrude Mildred Lake – W1	N 57° 1' 3.7" W 111° 43' 47.6"	Capped with 110 cm of peat-mineral mix (top 14 cm organic material) in 2005. *Populus tremuloides* and *Picea glauca* were planted at 1360 and 670 stems/ha respectively in 2007. In addition, *Alnus viridis* was planted at 250 stems/ha.
Syncrude Aurora – Fort Hills	N 57° 9' 49.4" W 111° 31' 58.4"	Capped with 90 cm of peat-mineral mix in 2005. In 2006, *Picea glauca* and *Populus tremuloides* were planted at 1218 and 882 stems/ha respectively. In addition, *Rosa acicularis* was planted at 230 stem/ha and *Amelanchier alnifolia* was planted at 321 stems/ha.

Thirty species plots are delineated on each site. There are thirty-two individual subplots (each 1 m x 1 m) within each, enough for one species with fleshy fruit, and buffers of 2 m between subplots. A 4 m buffer lies between species plots. For dry-seeded species, two species are included in a single species plot.

Sowing occurs in the fall a year after seeds are first harvested and again in the following spring. Soil/substrate is scuffled with a garden rake and seeds or fruit are broadcast. Soil is then raked and pressed to incorporate propagules and ensure good propagule/soil contact. Seeding rates are based on previous experience, seed size and seed availability. For fleshy-fruited species, the seeding rates are adjusted to obtain the same number of seeds in each plot for the two propagule types (e.g., for *Aralia nudicaulis* with a fruit sowing rate of 50 fruit/m^2 the seed sowing rate would be 250 seed/m^2 as each fruit bears approximately five seeds).

Plots are monitored for emergence/survival and vigour annually for up to five years. Many plots have been observed for three years, but a few have only been monitored once. Hence, results are tentative. In spring of 2012, light meter readings were taken in each sub-plot and these data used as a covariate in the analysis of variance. In this way, the reduced light availability due to competition from other vegetation is taken into account, making differences from other effects, such as season of sowing, more apparent among sites.

RESULTS

Each species is unique and emergence of one species isn't directly comparable to that of another, therefore we did not compare emergence percentages among species. However, it was possible to group species by their preferred establishment conditions.

Seven dry seeded species emerged equally well regardless of season. *Castilleja raupii* and *Symphyotrichum laeve* (purple paintbrush and smooth blue aster) emerged in similar proportions on all three experimental sites. Both *Solidago* species (*S. canadensis* and *S. simplex* – Canada and mountain goldenrod) as well as *Hesperostipa curtiseta* (western porcupine grass) emerged best on Aurora. *Vicia americana* (American vetch) was the only species to emerge best at Mildred Lake and *Anemone multifida* (cut-leafed anemone) emerged equally well on Aurora and Suncor, but in lower proportions on Mildred Lake.

The remaining dry-seeded species showed a preference for sowing season. *Bromus ciliatus* (fringed brome) and *Dasiphora fruticosa* (shrubby cinquefoil) emerged best from fall seed while *Anemone patens* (prairie crocus) emerged in greater proportions from spring sowing. *A. patens* and *D. fruticosa* emerged best on Aurora, but *B. ciliatus* showed no preference for site.

None of the fleshy fruited species showed a preference for sowing season alone, but propagule type was significant for seven species. Only *Viburnum edule* (lowbush cranberry) and *Shepherdia canadensis* (buffaloberry) emerged in higher proportions from entire fruit.

Amelanchier alnifolia (Saskatoon), *Arctostaphylos uva-ursi* (bearberry), *Prunus pensylvanica* (pin cherry), *Rosa acicularis* (prickly rose) and *Symphoricarpos albus* (snowberry) emerged best from cleaned seed. Of these, only *A. alnifolia* and *S. canadensis* emerged equally well on all the sites. The rest preferred Aurora.

There were also four species that emerged best under specific propagule/season combinations. *Fragaria virginiana* (wild strawberry) and *Rubus idaeus* (red raspberry) both emerged best from fall sown cleaned seed, whereas *Cornus sericea* (dogwood) and *Prunus virginiana* (chokecherry) preferred spring sown seed. *P. virginiana* and *R. idaeus* emerged best at Aurora, and *F. virginiana and C. sericea* emerged equally well on all three sites.

Betula papyrifera (paper birch), *Campanula rotundifolia* (harebell), *Cornus canadensis* (bunchberry) and *Sibbaldiopsis tridentata* (three-toothed cinquefoil) have emerged on one or more sites, but too few plants are available for statistical analysis.

The remaining 16 species have not emerged to date:

Alnus incana (river alder)
Alnus viridis (green alder)
Apocynum androsaemifolium (dogbane)
Aralia nudicaulis (sasparilla)

Cypripedium acaule (pink lady's slipper)
Geocaulon lividum (bastard toadflax)
Lilium philadelphicum (wood lily)
Lonicera caerulea (blue-fly honeysuckle)
Maianthemum canadense (lily of the valley)
Mitella nuda (bishop's cap)
Rhododendron groenlandicum (Labrador tea)
Rumex aquaticus (wild sorrel)
Schizachne purpurascens (false purple melic)
Trientalis borealis (northern starflower)
Vaccinium myrtilloides (dwarf blueberry)
Vaccinium vitis-idaea (bog cranberry)

CONCLUSIONS

These emergence trends can be used to broadcast seed or fruit at the appropriate time for the greatest chance at emergence and survival.

Although species can be grouped by their preferred season of sowing or propagule emergence, these trends bear no relation to how a species responds to a given experimental site. Each species is individual and often need to be treated differently from one another. By compiling emergence information alongside cleaning (Table 2), seed metrics (Table 3), and germination (Table 4) information, a proper establishment/revegetation plan can be made based on the species selected for revegetation.

There is a general trend toward greater emergence (both number of species as well as number of individuals) occuring at Aurora and Suncor than at Mildred Lake. This is probably, in part, a response to the warmer average soil temperatures recorded at these two sites in early June (Aurora – 16°C, Suncor – 15°C and Mildred Lake – 13.5°C) and likely throughout the season. Temperature differences can be attributed to various factors including soil type, slope and aspect. Greater vegetation cover at Mildred Lake may also cause cooler soil temperatures as well as compete for nutrients and water.

Table 2. Seed cleaning methods for native boreal species.

Cleaning Methods	**Species**
For achenes or caryopsis or for seeds formed in capsules that dry as they mature. Air-dry fruits in paper or Tyvek bags at 15-25˚C. Crush material or remove large chaff and crush remaining material. Sieve to remove seeds from chaff using appropriate size screens. Small chaff and dust can be removed by winnowing. If capsules are intact merely open capsules and empty seeds; sieve or winnow to remove chaff and dust. For awned species – break off awns with a de–awner or by hand.	*Alnus incana* *Alnus viridis* *Anemone patens* *Betula papyrifera* *Bromus ciliatus* *Campanula rotundifolia* *Castilleja raupii* *Hesperostipa curtiseta* *Lilium philadelphicum* *Mitella nuda* *Rumex aquaticus* *Rhododendron groenlandicum* *Schizachne purpurascens* *Trientalis borealis* *Vicia americana*
For small seeds enclosed in succulent fruit. Place pulpy fruits in water (use about 3:1 water with fruit) and place in a blender on low speed until fruits are fully macerated. Pour through sieve(s) to remove chaff smaller than seeds. Re-suspend residue in water and mix; allow seeds to settle and decant water with floating and suspended larger chaff. Repeat re-suspension step until seeds are clean; sieve and place seeds on paper towelling or cloths to dry. Dry at room temperature or up to 25˚C over a moving air stream.	*Amelanchier alnifolia* *Aralia nudicaulis* *Cornus canadensis* *Fragaria virginiana* *Geocaulon lividum* *Lonicera caerulea* *Maianthemum canadense* *Rubus idaeus* *Shepherdia canadensis* *Vaccinium myrtilloides* *Vaccinium vitis-idaea* *Viburnum edule*

Cleaning Methods	Species
<u>For small seeds enclosed in dry, pulpy fruit</u> Rub fruit between corrugated rubber in a box or on a large size sieve to remove pulp. Winnow the chaff off or suspend seeds and remaining chaff in water and mix. Allow the seeds to settle and decant off the lighter chaff. Repeat the suspension if necessary. Dry at room temperature or up to 25˚C preferably over a moving air stream.	*Arctostaphylos uva-ursi*
<u>For seeds with an attached pappus</u> Pull seeds from seed heads by hand or rub seeds with pappus between corrugated rubber in a box. Sieve to remove seeds from chaff using appropriate size screens. Small chaff and dust can be removed by winnowing. Alternately, pappus with attached seeds can be placed on a sieve with opening size large enough to let seeds through stacked on a sieve that will catch the seeds. Place a smaller sieve over the top sieve and direct a strong flow of air (such as that produced by a reversed vacuum) through the top sieve. Seeds will be removed from the pappus and lodge in the small mesh sieve.	*Anemone multifida* *Apocynum androsaemifolium* *Dasiphora fruticosa* *Sibbaldiopsis tridentata* *Solidago canadensis* *Solidago simplex* *Symphyotrichum laeve*
<u>For large, round, hard seeds in pulpy fruits</u> Mash fruits by hand or using a potato masher, apple-saucer, or ricer, or run through a hand meat grinder. Alternatively, use a food processor on low speed with blunt mashing blade (not a sharp blade) or use a blender with blades covered by plastic tubing or duct tape. Suspend residue in water and mix; allow seeds to settle and decant water with floating and suspended larger chaff. Repeat this step until seeds are clean; sieve and place seeds on paper towelling or cloths to dry. Dry at room temperature or up to 25˚C preferably over a moving air stream.	*Cornus sericea* *Prunus pensylvanica* *Prunus virginiana*
<u>For large, irregular-shaped, hard seeds in pulpy fruits</u> Process fruits through a tomato de-seeder (several times if necessary) to remove juice and some fruit pulp. Suspend residue in water and mix; allow seeds to settle and decant water with floating and suspended larger chaff. Repeat this step until seeds are clean; sieve and place seeds on paper towelling or cloths to dry. Dry at room temperature.	*Rosa acicularis* *Symphoricarpos albus* *Viburnum edule*

Table3. Seed and fruit metrics.

Species	Average Seeds/g	# of samples/ accessions	g/1000 seeds	Average Seeds/ fruit	Average # of Fruit/L	Average # of Fruit/kg
Alnus incana	2 013	23/7	0.51			
Alnus viridis	5 582	26/10	0.1996			
Amelanchier alnifolia	334	17/9	3.02	9	1 590	3 350
Anemone multifida	1 152	6/2	0.95			
Anemone patens	1 136	6/2	1.08			
Apocynum androsaemifolium	6 295	9/3	0.16			
Aralia nudicaulis	198	6/2	0.90	5	3 320	5 650
Arctostaphylos uva-ursi	168	67/20	6.31	6	1 690	3 720
Betula papyrifera	6 375	18/10	0.17			
Bromus ciliatus	586	18/6	0.19			
Campanula rotundifolia	31 810	9/3	0.03			
Castilleja raupii	13 970	26/7	0.07			
Cornus canadensis	151	3/1	6.65	1	2 780	7 890
Cornus sericea	38	9/6	26.71	1*	2 460	4 900
Dasiphora fruticosa	5 353	21/8	0.20			
Fragaria virginiana	2 598	24/8	0.39	35	2 370	3 940
Geocaulon lividum	23	6/2	42.83	1	2 310	3 110
Hesperostipa curtiseta	81	15/5	12.38			
Lilium philadelphicum	1 340	10/6	1.21			
Lonicera caerulea	1 900	11/4	0.54	10	2 450	4 350
Maianthemum canadense	111	9/3	9.05	1.3	8 370	14 700
Mitella nuda	3 606	9/3	0.28			
Prunus pensylvanica	26	14/7	39.64	1	1 620	3 310
Prunus virginiana	15	10/6	69.67	1	1 200	2 090
Rhododendron groenlandicum	53 230	10/6	0.03			
Rosa acicularis	104	13/7	9.88	23	630	1 130

Species	Average Seeds/g	# of samples/ accessions	g/1000 seeds	Average Seeds/ fruit	Average # of Fruit/L	Average # of Fruit/kg
Rubus idaeus	1 243	18/6	1.23	37	808	1 160
Rumex aquaticus	1 096	18/6	0.95			
Schizachne purpurascens	591	5/2	1.70			
Shepherdia canadensis	165	42/12	6.13	1	5 640	8 090
Sibbaldiopsis tridentata	2 611	6/2	0.39			
Solidago canadensis	11 740	15/5	0.11			
Solidago simplex	10 020	12/4	0.13			
Symphoricarpos albus	208	11/7	4.89	2	3 220	10 000
Symphyotrichum laeve	5 048	13/7	0.20			
Trientalis borealis	2 269	16/6	0.45			
Vaccinium myrtilloides	6 810	46/16	0.15	37	2540	5 240
Vaccinium vitis-idaea	4 880	52/17	0.21	12	3 190	7 050
Viburnum edule	46	32/14	22.07	1	1 770	3 030
Vicia americana	77	6/2	13.13			

Table 4. *Germination recommendations.*

Species	Germination recommendations
Alnus incana	Cold stratify fresh or year old seeds for at least 4 weeks.
Alnus viridis	Cold stratify fresh or year old seeds for at least 4 weeks.
Amelanchier alnifolia	Cold stratify fresh or year old seeds for 16 weeks.
Anemone multifida	No pre-treatment required.
Anemone patens	Cold stratify fresh seed for 4 weeks or allow one year after-ripening.
Apocynum androsaemifolium	Fresh seed require no pre-treatment.
Arctostaphylos uva-ursi	Seed germination is highest when year-old seeds are soaked in concentrated sulphuric acid for a period of four and a half hours and subjected to at least 8 weeks of warm stratification followed by at least the same length of cold stratification.
Betula papyrifera	4 weeks stratification of fresh or 1 year old seeds.
Bromus ciliatus	Cold stratify fresh seeds for at least 4 weeks.

Species	Germination recommendations
Campanula rotundifolia	No pre-treatment required.
Castilleja raupii	Cold stratify fresh, 1 year or 2 year old seeds for at least 4 weeks. Seeds lose little viability for at least six years.
Cornus canadensis	Cold stratify seed for 20 weeks.
Cornus sericea	Cold stratify fresh, 1 year or 2 year old seeds for 4 weeks.
Dasiphora fruticosa	No pre–treatment required.
Fragaria virginiana	Cold stratify fresh or year old seeds for up to 12 weeks.
Hesperostipa curtiseta	Cold stratify year old seeds for 4 weeks.
Lilium philadelphicum	No pre–treatment required.
Lonicera caerulea	Cold stratify fresh seeds for at least 4 weeks.
Maianthemum canadensis	Cold stratify fresh seed for 8 or 12 weeks.
Rhododendron groenlandicum	Stratify fresh or one year old seeds for 4 weeks.
Rubus idaeus	Acid scarify seeds for one minute prior to 16 weeks warm stratification followed by 8 weeks cold stratification.
Rumex aquaticus	Cold stratify fresh or year old seeds for 4 weeks.
Schizachne purpurascens	Fresh seeds require no pre-treatment
Shepherdia canadensis	Cold stratify mechanically scarified seed for 8 weeks.
Sibbaldiopsis tridentata	Cold stratify fresh seeds for at least 4 weeks.
Solidago canadensis	No pre–treatment required.
Solidago simplex	Fresh seeds require no pre-treatment.
Symphyotrichum laeve	No pre-treatment required.
Trientalis borealis	Warm stratify year old seeds for 16 weeks.
Vaccinium myrtilloides	Cold stratification did not improve germination but greater germination was found when tested for period of 60 days or more.
Vaccinium vitis-idaea	Cold stratify fresh or 1 year old seeds for at least 12 weeks.
Viburnum edule	Warm stratify seed for 16 weeks followed by cold stratification for 12 weeks.
Vicia americana	Mechanically scarify seeds.

BIOLOGICAL SOIL CRUSTS AND NATIVE SPECIES FOR NORTHERN MINE SITE RESTORATION

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ABSTRACT

The nitrogen cycle is highly sensitive to pollutants and restoration of this biogeochemical pathway is essential to ensure long-term sustainable ecosystems. In a greenhouse trial, the growth and nitrogen fixation rates of *Dryas drummondii*, *Hedysarum alpinum*, *Oxytropis campestris* and *Lupinus arcticus* were determined in both tailings and cover soils with amendments of rhizobia and biochar. In a growth chamber trial, Pure *Nostoc commune* culture, Dried *Nostoc* spp. and biological soil crust (BSC) slurries derived from mature soil crusts were applied with and without biochar to tailings. *L. arcticus* had the highest biomass and all species with the exception of *D. drummondii* showed nitrogen fixation after 3 months, with higher rates observed in cover soils. Nodulation and nitrogen fixation only occurred in herbs given a rhizobia innoculum, indicating the need for microbial amendments. All mature BSC treatments had significantly higher rates of nitrogen fixation compared with other treatments. Inclusion of biochar did not significantly increase the rates of nitrogen fixation for BSCs, but did influence nitrogen fixation by native herbs. Including local native nitrogen-fixing species in seed mixes and establishing nitrogen-fixing BSCs may reduce the application rates of artificial fertilizer and promote plant community growth while establishing primary successional processes.

Key Words: Biochar, Cryptogamic Crusts, Indigenous Species, Nitrogen Fixation, Revegetation, Soil Amendments.

INTRODUCTION

Metalliferous mine tailings are often a source of heavy metal pollution as a result of removal by wind and water, as well as, leaching of products of mineral weathering into nearby watercourses (Liu et al. 2012; Shu et al. 2001). Physical and chemical techniques provide a means to reduce dust and water erosion, however phytostabilization can provide a relatively simple and cost effective way to reduce these risks (Bradshaw 1997; Lan et al. 1998; Liu et al. 2012; Ye et al. 2002). Natural recolonization of plants on mining impacted soils is often limited since these degraded materials often have no aggregate structure or organic matter and are deficient in nutrients (N and P), as well as having high toxicity associated with metals and metalloids (Pb, Zn, Cu, Cd, Mn, Ni and As), requiring intensive use of amendments and chemical fertilizers (Huang et al. 2011; Petrisor et al. 2004; Ye et al. 2002).

Establishment of native pioneer species can not only improve soil characteristics by enhancing organic content and supplying needed nutrients, but may also reduce long-term soil toxicity, so that more sensitive plants can establish leading to a healthier more diverse ecosystem (Chan et al. 2003). Indigenous or native species are preferable to exotic species because they are most likely to fit into fully functional ecosystems and are climatically adapted (Chaney et al. 2007; Li et al. 2003; Sheoran et al. 2010). This may be particularly important in harsh climates typical of northern Canada.

The nitrogen cycle is highly sensitive to pollutants and restoration of this biogeochemical pathway is essential to ensure long-term sustainable ecosystems. Nitrogen fixing plants may be important in soil restoration due to the addition of C and N and a more favourable balance between the production and immobilization of inorganic N (Myrold and Huss-Danell 2003; Rhoades et al. 2001; Wurtz 1995). In highly disturbed N-limited ecosystems biological nitrogen fixation is important for plant growth and nitrogen fixing legumes are often used in restoration efforts (Bradshaw 1997; Broos et al. 2004; Liu et al. 2012; Reichman 2007; Tordoff et al. 2000). However, these legumes are often not native to the area of restoration.

Biological Soil Crusts (BSCs) are early successional communities, composed of bacteria, cyanobacteria, algae, mosses, liverworts, fungi and lichens. BSCs are found to occur naturally throughout northern Canada and these crusts may act as keystone communities in establishing primary successional processes and returning disturbed ecosystems to a desirable trajectory (Bowker 2007). A few studies have shown inoculation of mine tailings with BSCs may be a means to accelerate the restoration process (Liu et al. 2012; Spröte et al. 2010). BSCs may play several important roles in restoration processes, including reduction of soil erosion, increased soil organic matter, carbon and nitrogen content and positively influencing local hydrology and vascular plant establishment (Li et al. 2012; Zhao et al. 2006, 2010).

Due to the lack of organic matter, low pH and high metal content the conditions on many mine sites may be too harsh for establishment of vegetation. Hence, the use of soil amendments may be necessary to allow for successful germination and growth. Several studies have found biochar, a product that results from the oxygen limited pyrolysis of various biological ingredients, can result in significant decreases in the bioavailability of heavy metals associated with mine impacted soils (Beesley and Marmiroli 2011; Fellet et al. 2011; Namgay et al. 2006) and simultaneously improve physical, chemical and biological soil properties (Laird et al. 2010).

The objective of this study is to examine native herb and biological soil crust species and biochar soil amendments to determine optimal formulations that improve soil conditions and promote long term re-vegetative success and nitrogen input in northern mining impacted soils.[5]

METHODS

Nitrogen-Fixing Herb Greenhouse Trial

Soil amendments and nitrogen-fixing herb species were examined in a greenhouse trial at the Yukon Research Centre, Whitehorse, Yukon. Tailings and mining impacted soils for the greenhouse trial were taken from the Keno Hill Silver District located 330 km north of Whitehorse, Yukon, Canada (63°55’26.4N, 135°29’76.1W). The tailings are highly variable with a pH ranging from 5.7 to 8.4 and texture varying from silt loam to sand. The tailings exceed the Canadian Council of Ministers of the

[5] An extended version of this paper has been submitted to Restoration Ecology.

Environment (CCME) industrial soil quality guidelines for allowable levels of Antimony (Sb), Arsenic (As), Cadmium (Cd), Copper (Cu), Lead (Pb), Silver (Ag), Titanium (Ti) and Zinc (Zn). Mining impacted soils from Husky SW (63°54'18.9 N, 135°31'45.1W) were also collected on-site. The Husky SW soil is currently being used in an engineered cover design trial as a cover soil. Cover soils pH was 8.0 to 8.4 with a loam texture and organic carbon content of 47%. The soils exceed the CCME industrial soil quality guideline for As.

We used a full factorial design with 2 soil types (Cover and Tailings), 4 local native nitrogen-fixing species (*Dryas drummondii* (Richardson ex Hook.), *Hedysarum alpinum* (L.), *Oxytropis campestris* (L.) and *Lupinus arcticus* (S. Watson)) and 4 soil amendments. Each treatment combination had 12 replicates for a total of 384 samples. Each sample comprised an individual container that was 3.8 cm in diameter with a volume of 164 ml (Ray-Leach Tubes, Stuewe & Sons, Tangent, Oregon).

Seeds from the 4 local native nitrogen-fixing species were collected throughout August 2012. In addition, for each species the belowground systems of a number of plants were excavated, examined and sampled for rhizobia or frankia nodules.

Three different soil amendments and a control treatment were used: biochar (BC), rhizobia (R), biochar and rhizobia (BCR) and a control with fertilizer only (C). The biochar (BC) was a Phosphorus-rich bonemeal biochar (2-14-0). The biochar (1 kg/m^2) was mixed with deinonized (DI) water and 5 ml of biochar slurry was added to each container. Nodules previously collected were masticated in DI water to create rhizobia slurries or for *D. drummondii* a Frankia slurry. Seeds receiving the R and BCR treatments were soaked for 3 hours in the slurry prior to planting. Slurries contained 1.4 mg/ml of masticated nodules (wet weight). Containers receiving the R treatment were given 2.5 ml of slurry and 2.5 ml of DI water. All containers received 2 seeds of one of the 4 species and fertilizer (19:19:19) at a rate of 110 kg/ha. The trial was initiated on September 11, 2012.

The greenhouse conditions and watering were controlled to reflect typical summer growing conditions in the Keno area. Temperature was 11°C with no light from 22:00 to 4:00 and 16°C with 175 $\mu mol/m^2/s$ of light from 4:00 to 22:00. Each replicate was watered every second day with 6 ml of DI.

From December 6 to 13, 2012 containers were sampled for germination rate, number of observable nodules, above and belowground biomass and nitrogen fixation. Measurements of N_2-fixation were made using acetylene reduction assays (ARA) (Stewart et al. 1967). Plants were harvested from each container (with belowground systems kept intact) and placed in a separate 60 ml amber glass vial with a Teflon septa cap. Each amber vial was injected with 10% (v/v) acetylene gas (C_2H_2) and incubated in the dark at 20°C for 4 hours. Ethylene concentrations were measured with a portable gas chromatograph (SRI 8610A, Wennick Scientific Corporation, Ottawa, ON, Canada) fitted with a Porapak column (Alltech Canada, Guelph, ON, Canada) and a flame ionization detector.

Biological Soil Crust Growth Chamber Trial

Both the cover soils from Husky SW and Valley Tailings used in the greenhouse trial were used in the growth chamber trial. The substrates were collected throughout the summer of 2012 and were autoclaved at 120°C for 1 hour prior to use to provide a sterile medium free of any pre-existing soil microorganisms. Autoclaved soils were lightly packed into petri dishes (1.2 cm height, 60.82 cm^2 surface area) leaving 0.5 cm for addition of slurry treatments to the surface.

Six different treatments were applied as slurries to the Valley Tailings (T): Pure *Nostoc commune* (Vaucher ex Bornet & Flahault) culture (UTEX Culture Collection of Algae, University of Texas) with biochar (NC BC T) and without biochar (NC T), Dried *Nostoc* spp. collected from grassland near Haines Junction, Yukon with biochar (ND BC T) and without biochar (ND T) and biological soil crust slurry from mature soil crusts collected at Husky SW with biochar (S BC T) and without biochar (S T). Only two treatments were applied to the Husky SW cover soils (C): Biological soil crust slurry from mature soil crusts collected at Husky SW with biochar (S BC C) and without biochar (S C). Each treatment on each soil type had 10 replicates (i.e., petri dishes) for a total of 80 samples. For treatments receiving bonemeal biochar, biochar was added at a rate of 1 kg/m^2. All treatments also received a commercial fertilizer (19:19:19) that was pulverised with a mortar and pestle and added at a rate of 110 kg/ha (i.e., 5.9 g per petri dish). For each treatment, slurries were created by adding nitrogen fixers/crust, biochar and fertilizer to 100 ml of DI water and 10 ml of slurry was added to each replicate.

All samples were placed in a growth chamber (Conviron Adaptis A1000, Winnipeg, MB, Canada) on September 10, 2012 which had a diurnal cycle of temperature ranging from 9.9 to 19.3°C, relative humidity from 47 to 81% and light from 0 to 200 $\mu/m^2/s^1$ with darkness from 22:00 to 4:00 hrs. DI water (6 ml) was added to each petri dish every second day for the duration of the experiment.

From December 19 to 22, 2012 petri dishes were randomly selected and net photosynthesis, dark respiration and nitrogen fixation were measured for each replicate. Each petri dish was placed within a 450 ml clear glass incubation chamber and sealed with high vacuum grease. Rates of net photosynthesis and dark respiration were calculated from changes in CO_2 concentration within incubation chambers over approximately 30 minutes (LI-840A CO2/H2O analyzer, Li-Cor, Lincoln, Nberaska, USA). Nitrogen fixation was measured immediately after photosynthesis and respiration using the same ARA method as described above except that samples were incubated for 4 hours under at 20°C and 200 $\mu/m^2/s^1$ of light.

Data from the greenhouse and growth chamber trials were examined to ensure the assumptions of Analysis of Variance (ANOVA) were met and log transformations were performed on some variables. Greenhouse trial data was analyzed using a full factorial ANOVA. All analyses were conducted in R (R package version 2.1.50).

RESULTS

Nitrogen-Fixing Herb Greenhouse Trial

Overall cover soils had significantly lower germination rates (42% vs. 53%, $p<0.01$), but higher rates of nodulation (3.8 nodules vs. 0.3 nodules, $p <0.001$) and nitrogen fixation (209 µmol ethylene/m^2/hr vs. 23 µmol ethylene/m^2/hr, $p<0.001$) compared with tailings. Aboveground and belowground biomass were not significantly different between the two substrate types ($p = 0.47$ and $p = 0.63$ respectively).

O. campestris had significantly lower germination rates (28%) compared with all other species (*D. drummondii* = 52%, *H. alpinum* = 51%, *L. arcticus* = 59%), which did not differ significantly from each other. The BC treatment (34%) had lower germination compared with all other soil amendment treatments (BCR = 54%, F = 51%, R = 52%, ANOVA, $p<0.001$ for all comparisons).

L. arcticus (62 g/m^2) had significantly higher average aboveground biomass and belowground biomass compared with all other species (ANOVA, $p<0.001$ and $p<0.05$ for above and below comparisons respectively) (Figure 1). *H. alpinum* had significantly higher average belowground biomass than *D. drummondii* and *O. campestris* ($p<0.05$ for both comparisons). The BC soil amendment treatment had lower average aboveground biomass (8.7 g/m^2) compared with all other soil amendments (BCR = 24 g/m^2, R = 19g/m^2, F = 26 g/m^2, $p<0.05$ for all comparisons). The BC soil amendment treatment (28 g/m^2) had significantly lower average belowground biomass than the BCR (61 g/m^2) and R (56 g/m^2) treatments ($p<0.05$ for both comparisons).

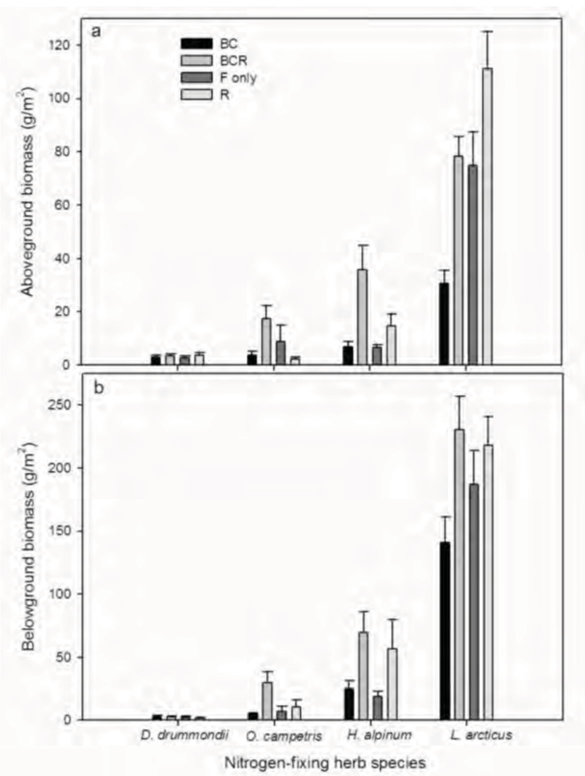

Figure 1. Aboveground (a) and belowground (b) biomass of four native nitrogen-fixing herb species after 12 weeks of growth in a greenhouse trial. All species were fertilized and four soil amendment treatments were applied to each species: biochar (BC), biochar and rhizobia (BCR), fertilizer only (F only) and rhizobia (R). Bars are means with SE.

The R and BCR treatments had higher average rates of nodulation (4.7 and 3.6 nodules respectively) than the BC and F treatments (0 and 0.02 nodules respectively, $p<0.001$ for all comparisons). *L. arcticus* had the highest average rate of nodulation (3.9 nodules) and was significantly higher than both *O. campestris* (1.3 nodules) and *D. drummondii* (0 nodules) ($p<0.05$ and $p<0.001$ respectively). *H. alpinum* had the second highest average rate of nodulation (3.1 nodules).

All species with the exception of *D. drummondii* demonstrated nitrogen fixation after 3 months. Only those samples treated with the rhizobia innoculum (i.e., CR, TR, CBCR and TBCR) had nitrogen fixation above our detection limit (10 µmol ethylene/m^2/hr) (Figure 2). *H. alpinum* had significantly higher mean rates of nitrogen fixation (288 µmol ethylene/m^2/hr) compared with both *O. campestris* (50 µmol ethylene/m^2/hr) and *L. arcticus* (11 µmol ethylene/m^2/hr) (ANOVA, $p<0.001$). Average nitrogen fixation was significantly higher for the BCR treatment (188 µmol ethylene/m^2/hr) than the R treatment (45 µmol ethylene/m^2/hr) ($p<0.001$). The highest rates of nitrogen fixation occurred in cover soils with the BCR treatment (Figure 2). Specifically, *H. alpinum* (*) had higher nitrogen fixation than *L. arcticus* (*) in CBCR and *O. campestris* had higher nitrogen fixation in cover soils ($) than in tailings ($) for the BCR treatment (Figure 2).

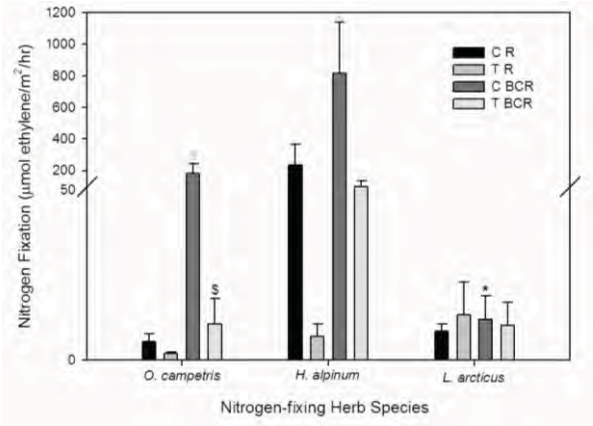

Figure 2. Nitrogen fixation by three herb species in four different treatments after 12 weeks for growth in a greenhouse trial. The treatments are cover soils with rhizobia (CR), tailings with rhizobia (TR), cover soil with biochar and rhizobia (CBCR) and tailings with biochar and rhizobia (TBCR). Bars are means with SE.

Biological Soil Crust Growth Chamber Trial

Establishment and growth of biological soil crusts derived from mature crusts on-site (S) were highly successful on both tailings and cover soils within the growth chamber. Lichens, mosses, *Nostoc* spp. and the occasional recruitment of naturally occurring grasses were observable after 10 weeks of incubation (Figure 3).

Figure 3. Establishment and growth of biological soil crust from a slurry derived from mature crust found at Keno Hills on cover soils immediately after application (a) and following 10 weeks of incubation in the growth chamber (b).

There were no significant differences in net photosynthesis or dark respiration between the different cyanobacterial soil amendments (ANOVA, p=0.47 and p=0.17 respectively). However, biochar may influence both photosynthesis and respiration, although these differences are likely strongly influenced by the substrate type and cyanobacterial amendment. When only treatments in tailings were considered, treatments with biochar tended to have higher rates of net photosynthesis (ANOVA, p=0.09). Slurry amendment treatments with biochar had lower rates of dark respiration on both tailings and cover soils (ANOVA, p<0.05).

All slurry treatments created from biological soil crusts harvested from Keno (*) had significantly higher rates of nitrogen fixation compared with both the pure *Nostoc commune* culture (*) and dried *Nostoc* sp. slurries (*) (ANOVA, p<0.001 for all comparisons; Figure 4), with the exception of the ND T and S T treatments which were not significantly different ($) (ANOVA, p=0.09).

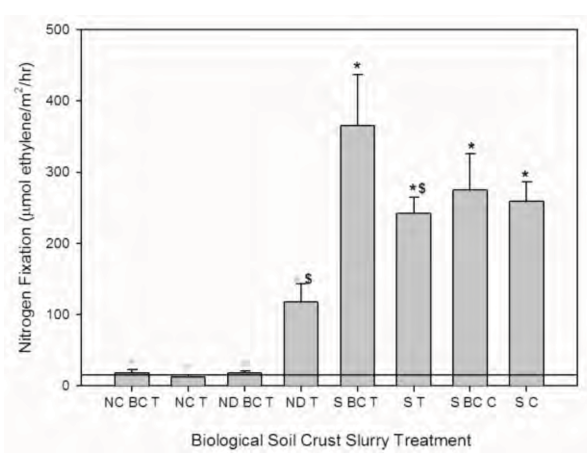

Figure 4. Nitrogen fixation rates of biological soil crust slurry treatments after 101 days in a growth chamber experiment. Treatments are Nostoc commune with biochar (NC BC T) and without biochar (NC T) on tailings, Nostoc spp. dried with biochar (ND BC T) and without biochar (ND T) on tailings, Biological soil crust slurry with biochar (S BC T) and without biochar (S T) on tailings and on cover soils (S BC C and S C) respectively. Bars are means with SE. Reference line indicates the nitrogen fixation detection limit.

Both the NC and ND treatments were generally below our detection limit for nitrogen fixation with the exception of the ND T treatment. Inclusion of biochar in the ND treatment resulted in significantly lower nitrogen fixation rates ($p<0.001$). There were no significant differences in nitrogen fixation between the substrate types (tailings versus cover) for the slurry treatment (S) ($p = 1.00$) or for the slurry treatment with biochar (S BC) ($p = 0.99$). Comparison of biochar treatments within each substrate type were also not significantly different (tailings, $p = 0.99$; cover soils, $p = 1.00$).

DISCUSSION

Nitrogen-Fixing Herb Greenhouse Trial

The main limiting factors for establishment of vegetation on mining impacted soils, especially on tailings, are high levels of acidity, low nutrient content, especially nitrogen, low water-holding capacity, low organic matter and excess salinity (Chan et al. 2003; Petrisor et al. 2004; Ye et al. 2002). Identification of northern native species that are able to overcome these limitations is needed. We had relatively high germination rates with the exception of *O. campestris*, which is likely due to limitations in our seed preparation procedure. *L. arcticus* had the highest above and belowground biomass and no differences in biomass were observed between the tailings and cover soils. In addition, we found *L. arcticus* to have the highest rate of nodulation compared with the other species examined. Therefore, of the species examined here, *L. arcticus* may be an important species to consider in restoration efforts aimed at promoting biomass accumulation and N input.

Nodulation and nitrogen fixation only occurred in samples that were given a rhizobia innoculum, indicating that nodulation in these soils is unlikely to occur naturally within a few months and the use of nitrogen-fixing species in northern reclamation may require microbial amendments. Nodulation may play a critical role in the growth and biomass accumulation of plants. Chan et al. (2003) observed that the nitrogen fixing *Sesbania* spp. had more nodules when they were inoculated and inoculated plants generally produced higher biomass.

Surprisingly, in our greenhouse trial we found the biochar only treatment (BC) had lower rates of germination, aboveground and belowground biomass. Under less harsh environmental conditions (i.e., watered every second day) and in soils with higher water-holding capacity, biochar may play a less important role in promoting germination. We found nitrogen fixation for both the R and BCR treatments, but the BCR treatment had significantly higher rates. Therefore, a combination of both a rhizobia innoculum and biochar may best promote nitrogen fixation and hence N input on mine impacted sites. Robertson et al. (2012) did not find any differences in the rate of nodulation of *Pinus contorta* or *Alnus viridis ssp. sinuata* in sub-boreal forest soils amended with biochar. However, nitrogen fixation was observed to continue for longer in biochar-treated systems versus non-biochar treated systems.

Most nitrogen-fixing microorganisms are thought to have an optimum soil pH near 7 and a higher diversity of free-living nitrogen fixers has been detected in tailings with a more neutral pH (Zhan and Sun 2011). In another study conducted *in-situ* on the Valley Tailings we found that the combined treatment of dolomite lime (54.6% $CaCO_3$, 41.5% $MgCO_3$) and bonemeal biochar resulted in higher germination and aboveground biomass of native grasses (Stewart et al. 2013) indicating that bonemeal biochar may reduce the availability of toxic heavy metals. Under suitable pH conditions inoculation can increase plant resistance to some toxic metals by reducing plant uptake, while also stimulating the absorption of nutrients (Petrisor et al. 2004).

Biological Soil Crust Growth Chamber Trial

The establishment, growth and nitrogen fixation of BSCs applied as slurries derived from mature crusts on-site (S) was highly successful with establishment of lichens, mosses and *Nostoc* spp. globules within 3 months. Xiao and Zhao (2008) also demonstrate that it is feasible to inoculate and cultivate artificial BSCs using a method of crushing and broadcast sowing natural BSC collected on-site. In their study, artificial crust coverage reached 30 to 60% and the main BSCs components were the same as the collected crusts. Nitrogen fixation from dried *Nostoc* sp. (ND) and pure *N. commune* culture (NC) were very limited. Nitrogen fixation rates for NC were below our detection limits and it is likely that these cyanobacteria were unable to survive and/or fix nitrogen on the tailings and cover soils. The rapid transition from an aqueous (i.e., growth medium) to solid (i.e., tailings and soils) environment may account for the low success rate with this type of amendment.

N. commune form macroscopic colonies that consist of extracellular polysaccharides (EPS) and filamentous cells embedded in EPS (Tamaru et al. 2005). Extracellular polysaccharides of *N. commune* are crucial for the stress tolerance of cellular functions, including photosynthesis, respiration and nitrogen fixation and may serve as a sink for excess energy during desiccation and during freezing and thawing (Tamaru et al. 2005). In the environment these cyanobacteria are exposed to natural drying and wetting cycles and *Nostoc* cells produce EPS in response to stress conditions. However, laboratory cultures that have not been exposed to these desiccation cycles generally have only small amounts of EPS and therefore, may be highly sensitive to desiccation (Tamaru et al. 2005).

We did not observe any significant differences in net photosynthesis, dark respiration or nitrogen fixation between the different biochar treatments. However, we did observe slight increases in net photosynthesis with biochar on tailings and slightly higher nitrogen fixation for slurries (S) with biochar, which may be related to increased moisture availability for cyanobacteria in this more porous well-drained substrate. Studies indicate that the variability of carbon fixation and nitrogen fixation are largely dependent on the amount of time BSCs are wet and the successional stage of crust development (Li et al. 2012; Zielke et al. 2002, 2005). Nitrogen fixation changes with crust succession due to changes in species composition, increases in biomass during BSCs development and increasing polysaccharide material, which slows water loss and lengthens activity time (Belnap 1996; Zhao et al. 2010). Although, our slurry treatment (S) would represent a later successional stage of crust development (i.e., dominance of lichens and mosses), all of our crust treatments were of the same age and all had the same moisture treatment (i.e., watered every second day), which may in part account for why we did not detect any significant differences in photosynthesis between our different BSC treatments. Field studies are needed to examine the establishment, growth and nitrogen fixation rates with biochar under natural conditions. In addition, further studies are required to determine if biochar does increase moisture availability to cyanobacteria.

CONCLUSIONS

Development and identification of soil amendments and native species directly impacts eco-restoration revegetation success while simultaneously helping to remediate soils affected by pollutants. Currently there are very few native species available for restoration in northern Canada. Our study indicates that native nitrogen-fixing herbs and biological soil crusts demonstrate a strong potential for use in restoration efforts in the North.

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NORTHERN BIOCHAR FOR NORTHERN REMEDIATION AND RESTORATION

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ABSTRACT

Biochar is a soil amendment that results from heating various biological ingredients, such as wood, fish or animal bone under oxygen limited conditions and has proven to promote plant growth, as well as, hydrocarbon degradation at contaminated sites in southern climates. We are working to identify different types of biochar that promote hydrocarbon degradation, as well as, re-vegetation success in mining-contaminated northern soils. Preliminary results from hydrocarbon contaminated soils indicate that under frozen conditions, 3% biochar was significantly effective, reducing the F2 and F3 fraction by up to 22%. In northern sandy soils biochar may improve the texture of the soil, enhancing water holding capacity, porosity, surface area and the availability of water under frozen conditions, which in turn may stimulate microbial activity that appears to be a driving factor in petroleum hydrocarbons degradation. The application of a biochar and lime treatment to mine tailings led to significantly higher germination rates and aboveground biomass compared with fertilizer only. There are a number of potential mechanisms by which biochar may have influenced germination and growth, including retention of soil moisture, increased temperature at the surface due to low albedo, increased nutrient retention and reduction in bioavailability of heavy metals.

Key Words: Hydrocarbons, Heavy Metals, Mine Tailings, Re-Vegetation, Frozen Soils.

INTRODUCTION

Bioremediation and restoration in northern Canada is a slow process with unique challenges related to cold-climate soil chemistry, short growing seasons and geographical isolation. Both hydrocarbons and heavy metals are common pollutants in the North (Mohn and Stewart 2000; Thomas et al. 1992). With increasing industrial activity in northern Canada there is a growing need to develop northern-specific remediation and restoration technologies.

Conventional methods of hydrocarbon remediation rely on fertilizer additions and soil turning to stimulate the microbial community to catabolize organic contaminants; however this approach has not proven to be consistently successful in polar environments (Paudyn et al. 2008). Biochar is a novel amendment that results from the oxygen-limited pyrolysis of various biological ingredients, which has received interest because of its ability to enhance growth and carbon storage in agricultural soils (Lehmann and Joseph 2009; Lehman et al. 2006). Biochar could have parallel effects on petroleum hydrocarbon (PHC) contaminated soil and has the potential to increase bioremediation rates. Few studies have been conducted on PHC contaminated soils in cold environments, but there have been successful reports of increased PHC degradation rates in biochar amended soils as compared to the control (Dias et al. 2012; Theis and Rillig 2009). However, the physical, chemical, and biological mechanisms driving these results are not well understood and further investigation into these mechanisms is required. Aged-diesel contaminated soil from Iqaluit, NU was used in a laboratory trial which investigated the use of biochar as an amendment to enhance PHC degradation. The aim of this study is to determine if biochar enhances PHC degradation in northern soils and to link this degradation to θ_{liquid} content and its affect on the soil microbial community.

Due to the lack of organic matter, low pH and high metal content the conditions on many mine sites may be too harsh for vascular plants to establish. Hence, the use of soil amendments may be necessary to allow for successful germination and growth. Several studies have found biochar can result in significant decreases in the bioavailability of heavy metals associated with mine impacted soils (Beesley and Marmiroli 2011; Fellet et al. 2011; Namgay et al. 2006) and simultaneously improve physical, chemical and biological soil properties (Laird et al. 2010). Biochar has many benefits for the environment and has been investigated extensively in southern climates, but very few studies have examined its use in northern mine site reclamation and restoration. The aim of this study is to examine various soil amendments and native species to determine optimal formulations that improve soil conditions and promote long term re-vegetative success in northern mine impacted soils. Determining the most appropriate reclamation and restoration option is heavily dependent on site characteristics and the contaminants of concern; however, development of soil amendments that are effective in northern soils with multiple contaminants may provide important cost-effective technologies.

METHODS

Hydrocarbon Remediation Trial

The soil had a background TPH concentration of 600 mg kg^{-1}, neutral pH (7.5), sandy texture (93.9% sand) nutrient deficiencies (i.e., 0.48 NO_3 mg kg^{-1}, NH_3 0.95 mg kg^{-1}, PO_4 0.23 mg kg^{-1}), and low moisture (9.81 gg^{-1}), and organic carbon contents (0.67% OM). Prior to trial setup, the soil was homogenized and amended in batches with urea (46-0-0) and mono-ammonium phosphate (11-52-0) fertilizer, compost, and bonemeal biochar. After the addition of amendments to each treatment, the soil was re-homogenized, weighed into amber glass vials and plugged with sterile cotton balls to prevent anaerobic conditions. Water was added to each vial to bring the soil up to 60% of field capacity, which was maintained throughout the experiment.

PHC degradation under thawed conditions was compared using control and amended treatments; fertilizer, 3% (w/w) bonemeal biochar plus fertilizer, 6% (w/w) bonemeal biochar plus fertilizer, 5% (w/w) compost plus fertilizer, and 10% (w/w) compost plus fertilizer. The vials were incubated under thawed (10°C) conditions and destructively sampled over 0, 30, 60, and 90 days. PHC degradation under frozen conditions was also compared using control and amended treatments; fertilizer and 3% (w/w) bonemeal biochar plus fertilizer. These vials were incubated at -5°C and destructively sampled over 0, 30, 60, and 90 days. Biochar and compost was added on a weight to weight (w/w) basis while urea and mono-ammonium phosphate fertilizer was added at a C_{TPH}:N:P ratio of 100:9:1 (Chang et al. 2010).

After destructive sampling, the vials were stored at -80°C until they were analyzed for total petroleum hydrocarbons (TPH) using ASE and GC-FID analysis, for θ_{liquid} content using time-domain reflectometry (TDR) methods and for total PHC-degrading microbial populations using Most Probable Number (MPN) counts. Gravimetric moisture content was determined for each sample and TPH was extracted using an ASE 200 Accelerated Solvent Extraction System; a patented technique developed by Dionex® to extract liquids from solid and semisolid matrices (Thermo Scientific, Sunnyvale, CA). Following ASE, the PHC extracts were run through sodium sulphate and silica gel cleanup columns to remove water and polar organic compounds, respectively (CCME 2001). Finally, PHC concentration in fractions F2 and F3 were quantified using a Varian 3800 CP gas chromatograph fitted with a flame ionization detector (Varian, Santa Clarita, CA).

Volumetric water content was measured by TDR, a technique that measures the dielectric constant of the medium. Because the dielectric constant of water ($K_{water} = 80$) is much higher than other soil constituents ($K_{air} = 1$; $3 > K_{soil} < 7$;$< K_{ice} = 3.2$), the dielectric constant of the medium is proportional to the amount of θ_{liquid} (Topp et al. 1980). This technique is capable of measuring small quantities of θ_{liquid} down to 0.05 m^3 H_2O m^{-3} soil (Topp et al. 1980). Soil θ_{liquid} content was measured using calibrated TDR probes and a Tektronix 1502B cable tester (Tektronix, Beaverton, OR). Waveform data were collected, compared to the gravimetric moisture content and plotted as a linear equation to obtain calibration.

Culturable, aerobic, diesel-degrading microbial populations were enumerated using 96-well microtiter plates. Filter sterilized No. 2 fuel oil (F2) was used as the selective substrate to quantify total PHC-degrading microorganisms (Haines et al. 1996). Minimal salts medium (180 µL) was added to each well and supplemented with 3 µL of 5000 mg/kg^{-1} F2 diesel (Yergeau 2009). Soil samples were diluted in a saline phosphate buffer solution (PBS); 1 g of soil was mixed with 9 mL of PBS to create a 1:10 dilution. The first row of wells was inoculated with 20 µL of the 1:10 dilution. The subsequent wells in each column were inoculated by transferring 20 µL from the previous well to create a dilution series ranging from 10^{-2} to 10^{-7}. The last row remained un-inoculated to serve as a sterile control. Following incubation at 23°C for three weeks, 50 µL of filter-sterilized iodonitrotetrazolium (INT) violet (3 g/L) was added to identify positive wells (Haines 1996). In positive wells, INT is reduced to an insoluble formazan that deposits as a red precipitate in the presence of respiring microorganisms (Wrenn and Venosa 1996). Red or pink positive wells were scored after an overnight, room-temperature incubation with INT. Final cell numbers were derived using an MPN calculator (Jarvis et al. 2010).

Tailings Soil Amendment Trial

The Keno Hill Silver District is one of the world's highest-grade silver districts located 330 km north of Whitehorse, Yukon, Canada. It is estimated that approximately 4,050,000 tonnes of tailings were deposited at a 130 ha site, known as the Valley Tailings, located in the McQuesten River Valley (63°55'26.4N, 135°29'76.1W). The tailings are highly variable with a pH ranging from 5.7 to 8.4 and texture varying from silt loam to sand. The tailings exceed the Canadian Council of Ministers of the Environment (CCME) industrial soil quality guidelines for allowable levels of Antimony (Sb), Arsenic (As), Cadmium (Cd), Copper (Cu), Lead (Pb), Silver (Ag), Titanium (Ti) and Zinc (Zn).

The original site consisted of boreal vegetation including small trees (*Picea glauca* (Moench) Voss, *Picea mariana* P. Mill., *Populus tremuloides* Michx., and *Populus balsamifera* L.), as well as, shrubs (*Salix* spp.) and moss mats (*Sphagnum* spp., *Pluerozium* spp.). Vegetation in the Valley Tailings area was eventually covered by tailings, which range from 0.1 to over 4m in thickness (Keller et al. 2010). Mean

January and July temperatures are -26.9°C and 15.6°C respectively. However, summer temperatures in the region can exceed 25°C and winter temperatures -50°C. The average total precipitation is 322 mm and discontinuous permafrost is found throughout the area (Clark and Hutchinson 2005).

We examined 8 soil amendment treatments and 1 control treatment (fertilizer and seed only). Soil amendments were applied to the Valley Tailings using a randomized block design with 14 blocks (5m x 10m). Leaving at least 1 m^2 between treatments, the 9 treatments (1 m^2 each) were randomly assigned within each block. Each soil amendment treatment had 14 replicates for a total of 126 plots.

The soil amendment treatments included the following materials: biochar (1 kg/m^2), smectite (calcium bentonite) (750g/m^2), dolomite lime (54.6% $CaCO_3$, 41.5% $MgCO_3$) (484 g/m^2), wood mulch (193 g/m^2) and the tackifier Guar Gum (12.7 g/m^2). The biochar (BC) was a phosphorus-rich bonemeal biochar (2-14-0) pyrolyzed at a temperature of 450°C for 6 hours. The grain size of the finished product was ≤ 2mm. The 8 soil amendment treatments were: biochar (BC); biochar and smectite (BCS); biochar and lime (BCL); smectite and lime (SL), biochar, smectite and lime (BCSL); mulch (M); mulch and lime (ML); biochar, smectite, lime and mulch (BCSLM). All soil amendment treatments were fertilized (19:19:19) at a rate of 110kg/ha and seeded at a rate of 30kg/ha with a native grass seed mix, containing Violet Wheatgrass (*Agropyron violaceum* Hornem. – 40%), Sheep Fescue (*Festuca ovina* L. – 23.3%), Rocky Mountain Fescue (*Festuca saximontana* Rydb. – 23.3%) and Glaucous Bluegrass *(Poa glauca* Vahl. – 13.4%). Guar Gum was present in all treatments. All of our amendments were applied as a slurry to simulate delivery via hydroseeding using 7.5L of water per m^2.

Plots were established between July 1-3, 2012. Germination was determined by counting the number of emerging stems in 0.25m^2 subplots for each replicate from July 25-26, 2012. Above and belowground biomass were sampled from August 27-29, 2012 in 0.5 m^2 subplots for each replicate. All biomass was dried at 90^{o}C for 36 hours before determining dry weights.

Data from the hydrocarbon and soil amendment trials were examined to ensure the assumptions of Analysis of Variance (ANOVA) were met and log transformations were performed on some variables. A two-way ANOVA was used to detect significant differences between treatments over the duration of the hydrocarbon experiment. Soil amendment trial data was analyzed using a blocked ANOVA. All analyses were conducted in R (R package version 2.1.50).

RESULTS

<u>Hydrocarbon Remediation Trial</u>

Amended and control treatments had variable effects on TPH degradation under frozen versus thawed conditions (Figure 1). Under frozen conditions, 3% (w/w) bonemeal biochar significantly reduced the F3 fraction by 22%, as compared to fertilizer which only stimulated degradation by 3%. In the F2 fraction, both the 3% (w/w) biochar and fertilizer amendments stimulated degradation, by 42% and 33%, respectively. None of the amended and control thawed treatments exhibited a significant effect on TPH degradation, as compared to the standard fertilizer treatment.

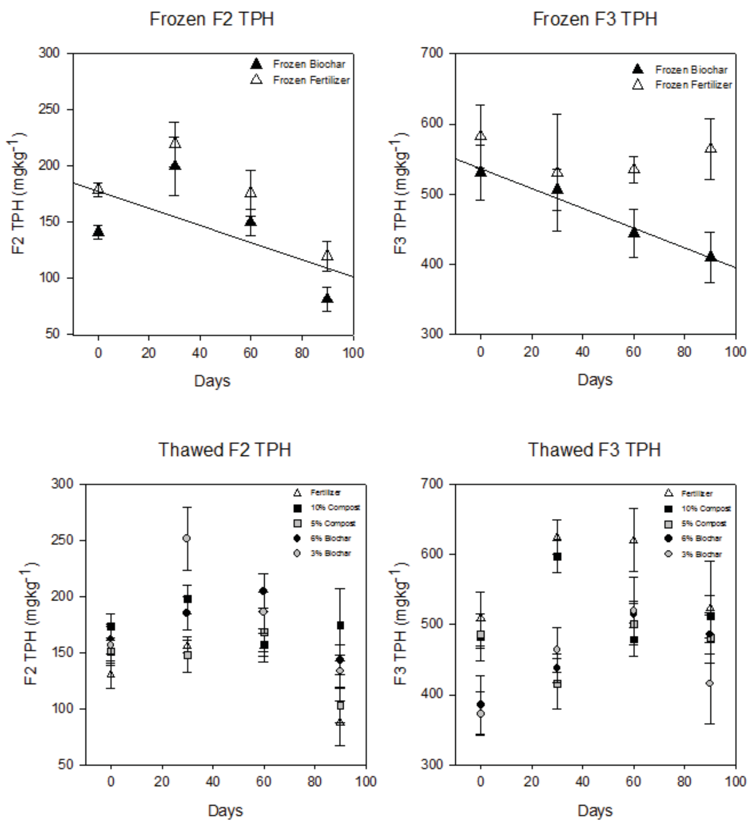

Figure 1. *PHC degradation in fractions F2 and F3 was monitored over 90 days in frozen and thawed soils. Under thawed conditions, no significant effect on degradation was observed in any treatments. However, under frozen conditions, 3% (w/w) biochar decreased concentrations in the F3 fraction, as compared to the standard fertilizer treatment. Both 3% (w/w) biochar and the fertilizer treatment stimulated degradation in the F2 fraction. The addition of biochar appears to have a significant effect on heavier PHC compounds, under frozen conditions.*

Under frozen conditions, θ_{liquid} of soil amended with 3% (w/w) biochar was higher than the fertilizer control, although differences were not significant. Trends suggest that biochar amendments can increase the θ_{liquid} content in frozen soils by increasing surface area and soil-water interactions.

Most probable number (MPN) counts indicated that total PHC-degrading populations were significantly higher in soil amended with 3% (w/w) bonemeal biochar incubated under frozen conditions (Figure 2a). Surprisingly, MPN with 3% (w/w) bonemeal biochar incubated under thawed conditions was not significantly different than the fertilizer control (Figure 2b).

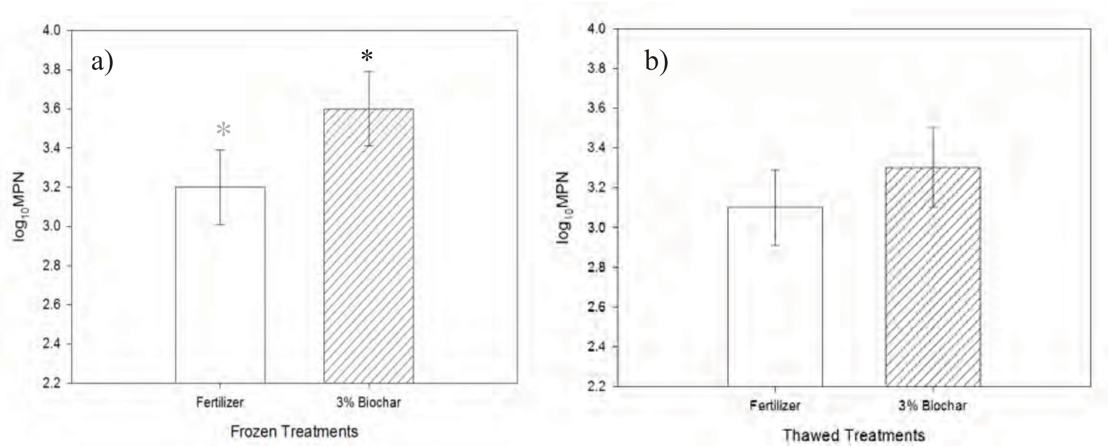

Figure 2. MPN counts from soils amended with bonemeal biochar as compared to the control treatment, incubated for 90 days under frozen (a) and thawed (b) conditions. Different shaded asterisks (*) indicate significantly different PHC degradation (ANOVA, p<0.05).

<u>Tailings Soil Amendment Trial</u>

Of the 8 amendment treatments examined, the biochar and lime treatment (BCL) had significantly higher germination rates (Figure 3) and significantly higher aboveground biomass (Figure 4a) when compared with the fertilizer only control treatment (CT). The BCL treatment also had higher germination rates than treatments that included mulch (i.e., M, ML and BCSLM).

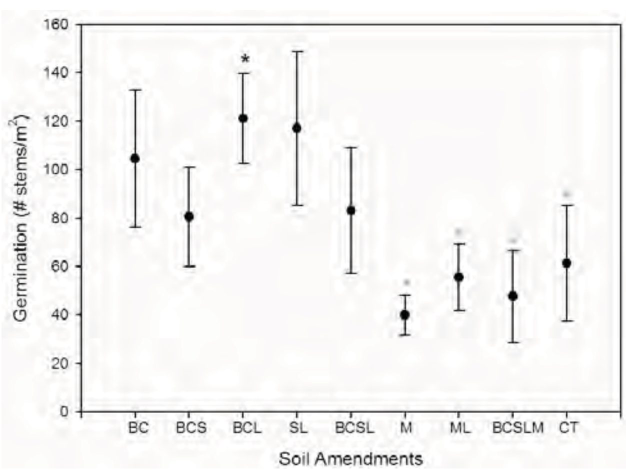

Figure 3. Germination of native grasses with different soil amendments 22 days after application at the Valley Tailings, Keno, YT. Soil amendment treatments are Biochar (BC), Biochar and Smectite (BCS), Smectite and Lime (SL), Biochar, Smectite and Lime (BCSL), Mulch (M), Mulch and Lime (ML), Biochar, Smectite, Lime and Mulch (BCSLM) and Control with fertilizer only (CT). Germination of the BCL treatment is significantly higher (*) than the M, ML, BCSLM and CT treatments (*) (ANOVA, p<0.05 for all comparisons).

None of the soil amendment treatments had significantly higher belowground biomass compared with the control. Average belowground biomass was slightly higher in the BCL treatment (2.9 g/m^2) compared with the control (2.0 g/m^2), however not significantly. The SL treatment had significantly higher belowground biomass compared with BCS, BCSL, ML and BCSLM (Figure 4b).

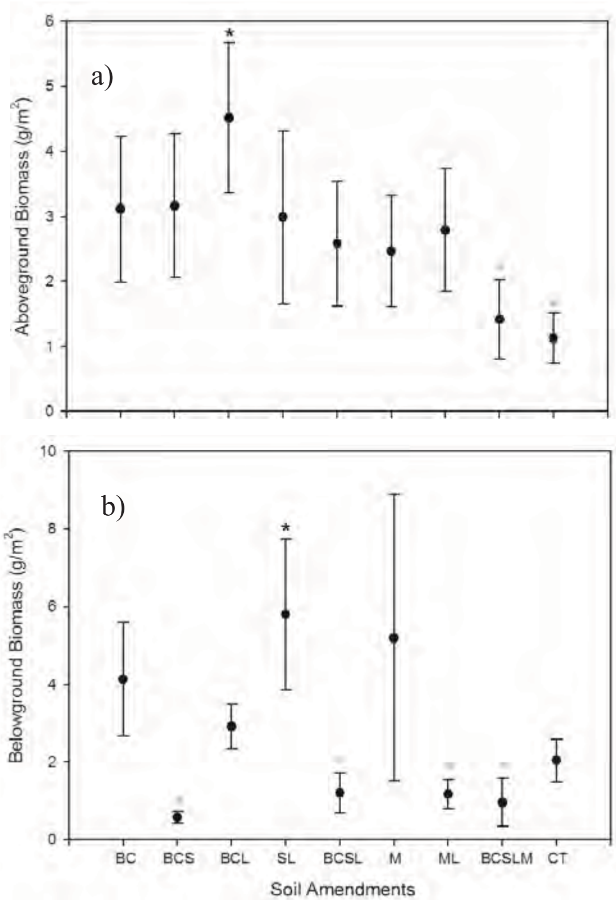

Figure 4. Aboveground (a) and belowground (b) biomass of native grasses with different soil amendments after 2 months grow on the Valley Tailings, Keno, YT. Soil amendment treatments are Biochar (BC), Biochar and Smectite (BCS), Smectite and Lime (SL), Biochar, Smectite and Lime (BCSL), Mulch (M), Mulch and Lime (ML), Biochar, Smectite, Lime and Mulch (BCSLM) and Control with fertilizer only (CT). The aboveground biomass of the BCL treatment is significantly higher (*) than the BCSLM and CT treatments (*) (ANOVA, p<0.05 for all comparisons). Note: Belowground biomass was only sampled within 5 plots per treatment.

DISCUSSION

Hydrocarbon Remediation Trial

Under frozen conditions, the addition of 3% (w/w) bonemeal biochar is effective at accelerating the degradation of PHC contaminated soils from a northern landfarm. Microbial activity appears to be a driving factor in the degradation of PHCs under frozen conditions, as amended soils had higher numbers of PHC-degrading microorganisms. Dias et al. (2012) similarly found higher PHC-degrading bacterial counts in biochar amended soils, as compared to the control. In soils with adequate nutrients, temperature, and θ_{liquid} there may be limited response from biochar applications (i.e., no significant differences in degradation were detected under thawed conditions). However, biochar can compensate for deficient soils that are limited by temperature, θ_{liquid}, nutrient supply rates and enhance microbial activity and PHC degradation. Areas with low rainfall and coarse textured or nutrient deficient soils have benefited the most from the addition of biochar to agricultural lands (Lehmann et al. 2006). Similar to previous studies, we found that biochar can increase θ_{liquid} in coarse textured soils under frozen conditions, due to increases in soil porosity and surface area (Amonette and Joseph 2009; Chan and Xu 2009; Downie et al. 2009).

Although the frozen and thawed treatments had similar soil properties and fertilizer additions, frozen soils are limited by temperature, θ_{liquid}, and nutrient and gas exchange. Liquid water (θ_{liquid}), limits microbial activity in frozen soil as unfrozen water allows the diffusion of microbial substrates and waste products (Ostoumov and Siegert 1996). Changes in θ_{liquid} may directly impact microbial activity and subsequently alter nutrient supply rates, which can also influence microbial activity and PHC degradation in frozen soils (Harvey et al. 2008; Harvey 2011). Trends suggest that θ_{liquid} content is higher in biochar amended soils so we expect that there will be significant differences between nutrient supply rates in biochar amended versus the control treatment under frozen conditions. We also expect that the abundance of catabolic functional genes will also be higher in biochar amended soils as compared to the control, under both frozen and thawed conditions. In subsequent experiments, we will attempt to examine linkages between functional gene copy numbers and microbial biomass to θ_{liquid} content, which influences nutrient supply rates and degradation rates.

If a deficient soil is lacking in essential macro and micronutrients, biochar additions may not directly supply enough nutrients, therefore, fertilizer additions may be necessary. However, biochar has a high cation-exchange capacity (CEC) and can reduce nutrient losses in the soil (Beesley et al. 2011); therefore, for soils with limited CEC or θ_{liquid}, biochar additions may be beneficial.

Current bioremediation strategies are targeted toward the short summer months (2 to 4 months/year), but this is often an insufficient amount of time to meet remediation targets and environmental criteria. Our results suggest that with biochar, remediation may be active under frozen conditions extending remediation periods and providing new cost-effective remediation strategies.

Tailings Soil Amendment Trial

Phytostabilization of mine tailings is highly difficult, not only due to phytotoxic effects of elevated heavy metal concentrations, but also due to extreme pH values, low fertility, low water-holding capacity and unfavorable substrate structure (Fellet et al. 2011). The BCL treatment promoted both germination and growth of the native grass seed mix. There are a number of potential mechanisms by which biochar may have influenced germination and growth including retention of soil moisture, increased temperature at the surface due to reduced albedo, increased nutrient retention and reduction in bioavailability of heavy metals. Biochar has a high surface area and porosity and can increase the water-holding capacity of the soil (Fellet et al. 2011). Biochars with a high volume of macropores greater than 50 nm diameter can make water available to plants (Lehmann and Joseph 2009). Higher moisture availability at the surface may have contributed to both higher rates of germination and initial growth.

The functional groups of biochar influence the sorption process depending on the nature of their surface charge so that both transition metals and non-transition metals can be sorbed onto the surface of biochar particles (Amonette and Joseph 2009). Several studies have found reduced availability or leachability of heavy metals following the application of biochar to contaminated soils (Beesely et al. 2011). In situ immobilization of metals using soil amendment processes is increasingly being considered as an effective and low cost remediation alternative (Fellet et al. 2011; Kumpiene et al. 2008; Mench et al. 2007). While liming is one of the oldest and most widely used metal immobilizing soil treatments, the effects of liming gradually reduce over time due to the dissolution and leaching of the liming agent, especially in highly acidic soils (Ruttens et al. 2010). Biochar however, is highly recalcitrant (Steiner et al. 2007) and its effects may persist over long time periods. In addition, most biochars tend to have neutral to basic pH and therefore commonly have a liming effect. However, it should be noted that this liming effect has been shown to increase As mobility and restrict re-vegetation (Beesely et al. 2011). Care should be taken to determine how various types of biochar may interact with the elements and conditions at a given site. Further studies are needed to examine the influence of biochar application methods and the adsorption and long-term immobilization of heavy metals with biochars.

WORK IN PROGRESS

In addition to the above studies our project is also working to overcome the logistical and technical hurdles associated with biochar production in the North, specifically in both Whitehorse, YT and Iqaluit, NU. Three further trials examining the use of biochars for hydrocarbon remediation are being undertaken: 1) a laboratory trial comparing the rates of PHC degradation in frozen soils amended with biochars made from wood, fishmeal and bonemeal feedstocks; 2) Land treatment facility trials in Whitehorse, YT and Iqaluit, NU to determine the specificity of biochar and role of direct additions of nutrients via biochar by evaluating PHC degradation under different biochar formulations; and 3) Land treatment trials in Whitehorse, YT and Iqaluit, NU to determine methods of applying biochar to soils (i.e., surface application, injection or homogenization) that optimize θ_{liquid}.

CONCLUSIONS

Soil amendments that promote the degradation of hydrocarbons, reduce the bioavailability of heavy metals, improve soil conditions and promote long term re-vegetative success are needed in northern Canada. By providing technological solutions for northern remediation and restoration, we aim to increase PHC bioremediation and mine restoration success in the North. Providing effective biochar formulations, optimal application rates and efficient application technologies specific to local restoration needs is essential to alleviate current challenges to cost-effective remediation.

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THE USE OF CHEMOX® TO OVERCOME THE CHALLENGES OF PHC CONTAMINATED SOIL AND GROUNDWATER AT REMOTE SITES

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ABSTRACT

A major theme for working in northern remote sites is overcoming logistical and technical challenges related to short season and limited access to remediate petroleum hydrocarbon impacted soil and groundwater. Chemical oxidation remediation approaches (ChemOx®[6]) can provide solutions to the unique challenges posed by these remote sites and allow expedited remediation within limited field seasons, often with less intrusive work activities common to other traditional remediation methods.

Successful chemical oxidation remediation integrates a strong understanding of oxidant chemistry and field application experience, whether using in-situ (ISOTEC®[7]) or ex-situ (EXCO®[8]) approaches. The use of hydrogen peroxide based oxidants has been widely accepted for expedited reactions with petroleum hydrocarbon contaminated mediums. However, historically, misperceptions and unreasonable expectations have often been applied to the design of these programs. By combining a strong application approach, unique transportation and logistics solutions, such as air transport, and working within a reasonable and realizable objective, chemical oxidation is well suited to handle the uncommon challenges of remote programs.

Key Words: Chemical Oxidation, Ex-situ, Catalyzed Hydrogen Peroxide, Petroleum Hydrocarbons, Remediation, Remote Site.

INTRODUCTION

Petroleum hydrocarbon contamination of soil and groundwater resources is a challenge to many industries. In some cases the remote nature and climate of these sites requires a solution that does not allow for, or limits, the effectiveness of conventional remediation options. With increased public awareness and stakeholder involvement, many of these conventional options, such as source removal and landfilling, are losing popularity for economic, social and sustainability reasons. As the distance and access to conventional remediation solutions increases, the remediation planning for remote and/or seasonally accessible contaminated sites often shifts to alternative solutions.

The general concepts of alternative remediation strategies are well known, with most of the concepts well documented in standard environmental engineering references. However, much progress and innovation largely goes unpublished, keeping clients uninformed about the advancements and options that exist,

[6] ChemOx® is a registered trademark for catalyzed hydrogen peroxide solutions, owned by TRIUM Environmental Inc.
[7] ISOTEC® is a registered trademark for in-situ chemical oxidation services, owned by TRIUM Environmental Inc.
[8] EXCO® is a registered trademark for ex-situ chemical oxidation services, owned by TRIUM Environmental Inc.

especially early in remediation planning phases when time may allow combined treatment strategies to be implemented without the pressure of last minute regulatory or external third parties. This lack of available reference material and the historical misapplication and unreasonable expectations often placed on these technologies have resulted in market hesitation and uncertainty towards adopting alternative processes. Strong applied knowledge, integrated with a qualified specialist and proactive establishment of realistic objectives can overcome these limitations.

HYDROGEN PEROXIDE CHEMISTRY AND APPLICATION MECHANISMS

Chemical oxidation is the degradation of contaminants by chemicals referred to as oxidants, such as catalyzed hydrogen peroxide (H_2O_2). Catalyzed H_2O_2 can be preferred for its oxidation potential, active/rapid destruction mechanisms (<24 hour reaction timeframes), clean (no residual ions) and pH balanced reactions, and cost effectiveness. For the reasons above, except for the destruction of the petroleum hydrocarbon parameters, any alteration of the subsurface geochemistry is quickly reversed due to the short reaction timescale and pH control, the extent of which is determined by the buffering capacity of the soil and water matrix, and the blending scenario of the catalyzed H_2O_2 (i.e., ChemOx®). A complete reaction of catalyzed H_2O_2 and petroleum hydrocarbons results in mineralization (degradation) of the contaminants to carbon dioxide (CO_2), water and heat. Catalyzed H_2O_2 also provides a post-reaction oxygenated environment, which when combined with base nutrients, encourages indigenous bacterial populations for longer term polishing of contaminants after treatment.

Often, the most important components to implementation of a catalyzed H_2O_2 remediation program are the expediency of the reaction and flexibility of application. However, the inherent limitation of chemical oxidation is the stoichiometric demands of high concentrations of the contaminants. This stoichiometric demand – combined with the depth, location, and extent of the contaminant impacts in the soil/groundwater mediums at each site – dictate the appropriate application method, whether in-situ or ex-situ. The general criteria for selection of an in-situ or ex-situ method are presented in Table 1.

Table 7. General Application Technique Selection Criteria

In-situ	Ex-situ
✓ Groundwater Bearing Zone	✓ Vadose/Unsaturated Soils
✓ Dissolved Phase/Down gradient Plumes	✓ Less than 4 metres below grade
✓ Greater than 4 metres below grade	✓ Low Permeability Unconsolidated Soils
✓ Permeable Unconsolidated Soils	✓ Aged/Weathered Mediums (i.e. Bio-Piles)
✓ Fractured Bedrock	✓ Single Events/Rapid Closure
✓ Multiple Events	

As indicated above, when impacted material exists primarily as dissolved phase and in the saturated groundwater zone, TRIUM will apply our in-situ chemical oxidation (ISOTEC®) approach. This approach involves installation of dedicated injection wells and multiple treatment events. Injection wells allow greater flexibility and lower costs for setting up an effective multi-event treatment program. Multiple events are necessary to capture the occurrence of "rebounding", which is a common concern of in-situ processes. Rebounding should be expected in most programs and is actually an indicator of success of the process, as contaminants must desorb from the soil to become available/contactable by the oxidant, which occurs in an aqueous phase reaction. Several reaction processes occur during an oxidation program that govern desorption of the contaminants including soil/water partitioning coefficients, and partial oxidization intermediates. Therefore, in-situ programs should always be considered on a longer time frame and with awareness of the required natural processes that must occur to support the treatment method.

When unsaturated conditions or low permeability soils are treated, TRIUM applies our ex-situ chemical oxidation approach (EXCO®). EXCO® is effective as it allows enhanced contact (i.e., mechanical mixing) between the oxidant and the soil in its original location, and also allows for optimization of the soil conditions prior to application. EXCO® applications can target either treatment in established treatment cells/pads or employ an in-place mixing method. The processes also allow the treatment to progress discretely with only the required volume of soil for treatment being open/exposed at any given time. This reduces equipment requirements, volatilization, cross contamination and other regulatory concerns, while ensuring that optimal treatment conditions are achieved in very short timeframes.

As the timeframe of treatment using the EXCO® approach with catalyzed H_2O_2 is measured in days, closure sampling of treated soils can occur within 72 hours, providing a rapid turnaround and higher degree of treatment certainty. In addition to economic advantages, by treating the material on site, the inherent safety risks and carbon offset associated with transporting impacted soils is greatly reduced. Many clients consider this reduced health and safety risk and sustainability measure a value add to the process, and will incorporate these soft costs in selecting a remediation technique.

CASE STUDY – REMOTE SITE

The advantages of the ChemOx® approach were evaluated through a technology matrix evaluation conducted by the client for a remote, air access only, site. The outcome of the review indicated that the logistics and expediency of the EXCO® process would be best suited to achieve the objectives of the program. Specific details and concentrations have been altered in this paper for client disclosure however the outline of the process remains consistent.

Project Logistics

Among many logistical considerations, the program was limited by a short field season and fly-in only access. On-site facilities and equipment requirements were addressed by the client that would allow the on-site operation to run safely, however a primary challenge of transporting the oxidant to site had to be overcome. As the EXCO® program would require several thousand kilograms of 50% catalyzed H_2O_2, an air transportation exemption was required. Under the Transportation of Dangerous Goods Regulation (TDG), the allowable air transportation of hydrogen peroxide was limited to 20% concentration in 5 litre containers. Working with the client, a review was completed with Transport Canada, which resulted in approval and issuance of an equivalency certificate allowing handling and transportation of the oxidants by air at between 20% and 60% concentration in 200 L drums (with over-packs).

Baseline Conditions of Impacted Soil

The impacted soil targeted for treatment at the site comprised a coarse-grained soil, in an alluvial area. Based on laboratory analytical results from early assessment programs at the site, light extractable petroleum hydrocarbons (LEPH) and heavy extractable hydrocarbons (HEPH) were reported at <3,000 parts per million (ppm). Upon further investigation once heavy equipment resources were available, LEPH and HEPH concentrations were encountered ranging from non-detect to one order of magnitude higher than anticipated.

Due to the depositional setting, it was suspected that the coarse-grained soils could maintain elevated natural total organic carbon (TOC) in the water phase. Elevated TOC often results from naturally occurring phytogenic interferences, and can act as a scavenger of the oxidant, which is referred to as Natural Oxidant Demand (NOD). High carbon contents also sequester petroleum hydrocarbons making them less "available". This can be an important consideration in determining plume migration and contaminant transport.

A background TOC assessment was conducted at this site, and an analytical interference of phytogenic compounds was observed to cause bias to standard analytical methods and analysis. Most jurisdictions have protocols for determining "background" concentrations including methods for making the distinction between naturally occurring phytogenic hydrocarbons and petrogenic petroleum hydrocarbons. These methods begin with taking background soil samples that are representative of the area in contest, and requesting additional laboratory preparation, sample cleanup or analysis. For phytogenic hydrocarbons a silica gel cleanup is one laboratory method that can be applied, and is not necessarily automatically included in the standard method for analyzing petroleum hydrocarbons. For example, in British Columbia, this silica gel cleanup procedure is refereed to as a Method 10 cleanup. Soil samples were collected from the area surrounding the site to characterize the background concentrations/interferences and for consideration in adjusting remediation guideline endpoints. Additionally a background cell was selected to determine if the catalyzed H_2O_2 could introduce interference to the analytical method by reacting with the phytogenic compounds.

The results of the phytogenic and oxidant interference testing indicated that interference was observed for HEPH parameters, ranging between 300 ppm and 500 ppm. These "Background Correction" values could then be used to adjust HEPH concentrations when evaluating the chemical oxidation process performance.

Series of Tests

The program was conducted as a series of tests, which consisted of i) preliminary bench-scale test, ii) field pilot application, and iii) secondary bench-scale test. The field pilot application program was conducted following the preliminary bench-scale test, and was based on the limited sample and analytical data available at that time. Once mobilized to site for the pilot application, concentrations of LEPH/HEPH in the main treatment area were found to be higher than the concentrations used in the preliminary bench-scale program. This was a limiting factor for the pilot application program as the stoichiometric oxidant demand to address these higher concentrations exceeded the amount of catalyzed H_2O_2 mobilized to the site. The objective of the pilot application was shifted to focus on a demonstration of potential reductions at these higher concentrations and a secondary bench-scale test was conducted following the pilot program. The secondary bench-scale test would prove out the oxidant's ability to achieve the required endpoints and provide projections for financial thresholds and technology optimization factors.

1st Bench-scale Test

Prior to commencing the field pilot a bench-scale test was conducted on the initial soils collected from the site. The oxidant loading from the bench-scale program that performed best was then used to design the field pilot program. In the first bench-scale testing program, reductions of between 60% and 80% were reliably reproduced, indicating that the catalyzed H_2O_2 could achieve the preliminary program objectives. Based on the results of the 1st bench-scale, an oxidant loading of >100 kg/m^3 (50% catalyzed hydrogen peroxide) was deemed most effective to overcome the elevated TOC and petroleum hydrocarbon concentrations.

Field Pilot Scale Test

The oxidant loading and design parameters from the bench-scale were carried out in the field over a two week period. Two pilot cells of 50 m^3 each were prepared at the site, and each cell was treated with >100 kg/m^3 of the catalyzed H_2O_2. Mixing was achieved using an excavator equipped with a standard digging bucket. As the soils were found to be at optimum moisture conditions, the catalyzed H_2O_2 could be applied directly to the soil aliquots with dilution occurring from the moisture already present in the soil.

As the actual LEPH/HEPH concentrations were higher than the concentrations used in the first bench-scale program, the expectations of the oxidant performance were adjusted to determine the maximum reductions that could be achieved. Throughout the program, composite and discrete samples were collected to track reductions throughout the program and validate the effectiveness of each sampling technique.

Petroleum hydrocarbon samples were collected on Day 1 (representing pre-treatment concentrations), on Day 3 of treatment (to capture assumed homogenous soil concentrations due to mixing) and at the completion of the program.

The petroleum hydrocarbon levels in the homogenized samples were reduced on average by 33% compared to the pre-treatment (unhomogenized) samples. The petroleum hydrocarbon levels in the post-treatment samples from Cell 1 and Cell 2 had an average reduction of 36% and 23% respectively, compared to the homogenized concentrations.

The percentages described above, can be further expressed by the overall reduction of each relative to the oxidant performance expectations for the oxidant loading that was applied to the site. In Cell 1, the overall reductions achieved were 1.3 to 2.3 times greater than the theoretical reductions. In Cell 2, the overall reductions achieved were -2 to 1.1 times greater than the theoretical reductions. The sample point representing the -2 times reduction is considered to be a result of sample homogeneity and process interference (other activities that were being conducted at the time).

Secondary Bench-scale

Following the field pilot program, a secondary bench-scale test was conducted with soil selected from an area of high impact at the site (remaining after the field pilot). The purpose of the secondary bench-scale test was to demonstrate the oxidant performance to achieve the required endpoints and provide projections for financial thresholds and technology optimization. One of the important variables arising from the field program was the timeframe between applications to allow equilibrium between soil and water due to the elevated TOC. The secondary bench-scale dosing strategy was modified to include a 5 to 7 day pause between ChemOx® applications to determine the effect of the equilibrium "stall".

Three applications of the catalyzed H_2O_2 were completed, observing the 5 to 7 days between treatments. At the completion of the bench-scale, all three sets of samples achieved the regulatory guideline objective. The average reduction of petroleum hydrocarbon concentrations in all three sets of samples was >90%, and was able to demonstrate that catalyzed H_2O_2 could achieve the program endpoints.

During the second bench-scale program, the reductions observed in the field were plotted against the theoretical oxidant demand. It was observed that the oxidant demand to achieve the reductions noted was trending in a non-linear curve, and the oxidant was "outperforming" the theoretical stoichiometric oxidant demand. This is believed to occur as a result of the amount of "available" petroleum hydrocarbons being greater as the concentration increased. The performance of the second bench-scale program was then plotted after the field data points on this curve to determine how the efficiencies of the catalyzed H_2O_2 would be affected as the concentrations decreased. Figure 1 illustrates this non-linear curve and demonstrates the program performance to illustrate the actual oxidant demand required to achieve the program objectives.

Although, the findings indicate that an oxidant dosing slightly greater than the theoretical oxidant demand would be required, the efficiencies of the delayed treatment timeframes (5 to 7 days between dosing) could overcome the scavenging and sequestering effects of the high TOC. The blue line represents the laboratory reported values, with the red line showing the adjusted values based on the phytogenic interference caused by the elevated TOC. Actual values cannot be presented, however the trend of the graph remains accurate.

DISCUSSIONS AND CONCLUSIONS

Throughout the sequence of tests, the application of TRIUM's EXCO® process using catalyzed H_2O_2 demonstrated that a chemical oxidation approach could achieve the safety and technical endpoints established for the program. The field pilot EXCO® program was conducted efficiently and demonstrated that high concentration hydrogen peroxide could be transported safely by air. Actual site conditions revealed that petroleum hydrocarbon concentrations were greater than estimated and therefore reductions to achieve the regulatory guidelines were not achieved during the field pilot given the

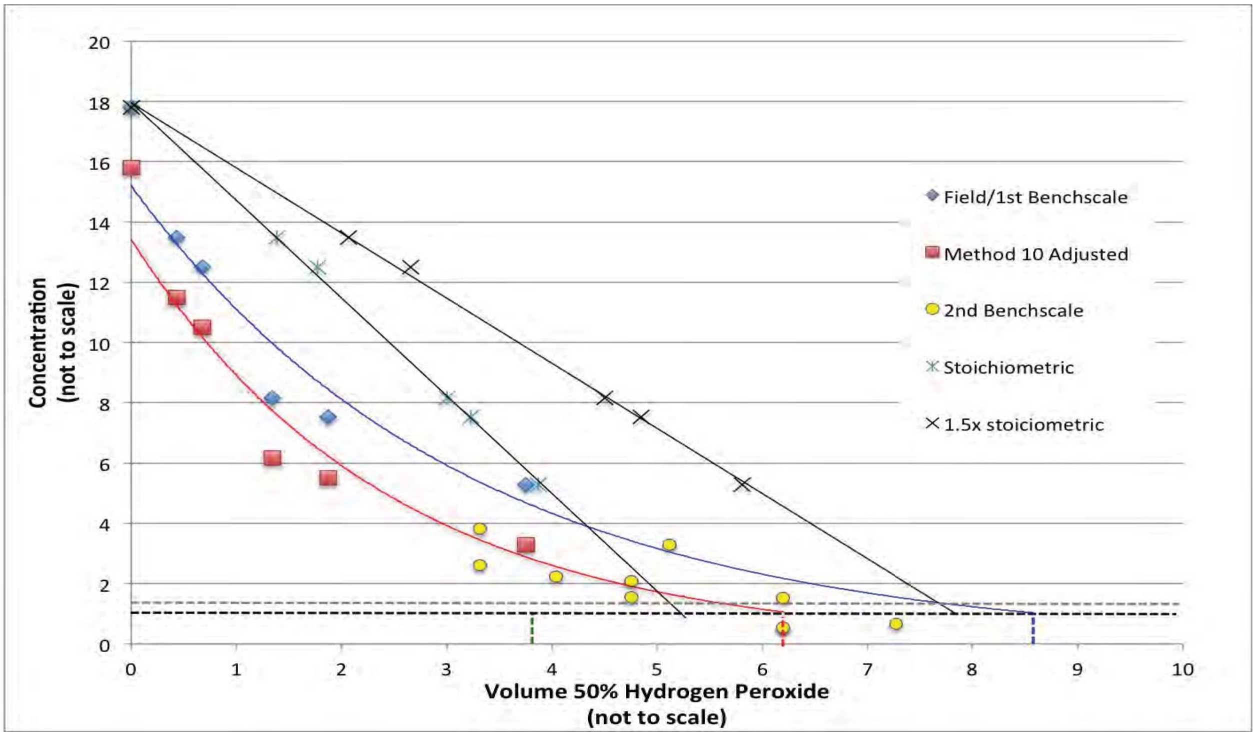

Figure 10. Field/Benchscale Performance vs. Theoretical Oxidant Demand

theoretical oxidant demand relative to the volume of catalyzed H_2O_2 delivered to the site. The follow-up second benchscale program proved that a modified application strategy using the catalyzed H_2O_2 could achieve the regulatory guideline objective, and provided the client the required parameters and estimations for consideration of full-scale treatment.

Given the difference in actual site conditions relative to the benchscale design parameters, the chemical oxidation approach exceeded the original design expectations, and the program was able to demonstrate the ability of chemical oxidation remediation strategies to overcome many of the unexpected challenges presented by remote sites.

POSTERS


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PRACTICAL FIELD USES OF REMOTE SENSING

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ABSTRACT

Technological advances have made the field use of map products derived from remote sensing much more accessible. It is now possible for nearly anyone to take a tablet holding dozens of high resolution maps into the field, then view, navigate and interpret those maps instantly. In a companion paper "Remote Sensing In Reclamation Monitoring: What Can It Do For You?" we demonstrate that there are many valuable remote sensing-derived map products including vegetation cover type and vegetation growth trends. We emphasize here that these can be created and distributed to laptops, tablets and smartphones, providing access to powerful map products anywhere.

Remote sensing map products provide the context in which physical and vegetative features are distributed on the landscape and help to plan field work. As an example, we are using remote sensing map products that identify important habitats to pre-select landing sites for a helicopter based survey and ensure comprehensive sampling.

Once we are on site, close comparison of what we see on the ground to the map data allows us to better interpret the imagery, modify our sampling, or adjust for factors we could not anticipate before the field work. We also use many different map products to view information about the landscape that we cannot interpret visually. For example, sparse vegetation cover observed at a sample site may be a result of limiting physical factors, or because of a short time since the last disturbance. Historical remote sensing map products on a portable device could show that the area was barren 10 years ago, thus indicating recovery is underway.

Following the fieldwork, we use the field sampling to calibrate, validate and improve the accuracy of our preliminary maps. Correlating on-the-ground vegetation survey data with remote sensing data allows us to quantitatively map the vegetation cover and distribution across large study areas, including areas we could not visit.

The incorporation of remote sensing data provides a more thorough understanding of the observed landscape, its history, and its future. This poster will show examples of these different field uses of remote sensing products from actual reclamation projects, and demonstrate some of those products on a GPS-enabled tablet computer for the area around Whitehorse and for the Faro Mine north of Whitehorse, which was the subject of a mine tour for this conference.

PROJECT CASE STUDY – COMPOSITE SOIL COVER FOR SULPHIDE TAILINGS AT MINE SITE IN NORTHEASTERN ONTARIO, CANADA

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ABSTRACT

During the Fall of 2006, the Ontario Ministry of Northern Development and Mines tendered a construction project to provide a soil cover over the North Impounded Tailings (NIT) area at the abandoned Kam Kotia Mine site in Northeastern Ontario, Canada. The soil cover would effectively impede the entry of water and oxygen into the high sulphide tailings, substantially reducing acid generation and metal leaching effects from within the tailings. The final design incorporated waste rock, sand, gravel layers, a Geosynthetic Clay Liner (GCL), clay and granular cover soils. During the Fall of 2006 and Winter of 2007, Hazco E&D Services implemented the construction of this design. Overall, 800,000 m^2 of GCL was deployed and covered. Deployment of the GCL was undertaken by Terrafix Environmental Inc. under the supervision of Earth Tech Engineering. This paper summarizes the design and construction of this composite soil cover system.

Key words: Mine Closure, Geosynthetic Clay Liner.

INTRODUCTION

The Kam Kotia Mine in Timmins, Ontario operated intermittently from the 1940s until 1972 producing copper, zinc and gold. Following closure approximately 3 million tonnes of acid-generating sulphide tailings and 500 thousand tonnes of acid-generating waste rock was left on the surface at the site. These waste materials have since evolved to become significant sources of Acid Rock Drainage (ARD) and Metal Leaching (ML), which has had a significant impact on the surrounding environment. The Ontario Ministry of Northern Development and Mines (MNDM) has implemented a multi-staged rehabilitation program at the site to mitigate the ARD-ML effects of the waste deposits. Several phases have been completed to date. This paper will only look at the composite soil cover system installed during the fall of 2006 and winter of 2007. The MNDM currently collects and treats ARD runoff and seepage and operates a High-Density Sludge (HDS) treatment plant to treat ARD impacted runoff. Wardrop Engineering Inc., and SENES Consultants Ltd. were retained to design a dry soil cover for the North Impounded Tailings (NIT) in 2004 and 2005. This design was tendered for construction in 2006 and was awarded to Hazco Environmental & Decommissioning Services. The design goal for the project is to provide a “dry” soil cover to minimize the infiltration of water and also limit the ingress of oxygen into the tailings. This construction design will reduce the quantity, acidity and metal loading of the leachate reporting to the site’s drainage collection system to the point where passive treatment technology could be implemented and the HDS plant could be taken out of service.

DESIGN OF THE COMPOSITE SOIL COVER

Ontario's Ministry of Northern Development and Mines commissioned Wardrop and SENES to design a soil cover over the NIT, shown on Figure 1. Earth and rock borrow materials were characterized and preliminary designs were assessed using parameters calculated from standard geotechnical soil testing. Preliminary hydrogeological and geochemical models were run on four design options. The two designs utilizing compacted local clay and a GCL were found to be equivalent in performance and cost. A second round of more detailed laboratory testing was done, including calculating void ratios, freeze-thaw permeability, water retention curves and oxygen diffusion coefficients. After incorporating this data into the analyses, the optimal final design incorporated waste rock, sand, gravel layers, a GCL, clay and granular cover soils.

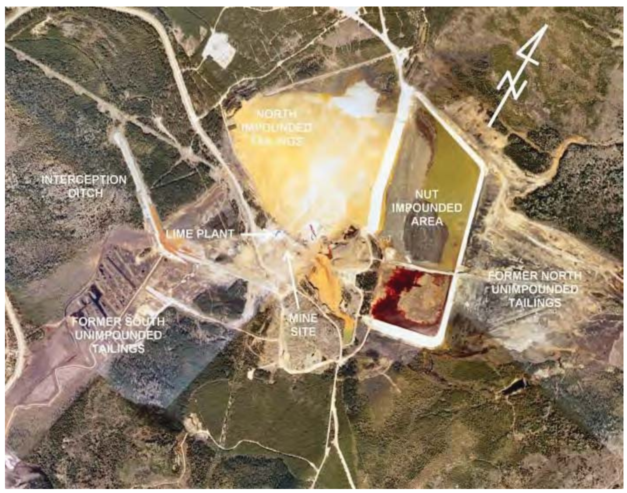

Figure 1. Aerial view of the abandoned Kam Kotia Mine site prior to implementation of a multi-staged rehabilitation program.

<u>Material Testing and Hydrological Modelling</u>

A comprehensive sampling, geotechnical testing program, and chemical analyses of the waste rock was undertaken to characterize the borrow materials available from the surrounding glaciofluvial (granular) and glaciolacustrine (clay) deposits. In addition to natural aggregate sources, the mine waste rock was also sampled and tested for use in the cover. This material testing and Hydrological Modelling program was conducted under the direction of Mr. Andrew Mitchell, P.Geo., formerly of Wardrop Engineering, and Mr. Jeff Martin, P.Eng., of SENES Consultants Ltd.

Additional detailed testing such as oxygen diffusion, freeze-thaw permeability and moisture retention were conducted under the direction of Dr. Michel Aubertin at Ecole Polytechnique in Montreal, Quebec.

The complete hydrological modelling design and results including the performance of four cover option models was presented at the 58th Canadian Geotechnical Conference in Saskatoon (unfortunately the paper was never published in the conference proceedings). A request for this paper can be made to the author of this case study paper.

Cover Options and Final Design

The four cover options modelled by Wardrop and SENES are as follows, starting from the tailings surface upward:

1: 0.3 m rock / 0.25 m sand / 0.5 m clay / 0.5 m sand
2: 0.25 m rock / 0.3 m sand / 1.0 m clay / 0.5 m sand
3: Cover incorporating a geosynthetic clay liner (GCL) with appropriate sand bedding and cover.
4: Cover incorporating a synthetic geomembrane (PVC) with appropriate sand bedding and cover.

The final laboratory test program indicated that the silty clay from the site borrow pit was highly frost susceptible and that there could be an increase in the permeability of up to two orders of magnitude with repeated freeze-thaw cycling. This raised concerns for the longevity of a cover designed with clay as the sole water and oxygen barrier. To overcome this propensity of the clay, it was decided that a GCL would be incorporated into the final design. In addition to the GCL, some other final refinements of the design were incorporated to provide a more robust and durable installation. The final cover, for the tailings surface upwards is as follows:

1. A basal layer of crushed mine waste rock of 300 mm thickness. This layer formed both an effective capillary break due to its coarse grain size distribution as well as adding structural stability to the design. In addition, using the waste rock in this manner provided an opportunity to deal with this acid-generating waste as part of another element of the mine site rehabilitation, which precluded needing to cover the waste rock pile in a later phase of work – at additional cost.

2. A layer of granular fill of 300mm thickness. This layer completed the required thickness for an effective capillary break as well as providing a suitable subgrade for synthetic liner installation.

3. A polypropylene coated GCL forms the water and oxygen barrier to effectively isolate the tailings from the ingress of water and air into the tailings mass from above. The polypropylene coated product was selected in the final design since it has an order of magnitude lower hydraulic conductivity than the figures assumed in the modelling, which adds conservatism in the design at little additional cost.

4. A layer of silty clay, 300 mm thick is placed directly over the GCL to ensure full hydration throughout the service life of the cover. In addition, the lower permeability of the clay will act as a secondary barrier in addition to the GCL, enhancing the oxygen barrier.

5. A layer of granular soil, 500 mm thick isolates the clay and GCL from physical disturbance and also provides a store and release function to mitigate the effects of sustained high precipitation or drought. The thickness of this layer was increased from the preliminary designs to provide greater protection from frost and root penetration. In addition, it provides the required confining stress on the GCL and enhances the durability of the cover.

6. A layer of organic mulch an topsoil, 100 mm thick, obtained from the removal of overburden from the clay source will be turned into the upper 50 to 75 mm of the granular layer to provide a growth media for surface vegetation.

The final design incorporated into the cover to provide greater resistance to frost-induced disruptions and a layer of the silty clay available locally was incorporated into the sequence to provide continual hydration of the GCL, which is essential to maintaining low gas permeability in the comparatively thin bentonite clay layer afforded by the product.

NEW GEOSYNTHETIC CLAY LINER / POLYPROPYLENE COATED

A geosynthetic clay liner containing a polypropylene coating was used to provide a unique hydraulic property. This product, which has been available since 1999 adopts merging a typical textile coating procedure to that of a needle-punched geosynthetic clay liner. Originating from the textile industry, this process yields a composite clay geosynthetic barrier (GBR-C) / geosynthetic clay liner (GCL) product with unique hydraulic properties and physical performances that make it well suited to many new design approaches. The product includes a polypropylene coating applied to the woven geotextile side of a GCL, providing a low permeability typical for a geomembrane at 5×10^{-13} cm/sec (ASTM E96).

PROPERTIES / TEST METHODS ON POLYPROPYLENE COATED GCL

Testing of the polypropylene coated GCL for this project was done by Sageos/CTT Group (Canada) under the supervision of Earth Tech Engineering (Winnipeg). Hydraulic testing on the coated GCL is a difficult task in the traditional permeameter due to the lower flow characteristics of this new GCL – attributable to the polymer membrane coating. A typical non-coated GCL will yield permeability values on the order of 3×10^{-9} cm/s under 35 kPa effective confining stress and 14 kPa head pressure, when testing in accordance with the Hydraulic Conductivity Test Method ASTM D5084. Polypropylene coated GCLs have shown to force side wall leakage to occur, thus making it difficult to measure the performance in the traditional permeameter.

To more accurately determine the true flow through the membrane portion of this type of GCL, a water vapour transmission test was performed. An equivalent hydraulic permeability (k) was calculated using the procedure outlined by Koerner (1997). Via ASTM E96, a value of less than 5×10^{-13} cm/s is achieved.

The coating is typically applied to the woven portion of the GCL. This coating has added another dimension to GCLs with an increase in peel values and internal shear values. The fibres which have been needled through the composite are subjected to the coating and as the coating becomes an integral part of the GCL, the fibres are bonded within the coating. Another added benefit of a polypropylene coated GCL is its effectiveness as a root inhibitor (Lucas 2002).

PROJECT OVERVIEW

Construction of the 80 ha composite cover soil system started in early November of 2006 with an initial deployment and completion of 10,000 m^2 on the first day of the project. Initial plans were to deploy 120,000 m^2 during the fall prior to closing the project down for the winter. These initial plans were changed and 800,000 m^2 was deployed from November of 2006 to February of 2007. Deployment rates at times reached over 30,000 m^2 per day.

Although the GCL can be deployed at a high rate, this deployment is restricted by the cover soil placement over the GCL. For this project the cover soil used was a silty clay which was available locally at the mine site.

The GCL was deployed over a stable subgrade of granular fill as mentioned previously in Section 2.2. A 0.3m overlap was done, recorded, and supervised by Earth Tech Engineering (shown in Figure 3).

Figure 2. Hazco's truck fleet carrying the clay from a nearby clay source available on site to provide a cover soil over the geosynthetic clay liner.

Figure 3. Deployment of the GCL. 30cm overlap being done. Polypropylene side of the GCL facing down.

Following the deployment of the GCL, loose bentonite was placed between panel edges. Loose granular bentonite should be placed between the panels at a rate of 2 kg per lineal metre of seam if the GCL is the primary hydraulic seal. The addition of bentonite to the seam is optional when the GCL will be acting as leak isolator for an overlying membrane. Spreading of the loose bentonite is shown in Figure 4.

Following the deployment of the GCL, a 0.3 m thick silty clay layer was placed directly over the GCL as shown in Figure 5.

Figure 4. Spreading of loose bentonite between GCL sheets to provide a continuous hydraulic seal.

Figure 5. Clay being applied over the GCL.

SUMMARY

One can obtain many types of GCLs: stitched, glued, needle-punched, different bentonite content, different geotextile weight, scrim reinforced, and enhanced polymer, etc. This list is long. Over the years and through increased use, this area of geosynthetics engineering seems to see ever-cheaper GCLs being requested for particular projects. This may mean thinner textiles and/or less bentonite, almost to the point of becoming less a GCL than a double-layered textile. What is the lowest mass per unit area of bentonite that a GCL can have and still achieve, for example, the quoted manufacturer's hydraulic conductivity? It's getting to the point where 12 kg/m^2 seems acceptable, down from the standard 18 kg/m^2. The design community originally used 24 kg/m^2. There has to be a limit, but only specific project engineers will ask for those limits. (Maubeuge 2002).

Engineers when asking for a geosynthetic clay liner must request an accurate breakdown of the GCL instead of simply asking for a GCL. The recommended list of data when requesting for a GCL is as follows:

- Top geotextile shall be X g/m^2 nonwoven.
- Bottom geotextile shall be X g/m^2 woven or X g/m^2 scrim-reinforced nonwoven.
- Swell index of the bentonite.
- Fluid loss of the bentonite.
- Bentonite mass per unit area at X moisture content.
- Grab strength of the GCL.
- Peel strength of the GCL.
- Permeability of the GCL.
- Index flux of the GCL.

- Internal shear strength of the GCL.
  Scrim = woven.
  Scrim-reinforced nonwoven = woven + nonwoven.

Figure 6. Needle-punching board.

All GCLs have a nonwoven top geotextile for needle-punching purposes.

The bottom geotextile is either a woven on its own or a scrim nonwoven geotextile if required for rough soil conditions or steep slope applications.

The Geosynthetic Research Institute recommends that the bottom geotextile of a GCL must contain a scrim-reinforced nonwoven geotextile. Possible failures which may occur by not using a scrim-reinforced bottom geotextile are internal erosion of the bentonite through the geotextile in hydraulic head conditions (Rowe and Orsini 2002), and possible shrinkage of the GCL itself in the composite lining system (GRI White Paper – 2005).

As per the Geosynthetic Research Institute's White Paper of April 2005: Do not use GCLs with needle-punched nonwoven geotextiles on both sides unless one of the geotextiles is scrim-reinforced. There are numerous possibilities in this regard, but all should have a woven component embedded within, or bonded to, the nonwoven component.

The project described herein contained a bottom woven geotextile. As mentioned, a polypropylene coating is applied to the GCL used in this case to decrease the permeability of the product (GCL) to the range of a geomembrane.

One should never use trade names when requesting an item for a specific project. One should always list the testing values required from a specific product, i.e. ASTM testing values. Products and their names change over time hence the requirement to avoid using trade names. Another factor is to avoid having the purchasing agent and/or general contractor make a decision during the tender process.

Remember, you get what you pay for in life. Want cheap? Expect it, but don't expect quality and performance from it. Someone will sell it to you. Will they provide you with a warning? It's to be hoped that they will provide you with the limitations of the product. Want something reliable? Every company can offer reliability at a reasonable price. Want the "crème de la crème" with all the built-in safeguards and back-up systems? Every manufacturer would love to sell its premium brand, but expect to pay a premium for that. In our world, however, the cheapest price often prevails. A manufacturer's premium brands probably represent only 10% of their overall sales – if that.

Products are never equal when using trade names. Products are equal when values are provided and hence can be compared. Ask for them in your tender request.

ACKNOWLEDGEMENTS

The writer would like to acknowledge the contribution of a number of people and contributors to the paper.

Mr. Andrew Mitchell, Mr. Jeff Martin, and Mr. Christopher Hamblin of the Ministry of Northern Development and Mines for supplying the original design paper for this project, which as mentioned, was never published during the 58th Canadian Geotechnical Conference.

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ASSESSMENT OF SAWMILL WASTE BIOCHARS FOR THE PURPOSE OF HEAVY METAL REMEDIATION

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ABSTRACT

Over recent years, biochar has become of particular interest for remediation purposes, not only because it has the ability to reduce the bioavailability of pollutants, but also because it can be produced inexpensively from virtually any type of organic material. However, it has been widely demonstrated that the characteristics, and thus the performance of biochar, can be greatly affected by the feedstock and pyrolytic conditions used in its production. Taking this into consideration, it is important to understand a particular biochar before it is applied in the field. The goal of the presented research is to assess a number of biochars made from sawmill waste for the purpose of heavy metal remediation. This includes characterizing the leachable fraction of the biochar, as well as an assessment of its capacity to adsorb heavy metals.

DETERMINATION OF OPTIMAL SUBSTRATE TO MAXIMIZE THE REVEGETATION OF COVER WITH CAPILLARY BARRIER EFFECTS

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ABSTRACT

Using a vegetal barrier can be an efficient approach to protect cover with capillary barrier effects (CCBE) from invading trees. CCBE are used in mine environments and remediation to prevent oxygen flux from reaching reactive mine tailings, generating acid mine drainage. However, the long-term performance of CCBE can be impeded by tree establishment. After five years the roots of trees established on CCBE e.g., *Salix sp.* and *Populus balsamifera* have already exceeded the protecting layer depth and are growing in the water retention layer (approx. 60 cm). It has been suggested that plants with allelopathic effects can be used to limit the bio-intrusion of trees. *Kalmia angustifolia* L. and *Rhododendron groenlandicum* ([Oeder] Kron & Judd), two common shrubs in the boreal region, have been reported having a strong allelopathic effect (AE). Allelopathy is defined as the inhibition of germination and/or growth of certain species of plants by the presence of other plants. Due to hostile environmental conditions, the establishment of allelopathic species can be difficult on the CCBE. The aim of this project was to use different mixture of industrial waste as substrate to maximize the revegetation of CCBE with allelopathic species. This project also allows the valorisation of wastes with low economic values without impacting the ecosystem. In this project seven substrates were tested in the greenhouse and the field. Twenty plants of each species were grown with the different mixes of substrate. For the greenhouse trial, the results show that all substrates result in some gain of biomass, but a significant gain was observed for two of them.

For the field part, a randomized complete block design was installed in 2011 on a CCBE (Abitibi-Témiscamingue, Québec). The site is characterized by two ecotopes distinguished by their level of water saturation (wet and dry). In each ecotope, three experimental blocks with 21 plots in each block were established. After one year of growth, the rate of survival of AE species was measured. The preliminary results show that the rate of survival is higher in control plots than in substrate-amended plots, probably because we measured a loosening in other plots. Also, the rate of survival is higher in the wet ecotope. This study confirms that a mix of industrial waste is a good option as a plantation substrate. However, preliminary results highlight the necessity of improving the physical properties and water retaining capacity of the substrates. More details of our study will be provided in the poster.

OIL SANDS RESEARCH AND INFORMATION NETWORK: CREATING AND SHARING KNOWLEDGE TO SUPPORT ENVIRONMENTAL MANAGEMENT OF THE MINEABLE OIL SANDS

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ABSTRACT

The Oil Sands Research and Information Network (OSRIN) is a university-based, independent organization that compiles, interprets and analyses available knowledge about managing the environmental impacts to landscapes and water impacted by oil sands mining and gets that knowledge into the hands of those who can use it to drive breakthrough improvements in regulations and practices. OSRIN is a project of the University of Alberta's School of Energy and the Environment (SEE) and was funded with grants from Alberta Environment and the Canada School of Energy and Environment Ltd. OSRIN's mandate is to create and share knowledge so that (1) Alberta can continue to improve environmental management of the mineable oil sands, and (2) Albertans and others are better informed about oil sands impacts, research and management. We do this by funding research and sharing knowledge through our website.

Keywords: Oil Sands, Environmental Management, Research.

INTRODUCTION

The Oil Sands Research and Information Network (OSRIN) is a university-based, independent organization that compiles, interprets and analyses available knowledge about managing the environmental impacts to landscapes and water impacted by oil sands mining and gets that knowledge into the hands of those who can use it to drive breakthrough improvements in regulations and practices. OSRIN is a project of the University of Alberta's School of Energy and the Environment (SEE). OSRIN was launched with a start-up grant of $4.5 million from Alberta Environment and a $250,000 grant from the Canada School of Energy and Environment Ltd.

OSRIN provides:

- Governments with the independent, objective, and credible information and analysis required to put appropriate regulatory and policy frameworks in place
- Media, opinion leaders and the general public with the facts about oil sands development, its environmental and social impacts, and landscape/water reclamation activities – so that public dialogue and policy is informed by solid evidence
- Industry with ready access to an integrated view of research that will help them make and execute environmental management plans – a view that crosses disciplines and organizational boundaries

OSRIN's mandate is to create and share knowledge so that: (1) Alberta can continue to improve environmental management of the mineable oil sands, and (2) Albertans and others are better informed about oil sands impacts, research and management. We do this by funding research and sharing knowledge through our website.

Funding

OSRIN was launched in 2009 with a start-up grant of $4.5 million from Alberta Environment and a $250,000 operating grant from the Canada School of Energy and Environment Ltd. Since then OSRIN has received a small amount of additional income such that the total budget available to pay for research and administration was slightly less than $4.8 million. We have spent $3.97 million to date.

Governance

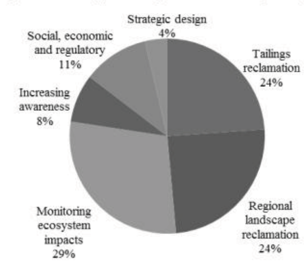
OSRIN has a 10-member Board of Directors comprised of seven provincial government representatives from key environmental, energy and research ministries or agencies and three academic research representatives. The Board provides advice and guidance on research program and project priorities and assists in disseminating OSRIN research results within their respective organizations.

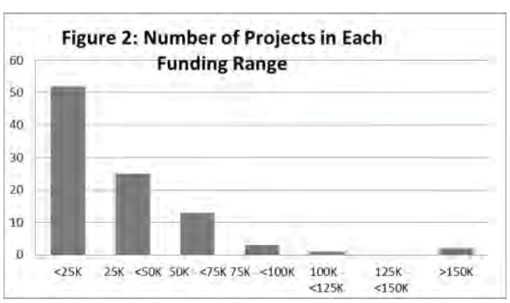
RESEARCH

OSRIN has identified six program areas in which we are funding work. Within each program area we fund projects to scope out the state of knowledge, identify knowledge gaps, and provide insights regarding research priorities.

The six research program areas are described below, with examples of some projects funded in each area. Not surprisingly the majority of funding to date has gone to tailings, reclamation and monitoring research (Figure 1).

Figure 1: Project Expenditures by Program





We tend to focus on small, short-duration projects costing less than $50K to allow us to undertake more research for the available funds (Figure 2). Generally the projects synthesize existing knowledge (e.g., literature reviews, technology catalogues) or bring experts together to discuss the state-of-knowledge (workshops). However OSRIN also funds early stage research that helps set the basis for longer-term, more expensive field and demonstration trials (usually new technologies, or new applications of existing technologies).

Tailings Reclamation

This program seeks to identify challenges that must be addressed in accelerating the reclamation of tailings ponds and tailings disposal areas and to catalyze necessary research, demonstration and development efforts to resolve them. Sample example projects include:

- A review of tailings dewatering technologies (note this was completed before the recent comprehensive tailings technology review by government and industry)
- A catalogue of analytical methods for naphthenic acid
- A review and assessment of emission measurement technologies for air pollutants from tailings ponds

Regional Landscape Reclamation

This program focuses on providing the knowledge necessary to support development of regional reclamation targets as well as site- and mine-level objectives. Some example projects include:

- Recommendations for planting trees on tailings dams in consideration of dam safety concerns
- An assessment of climate change impacts on revegetation success
- A workshop and a report on ecological resilience of reclaimed lands

Monitoring Ecosystem Impacts

This program focuses on components of a comprehensive, robust system in Alberta to monitor the effects of oil sands mining operations on ecosystem health – a system that is scientifically sound and has the confidence of the general public. Some example projects include:

- A paper and workshop outlining the characteristics of a world class environmental monitoring system for the oil sands (note that this was released before the provincial and federal government panel reports and monitoring system announcement)
- An assessment of isotope and geochemical tracers for fingerprinting tailings waters and the ability of the technique to determine occurrence of tailings water in the Athabasca River
- An evaluation of the use of wildlife as biomonitors of ecosystem health

Increasing Awareness

This program aims to increase awareness of OSRIN and oil sands issues through an active website presence, sponsoring oil sands related conferences, digitizing historical information and publication of OSRIN research results. Some example projects include:

- A review of the information sources journalists use to develop oil sands stories
- Three years co-sponsoring the iGEM (International Genetically Engineered Machines) Oil Sands Challenge with the Oil Sands Leadership Initiative
- A report and workshop exploring the potential for a reclamation knowledge network

Social, Economic and Regulatory

This program seeks to identify social, economic and regulatory issues that may affect environmental management of oil sands and to evaluate the effectiveness of environmental management in addressing social, economic and regulatory issues. Some example projects include:

- Three reports assessing different facets of the province’s Mine Financial Security Program
- A review of federal legislation applicable to oil sands

- A plain language summary of human health risk assessment (HHRA) and a report on the role of naphthenic acids in HHRAs

Strategic Design

This program focuses on the development and refinement of OSRIN's strategic intent and program delivery. Most of the work in this program was undertaken in the first year. A key project in this program was development of a detailed research strategy roadmap, logic model, and a description of OSRIN's goals, mandate and approach.

INFORMATION

OSRIN creates and shares information in a variety of ways including: our website, our research reports, our bibliography, digitizing historical research, and networking. As noted above under the Increasing Awareness program we also support oil sands related conferences to ensure people have access to technical and policy information.

Website

OSRIN's website (http://www.osrin.ualberta.ca/en.aspx) is our primary method for disseminating information and is your gateway to the world of oil sands information. The website contains:

- What's New – current events updated daily
- Did You Know – interesting tidbits to whet your appetite
- Publications – links to OSRIN's publications
- Newsletter – 180 subscribers to a bi-weekly e-mailed updates on website content and OSRIN activities
- Website Links – links to an array of information sources
- Videos – see and hear a variety of opinions; an excellent teaching tool
- Upcoming events – conferences, workshops etc. relevant to oil sands
- Who's who – a listing of people involved in oil sands work

Research Reports

OSRIN has released 38 technical research reports to date as well as 9 staff reports. OSRIN has also released a video (both full length and sub-divided into blocks for easy mobile device access). As of July 5, 2013 there have been over 22,000 copies of OSRIN products downloaded. A full listing of products is available at http://www.osrin.ualberta.ca/en/OSRINPublications.aspx and all of the reports are accessible on the University of Alberta's Education & Research Archive site at http://hdl.handle.net/10402/era.17209.

Oil Sands Environmental Management Bibliography

OSRIN has partnered with the Cumulative Environmental Management Association (CEMA) to develop an on-line, searchable bibliography for oil sands related information (http://osemb.cemaonline.ca/rrdcSearch.aspx). With over 2,700 references from academia, government, industry and other organizations spanning the years 1914 to 2014 the bibliography is the best place to start your oil sands research (Figure 3).

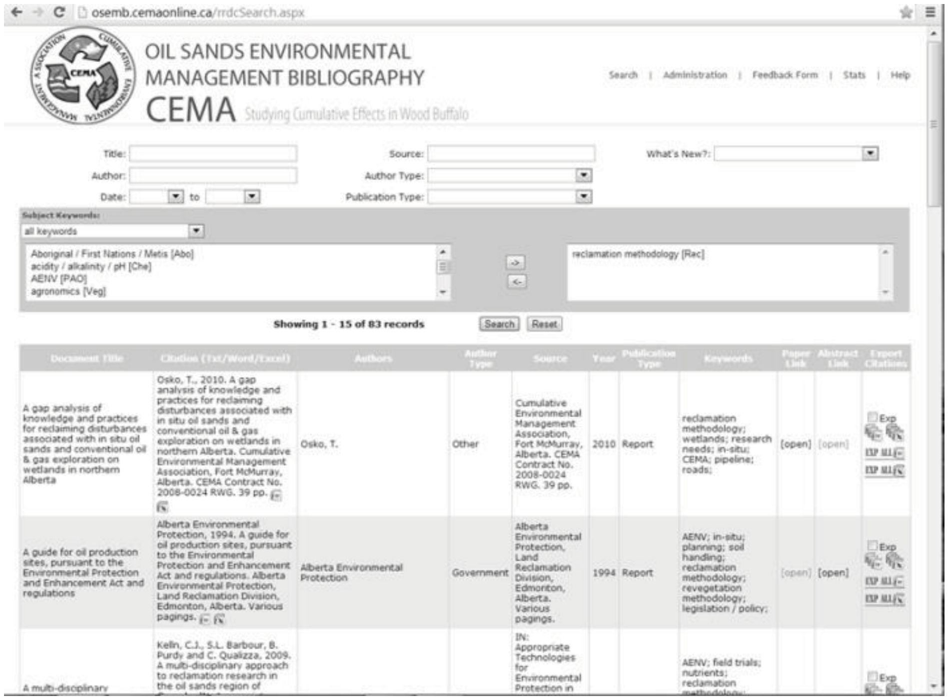


Figure 3. Oil Sands Environmental Management Bibliography

Digitizing Historical Reports

OSRIN has digitized 320 government research and policy documents from the 1970s and 1980s to ensure this valuable content is not lost to those who look primarily to the internet for information. Further information on the types of documents available as well as links to full listings of available reports and the sites where the individual reports can be downloaded are provided at http://www.osrin.ualberta.ca/Resources/DigitizedReports.aspx.

Networking

OSRIN provides a key link between researchers, ENGOs, the general public and government and has responded to numerous requests for information and contacts. The Executive Director's membership in CEMA's Reclamation Working Group and Land Working Group, plus contacts in government, academia and industry, provide opportunities to make people aware of activities that others are undertaking that may have an impact on their own work. The Executive Director is also active in making presentations in a variety of forums to explain OSRIN's work and promote recognition of the extensive body of information that has been generated about environmental management of the oil sands (see http://www.osrin.ualberta.ca/AboutOSRIN/News%20and%20Activities.aspx).

MINERALOGICAL AND GEOCHEMICAL CONTROLS ON METAL SEQUESTRATION IN THE KENO HILL SILVER DISTRICT

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ABSTRACT

The Keno Hill Ag-Pb-Zn district (Yukon, Canada) has been mined intermittently between 1913 and 1989 and hosts numerous watercourses impacted by historical mine water drainage. Naturally occurring sequestration of total and dissolved Mn, Fe, Zn, Cd, As and Pb is observed along the course of the drainage leaving the No Cash (NC) and Husky South West (HSW) mine adits. In this poster we will discuss the attenuation of these elements in precipitates forming downstream from the adits. Samples of stream sediment were collected at regular intervals downstream and at different depths within the sediments. Bulk geochemistry showed that the concentration of Mn, Fe, Zn, Cd, As and Pb decreased with distance from the adits.

Electron microprobe and laser ablation **inductively coupled plasma mass spectrometry** allowed detailed chemical analysis of Mn- and Zn-rich colloidal coatings on sediment grains and of individual particles in both NC Creek and HSW drainage sediments. These techniques showed that in both drainage systems Zn was precipitated with Mn, whereas As was predominantly associated with Fe. Synchrotron-based micro-X-ray fluorescence mapping, micro-X-ray diffraction, and Zn K-edge micro-X-ray absorption near edge structure spectroscopy identified hetaerolite ($ZnMn_2O_4$), hydrozincite ($Zn_5(CO_3)_2(OH)_6$) and Zn-sorbed on ferrihydrite ($Fe_5O_3(OH)_9$) in Mn-Zn-rich colloform coatings on silicate gains in the stream sediments of No Cash Creek. Work is ongoing to characterize the major hosts of Zn in the sediments deposited from the Husky SW adit.

OIL SANDS VEGETATION COOPERATIVE – A COORDINATED EFFORT TO HARVEST AND BANK SEEDS FOR RECLAMATION IN NORTHEASTERN ALBERTA

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ABSTRACT

Development of the oil sands in northeastern Alberta is an important contributor to the economies of both Alberta and Canada and is significantly changing the landscape on large tracts of land. Reclamation of resulting disturbances is ongoing and progressive throughout the life of the project. Key steps involved in reclamation include landform construction, replacement of reclamation material (i.e., soil) and planting native boreal plant species to create functional, self-sustaining ecosystems. Large numbers of propagules from a variety of individual plant species are required to accomplish this goal. As the number of leases being developed grows and as each company expands their footprint, there has been a greater and greater need for propagules to progressively reclaim the land. It has become apparent that a concerted effort by all industrial operators in the area is necessary to maximize the ability of individual companies to reach their revegetation objectives.

To ensure a steady supply of native plants for use on reclaimed sites in northeastern Alberta, large quantities of seed need to be harvested and banked while preserving the integrity of the natural refugia. In 2009, the Oil Sands Vegetation Cooperative (OSVC) was formed with six member companies. The goal is to ensure a consistent and constant supply of seeds and propagules of native boreal plants for reclamation efforts in the mineable oil sands region of Alberta. A coordinated seed harvest has been developed with methodologies based on the Forest Genetic Resource Management Standards. To date, significant quantities of seed representing 25 boreal species have been harvested, extracted, registered and banked. The OSVC has also commissioned a comprehensive research project to examine seed viability, dormancy and storage conditions for a variety of plant species.

RATROOT (*ACORUS AMERICANUS*) PROPAGATION AND ESTABLISHMENT ON CREATED WETLANDS IN THE OIL SANDS REGION OF ALBERTA

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ABSTRACT

Ratroot (*Acorus americanus*) is an ecologically significant boreal wetland species: it provides food and cover for waterfowl and the rhizomes provide a food source for small mammals. This species is also culturally important to local Aboriginal peoples. It is one of the most widely known and used of all the medicinal and food plants. Although ratroot is a constituent of many boreal wetlands, natural ingress of ratroot has not been observed on any constructed wetlands on reclaimed oil extraction sites in northeastern Alberta.

A program was initiated in 2009 to document and characterize natural populations of ratroot and to investigate the potential for introducing ratroot to the constructed wetlands in the oils sands region. Specifically, propagation and establishment methods were evaluated.

Seven naturally occurring populations were documented in the Fort McMurray area. Seeds and rhizomes were harvested and propagation tested. Resulting plants were used to evaluate establishment at constructed wetlands.

Seeds germinate well in the presence of light. As seeds age they develop dormancy and germinate best if stratified and if incubated at higher temperatures. Plants can be vegetatively propagated by rhizome cuttings provided each cutting has at least some root and shoot tissue.

Plants established on artificial mats in constructed wetlands grew and developed well but high mortality was observed due to herbivory. Larger plants (those produced from rhizome cuttings) were more successful than smaller seedlings, and plants placed into Biohaven™ mats survived and grew better than those placed on the foam mats. Plants out-planted along shorelines grew and developed well; they flowered and produced seeds.