41st CLRA National Annual General Meeting and Conference

McIntyre Arena, Timmins, Ontario

June 26-29, 2016

PROCEEDINGS

CLRA | ACRSD

Canadian Land Reclamation Association

Association canadienne de réhabilitation des sites dégradés

PHYSICAL AND CHEMICAL INVESTIGATION OF WASTE ROCK FROM PAST OPERATIONS AT THE DETOUR LAKE GOLD MINE

Jeff Bain[1], Brayden McNeill[1], Aileen Cash[1], Mark Steinepreis[1], Lianna Smith[1], David Blowes[1], G.Ward Wilson[2], Jim Robertson[3] and Marie-Hélène Turgeon[3]

[1]Department of Earth and Environmental Sciences, University of Waterloo, 519-888-4567, jbain@uwaterloo.ca
[2]Department of Civil and Environmental Engineering, University of Alberta
[3]Department of Environment and Sustainability, Detour Gold Corp., Toronto , Ontario

Key Words:
Mine drainage, gas transport, acid-base accounting, sulfide oxidation, permeability

Abstract
In August 2011 researchers from the University of Waterloo, the University of Alberta and Detour Gold Corp. began a research program at the Detour Lake Gold Mine (DLM). This research program involves extensive field and laboratory studies and data analysis (*e.g.,* internal structure, grain size, porewater, pore gas) focused on four waste-rock stockpiles deposited at the DLM site during operations between 1983 and 1999, and which had 13 years of post-closure with covers. The goal of the study is to understand the processes that control the movement and composition of porewater in the historic waste-rock stockpiles. Although some of the waste rock at DLM is potentially acid generating (PAG) and has the potential to release dissolved constituents to the environment, most (83%) of the waste-rock is non-acid generating (NAG). The current investigation showed that the distribution of PAG and NAG waste-rock in the historic stockpiles and the degree of oxidation are variable, and zones of oxidized waste rock are found throughout these piles. The degree of oxidation appears to be related to the waste-rock type and amount of sulfide minerals rather than the location in the stockpile. Elevated concentrations of dissolved sulfate and depleted concentrations of pore-gas O_2 indicate that sulfide oxidation is occurring, but the concentrations of dissolved metals in the matrix porewater are generally low. A predominance of neutral pH porewater and paste-pH results for the waste-rock matrix suggest that ARD has not set in, however, acidic paste pH values in about 20% of the samples indicates that once the pH neutralization capacity of the waste rock is depleted, acidic conditions can develop. A thin soil cover placed on the stockpiles in the 1990s restricts the rate/extent of sulfide oxidation. Analysis of these decades-old waste-rock stockpiles will provide DLM with insight into how to design the new waste-rock stockpiles in a manner that will minimize the risk of seepages that may evolve during operations as well as for the post closure period decades into the future. This knowledge will also improve Detour's ability to manage new waste-rock stockpiles that are being designed and permitted for the new West Detour developments. The research project is continuing with additional financial support from DLM and NSERC and technical support from the staff and graduate students from the Universities of Waterloo and Alberta, and Carleton University.

Detour Gold Mine Site Description

Detour Gold Corporation (Detour Gold) is operating an open pit mine at their Detour Lake property in northern Ontario, approximately 180 km northeast of Cochrane, Ontario. The topography around the mine is typical of the James Bay glacial lowlands underlain by glacially deposited clay, silt, sand and gravel in depths ranging from a few meters to tens of meters. Some of the sandy sediments at the site are carbonate-rich (several percent $CaCO_3$).

The ore deposit is situated in the area of the former Detour Lake mine, which was operated by Placer Dome Inc. from 1983 to 1999. The deposit contains an open pit mineral reserve of 16.4 million ounces of gold, which will be recovered over a mine life of 23 years. The orebody is near surface and is being mined as an open pit with a waste to ore ratio of 3.7. Mining of the orebody is expected to produce on the order of 500 million cubic meters (1.5 billion tonnes) of waste rock over the life of mine. Mineralization is accompanied by sulfide minerals including pyrite, pyrrhotite and chalcopyrite, and lesser amounts of pentlandite and arsenopyrite (AMEC, 2010). Approximately 83% of the waste rock has a low potential to generate acid, as the neutralization potential (NP) to acid potential (AP) ratios (NP/AP = NPR) are > 1.5. Approximately 17% of the waste rock is predicted to have the potential to generate acid (potentially acid generating, PAG), as the NPR is < 1.5 (BBA, 2014). To facilitate management of ARD if it occurs, PAG rock is being stored separately from NAG rock.

Previous operations at the Detour Lake site generated waste-rock stockpiles (WRS1, 2, 3 and 4) dating from 1983 to 1999. Around the year 2000, these stockpiles were resloped and covered with 0.3-1.5 m of overburden, to meet closure requirements of the previous mine owner (Placer Dome, 2001). Monitoring indicates that these stockpiles had not released dissolved metals or acidic drainage to the environment during the 17 years of operation, nor in the 13 year post closure period.

Objectives

DLM is considering installing a low permeability soil cover on new PAG waste rock areas to minimize the potential for ARD. Four of the historic waste rock stockpiles are being examined for this project to evaluate how chemical conditions have evolved in the historic WRS over the past 15-30 years and to determine which physical and structural characteristics contributed to this behaviour.

Methods

Historic WRS 1 and WRS 2 are located within the footprint of the open pit and were relocated. They were examined for internal structure, physical, chemical and biogeochemical characteristics during relocation. WRS 3 and 4 fall outside the open pit footprint. They are being evaluated for mineralogical, chemical and gas transport characteristics in the long-term using instruments (e.g. thermistors, moisture sensors, tensiometers, suction lysimeters, pore-gas sampling points) installed into boreholes. The research program has included test-pit and vertical profile excavation and borehole drilling and instrumentation programs in 2011-2012, followed by data collection and analysis since 2012. A large number of samples collected during these programs have been processed to determine the physical and hydrological characteristics of WRS 1 and 2 (Cash, 2014; Cash et al., 2014). The composition of the pore water and pore gas within WRS 3 and 4 and chemical/mineralogical characterization of the waste rock solids has been extensively examined in all four stockpiles (McNeill, 2016). NSERC funding was obtained in 2012 to support this project and a second grant was received in 2015 for a further five year program to continue these studies.

Observations

Observations from more than 20 test pits (< 5 m depth) and full-thickness profile excavations in WRS 1 and 2 indicated that these stockpiles are structured with multiple benches 10-15 m high, separated by compacted traffic surfaces (Figure 1). The soil cover is typically 0.3 to 1 m thick, and consists of silty-sand glacial overburden with rounded boulders up to about 0.3 m diameter.

Structural features identified during the excavation study indicate that the stockpiles were likely constructed as a combined push dump and paddock dump. Areas with angle of repose slopes and a trend of grain size coarsening downward, representative of push-dump construction, were visible in the interior of the stockpiles. Areas with an absence of bedding structure are probably the result of smaller tip faces or paddock dumping.

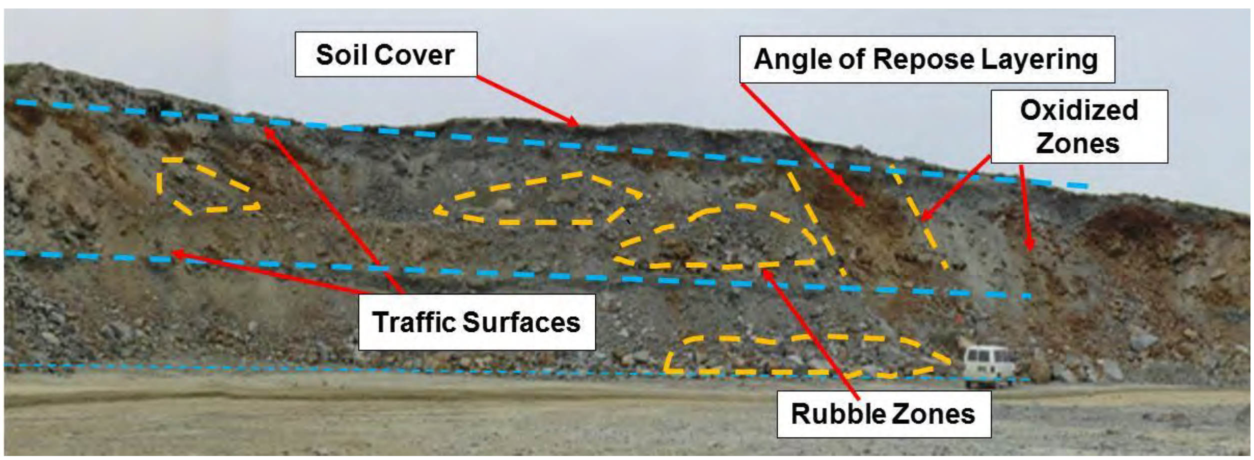


Figure 1. Excavation face (approximately 20 m vertical) on WRS1, showing internal structural features and zones of oxidation

Waste rock composition and weathering are variable in the stockpiles. Areas of weathered (oxidized) waste rock were found at throughout WRS 1 and 2, and a distinct oxidation profile was not evident. The presence of oxidation appears to be related to the waste rock type (PAG material) and amount of sulfide minerals rather than the location in the stockpile. A predominance of neutral paste pH results for fine material from WRS 1 and 2 (Figure 2) suggest that ARD has not set in. Acidic paste pH values in about 20% of the samples indicates that after a 30 year lag time, the pH neutralization capacity of the waste rock has depleted in some areas, allowing localized acidic conditions to develop.

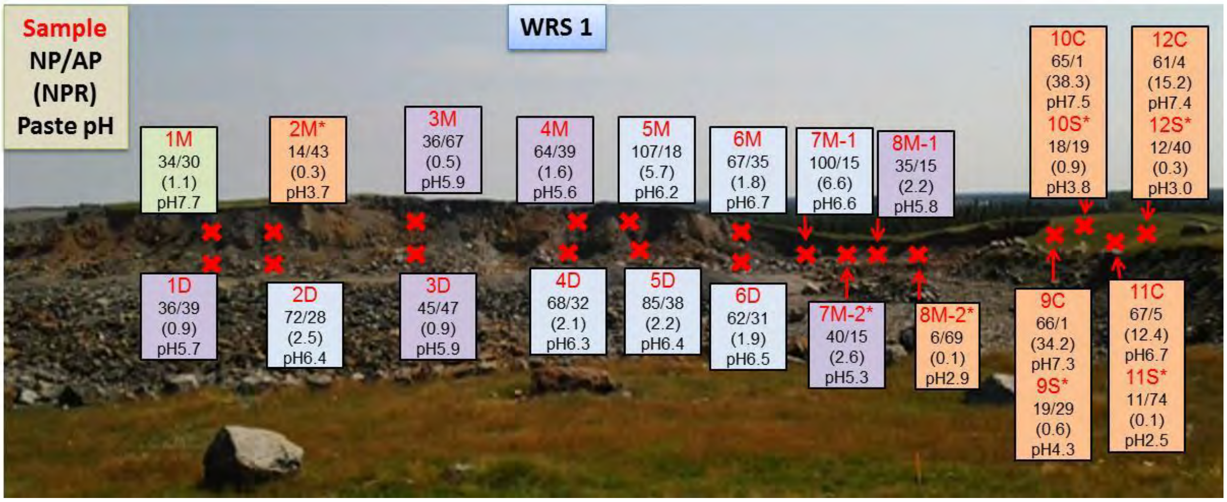


Figure 2. NPR and paste pH values for waste-rock matrix samples. The box shading groups the samples by pH.

Solid phase and aqueous chemical characterization of the waste-rock stockpiles

Analyses indicate that the whole rock composition and major mineralogy of WRS 1-4 are uniform. Total carbon and sulfur analyses indicate that WRS 1, 2 and 4 have similar, but variable C and S content (Table 1), and WRS 3 has a higher C content and a significantly lower S content. Acid-base accounting (ABA) conducted with these results indicates that approximately 60% of the waste rock samples collected from WRS 1, 2 and 4 are potentially acid generating (PAG) and have a neutralization potential (NP) to acid potential (AP) ratio (NP/AP = NPR) of < 1.5. WRS 3 is lower in sulfur compared to WRS 1, 2, and 4 and has fewer PAG samples. In these respects, most of the new waste rock at Detour is more like the rock in WRS 3.

Table 1. Carbon and sulfur analysis on samples from historic WRS 1, 2, 3 and 4 (range and average value).

	WRS 1	*WRS 2*	*WRS 3*	*WRS 4*
Carbon (wt %)	1.7-0.02 (0.5)	2.8-0.03 (0.63)	2.5-0.4 (1)	0.9-0.2 (0.6)
Sulfur (wt %)	4.3-0.01 (1.4)	3.9-0.03 (1.2)	0.5-0.05 (0.13)	1.96-0.03 (0.8)

In-situ monitoring of WRS 3 and 4 has provided several insights into the hydrological and chemical conditions of these stockpiles. Air permeability measurements of the cover (10^{-11} m^2) are 1-2 orders of magnitude lower than in the waste rock (~10^{-10} to 10^{-9} m^2). Diffusive gas transport dominates at air permeability values below 10^{-10} m^2 (Pantelis and Ritchie, 1992; Amos et al., 2015), thus the cover is restrictive to advective gas transport even during dry summer months, a key factor in reducing the rate of sulfide oxidation. Pore gas O_2 is sub-atmospheric at WRS 3, due to oxidation of sulfide minerals and the presence of the low permeability cover that restricts the replenishment of oxygen (Figure 3). Increased CO_2 levels in WRS 3 are associated with acid production and neutralization by carbonate minerals.

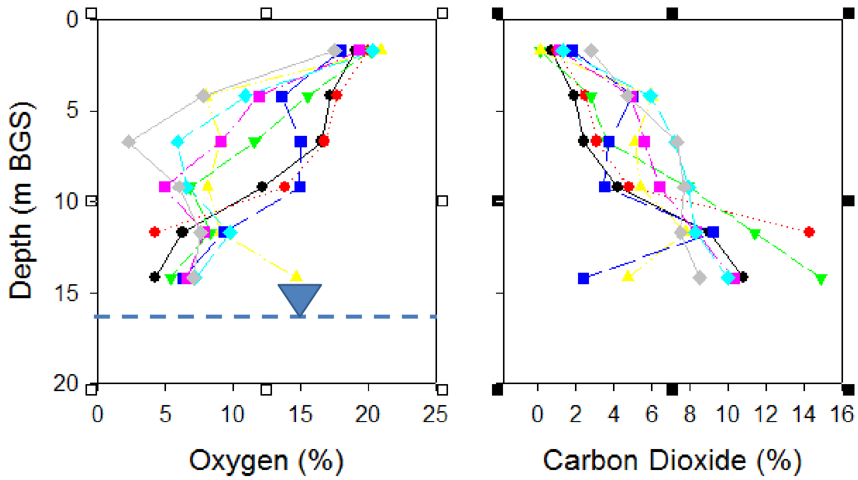


Figure 3. Pore gas O_2 and CO_2 concentrations versus depth in WRS 3 on different dates in 2012.

The pore water composition in the unsaturated zone of WRS 3 and WRS 4 also indicates that sulfide oxidation is ongoing in both stockpiles. Pore-water chemical conditions at WRS 3 and 4 (Table 2) are similar. Pore-water pH is near neutral (6-8) throughout both stockpiles. Elevated sulfate concentrations and depletion in total alkalinity indicate that sulfide oxidation is ongoing and acidity released is being neutralized internally. Metal and anion concentrations are generally higher at WRS 4 (higher PAG content) than at WRS 3. Dissolved metal concentrations are low at both piles indicating that metals released by sulfide oxidation are probably being attenuated within the stockpile. Natural attenuation of the metals and sulfate may also be occurring at the base of the waste rock piles, where infiltrating water passes through organic-carbon rich peat. Despite ongoing sulfide oxidation within the stockpiles, monitoring conducted by DLM indicates that the stockpiles are not currently releasing dissolved metals or acidic drainage to the environment.

Table 2. Selected water chemistry parameters for porewater in the unsaturated zone of WRS 3 and 4 (range and average values). Porewater chemistry for WRS 1 and 2 is not available.

	pH	Eh (mV)	Alkalinity (mg/L as $CaCO_3$)	SO_4 (mg/L)	Al (µg/L)	Fe (µg/L)	Mn (µg/L)	Ni (µg/L)	Cu (µg/L)	Zn (µg/L)
WRS 3	7.6-8.6	197-498	160-1550	670-42	1-102	3-121	7-1054	4-10	2-150	82-1163
		374	700	280	27	29	216	6	17	225
WRS 4	6.97-8.6	197-498	19-675	34-2401	1-34	2-3164	0-1785	0-146	0-80	2-329
		366	196	1136	10	129	127	20	9	58

Project outcomes

- The high NP values of the waste rock result in a long lag period (more than 30 years) before acidic conditions set in

- Careful management of the new waste rock is required to prevent future generation of low-quality drainage
    - The new NAG waste rock piles appear to be similar to WRS 3
    - The new PAG waste rock piles appear to be similar to WRS 1, 2 and 4
- New waste-rock stockpile construction techniques (larger equipment, more size segregation) are likely to lead to more extensive advective gas transport. Advective gas transport may be addressed through:
    - Disruption of gas transport on pile slopes
    - Use of low permeability covers to limit advective gas transport
    - Improved cover design to limit diffusive gas transport and downward migration of sulfide oxidation products

Summary

The findings from our investigations improve our understanding of the contributions of physical structure, cover integrity, construction method, mineralogy and geochemical characteristics toward ARD generation in the old waste-rock stockpiles, which had 17 years of operating history and 13 years of closure history with covers. It is anticipated the ARD-generating and -limiting mechanisms in the new stockpiles will be similar. This knowledge can be used by DLM to guide the design and management of the new waste-rock stockpiles to lessen the risks of ARD generation in both the short and long term. The research project is continuing with additional financial support from DLM and NSERC and technical support from the staff and graduate students from the Universities of Waterloo and Alberta, and Carleton University.

References:

AMEC, 2010. Detour Lake Project. Metal Leaching and Acid Rock Drainage Characterization Report – Volume 1. August 2010. Prepared by AMEC Earth & Environmental. 1471 pp.

Amos, R.T., Blowes, D.W., Bailey, B.L., Sego, D.C., Smith, L., Ritchie, A.I.M. 2015. Waste-rock hydrogeology and geochemistry. Applied Geochemistry, 57: 140-156.

BBA, 2014. Detour Lake Mine. NI 43-101 Technical Report. Prepared by BBA in cooperation with SGS, Feb. 4, 2014. 363 pp.

Cash, A. 2014. Structural and Hydrologic Characterization of Two Historic Waste Rock Piles. M.Sc. Thesis, Department of Civil and Environmental Engineering, University of Alberta.

Cash, A., Wilson, G.W., Blowes, D.W., Amos, R.T., McNeill, B. and Robertson, J. 2014. Structural and Hydrologic Characterization of Two Historic Waste Rock Piles at Detour Lake Mine. Presented at the 22nd BC MEND ML/ARD Workshop, Vancouver, B.C.

McNeill, B. 2016. Geochemical and Microbiological Characterization of the Historic Waste Rock Piles at the Detour Lake Gold Mine. M.Sc. Thesis, Department of Earth and Environmental Sciences, University of Waterloo.

Pantelis, G., and Ritchie, A.I.M. 1992. Rate-limiting factors in dump leaching of pyritic ores. Applied Mathematical Modeling, 16: 553-560.

USE OF A CONCEPTUAL MODEL IN ADVANCE OF NUMERICAL SIMULATIONS TO DEMONSTRATE AN UNDERSTANDING OF LOADING FROM A RECLAIMED WASTE ROCK PILE

C. Bradley[1], G. Meiers[1], D.Mayich[2], M. O'Kane[3], J.Shea[4]

[1]O'Kane Consultants Inc., Fredericton, New Brunswick, Canada

[2]David Mayich Consulting, Sydney, Nova Scotia, Canada

[3]O'Kane Consultants Inc., Calgary, Alberta, Canada

[4]Public Works and Governments Services Canada, Sydney, Nova Scotia, Canada

Key Words: geomembrane, net present value, mass balance, drain-down, seepage, acidity, sulfate.

ABSTRACT

A conceptual model is an initial step in demonstrating an understanding of site conditions to inform on long-term impacts from waste storage facilities. This understanding provides the basis for additional work, including numerical simulations should they be required, and identifies appropriate objectives. In conjunction with numerical simulations, a representative conceptual model establishes a high degree of confidence in simulated results. A conceptual model also serves as a delivery mechanism for conveying information to site personnel or management who may not have the technical knowledge or time required to interpret results. Although developing a conceptual model may seem like an unnecessary step, it is an instrumental skill which needs to be cultivated.

The Victoria Junction waste rock pile was reclaimed with an engineered cover system that includes a geomembrane layer and is presented as a case study. The waste rock pile is located on Cape Breton Island near Sydney, N.S., Canada, and was reclaimed in 2006. Using site information, a conceptual model was developed to include physical, flow, and geochemical components for predicting loading and long-term impacts to the receiving environment. The case study details the conceptual model and methodology employed, and highlights the importance of implementing a monitoring program to ensure sufficient information is collected to adequately inform the conceptual model. It was demonstrated through the conceptual model that a drain-down seepage analysis was required, while rigorous groundwater flow and contaminant transport modeling was not. The conceptual model identified that long-term acid loading from the waste rock pile will be limited to levels that allow for closure objectives to be achieved. Consideration may be required in long-term planning given that the longevity of geomembranes are in the order of 500 to 1,000 years and that stored and potential acidity will remain for ~580,000 years.

Net present value and a representative conceptual model were used to demonstrate the financial and environmental benefit of the reclaimed waste rock pile compared to the

collection and treatment of basal seepage in perpetuity. While the net present value was similar for both alternatives the conceptual model demonstrated that further improvement in surface and groundwater quality was observed through the reclaimed scenario that would not have been realized through collection and treatment.

1. INTRODUCTION

A representative conceptual model captures something real and in its simplest form is an analytical tool used to organize site information and ideas. A site conceptual model for mine waste management facilities includes supporting physical, flow and geochemistry components. The physical component of the model describes the waste storage facility and surrounding landform, and the flow component describes the movement of water through the physical component, both above and below the ground surface. Waste material is characterized in the geochemistry component and source terms are tested for accuracy when coupled with the physical and flow components. Combined, these three components demonstrate an initial understanding of site conditions and may inform on long-term trends such as impacts to the receiving environment.

A functional conceptual model must remain flexible and evolve as new site information becomes available and context for the site is established. A conceptual model that is inclusive but yet simple and demonstrates site processes serves as tool for the engagement of management / site personnel in a forum of active collaboration. A collaborative effort that occurs throughout the project allows for continuous refinement of the conceptual model through intimate site knowledge which may or may not be included in reports.

The conceptual model is a holistic approach that ultimately informs management on decisions regarding site practices in effort to meet closure objectives. It also provides the basis for additional work to support the model and identifies appropriate objectives. Should numerical simulations be required, a conceptual model establishes an understanding for anticipated outcomes for simulations before they are completed, hence a high degree of confidence is established in the results.

This paper presents a case study in which a conceptual model was developed to assess and predict long term performance (i.e. loading to the receiving environment) of the reclaimed Victoria Junction (VJ) waste rock pile (WRP) located in Cape Breton, Nova Scotia, Canada. The conceptual model demonstrates that seepage numerical modeling was required to inform on drain-down from the WRP but groundwater flow and contaminant transport numerical modeling was not. All too often there is a desire to proceed to numerical simulation before a representative conceptual model has been developed.

The VJ WRP was reclaimed with an engineered cover system that includes a geomembrane layer (high-density polyethylene) with the objective of limiting the ingress of meteoric water and oxygen to the underlying waste material. Reduced loads to the receiving environment have been observed and are expected to continue provided cover system performance is maintained, which is largely dependent on the geomembrane, as well as the overlying layers and overall stability of the landform. The conceptual model indicates that stored and potential acidity will be leached from the reclaimed WRP for ~200,000 years, while the longevity of geomembranes is estimated to be upwards of 1,000 years (Rowe, 2005). Studies conducted by Benson et al (2010) estimate the minimum lifespan of geomembranes used in cover systems is on the order of 55 to 125 years.

While it is perceived that any leakage through a geomembrane constitutes a failure from their traditional use as a liner, it may not be the case for meeting closure objectives for waste storage facilities in the mining industry. Although there is a considerable range in the estimates of geomembrane lifespan, it is conceivable that a geomembrane will performing satisfactory as a barrier beyond this period as long as appropriate design considerations are included in the cover system. The key is adequate information to inform on long term performance and to develop an understanding of risk for impacts to the receiving environment.

2. VICTORIA JUNCTION

2.1 PHYSICAL MODEL

The reclaimed VJ WRP is located on the site of a historic coal preparation plant approximately 5 km northeast of Sydney and has a footprint of approximately 26 ha and a height of 40 m. The WRP is located in a topographic low and thought to be constructed on the edge of a post-glacial lake created by a large depression in bedrock. Geology generally consists glacial till over sedimentary bedrock. There is a sand deposit in the wetland immediately north of the WRP, along with a peat layer that encompasses the wetland.

The VJ site has evolved over many years to accommodate operational changes to the coal preparation plant and as part of the remediation activities post-closure of the facility. The coal preparation plant operated from the mid 1970’s to 2000. During operations, the coal preparation plant washed up to 4 million tonnes of raw coal per year, of which 15–20% was placed into the WRP and 3% into tailings ponds. Two coal tailings ponds were constructed as required by processing and storage demands, and were eventually relocated within the WRP and encapsulated with waste rock. Three additional tailings ponds were constructed within the confinements of the WRP. It was general procedure to keep the coal tailing ponds covered with supernatant water to prevent oxidation from occurring. Upwards of 10 million tonnes of potentially acid forming (PAF) waste was placed in the WRP.

In 1987 a 1 m thick bentonite wall was constructed along the north and east toe of the WRP to reduce acid rock drainage and metal leaching (ARD/ML) moving through shallow overburden into the wetland. A toe-drain collected upwelling seeps that were treated through an active treatment plant. The toe-drain was upgraded to the leachate collection system (LCS) in 2006 during construction of the cover system, and encompassed the entire perimeter of the WRP. A basic lime slurry addition water treatment plant was located adjacent to the WRP for the collection and treatment of ARD/ML-affected runoff. This was replaced in 1994 with another lime slurry water treatment plant that included various surge, settling and polishing ponds before discharging into Smiths Brook east of the pile. In 2003, six pump-and-treat wells were installed into bedrock to the north of the WRP to intercept deeper impacted groundwater. A wet well was installed north of the lower surge pond to intercept ARD/ML impacted water percolating through the base / lining of the lower surge pond. The lower surge pond received water from the pump-and-treat wells, the LCS, and the wet well through a series of pumps. Fig.1 shows key landform features and remediation measures.

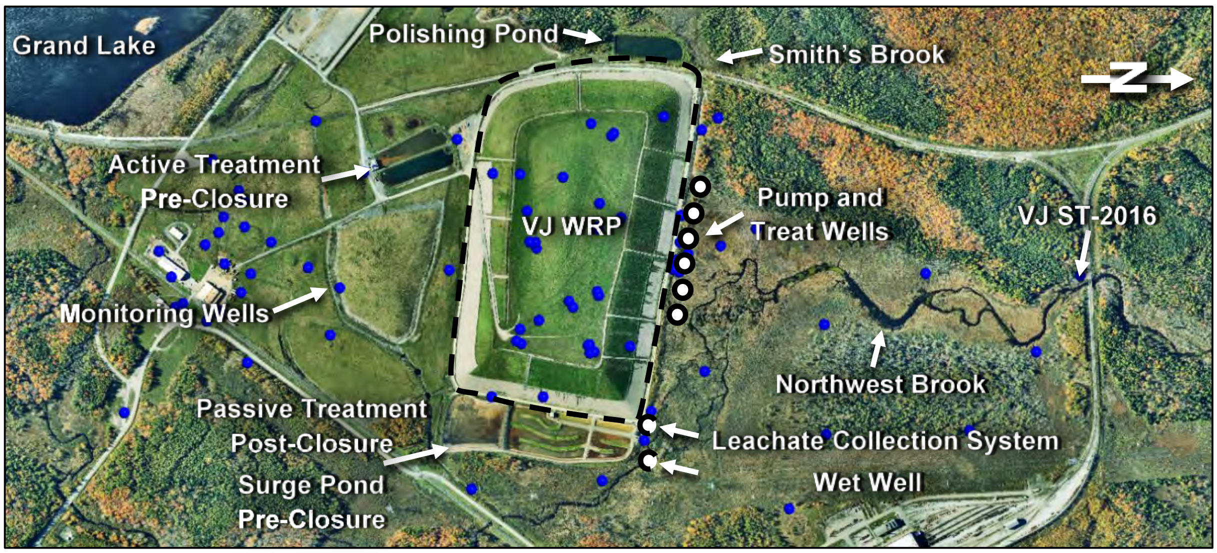


Figure 1. Victoria Junction WRP and site features. Photo taken post-closure with passive treatment.

An engineered cover system was implemented as part of the closure plan for the WRP in 2006. The cover system was designed based on several closure objectives:

- providing a measurable positive effect on the environment compared to the impact pre-closure;
- minimizing the impact to wetlands connected through Northwest Brook; and
- site aesthetics in accordance with industry standards and public expectations.

The cover system comprises a till growth medium layer placed over a granular drainage layer and a 60 mil HDPE geomembrane. The cover system was designed to limit the ingress of meteoric water and oxygen, thus limiting the transport of stored acidity and generation of potential acidity. In 2013, the active water treatment plant was decommissioned and a passive water treatment system constructed in the lower surge pond to the east of the WRP, as shown in Fig. 1. The passive treatment system is recharged with lime and polymer, and ARD/ML impacted water is actively pumped from the LCS into the treatment system.

2.2 FLOW MODEL

The VJ WRP is located within the Northwest Brook watershed. Northwest Brook flows from Grand Lake, located approximately 100 m south of the site, around the east side of the WRP and through the wetland into the Atlantic Ocean approximately 6 km north of the site. Two sampling locations upstream of the WRP provide water quality prior to mixing with ARD/ML and one sampling location downstream of the WRP (VJ ST-2016) is a receptor for ARD/ML and used to evaluate load to the receiving environment. Smith's Brook is a small tributary flowing into Northwest Brook northwest of the WRP and provides alkalinity to the wetland from an open limestone channel constructed within the lower portion of the brook. There are also alkalinity contributions to the system from the active / passive treatment discharge water, two limestone pools imbedded within a surface runoff collection ditch, and natural alkalinity in flows from Grand Lake and groundwater.

Groundwater elevations in the vicinity of the WRP are relatively shallow, approximately 1-5 m below the ground surface, and flows are generally towards the north-northeast. Groundwater discharges north of the WRP, supported by the vertical hydraulic gradients, presence of the wetland, and artesian flow in some of the monitoring wells. Recharge is anticipated to occur to the west and east of the WRP based on the topography. Flow contours were generated for the site from monitoring wells and extended to encompass the landform by interpreting groundwater elevations from topography and water elevations in local water bodies. Contours indicate flow is focused towards VJ ST-2016 (see Fig. 2) and supports the use of this location as a receptor of ARD/ML in the geochemistry model.

Figure 2. Groundwater contours in the vicinity of the VJ WRP.

In general there are strong downward hydraulic gradients below the WRP and weak upward hydraulic gradients in the wetland north of the WRP. The upward hydraulic gradients in the wetland are attributed to the topographic highs surrounding the site and are anticipated to have existed prior to placement of the WRP. The strong downward hydraulic gradients are likely the result of the large hydraulic heads within the WRP caused by maintaining a supernatant water cover over the coal tailings ponds and the high ingress of meteoric water into the bare waste, and have reversed the natural upward hydraulic gradient below the pile. The cover system has reduced the ingress of meteoric water from an estimated 400 mm per year to less than 1 mm per year, based on pre-cover system analysis by AMEC (2005) and estimated leakage through defects as described by Giroud et al. (2000) coupled with three years of cover system field performance monitoring data. Drain-down is expected to occur until the WRP reaches a steady state water content. A portion of drain-down has already occurred, evident by decreased water levels within the WRP following placement of the cover system in 2006. Drain-down, along with net percolation, will contribute loadings to the receiving environment through basal seepage.

There are two components to drain-down for the WRP: saturated drain-down, which will contribute greater loading over a shorter period; and unsaturated drain-down, which will occur over a longer time frame and generate a lower loading. Fig. 3 shows water levels (i.e. saturated drain-down) in two of the monitoring wells that were completed in the tailings layer one year prior to placement of the cover system and the decrease in water levels over time. The rate of decrease in water levels illustrates a more rapid drain-down shortly after placement of the cover system, followed by a gradual decrease. The trend in the rate of drain-down was extrapolated with trend lines to estimate the cessation of saturated drain-down. It is estimated that the drain-down of water perched within the pile is currently 70 mm/yr and will continue for another 10–20 years, at which time the unsaturated drain-down of water will result in a much lower rate of basal seepage. Fig. 4 presents the decrease in water levels through a cross section of the WRP over the course of reclamation.

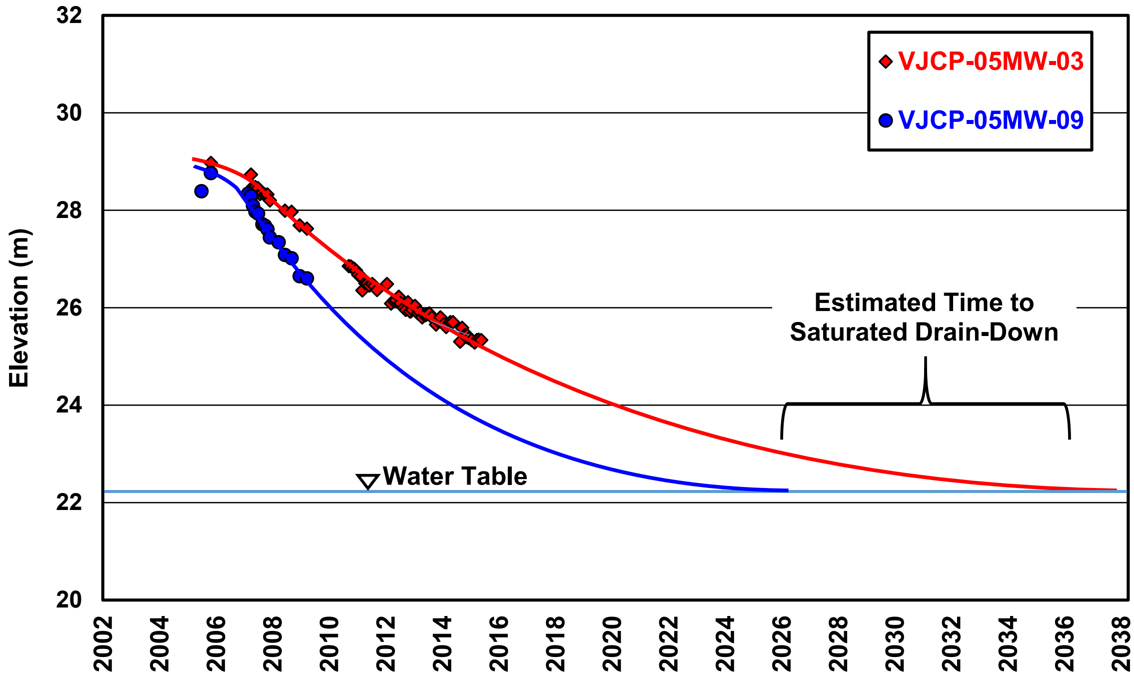


Figure 3. Estimated time to cessation of saturated drain-down in the WRP.

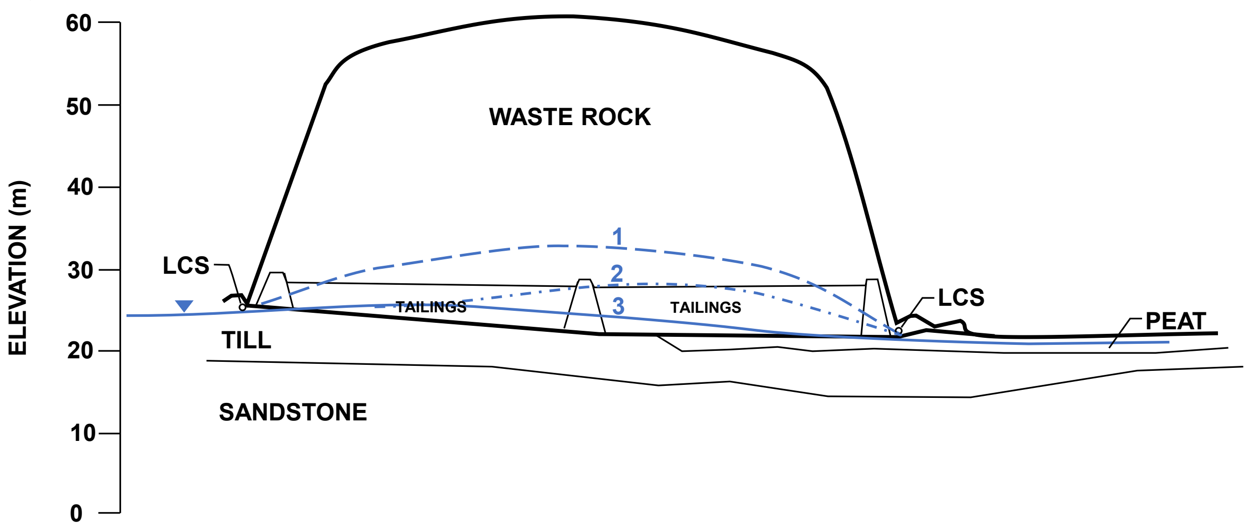


Figure 4. South-North cross section illustrating the decrease in water elevation in the WRP.

Lines 1, 2 and 3 in Fig. 4 represent the observed decrease in water levels following installation of the cover system and associated reduction in water ingress. Line 1 was generated from water levels observed prior to placement of the cover system and shows water perched to an elevation of approximately 33 m, approximately 10 m above the base of the WRP. Line 2 represents the current elevation of water levels, a decrease of approximately 5 m since 2006. The predicted water table elevation at cessation of drain-down was determined from water elevations up- and downstream of the WRP and wells within the footprint of the pile screened in underlying till. The water elevations for wells completed in the till have not changed over time and are reflective of Line 3, indicating that the final water table will likely mound within the base of the pile at approximately 3 m and diminish to near zero at the LCS. Groundwater flow through the WRP is estimated based on a cross section 3 m high and the entire width of the WRP.

2.5 GEOCHEMISTRY MODEL

Loading to the receiving environment has evolved over time and can be characterized by three distinct phases:

- Phase 1 – pre-cover system with active treatment,
- Phase 2 – post-cover system with passive treatment, and
- Phase 3 – long-term post-cover system with passive treatment.

An acid load mass balance was developed for each phase to quantify loading to the receiving environment (Figs. 5, 6 and 7, respectively), with the first two phases providing the basis for long-term predictions.

In the pre-cover system phase, two sources contribute loading to the receiving environment: basal seepage (groundwater mounding and net percolation) and runoff from the site. Sinks, or components acting to reduce this load, include the groundwater collection system (pump-and-treat and wet wells, and LCS) and the active treatment system. The source term for basal seepage was based on measured water quality at the base of the WRP and an iterative process using the acid load mass balances for Phase 1 and Phase 2. Although not necessarily required to determine loading to the receiving environment, runoff was back calculated from the other components for completeness. Grand Lake also contributes a background / natural acid load to the system and the load is essentially a result of the high flow rates observed in Northwest Brook rather than elevated concentrations. The load removed by groundwater alkalinity was calculated based on the decrease in alkalinity observed between groundwater quality up and down-gradient of the WRP.

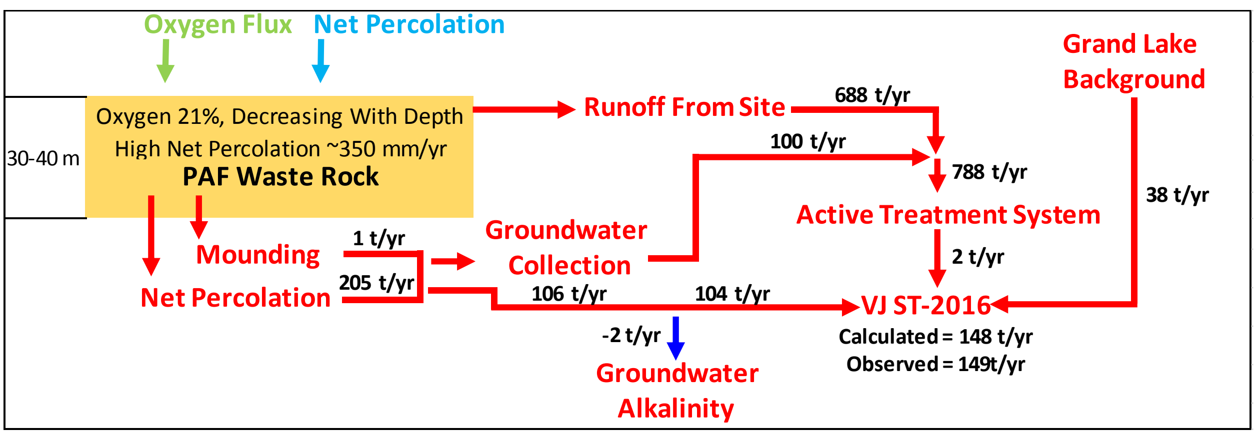


Figure 5. Phase 1 – Acid load mass balance pre-cover system.

The total calculated acid load at VJ ST-2016 is 148 t/yr, and a difference of 1 t/yr from the observed load would suggest that the flow and geochemistry models closely represent site conditions. The total acid load generated is 894 t/year, of which runoff is the largest contributor at 688 t/year. The active treatment system neutralized 788 t/year, while 2 t/year were neutralized by natural alkalinity in groundwater. In terms of basal seepage, approximately 49% is intercepted by the active treatment system, with the remaining reporting to groundwater flow, which is key in developing an understanding of loading to the receiving environment.

The mass balance changes substantially after installation of the low flux cover system and remediation of the site, both in terms of the load produced from surface runoff and basal seepage. The passive treatment system is introduced in this phase and replaces the active treatment system. A drain-down component is also introduced as a result of the decrease in net percolation. The total acid load generated from the site is reduced from 894 t/yr to 43 t/yr and clean runoff from the site represents 688 t/yr of this reduction. Approximately 23% of basal seepage is intercepted and treated before being discharged. The total calculated acid load at VJ ST-2016 is 68 t/year, compared to the current observed load of 66 t/year.

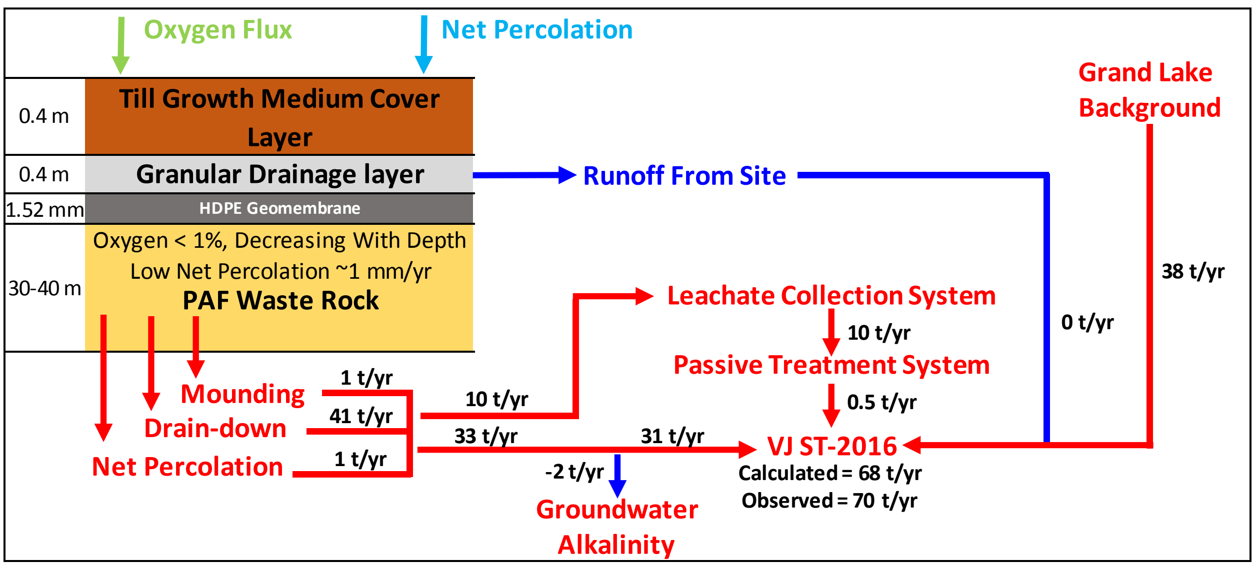


Figure 6. Phase 2 – Acid load mass balance post-cover system.

While an acid load of 66 t/yr was observed at VJ ST-2016, the pile contributes approximately 44% of this load. The average acidity concentration in Northwest Brook at the outlet to Grand Lake and at VJ ST-2016 is approximately 1 mg/l and 1.5 mg/l, respectively. This highlights that the acid load at VJ ST-2016 is primarily the result of the high flow rates in Northwest Brook rather than elevated concentrations. The acidity concentration of basal seepage from the WRP is approximately 2,250 mg/l, and load is attenuated as a result of the low flow rate from the pile. It is important to note that there is also an alkalinity load at VJ ST-2016 of approximately 540 t/yr, which is able provide the required buffering capacity. The natural alkalinity load from Grand Lake alone is greater than the observed acid load at VJ ST-2016 at approximately 120 t/yr. Considering the amount of alkalinity present at VJ ST-2016, there is a net negative acid load of approximately -470 t/yr.

Although the total acid load generated from the site was reduced by approximately 95% (894 to 43 t/year), the acid load at VJ ST-2016 decreased by approximately 56%. The pre-cover system and current mass balances provide context for the observed water quality at VJ ST-2016 in that a proportional decrease in loading was not observed after changes in water collection and treatment. This is primarily attributed to decommissioning the pump-and-treat wells and wet well, as well as the natural contribution from Grand Lake.

Using the mass balances and conceptual model for pre-cover system and current conditions, a mass balance was developed to predict loadings to the receiving environment 100 years post-cover system. It is estimated that saturated drain-down will be completed in another 10 to 20 years and the bulk of water from unsaturated drain-down will have occurred in 100 years; therefore, the acid load from drain-down will be negligible. The period of unsaturated drain-down is supported by numerical modeling completed for another reclaimed coal WRP with similar waste. Contributions to the LCS, and, therefore, the passive treatment system, are anticipated to decrease as drain-down completes. As a result, it is estimated that the load captured by the LCS will be negligible. Any contribution is anticipated to be from groundwater mounding and seasonal fluctuations in groundwater elevation. The mass balance closed for the 100-year period would suggest a decrease in acid load to 38 t/year, equivalent to the background load from Grand Lake.

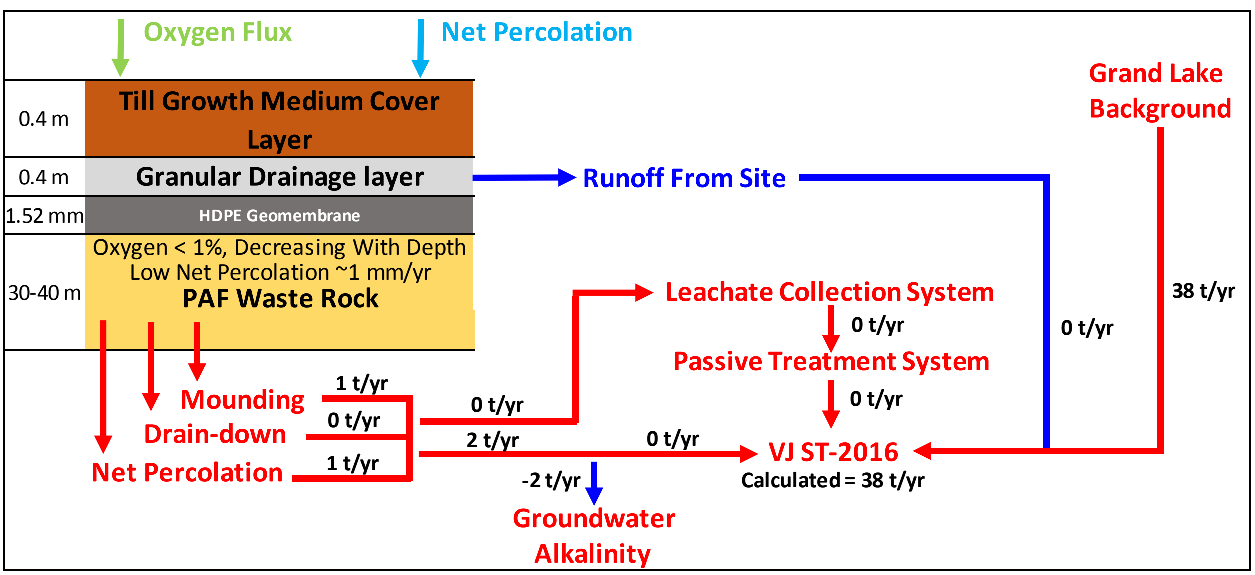


Figure 7. Phase 3 – Acid load mass balance 100 years post-cover system.

This provides context for what benefit the passive treatment system currently provides. Based on the current mass balance, it captures approximately 23% of basal seepage, reducing loading to the receiving environment from 77 t/year to 68 t/year. Loading to the receiving environment without the reduction from the passive treatment system would still be below pre-cover system loading and site closure objectives would still be met. Where the load captured by the passive treatment system is anticipated to decrease as saturated drain-down completes, a strategy for decommissioning the passive treatment system may be to maintain its operation for 10–20 years until saturated drain-down is complete and further improvements to the wetland are observed.

The conceptual model developed for the site identified groundwater mounding and drain-down as having the greatest uncertainty, and the distribution of load between these components has an impact on the predicted long-term loading. As a result seepage numerical simulations were undertaken to quantify and validate the drain-down estimates, which were then used to validate the mounding component. Results indicate that saturated and unsaturated drain-down in the post-cover system mass balance is approximately 40 mm/yr, resulting in approximately a 20 t/yr decrease in load. In order to calibrate the model the saturated hydraulic conductivity of the till underlying the WRP was increased approximately one order of magnitude to 4.0×10^{-4} cm/s, still within the range reported in the literature. Updating the conceptual model with this saturated hydraulic conductivity increased the load due to groundwater mounding by approximately 20 t/yr, which made up for the loss in loading from drain-down. Considering the changes in load from drain-down and groundwater mounding and assuming that the LCS will continue to capture 10 t/yr, the long-term load 100 years post-cover system is approximately 50 t/yr, which includes 38 t/yr from Grand Lake.

The acid load mass balances provide a strong understanding for loading to the receiving environment, and enabled long term predictions 100 years post-cover system. Coupled with the seepage numerical simulations there is a high degree of confidence in the results. The results indicate that loading has decreased substantially following placement of the cover system and will continue to decrease as drain-down diminishes. Leapfrog Hydro was used to develop groundwater plumes from monitored water quality to provide an understanding for how groundwater has evolved in conjunction with site reclamation activities.

Extensive groundwater sampling was conducted from 2002-current and permitted the generation of six 3D groundwater plume models of mean yearly sulfate concentrations to visualize groundwater plume evolution over the period (Figs. 8a-f). An improvement in groundwater quality (i.e. a decrease in the plume's extent and concentration) was observed over the monitoring period and three key phases of groundwater evolution were identified:

- significant improvement in water quality following placement of the cover system in 2006 and 7 years of active deep bedrock water treatment (Figs. 8a to 8b);
- a slight decline in water quality immediately following the decommissioning of the pump-and-treat wells (Figs. 8b to 8c); and

- steady improvement in water quality over the short term with trends that support the flow model, thus strengthening the conceptual model as a whole (Figs. 8c to 8f).

Groundwater quality improvements observed in the modelled plumes are supported by the acid load mass balance. Pockets of high concentration appear to be dissipating and there is also less impact to deep groundwater as vertical gradients in the WRP have diminished as a result of the decreased flux through the cover system

As the conceptual model demonstrates, longevity of ARD/ML is controlled by the rate of transport of acidity from the WRP, which in the long-term is controlled by groundwater mounding and net percolation. Using a simple analysis under the assumption that potential acidity will be generated at a rate greater than the mobilization of stored acidity, the longevity of ARD/ML will be a function of the loading from each component. The long-term loading rate for both groundwater mounding and net percolation is 20 t/yr and 1 t/yr, respectively. Based on the ABA analysis and the physical model, there is approximately 670,000 t of total acidity, of which 90,000 interacts with groundwater and 580,000 interacts with net percolation. Groundwater mobilizes acidity at a much greater rate than net percolation. Acidity mobilized by groundwater mounding will be lost over ~4,500 years. The acidity mobilized by net percolation will be lost over ~580,000 years.

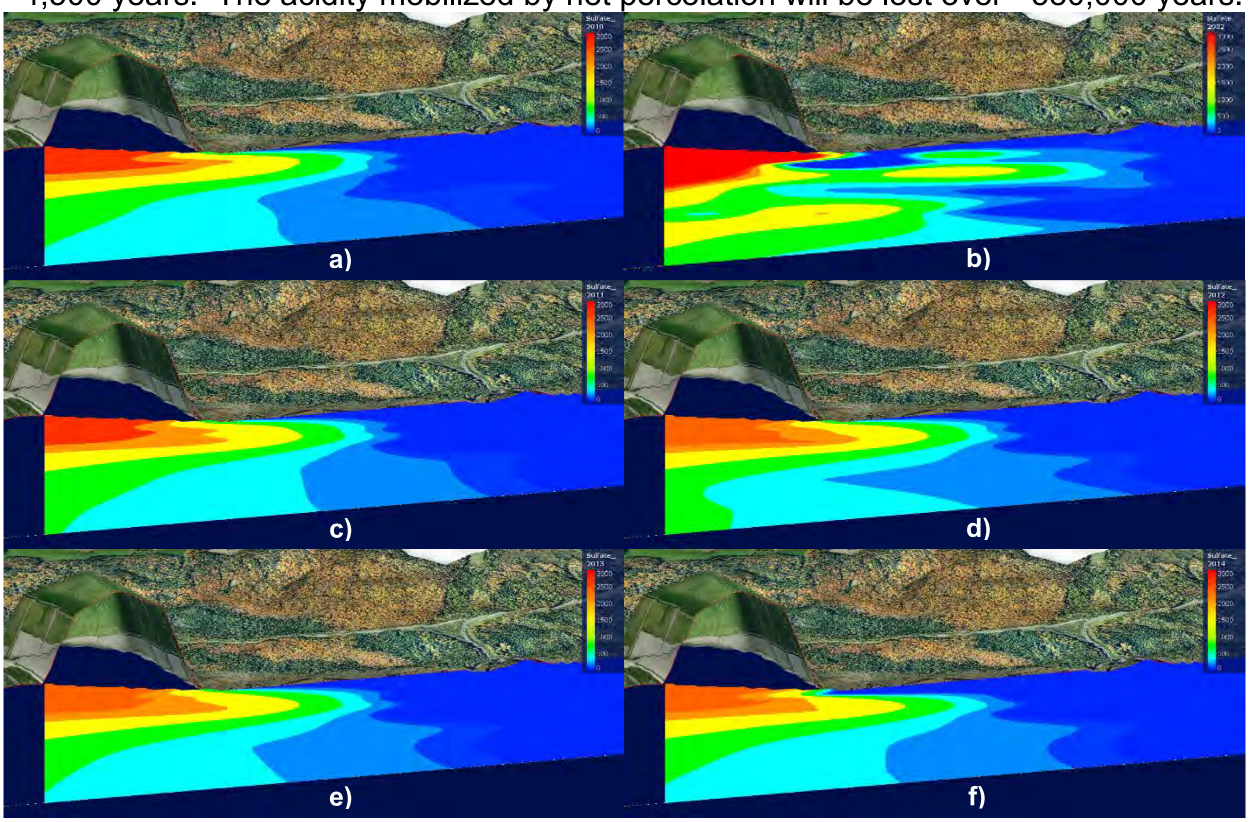


Figure 8. Groundwater sulfate plumes: a) 2002, b) 2010, c) 2011, d) 2012, e) 2013, f) 2014.

3. NET PRESENT VALUE

The mining industry is frequently challenged with the choice to either treat in perpetuity or undergo reclamation activities to manage ARD/ML generating waste. The VJ site operated with active water treatment for more than 30 years prior to reclamation and provides a unique opportunity to evaluate the decision to engage in reclamation. Comparing the two alternatives requires an analysis such as net present value (NPV), a tool often used to quantify closure costs by discounting future cash flows to a common denominator.

An NPV analysis for VJ was completed beginning in 2006 for both the treatment and reclamation alternatives for a 100 year period. Due to the time value of money, a 100 year period is essentially the same as in perpetuity (Phillip and Myers, 2003). Closure costs are typically quite difficult to forecast in the early stages of mine closure planning. Future costs for VJ for both the treatment and reclamation scenarios were based on past operating and maintenance expenditures. This provides a good estimate of future costs, although there are still many assumptions and variables such as discount rate, changes in the price of commodities, and actual improvements observed in the receiving environment. The NPV for each scenario under different discount rates is presented in Table 1. Although the treatment scenario has a lower NPV at a 4% discount rate, one must consider the risk (i.e. probability and impact) of each scenario, both from a financial and environmental stand point. For example, the load generated from the site under each scenario are significantly different, each with their own unique financial and environmental risks. In the conceptual model it was demonstrated that under active treatment approximately 49% of the basal seepage load was collected and treated. It is likely that water quality in the receiving environment would have taken a different trajectory than the cover system scenario given the acid load mass balance. The conceptual model also shows that a system failure under the treatment alternative would have a much greater impact to the receiving environment, and therefore financial impact, with an acid load generation of 894 t/yr compared to 43 t/yr in the reclaimed alternative. Although the NPV analysis has many assumptions, the decision to install the cover system was obviously not solely dependent on NPV. The goal of site closure was to minimize cost while mitigating risk.

It is also important to note that if the discount rate was reduced to say 1%, the difference between NPVs for the two scenarios would be substantially different, with the collect and treat option being substantially higher. The key point to highlight in making this comparison is that hindsight perspective for the VJ WRP closure activities provides a good argument for discount rate being an output from an NPV analysis, and to use this understanding to inform risk, rather than how discount rate is typically utilized (i.e. as an input to NPV analysis).

Table 1. Net present value for collection and treatment in perpetuity, and reclaimed WRP.

Discount Rate (%)	Treatment ($)	Reclaimed WRP ($)

1	29.5M	16.1M
2.5	17.0M	14.6M
4	11.2M	13.8M

4. CONCLUSION

An understanding for current and long-term loading to the receiving environment was developed through the use of a conceptual model. The VJ site has transitioned from active to passive treatment following placement of a low flux cover system, with continued improvements observed in the receiving environment and a reduction in loading from the WRP. Although the reclaimed WRP had a higher NPV (4% discount rate) than treating ARD/ML affected water in perpetuity, loading to the receiving environment was reduced, and environment and financial risks were mitigated.

The total acidity in the VJ WRP is estimated to be lost over 580,000 years, assuming that cover system performance can be maintained over the long-term. While there is variability in the estimated lifespan of the geomembrane layer, it is likely far less than the time required to release the total acidity in the WRP. Degradation of the geomembrane is inevitable, both in the short-term during initial placement procedures, and over the long-term due to service stresses, anthropogenic activities, animal bioturbation, and the effects of vegetation, leading to increased net percolation and therefore loading to the receiving environment. It is imperative that cover systems that incorporate a geomembrane layer are designed to consider the effects of short and long-term degradation to reduce the risk of increased load to the receiving environment.

A well-designed monitoring program is critical to the conceptual model and requires sufficient information is collected, both spatially and temporally. While groundwater flow and contaminant transport numerical modeling is commonly used in predicting long term impacts to the receiving environment, it was not required in this instance given the conceptual model and site information. This case study illustrates the importance and opportunity for using the conceptual model to communicate performance and risk, and ultimately inform management on decisions regarding site practices in order to meet closure objectives.

5. REFERENCES

Benson, C. H., Albright, W. H., Fratta, O., Tinjum, T. M., Kucukkirca, E., Lee, S. H., Scalia, J., Schlicht, P.D., Wang, X. 2010. Engineered Covers for Waste Containment: Changes in Engineering Properties and Implications for Long-Term Performance Assessment, NRC Job Co.

Giroud, J. P., Zornberg, J. G., Beech, J. F. 2000. Hydraulic design of geosynthetic and granular liquid collection layers comprising two different slopes. Geosynthetics International, 7(4-6): 453–489.

Phillip, M., Myers, K. 2003. Regulatory Perspective — Development of Closure Plan With Adequate Financial Assurance for the Protection of Ground Water Quality at the Chino Copper Mine. 6th International Conference on Acid Rock Drainage, (July): 671–676.

Rowe, R.K. 2005. Long-term performance of contaminant barrier systems. Geotechnique, 55(9):631-678.

OVERVIEW OF REHABILITATION RESEARCH OF DIAMOND MINE WASTES IN THE HUDSON BAY LOWLAND

Daniel Campbell
School of the Environment and Vale Living with Lakes Centre
Laurentian University, Sudbury, ON P3C 2J6
dcampbell@laurentian.ca

Keywords: reclamation; best management practices, kimberlite; winter roads; peatlands; uplands; reference conditions; primary succession; technosols; plant traits

The Hudson Bay Lowland (HBL) is a vast and almost pristine subarctic region in Ontario's Far North, forming the third largest wetland in the world (Abraham and Keddy, 2005). These wetlands consist mostly of peatlands, also known as muskeg, underlain by alkaline carbonate deposits and bedrock, with intrusions of economically-significant mineral deposits. This region is rapidly becoming a centre of Canadian mining activity, with active mining at the De Beers Canada Victor Mine and extensive exploration and advanced permitting in the vicinity of the Ring of Fire. Good environmental management is a primary concern in this region.

One of my research goals is to develop protocols for ecosystem rehabilitation after mining in the HBL, in a similar fashion as has been developed for the peat extraction industry in eastern Canada (Quinty and Rochefort, 2003). Such protocols would detail the best management practices (BMPs) needed to return valuable ecosystem services to mining-disturbed landscapes. Much sound information can be borrowed from other high boreal or subarctic regions with extensive development, such as from the Alberta bitumen mines, but local research is also required to build sound BMPs for the HBL.

My students and I have conducted research around the De Beers Canada Victor Mine on (i) the restoration of disturbed peatlands areas; (ii) the selection of suitable reference conditions to determine mine waste reclamation targets; (iii) upland substrate mixes suitable for natural vegetation made from mine wastes and organic materials; (iv) best plant species to reclaim upland mine sites; (v) field tests of the abilities of soil mixes to support native plant species; and (vi) best protocols for seed collection and the valuation. In this presentation, I am reporting on the first four of these research themes.

Disturbances to peatlands represent the largest area of mining disturbance. Some of these peatlands have been transformed to novel upland ecosystems and cannot be returned to peatlands. However, many of these peatlands have linear disturbances such as winter roads and buried pipelines, with disturbed or no vegetation, but suitable peat substrates and hydrology. We have shown that the plant cover and species composition of most winter road clearances over peatlands recover within a decade of abandonment, with the exception of the dwarf trees such as tamarack and black spruce, which will take several decades to return (Campbell and Bergeron, 2012). However, some peatland habitats, such as bogs, have slow recovery (Campbell and Corson, 2014) and will require active rehabilitation. We have tested existing BMPs developed to restore mined peatlands in southeastern Canada, and have shown that these can be simplified, by just spreading moss fragments over bare peat substrates in order to rehabilitate the peat-forming function (Corson and Campbell, 2013).

The rehabilitation of the upland mine wastes is more challenging, in part because of the local rarity of upland habitats, which are either found as islands within the vast peatlands or, paradoxically, along the rivers. We have surveyed these upland habitats and have shown that the carbonate-rich silt loam overburden produced by the Victor Mine resembles these natural soils (Garrah, 2013; Fig. 1). Mixes of silty overburden and peat are the most promising as a viable substrate to return sites towards regional representative upland ecosystems. The vegetation along the river has many early succession species, suitable for the rehabilitation of mine sites (Garrah, 2013). We have concentrated on these species and have attempted to identify their abilities to disperse, establish and grow, using a plant trait approach (Laurin, 2012). This information will help identify appropriate species to rehabilitate upland mine waste environments.

These studies together will help build sound BMPs for ecosystem rehabilitation in the Far North of Ontario. We have worked primarily around the Victor Mine, but we are confident that many of these ecosystem rehabilitation approaches can aid similar mines, even in lower carbonate terrain around the Ring of Fire and elsewhere in the subarctic.

Literature cited

Abraham, K.F. and Keddy, C.J., 2005. The Hudson Bay Lowland. In: L.H. Fraser and P.A. Keddy (Editors), The World's Largest Wetlands, Ecology and Conservation. Cambridge University Press, Cambridge, pp. 118-148.

Campbell, D. and Bergeron, J., 2012. Natural revegetation of winter roads on peatlands in the Hudson Bay Lowland, Canada. Arctic, Antarctic, and Alpine Research, 44(2): 155-163.

Campbell, D. and Corson, A., 2014. Can mulch and fertilizer alone rehabilitate surface-disturbed subarctic peatlands? Ecological Restoration, 32(2): 153-159.

Corson, A. and Campbell, D., 2013. Testing protocols to restore disturbed *Sphagnum*-dominated peatlands in the Hudson Bay Lowland, Canada. Wetlands, 33(2): 291-299.

Garrah, K., 2013. Upland Ecosystems in the Hudson Bay Lowland as Reference Conditions for the Rehabilitation of Mine Waste Piles, Laurentian University, Sudbury.

Laurin, C., 2012. Identification of candidate plant species for the restoration of newly created uplands in the Subarctic: A functional ecology approach, Laurentian University, Sudbury.

Quinty, F. and Rochefort, L., 2003. Peatland restoration guide, 2nd ed. Canadian Sphagnum Peat Moss Association and New Brunswick Department of Natural Resources and Energy, Québec, 106 pp.

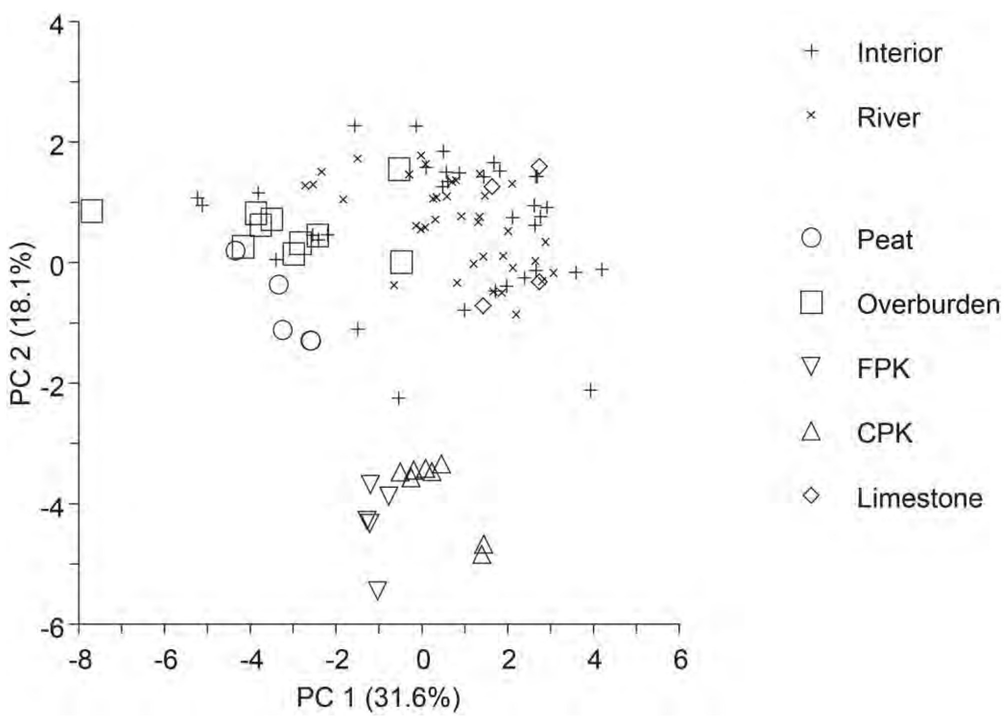


Fig. 1. Principal component analysis of the bioavailable elements in waste materials from the Victor Mine (n = 32) as compared to reference sites in isolated interior uplands (n = 35) and on uplands along the Attawapiskat River (n = 37). The waste materials include peat, silty loam overburden, fine and coarse processed kimberlites (FPK and CPK) and limestone.

MONITORING THE BEHAVIOUR OF SLUDGE IN THE VADOSE ZONE[1]

M. Coleman[2], K. E. Butler[3], N.J. Susak[4]

[1] Michele Coleman, NB Power, Fredericton, NB, E3B 4X1, Canada;
[3] Karl E. Butler, Dept of Earth Sciences, University of New Brunswick, Fredericton, NB E3B 5A3, Canada;
[4] Nicholas J. Susak, Dept of Earth Sciences, University of New Brunswick, Fredericton, NB E3B 5A3, Canada;

Additional Key Words: apparent conductivity, electrical resistivity, lime neutralization sludge deposition, acid mine water, acidity, coal mining, waste rock test pits

Abstract: The Fire Road Mine coal mine in eastern Canada has been a source of acid mine drainage since the mid 1980's. Lime neutralization treatment has been ongoing and lime treatment costs and mine water acidity levels have dropped significantly over time. Placement of the resulting treatment sludge back onto and into the backfilled mine site may be a factor in reducing the mine water acidity. One of the originally defined benefits of placing the sludge back into the waste rock was that the sludge would fill up the void spaces and possibly decrease the rate of oxygen diffusion to the waste rock that was above the ground (mine) water elevation. This would possibly reduce the rate of acid generation.

There were questions about how far the sludge migrated into the vadose zone and the characteristics of that sludge over time. Excavations in the early 1990's indicated that the sludge only dried out near the surface but remained moist at depth. Discussions ensued about variations in conductivity over time being an indication of reduced moisture content in the sludge as it dewatered over time.

Geophysical research projects have been conducted since 2001 to identify variations in conductivity across the backfilled mine sites by the University of New Brunswick, Department of Earth Sciences. The geophysical surveys have been useful for corroborating the decrease in acidity as identified by annual and biannual groundwater well monitoring surveys. The geophysical surveys have

---

[1] Oral Presentation (abbreviated and updated version) presented at the 41st CLRA's National Annual General Meeting and Conference, Timmons, Ontario, June 26-29, 2016.
Also presented at Oral paper (longer version) presented at the 2016 National Meeting of the American Society of Mining and Reclamation, Spokane, WA, Reclaiming the West, June 4 - 9, 2016Published by ASMR; 1305 Weathervane Dr., Champaign, IL 61821.

also been instrumental in identifying the distribution and changing conductivity of the lime treatment sludge in the waste rock.

During the early Spring of 2016 ground water chemistry, apparent conductivity (EM31) and electrical resistivity (ERI) surveys along with test pit trenching were conducted to provide information as to:

1. Is the groundwater acidity continuing to decrease?
2. Is sludge still present in the vadose zone?
3. Why are some previously highly conductive zones becoming less conductive over time?
4. Does the sludge dry out and become less conductive over time?
5. For future investigations, is the benefit of depositing sludge in the vadose zone temporary, or is it long term.

The ground water well monitoring survey illustrated (Figure 1) that acidity levels continued to decrease across the mine site.

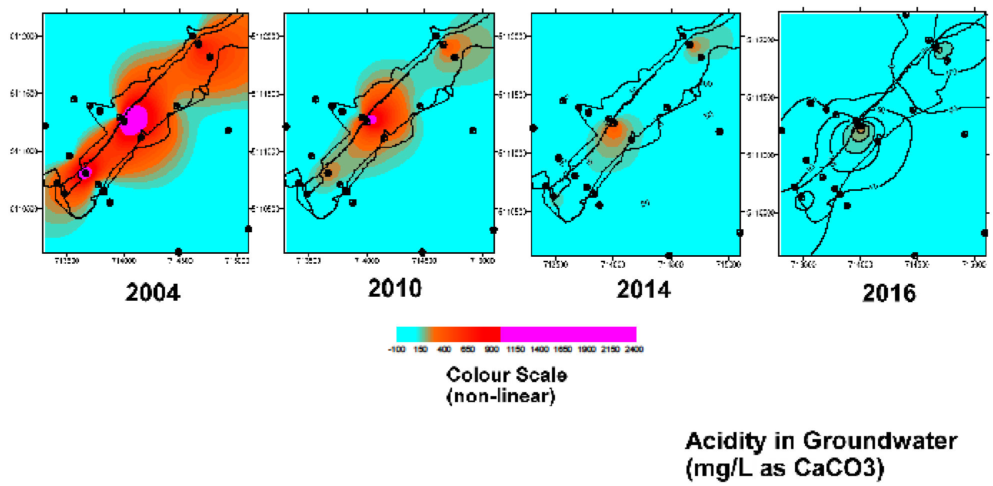


Figure 1 Decreasing acidity in groundwater as determined from monitoring wells across the Fire Road mine site 2004- 2016.

Apparent conductivity and electrical resistivity surveys were re-acquired in areas that had not been surveyed or received sludge deposition for more than a decade by the University of New Brunswick, Department of Earth Sciences. Then, test pits were excavated to allow in-situ measurements of electrical resistivity and sampling for water content in areas where sludge is present or absent. This data was compared to the conductivity /resistivity survey results acquired from surface surveys. This information was also compared to soil texture and soil chemistry from the finer material collected at selected intervals along the test pit wall.

Electromagnetic apparent conductivity (EM31) (Figure 2) and electrical resistivity imaging (ERI) surveys have been instrumental in identifying the locations of highly conductive mine water and what we had postulated is the treatment sludge, which had settled out in the vadose zone during disposal. However, the conductivities of these zones varied across the site, across seasons and over time. Observations from previous geophysical surveys over several seasons indicated that the sludge may dry out and become less conductive.

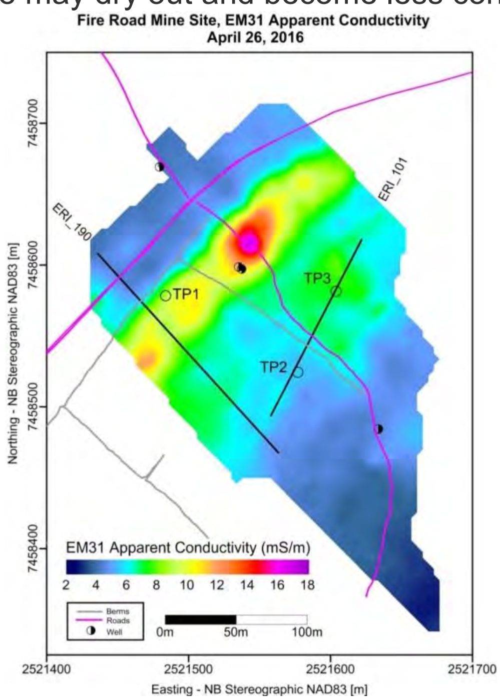


Figure 2 Apparent Conductivity (EM31) in 2016. Note the locations of the test pits (TP1, TP2, TP3) and ERI survey lines 101 and 109.

Along with results of the 2016 geophysical surveys, areas of known sludge deposition history were identified.   Test pit locations were selected on the basis of known sludge deposition based on surface and geophysics information (test pit 1), location of no evidence of sludge deposition (test pit 2) and area of uncertain deposition of some undetermined amendment (test pit 3), as indicated in Figure 2. Test pits (trenches) were excavated to determine presence/ absence of fine grained material in the waste rock void spaces, soil texture and chemistry including pH, major cations and cation exchange capacity. Not surprisingly, test pit 1 had the highest calcium content and the highest conductivity, the soil texture of test pit 2 with no amendment had very little fine grained material in the void spaces between the waste rock, and the unknown material fine material in test pit 3 with the moderate conductivity level contained higher magnesium levels. The surprising result was the consistent low pH (3.5-4) of the fine grained material across all test pits.

ERI results support the hypothesis that resistivity imaging can identify areas where sludge is likely resident in the vadose zone. This information was beneficial to determining the behaviour of the sludge in the vadose zone and initiated discussions about the whether the benefits of sludge in this zone are temporary or long term. Improving the mine water chemistry to "zero lime demand" is the ultimate goal for mine water treatment at this mine.

USE OF QUANTITATIVE ECOLOGICAL AND HUMAN HEALTH RISK ASSESSMENT TO INFORM THE RECLAMATION PROCESS IN THE TIMMINS GOLD CAMP

I.Collins[1], T.Sulatycky[2] L. Alves Beese[1], D. Hart[1] and M. Aziz[2]
[1] EcoMetrix Incorporated, 6800 Campobello Rd., Mississauga, Ontario, L5N 2L8, Canada, icollins@ecometrix.ca
[2] Goldcorp – Porcupine Gold Mines, 4315 Gold Mine Rd, South Porcupine, ON P0N 1H0, thomas.sulatycky@goldcorp.com

Key Words: exposure, toxicity, hazard quotient

Porcupine Gold Mines (PGM) has extensive mining-related land holdings in and around the City of Timmins, Ontario. Many of these holdings consist of closed open pit and underground mines and mine waste management facilities that are at various stages of the closure and rehabilitation process. PGM is currently considering reclamation options for several of its sites, and is going beyond the traditional options selection process by utilizing quantitative Ecological and Human Health Risk Assessment (EHHRA) to inform its reclamation choices as well as technical, financial and other considerations. EcoMetrix Incorporated (EcoMetrix) is undertaking these analyses on PGM's behalf for its Aunor-Delnite, Pamour, and McIntyre and Coniaurum sites.

Generally, EHHRA work in Canada can be undertaken using federal risk assessment guidance (*e.g.,* Health Canada, Environment Canada, Federal Contaminated Sites Action Plan, or Canadian Council of Ministers of the Environment) as well as provincial guidance (*e.g.,* Ontario Regulation 153/04 for redevelopment of brownfield properties). EcoMetrix has adapted the relevant federal and Ontario risk assessment guidance within a mine closure context in order to assist PGM with its closure planning for these sites.

In general, risks from mine-related constituents in the environment are determined by two components: exposure and toxicity. Both components must be present for an environmental risk to exist. In this context, EHHRAs assess exposure and toxicity separately and then combine the results of each assessment to characterize risks. In this context, the risk characterization results are then used to prioritize source control options at closure. Uncertainties and assumptions made are also analyzed and evaluated in the EHHRA. The specific content of each section of the EHHRA is summarized in the following sections.

Problem Formulation

The first, and most important stage in an EHHRA is the Problem Formulation. In this section, the objectives of the EHHRA are defined, as a basis for design of the study. For our sites, the main objectives are to estimate human health and ecological risks for aquatic receiving environments and their associated riparian zones under current environmental conditions, and to develop Risk-Based Design Objectives (RBDOs) to underlie the design of closure options at each site. These RBDOs are being used by EcoMetrix' closure planning team to identify appropriate closure options for each site.

The Problem Formulation also presents an analysis of who or what may be exposed to which environmental Constituents of Potential Concern (COPCs), as well as where and how these exposures may take place. EcoMetrix has relied on previous reports such as Environmental Effects Monitoring (EEM) studies, annual monitoring reports, and existing closure plans prepared for all three sites to determine the answers to these questions.

EcoMetrix has consulted with Ministry of the Environment and Climate Change (MOECC) and Ministry of Natural Resources and Forestry (MNRF) to ensure that this section is complete. The end result of this analysis is the Conceptual Site Model (CSM), which summarizes visually how constituents move from potential source areas on a mine site, such as a tailings impoundment, through the environment to the human and ecological receptors that may be exposed. An example of a CSM diagram is presented in Figure 1.

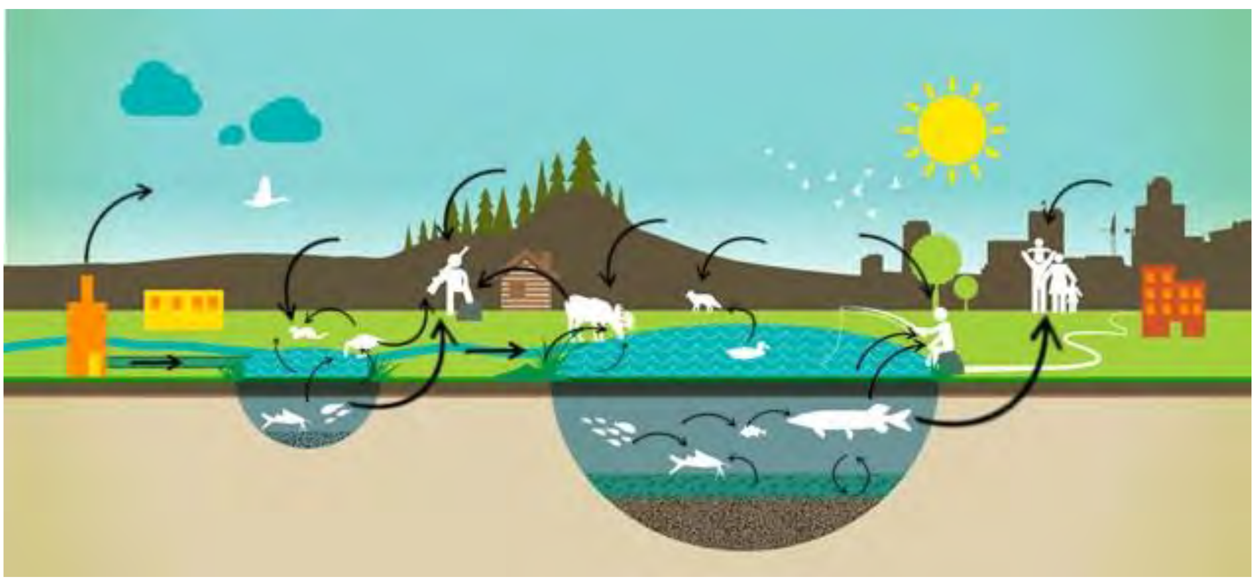


Figure 1: Illustration of a Conceptual Site Model (CSM)

Exposure Assessment

In the Exposure Assessment, EcoMetrix has investigated how much each human and ecological receptor is exposed to each COPC. Based on the CSM, an environmental fate and transport (pathways) model has been constructed that estimates the exposure value (EV) for each person or ecological receptor identified in the Problem Formulation. EV estimates are calculated using conservative exposure factors so that potential exposures are not underestimated. Site-specific data have been used wherever available; for these sites, measured concentrations of constituents in surface water, sediment, soil, terrestrial and aquatic plant tissues, and benthic invertebrate tissues have been used to quantify exposure.

Toxicity Assessment

In the Toxicity Assessment, also called an Effects Assessment in an ecological context, an effect level or benchmark value (BV) is identified for each COPC. The derivations of BVs are conservative by design, and may use the lowest available effect levels or include uncertainty factors to make sure the assessment does not underestimate risks. For humans, potential cancer, non-cancer, and developmental (fetal) effects have been evaluated. For ecological receptors, survival, growth, and reproductive effects were evaluated. In order to accomplish this task, EcoMetrix searched the literature for the health effects that each COPC might cause, summarized the results of the studies underlying the investigation of these health effects, and tabulated the numerical results.

Risk Characterization

In this section of the report, EcoMetrix has combined the results of the Exposure and Toxicity (Effects) Assessments into single measures of risk. Given the conservatisms in the Exposure and Toxicity (Effects) Assessments, these measures of risk are expected to be overestimates of true health risks. For non-cancer and developmental effects in humans, and for ecological receptors, the risk measure is the Hazard Quotient (HQ), which is calculated as the EV divided by the BV. For cancer effects in humans, the risk measure is the Incremental Lifetime Cancer Risk (ILCR). EcoMetrix has compared these measures to the targets set out in the various federal and provincial guidance documents to determine whether risks to human or ecological receptors could be ruled out. The relative magnitudes of the HQs and ILCRs have therefore been discussed in the context of uncertainties and overall conservatism to allow prioritization of environmental risks to be undertaken.

Risk-Based Design Objectives

For the higher priority human and ecological health risks, mitigation options were considered to be necessary at closure to protect the potentially affected receptors. Closure and reclamation options have therefore been developed that target the sources of, and pathways for, the high priority COPCs to mitigate potential health risks. These options have been developed using RBDOs. The derivation of a RBDO starts from the target measure of risk according to federal or provincial guidance, and works backwards through the risk assessment process to arrive at an exposure concentration that is not expected to result in a human or ecological risk.

Once all of the required RBDOs have been estimated using this reverse process, the Risk Characterization step is re-evaluated to demonstrate that with closure options in place, high priority health risks in the riparian receiving environment will have been reduced to an acceptable level. The closure options meeting the RBDOs can then proceed to further technical and financial analyses, as well as any other analyses deemed appropriate, so that a scientifically defensible closure plan that is protective of humans and the environment can be developed.

COLLECTION, PROPAGATION AND DISPERSION OF SHRUBS AND NON VASCULAR SPECIES FOR RECLAMATION IN HARSH NORTHERN ENVIRONMENTS

Sarah A. Ficko[1], M. Anne Naeth[1]
[1]University of Alberta, Edmonton, AB, T6G 2H1

Key Words: Revegetation, shrub cuttings, lichens, reclamation, arctic, tundra, Northwest Territories

Introduction

Extraction of natural resources can leave large areas of disturbed land, partially or completely stripped of vegetation, making them unstable, erosion prone and unable to provide food or habitat for fauna. Shrubs and non-vascular species in biological soil crusts (e.g. lichens) are integral components of tundra ecosystems. Without reclamation, it could take hundreds to thousands of years for disturbed areas to recover naturally due to extreme environmental conditions.

Assisted revegetation is intended to accelerate plant establishment and growth on disturbed sites. Current reclamation practices are limited by lack of native plant material, harsh environmental conditions, high costs and lack of previous regulatory requirements. Many northern shrub species have low, unknown, or cyclic seed production, which creates challenges for collection and storage. Due to the lack of commercial seed suppliers in the north, propagation of shrub species by cuttings has high potential to create a consistent source of plant material to reclaim large areas, if timely root development can be promoted. Lichens, mosses and biological soil crusts play key roles in harsh environments as they stabilize soil, modify infiltration, increase soil fertility, and prevent erosion. However, only limited research has been conducted on their use in reclamation due to their slow growth rates and historical perception as being less important compared to vascular plants.

Researching innovative, cost effective and sustainable methods to reclaim disturbed northern land and develop self-sustaining communities will create techniques to restore disturbed land and assist with conservation of one of the few remaining natural environments worldwide. The objective of this research program is to develop and improve methods for collection, propagation and dispersion of native shrub and lichen species in harsh northern environments.

Methods

Shrub cuttings

Shrub cuttings were collected from eight dominant tundra species at Diavik Diamond Mine, Northwest Territories, located approximately 300 km northeast of Yellownife. Cuttings were collected from *Arctostaphylos rubra* (Rehder & Wilson) Fernald (red bearberry), *Betula glandulosa* Michx. (bog birch), *Empetrum nigrum* L. (crowberry), *Ledum* decumbens L. ssp. *decumbens* (Aiton) Lodd. ex Steud. (marsh labrador tea), *Loiseleuria procumbens* (L.) Desv. (alpine azalea), *Salix* sp (willow species). *Vaccinium uliginosum* L. (bog bilberry) and *Vaccinium vitis-idaea* L. (bog cranberry).

The effects of three common horticultural practices affecting rooting of shrub cuttings were evaluated. The main objectives were to determine if the concentration (0, 0.1, 0.4, 0.8 %) of a common growth hormone (indole-3-butyric acid (IBA)), soaking length (0, 1, 3, 5, 10, 20 days) and time of year of collection (spring, summer, fall) could promote root initiation and development in growth chamber experiments over 60 days to create a more consistent source of plant material available for reclamation of disturbed northern sites. Treatment choices were selected based on a review of scientific literature, common horticultural practices and ease of application.

Lichens

Lichens were collected at Diavik Diamond Mine for a multi year field experiment. Specific objectives are to determine the effects of substrate type (crushed rock, lake sediment, processed kimberlite), containment type (none, jute, erosion control material, erosion control material and jute, woody debris, woody debris and jute, tundra soil, tundra soil and jute), and placement type (none, slurry, dry placement) on growth and survival of lichens. One hundred grams of sieved lichen material was used per 50 x 50 cm plot.

Results and Discussion

Shrub cuttings

Cuttings from all eight species collected at Diavik have the capacity to produce roots, although cuttings with roots across all treatments in spring, summer and fall, ranged from 1 % for fall *Arctostaphylos rubra* cuttings to 88 % for spring *Salix* sp. cuttings. Time of year of collection had a strong influence on root development for *Salix* sp. and *Vaccinium vitis-idaea* cuttings. Effects of soaking length and IBA concentration were less clear due to limited root initiation for most species, and interactions with time of year of collection. While species specific factors play a role in root initiation and development, other factors including environmental conditions may also be involved.

Lichens

After one year, preliminary results indicate that lichen fragment retention was enhanced with jute, lichens were more frequently associated with micro topography, and all treatments with dispersed lichen fragments (dry or slurry) had similar species frequency. Plots with no lichens had very few lichens. Most lichen fragments appeared health. A final assessment of lichen plots will occur in August 2016.

Preliminary Conclusions

Cuttings from all eight shrub species have the ability to produce roots, but root initiation and development was species specific. Time of year of collection had the strongest influence on root initiation and development. Placement of lichen fragments on plots significantly increased presence of all species monitored, while type of placement (dry, slurry) did not appear to play a significant role in species survival or frequency. Plots with jute appeared to have the highest retention of lichen fragments across all treatments. On-going and future research will potentially lead to more informed northern revegetation guidelines.

Acknowledgements

We thank Diavik Diamond Mine Inc. for their sponsorship of this research, including in-kind and other support. Funding was also provided by a Natural Sciences and Engineering Research Council of Canada (NSERC) CREATE grant to the Land Reclamation International Graduate School. Scholarships and grants were received from NSERC, Alberta Innovates Technology Futures, Alberta Innovation and Advanced Education, Association of Canadian Universities for Northern Studies, Helmholtz Alberta Initiatve, UAlberta North (formerly Canadian Circumpolar Institute), and the University of Alberta.

EVALUATION OF THE PERFORMANCE OF REHABILITATION ACTIVITIES AT THE HOLLINGER TAILINGS MANAGEMENT AREA (HTMA)

Brian Fraser[1], Sarah Barabash[1] and Thomas Sulatycky[2]
[1]EcoMetrix Incorporated, Mississauga, ON, L5N2L8
[2]Porcupine Gold Mines – Goldcorp Inc., Timmins, ON, P0N1H0

Key Words: aquatic habitat and biota, water quality trends, monitoring program revisions

Introduction

The Hollinger Tailings Management Area (HTMA, or Site) is located in Timmins, Ontario northeast of the intersection of Highways 655 and 101, and in part occupies a portion of the historical basin of Gillies Lake. The Site is owned and maintained by Goldcorp Canada Ltd. Porcupine Gold Mines (PGM). The HTMA consists of two distinct historical mine waste deposits, the Hollinger (or Gillies Lake) Tailings Area (HTA) and the McIntyre Concentrate Dump (MCD). Deposition of combined gold tailings to the Gillies Lake basin by Hollinger Mines took place from approximately 1917 to the late 1950's. The MCD was used by the McIntyre Mine for storage of off spec sulphide rich gold concentrate from the sulphide flotation process in the mill between the 1933 and 1956, at which time it was filled to capacity. In 1988 and 1989, a gold recovery project took place at the site that consisted of re-mining the tailings and sulphidic concentrate, resulting in the exposure of large areas of unoxidized tailings and concentrate, and subsequent generation of acid rock drainage that drained towards Town Creek.

Rehabilitation Activities at the HTMA

Plans for rehabilitation at the Site were documented in the HTMA Closure Plan (SENES, 2007) and its subsequent amendment (Goldcorp Canada, 2009). For the purposes of Site closure planning the HTMA was divided into six closure areas including (Figure 1):

- Area 1, the McIntyre Concentrate Dump;
- Area 2, Gillies Pond, a remnant of Gillies Lake within the HTMA;
- Area 3, situated southwest of Gillies Pond;
- Area 4, the western portion of the site immediately adjacent to Highway 655
- Area 5, situated at the north end of the site adjacent to the McIntyre Tailings Management Area; and,
- Area 6, Town Creek and the area northwest of Gillies Pond.

Figure 1. Location of the Hollinger Tailings Management Area (HTMA) in Timmins, ON

During the closure planning process PGM collaborated with the City of Timmins and Mattagami Region Conservation Authority (MRCA) to determine a long-term land-use plan that would benefit the community. According to the plan, the Site will be maintained as a green space for walking and hiking and will not be used for activities that could physically disrupt the vegetation. The overall objectives of the rehabilitation work at HTMA included:

- Improving surface water quality in Town Creek and groundwater quality on site in the long term;
- Improving the long-term physical stability of the tailings to eliminate mine hazards, stabilizing soils to reduce steep slopes and limit surface erosion so that no further tailings enter Town Creek; and,
- Improving the aesthetics of the HTMA by removing debris and vegetating the surface.

Closure and rehabilitation works at the Site were implemented in two phases (AMEC, 2014a; 2014b). Phase 1 included the following general activities that took place during 2009 and 2010:

- Construction of temporary and permanent outlet control structures for Gillies Pond;
- Diversion of drainage from the MCD and the southern portion of Area 3 into the south end of Gillies Pond;
- Diversion of drainage from the northern portion of Area 3 into the north end of Gillies Pond;
- Rehabilitation of Town Creek and construction of a new channel from the outlet of Gillies Pond to a point 500 metres (m) downstream of Hwy 655; and
- Rehabilitation of Areas 5 and 6.

Phase 2 was completed in 2011 and 2012 and included the following general activities:

- Relocation of the upper 0.5 m of acid generating tailings from the southern portion of Area 3 into Gillies Pond to prevent further acid generation;
- Relocation of concentrate from the MCD into Gillies Pond to prevent further oxidation and acid generation;
- Application of a cover to southern portion of Area 3 and the MCD comprising sand and gravel topped with a vegetative medium layer (biosolids);
- Reshaping the banks of Gillies Pond and placing rip rap on the slopes for erosion protection; and,
- Rehabilitation of the former settling ponds located east of Gillies Pond.

The application of biosolids in the fall of 2012 was the final rehabilitation activity at the HTMA. At present the primary activities at the Site are related to monitoring.

Assessment of the Performance of the Rehabilitation Activities

PGM retained EcoMetrix Incorporated (EcoMetrix) to assess the performance of the rehabilitation measures at the HTMA (EcoMetrix, 2015). Among other things, the assessment comprised the following:

- A summary and interpretation of a post-rehabilitation biological monitoring survey in Town Creek;
- An analysis of water quality trends in, and treatment requirements for, Gillies Pond; and,
- An evaluation of water quality in Town Creek post-rehabilitation.

The first of two planned post-rehabilitation biological monitoring surveys in Town Creek was implemented in the fall of 2014. The survey included the characterization of water quality, sediment quality, benthic macroinvertebrate community structure, fish community structure and aquatic habitats in Town Creek (Azimuth 2015). Based on the results of this study it can be concluded that:

- The channel constructed to convey drainage from the HTMA to Town Creek is functioning as designed, as is available as aquatic habitat over its entire length.
- The establishment of riparian vegetation is advancing along the entire length of the channel, dominated at this time by herbaceous plants (grasses) with a limited number of poplar and willow seedlings having emerged in the downstream section.
- Metal levels in creek sediments show an increasing trend with increasing distance downstream of the HTMA and were within the range of those measured in previous assessments (Minnow, 2001, 2003).
- Benthic invertebrate density and diversity was similar at the three survey stations with some shift in community composition from upstream to downstream, likely associated with habitat differences. Benthic invertebrate community endpoints were similar to those measured in previous assessments (Minnow, 2001, 2003).
- Fish were collected along the entire study area length, which included the constructed channel and Town Creek to its mid-reach, indicating that water quality even in the most upstream areas of the system is not a limiting factor to fish presence. There appear to be no complete barriers to upstream fish

migration from the Mattagami River to the mid-reach of Town Creek, though some limits to fish passage within certain areas of the creek appear to be present. Fish abundance and diversity was higher in 2014 than it was in previous assessments (Minnow, 2001, 2003).

Regular surface water sampling at the Gillies Pond Outlet has been conducted since 1991. Historically water quality at this location was influenced by both source term loadings associated with the tailings deposited in the HTA, as well as the solids stored in the MCD. One of the primary objectives of the rehabilitation measures implemented at the Site was to address these source term contributions for the purpose of improving water quality downstream in Town Creek.

For the purpose of the analysis of water quality trends in the HTA constituents of potential concern (COPCs) including pH, arsenic, copper, lead, nickel, total suspended solids (TSS) and zinc were considered. Following implementation of rehabilitation measures at the HTMA the following has been noted. The pH in Gillies Pond has been circum-neutral to moderately alkaline, has been within the discharge limit range of 6.0 to 9.5 at all times and has not been subject to the depressed pH and seasonal patterns seen between the late 1990s and 2008 (Figure 3). Concentrations of arsenic (Figure 4) and metals (Figures 5 through 8) have decreased by several orders-of-magnitude, have met their respective discharge limits at all times and have not shown the seasonal pattern seen between the late 1990s and 2008. The TSS levels (Figure 9) have become less variable following the implementation of rehabilitation measures, have been below the daily and monthly limits since 2009 and have been typically less than 5 mg/L.

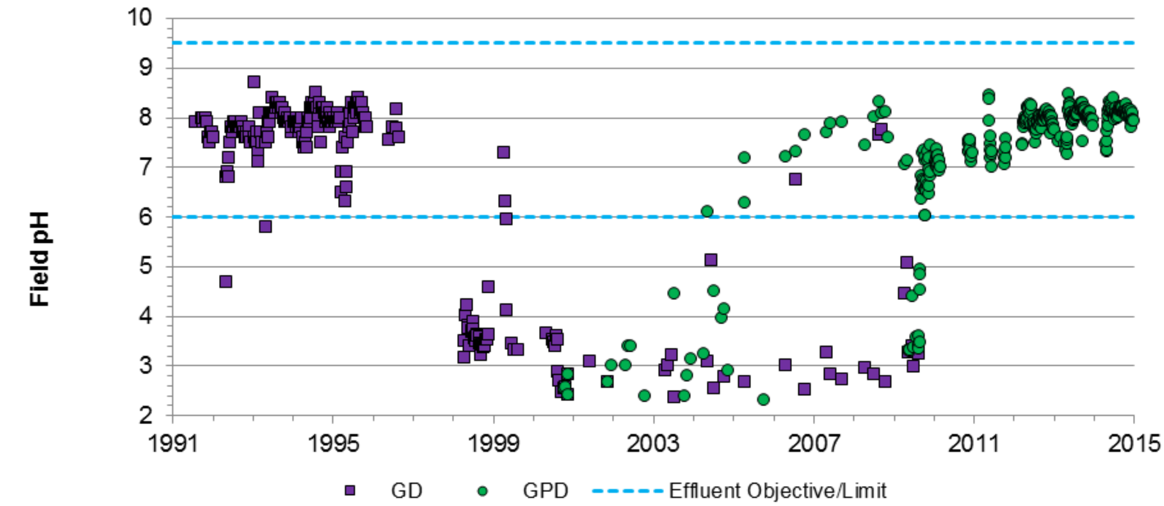


Figure 3. pH Levels in Gillies Pond 1991 through 2014

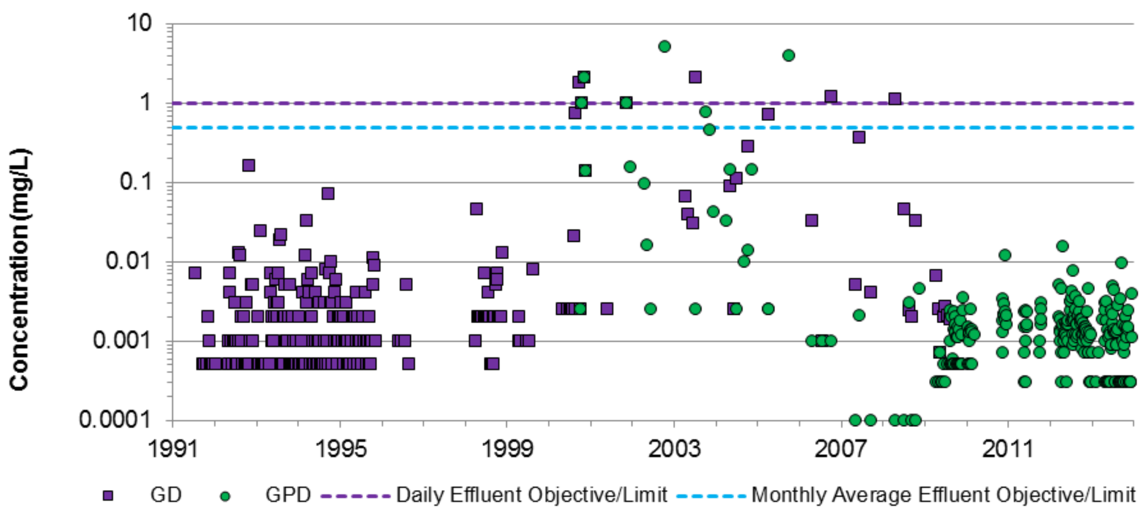


Figure 4. Arsenic Concentrations Levels in Gillies Pond 1991 through 2014

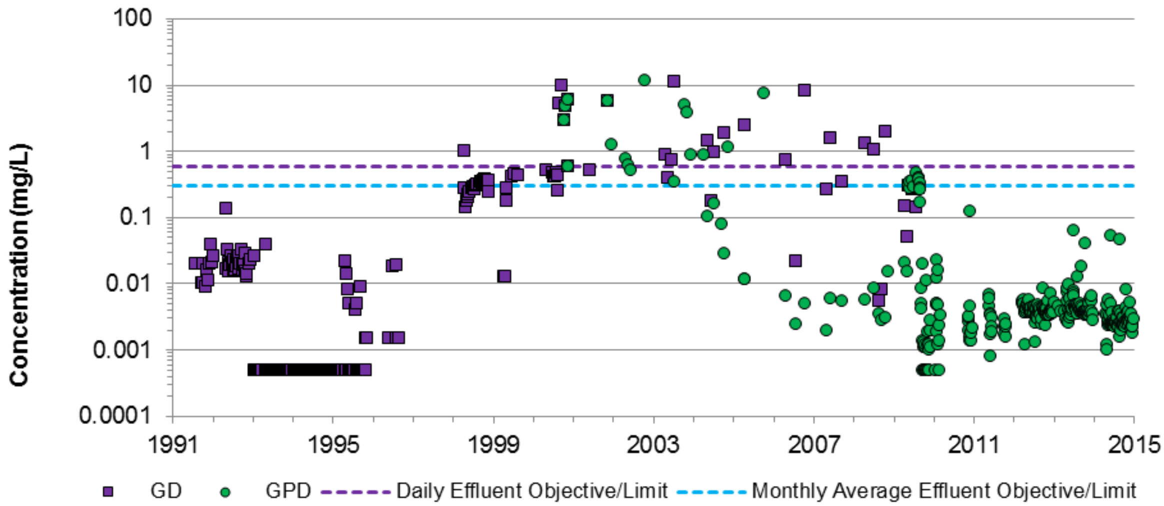


Figure 5. Copper Concentrations in Gillies Pond 1991 through 2014

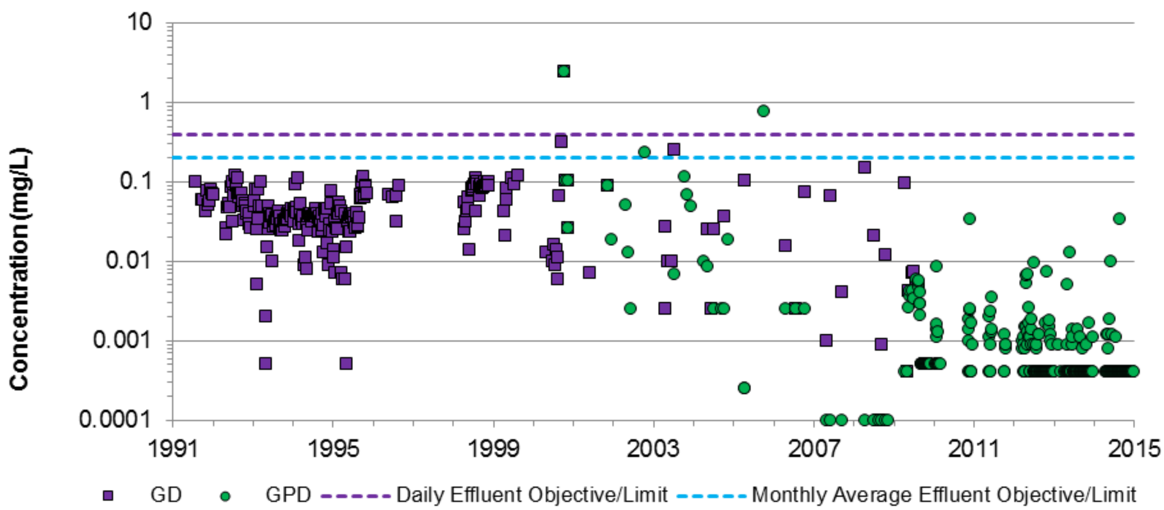


Figure 6. Lead Concentrations in Gillies Pond 1991 through 2014

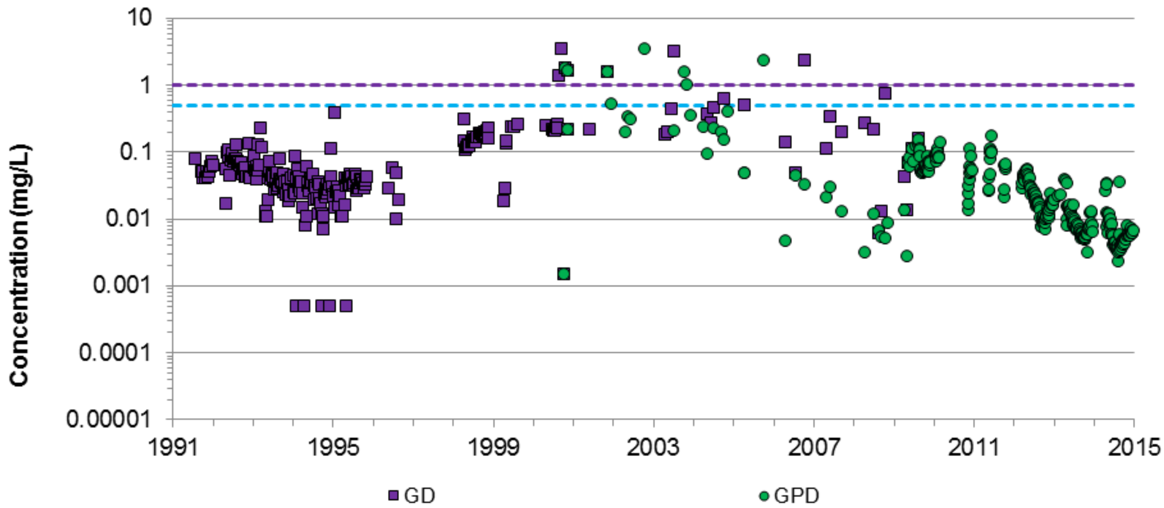


Figure 7. Nickel Concentrations in Gillies Pond 1991 through 2014

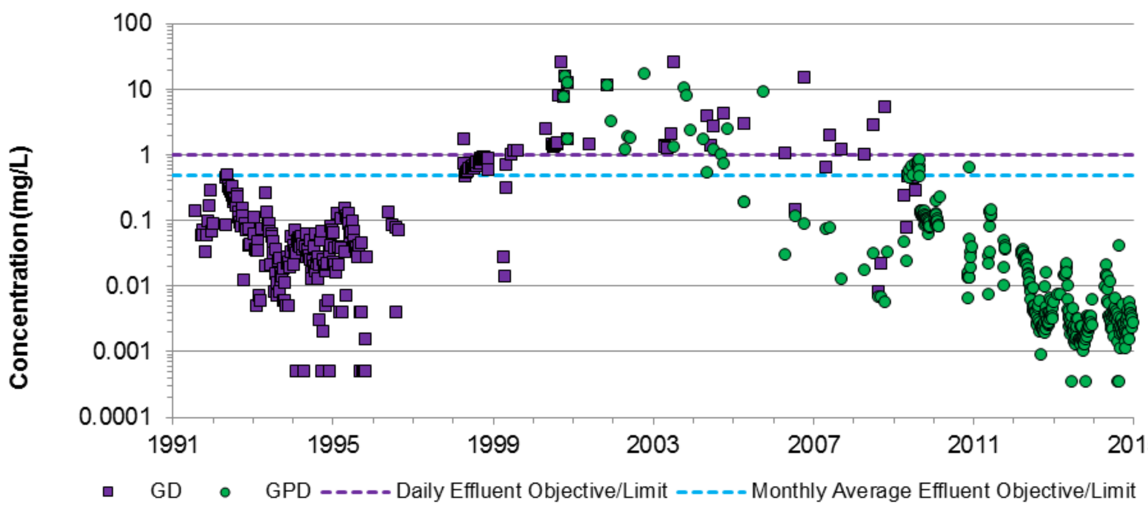


Figure 8. Zinc Concentrations in Gillies Pond 1991 through 2014

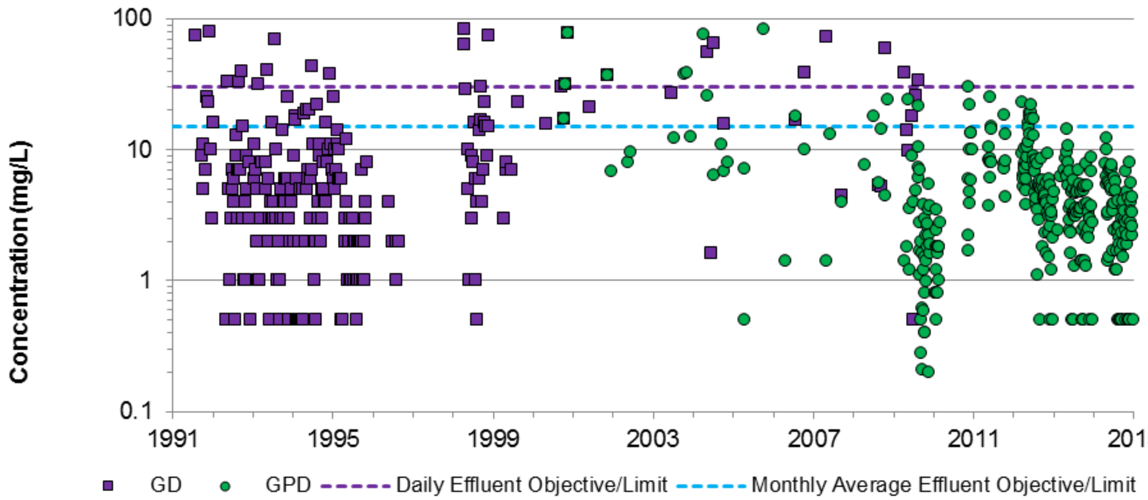


Figure 9. TSS Levels in Gillies Pond 1991 through 2014

Regular surface water sampling in Town Creek has been conducted since 1998. The influence of HTMA discharge on water quality in Town Creek prior to rehabilitation measure implementation was more significant in the headwater area of the creek than it was in the mid and lower reaches. In the headwaters water quality largely mimicked that of the HTMA discharge, whereas in the mid and lower reaches COPC concentrations varied considerably. Improvements in water quality in the upper reach of Town Creek (monitoring station DS655 at the Hwy 655 crossing) were observed immediately following the implementation of rehabilitation measures at the HTMA. The pH levels became circum-neutral to slightly alkaline (Figure 10), arsenic (Figure 11) and metal (Figure 12 through 15) levels decreased by one to two orders of magnitude and TSS levels (Figure 16) have decreased and are generally less variable. Since implementation of rehabilitation measures at the HTMA COPC levels have met site-specific surface water quality objectives (SWQOs) in almost all instances and moreover COPC levels are generally close to or below Provincial Water Quality Objectives (PWQOs).

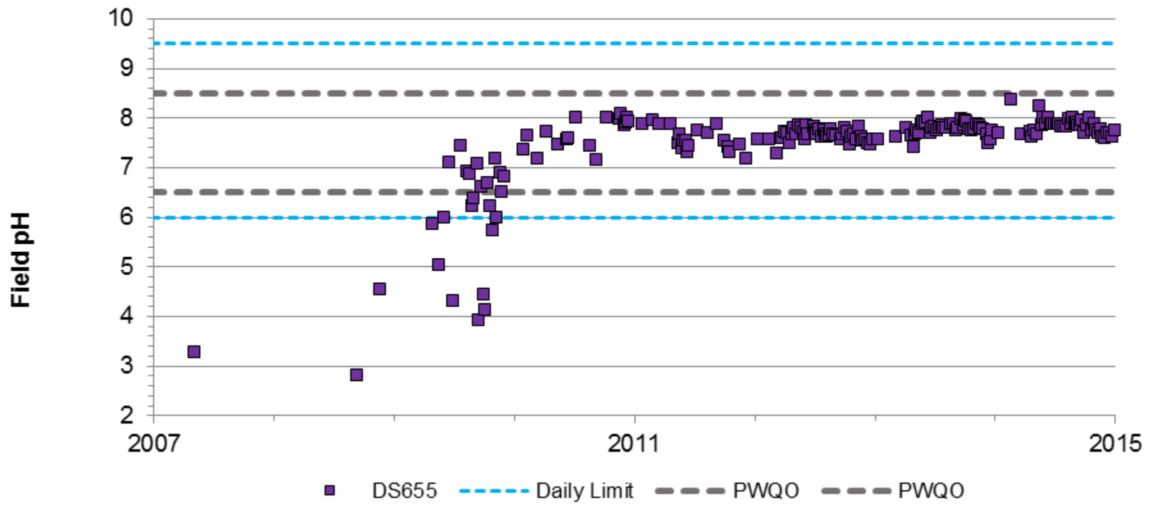


Figure 10. pH Levels at Town Creek Monitoring Station DS655, 2007 through 2014

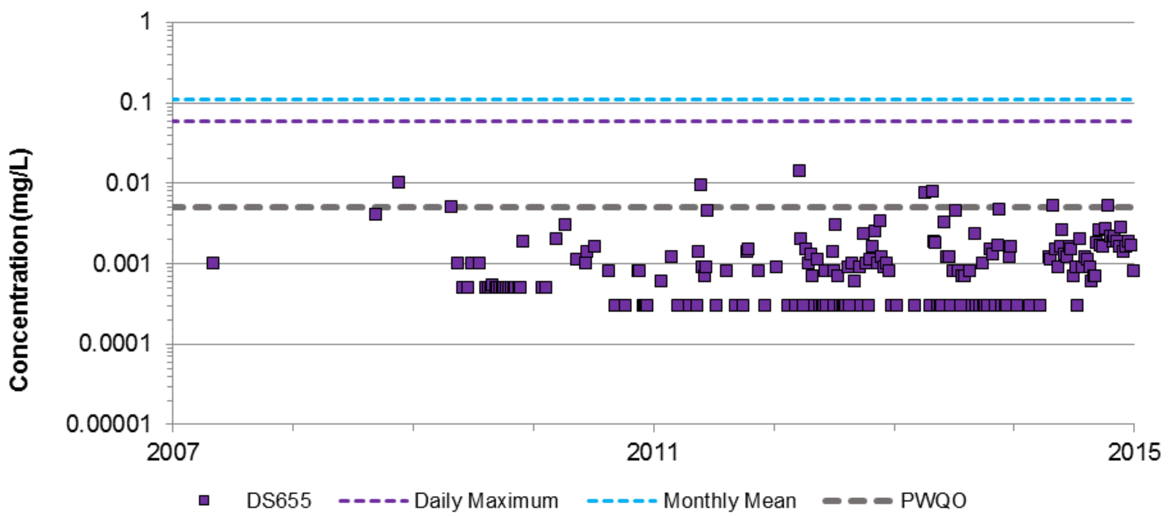


Figure 11. Arsenic Concentrations at Town Creek Monitoring Station DS655, 2007 through 2014

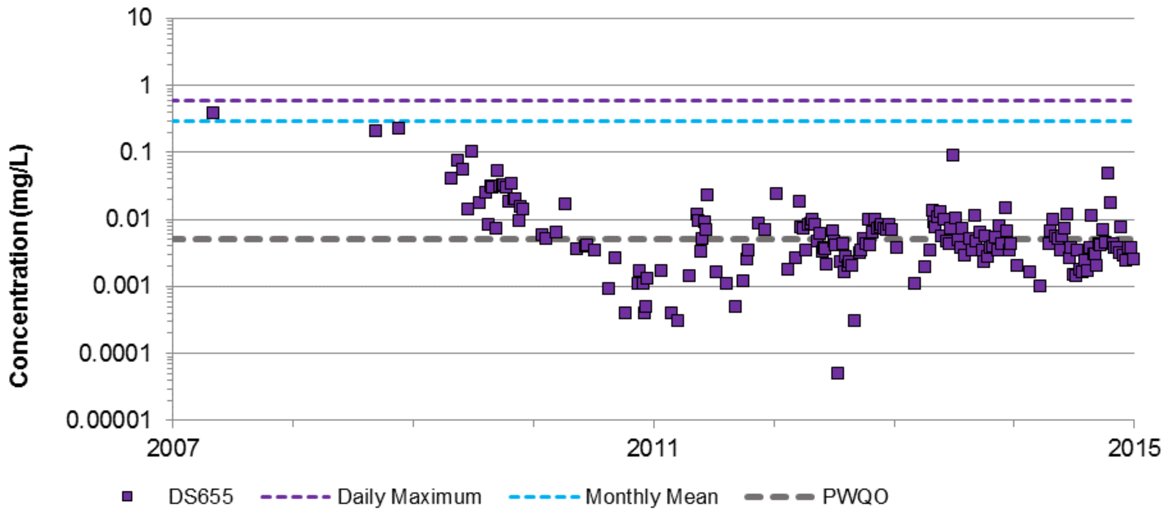


Figure 12. Copper Concentrations at Town Creek Monitoring Station DS655, 2007 through 2014

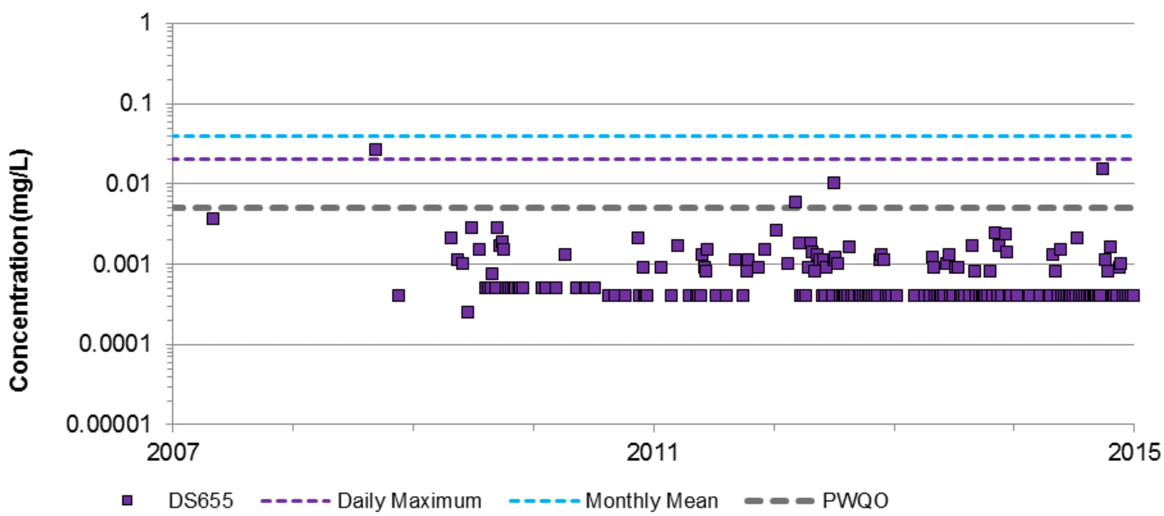


Figure 13. Lead Concentrations at Town Creek Monitoring Station DS655, 2007 through 2014

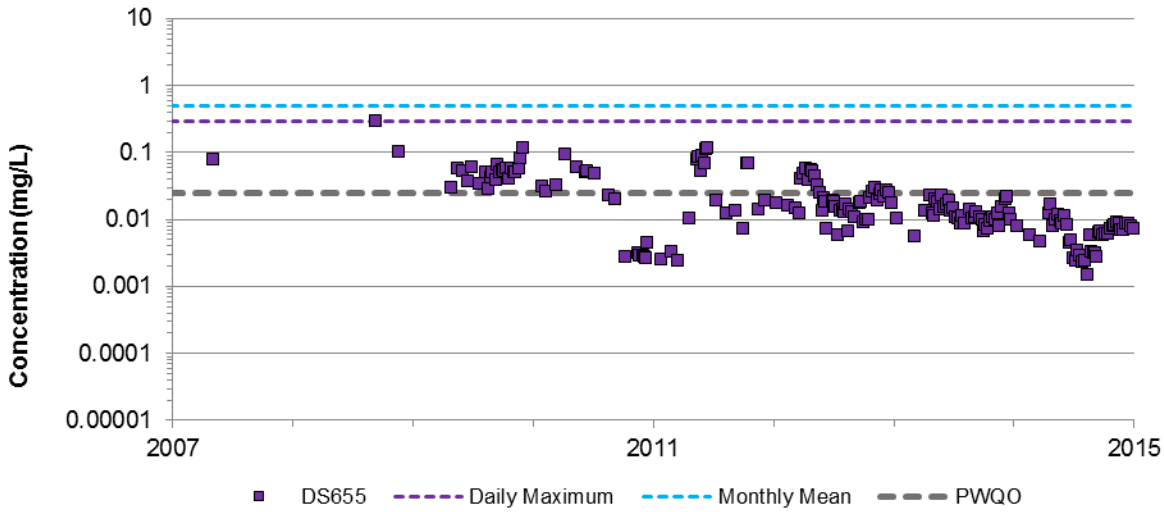


Figure 14. Nickel Concentrations at Town Creek Monitoring Station DS655, 2007 through 2014

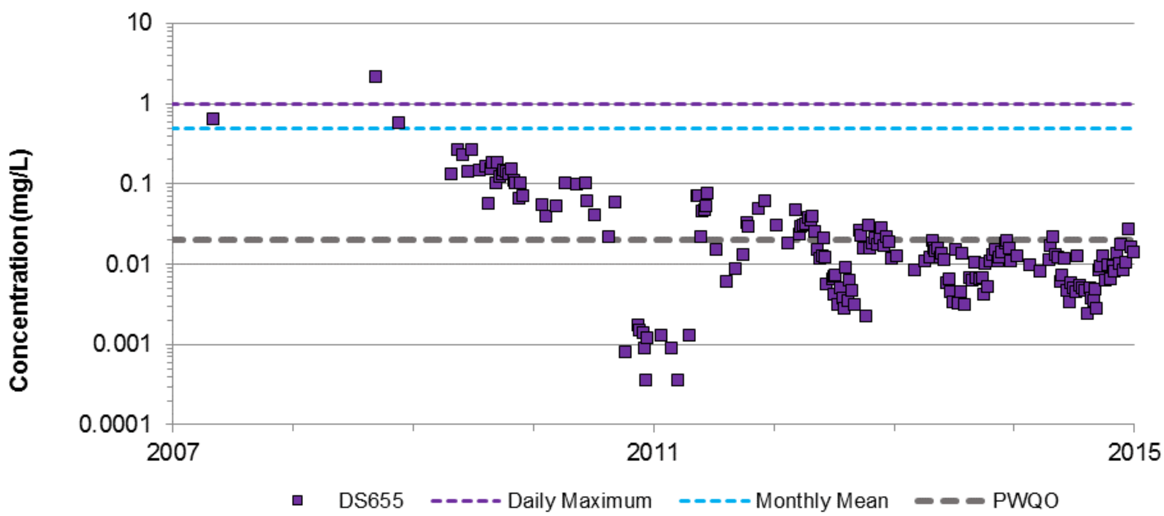


Figure 15. Zinc Concentrations at Town Creek Monitoring Station DS655, 2007 through 2014

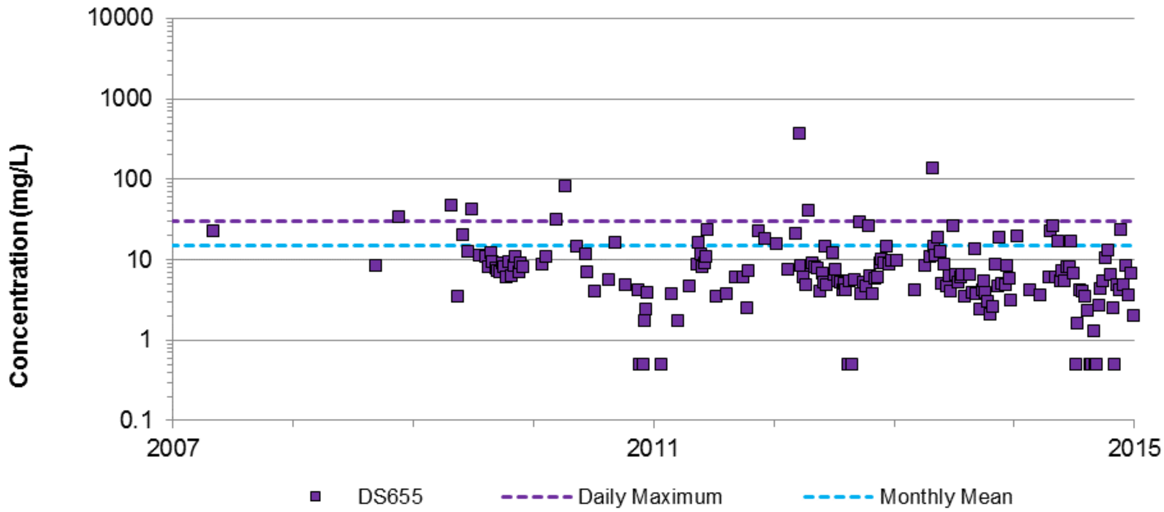


Figure 16. TSS Levels at Town Creek Monitoring Station DS655, 2007 through 2014

AMEC (2007) developed an empirical model to predict post-rehabilitation water quality at the Hwy 655 crossing (monitoring station DS655). The predictions, as well as actual concentrations over the period 2012 through 2014 at DS655 are summarized in Table 2. Overall it is evident that the HTMA rehabilitation measures had a greater positive influence on water quality than predicted. The pH values at DS655 post-rehabilitation exceed predictions for both the spring freshet and summer low flow conditions by 1 to 2 pH units on the low end of the prediction and by between 0.5 and 1.0 pH units on the high end. For each of the other COPCs monitoring data collected between 2012 and 2014 are one to two orders of magnitude less than the values predicted for the post-rehabilitation era.

Table 1: Comparison of Predicted Post-Rehabilitation CPOC Predictions to Actual 2012 to 2014 Monitoring Data

CPOC	**Predicted Concentrations (mg/L)**		**Actual Concentrations (mg/L)**	
	Freshet	Low Flow	Freshet	Low Flow
pH	5.2 to 7.0	6.0 to 7.5	7.2 to 8.0	7.4 to 8.0
Arsenic	0.006 to 0.02	0.001 to 0.1	0.0003 to 0.0077	0.0003 to 0.0052
Copper	0.03 to 0.2	0.01 to 0.3	0.0026 to 0.0135	0.0003 to 0.0177
Lead	0.01 to 0.1	0.01 to 0.1	0.0004 to 0.0015	0.0004 to 0.0152

Nickel	0.05 to 0.2	0.01 to 0.2	0.0045 to 0.0227	0.0006 to 0.0227
Zinc	0.2 to 0.9	0.1 to 1.0	0.0006 to 0.0263	0.0022 to 0.03

Recommendations for Revisions to the Site-wide Monitoring Program

Based on the results of the performance assessment of the rehabilitation measures at the HTMA recommendations for revisions to the current site-wide monitoring program have been developed. The program will continue to comprise three primary components including groundwater, surface water, and aquatic biology. Monitoring for each of these components has been defined in terms of: sampling locations; sampling frequency; sample collection protocol including analytes and/or endpoints of interest; sample analysis protocols; and, interpretation of results. General requirements and frequency for reporting, as well as for quality assurance and quality control provisions have also been considered. Site-wide monitoring program commitments will be defined and harmonized in consideration of current Site-environment interactions, post site-wide remediation, as part of the planned 2016 Closure Plan Amendment.

Summary and Conclusions

Over the period 2009 through 2012, PGM implemented rehabilitation measures whose purpose was to, in part, improve the long-term surface and groundwater within Town Creek. Improvements measured in water quality at the HTMA, and downstream in Town Creek, were observed immediately following implementation of rehabilitation measures. The effectiveness of the rehabilitation measures are reflected by the extent to which the quality of water released from the HTMA has improved and stabilized and by all COPCs that have met their respective discharge limits at all times. The improvements in water quality at the HTMA are also reflected in Town Creek, particularly in its upper reach where the influence of the HTMA discharge is more conspicuous. Following the implementation of rehabilitation measures COPC levels have met respective SWQOs in almost all instances and moreover COPC levels are generally within the range of, or below PWQO levels. The results of the 2014 Town Creek biological survey indicated that water quality even in the most upstream areas of the system is not a limiting factor to fish distribution. Overall, based on the data and analyses presented herein it can be concluded that the rehabilitation measures implemented at the HTMA have been successful in meeting the Closure Plan objectives as it concerns surface water quality.

Literature Cited

AMEC Environment & Infrastructure, a Division of AMEC Americas Limited (AMEC). 2007. Predictions of Chemical Concentrations at Highway 655 after Closure of the Hollinger Tailings Management Area. June 2007.

AMEC Environment & Infrastructure, a Division of AMEC Americas Limited (AMEC). 2014a. Hollinger Tailings Management Area Rehabilitation Project Summary. January 2014.

AMEC Environment & Infrastructure, a Division of AMEC Americas Limited (AMEC). 2014b. Hollinger Tailings Management Area Summary As-Built Report. January 2014.

Azimuth Environmental Consulting Inc. (Azimuth). 2015. Goldcorp Canada Ltd. Hollinger Tailings Management Area 2014 Biological Monitoring. March 2015.

EcoMetrix Incorporated (EcoMetrix). 2015. Hollinger Tailings Management Area Comprehensive Evaluation Report. March 2015.

Minnow Environmental Inc. (Minnow). 2001. Biological Assessment – McIntyre Concentrate Dump Closure.

Minnow Environmental Inc. (Minnow). 2003. Biological Assessment Town Creek Upstream of Highway 655.

Porcupine Gold Mines (PGM). 2009. Hollinger Tailings Management Area Closure Plan Amendment. April 2009.

Senes Consultants Ltd. (Senes). 2007. Closure Plan for the Hollinger (Gillies Lake) Tailings Area. January 2007.

HOW TO ASSESS THE BIO IN BIOPROCESSES? CONTRIBUTIONS OF MICROBIAL COMMUNITY PROFILING TO MINE PERMITTING, OPERATIONS, AND RECLAMATION ACTIVITIES FROM THE PAST 5 YEARS.

Vanessa Friesen[1], Jordyn Bergsveinson[1], and Monique Haakensen[1]
[1] Contango Strategies, Saskatoon, Saskatchewan, S7N 4N1

Key Words: Innovation, environment, evidence-based decision making, genomics, bioremediation, passive water treatment, biofouling, MCP testing

Microbes are the driving force in many processes, acting as catalysts to facilitate biogeochemical reactions that influence mining operations and remediation efforts. Despite this influence, microbes have often been overlooked in mining-associated processes. Historically, this has largely been due to the inability to effectively test and interpret mining-associated microbiological samples and data in a way that is useful to inform decisions. However, technology has advanced dramatically over the past 5 years, and genetic (genomic) and growth-based microbial community profile (MCP) testing is now being applied to diverse mining processes and reclamation activities to inform and de-risk decision making (Fig. 1).

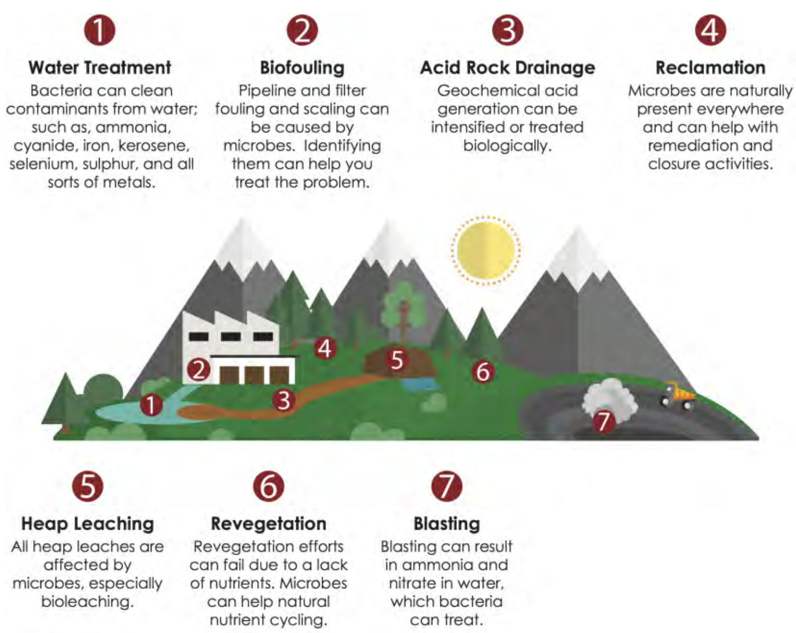


Fig. 1 – Examples of how microbes impact the mining sector

Microbial community profiling is now being used globally to aid mines with remediation, water treatment, consultation on acid rock drainage, and process optimization. The current state of technology is presented here as mini case studies from the past 5 years with example applications of MCP testing in mine settings. Examples range from permitting, design and optimization of reclamation activities, and treatment and prevention of fouling in operational processes.

Case study 1: Treatment and prevention of fouling in operational processes

Biofouling can occur when microorganisms accumulate on surfaces, blocking pipes and filters, and preventing process flow. To effectively treat and prevent biofouling, the microorganisms responsible for the build up need to be identified to determine treatment and mitigation strategies, as well as putative sources of the problematic organisms. Historically, identification may have been overlooked, and the problem would be addressed as a black box, or sometimes microscopic or metabolic testing may be attempted, which are unable to provide the resolution needed to identify the culprit organisms.

The selected case study is an example of identifying biofilms that were building up on distribution and picket fences in settlers used to separate copper from a pregnant leach solution. The build-up was problematic for process flow and required routine removal. Identification of the problematic organism through genetic MCP testing enabled recommendations for treatment and future prevention, as well as identified putative sources of the organism in the process.

Case study 2: Selection of plants for ammonia treatment during operations

A constructed wetland was being designed for operational treatment of ammonia (from blasting residues) for an underground mine in Canada. Ammonia oxidation is a process performed by bacteria in the wetland, but these bacteria are often found associated with the roots of plants. In order to effectively select the best plant sources for the wetland (i.e., those which would bring along beneficial bacteria with them), the microbial populations hosted by the roots of native *Typha* and *Phragmites* from different borrow sources near the construction site were compared with MCP testing.

This testing allowed for the selection of *Typha* from two different borrow sources for use in the constructed treatment wetland. Two sources were selected because the types of ammonia-oxidizing organisms were different in these borrow locations, and therefore inclusion of both increases the diversity of organisms that can perform the needed function in the wetland. In the context of treatment wetlands, the diversity of microbes present that can perform a desired function (in this case, ammonia oxidation/nitrification) can be regarded as a measure of robustness to treat the water under a wider range of conditions and changes in water chemistry.

Case study 3: Assessment of passive water treatment potential and testing through freeze-thaw for permitting

A mine project in the Northwest Territories is predicted to have seepage in closure that requires treatment. To assist in permitting, a site assessment was performed to delineate attenuation of constituents of concern that are occurring in a natural wetland on site, to inform site-specific design and testing of constructed treatment wetlands for closure.

Microbial communities were assessed along a watershed that receives seeps that are naturally high in arsenic, alongside other analyses for physicochemical parameters at the site. Using this information from the site assessment, a passive treatment wetland was designed and constructed at pilot scale to mimic natural conditions at the site that were found to improve arsenic treatment. The pilot-scale design successfully achieved targeted reducing and oxidizing conditions (in treatment cells designed for these respective conditions as part of a treatment train), and demonstrated the stability of key microbes through a freeze thaw cycle (Fig. 2). With historical tools, the microbial aspects of natural attenuation at the site and subsequent pilot-scale designs and testing through a freeze thaw cycle would have been poorly defined, leading to the inability to predict robustness and optimize performance.

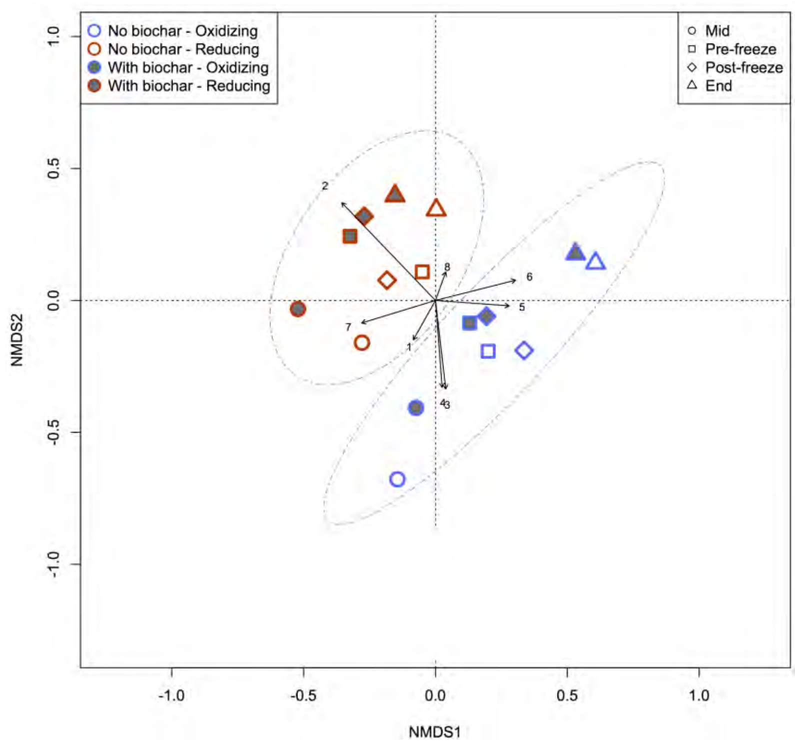


Fig. 2 – Microbial populations in pilot-scale treatment wetlands through freeze-thaw testing

Multivariate statistical analysis can be used to suggest relationships between microbial populations in different wetland designs over time and through a freeze-thaw cycle. Oxidizing and reducing conditions targeted in the treatment wetlands are outlined with dotted ellipses.

UNESCO GLOBAL GEOPARK NETWORK (GGN)… ABSTRACT

Beginning in 1998, the UNESCO Global Geopark initiative now extends to 33 countries and features 120 “Parks”. Canada is a signatory member of UNESCO. The federal government has assigned the task of overseeing Geopark development to the Canadian Federation of Earth Sciences.

The Geoparks are becoming very popular worldwide, due to their combination of conservation, sustainable development, and community involvement. The single constant is the requirement for each park to exhibit “significant geological heritage”. As such, many of the Parks are developed in former (and active) mining regions.

In Canada there are currently 2 recognized Global Geoparks. They are “Stonehammer”, located at St. John NB, where the focus is on plate tectonics, fossils, glaciation, geological education, and adventure tours. The second site is “Tumbler Ridge”, in northeastern BC, within an active coal mining region, where the focus includes dinosaur tracks, waterfalls and mountain hiking. A number of other mining communities are currently being considered, such as the Fort Mc Murray area (Wood Buffalo), Sudbury, and the Temiskaming Rift Valley of Ontario and Quebec.

The GGN has a particular interest in developing (or maintaining) a sustainable economy that is based on geotourism in regions that have a strong mining heritage. This is featured most prominently in the “Tuscan Mining Park”, in Italy. Centuries of active mining ended in the 1990’s, but educational geological and mining tours of the former mines and smelters have replaced some of the lost jobs.

CLRA members have always worked toward the protection and redevelopment of the natural environment following mining activity. Members should now consider redirecting part of their activities to the redevelopment of a **sustainable economy** that is based on the geological assets of former Canadian mining regions. To this end, a link with the GGN could prove effective.

Graham Gambles, President, RockWalk Park Inc., Box 586, Haileybury, ON, P0J 1K0, 705-672-3105, gamblesgraham@yahoo.ca

INTRODUCING THE UNESCO GLOBAL GEOPARK NETWORK

Graham J. Gambles
RockWalk Park Inc., Temiskaming Shores, ON, P0J 1K0

Key Words : aspiring, Canadian, criteria, geology, geoheritage, geotourism, agritourism, sustainable, mining, reclamation

“UNESCO” is the acronym for the “United Nations Educational, Science, and Cultural Organization”. Canada is a signatory member of this international body and actively participates in UNESCO programs.

Established in 1998, the UNESCO Global Geopark Network (GGN) now involves 120 Geoparks spread out over 33 countries. As of June, 2016, there are two full-fledged UNESCO Global Geoparks in the “Canadian Geoparks Network” (CGN) plus an additional ten “Aspiring” Canadian Geoparks that are in various stages of development.

Our lives are shaped by geology and landscape. Collective decisions on where we settle, the crops we grow, the water resources that we require, the climate itself, all resonate back to geological opportunities…and limitations. Those of us who are directly linked to the extractive resource industry are well aware that what we mine, the energy resources that we chose to use, and the natural hazards involved in procuring these resources, are directly linked to geology.

But what is the geological awareness level of the general population (in any country)? Are they knowledgeable of the implications of geology in their lives? UNESCO advisory staff looked at this issue in the closing years of the last century and came to the conclusion that no, the essential components of “Earth Science” understanding were not sufficiently present in the public domain on a global level.

Previous experience with “The Man and the Biosphere Programme”, created in the early 1970’s by UNESCO, was highly successful. Hundreds of sites in more than a hundred countries became the “World Network of Biosphere Reserves”. These closely managed facilities are places for learning about sustainable development, as it relates to the human connection to biology. Similarly, the UNESCO “World Heritage Sites” initiative promotes the conservation of natural and cultural sites of outstanding universal value.

Could a new program be developed for the Earth Sciences as a whole, through actively engaging with local communities, with the aim of educating the global population about global geodiversity? The vision included the concept that Geoparks would be of significant size, and would have the potential to include both Biosphere Reserves and World Heritage Sites within the parks boundaries.

THE ESTABLISHED CRITERIA

There are four basic features that are fundamental for the development of a Geopark. These standards must be met and confirmed by representatives of the country that nominates a specific park to the Global Geopark Network.

- GEOLOGICAL HERITAGE of international value, based on peer-reviewed, published research on the areas geological sites, must be proven to exist.
- MANAGEMENT by a body having legal existence that is recognized under national legislation must exist. It will be represented by all relevant local and regional authorities and principal private partners. The organization must develop a comprehensive management plan to protect the landscape and conserve cultural identity.
- VISIBILITY to visitors, as well as the local population must exist, in order to promote sustainable local economic development, primarily through Geotourism and Agritourism. It is important to establish a corporate identity.
- NETWORKING, both on the National and International scale is obligatory in order to be a member of the UNESCO Global Geopark Network (GGN). By working together across borders, the GGN contributes to increasing understanding among different communities and enhances peace-building processes.

  Criteria satisfaction is evaluated during biennial meetings of the International Conference on Geoparks, which also is in charge of undertaking periodic reviews.

TOP 10 FOCUS AREAS OF UNESCO GLOBAL GEOPARKS

Natural Resources

UNESCO Global Geoparks inform people about the sustainable use and need for natural resources, whether they are mined, quarried, or harnessed from the surrounding environment, while at the same time promoting respect for the surrounding environment and the integrity of the landscape.

Geological Hazards

The promotion of awareness for local geological hazards, including volcanoes, earthquakes and tsunamis, may help prepare populations by developing disaster mitigation strategies. These efforts build important capacity and contribute to the development of more resilient communities that can effectively respond to disaster.

Climate Change

Geoparks may hold the history of ancient climate change in the geological record, and the facilitors can be the educators on current climate change. The Geoparks can influence best practices for the population by utilising renewable energy and employing the best standards of “green tourism”. Such community and educational activities or projects are important to raise awareness on the potential impact of local climate change

Education

All UNESCO Global Geoparks must develop and operate educational activities for all ages in order to spread awareness of our geoheritage and its links to other aspects of our natural, cultural, and intangible heritages. Education can be offered in both formal and informal packages, and can be designed in a format that will allow both adults and retired people the opportunity to teach others.

Science

Geoparks are encouraged to work with academic institutions to engage in active scientific research in the Earth Sciences, and other disciplines as may be appropriate, to advance our knowledge about the earth and its processes. In addition, a Geopark must take great care not to alienate the public from science, and must avoid the use of technical jargon in programs aimed at attracting the interest of the general public.

Culture

UNESCO Global Geoparks are fundamentally about people, and about exploring and celebrating the links between communities, our practices, and the Earth. Many Geoparks have strong links to the arts communities, which allow the synergy released by the constructive arts and science combination, to become active in a unique manner. Mythology, folklore, building traditions and farm practices all come into play.

Women

The Global Geopark Network has a strong emphasis on the empowering of women through focussed education programmes and through the development of women’s cooperatives. Geoparks are a platform for the development of sustainable local cottage industries and craft products. Women often operate accommodation services for visitors.

Sustainable Development

Despite the presence of important geological heritage, it is equally important that the area has a plan for the sustainable development of the people who live there. This may take the form of sustainable tourism activities, or sustainable agriculture and food tourism. UNESCO Global Geopark status does not imply restrictions on ANY economic activity that complies with indigenous, local, regional, and/or national legislation.

Local and Indigenous Knowledge

Geoparks actively involve local and indigenous peoples, preserving and celebrating their culture. Local and indigenous knowledge, practices, and management systems, alongside science, are included in the planning and management of the Global Geopark.

Geoconservation

UNESCO Global Geoparks are areas that use the concept of sustainability, value the heritage of “Mother Earth”, and recognize the need to protect it. The defining geological sites in UNESCO Global Geoparks are protected by indigenous, local, regional, and/or national law and appropriate management authorities and agencies that will provide the necessary monitoring and maintenance of these sites. However, Global Geoparks do not interfere with the removal of geological material from licensed mines, quarries, etc. that are subject to regulation under national and/or international legislation.

UNESCO GLOBAL GEOPARKS AND MINING

Most member Geoparks have incredible geologic landscapes to enjoy, but they do not have an active history in quarrying or mineral extraction. However, the small subset that does represent the international mining industry provides a glimpse into the potential of incorporating mining heritage into the developing field of sustainable Geotourism. There are clear indications that the UNESCO Global Geopark Network is concerned about the loss of employment opportunities and cultural values when a long term mining region is permanently closed. However, there is economic value in mining heritage.

Could the GGN be suggesting that mine reclamation specialists need to do more than secure a minesite from pollutant discharge while re-establishing traditional vegetative cover? Are they saying that mine reclamation includes the re-establishment of meaningful, sustainable economic opportunities among the long term residents in a traditional mining town? Does the UNESCO Global Geopark initiative indicate that there are values to be consciously preserved in the infrastructure of the mining complex, as well as in the infrastructure of the surrounding communities themselves?

The answer may be in the evaluation of mining and quarry areas that are part and parcel of a current regional Geopark. Here is a list of Global Geoparks that involve various levels of mining history and should be researched (on the internet) by individuals involved in mineland reclamation.

Tuscan Mining Park UNESCO Global Geopark (Italy)

Geological and Mining Park of Sardinia UNESCO Global Geopark (Italy)

Troodos UNESCO global Geopark (Cyprus)

Swabian Albs UNESCO Global Geopark (Germany)

Bakony-Balaton UNESCO Global Geopark (Hungary)

Ore of the Alps UNESCO Global Geopark (Austria)

Terras de Cavaleiros UNESCO Global Geopark (Portugal)

Copper Coast UNESCO Global Geopark (Ireland)

Idrija UNESCO Global Geopark (Slovenia)

Karawanke/Karawanken UNESCO Global Geopark (Slovenia/Austria)

Yanqing UNESCO Global Geopark (China)

Tumbler Ridge UNESCO Global Geopark (Canada)

PRESENT AND FUTURE UNESCO GLOBAL GEOPARKS IN CANADA

To assist in Geopark development, Canada initiated the Canadian National Committee for Geoparks (CNGG) in 2009. The first Geopark to achieve international recognition was Stonehammer, in 2011, located at St. John, New Brunswick. It was followed by Tumbler Ridge in 2014, located in north-eastern British Columbia.

Membership of the CNGG comes from all parts of Canada. The CNGG assists Geopark proponents in the development of plans, advising on the strengths and weaknesses of Aspiring Geoparks in Canada. Figure 1 outlines the Canadian Geopark Network as it was defined in early 2016, with 2 fully accredited UNESCO Global Geoparks and 10 Aspiring Canadian Geoparks.

The UNESCO Global Geopark program is a long term venture. As only 2 Geoparks in any one nation can achieve international recognition in any given year, it will be a long time before there are representative sites in every Province and Territory. This is a primary reason why Canada should look at developing Geoparks that are much larger in size than what is normal in most parts of the world.

Note that this program is dependent on individual citizens, community organizations, not-for-profit agencies, and private companies to take the initiative and develop Geopark projects. These are “grassroots” projects. They do not originate in the offices of Federal, Provincial, or Territorial governments, although the support and cooperation of these organizations is essential in the long term. All Provincial governments have staff who are familiar with the program.

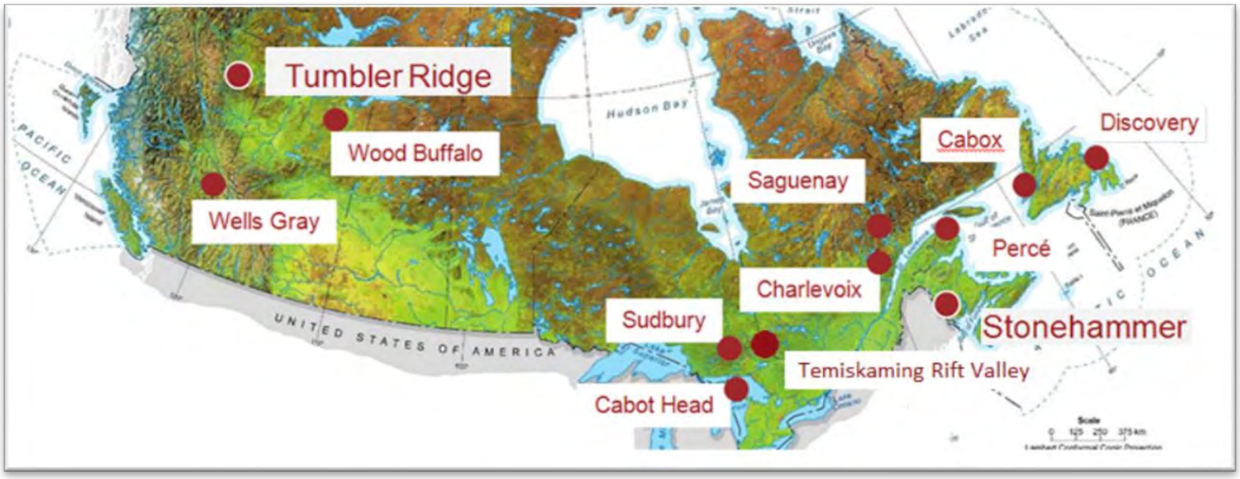


Figure 1: UNESCO Global Geoparks and Aspiring Canadian Geoparks

THE OPPORTUNITY FOR CLRA

When the Canadian Land Reclamation Association originated in the mid 1970's, there was a vision that the organization would bring all elements of land reclamation science together in one professional network. However, in reality, the science had not progressed much further than the focus of the original Ontario Cover Crop Committee at the University of Guelph.

Science is progressive and reclamation professionals now evaluate a whole host of new technologies in an effort to build the most suitable reclamation plan for each individual project. Most research is done with the goal of enhancing the biological opportunities for the renewal of the derelict site. However, UNESCO, through the Global Geopark initiative, is suggesting that there is another component in (mine) land reclamation that needs to be addressed. That is the requirement of developing sustainable human opportunities in the shadow of abandoned mine operations. This is a cultural need of humans, who are also part of the biosphere.

A century ago, mine operators abandoned projects without thought, when ore turned to waste. Environmental consideration had not yet evolved. Workers were left to fend for themselves as best they could. Today, most governments encourage the eradication of all traces of a mines existence, in addition to the necessary protection of the biosphere from pollutants. Workers are financially encouraged to retrain for new careers and move.

But through it all, the human community that is left behind remains culturally linked to the reason for its original existence. They are mining towns with a proud history, inhabited with the offspring of generations of miners. To destroy all evidence of their cultural heritage is an affront to those who stay and try to achieve a sustainable way of life in their home town, in their geoheritage. UNESCO suggests this idea to be valid.

In Cobalt, Agnico Eagle recognized that there is a human need for this community to maintain its mining heritage. As part of its corporate commitment to the community, the company representatives are expanding their reclamation planning to allow for geotourism opportunities. As such, there is a direct link being created between Cobalt and the historic mining communities of Europe, who are finding a new future as part of the UNESCO Global Geopark Network.

Cobalt will be a core community in Temiskaming Rift Valley Aspiring Canadian Geopark.

CLRA members and the organization itself have the opportunity to evolve over the next few years and align reclamation activities more closely to the human cultural needs in former mining communities. The public has invested heavily in the infrastructure that supported the mines. Reclamationists are well positioned to broaden their scope to include a sustainable future for mine communities, as an integral component of a successful land rehabilitation project.

CAN FAST-GROWING PLANTATIONS FACILITATE FOREST TREE RECRUITMENT WHILE LIMITING SOIL EROSION IN MINE WASTE ROCK SLOPES?

Marie Guittonny-Larchevêque [1], Hugo Bouchard [1], Arnaud Remaury [1], Suzanne Brais [2], Jane Rickson [3]

[1] Institut de recherche en mines et en environnement – Université du Québec en Abitibi-Témiscamingue, Rouyn-Noranda, QC, J9X 5E4

[2] Institut de recherche sur les forêts – Université du Québec en Abitibi-Témiscamingue, Rouyn-Noranda, QC, J9X 5E4

[3] Cranfield Soil and Agrifood Institute – Cranfield University, UK, MK43 0AL

Key Words: forest reclamation, hybrid poplar, mine closure, mine revegetation, root development, soil erosion, tree recruitment

Introduction

In forested regions, mine wastes are poor quality substrates for tree establishment, especially waste rocks. Waste rocks are one of the two main solid mine wastes produced by mine sites, along with mill tailings. They come from the rock materials surrounding the ore, which are extracted by explosion and stockpiled on tens of meters (Brooks, 1990). Tree plantation after soil layering could catalyze plant succession on waste rock slopes and accelerate the conversion of revegetated waste rocks into forest, restoring the traditional use of closed mine sites. But in waste rock slopes, soil erosion may limit revegetation success. Herbaceous species are seeded in waste rock slopes to rapidly stabilize soil surface and to limit soil erosion (Helm, 1995). However, used herbaceous species often compete for water, nutrients, and light with planted trees (Rizza et al., 2007; Franklin et al., 2012), and are low quality nurse plants to allow natural colonization of revegetated slopes by forest trees. On the contrary, the use of fast-growing tree plantations on waste rock slopes could facilitate forest trees recruitment by changing the undergrowth microclimatic conditions while limiting soil erosion. On one hand, fast-growing trees like hybrid poplar develop an extensive root system which rapidly colonizes the available soil volume (Douglas et al., 2010; Wilkinson, 1999), as well as a canopy which protects the soil from rain drop impacts. On the other hand, the facilitation theoretical model (Connell and Slatyer, 1977) tells us that pioneer tree species like poplar, which are able to colonize disturbed sites, prepare the environment for tree species which come later in forest succession.

Thus fast-growing poplars may be used to quickly stabilize the soil against erosion while facilitating forest recolonization. However, the plantation design that should be used to achieve both objectives remains unknown. To overcome this knowledge gap, a hybrid poplar plantation was established in 2013 on waste rock slopes of the Canadian Malartic gold mine in Quebec. The influence of plantation design in terms of tree spacing and combination with hydroseeding of herbaceous species was evaluated on: 1) soil erosion, vegetation and litter cover, and root development in superficial soil, and 2) microenvironmental conditions controlling seed germination, seedling development and survival of forest trees.

Experimental setting

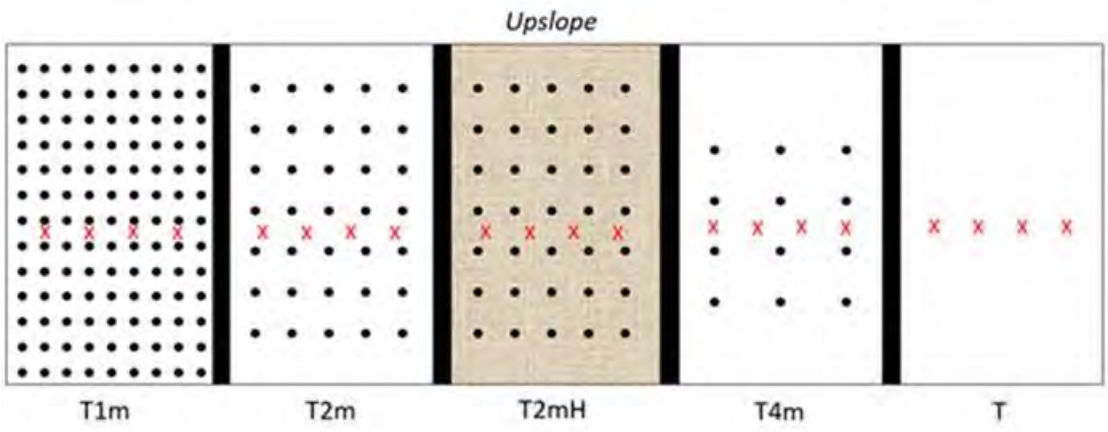


Figure 1. Representation of one repetition block (among three) of the experimental plantation established in 2013: black dot represent trees. Five treatments were tested: three tree spacing (T1m: 1x1m, T2m: 2x2m, T4m: 4x4m), hydroseeding of agronomic herbaceous species with planted trees at 2x2m (T2mH), or soil cover without revegetation (T: no trees, no hydroseeding).

Results

Along the three years after planting, no treatment effect was found on planted tree survival, which remained close to 100%. Soil loss was the lowest in plots with planted trees at 2x2m and hydroseeding (T2mH) while it was the greatest in control plots without planted trees and hydroseeding (T, Figure 2). Plots planted with hybrid poplars without hydroseeding showed intermediate soil losses. Undergrowth cover and herbaceous root length density were greater in hydroseeded plots compared to other plots, but the difference attenuated with time thanks to natural vegetation colonization of plots without hydroseeding. Tree canopy cover and root length density were greater in high density plantations (1x1m) compared to the other spacing treatments (2x2, and 4x4m).

Plots with intermediate tree density (2x2m) showed the greatest number of naturally colonizing seedlings of poplars and willows. Germination rates of tree species coming later in the boreal forest succession, i.e. *Picea glauca* [Moench] Voss and *Abies balsamea* [L.] Miller, increased in 1x1m planted plots compared to other treatments concomitantly to an increase in the volumetric water content of superficial soil with planting density (Figure 3). Finally, the hydroseeded treatment provided the lowest abundance of pioneer trees and balsam fir seedlings.

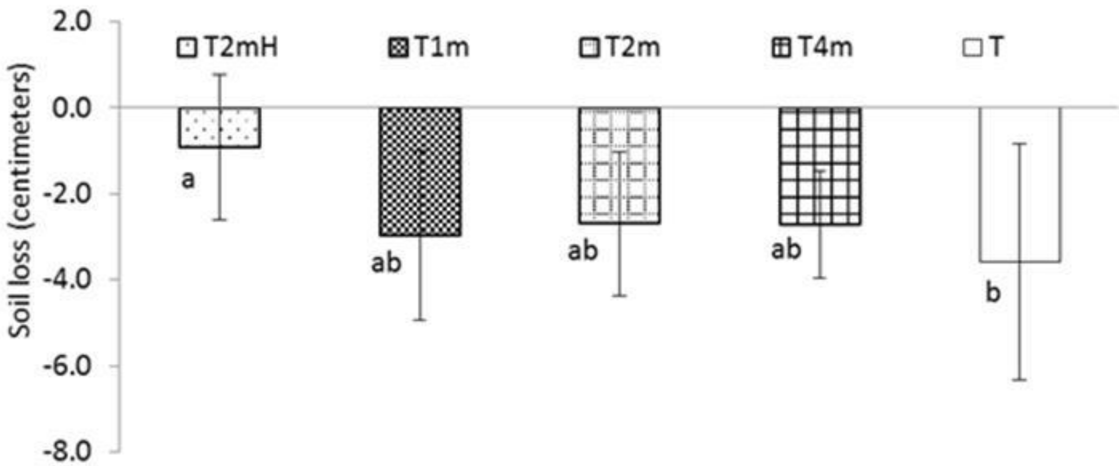


Figure 2. Mean soil loss (cm) during spring snow melt in the two-year-old plantation (2014) among the five tested treatments: three tree spacing (T1m: 1x1m, T2m: 2x2m, T4m: 4x4m), hydroseeding of agronomic herbaceous species with planted trees at 2x2m (T2mH), or control with soil cover but no revegetation (T). (N=12). Bars denote SE. Means that do not differ at the 0.05 level are noted with the same letter (a < b)

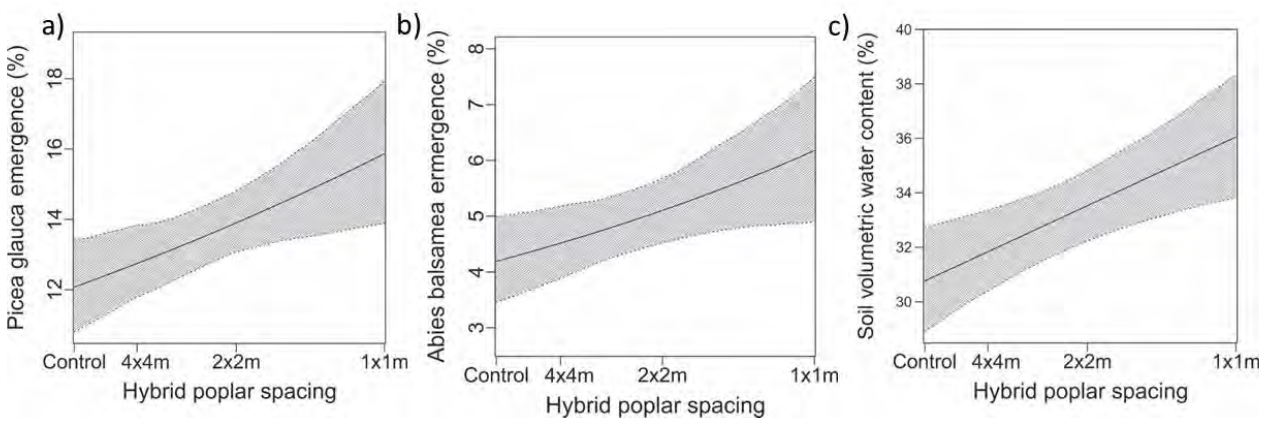


Figure 3. Seedling emergence rates of (a) *Picea glauca* and (b) *Abies balsamea* in relation to hybrid poplar spacing after spring seeding in the 3-year-old plantation (2015), varying in accordance with the increase in (c) soil volumetric water contents along the same gradient. Results presented in (a) and (b) summarize total summer emergence, whereas results in (c) are based on the average value measured from mid-May to mid-June, when the largest proportion of emergences were recorded for both species. Fitted values and 90% confidence intervals were estimated by Monte Carlo simulations from generalized and linear mixed-effects models.

Conclusion

At the short term, hydroseeding of herbaceous species remained the most efficient method to limit soil erosion in waste rock slopes thanks to a greater aerial cover and root development of the undergrowth vegetation as soon as the first year after planting. On the other hand, planted tree growth, as well as the success of forest tree recruitment, were decreased in plots hydroseeded with herbaceous species.

Fast-growing plantations also decreased soil erosion compared to control plots without trees and without hydroseeding. Their positive effect improved the second year after planting according to tree growth and undergrowth vegetation development.

Finally, the treatment with high planting density (1x1m) presented greater soil water volumetric content the third year after planting, which was favorable to *Picea glauca* and *Abies balsamea* recruitment compared to other treatments. Moreover, the same treatment with high planting density could better protect the soil from erosion from the third year after planting onwards because it presented a greater tree canopy cover and a greater root development of trees compared to lower planting densities.

Acknowledgments

This research was funded by the Canadian Malartic Partnership and NSERC (the Natural Sciences and Engineering Research Council of Canada) through a Collaborative Research and Development (CRD) grant to M. Guittonny-Larchevêque.

Literature cited

Brooks, B.W., Peters, T.H., Winch, J.E. 1989. Manual of methods used in the revegetation of reactive sulfide tailings basins. Mine Environment Neutral Drainage (MEND) Program. Project 2.24.1. 102p.

Connell, J.H., Slatyer, R.O. 1977. Mechanisms of succession in natural communities and their role in community stability and organization. American naturalist: 1119-1144.

Douglas, G.B., McIvor, I.R., Potter, J.F., Foote, L.G. 2010. Root distribution of poplar at varying densities on pastoral hill country. Plant Soil 333: 147-161.

Helm, D. 1995. Native grass cultivars for multiple revegetation goals on a proposed mine site in southcentral Alaska. Restor. Ecol. 3: 111-122.

Franklin, J.A., Zipper, C.E., Burger, J.A., Skousen, J.G., Jacobs, D.F. 2012. Influence of herbaceous ground cover on forest restoration of eastern US coal surface mines. New Forest 43: 905-924.

Rizza, J., Franklin, J., Buckley, D. 2007. The influence of different ground cover treatments on the growth and survival of tree seedlings on remined sites in eastern Tennessee. National Meeting of the American Society of Mining and Reclamation, Gillette, WY, 30 Years of SMCRA and Beyond, 2-7 Juin 2007. R.I. Barnhisel (Ed.). Published by ASMR, Lexington, KY: 663-677.

Wilkinson, A.G. 1999. Poplars and willows for soil erosion control in New Zealand. Biomass and Bioenergy 16: 263-274.

CONSTRUCTED WETLAND DESIGN AND OPTIMIZATION FOR METAL AND METALLOID TREATMENT AT THE MINTO MINE, IN THE YUKON, CANADA

Monique Haakensen[1], Vanessa Friesen[1], Ryan Herbert[2]
[1] Contango Strategies, Saskatoon, Saskatchewan, S7N 4N1
[2] Minto Mine, Capstone Mining Corp, Whitehorse, Yukon, Y1A 2V3

Key Words: Passive water treatment, copper, selenium, cadmium, mine closure, cold climate, treatment wetland, phased approach, on-site demonstration, biogeochemistry

Constructed wetland treatment systems (CWTSs) can treat various contaminants in water when designed and executed in a scientifically guided manner. A site-specific CWTS is being developed for Capstone Mining Corporation's Minto Mine (Yukon), using a scaled phased approach to allow for flexibility for modifications and optimisations along each step. The phases are: (1) site assessment and information gathering, (2) technology selection and conceptual design, (3) pilot-scale testing and optimisation (controlled environment), (4) onsite demonstration-scale confirmation and optimisation and (5) full-scale implementation. The first three phases have been completed successfully, and a year of testing performed on site for Phase 4.

Innovations in the design development included application of genetic and growth-based microbial community profiling (MCP testing) to guide system design in a site-specific context. Pilot-scale testing allowed for selection of the optimal design from several options and for testing of different predicted closure water chemistries (Fig. 1), including concentrations beyond the most elevated predictions, including associated near-term and long-term scenarios (with and without nitrate, which could be present from blasting residues and affect selenium treatment). The pilot-scale CWTS confirmed plant amenability to transplantation, and the design selected for further testing achieved 92% removal of copper (mean influent 146 µg/L, outflow 11.3 µg/L) and 41% removal of selenium (mean influent 10.2 µg/L, outflow 6 µg/L; Fig. 1).

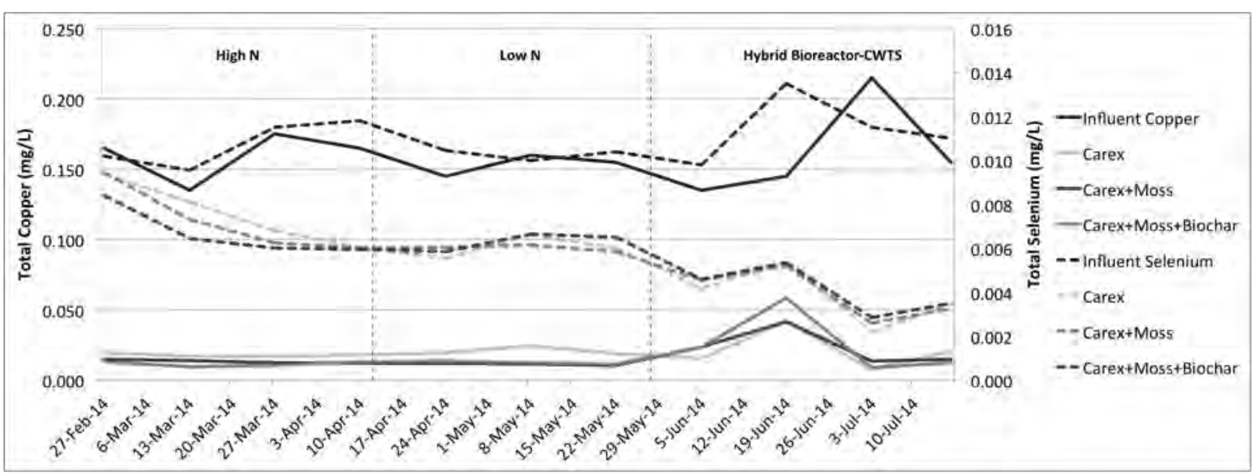
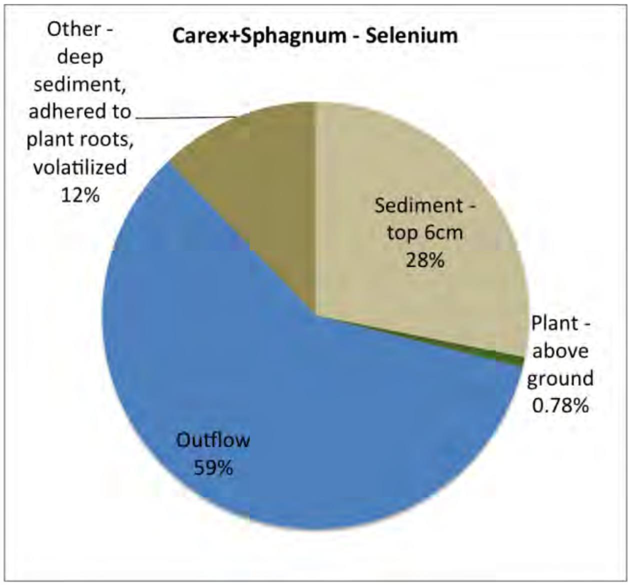


Fig. 1 – Performance of selenium and copper treatment over time at pilot-scale

A mass-balance of the pilot-scale systems confirmed that elements were sequestered to sediments, with less than 0.5% of the copper and 2% of the selenium transferred to the plant leaves (Fig. 2). Removal rate coefficients were developed from the pilot-scale wetlands to allow for appropriate modelling and sizing of on-site systems. Further, MCP testing and physicochemical parameters (that explain the performance and are used as decision variables) outlined expected system maturation timelines through pilot-scale testing and setting subsequent expectations for on-site implementation.



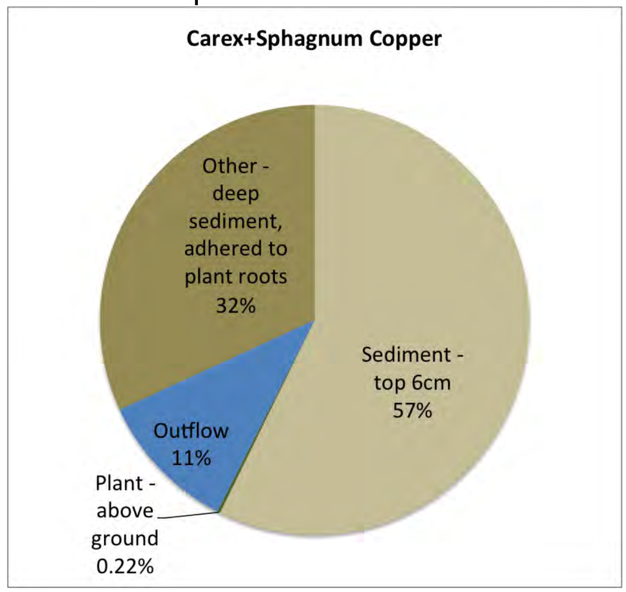


Fig. 2 – Mass balance of selenium and copper in pilot-scale treatment wetland

The optimised design was built at demonstration-scale (Phase 4) on-site in 2014 (Fig. 3), and has been operating since, following a path of maturation as predicted by the pilot-scale systems (Fig. 4). This maturation will be discussed in this presentation in the context of the previous 3 Phases. Highlights include a better than anticipated plant establishment, and initial performance during the commissioning period.

Fig. 4 – Demonstration-scale treatment wetland on site, 2015.

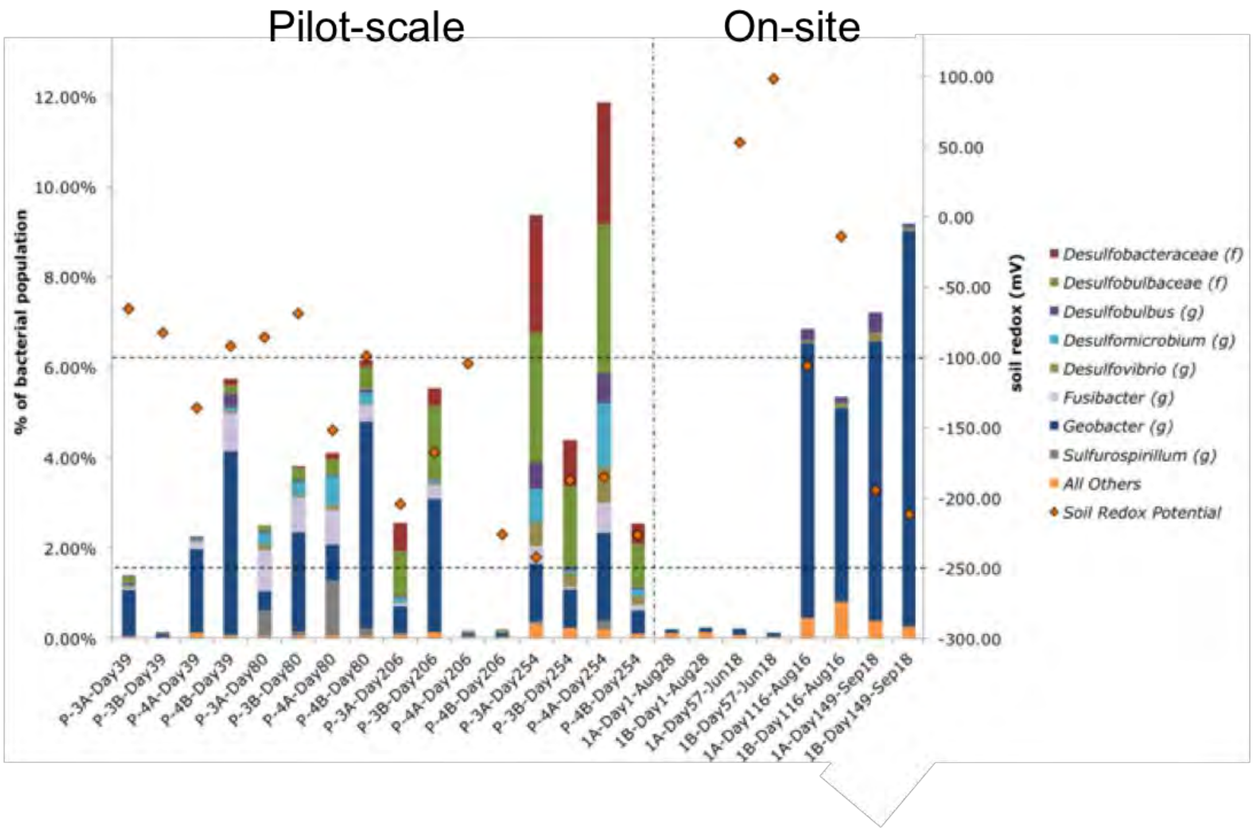


Fig. 5 – Comparison of soil redox (right axis, diamonds) and sulphide-producing microbial population (left axis; bar chart) for pilot-scale (off site) and demonstration-scale (on site) systems. Horizontal dashed lines indicate targeted soil redox range, as suggested by oxidation-reduction chemistry and refined by pilot-scale testing.

Effect of Liming on Ni Bioavailability and Toxicity to Oat and Soybean Grown in Field Soils Containing Aged Emissions from a Ni Refinery

S. Cioccio[1], Y. Gopalapillai[2], T. Dan[3] and B. Hale[2*]
[1]Golder, Mississauga, ON
[2]School of Environmental Sciences, University of Guelph, Guelph, ON
[3]Stantec, Guelph, ON

Keywords: soil pH, dolomitic lime

1 Introduction

Mitigating toxicity of soil Ni through liming might manifest itself through lower bioavailability of soil Ni, thus lower tissue concentrations of Ni. Bioavailability of dissolved metals is reduced as soil pH increases by reducing the free metal ion in solution, as competition with H^+ for negatively charged sites on soil solids decreases [1]. Plant uptake of dissolved Ni may also increase with soil pH above 5, as competition between protons and metal cations for binding sites on the biotic ligand also decreases [2,3]. However, the net effect is often a decrease in uptake. Particularly since Ni solubility also decreases with the increase in pH, and specific to Ni, addition of dolomitic lime could increase the pore water concentration of Mg^{2+}, which has been shown to reduce Ni toxicity to soil-dwelling organisms by competing with Ni for the biotic ligand [2,4]. Finally, tissue concentration of an essential trace element, such as Ni, could be somewhat influenced by the organism's tendency towards homeostasis [5], which can confound the relationship between estimated metal bioavailability and the observed toxicity. The overall objective of the present work was to generate higher-tier in situ toxicity data for elevated Ni in soils, which would confirm the observations from the several pot studies of soils from this site, as well as the framework for predicting soil Ni toxicity identified by the European Union Risk Assessment Report (EU RAR) [6]. The present multi-year study of agronomic yield of field-grown oat and soybean, occurred in three adjacent fields that had received emissions from a Ni refinery for 66 years. The soil Ni concentration in the plots ranged between 1300 and 4900 mg/kg, and each field was amended with either 50, 10 or 0 tonnes/ha of crushed dolomitic limestone. Dolomitic lime is the preferred method for restoring agricultural soils, and it contains both Ca and Mg. The goal of the present study was to test the effect of liming on agronomic yield of soybean and oat, as well as its effect on the plant availability of soil Ni.

2 Materials and Methods

Three agricultural fields near the Ni refinery in Port Colborne, ON which are known to have elevated concentrations of Ni were amended with crushed dolomitic limestone at either 50, 10, or 0 t/ha in the summer prior to the first field trial. In the first year of the study, half of the plots in each of the three fields were planted with oat (*Avena sativa* var. Rigodown), and the other half were planted with soybean (*Glycine max* var. OAC Bayfield). In the second year of the study, all plots were planted with soybean (OAC Bayfield) (as none of the soybean plots in the previous year produced useable data) and in the third year of the study, half of the plots in each field were planted with oat and the other half of the plots were planted with soybean. Agronomic yield for each plot was determined when the plants were ready for commercial harvest. The absolute yield for each plot was also expressed relative to the county average yield for that crop year, which was 2.2 t/ha (oat; 2005), 3.1 t/ha (soybean; 2006), 2.4 t/ha (oat; 2007) and 2.2 t/ha (soybean; 2007).

3 Results and Discussion

For soybean, relative yield was higher at 10 and 50 t/ha added lime than at 0 t/h added lime ($p < 0.05$) (Figure 1a), whereas for oat, relative yield showed no significant difference among the liming treatments ($p = 0.09$) although the pattern of response to the three liming treatments was similar to that for soybean (Figure 1b). This comparison of main effect means should be carefully interpreted, as the mean (and median, data not shown) soil [Ni] at 10 t/ha added lime was greater than at both 0 t/ha and 50 t/ha added lime, so 'effect of lime' is confounded with exposure concentration. Despite this confounding factor, liming appeared to be effective at improving agronomic yield of soybean and oat grown in soils with [Ni] ranging to nearly 5000 mg/kg, although the optimal rate of lime addition could not be identified. Without the confounding of lime addition with exposure concentration, it is possible that agronomic yield of both species would have been clearly greatest at 10 t/ha, because it substantially reduces $CaCl_2$-extractable soil [Ni] and results in a soil pH closer to optimal for cultivating most grain crops (i.e. 6) than results from adding 50 t/ha lime to soil [7]. Pagani and Mallarino [8] demonstrated that yields of soybean and corn in uncontaminated soils were sometimes improved by liming, at rates of 9, 13.5 or 22.4 Mg/ha CCE equivalent, whether or not the lime source was calcitic, dolomitic, or pure $CaCO_3$. Soil pH was increased from about 5.7 to between 6.5 and 7.3, depending on the source of lime.

4 References

1. Allen HE. 2002. Bioavailability of metals in terrestrial ecosystems: importance of partitioning for bioavailability to invertebrates, microbes, and plants. SETAC Press, Pensacola, USA.
2. Lin Y, Di Toro DM, Allen HE. 2015. Development and validation of a terrestrial biotic ligand model for Ni toxicity to barley root elongation for non-calcareous soils. Environ. Pollut. 202:41–49.
3. Rooney CP, Zhao F-J, McGrath SP. 2007. Phytotoxicity of nickel in a range of European soils: Influence of soil properties, Ni solubility and speciation. Environ. Pollut. 145:596–605.
4. Owsianiak M, Rosenbaum RK, Huijbregts MAJ, Hauschild MZ. 2013. Addressing Geographic Variability in the Comparative Toxicity Potential of Copper and Nickel in Soils. Environ. Sci. Technol. 47:3241–3250.
5. Marschner P. 2012. Miner. Nutr. High. Plants Third Ed. Academic Press, San Diego, pp 645–651.
6. European Union. 2008. European Union Risk Assessment Report: Nickel and Nickel Compounds. European Communities, Luxembourg.

7. Castro GSA, Crusciol CAC. 2013. Effects of superficial liming and silicate application on soil fertility and crop yield under rotation. Geoderma. 195:234–242.
8. Pagani A, Mallarino AP. 2012. Soil pH and Crop Grain Yield as Affected by the Source and Rate of Lime. Soil Sci. Soc. Am. J. 76:1877–1886.

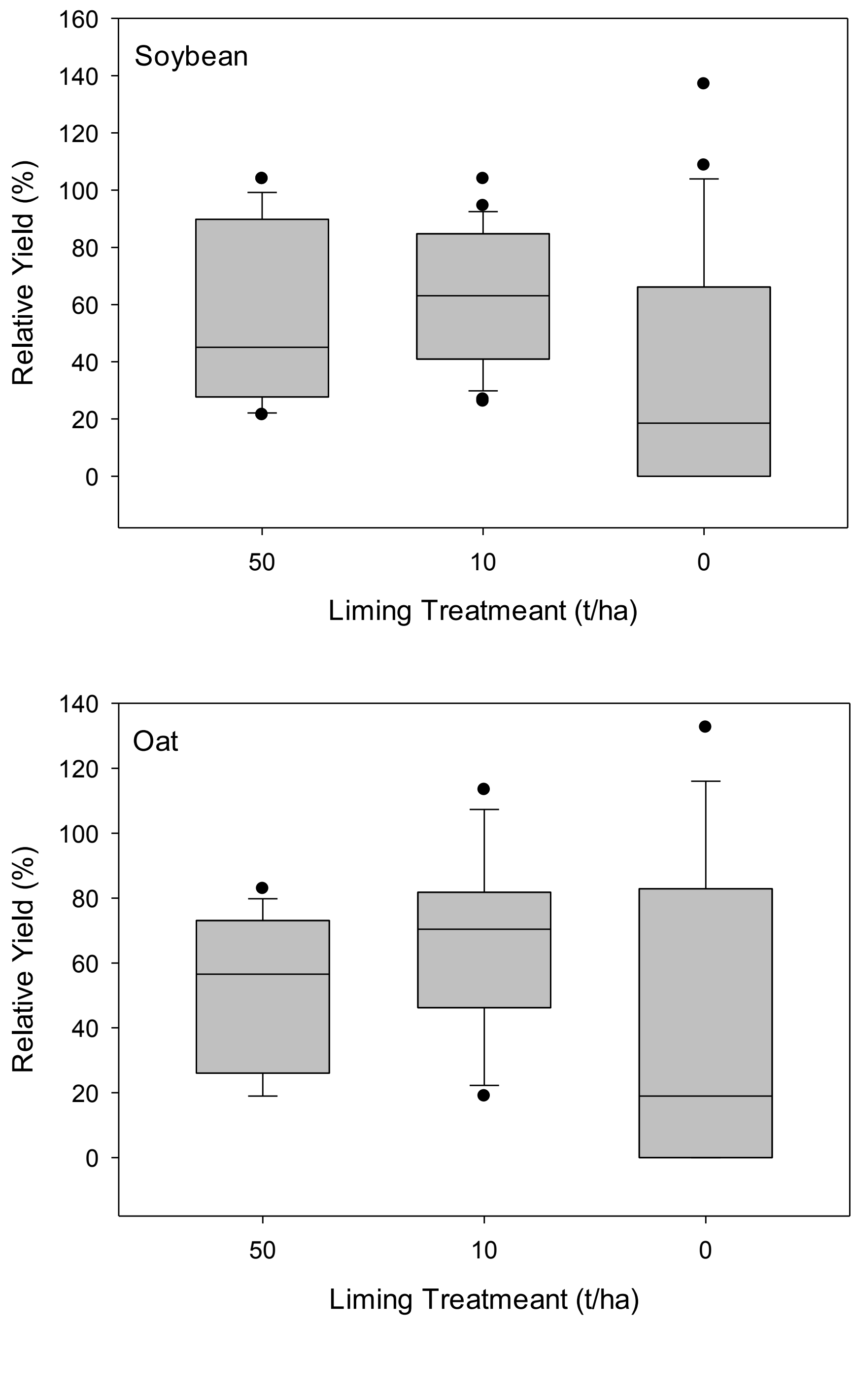


Figure 1: Agronomic yield of soybean and oat relative to county average values for the year, for each of three liming treatments.

TESTING SOIL MIXTURES FOR RECLAMATION OF DIAMOND MINE WASTES IN A SUBARCTIC REGION

Andrea Hanson[1, 3] and Daniel Campbell[2, 3]

[1]Department of Biology, Laurentian University, Sudbury, ON, P3E 2C6
[2]School of the Environment Laurentian University, Sudbury, ON, P3E 2C6
[3]Vale Living with Lakes Centre, Laurentian University, Sudbury, ON, P3E 2C6

Key Words: mine reclamation, Hudson Bay Lowlands, processed kimberlite, Ontario's Far North

Introduction

Subarctic regions are recognized as one of the largest remaining pristine landscapes on earth (Walker, 1997). Within these regions, strong connections exist between the physical, chemical, and biological components of each ecosystem (Walker, 1997), and are recognized as vulnerable due to their sensitivity to change and disturbance (IUCN, 1993). Subarctic and arctic regions have been subject to increasing pressure from human activities, including mining, which have led to progressive degradation of these pristine environments (Deshaise *et al*., 2009). Rehabilitation efforts in subarctic regions have been minimal due to the lack of proper guidelines. This is mainly due to a limited understanding of fundamental ecosystem processes that exist within subarctic regions (Deshaise *et al*., 2009). Reclamation strategies aim to improve conditions, such as developing functional soils, to create conditions suitable for vegetation establishment (Munro, 2006). Therefore, it is important to understand the soil properties and relationships that can influence reclamation within subarctic regions.

Our research focuses on the reclamation of diamond mine wastes in subarctic regions, specifically at the De Beers Victor Diamond Mine. The De Beers Victor Diamond Mine is located within the Hudson Bay Lowlands, within Canada's subarctic. Currently, it is the only active mining development within the Hudson Bay Lowlands. Mines that develop in northern regions can be a challenge to reclaim due to natural aspects of subarctic environments as well as the remote location of these mining developments. Due to their remote location, there is often a shortage of material that can be used to create cover soils during the process of mine reclamation.

The goal of our research is to examine how mines in northern regions, such as the De Beers Victor Diamond Mine, can use readily accessible material, including their mining by-products, to create cover soils for reclamation. To achieve this, various soil mixtures were created using local mineral and organic substrate materials at the Victor Mine, and their success and performance will be evaluated. The main objectives of this research are to (1) determine which mixture(s) of mineral substrates promote the most suitable conditions for the establishment of vegetation, and (2) determine how the quantity of organic matter influences vegetation establishment.

Methods

During the summers of 2013 and 2014, test mixtures were constructed on two areas of mine waste at the Victor Mine. In total, 48 test mixtures were constructed, measuring 5m x 5m. The raw materials used to create the soil mixtures included a silty loam marine overburden (OB), coarse and fine processed kimberlite (CPK and FPK respectively), and peat. Using these raw materials, four mineral mixtures were created, and tested with either 20% or 40% peat applications. The mineral mixture combinations include 100%OB, 50:25:25 OB:CPK:FPK , 50:50 OB:CPK, and 25:50:25 OB:CPK:FPK. During the summer of 2014 each test mixture was fertilized at a rate of 12.5g/m^2 (8-32-16 NPK), a local microbial inoculation mixture was applied, and were seeded with a variety of native nitrogen-fixing and non nitrogen-fixing plant species.

During the summer of 2015 various physical, chemical and biological parameters were examined to evaluate the success and performance of each mixture. The physical properties examined included root penetrability, texture, soil surface moisture, air temperature, bulk density, organic matter content, surface roughness and changes in moisture and temperature with depth. The chemical properties examined included soil pH, electrical conductivity, bioavailable nutrients, and C, N, and S content. The biological properties examined included microbial activity, aboveground biomass, and total plant cover and composition. These tests were also preformed on selected reference sites in order to compare the performance of our test mixtures in comparison to natural native soil conditions, a peat-OB mixture currently being used to reclaim a stockpile at the Victor Mine, and untreated waste rock piles at the Victor Mine.

Results

Results at this stage in our research show significant differences between several physical, chemical, and biological parameters of the mineral mixtures including soil bulk density, surface moisture, electrical conductivity, pH, and total plant cover. The greatest differences within these parameters were observed between the mineral mixtures containing 100% OB, and those containing high amounts of processed kimberlite. The mixtures containing 100% OB had greater surface moisture measurements, lower bulk densities, greater electrical conductivity measurements, lower pH measurements, and greater total plant cover scores during the first year of growth (Fig. 1) compared to the mixtures containing high amounts of processed kimberlite.

Significant similarities were also found between several physical, chemical and biological parameters of the test mixtures in comparison to native soil reference site conditions, in terms of bulk density, root penetrability resistance measurements, and electrical conductivity measurements. The test mixtures and peat-OB mixture had statistically significant similar estimated total plant cover scores.

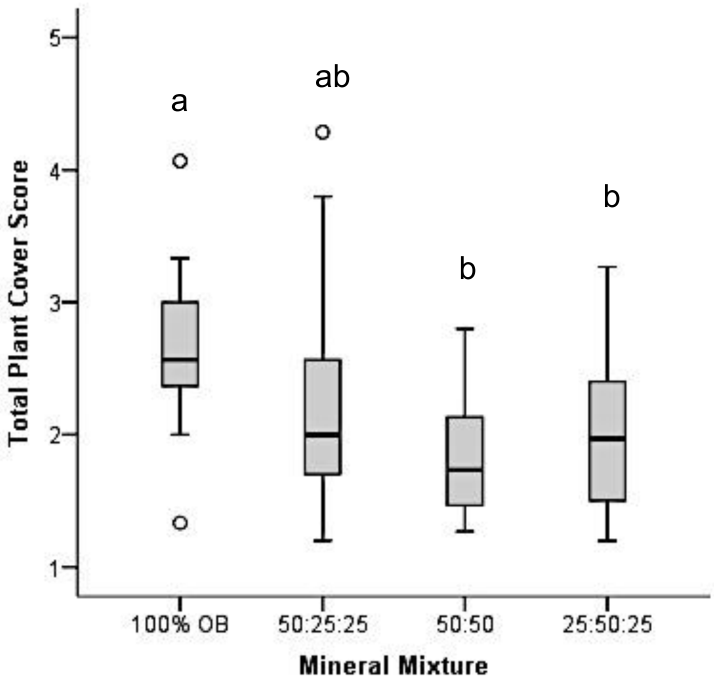


Fig. 1. Total plant cover variation observed on the mineral mixtures containing 20% and 40% peat in late August 2015 with post-hoc Tukey's test results (mineral mixture ratio OB:CPK:FPK, $P<0.05$). The scoring system used is as follows: 1=≤1%, 2=2-5%, 3=6-20%, 4=21-50%, 5=≥50% cover.

Conclusions

This research is still ongoing, however, we are observing that our test soil mixtures containing 100% OB were the most successful in terms of vegetation establishment during the first year of growth, and that our test mixtures resemble reference site conditions. During the summer of 2016 second year growth will be evaluated. This research will provide the De Beers Victor Diamond Mine with suggestions for a cover soil to use during mine reclamation. It will also provide insight into the challenges faced when restoring disturbed subarctic environments, and will be used to develop reclamation protocols for mine waste within the subarctic regions of Canada and others globally.

Acknowledgments

We would like to acknowledge the funding support from NSERC, the Goodman School of mines, and Laurentian University. Also, we would like to thank the CLRA for their support through the Tom Peter's Memorial Mine Reclamation Student Award. We would also like to thank the employees at the De Beers Victor Mine for their constant support of this research project through

2013 to present. Finally, we would like to thank the field assistants and other graduate students involved throughout the years, as well as the guidance and support from committee members as this research progresses.

Literature Cited

Deshaise, A., Boudreau, S., and Harper, K.A. 2009. Assisted Revegetation in a Subarctic Environment: Effects of Fertilization on the Performance of Three indigenous Plant Species. *Arctic, Antarctic and Alpine Research*. 41(4). 434-441.

IUCN. 1993. *Oil and Gas Exploration and Production in Arctic and Subarctic Onshore Regions*. IUCN Gland, Switzerland and Cambridge UK, with E&P Forum, London UK. pp56.

Munro, J.W. 2006. Ecological Restoration and other conservation practices: the difference. *Ecological Restoration*. 24. 182-189.

Walker. D.A. 1997. Arctic Alaskan Vegetation Disturbance and Recovery. *Disturbance and Recovery in Arctic Lands*. Kluwer Academic Publishers: Netherlands. 457- 479.

MINE CLOSURE STRATEGY IN THE NORTH: THE USE OF SEMI-PASSIVE BIOLOGICAL TREATMENT FOR REMOVAL OF As, Sb AND Se FROM MINE IMPACTED WATER

Dr. Amelie Janin[1], Dr. Steve Wilbur[2], Isobel Ness[1]

[1] Yukon Research Centre, Yukon College, Whitehorse, YT, Y1A 5M2

[2]Victoria Gold Corporation, Vancouver, BC, V7X 1K8

Key Words: bioremediation, passive water treatment, sulfate-reducing bacteria, anaerobic bioreactor

Passive water treatment technologies are increasingly being considered for mine site closure in the Yukon. Efforts are currently underway in Yukon to test, compare and contrast passive treatment technologies with conventional technologies. This study aimed to provide additional information about the effectiveness of passive treatment technologies for mine water treatment in cold climates. To test the hypothesis that bioreactors can effectively treat mine-impacted water at low temperatures, four bench-scale, continuous flow bioreactors were assessed for their potential to remove As, Se and Sb from mine effluent over a one year period. More specifically, the objectives of this study were to: 1) assess the efficiency of removal of As, Sb and Se from a synthetic drainage with relatively high initial concentrations of the three metals in cold conditions, as well as from actual leachate collected at the Eagle Gold site, a proposed gold mine in central Yukon, 2) evaluate the effect of using wood chips as part of the composition of the bioreactor, and 3) assess the effect of the freeze/thaw transition on the bioreactors' performance.

Protocols were: Phase 1 (P1), the bioreactors were operated with the addition of liquid methanol as a carbon source, and in an environment with uncontrolled temperature during the fall until the bioreactors froze solid; Phase 2 (P2), the bioreactors were thawed in a fridge at a stable temperature of 6°C; Phase 3 (P3), the bioreactors were operated and monitored at 6°C; Phase 4 (P4), the bioreactors were operated at 6°C in the same conditions but without the carbon addition.

Results show that all bioreactors significantly decreased As, Sb and Se concentrations when carbon was added independent of influent concentration. Using the highly concentrated synthetic metals drainage with an average of 4.9 mg/L As, 0.14 mg/L Sb and 0.47 mg/L Se with a 1% methanol addition, the removal efficiencies were >93% , >96%, and >99% for As, Sb, and Se, respectively over phases 1 to 3. The results indicate that the use of spruce chips in the bioreactor substrate (C10 and C11) helped improve removal efficiencies, while mitigating the effect of freeze/thaw . It is suggested that the wood chips provide an adequate support to either protect and/or favor biofilm growth, which may promote sulfate reduction and metal sulfides precipitation. Finally, the efficacy for adding carbon was assessed by comparing the results for a 153-day period with 1% methanol added to a 220-day period without methanol added. Sb and Se were affected by the lack of methanol but high performances were still achieved with >95% and >88% removal of Sb and Se respectively over the 220 days (The Sb performances figure is not shown here resemble Se removal shown in Fig 1). On the other hand, As concentration rose steadily in the effluent once liquid carbon addition was stopped (Fig 2), suggesting that supplementing the reactor with liquid substrate is required to remove As in cold temperature. We surmise that adding methanol provided an easily biodegradable carbon source, which is much needed by the microorganisms responsible for As removal at cold temperatures.

This study is one of very few studies reported in the literature that demonstrates Sb removal from water by an anaerobic bioreactor. Overall, it demonstrates the potential application of passive anaerobic bioreactors as a technique to remove As, Sb and Se from mine water effluent in a cold climate where freeze/thaw happens as long as easily biodegradable carbon is available to the microbes. It also suggests that the addition of liquid carbon to the bioreactor may be required, especially for As removal in cold temperatures and freeze/thaw conditions occur. Further study would be needed to identify the temperature threshold under which the addition of carbon source is required and to scale up the bioreactor. It is suspected that better flows (less preferential pathways) would be achieved in a larger unit and would support removal efficiencies.

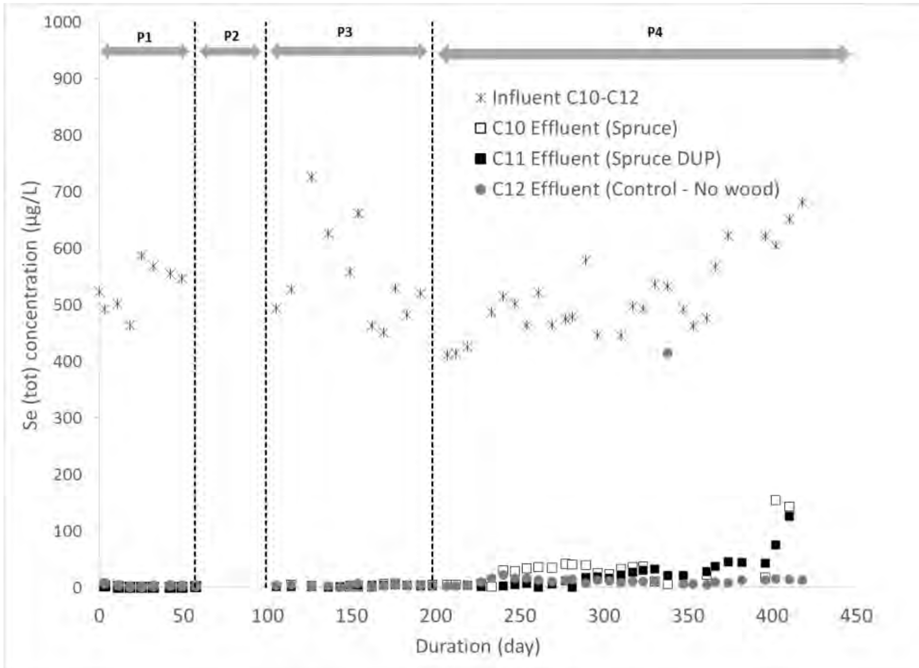


Figure 1. Se concentration in the influent and effluents from the C10-12 reactors (Phases 1-4; quantification limit is 0.7 ug/L, value below the limits are reported as 0.7)

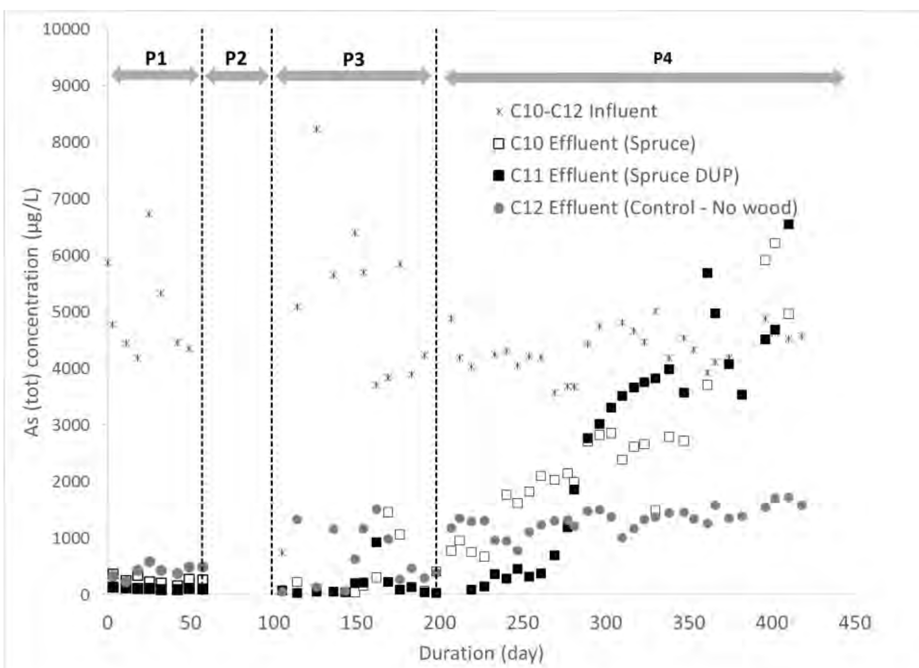


Figure 2. As concentrations before and after treatment by the bioreactors C10 and C11 (duplicates, wood-amended) and C12 (control, no organic amendment) (Phases 1-4; quantification limit is 0.8 ug/L, value below the limits are reported as 0.8)

SOIL MICROARTHROPODS: A PRELIMINARY STUDY OF THEIR POTENTIAL USE AS BIOINDICATORS IN ECOLOGICAL RESTORATION

Cassieanna Krane[1], Jesse Hoage[1], Damien Mullin[1], Erica Welch[1], Samantha Oliviera[1], Graeme Spiers[1,2], Peter Beckett[1]
[1] Department of Biology, Laurentian University, Sudbury, Ontario P3E 2C6
[2] School of the Environment, Laurentian University, Sudbury, Ontario P3E 2C6

Abstract:
This study introduced students to the use of the soil microarthropods as bioindicators of soil health by using population distribution data to evaluate and compare ecological health at selected reclaimed sites in the Sudbury area. The area has undergone extensive ecological degradation due to anthropogenic impacts such as forestry and smelting, followed by decades of revegetation treatments. The treatments included the application of dolomitic limestone and fertilizer, seeding with grasses and legumes followed by tree planting and, most recently, the selective placement of imported forest-floor transplant plots on selected regreened sites. The morphology of collected Humus Forms from both untreated and treated sites were described in detail, with soil arthropods being then extracted using a series of Berlese-Tullgren funnels for estimates of population membership and diversity which were then interpreted as indicators of ecological health status. The results from this small preliminary study indicated that the transplant plots have greater humus form profile development, an observation suggestive of higher levels of arthropod activity and species diversity producing more organic detritus promoting higher microbiological decomposition rates. The lack of extensive sample replication, coupled with identification of the arthropods mainly to the order level, limit any definitive conclusions from being drawn from this study, although it most certainly served as a valuable learning experience.

Key Words: forest floor transplants, Berlese-Tullgren extraction, ecosystem restoration, Sudbury

Introduction

Soils are the very basis for the majority of life on earth, and the importance of the diverse array of organisms which inhabit these ecosystems underfoot cannot therefore be underestimated (Orgiazzi *et al*. 2016). As key decomposers (Latif, 2013), soil dwelling arthropods are included amongst these critical soil inhabitants, and as their populations are often seen as indicative of ecosystem health they are termed bioindicators by many (Bardgett 2002). Crucial as they are in the functioning of soils, observing these arthropods is not however straightforward, with specialized techniques and tools required for their extraction and examination (Orgiazzi *et al*. 2016). The application of these techniques vary and, although a subject of debate within the scientific community, researchers agree that the observation method selected depends on the ecology and behaviour of the target species (Yi *et al*. 2012).

Reviewed studies which conducted arthropod sampling in order to assess their abundance, richness and diversity when evaluating the health of an ecosystem used some variation of the Berlese – Tullgren funnel, with a few employing a modified version, such as one which used an additional chemical gradient (Freedman and Hutchinson, 1980), and another which employed Murphy's split funnel, as well as a simple funnel without heat in order to tease out acarina and collembola, and nematodes and enchytraeids respectively (Marshall, 1974). The Berlese-Tullgren funnel did, however, appear to be known for being a simple yet biased method as it is unable to obtain a complete picture of the soil arthropods present (Andre 2002).

The depth of the soil cores collected ranged between 5 cm (Andrés and Mateos, 2006) to 15 cm (Marshall, 1974), though 10 cm cores were the most common (St.John et al. 2002; Lindberg and Bengtsson, 2006; Iloba and Ekrakene, 2009). All studies performed multiple replications per plot, ranging from 2 (Creamer et al. 2008; St. John et al. 2002), to 5 (Komulainen and Mikola, 1995), and in one case, 60 (Freedman and Hutchinson, 1980). One of the studies surveyed outlined the techniques which they used to identify, sort, and extract the soil microarthropods, which was by examining the microarthropods suspended in a solution under a dissection microscope, and extracting and sorting them using a pipette (Iiloba and Ekrakene, 2009).

The study region of Sudbury, Ontario has been subjected to an intensive regreening effort following extensive environmental degradation beginning in the 19th century due to anthropogenic activities such as forestry, mining and smelting (Courtin 1994). As a result of this industrial damage, 20 000 hectares were completely barren and devoid of vegetation, with an 80 000 additional hectares being semi-barren at the initiation of large scale reclamation projects in the 1970s (VETAC 2007). The loss of vegetation due to forestry, fire, and the release of environmental toxins such as copper, nickel, and sulphur dioxide from smelting also promoted severe soil erosion (Freedman and Hutchinson, 1980). However, by 2007, the regreening program, a well-coordinated community initiative operating through multiple partnerships, had led to: (1) the planting of 9.5 million trees, (2) the planting of 280 thousand shrubs, (3) and the liming, seeding, and fertilizing of approximately 3500 hectares of land (VETAC 2015, figure 1).

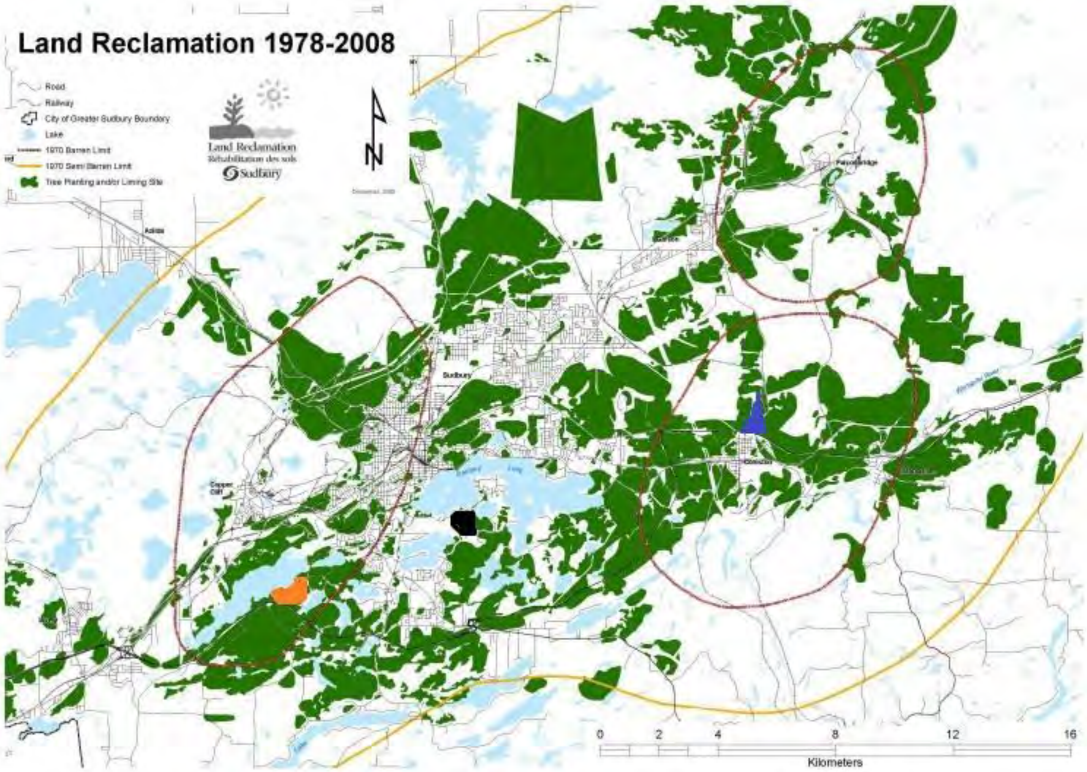


Figure 1. Map of the Sudbury area, with the location of the historically barren zones (outlined in red) and semi-barren zone (outlined in yellow). The map also indicates (in green) all of the areas to which lime was applied, and/or trees were planted, between the years 1978 and 2008. The three experimental sites have also been highlighted, Kelly Lake (orange), Laurentian University (black), and the Jane Goodall trails (blue) (VETAC, n.d).

Despite all of the regreening efforts, the Sudbury Area Risk Assessment completed in 2007 recognized that the land reclamation project had neither restored woodland understory species (Beckett et al., 2007), nor had terrestrial insect communities recovered to the extent as had plant communities (Babin-Fenske and Anand, 2010). These understory species are normally established as a component of the succession process, and their continued absence is likely due to a combination of unfavourable conditions such as thin, nutrient-poor soils, and lack of a suitable seed-bank (Kozlov and Zevera 2007). In an effort to reintroduce these missing species, including herbaceous plants, mosses and lichens, as well as other taxa such as arthropods, the addition of a new initiative to the regreening program in 2010 was the import of forest floor plots (VETAC 2010). Preliminary observations have indicated that some of these new understory species are surviving and thriving in their new habitats (Santala 2014).

This report outlines preliminary work investigating the soil arthropod communities within these transplanted materials as soil arthropods are often measured as bioindicators when evaluating site health, especially when assessing site recovery following environmental stressor disruptions such as drought (Lindberg and Bengtsson, 2006), pesticide use (Latif, 2013; Iiloba and Ekrakene, 2009), and mining activities (Creamer et al., 2008; St.John et al., 2002; Andrés and Mateos, 2006). For example, a study by Southwood et al. (1982) which found that (1) arthropod richness was correlated with tree abundance, and (2) arthropod abundance increased with community species richness, suggests that, if the transplant plots increase understory vegetation species richness, the abundance of arthropods should also increase.

Surface humus form samples were collected from several sites within the region, including regreened, transplant, and untreated plots, to assess the diversity of soil arthropods present in these plots, if they may serve as indicators of ecological health, and especially whether the transplant plots hosted the most diverse communities. A detailed description of the soil humus forms at the sites was completed to provide an understanding of the humus development rates as an indicator of soil organism plant degradation activity, an additional simple visual characteristic to evaluate site health.

Methods

Study Sites

The study region within the City of Greater Sudbury, Ontario, Canada is located within the northern region of the Great Lakes – St. Lawrence forest zone which experiences hot summers and cold winters, with temperatures ranging from above 30 °C to below -30°C, and a mean annual precipitation of approximately 870 mm. Tree species, either planted or volunteer colonizers, include White Pine, *Pinus strobus*, Red Pine, *Pinus resinosa*, Jack Pine, *Pinus banksiana*, White Spruce, *Picea glauca*, Black Spruce, *Picea mariana*, White Birch, *Betula papyrifera*, Trembling Aspen, *Populus tremuloides*, and Red Maple, *Acer rubrum* (SARA group 2004).

Samples were collected from three sites throughout the Sudbury area, two located in historically barren areas, Kelly Lake (KL) and the Jane Goodall trails (JG), while the third, from Laurentian University (LU), was located within the historically semi-barren zone. Four separate vegetation zones were sampled at the KL site, an oak-poplar, stand, a White Pine stand, a White Birch stand, and a barren area. The birch stand the barren zone had not been limed as a part of the re-greening program. The vegetation at both the JG site and the LU site was dominated by the typical Sudbury mixed forest species.

Transplant Plots

Forest floor mats, measuring 0.64m long, 0.54m wide, and 0.10m thick were taken from undisturbed donor sites approximately 50km South of Sudbury and placed at disturbed, recipient sites as 4m by 4m experimental plots to encourage the survival of vegetation even if the edges of the plots dried out (Santala 2014). The mats were 0.10m thick to allow for the shallow root systems within the humus forms to be harvested and, thus, replanted (Santala 2014).

Sampling and Analysis

Soil samples were taken in the form of soil cores, which were extracted using soil corers fashioned from 10 cm sections of 10 cm diameter aluminium pipe. Six soil samples were collected at the KL site, two were collected at the LU site, and three were collected at the JG site. At the KL site, two samples were collected from both the regreened oak-poplar and the white pine stands, with each having a sample taken from both a transplant and a control plot. A single sample was taken at the KL untreated, barren zone birch stand. At the regreened LU site, one sample was taken from a transplant plot, while another was taken from a control plot. At the JG site, two samples were taken from transplant plots, one of which was introduced in 2010 and the other in 2015, while the third sample was from a control plot.

The soil arthropods were extracted from the cores using a Berlese-Tullgren funnel installed below a 60w incandescent lightbulb to establish a temperature gradient. The arthropods were then stored suspended in a sugar-alcohol-water solution prior to identification to Order level on examination under a dissection microscope of a thin film of the organism-rich solution into a petri dish. Humus form samples were also taken to enable detailed pedological descriptions. The Shannon – Wiener index of diversity was used to quantify the diversity of arthropod orders found in each sample.

Results

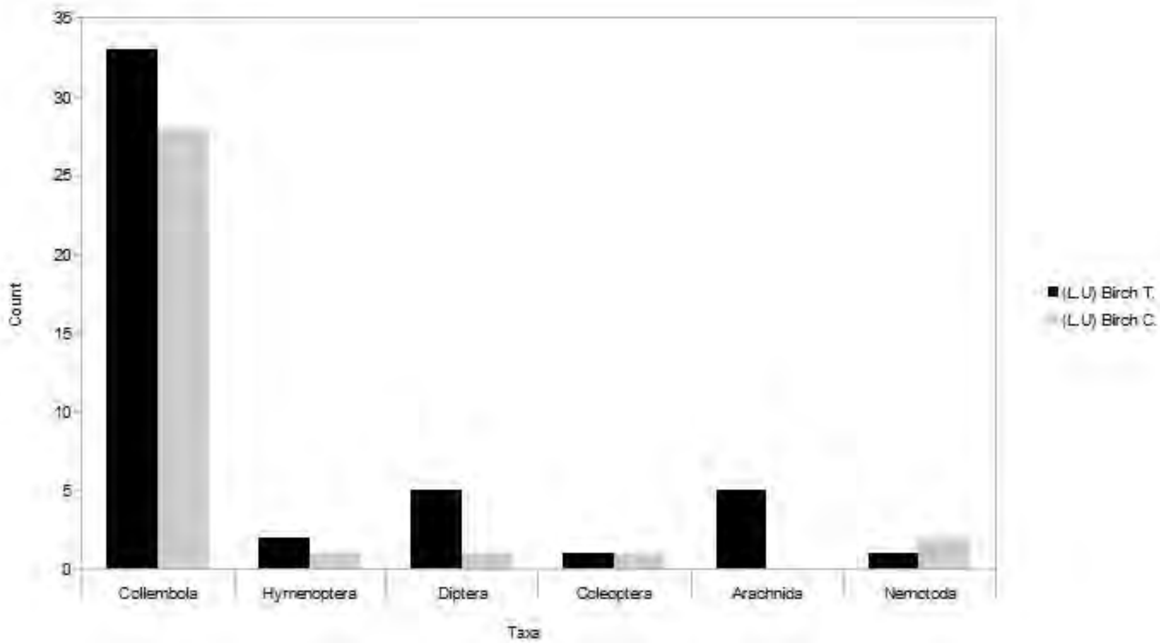
Of all of the arthropod orders, acarina was the most abundant. The proportion of total arthropods counted in the sample which were acarina ranged from 23% in the KL White pine sample to 84% in the KL barren sample, with the abundance of acarina ranging between 6 individuals in the JG 2010 transplant to 482 in the KL barren sample. Collembola were the only other order found in all ten of the samples analyzed. There were 14 orders of arthropods identified at the KL site (Figure 2), 11 at the JG site (Figure 3), and 7 at the LU site (Figure 4).

Figure 2. Abundance of all arthropod orders identified in the samples collected at the Kelly Lake sites; two from the oak-poplar stand (Control (C), Transplant (T5)), one from the unlimed birch stand, one from the unlimed barren zone, two from a white pine stand (Control (C), Transplant (T4)).

Figure 3. Abundance of the arthropod orders identified in the samples collected from the Jane Goodall trails site; one from a transplant established in 2010, the other from a control plot.

Figure 4. Abundance of each of the arthropod orders identified in the samples collected at the Laurentian University site; both from a birch stand, one from a control (C) plot, one from a transplant (T) plot.

The diversity index values for arthropod orders found in these samples ranged from 0.6, for the KL barren sample, to 1.35 for the LU birch stand transplant sample (Table 1).
Th ... e
JG
Th
exc
wh



Tab
take
The
an
star
wer

Sample	Diversity Index (H)
(K.L) Oak-Poplar Control	0.77
(K.L) Oak-Poplar Transplant 5	0.74
(K.L) Birch Unlimed	0.95
(K.L) Barren 1	0.6
(K.L) White Pine Transplant 4	0.99
(K.L) Limed Pine Control	1.05
(J.G) 2010 Transplant	1.29
(J.G) Control	0.96
(L.U) Birch Transplant	1.35
(L.U) Birch Control	0.93

Table 2. Humus form profile descriptions for seven soil samples, three from the Jane Goodall trails site (J.G), two of which were from transplant plots, while the third was a control. Two samples were taken from a birch, *Betula papyrifera*, stand at the Laurentian University site (LU), one of which was a transplant, while the other was a control. Two samples were from an Oak-Poplar, *Quercus* sp. *Populus* sp., stand at the Kelly Lake site (KL). (Klinka 1981, 1993)

Sample Name	Horizon Name	Classification
(J.G) 2010 Transplant	Ln Fa Hgw	Lignomoder
(J.G) 2015 Transplant	Ln Fr Hc	Lignomor
(J.G) Control	Ln Fr Hcw	Lignomor
(L.U) Birch Control	L (<1 cm) Fm (1 cm) Fr (1.5 cm) Hc (1 cm)	Rhizomor or Lamimor
(L.U) Birch Transplant	L (<0.5 cm) Fz (<0.5 cm) Hc (3 cm)	Leptomoder
(K.L) Oak-Poplar Control	Lv (<0.5 cm) Frm (1-1.5 cm) Ah (3 cm)	Lamimoder
(K.L) Oak-Poplar Transplant 5	L/F (transplant) (1.5cm) Ah (3 cm)	Mullmoder

Discussion

At both the LU and the JG sites the arthropod abundance for the samples taken from the transplant plots had higher indices of diversity than their respective control plots. The results were slightly more complicated at the KL site, as neither the oak-poplar, nor the white pine transplant plots had higher diversity indices than their respective controls, though that of the barren zone was the lowest of all samples analyzed. In the case of the oak-poplar and white pine samples at KL the differences in diversity indices were relatively minor, at 0.77 to 0.74 and 0.99 to 1.05 respectively. For the LU and JG samples, however, the differences were much greater, being 1.29 to 0.96 and 1.35 to 0.93 respectively.

The results obtained for the unlimed KL barren zone sample were somewhat expected, as the acidic soils of the area support very little vegetation cover (SARA Group, 2004). Given this combination of environmental factors, the soils at the barren site may not have been favourable to a wide range of species (Creamer *et al.* 2008). Interestingly, however, the stunted birch covered barren zone sample did have the highest abundance of acarina of all of the samples analyzed. This high abundance is not necessarily correlated with high diversity of acarina species, as they were not identified to that level, and therefore may have been a case of only a few colonizing species dominating an area, as it is acknowledged to occur in many instances of stressed ecosystems, such as those polluted by metals (Creamer *et al*. 2008).

Nematodes, normally exceedingly common in soils worldwide, were only identified in low abundances in five samples. This observation may be due to multiple factors, including high metal concentrations, such as copper, being observed as limiting nematode abundance (Creamer *et al*. 2008). There is also the possibility that nematodes were not properly extracted from the soil samples because of a high drying and temperature gradient during the extraction phase, leading to few organisms for identification under the dissection microscopes.

Colonization of polluted areas by microarthropods has been demonstrated to be limited to only a few species, even if there are revegetated areas nearby, as they disperse very slowly (St.John *et al*. 2002). This limited dispersal potential, along with the hypothesis that the degraded Sudbury soils may be less habitable to many species, may explain why diversity was greater in the transplant plots than in the control plots at the LU and JG sites. Studies previously conducted on revegetated mine tailings in Sudbury demonstrated that the species richness and diversity of acarina was less on tailings, even on sites revegetated 40 years previously, compared to nearby control plots (St.John *et al*. 2002). The tailings study also found the density of mite species to be similar in the old tailings plots and control plots, with species compositional diversity indicating the plots to be no more than 60% similar (St.John *et al*. 2002). In this latter study, the abundance was high in the low diversity stressed system, an observation similar to that found in this study. However, direct comparisons between the two studies cannot be conclusively drawn as that study identified mites to the species level, specifically, whereas this broad study identified only arthropod orders. The results, based on the parameters measured, in the current study cannot conclude whether the soil microarthropods from the transplant plots have migrated outwards. However, as soil microarthropods migrate exceedingly slowly (Creamer *et al*. 2008), the five year period between the import of the transplants to the Sudbury regreeening plots and the sampling period in the Fall of 2015 may have been insufficient to allow for significant migration.

The litter layer (L) on the soil profiles at the LU site was similar. The regreened control plot humus form had two distinct fermentation layers (F), neither of which was zoogenous in nature as observed for the transplant plot. In the transplant plot, the litter layer (L) and the fermentation layer(s) (F) were thicker in the control plot than in the transplant plot. These observations may support the hypothesis of a thriving and active microarthropod community in the transplant plot speeding decomposition (Latif, 2013), thinning the litter and fermentation layers, and leaving abundant fecal matter in the form of a zoogenous fermentation layer (Klinka *et al*. 1981). The JG site had two transplant plots, one installed in 2010, and the other in 2015 only months before the sample was taken. The results of the JG soil profiles did not point towards a clear pattern in soil arthropod activity, with the younger 2015 transplant being more similar to the regreened control than the 2010 transplant. The 2010 transplant was the only one of the three to have a humus layer (H) composed primarily of faunal droppings. Both the control and transplant profiles from the KL oak-poplar site had less than 2 cm of litter and fermentation materials, with no clear humic layer (H). In contrast, the thin organic layer was over an organo-mineral layer enriched with humic materials (Ah). Thus the limed control plot was different from the transplant plot, with the fermentation layer being distinguished from the litter layer (L), whereas the transplant had a surface layer which was a mix of litter and fermenting (F) materials. These results may, as with those for the LU sites, support the hypothesis that the transplant plots contained a more active microarthropod community which aided in the decomposition of the annual litter contribution from the understorey and canopy vegetation.

The use of the simple Berlese – Tullgren funnel was well suited to this preliminary study. Similar studies surveyed also appear to use a variant of the technique, with the simplicity of the method for this preliminary class-based study being critical in spite of the potential lack of reproducibility and accuracy. The collection of 10 cm deep soil samples was suitable as the studied soils are commonly shallow, often with less than a 10 cm surface humus form, a depth described in the literature as containing the majority of soil microarthropods (St.John *et al*. 2002; Marshall, 1974). There may have been several sources of error in the identification and enumeration of these microarthropods, with the identification being completed by several students, with potential discrepancies in observation and use of the systems for identification. The lack of extensive sample replication also limits the strength of conclusions drawn from the results which fail to truly account for variation, and therefore may not present a complete picture as to the number and diversity of microarthropods present at each site.

Conclusion

Although the results of this study were inconclusive, the preliminary findings appear to support the hypothesis that the transplant plots introduced into the area may host a greater diversity of microarthropod orders. The barren site at KL had the least diverse microarthropod community, strongly supporting the hypothesis that microarthropod diversity is potentially indicative of ecosystem health. The transplant plots had humus form profiles with thinner litter and fermentation layers, as well as more evident faunal droppings, than their respective regreened control plots, indicative of greater, more productive, or more diverse microarthropod communities in those plots.

In future studies, the systematic errors in this study might be mitigated by increasing the replication, and by having a team of researchers more comparable in their methods. The impact of these transplant plots could be more thoroughly understood by determining with a greater degree of precision which microarthropod taxa inhabit these transplants, which groups disperse from the transplants, and at what rate this dispersal occurs. This detailed faunal studies could be completed by sampling on a gradient over a series of years, and identifying the microarthropods found to the genus or even species level.

The Sudbury Story was once only a cautionary tale, an example of the harm which we could wreak on our environment as a consequence of unmanaged industrial releases to the host environment. The continued successes of the Sudbury regreening program however, further supported by these preliminary findings, have since expanded that cautionary tale into one of inspiration for all the world to see.

Literature Cited

André, H. M., Ducarme, X., Lebrun, P. & Andre, H. M. 2002. Soil Biodiversity: Myth, Reality or Conning? *Oikos* 96, 3–24.

Andrés, P., and E. Mateos. 2006. Soil Mesofaunal Responses to Post-mining Restoration Treatments. *Applied Soil Ecology.* 33: 67-78.

Babin-Fenske, J., and M. Anand. 2010. Terrestrial insect communities and the restoration of an industrially perturbed landscape: Assessing success and surrogacy. Restoration Ecology 18(S1): 73-84.

Bardgett, R. D. 2002. Causes and consequences of biological diversity in soil. *Zoology (Jena).* 105, 367–374.

Beckett, P., J. Braun, J. O'Hara, M. Williams, S. Monet, and T. McCaffrey. 2007. Ecological status of the Sudbury land reclamation program. Presentation presented at the 2007 Sudbury Mining and Environment Conference in Sudbury, Ontario. 26 Slides. URL: http://sudbury-mining-environment.ca/2007Presentations/Session%204%20-%20A/3%20Beckett%20Peter.pdf

Courtin, G. M. The last 150 years: a history of environmental degradation in Sudbury. 148, 99–102 *Sci. of the Total Env.* (1994).

Creamer, R. E., D. L. Rimmer, and H. I. J. Black. 2008. Do Elevated Soil Concentrations of Metals Affect the Diversity and Activity of Soil Invertebrates in the Long-term? *Soil Use & Management.* 24: 37-46.

Freedman, B., and T. C. Hutchinson. 1980. Effects of Smelter Pollutants on Forest Leaf Litter Decomposition near a Nickel–copper Smelter at Sudbury, Ontario. *Can. J. Bot.* 58: 1722-736.

Iloba, B.N. and Ekrakene. 2009. Soil arthropods recovery rates from 5 – 10 cm depth within 5 months period following dichlorov (an organophosphate) pesticide treatment in designated plots in Benin City, Nigeria. *Journal of Entomology and Nematology.* 1: 43-49.

Klinka, K., Green, R.L., Trowbridge., Lowe, L.E. 1981. Taxonomic Classification of Humus Forms in Ecosystems of British Columbia. Province of British Columbia Ministry of Forests. Victoria, B.C.

Komulainen, M., and J. Mikola. 1995. Soil Processes as Influenced by Heavy Metals and the Composition of Soil Fauna. *The Journal of Applied Ecology.* 32: 234-41.

Kozlov, M. V. & Zvereva, E. L. Industrial barrens: Extreme habitats created by non-ferrous metallurgy. *Life Extrem. Environ.* 69–97 (2007).

Latif, M. A.A. 2013. *Ecological Role of Animal Diversity in Soil System (A Case Study at El-Rawakeeb Dry Land Research Station, Sudan)*. Proc. of 1st Annual International Interdisciplinary Conference, Azores, Portugal. N.p.: n.p. Web.

Lindberg, N., and J. Bengtsson. 2006. "Recovery of Forest Soil Fauna Diversity and Composition after Repeated Summer Droughts." *Oikos* 114: 494-506.
Marshall, V. G. 1974. Seasonal And Vertical Distribution Of Soil Fauna In A Thinned And Urea- Fertilized Douglas Fir Forest.*Can. J. Soil. Sci.* 54: 491-500.
Orgiazzi, A., Bardgett, R.D., Barrios, E., Behan-Pelletier, V., Briones, M.J.I., Chotte, J-L., De Deyn, G.B., Eggleton, P., Fierer, N., Fraser, T., Hedlund, K., Jeffery, S., Johnson, N.C., Jones, A., Kandeler, E., Kaneko, N., Lavelle, P., Lemanceau, P., Miko, L., Montanarella, L., Moreira, F.M.S., Ramirez, K.S., Scheu, S., Singh, B.K., Six, J., van der Putten, W.H., Wall, D.H. (Eds.), 2016, Global Soil Biodiversity Atlas. European Commission, Publications Office of the European Union, Luxembourg. 176 pp.
Santala, K.R. 2014. The introduction of native forest floor plant species into the industrially disturbed forests of Sudbury, Ontario, Canada. MSc. Biology Thesis. Laurentian University Sudbury, Ontario, Canada.
Santala, K. R. S. Monet; T. McCaffrey, D. Campbell, P. Beckett, and P. Ryser. 2016. Using turf transplants to reintroduce native forest understory plants into smelter-disturbed forests. *Restor. Ecol*. 24(3): 346 - 353.
SARA Group. 2004. *Sudbury Soils Study Combined Soils Reports Executive Summary*. Rep. Vol. 2.0. Sudbury, Ontario: Sudbury Soils Study.
Southwood, T.R.E., V.C. Moran, and C.E.J. Kennedy. 1982. The richness, abundance, and biomass of the arthropod communities on trees. Journal of Animal Ecology 51: 635-649.
St. John, M., G. Bagatto, V. Behan-Pelletier, E.E. Lindquist, J.D. Shorthouse, and I.M. Smith. 2002. Mite (Acari) colonization of vegetated mine tailings near Sudbury, Ontario, Canada. *Plant and Soil.* 245: 295-305.
Yi, Z., F. Jinchao, X. Dayuan, S. Weiguo, and J. C. Axmacher. 2012. A Comparison of Terrestrial Arthropod Sampling Methods. *Journal of Resources and Ecology.* 3: 174-82.
Vegetation Enhancement Technical Advisory Committee (VETAC). 2007. Annual Report 2007. Technical Report: 21pp. URL: http://sudbury-mining _ environment.ca/2007Presentations/Session%204%20-%20A/3%20Beckett %20Peter.pdf Accessed 20 February 2016.
Vegetation Enhancement Technical Advisory Committee (VETAC). 2010. Regreening Sudbury: Annual Report 2010: Land Reclamation Program. Technical Report: 30pp. URL:http://www.greatersudbury.ca/content/div_landreclamation/documents/2010%20Final%20Annual%20Report.pdf
Vegeatation Enhancement Technical Advisory Committee (VETAC).2015. Regreening Sudbury: Annual Report 2015: Land Reclamation Program. Technical Report: 32pp. URL:http://www.greatersudbury.ca/living/environmental-initiatives/regreening-program/pdf-documents/2015- regreening-program- annual-report/
VETAC. "Forest Floor Transplants." *Environmental Initiatives*. City of Greater Sudbury, n.d. Web. 2016.

The million dollar mistake

Steve Lalancette, president

Regeneration

In business, a million dollar mistake refers to an error that has three main characteristics. First, it's a big one. It has large consequences that are hard to correct. Second, it's definitely not intentional. Those responsible were doing their best. However, third and most importantly, it was avoidable.

Over the last 20 years, academic and commercial research and development initiatives have led to the creation of new restoration techniques where acid generating tailings are sequestered in highly stable structures. Covers with capillary barrier effects (CCBE), elevated water tables and covers with geomembranes are now commonly used as tools to rehabilitate mining sites no longer in operation. Prediction models estimated that these structures could remain stable for hundreds of years… Or so we thought.

Often considered as "icing on the cake", vegetation has not usually been taken into particular consideration in the design of these structures. The lack of understanding of biological concepts such as ecological succession or plant nutrition has led to the false beliefs that trees would not grow in what was seen as inhospitable conditions or that herbaceous covers would prevent their colonization.

But they do grow. In certain cases tree growth can be prolific and very rapid. And they do so on structures that were not designed to tolerate them.

With their extensive and often deep root systems, trees are now threatening the very integrity of structures that were constructed at high cost. Roots will grow across the capillary barriers of CCBEs and through most geomembranes. Through evapotranspiration, they also have a very important impact on water tables, each tree pumping hundreds of liters of water into the atmosphere every day. This will affect the saturation levels in CCBEs and the height of water tables, sometimes bringing them far below planned levels.

Therefore as an industry, we have made a million dollar mistake. The mistake was not intentional, but why was it avoidable?

A simple (if not simplistic) way to understand why tree grow on what looks like inhospitable structures is to understand what they are made of. First, they contain between 85% and 95% of water. Both CCBEs and elevated water tables contain water as part of their structure that can serve as water reserves for trees. Structures using geomembranes may not offer a water reserve *per se* but simple water retention from rainfall is often enough to initiate growth. Second, the three major elements composing the dry mass of a tree, carbon, oxygen and hydrogen (up to 95% of dry mass, *i.e.* excluding water), are obtained from the atmosphere or from water.

Simplistically put, 99% of elements composing total tree mass are always available. Moreover, since trees use the sun as a source of energy, they do not need organic matter.

If we look at the remaining 1% of mass, we find that certain elements are essential to tree growth and survival even if required in small quantities. Three of them, nitrogen, phosphorus and potassium, are critical. They are usually present on mining structures, but not in a form which trees can use. Nitrogen is present in large quantities in the atmosphere (N_2) and phosphorus and potassium in various mineral forms. However, trees require nutrients to be in solution.

This is not a permanent obstacle to tree growth. Most tree species can develop symbiotic relationships with mycorrhizal fungi. Theses fungi are able to access nutrients, such as nitrogen, phosphorus and potassium, in organic form and certain mineral forms (phosphorus and potassium). These sources are unavailable directly to plants. Other species, such as members of the *Alnus* genre, also form symbioses with nitrogen fixing bacteria (*Frankiae*) that transform atmospheric nitrogen into chemical compounds that plants can use. Their growth on a structure will increase the amount of nitrogen available to other plants and facilitate the arrival of other tree species.

Most of the other essential elements are required in very small amounts and are usually present as water contaminants therefore accessible to plants. If not, they are often available through their fungal partners.

In a nut shell, nothing on CEBBs, elevated water tables or coverages with geomembranes will prevent the growth of trees. In fact, often nothing is completely lacking as far as their nutritional needs are concerned. Conditions on these structures are far from ideal, but not so that tree growth will be null. It might be slow, mortality rates might be high, but in the end tree growth is almost unavoidable.

The biological processes that underline tree growth are well known and have been known for quite a while. The type of ecological succession we are seeing on mining structures has been studied on other types of structures that are very similar, such as dams and dikes built for the production of hydroelectricity. By failing to take this knowledge into consideration the rehabilitation industry, which I consider myself part of, has failed and made a million dollar mistake.

We must therefore ask ourselves why we made such a mistake in order to prevent them from happening again. Far from being absolutes, a few aspects of our industry have to be considered if we want to increase our ability to prevent such mistakes.

First, our industry is a young one. It simply did not exist 40 years ago. So even if it is easy to consider something as obvious after the fact, it is not always so clear beforehand. We still have much to learn and therefore must use an extra degree of caution when developing new solutions.

Second, our industry is complex. The rehabilitation industry includes a very wide array of expertise ranging from civil engineering to water treatment, hydrogeology to biology. There are therefore no such thing as a rehabilitation expert. Firms, academics and consultants must work together in order to tackle complex problems. Identifying the right people with the right expertise can be tricky, especially when we are unaware that certain aspects of a situation can be problematic simply because we are not an expert in that particular field. We must therefore refrain from making assumptions and refer to someone else as soon as we leave our own area of expertise. Partnerships and strategic alliances will become key to solve increasingly complex environmental issues.

Finally, we are a cost industry. We do not create revenues for our clients. Even if the environment is becoming an increasingly important part of corporate values, by its nature, it will always be under considerable strain to be as cost effective as possible. However, sacrificing on quality in order to reduce short term cost can be expensive in the long term.

Our failure to anticipate the impact of vegetation on rehabilitation structures in the mining industry is a good example of the type of challenges our industry is facing. Fortunately, solutions are now in place to control and prevent trees from colonizing confinement structures. Bio barriers and high resistance factor vegetation coverages will prevent tree seeds from germinating and might even kill young trees. New vegetation control techniques using fungi will kill adult trees and prevent their regrowth.

But for some structures, it is already too late and reconstruction cost will represent millions of dollars. Let us therefore see this as a reminder that we must remain open minded to the complementary expertise of our colleges (and competitors) if we want to avoid another million dollar mistake.

Invasion and Control of Exotic Cattails in Wild Rice Stands in Ontario

Peter Ferguson Lee and Kristi Dysievick, Lakehead University, Thunder Bay, ON, P7B 5E1, and John Kabatay, Seine River First Nation, Mine Centre, ON, P0W 1H0.
Key Words: southern wild rice, northern wild rice, cattails, invasive species

Introduction and Objectives:
Two species of wild rice are found in Canada (Aiken et. al, 1989). Southern wild rice (*Zizania aquatica* L.) is restricted to the lower Great Lakes where it is at the northern limit of its range in North America. The southern species has been largely extirpated by invasive carp and cattails. Northern wild rice (*Zizania palustris* L.) naturally occurs from the eastern seaboard of Canada inland to eastern Manitoba but its range has been extended westward through seeding efforts for commercial purposes. The open panicles of southern wild rice make it prone to seed loss and has no commercial value. This review describes the management efforts directed at both species. The particular issue of invasive cattails now affects the northern species and efforts are described to eliminate the potential catastrophic effect that occurred with the southern species.

The native species of cattails in Canada is *Typha latifolia*. Although the native species can form large monocultures in areas formerly occupied by a variety of other species, it is limited to depths of less than 25 cm. By comparison, the exotic, *Typha angustifolia*, or the hybrid, *Typha glauca*, can tolerate depths of up to 1.5 m (Grace and Harrison, 1986). *T. angustifolia* is thought to have spread from the eastern seaboard of North America inland to the Great Lakes in the late 19th century (Hotchkiss and Dozier, 1949). It has spread further westward in the 20th century. The distribution maps of Grace and Harrison (1986) show the species reached the Rainy River area by at least the time of their publication. The exotic cattails occupy the same niche as wild rice with the advantage of being a perennial and thus able to usurp the annual wild rice particularly when water levels increase from year to year. This invasive species is now having devastating effects on the largest stands of northern wild rice in the world (Lee, 2015).

Materials and Methods:
Study Locations: For southern wild rice, efforts were directed at re-establishing a self sustaining population of this species at the Royal Botanical Gardens in Hamilton, Ontario. The construction of a large carp barrier that prevented carp from entering the 320 hectare Cootes Paradise marsh, enabled re-vegetation to occur. Considered to be the historical dominant species, southern wild rice required a seed source. Surveys of reported sites of the species revealed only one source at Rondeau Provincial Park and seed was collected from this location. Cootes Paradise was assessed for suitable soil conditions and establishment of southern rice was conducted inside barriers preventing incursion of remaining carp. Planting success was evaluated and new establishment locations within the marsh selected.

For northern wild rice, efforts by the Seine River First Nation were directed at cutting cattails in Rainy Lake using a cutting bar apparatus attached to the front of their airboat. The culms of the cattails were cut at the sediment:water interface in the fall of

2014. The theory to the procedure is that removing the culms stops flow of oxygen to the rhizomes in the anaerobic sediment in winter, killing the plants (Sale and Wetzel, 1983). Results of the cutting trials were assessed in 2015 and effects on the nutrient regime of the underlying sediment where cattails were growing examined.

Results and Discussion:
Southern wild rice: Detailed analysis of soil conditions within Cootes Paradise revealed locations matching existing natural stands of southern wild rice. Small enclosures containing wild rice during the first year of production within the suitable areas were joined forming large enclosures. Rice seed was sufficient to enable complete coverage within the large enclosures by the second year. After ten years, a viable self sustaining population of southern wild rice exists in Cootes Paradise.

Northern wild rice: Cattail removal proved to be remarkably effective. In the areas that were cut, the cattails were completely eliminated (Fig. 1). Additionally, the native species that were present in the cut areas (water lilies, soft stem bulrush) were not affected and since these species were in low density, they had little effect on wild rice production. Adding to the success of the procedure, rice apparently remained viable in the seed bank in the area previously occupied by cattails. The cut area was completely filled with wild rice without seeding being required.

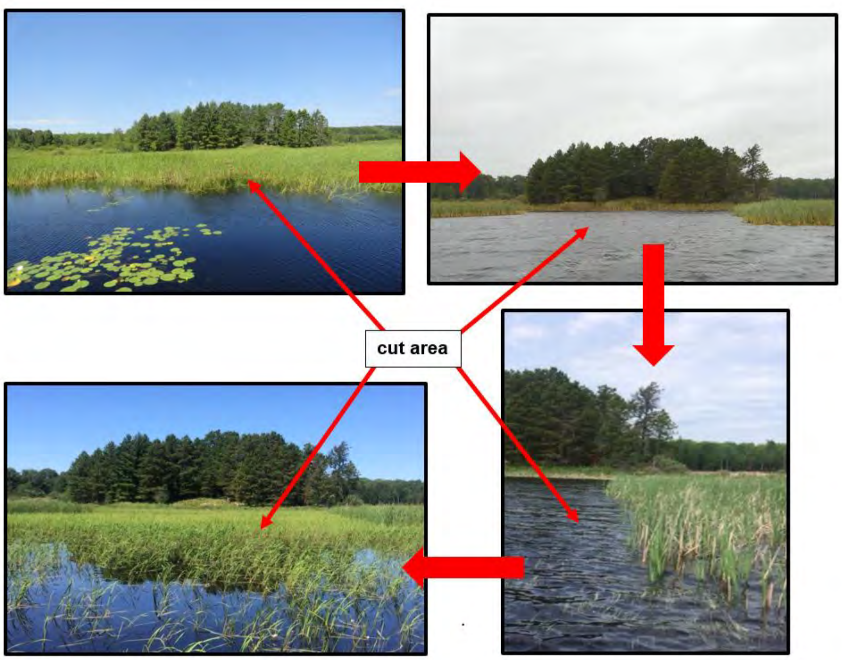


Figure 1. Sequence of cattail cutting trial on Rat River Bay, Rainy Lake. Upper left, prior to cutting cattails, upper right, after cutting, fall, 2014; lower right, spring, 2015 with rice in floating leaf stage; and lower left, re-established rice stand, fall, 2015.

The presence of cattails also had an effect on the underlying sediment (Table 1). Extractable nutrient values showed lower concentrations of P, NO_3, NH_4, and Mn in the cattail dominated area than the wild rice area while Ca, K, Na, Cu, Fe and Zn were all higher. Similar results were found for total nutrient and pore water nutrients. Plant tissue results showed that per 0.25 m^2, total C, total N, Ba, Ca, Cu, Mg, Mn, Na, P, S, Sr and Zn were all higher in the cattails' plant tissue compared to the wild rice study area. Perhaps most significant, there was a reduction in tissue nitrogen in the wild rice growing in the cut area where it exhibited chlorosis symptomatic of nitrogen deficiency.

Table 1. Extractable nutrients in sediment from cattails, natural wild rice and cut area.

Description	Mean	SD	Mean	SD	Mean	SD
Ext. Ca (µg / g)	1023.1	181.9	817.2	134.1	1150.0	168.5
Ext. K (µg / g)	5.9	2.2	5.9	4.1	10.1	4.4
Ext. Na (µg / g)	2.6	0.7	3.9	2.2	15.1	25.0
Ext. Cu (µg / g)	1.4	0.4	1.4	0.6	2.3	0.5
Ext. Fe (µg / g)	52.4	19.1	61.9	22.0	93.4	79.9
Ext. Mn (µg / g)	7.5	0.8	10.8	5.7	9.4	5.2
Ext. Zn (µg / g)	0.8	0.2	1.0	0.9	1.2	0.3
NH3+NH4 (µg / g)	0.7	0.3	2.3	1.0	0.6	0.4
N-NO3 (µg / g)	0.9	0.7	1.2	1.3	0.8	1.1
pH	6.0	0.1	5.6	0.1	6.0	0.0
Ext. P (µg / g)	13.3	3.7	8.3	5.0	7.8	4.1
Bulk Density (g/cm^3)	0.2	0.1	0.2	0.1	0.3	0.1

Conclusions:

Loss of southern wild rice stands was attributed mostly to invasive cattails but the possibility of additional remediation projects in Ontario for this species is unlikely. Northern wild rice, on the other hand, is a highly prized crop and intense interest in the species ensures its continued cultivation. Existing traditional stands are currently being reduced in size by invasive cattails. Underwater cutting of cattails prior to ice cover is a highly effective control practice and a method of stopping the extirpation of northern wild rice. Use of invasive cattails for remediation projects is inadvisable near wild rice areas.

Literature Cited:

Aiken, S.G., Lee, P.F., Punter, D., and J.M. Stewart. 1989. Wild rice in Canada. NC Press, Toronto. 130 pp.

Grace, J.B. and J.S. Harrison. 1986. The biology of Canadian weeds. 73. *Typha latifolia* L., *Typha angustifolia* L., and *Typha x glauca* Godr. Can. J. Plant Sci. 66: 361-379.

Hotchkiss, N. and H.L. Dozier. 1949. Taxonomy and distribution of North American cattails. Am. Midl. Nat. 41: 237-254.

Lee, P.F. 2015. Solving rice's future now! Prepared for Seine River First Nation. 97 pp.

Lee, P.F. 2005. Reintroduction of southern wild rice into Cootes Paradise. Prepared for Environment Canada Great Lakes Sustainability Fund. 39 pp.

Sale, P.J.M. and R.G. Wetzel. 1983. Growth and metabolism of *Typha* species in relation to cutting treatments. Aquatic Botany 15: 321-334.

VEGETATION RECOVERY ON ABANDONED BORROW PITS

Marie-Ève Marin[1,3], Sandrine Hugron[1,], Stéphane Boudreau[2,3] & Line Rochefort[1,3]
[1]Peatland Ecology Research Group, Université Laval, Qc
[2]Département de biologie, Université Laval, Qc
[3]Centre for Northern Studies, Université Laval, Qc

Key Words: road construction, mineral substrate, restoration

The formation of borrow pits results from the extraction of sand and gravel needed for road construction. The primary colonization of this inorganic substrate can be very slow, specially in stressful environments, as northern and alpine environments. Several factors, such as instability, poor water and nutrient retention and lack of propagules, limit the establishment of plants on abandoned borrow pits.

The research project presented aimed to develop solutions for the restoration of disturbed environments where mineral substrate is exposed by using native plants. Two approaches were considered. In the first place, an experimental approach allowed to test the effect of various substrate on the establishment of 3 typical vascular plants of the boreal forest, such as the glandular birch, the black spruce and the common Labrador tea. The different types of substrate tested were : (1) bare soil (2) amendment of peat, (3) glandular birch branch mulch, (4) fragmented community of *Racomitrium canescens* and *Stereocaulon paschale* and (5) dense naturally established communities of mosses and/or lichens. Preliminary results showed that the black spruce and the Labrador tea showed a better survival rate than the glandular birch, independently of the type of substrate on which it was grown. The preliminary results, taken after one growing season suggested that black spruce and Labrador tea would be good candidates for the restoration of disturbed environments where mineral soil is exposed, such as borrow pits. Among the various types of substrate tested, a branch mulch seamed to favor the establishment of transplanted seedlings on borrow pits by modifying the microclimatic conditions at the interface air-soil (as shown on figure 1), while fragmented community of mosses and lichens seems to favour seeds germination (as shown on figure 2). Those data were collected after one growing season. The results of vascular plants introduction after one year will be shown during the presentation. They were not available by the time this abstract was written.

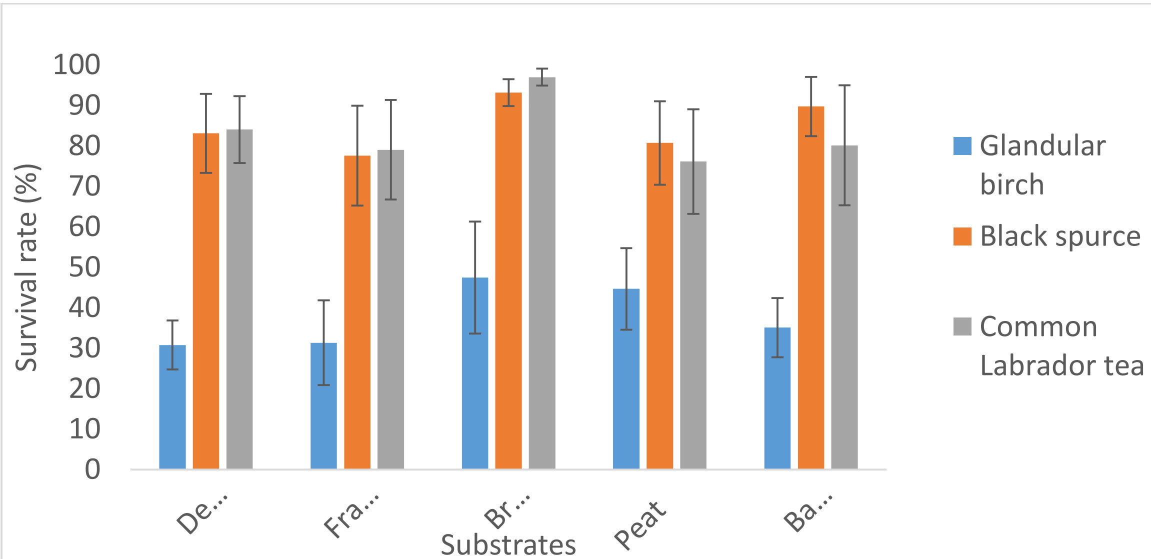


Fig. 1 : Survival rate of introduced seedlings on borrow pits after one growing season according to the substrate in which they were planted.

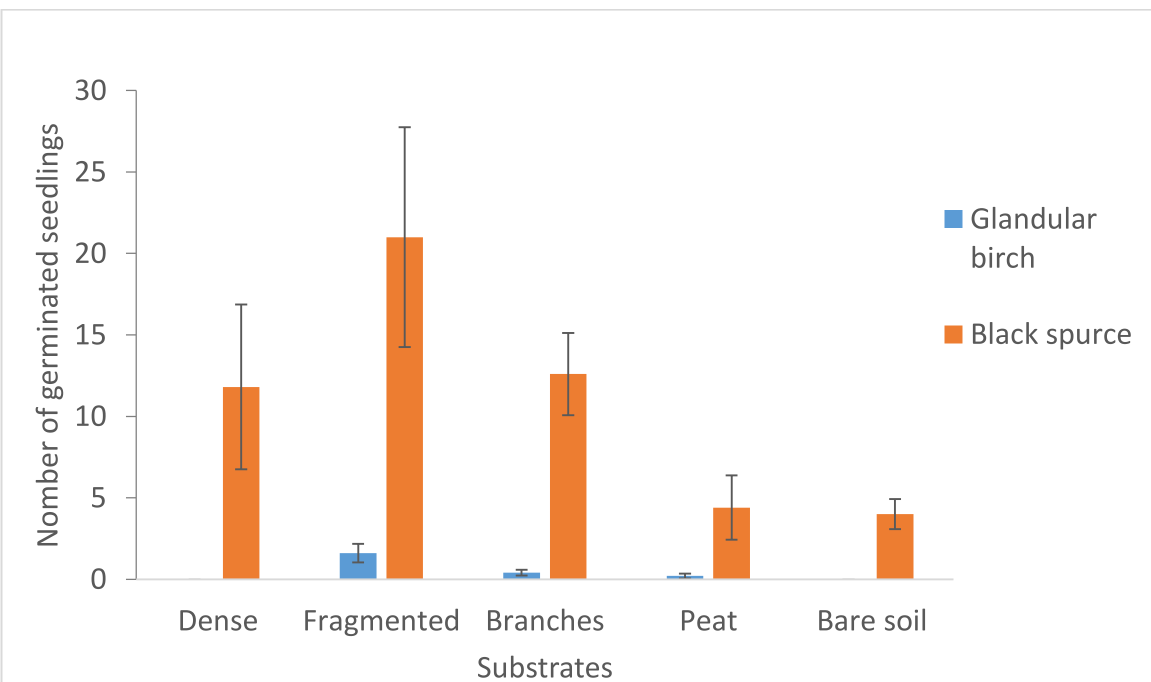


Fig. 2 : Quantity of germinated seeds on borrow pits after one growing season accordind to the substrate they were introduced in.

Secondly, plant establishment following restoration by the transfer of the organic substrate (including plant roots, seeds and aerial parts) on mineral roadsides was evaluated. The figure 3 illustrate the restoration technique. The results showed a fast vegetation recovery. The evaluation of several sites, allowed suggesting various options for the optimization of this technique.

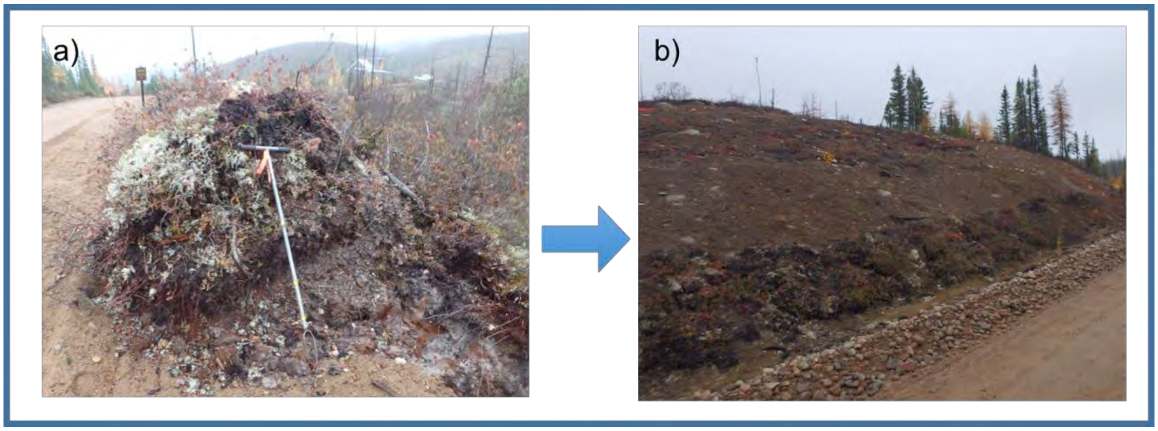


Fig. 3 : Illustration of the transfer of the organic substrate technique used to restore borrow pits and mineral road side. a) The pile of material conserved for the restoration. b) Organic material once spread on the road side.

SEASONAL AND OPERATIONALLY LINKED SULFUR GEOCHEMICAL CHARACTERIZATION OF A MINE OXIDATION RESERVOIR AND ITS INPUTS

Stephanie Marshall[1,2], Tara Colenbrander Nelson[1], Darla Bennett[1], Joe Fyfe[2] and Lesley A. Warren[1]

[1]McMaster University, School of Geography and Earth Sciences, Hamilton, ON, L8S4L8

[2]Sudbury Integrated Nickel Operations, A Glencore Company, Onaping, ON, P0M2R0

Key Words: Geochemistry, thiosalts, sulfur, mining, water, tailings, waste rock

The presence of sulfur oxidation intermediates (SOIs; e.g. $S_2O_3^{2-}$, SO_3^-, S^0, etc.) in mining wastewater is a significant industry-wide concern because of the potential for acidification and toxicity of receiving water bodies, resistance to treatment and the current lack of monitoring tools. Bacterially catalyzed transformation of sulfur species under real world conditions remains poorly constrained, limiting the development of biologically integrated indicators. As part of a larger project investigating the microbial links to mine water S chemistry for the water system of Ni, Cu mines in Northern Ontario, the objective of this investigation is to characterize the *in situ* S geochemistry of Glencore's Oxidation Reservoir (Onaping, ON) over seasonal, operational and annual scales. As the Reservoir receives diverse water inputs from three mining companies, waste rock, tailings lines and multiple watersheds, these are also being assessed over seasonal scales to establish their contributions to observed Reservoir geochemistry.

To date, water column survey data (pH, temperature, dissolved oxygen, conductivity, and oxidation-reduction potential) and samples (minimum of two depths per campaign), have been collected from the Reservoir and input sources at seven time points between September 2014 and May 2016 (Figure 1). In addition, two sampling campaigns in July 2015 bracketed effects related to mill production changes, i.e. the reduction and elimination of tailings discharge, may have had on the S geochemistry of the Oxidation Reservoir. For each water sample microbial bulk community structure (functional metagenomics - Illumina HiSeq), and geochemical characterization (S: H_2S, SOIs, SO_4^{2-}, total S; Organic carbon: TOC, DOC; Fe: Fe^{2+}, Fe^{3+}, total Fe; and N: NH_4^+, NO_2^-, NO_3^-) were carried out. The techniques used to gather such data include ICP-AES for total sulfur, HPLC for SOIs, ferrozine method for iron analyses, TOC analyzer for organic carbon, and colorimetric spectrophotometry for determination of the remaining species of sulfur and nitrogen.

Figure 1. Sample locations within the Strathcona Tailings Management System. Inputs (yellow stars) and the Oxidation Reservoir (white stars) denote the specific sampling locations.

The environmental survey data and associated cycling of sulfur, carbon, and nitrogen have been highly variable over space and time for both the inputs and the Reservoir (Figure 2). Preliminary analysis highlights the importance of inputs in observed Reservoir geochemical characteristics. Sulfur geochemical results indicate the widespread and often dominant presence of colloidal elemental sulfur (S^0) as a key SOI species (Warren et al. 2016). Collectively results indicate that Oxidation Reservoir S geochemistry will reflect input, seasonal, operational and microbial influences.

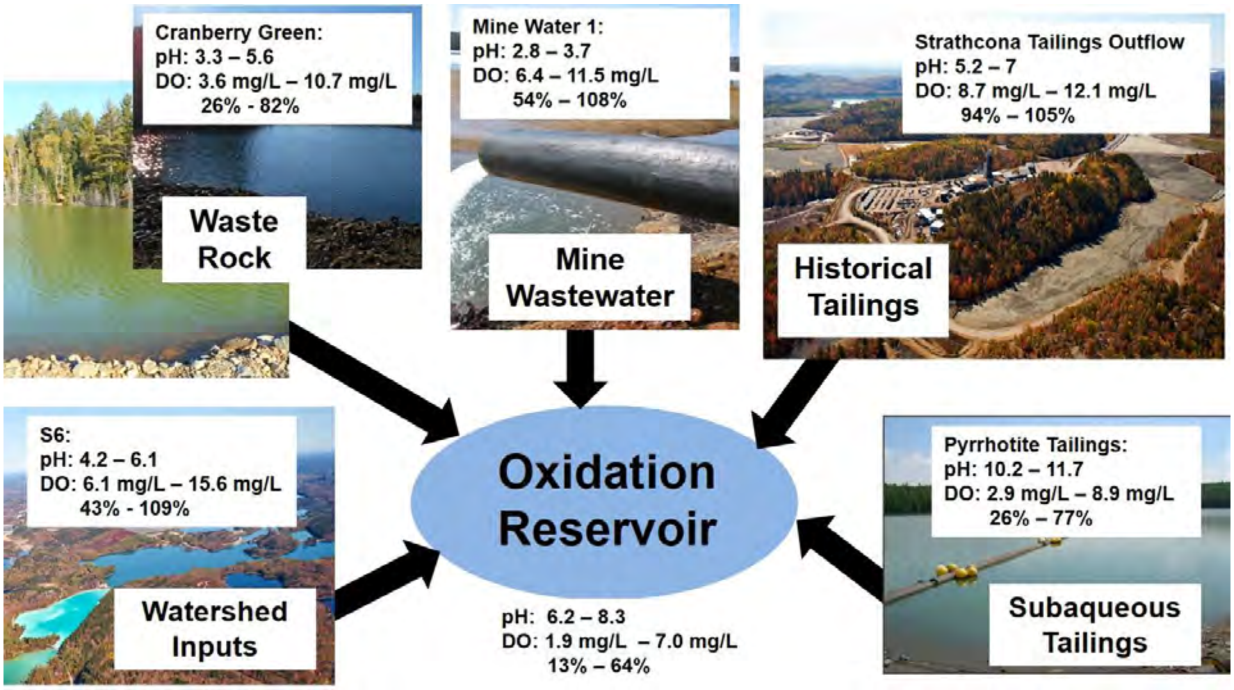


Figure 2. Spatial and seasonal variability of pH and dissolved oxygen (DO) of selected inputs and the Oxidation Reservoir from nine sample points during September 2014 to May 2016.

These results will establish the importance of the various inputs as well as climatic and operational variability in overall S geochemistry in this system. Their integration with microbial and experimental results will generate new understanding of the potential importance, roles and outcomes of bacterial activity on water quality.

Acknowledgements

Thank you to Joe Fyfe and Dr. Lesley A. Warren for the ongoing technical guidance and support. Funding for this project has been provided by Glencore, McMaster University, Ontario Genomics, and NSERC. Special thanks to Tara Colenbrander Nelson, Glencore technicians, Dr. Warren, and McMaster's Aquatic Microbial Geochemistry group for their valued contributions towards the field work and analyses.

Literature Cited

Warren, L.A., Nelson, T.C., Marshall, S., Bennett, D., and S. Apte. Sulphur geochemistry of mining wastewaters. *Minerals*. Forthcoming 2016.

Nanoflotation - using EDL collapse to increase the efficiency of flotation and submerged membrane filtration.

McEachern, Preston & David Bromley. Purlucid Treatment Solutions Ltd., 6312 Longmoor Way SW, Calgary AB, T3E5Z7 (587) 580-5096

Extended Abstract:
With most industrial water and wastewater treatment requirements, the objectives for treatment are relatively consistent. Typically they are separation and removal of colloidal particles, removal of free oil and grease and reduction in the parameters that either cause scaling when water is reused or injected in disposal wells or have an effect when released to the environment. We will review water treatment challenges for oil and metal mining industries and provide a case study that deploys flotation followed by ultrafiltration as a treatment solution. In this case study we demonstrate removal of suspended and colloidal solids to 0.01 µm in size at efficiencies that are 10-times the industry norm and at costs that make in-line treatment of tailings economically achievable. The case study will be used to review the science behind microflocculation through use of electric double layer (EDL) compression (the DLVO theory) and how this can enhance traditional coagulation using charge neutralization. Colloidal solids, no matter their charge, will repel each other because of the EDL on each solid. However once the solid enters a highly ionized/charged environment, the EDL collapses and van der waal attraction forces take over such that the solids attach to each other. Individual charge on the solids, in both cases, has no bearing on whether a solid repels or attracts one another. The use of microflocculation requires a novel design that can provide fixed microzones of high ionic charge and a reliable mechanism to capture the small colloids that are rapidly formed in these microzones; membrane treatment partially provides for these needs. Loss of flux rates due to fouling, the nemesis of membrane treatment, had to be solved to take advantage of microflocculation, a challenge overcome by the Nanoflotation system reviewed in this case study. Solids removed from the water can immediately be placed as reclamation fill as long as they pass soil criteria.

Flotation

Flotation has been used extensively in oilsands and metal mining for primary resource recovery. In oilsands, it represents the first stages of oil recovery and subsequent oil-water separation in the treatment of water for reuse. In metal mining it is the most widely used process for ore beneficiation. Flotation systems take on many forms, the simplest of which are tanks with or without settling plates (clarifiers) and a froth skimming system. Loading rates are expressed as the flow rate (m3/hr) divided by the tank cross sectional area (m2) and expressed as meters/hr (m/hr). Typical design loading rates range from 1-2 m/hr for simple clarifiers to 5-10 m/hr for dissolved air flotation systems. High rate DAFs where bubble density and attachment properties are enhanced with frothers is enhanced have more than doubled loading rates to 10- 20 m/hr. Despite these improvements, typical design rates for industrial waste waters remain 5 to 10 m/hr. Nanoflotation improves on these industrial design rates by allowing for designs in the range of 10 -20 m/hr using a surfactant-collector combination and a patent pending high intensity mixing system to enhance bubble surface attachment and rapid froth rise rates.

We tuned the surfactant-collector chemistry for optimal organic and total suspended solids (TSS) removal in a one-year trial treating oilsands tailings waters. The TSS in these waters consists of stable “gel” suspensions of primarily small (< 5 μm) aluminosilicate clays. Better than 97% removal of TSS was achieved in flotation consistently over a 4 month 24-7 run period at dosing rates of 0.028 mL/L of feed water for lower turbidity waters (<100 mg/L TSS) to 0.25 mL/L for higher turbidity conditions (900 mg/L TSS). Additional trials on water containing high concentrations of primarily colloidal silica (20,000 mg/L) did not demonstrate the same performance in the flotation system due to the low ionic charge of the silica colloids. These colloids were, however, efficiently removed in the subsequent ultrafiltration stage (see below).

Bubble density and size influence froth separation efficiency. Typical high rate DAF systems produce bubble densities of 1% and diameters < 0.1 μm. In Nanoflotation we focus on generating high bubble concentrations of 3% and larger bubble sizes in the 1 to 2 μm diameter range. In Nanoflotation, we achieve three features critical to efficiency; high bubble surface area, a highly charged bubble surface and larger bubbles. The charged surface of the bubble causes EDL collapse and better surface attachment while larger bubbles increase the rise rate to values exceeding 16 cm/s. A doubling of bubble size in this range (up to 2 μm) results in 3 to 4 times greater rise rates, with greater efficiencies achieved at higher temperatures.

Membrane filtration

Membrane filtration has not been used commercially in oilsands solids removal and dewatering primarily due to the high capital cost and an inability to control fouling. In metal mining, filtration is common in dewatering and water treatment. In dewatering, filter presses, belt presses and other non-submerged systems are common whereas in water treatment, submerged membrane filtration is increasingly more common.

Membrane strength, longevity and fouling negatively impact submerged membrane applications. This is particularly the case in oilsands applications where high organic and scale forming ion concentrations seriously curtail performance. Advances in membrane technology to overcome these issues have focused on a fixed precoat that both protects the membrane and repels charged ions that would otherwise foul the membrane and skin layer. The electorstatic repulsion is countered by the trans-membrane pressure (TMP) driving the solids and ions into the skin layer surface. When TMP exceeds a critical threshold (~1 bar), fouling rates can increase substantially.

Cross flow, where a volume of water is passed tangentially over the skin layer surface, is extensively deployed in conventional membrane systems to reduce fouling by removing accumulated solids. Cross-flow systems must have substantially higher tangential flow compared to the flow through the membrane such that pumping flow exceeds permeate (treated) water production by 10 to 20 times or more. In addition to high pumping volumes, cross-flow systems eventually foul.. Once the membrane skin layer fouls in situ cleaning using highly acidic and caustic solutions is required to recover flux rates resulting in reduced longevity and in some cases delamination of the skin layer and failure of the membranes themselves.

An alternate approach is to promote fouling of a temporary filter barrier similar to a skin layer with subsequent replacement *in situ*. These “dynamic membrane” systems use an inert material such as diatomaceous earth and are extensively used in removal of unreactive solids as a mud cake. Nanoflotation improves on the dynamic membrane concept with the patented replaceable skin layer (RSL) design (Fig. 1). We apply a precoat skin layer *in situ,* that creates highly charged microzones in the pore spaces. The skin layer is allowed to foul over an extended period and removed once TMP approaches 1 bar. The RSL technology allows for specific precoat chemistries that promote the agglomeration of ions and small colloidal particles within the precoat material. One formulation of the RSL targets the removal of fine tailings clays in the highly charged pore spaces within the precoat through EDL compression as proposed by the DVLO theory.

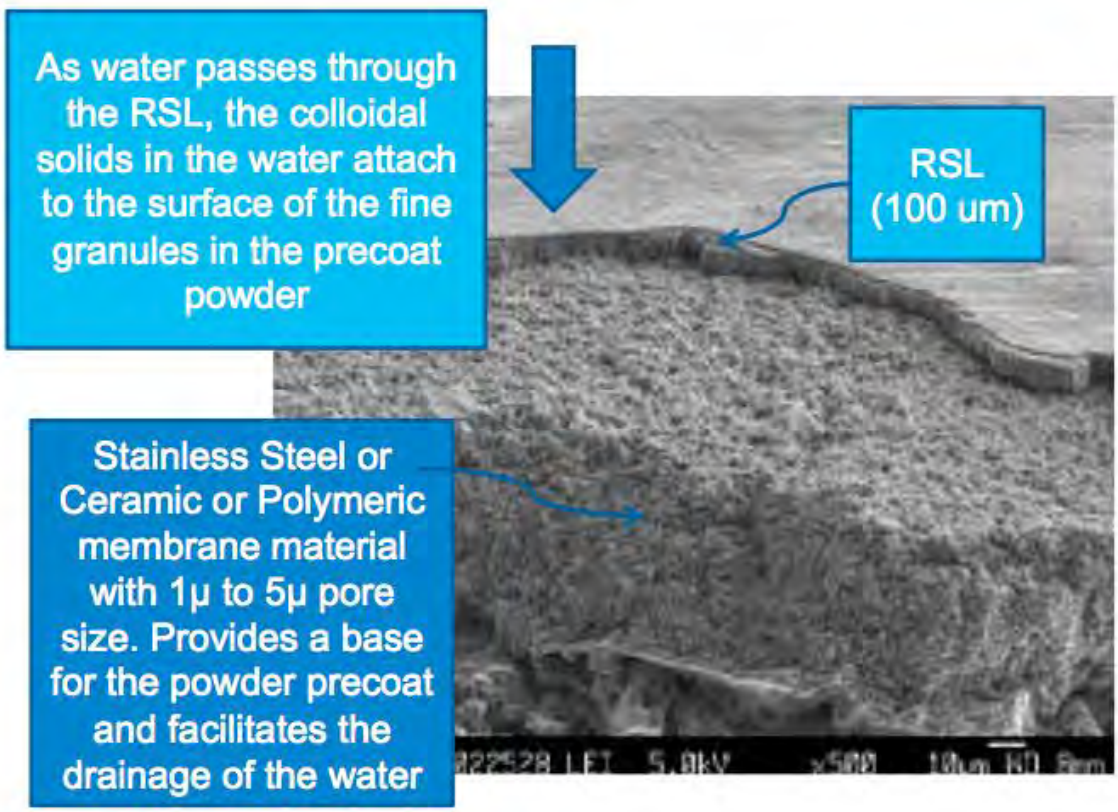


Fig. 1: Electron micrograph of the Nanoflotation RSL able to capture colloids >0.01 µm in size on a stainless steel membrane with a pore size of 1 µm.

The RSL patented technology has been deployed in various oilsands applications including tailings water treatment. In tailings treatment, stable suspensions of ultrafine clay were efficiently removed (Fig. 2) producing similar post treatment water quality of other leading ultrafiltration technologies but at a fraction of the capital cost and footprint.

For example, TSS removal averaged 100% over a four-month continuous treatment operation but was achieved with 10% of the membrane area, no cross-flow requirement, and 30-times fewer backwash cycles when run head-to-head with conventional systems.

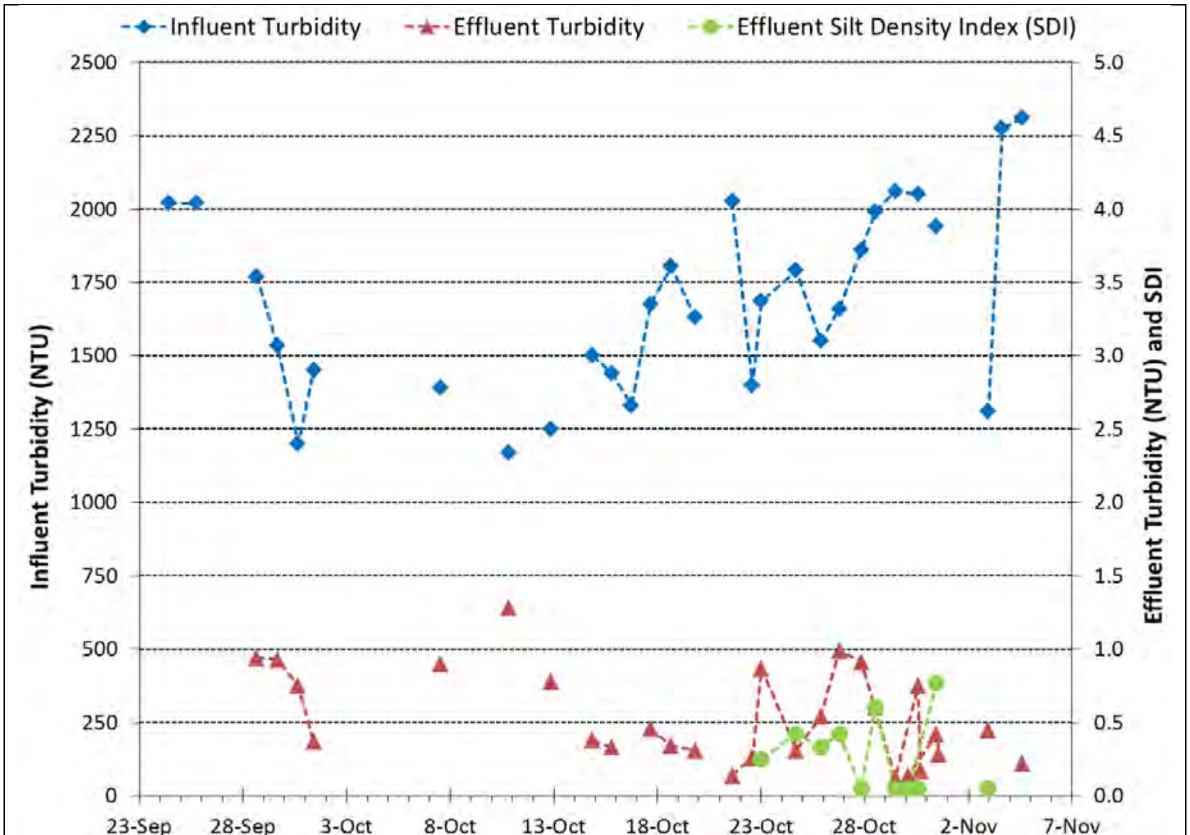


Fig. 2: Example of feed and permeate water quality during continuous treatment of oilsands mine tailings release water after the Nanoflotation RSL.

When applied to highly concentrated brines such as those from produced water and evaporative water treatment, the RSL membranes performed well with nearly 100% removal of dissolved silica, dissolved iron and colloidal particles including polymerized and unreactive silica (Table 1). For this to occur, it is necessary to convert the dissolved parameters to colloidal or suspended solids using targeted chemical manipulation which can be as simple as pH adjustment.

Table 1: Example performance of Nanoflotation treatment of a highly concentrated brine wastewater.

Scale Parameter	EBD/SAC feed concentration (mg/L)	NF Filtrate Concentration (mg/L)
Reactive Silica	13000	27.8
Total Barium	17.9	2
Total Strontium	73.9	51.4
Fe-Dis	2.2	0
Ca-Dis	1251	985.5
Mg-Dis	109	77.8

Table 2: Operational performance summary

Nanoflotation/ Replaceable Membrane Skin Layer vs Conventional Membrane Technology *Why is Nanoflotation better than conventional technology?*			Conventional Membranes	RSL Membranes
1. Very low material requirement per litre of water treated for membrane skin layer replacement			will add coagulant for pretreatment - 15 to 100 mg/l	Powder for precoat is 10 to 15 mg/l
2. Delta pressure (trans membrane pressure)-very low at 20 to 70 kpa (2 to 10 psi)		Note 3	No difference	No difference
3. Full H_2S operation if necessary			No difference	No difference
4. Can operate at high temperatures and pressure			Yes but high cost	Yes but low cost
5. Fully automated back wash and skin layer replacement			No	Yes
6. Back wash intervals			15 sec	6 to 15 hrs
7. Oil removal	99 to 99.9%		poor	excellent
8. Turbidity removal	99.9 to 99.99%		excellent	excellent
9. Average loading (flux) rate between backwashes (lph)	325 to 600	Note 1		10x higher
10. Flux recovery after b/w	98 to 99%		92%	99%
11. Max TSS application (mg/l)	150,000	Note 2	6000 mg/l	150.000 mg/l
12. Anticipated life of membrane	20 years		5 to 10 years	10 to 20 years
13. Pumping requirements vs conventional UF crossflow membranes	10%	Note 3	high energy	low energy 90% reduction
Note 1 Result: Significantly reduced footprint and membrane requirements				
Note 2 Result : Can be applied to sludges				
Note 3 Result : Very low energy requirements				

Summary

The Nanoflotation technology provides an effluent similar to other ultrafiltration membrane technologies but requires significantly less energy and capital costs (Table

2). In addition, the use of the RSL increases flux rates on the membrane significantly (10 to 15 times when it was compared against conventional ceramic membranes). The design allows for the use of heat and corrosive resistant membrane substrates such as stainless steel. With the significantly improved flux rates, the cost of using temperature resistant material results in an overall cost similar to or less than the typically low cost polymeric membranes that are limited to a temperature of 40^{o} C. The ability to treat at very high temperatures or in corrosive environments removes the need for energy intensive steps in the treatment process such as cooling or neutralization. For a facility that is dependent on steam generation, the Nanoflotation technology provides significant energy savings and correspondingly excellent CO_2 reductions.

PLANTS GROWING IN CLOSED MINES OF NORTHWESTERN ONTARIO ARE USEFUL FOR PHYTOSTABILIZATION POTENTIAL OF ARSENIC, ANTIMONY AND MOLYBDENUM IN MINE RECLAMATION

Christina M. Mol and Peter F. Lee

Lakehead University, Thunder Bay, Ontario, P7B 5E1

Key Words: Metals, Rhizosphere, Mining, Boreal, Bioavailability, Phytoremediation, Bioconcentration, Translocation, Contaminated Soils

Introduction

Northwestern Ontario is a boreal forest region on the Canadian Shield rich in mineral resources with active, closed and abandoned mines. These areas could be reclaimed with phytoremediation species to prevent wind and water erosion to reduce metals from entering the food chain and waterways (Salt *et al*., 1995; McIntyre, 2003; Pilon-Smits, 2005; Peer *et al*., 2005). Whereas organic contaminants can be degraded and reduced to less toxic components, metals are non-biodegradable, persist in waterways near the contaminated area, can endure within the soil for the unforeseeable future and be taken up by plants including agricultural crops and natural revegetation of the region (Ashraf *et al*., 2011; Nouri *et al*., 2008).

Naturally occurring plants on closed mine sites are best adapted for growth on soils with elevated metal composition (Salt, 1995). They are often highly tolerant of metals that remain directly from the mining processes that occurred in the area, via direct contribution from mining or indirectly from air, water and soil erosion (Freitas *et al*., 2004). Metallophytic plants have evolved to grow on elevated metal soils. They are useful for reclamation purposes of other industrial sites or to indicate potential ore bodies (Whiting *et al*., 2004). Ideal candidate species for rehabilitation are thriving in the elevated metal contaminated soils, produce high amounts of biomass and are well adapted to the local climate (Khan *et al*., 2000; Dickinson *et al*., 2009; Chen *et al*., 2012; Majumder and Jha, 2012).

Phytoremediation with native plants is a multifaceted approach using metallophytic species more suited to the local environment as compared to agricultural or introduced plantings (Oppelt, 2000; Pilon-Smits, 2005). Various phytoremediation strategies exist including phytoextraction and phytostabilization. Phytoextraction removes the contaminant from the soil profile and known as accumulators. If the concentration of the contaminant in the plant is extremely high in comparison to the concentration in the soil, these are known as hyperaccumulators (Kumar *et al*., 1995; Peer *et al*., 2006). Phytostabilizers do not uptake metals and other contaminants into the above ground tissues of the plant and called excluders (Wong, 2003). Phtyostabilization prevents the elevated metal contents of the soil from spreading into the food chain via plant uptake and helps contain the contamination to the affected area (Mendez and Maier, 2008). Metal pollution on these mine sites is directly influenced by the type of deposit mined, so phytostabilizers need to cater to each type of circumstance. Absence and/or excess of various metals and nutrients can affect the success of each type of plant as well as other characteristics such as lifespan, size, root systems, and predation. Identification of specific plant species can cater to these contaminated areas (Tordoff *et al*., 2000; Mendez and Maier, 2008).

This paper explores the phytostabilization potential of plants naturally regenerating on closed mines in Northwestern Ontario. Data will be presented on their associated soils including pH and the various metal concentrations. The sites chosen for this study are mines that operated in the 20th century in Northwestern Ontario, located on the Canadian Shield and are in the Lake Superior and Seine River Watersheds. Minerals mined varied but two mines had elevated levels of As, Mo and Sb. Remediation of these sites would help to meet government environmental requirements as well as reduce the impact on the local populations of humans and wildlife. Surface soil contaminants from these sites have the potential to runoff into surface waters, leach into ground waters, and negatively impact the local food chains. A combination of native boreal species can be planted on closed mines in order to restrain the movement of these potentially harmful metals. Chlorophyll content of the leaves are measured as a sign of plant health and can aid in identification of healthy metallophytes.

The objectives of the study were i) to identify plants growing on these closed mines, and ii) to examine the variation in the soil conditions natural regenerating species experience on these closed mines.. Plant communities will be found that are tolerant to metal stress.

Materials and Methods

Site Description

This study examines three closed mines areas in Northwestern Ontario with no replanting of plants or trees following closure. Located on the Canadian Shield, these mines sites sampled have been closed or not replanted for a minimum of 25 years and had restricted road access.

Steep Rock Iron Mines, encompassing just over 100 square kilometers, are located near Atikokan, ON and operated as a source of high grade hematite from goethite-hematite deposits for 30 years from the 1950s to the early 1980s (Ontario Geological Survey *et al*., 1972). The Winston Lake mine near

Schreiber, ON which produced primarily zinc, silver and copper with secondary amounts of gold began in 1988 and ended in the late 1990s (LaFrance *et al*., 2004). The third area consists of several properties managed by Premier Gold located in the Beardmore/Geraldton greenstone Belt: Northern Empire and Leitch mines. Northern Empire was and underground operation producing gold from 1934 to 1941 with other exploration occurring since that time period. Leitch was mined for gold from 1936 to 1968 (GEDC, 2005).

Field and Sample Preparation

Thirty metre long transects were placed on the mined rock piles, tailings areas and former building locations to determine naturally regenerating plant communities on the closed mines. On these transects samples were collected as follows: 15 1 m^2 quadrats of herbaceous plants, 6 5 m^2 quadrats of shrubs and 3 10 m^2quadrats of trees (Bagatto and Shorthouse 1999). Plants used for metal analysis in this investigation were identified and sampled based on abundance, amount of biomass in root and shoot tissue, healthy leaf colour and active growth on the sites. Chlorophyll content was used as a method to determine health status in their mine environment (Walters 2005). Chlorophyll content of species were obtained using the CCM-300 Chlorophyll content meter using an average of 3 readings per measurement (Gitelson, 1999).

Plant and soil samples were taken along these transects: 3 soil samples per transect and three plant samples per species. Plant samples were identified following local plant identification guides and verified at the Lakehead University Herbarium. The foliar samples were rinsed with distilled water, air-dried at room temperature for several weeks, and the samples were ground to a homogeneous powder. Analysis of metals in the plants encompassed all aboveground plant material in late August including twigs, leaves or needles, and flowers. Soil samples were collected in the rhizosphere of the plant, not always at the same depth due to plant type and variations in soil depth. Plants were dug out of the ground and shaken over a bag for the soil. Soils were air-dried and sieved with a 2mm mesh to remove plant matter and rocks.

Laboratory analysis

Analyses were done at the Lakehead University Environmental Laboratory (LUEL) according to the LUEL (2012) Quality Assurance/Quality Control (QA/QC) protocols. Soils and plants were analyzed for moisture,pH, conductivity, organic matter and total metal concentrations of Al, As, Ba, Be, Ca, Cd, Co, Cr, Cu, Fe, K, Mg, Mo, Na, Ni, P, Pb, Sb, Sr, Ti, Va, and Zn.

Statistical Analysis

Statistical analysis were performed using SPSS 23 package for Windows as follows:

i. For the transect data from the plant populations, herb, shrub and tree data were summarized using species richness (mean number of species and identity per mine) and the density (mean number of stems per mine)

(Magurran 2013). Differences in species richness and density at all stand levels among the three mines were examined using ANOVA.

ii. Plant species data were analyzed using ordination and classification (non-metric multi-dimensional scaling (NMS) and cluster analysis), to identify species with similar habitats. Vegetation data were screened for outliers, normality, and heteroscedasticity (McCune, 1996). The PROXSCAL algorithm with a Torgerson start and Chi-square measure for count data was applied because it allows similarity matrices to be used. Cluster analysis using Ward's linkage and Squared Euclidean distance was performed to confirm separation of the species.

iii. For the mine soils, mean and standard deviation of metals were determined for each mine. Data was log transformed to curtail skewness. Discriminant function analysis on the soil characteristics was performed. Statistical significance was defined as $P < 0.05$.

Results

Plant Characterization

Sampling at the mines resulted in a collection of 36 plant species, from 31 genera and 14 families, with richness and density data shown in

Table 1. Winston Lake and Premier had more plant species and a larger cover but less trees than Steeprock, which had an even richness of herbs, shrubs and trees. None of the species were found at all of the transect sites but willow (*Salix* spp. L), white birch (*Betula papyrifea* Marshall*),* goldenrod *(Solidago canadensis* L*),* hawkweed (*Hieracium canadense* Michx*), and* trembling aspen (*Populus tremuloides* Michx) were found at all three mining areas. Pearly everlasting (*Anapahlis margaritacea* L*)* was found solely at Winston Lake mine. At Steep Rock Mine, there are either older trees and very little understory with very few herbaceous species or areas with no trees, some shrubs and sparse herbs. Soils and surrounding water have vivid multicoloured hues with very low populations of unhealthy herbs. Many of these trees are seen with fungal mycorrhizae to aid in their growth. Sites investigated at Premier's properties showed stunted shrub-like trees and some herbaceous species but no large overstory. Winston Lake had no trees with some areas with shrubs and intermittent herbs.

Table 1 Mean number of herbaceous, shrub, and tree species (richness) and mean values of stand structure characteristics in three mines in northwestern Ontario*

	Herbs		Shrubs		Trees	
	Richness	Density (stems/m^2)	Richness	Density (stems/100 m^2)	Richness	Density (stems/ha)
Premier	4.7 $_a$	47.6 $_a$	2.4 $_a$	207.5 $_a$	0.0 $_a$	0.0 $_a$
Steep Rock	1.5 $_b$	12.3 $_b$	1.1 $_b$	183.9 $_a$	1.4 $_b$	10888.9 $_b$
Winston Lake	3.6 $_c$	42.1 $_a$	2.3 $_a$	281.1 $_a$	0.0 $_a$	0.0 $_a$

*Values within the rows with the same letters (a, b, and c) are not significantly different at the P <0.05 level.

The proximity values for the plant species at the three mines are represented by a two-dimensional NMS map based on the resulting raw stress of 0.05007 and a Stress-I value of 0.22377 (**Error! Reference source not found.**). The stress values reflect how well the solution summarizes the distances between the data so a low stress value shows a good fit ordination. Cluster analysis was run for 39 cycles to also determine the groups of plant species and compare to the NMS results (Figure 2). With a line drawn at the 6 distance in the cluster analysis, the majority of the plant species are separated in two clusters as well as outliers of the grass species of false melic grass (*Schizachne purpurascens* Torr.), and horsetail (*Equisetum* spp.) The next group of species features birdsfoot trefoil (*Lotus corniculatus* L), hawkweed (*Hieracium canadense* Michx), as well as raspberry (*Rubus idaeus* L), pearly everlasting and the shrub height of white birch. The last cluster is the remaining plant species, which can also be seen within the circle of the NMS diagram as seen in **Error! Reference source not found.**: white pine (*Pinus strobus* L), white spruce (*Picea glauca* Moench), willow, white birch, trembling aspen, red pine (*Pinus resinosa* Aiton), balsam fir (*Abies balsamea* (L) Mill), heart leaved aster (*Symphyotrichum cordifolium* L), fireweed (*Epilobium angustifolium* L), jack pine (*Pinus banksiana* Lamb.), blueberry (*Vaccinium angustifolium* Aiton), tamarak (*Larix laricinia* Michx), dandelion (*Taraxacum officinale* FH Wigg), yarrow (*Achillea millefolium* L), sedge (*Carex brunnescens* Pers.), daisy (*Leucanthemum vulgare* Lam), balsam poplar (*Populus balsamifera* L) , strawberry (*Fragaria vesca* L), cedar (*Thuja occidentalis* L), red clover (*Trifolium pratense* L), bladder campion (*Silene vulgaris* Poir.), goldenrod (*Solidago canadensis* L), and primrose (*Oenothera biennis* L).

Chlorophyll concentrations in vegetation at the mines are shown by Table 2. Some of the plant species with leaves that were too chlorotic to give a chlorophyll content via the CCM-300 meter. Chlorophyll concentrations of the leaves ranged from 166 mg/m^2 to 718 mg/m^2. Higher concentrations of chlorophyll were found at Winston Lake where no chlorosis was evident. Many leaves on the plant species at Premier had chlorophyll levels below detection for the meter but the plants that did give readings were higher values than the plants found at Steep Rock, except for white birch and red clover.

Soil Characterization

A summary of the soil analysis of each of the studied mines is given in

Table 3. Metals with elevated concentrations in the soil samples were As, Mo and Sb at Steep Rock at levels of 320.65 mg As kg^{-1}, 4.46 mg Mo kg^{-1} and 674.91 mg Sb kg^{-1} respectively and Premier sites at levels of 2245.36 mg As kg^{-}, 7.74 mg Mo kg^{-1} and 1472.3 mg Sb kg^{-1} respectively. Winston Lake had elevated amounts of zinc at 787.61 mg Zn kg^{-1} due to ore mined at the site. The pH at the mines varied from slightly basic at Premier with a pH of 7.9 to Winston Lake with 6.8 and Steep Rock with the more acidic conditions at 5.86. Each location showed similar bulk density ranging from 0.87 at Premier to 0.91 at Winston Lake and 0.93 at Steep Rock.

All soils chemistry data was used in the discriminant function analysis that classified 100% of the samples collected correctly (Figure 3). Function 1 explained 72.8% of the and function 2 explained 27.2%. Function 1 could be interpreted as the ratio of Fe (negative coefficient) to P, As, V, Pb, and Co (positive coefficient). Function 2 has Fe, Ca, Mn and K as the positive coefficients and Mg, Cr and Co as the negative coefficients. Each of the mines is completely separated with different soil characteristics and so plants found at all three locations are possible universal candidates for rehabilitation.

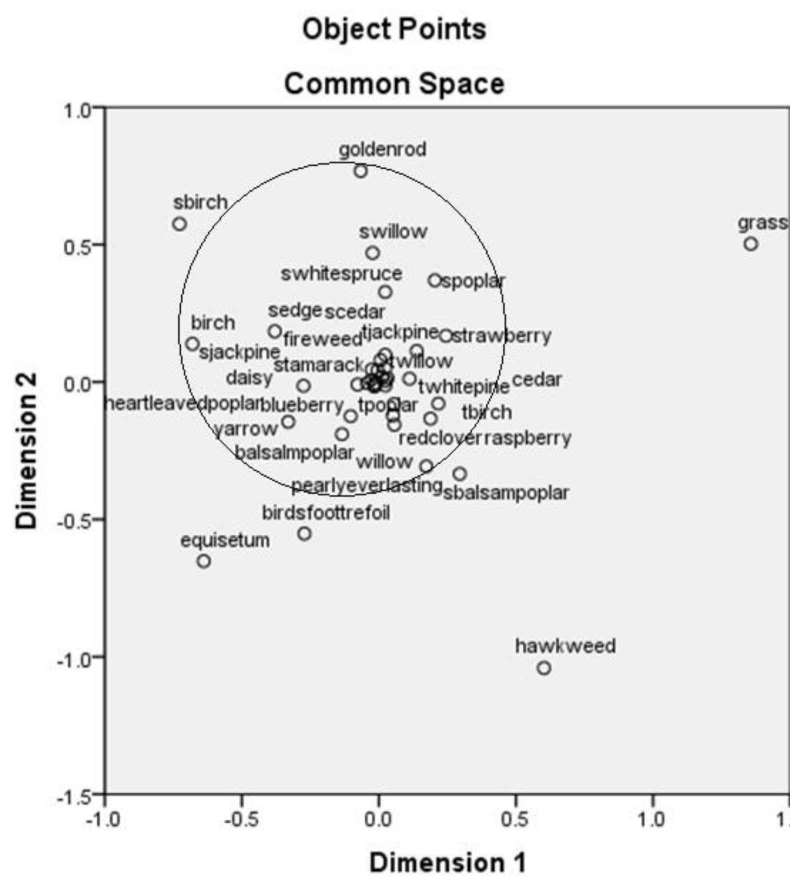


Figure 1 A two-dimensional ordination plot derived from non-metric multi-dimensional scaling (NMS) of 13 transects using herbaceous species composition and abundance data.

Table 2 Chlorophyll Concentrations of various plant species found at mine locations in Northwestern Ontario

			Mine Location					
			Premier		Steep Rock		Winston Lake	
			Concentration (mg/m^2)					
			Mean	Standard Deviation	Mean	Standard Deviation	Mean	Standard Deviation
Species	Balsam poplar	*Populus balsamifera*	535.7	116.3				
	White birch	*Betula papyrifera*	330.0	134.4	453.5	79.3	569.3	226.3
	Birdsfoot trefoil	*Lotus corniculatus*			479.7	76.6	349.0	215.0
	Blue spruce	*Picea pungens*					386.0	0.0
	Cedar	*Thuja occidentalis*	289.0		250.0			
	Goldenrod	*Solidago canadensis*	471.0		316.0	114.6	440.0	79.7
	Horsetail	*Equisetum* spp	166.0					
	Pearly everlasting	*Anaphalis margaritacea*					339.3	119.5
	Trembling aspen	*Populus tremuloides*			648.3	218.7	667.0	173.4
	Evening primrose	*Oenothera biennis*					244.0	
	Red clover	*Trifolium pratense*	278.0		538.5	47.4		
	Red pine	*Pinus resinosa*			300.5	68.6		
	Sedge	*Carex gynocrates*					347.0	0.0
	White pine	*Pinus strobus*			432.5	57.3		
	White spruce	*Picea glauca*	318.0		265.0	65.0	389.0	92.2
	Wild strawberry	*Fragaria vesca*			560.0			
	Willow	*Salix* spp.	540.8	149.8	405.3	130.8	718.0	174.4
	Yarrow	*Achillea millefolium*					351.5	111.0

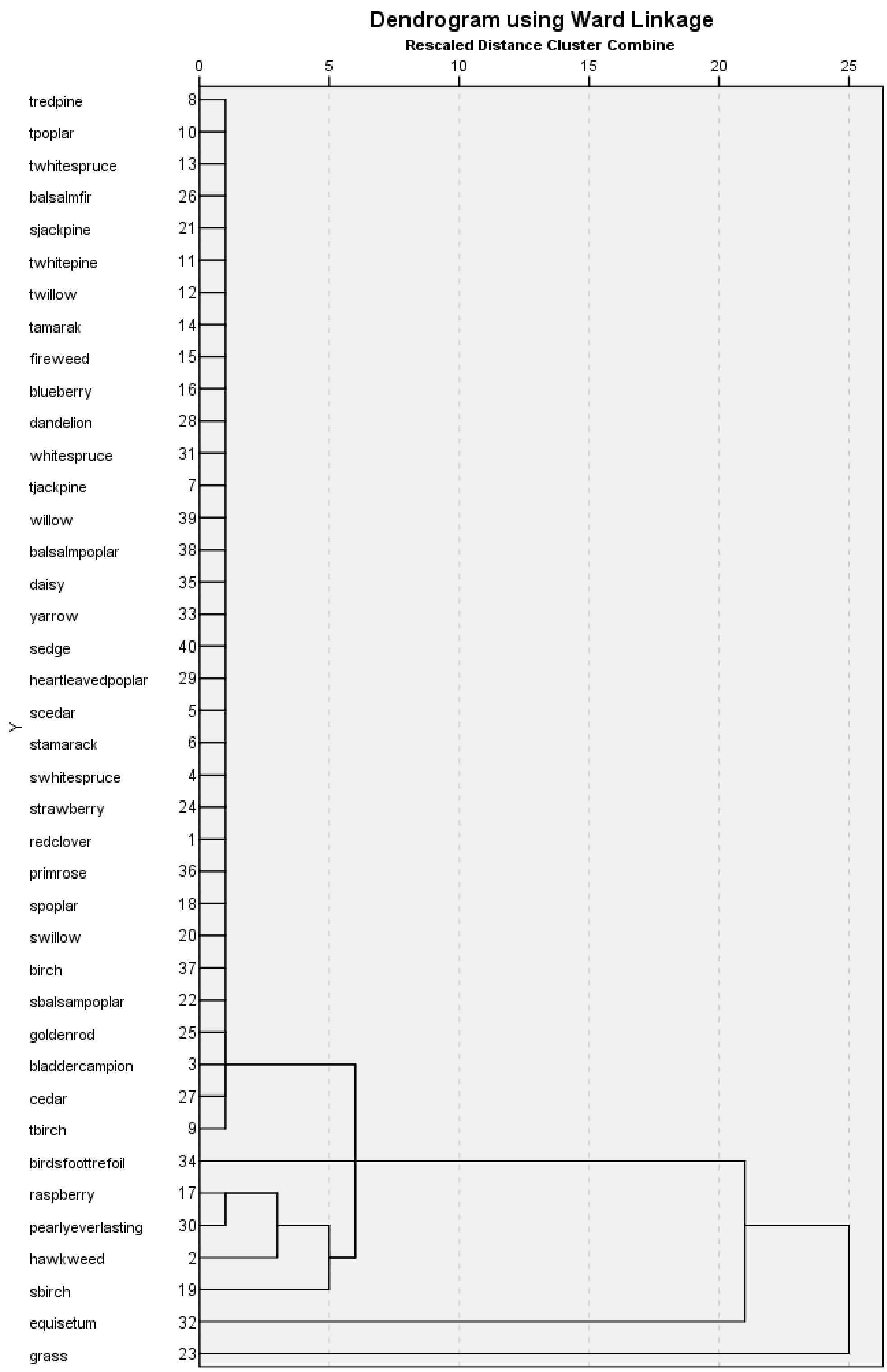


Figure 2 Hierarchical cluster analysis of 13 transects using herbaceous species composition and abundance data.

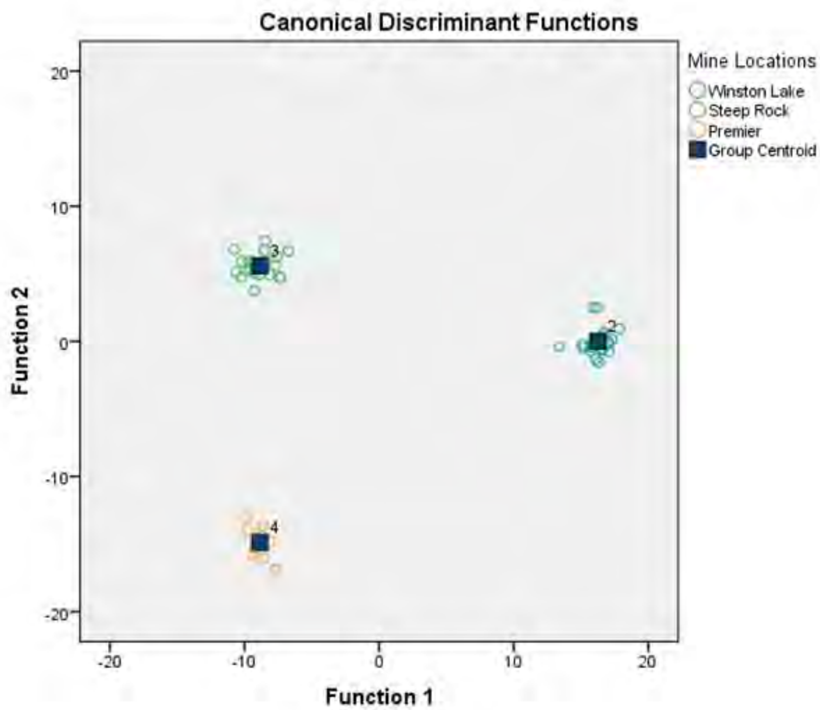


Figure 3 Discriminant Function Analysis Plot with all the soil variables at the mine locations. Standardized discriminating function 1 Fe -4.08, Ni - 1.70, K -1.38, Cr -1.29, Co and Pb +1.85, V +2.27, As, +2.28, P +2.34. Function 2 Mg -0.94, Cr -0.92, Co -0.84

Table 3 Soil chemistry characteristics of the studied areas on the three mines (mg kg^{-1}) includes total metal concentrations in mg kg^{-1}, % moisture, conductivity, bulk density, % organic matter and pH

	Mine Location					
	Premier		Steeprock		Winston Lake	
	Mean	Standard Deviation	Mean	Standard Deviation	Mean	Standard Deviation
% Moisture	19.54	7.87	9.23	3.52	15.54	6.23
Conductivity (us/cm)	347.35	157.86	139.79	101.31	235.71	229.09
Bulk Density (g/cm^3)	.87	.22	.93	.21	.91	.20
Organic Matter	2.72	1.13	5.80	1.16	4.57	7.23
pH	7.90	.17	5.86	1.94	6.36	.90
Aluminum	1.2%	0.3%	0.6%	0.2%	0.8%	0.2%
Arsenic	2245.36	3106.81	320.65	104.37	2.00	0.00
Barium	41.16	32.73	77.41	225.74	21.70	9.15
Calcium	2.2%	1.2%	1.7%	4.5%	0.19%	0.18%
Cobalt	17.83	8.57	25.12	11.74	2.47	1.09
Chromium	34.46	25.06	313.81	214.59	35.91	14.11
Copper	51.04	21.82	41.93	13.99	54.39	63.29
Iron	4.4%	1.5%	4.6%	17.6%	1.4%	0.36%
Potassium	2265.84	1275.54	573.20	340.97	304.39	151.20
Magnesium	1.0%	0.34%	0.81%	1.65%	0.39%	0.19%
Manganese	762.66	422.32	2624.90	2213.43	133.02	70.77
Molybdenum	7.74	24.37	4.46	12.06	2.00	0.00
Sodium	422.42	291.58	76.53	43.05	99.72	22.66
Nickel	53.66	43.90	124.94	39.50	20.10	8.84
Phosphorus	355.64	93.11	227.90	51.74	304.81	156.11
Lead	28.97	33.98	61.60	19.81	4.51	1.30
Antimony	1472.30	805.71	674.91	1001.75	2.00	0.00
Strontium	86.04	26.38	34.40	20.21	4.48	1.62
Titanium	298.14	152.31	306.89	162.44	410.64	75.70
Vanadium	32.17	20.57	125.90	65.92	23.83	4.50
Zinc	78.37	24.43	112.88	45.87	787.61	953.26

Discussion

Plant Characterization

Species richness and density was much lower than typical southern boreal forests in Canada. Very few species were tabulated compared to Haeussler *et al*. (2002). They found that species richness was higher in clear cut forests compared to old growth forests. Heavy mechanical soil disturbance and removed soil organic layers could drastically decrease the residual and resprouting species so as to shift to pioneering species growing from seeds and spores, providing an opening for non-native species invasion. The results of the NMS data and cluster analysis provide evidence of several factors: invasive species ability to adapt to the site conditions, type of soil conditions following mining operation and differences in the age of the stands due to time since closure. None of the mines had a completely unique set of plant species. The first group of plants in the NMS/Cluster analysis included the outlier species that have been classified as monocultural, invasive or exotic. With the alteration of the landscape, monocultures of these species occur due to their quick adaptation to the soil conditions, open sunlight, little competition and their ease of reproduction through seed or rhizomes (Bosdorff *et al*., 2005). They also tend to have hermaphroditic sex habits, extended flowering, small seeds, and a short lifespan (Cadotte and Lovett-Doust, 2001). Plants like *Equisetum* spp. can improve the soil compaction and lower the conductivity of the soil as well as improve soil nutrition (Young *et al.,* 2013). The next group of plants in the analysis were found at the Winston Lake location which had different soil conditions compared to the other sites, so these plants can be found on disturbed soils but not necessarily elevated metal contaminated soils. While all three sites were disturbed from mining operations, Winston Lake had levels of As, Sb and Mo in the soil considered normal to plants so plant species growing at this site are living on generally disturbed soils (Kabatas-Pendias, 2010). All of the other plants investigated in this study are in the last group of the analysis: white pine, white spruce, willow, white birch, trembling aspen, red pine, balsam fir, jack pine, blueberry, tamarack, dandelion, yarrow, sedge, daisy, balsam poplar, strawberry, cedar, red clover, bladder campion, and primrose . This group contains all of the older trees and are found on the majority of the transects. All of these plants can be considered potential candidates for rehabilitation purposes as they are found on a variety of disturbed soils and have a wide range of habitat for wildlife, and growth habits.

Soil Characterization

The As and Sb levels at the Premier sites and at Steeprock are similar (Jana *et al*., 2012). These levels of As and Sb are quite elevated according to Canadian standards of soil quality of 12 mg As kg^{-1}, and (CCME, 2007) or worldwide values of 0.05 to 4 mg Sb kg^{-1} and 1.5 to 3.0 mg As kg^{-1} soils from igneous rocks and 1.7 to 400 mg As kg^{-1} from sedimentary rocks (Kataba-Pendias and Mukherjee,, 2007; Smith et al., 1998). Canadian soil quality standards have Mo at 5 mg Mo kg^{-1} so Premier is the only location with average amounts of 7.74 mg Mo kg^{-1} while Steep Rock shows borderline levels just under the limit. The high amounts of Ca, Mg, K, and Na at each site can negatively influence the plant metabolism especially in the higher pH conditions (Wong *et al*., 1998).

Conclusions

The mining areas of Steeprock, Premier and Winston Lake show a range of plants with varying tolerance to soil metal concentrations of As, Mo and Sb with a range of accumulations. The main findings are i) a variety of plant species can be found at all three locations with few species specific to each mine, ii) the soil characteristics were quite different at each of the closed mines , and iii) there were species with the potential to be metal excluders including white birch, willow*.,* trembling aspen, goldenrod, pearly everlasting and tamarack.

Since differences exist between disturbed forest soils and man-made unweathered mine soils, difficulty arises when planting directly on mine soils. For better success at replanting mine soils with phytostabilization species, soil improvements could improve plant survival and growth. The addition of some topsoil or organic amendments improves soil moisture and nutrient availability. These could include woodbark, composts or another local waste source. Insulating layers of subsoil including building rubble, refuse, or uncontaminated rock would help buffer the planted species from lower underground metal contamination and increase the success of the seedlings and cuttings (Zhang *et al*., 2001).

Rehabilitation of contaminated soils on closed mines will have to include a variety of species for the specific metal contamination so as to mimic the diversity of the surrounding boreal forest. Some metallophytic species have a natural drought tolerance so as to withstand the dry conditions of the mine soil. Perennial species, species with wide ranging root systems, and those adapted to cold winters, low nutrient, low organic matter and compacted soils can be included in closure replanting plans for mines in Northwestern Ontario. Focus should be placed on pioneer plant species, rather than the climax coniferous species such as white spruce due to their poor health after planting on these mine sites. Plantings should include a mix of grasses, herbs, shrubs and trees that will colonize the surrounding area, increase organic matter and improve fertility and soil characteristics like water retention, aeration and wildlife habitat. Hyperaccumulator plants should be avoided for planting, actively eliminated from areas through weeding or only planted on areas scheduled for regular harvesting for metal removal so as to reduce the hazard for the future land uses. Future research should include test plantings in various metal concentration and soil types for ease of use. as well as with various organic matter, fungal and bacterial amendments.

Acknowledgements

Many thanks to Barrick Hemlo, the Lakehead University Environmental Laboratory and the National Sciences and Engineering Research Council of Canada.

Literature Cited

Ashraf, M.; Maah, M. & Yusoff, I. 2011. Heavy Metals accumulation in plants growing in ex tin mining catchment, Int. J. Enivron. Sci Tech. **8**(2), 401-416.

Bagatto, G. & Shorthouse, J. D. 1999. Biotic and abiotic characteristics of ecosystems on acid metalliferous mine tailings near Sudbury, Ontario, Canadian Journal of Botany **77**(3), 410-425.

Bossdorf, O.; Auge, H.; Lafuma, L.; Rogers, W. E.; Siemann, E. & Prati, D. 2005, Phenotypic and genetic differentiation between native and introduced plant populations , Oecologia **144**(1), 1--11.

Cadotte, M. W. & Lovett-Doust, J. 2001. Ecological and taxonomic differences between native and introduced plants of southwestern Ontario , Ecoscience, 230--238.

Canadian Council of Ministers of the Environment. 2014. Canadian Environmental Quality Guidelines . http://www.ccme.ca/.

Chen, B.-C.; Lai, H.-Y. & Juang, K.-W. 2012. Model evaluation of plant metal content and biomass yield for the phytoextraction of heavy metals by switchgrass , Ecotoxicology and Environmental Safety **80** (0), 393 - 400.

Dickinson, N. M.; Baker, A. J.; Doronila, A.; Laidlaw, S. & Reeves, R. D. 2009. Phytoremediation of inorganics: realism and synergies , International Journal of Phytoremediation **11**(2), 97-114.

Freitas, H.; Prasad, M. & Pratas, J. 2004. Plant community tolerant to trace elements growing on the degraded soils of São Domingos mine in the south east of Portugal: environmental implications , Environment International **30**(1), 65 - 72.

Greenstone Economic Development Corporation. 2015. Northern Empire mine - Beardmore . http://www.gedc.ca/article/northern-empire-mine--beardmore-225.asp.

Gitelson, A. A.; Buschmann, C. & Lichtenthaler, H. K. 1999. The Chlorophyll Fluorescence Ratio F735/F700 as an Accurate Measure of the Chlorophyll Content in Plants , Remote Sensing of Environment **69**(3), 296 - 302.

Haeussler, S.; Bedford, L.; Leduc, A.; Bergeron, Y. & Kranabetter, J. M. 2002. Silvicultural disturbance severity and plant communities of the southern Canadian boreal forest , Silva fennica **36**(1), 307--327.

Jana, U.; Chassany, V.; Bertrand, G.; Castrec-Rouelle, M.; Aubry, E.; Boudsocq, S.; Laffray, D. & Repellin, A. 2012. Analysis of arsenic and antimony distribution within plants growing at an old mine site in Ouche (Cantal, France) and identification of species suitable for site revegetation , Journal of environmental management **110**, 188--193.

Kabata-Pendias, A. 2010. Trace elements in soils and plants, CRC press.

Khan, A.; Kuek, C.; Chaudhry, T.; Khoo, C. & Hayes, W. 2000. Role of plants, mycorrhizae and phytochelators in heavy metal contaminated land remediation , Chemosphere **41**(1-2), 197 - 207.

Kumar, P. N.; Dushenkov, V.; Motto, H. & Raskin, I. 1995. Phytoextraction: the use of plants to remove heavy metals from soils , Environmental Science & Technology **29**(5), 1232--1238.

Lafrance, B.; DeWolfe, J. C. & Stott, G. M. 2004. A structural reappraisal of the Beardmore Geraldton Belt at the southern boundary of the Wabigoon subprovince, Ontario, and implications for gold mineralization , Canadian Journal of Earth Sciences **41**(2), 217--235.

Lakehead University Environmental Laboratory (LUEL). 2012. Water, Soil and Plant Tissue Protocols. Lakehead University, 65pp.

Magurran, A. E. 2013. Ecological diversity and its measurement, Springer Science & Business Media.

Majumder, A. & Jha, S. 2012. Hairy Roots: A Promising Tool for Phytoremediation, in T. Satyanarayana & Bhavdish Narain Johri, ed., Microorganisms in Environmental Management , Springer Netherlands, , pp. 607-629.

McCune, B.; Grace, J. B. & Urban, D. L. 2002. Analysis of ecological communities, Vol. 28, MjM software design Gleneden Beach, OR.

McIntyre, T. 2003. Phytoremediation of heavy metals from soils Phytoremediation , Springer, , pp. 97--123.

Mendez, M. & Maier, R. 2008. Phytoremediation of mine tailings in temperate and arid environments , Reviews in Environmental Science and Biotechnology **7**, 47-59.

Nouri, J.; Karbassi, A. & Mirkia, S. 2008. Environmental management of coastal regions in the Caspian Sea , International Journal of Environmental Science & Technology **5**(1), 43-52.

Ontario Geological Survey, Shklanka, R. & McIntosh, J. R. 1972. Geology of the Steep Rock Lake Area: District of Rainy River, Department of Mines and Northern Affairs.

Oppelt, E. T. 2000. Introduction to Phytoremediation (EPA/600/R-99/107), Technical report, National Risk Management Research LaboratoryOffice of Research and DevelopmentU.S. Environmental Protection AgencyCincinnati, Ohio 45268.

Peer, W.; Baxter, I.; Richards, E.; Freeman, J. & Murphy, A. 2006. Phytoremediation and hyperaccumulator plants, in Markus J. Tamas & Enrico Martinoia, ed., Molecular Biology of Metal Homeostasis and Detoxification , Springer Berlin Heidelberg, , pp. 299-340.

Pilon-Smits, E. 2005. Phytoremediation , Annu. Rev. Plant Biol. **56**, 15--39.

Salt, D. E.; Blaylock, M.; Kumar, N. P.; Dushenkov, V.; Ensley, B. D.; Chet, I. & Raskin, I. 1995. Phytoremediation: A Novel Strategy for the Removal of Toxic Metals from the Environment Using Plants , Nat Biotech **13**(5), 468--474.

Tordoff, G.; Baker, A. & Willis, A. 2000. Current approaches to the revegetation and reclamation of metalliferous mine wastes , Chemosphere **41**(1-2), 219 - 228.

Walters, R. G. 2005. Towards an understanding of photosynthetic acclimation , Journal of experimental botany **56**(411), 435--447.

Whiting, S.; Reeves, R. D.; Richards, D.; Johnson, M. S.; Cooke, J. A.; Malaisse, F.; Paton, A.; Smith, J. A. C.; Angle, J. S.; Chaney, R. L. & others 2004. Research priorities for conservation of metallophyte biodiversity and their potential for restoration and site remediation , Restoration Ecology **12**(1), 106--116.

Wong, M. 2003. Ecological restoration of mine degraded soils, with emphasis on metal contaminated soils , Chemosphere **50**(6), 775 - 780.

Young, I. W.; Naguit, C.; Halwas, S. J.; Renault, S. & Markham, J. H. 2013. Natural revegetation of a boreal gold mine tailings pond , Restoration Ecology **21**(4), 498--505.

Zhang, Z.; Shu, W.; Lan, C. & Wong, M. 2001. Soil seed bank as an input of seed source in revegetation of lead/zinc mine tailings , Restoration Ecology **9**(4), 378--385.

TAILINGS RECLAMATION AND REVEGETATION USING MUNICIPAL DEWATERED BIOSOLIDS

Jeff Newman, Terratec Environmental*, Glen Watson, Vale, Michael Payne, Black Lake Environmental

***Terratec Environmental, 200 Eastport Blvd, Hamilton, Ontario L8H 7S4**
Jnewman@terratecenvironmental.com

(905) 544 0444

Site remediation of severely disturbed lands such as mine tailings, strip mines, and logged areas has been progressing in other parts of North America for many years. In Ontario, residuals such as paper fibre residuals have been used for reclamation on a limited scale. Some active sites have implemented temporary solutions such as spreading chopped hay or straw or applying a chemical dust suppressant on tailings to reduce the amount of wind erosion from the surface of un-remediated areas. Although these practices are effective, they are costly, labour intensive and are not an economical long term strategy. An innovative method to address challenges of site remediation and dust control faced by the mining industry has been initiated by a unique partnership between Terratec Environmental Ltd (Terratec) and Vale Canada (Vale).

In 2012, Terratec entered into discussions with Vale to establish a collaborative trial project to utilize municipal biosolids to enhance vegetative growth on the mine tailings at their operation in Copper Cliff, Ontario.

Vale Canada's operation in Copper Cliff, situated within the City of Greater Sudbury, is an integrated mine, mill, smelter and refinery complex. For over 100 years, tailings from the milling operation have been deposited in the Copper Cliff Central Tailings impoundment, a 2,500 hectare (ha) facility. Since the early 1970's, a variety of projects and research trials have been done in an effort to re-vegetate the tailings area. Although there has been notable success, there were and still are large areas of bare or sparsely vegetated tailings which have led to erosion management challenges. Due to the fineness of the tailings wind erosion has been a significant issue when the surface dries. In an effort to control the tailings dust, Vale staff has spread either chopped hay or straw or a chemical dust suppressant. These practices, implemented to limit the effects of the wind erosion have been and are currently still effective but they have a significant cost in material and labour. Terratec and Vale wished to investigate a cost-effective alternative solution that would also provide significant additional environmental and aesthetic improvements such as wildlife habitat and overall improvements in biodiversity.

Biosolids are the nutrient-rich by-product from the treatment of municipal wastewater. The nutrients contained in biosolids include nitrogen, phosphorus and organic matter, in addition to essential micro-nutrients like copper, iron, molybdenum and zinc, which are vital to plant growth and soil fertility. Typically, biosolids are spread on agricultural fields as a commercial fertilizer replacement. However, when agricultural areas are not available due to weather or regulatory restrictions, alternative areas of utilization are necessary in order to keep this valuable resource from landfills.

It became apparent that the solution for each company's challenge was creating a partnership with one another. Vale could utilize biosolids from Terratec to establish a permanent vegetation cover on the tailings. Terratec could beneficially utilize this valuable resource for remediation purposes rather than landfilling. By implementing this program, both companies would benefit from innovative techniques to managing their respective challenges.

In 2012, Vale and Terratec proposed the establishment of an initial trial utilizing 2,000 dry tonnes (DT) of dewatered municipal biosolids. Application timing was dependent on the specific project and site suitability. Proposed project objectives were:

1. To establish a vegetative cover on the inactive portion of the tailings area to reduce or minimize and, ideally, eliminate wind and water erosion.
2. To establish a vegetative cover on the mine tailings to reduce water infiltration through absorption and utilization by the plants and evapotranspiration from the vegetation.
3. To establish a vegetative cover on the slopes of the tailings dams to reduce erosion potential.
4. To provide a growing media for hay and / or straw production for use in other areas of the tailings for dust suppression and to reduce the importation of purchased hay and straw for dust suppression.
5. To reduce the utilization of dust suppressant chemicals currently applied to the tailings.
6. To establish a vegetative cover on the mine tailings that would provide natural habitat enhancement and result in overall biodiversity improvements.

The utilization of biosolids is a sustainable environmentally prudent reclamation alternative to conventional practices for the mining sector. Conventional reclamation requires the use of materials that are mined or quarried from neighbouring lands or the use of chemical fertilizers and other soil amendment chemicals. Materials such as sand, clay, and topsoil extracted, are expensive to import and leave the source location "scarred". Considerable resources need to be spent to extract virgin soil and transport it to the mine site. The geographical magnitude of the Copper Cliff tailings operations makes the prospect of using soil from adjacent lands very impractical. Using soil would create large tracts of land that then would need to be repaired and revegetated using methods similar to closure practices for an aggregate extraction location. As a consequence, one problem would be solved at the expense of creating a second problem. By utilizing biosolids, Vale is keeping tonnes of nutrient-rich material out of landfills and providing a growth media for revegetation, revegetation which will provide for increased sustainability, while potentially growing their own erosion mitigating products, and reducing their reliance on imported remediation materials.

Currently, the Vale Copper Cliff Central Tailings Facility consists of an area of approximately 2,500 ha, 1,300 ha are inactive and are available for remediation with biosolids. The remaining area continues to be active for tailings disposal and water control. At the current application rate of 150 DT/ha there is the potential requirement of approximately 195,000 DT of biosolids. This could provide more than 30 years of biosolids utilization at an annual rate of 6,000 DT of material. The success of this project has led Vale to evaluate other sites in the Sudbury area for this type of remediation, ensuring a long term environmentally sustainability rehabilitation program.

In the future, Vale plans to harvest the vegetation from the reclaimed areas to be used to offset the importation of hay or straw from farming areas outside of Greater Sudbury where it is currently sourced. This will reduce fuel consumption for transporting this material and make more straw and hay available to the agricultural community in Northern Ontario.

Vale and Terratec are further developing the program by adding leaf and yard waste from the Greater Sudbury area to increase the quality of the amendment being placed on the Vale tailings operations. This will increase the understanding of how to optimize the mixture to address local conditions and potentially how to take this concept and correct similar problems in other areas of the province, where land reclamation efforts are in need of a cost effective solution.

Vale and Terratec through a unique partnership have developed a model for both the mining and the organics recycling industries in Ontario. They have demonstrated that the environmentally sustainable practice of combining and utilizing organic residuals from municipalities and industries is both economically and environmentally feasible as well as good sense in this age of Reduce, Reuse and Recycle.

EVALUATING AND PLANNING REHABILITATION IN AN HISTORIC MINING CAMP USING A RISK ASSESSMENT APPROACH IN COBALT, ONTARIO

Josée Noël[1]
Jean Cayouette[1]

[1]Agnico Eagle Mines Ltd., Val-d'Or, QC, J9P 0E9

Key Words: mining legacies, Cobalt, mine hazards, rehabilitation, mitigation, risk assessment, prioritization

Introduction

The Cobalt silver mining camp, in northeastern Ontario, was active starting in the early 1900's and has been the host to over 100 mines and 15 mills over the course of the last century. Cobalt is considered the "Cradle of hard rock mining in Canada"; it has greatly contributed to the development of Canada's mining expertise now exported all over the world. The rich mining history of Cobalt is evident and recognized.

Along with the heritage associated with these old mine sites also comes the potential for physical mine hazards (e.g. openings to surface), residual mine rocks, tailings areas and former industrial buildings/foundations to be present. These features can present a certain level of risks to public health and safety, as well as to the aquatic and terrestrial environments.

Agnico Eagle holds to this day the rights of a portion of the historic mining sites in the Cobalt-Coleman Area (about 230 properties). Some of these sites were operated and closed well before Agnico Eagle acquired the properties. From 1957 to 1989, Agnico Eagle Silver Division operated 23 mines, 4 mills and 1 refinery in the Cobalt-Coleman Area. Closure plans (7) were developed and filed with the Ministry of Northern Development and Mines (MNDM) between 1993 and 1998. Closure and rehabilitation work was done during the same period, and extended up to 2004.

In 2012, the MNDM asked for amendments to 1990's closure plans. Agnico Eagle thus engaged in a thorough inventory and investigation of their properties. The amount of information gathered during this exercise called for a system to be developed to support data analysis. We also needed to process the information in order to prioritize additional/required rehabilitation actions based on criteria that would take into account technical and economic factors, but also environmental and socio-cultural factors. A risk assessment approach was selected to reach that goal.

Risk Assessment Approach

The risk assessment approach consisted of 6 steps (Fig. 1). Steps are detailed in the following subsections.

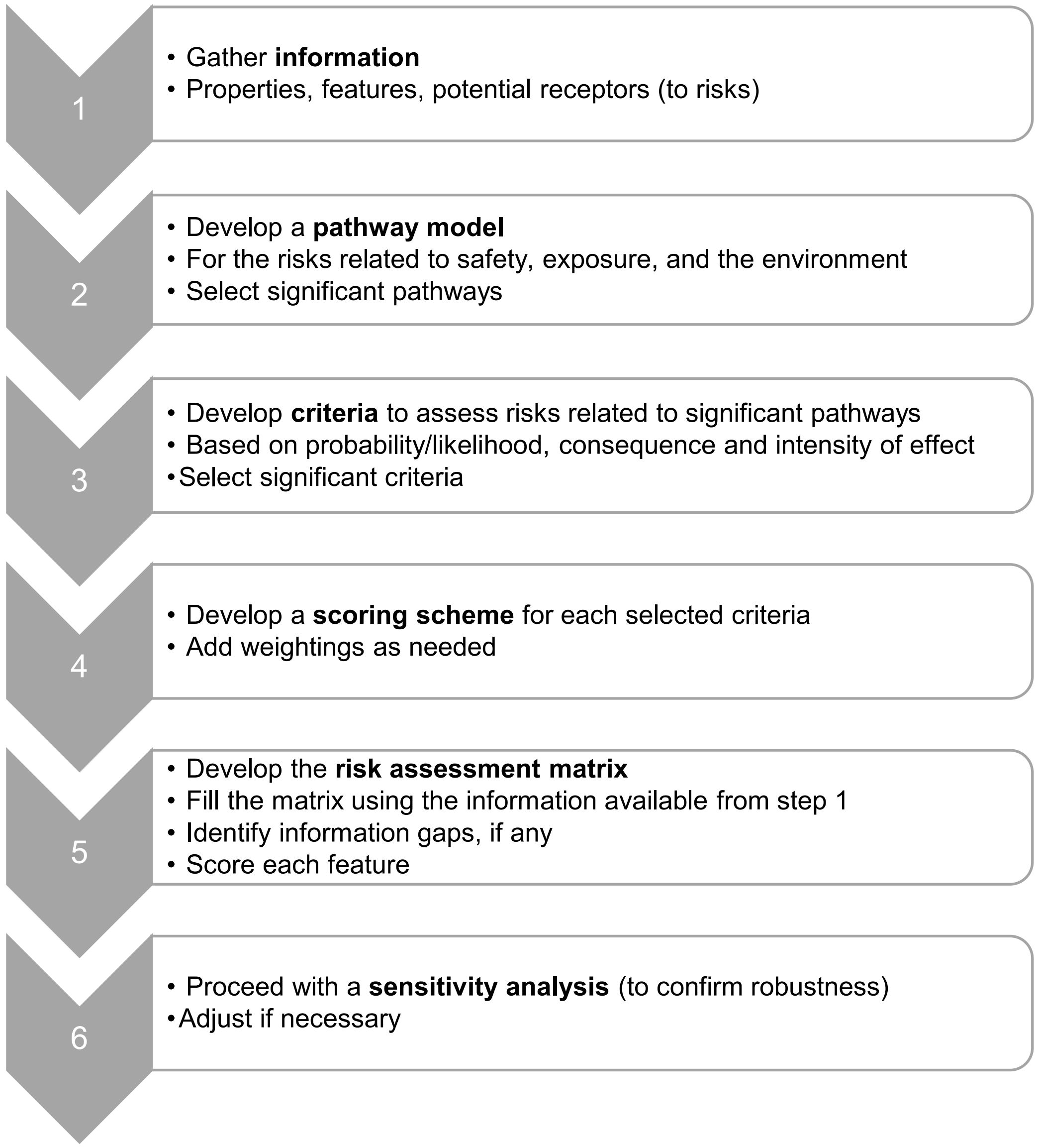


Figure 1: Legacy Mine Hazards Risk Assessment Tool Development Steps

Step 1: Information Compilation

From 2012 to 2015, Agnico Eagle conducted a review of the features on their Cobalt-Coleman properties in order to update the information on the status of potential mine hazards. The following types of features (Table 1) were identified. To ensure clarity and consistency when using the terminology, these were defined (Golder Associates, 2014).

Table 1: Types of Features encountered in the Cobalt-Coleman Historical Mining Area

Adits	Head Frames	Shafts
Buildings	Open Cuts	Stopes to Surface
Crown Pillars	Open Holes	Subsidence
Exploration Pits	Pits	Tailings Areas
Exploration Shafts	Portals	Trenches
Exploration Trenches	Raises	Waste Rock Piles
Foundations	Settling Ponds	Water Quality

Historical information[1] (map, reports) was used to find attributes (depth, dimensions, previous status, rehabilitation method, etc.) and general information on the features. Results of the studies done over the years in the area in terms of tailings and waste rock characterization, as well as surface and groundwater quality, were also compiled and analyzed. This was followed by field inspections and sampling programs to confirm/infirm data and to complete data coverage. All this information was inputted on Field Sheets, in a data management system, and then into a Georeferenced Information System (GIS).

This review identified 517 openings to surface, 300 crown pillars, 10 tailings areas, and 20 waste rock piles. Mitigation measures and rehabilitation already in place were inventoried as part the status of the features. Results indicated that 88% of openings to surface are considered rehabilitated, and 62% of crown pillars do not require further assessment. Some features would thus require additional work to ensure safety and/or to manage potential risks.

In addition to the information pertaining to the historical mining features, a field work program and mapping exercise was performed to complete the database related to the natural and social environment (receptors). Field and desktop work allowed for the mapping of the types of surrounding habitats, location of population, sensitive habitats, and species (notably, species at risk). All this information was also inputted into the GIS.

Step 2: Pathways

Two main categories of risk source were identified with reference to the legacy mine features in the Cobalt-Coleman Area. These are (1) the **physical hazards** (safety risk, e.g. falling in a hole), and (2) the **risks of exposure to contaminant** (for both the population and the environment). Three main receptors were identified: (1) **humans**, (2) **animals**, and (3) **ecosystems**. Table 2 summarizes the pathway analysis that was done to link the potential risks with potential receptors.

[1] It should be noted that the quality of the historical information may vary depending on the source. Notes were added to the database to take that into consideration.

Table 2: Pathway Analysis – Risks related to Legacy Mine Features in the Cobalt-Coleman Area

Receptor	Risk Source: Physical hazards	Exposure to contaminants: Soils[1]	Sediments	Air	Surface water	Groundwater
Humans Ingestion		○			√	○
Inhalation				√		
Safety	√					
Animals Aquatic wildlife			○		√	
Terrestrial wildlife	○	○			○	
Livestock						
Ecosystems Land-based		√		○		
Freshwater		√	○		○	

[1] Raw tailings/waste rock material is considered in this risk category

○ Potential Pathways √ Selected Pathways

Step 3: Criteria

The overall approach to develop criteria was to determine, for each risk source and their selected pathway(s), the **factors** that could influence the risks, could they be related to the:

- Probability/Likelihood;
- Consequence;
- Intensity; or
- Current mitigation.

A first exhaustive list of potential criteria was developed. Of these, only the ones that actually would discriminate the features were retained/selected (Table 3).

Step 4: Scoring Schemes and Matrix

All selected criteria were developed into scoring schemes using a scale from very low to very high. The criteria were developed using measurable units as often as possible in order for the tool to be objectively filled by any user. For more subjective criteria (e.g. potential for injury, consequence of failure), detailed scales were developed to describe as much as possible each option and try to eliminate most of the potential subjectivity.

Current mitigation was introduced in the scoring scheme using a multiplicator (from 0 to 1) lowering the feature's score when mitigation is in place.

Table 3: Criteria Selected for the Cobalt Legacy Features Risk Assessment

	Risk to public safety		Risk of exposure		Risk to the environment	
	Accessibility and risk of injury	**Stability**	**Particulate matters**	**Surface water**	**Surface water**	**Habitat**
Probability	Distance to nearest residence	Distance to nearest residence	Distance to nearest residence	Accessibility of impacted water	Downstream distance to Lake Timiskaming	Risk of material to migrate into surrounding environment
	Ease of public access	Ease of public access			Probability of contact	Mine waste storage area
Intensity	Height/depth of feature	Potential of failure (POF)	Arsenic concentration	Arsenic concentration		
Consequence/ Likelihood	Potential injury	Consequence of failure	Mine waste storage area	Use of surface water	Contribution to arsenic concentration in Farr Creek	Type of downstream habitat
Mitigation	Current mitigation	Current mitigation	Current mitigation	Current mitigation	Current mitigation	Current mitigation

The risk assessment matrix was then developed with these criteria and scoring schemes. To better reflect the importance of the risks in the overall analysis, the tool allows for the possibility to weight both the risks and/or the pathways.

Step 5: Fill the Matrix and Score Features

Once the risk assessment matrix was fully developed, it was tested, adjusted, and then the assessment took place. Each feature was inputted into the matrix, described, and then evaluated using the applicable set of criteria. Total score for a given feature was the sum of all applicable risk mitigated scores. So the more residual risks there was for a given feature, the higher the total score was. Features scores can be summed to give an overall score for a given property.

Step 6: Sensitivity Analysis

A sensitivity analysis was finally done to ensure robustness of the tool and the results. Since different features (e.g. open holes vs tailings areas) were compared all together, attention was given to evaluate representativeness of the results. The basis for comparison needed to be clear and solid for the tool and results to be used with confidence.

Results: Ranking to Inform Action Plan

Once all the features were scored, it allowed for the ranking of all of them. Priority for additional rehabilitation work will be given to the features with the highest scores. Scores could also be used to determine if additional rehabilitation work is required, taking inherent risks into account. Property's score will also be looked at to identify areas that could benefit the most from additional rehabilitation efforts, taking into account proximity for enhanced effectiveness.

This approach allowed for effective use of the information available on the features in order to assess their potential/residual risks. The tool was used to score and compare legacy mine features from a health and safety, social and environmental risk perspective. It provided with a detailed and documented tool to help communicating priorities to stakeholders.

References

Golder Associates. 2014. Physical Mine Hazards, Far Field Areas (Closure Plan Amendment #1). Agnico Eagle Mines Ltd. Cobalt – Coleman Area Properties. Report Number 1405040, December 2014. 29 pages + figures and appendices.

RESTORATION OF A SPHAGNUM-DOMINATED PEATLAND IMPACTED BY A MINERAL ROAD BY THE BURIAL UNDER PEAT LAYER METHOD

Kathy Pouliot[1] and Line Rochefort[2]

[1] Peatland Ecology Research Group (PERG) and Centre for Northern Studies (CEN), Université Laval, Québec, Qc, G1V 0A6. [2] Full professor, Researcher and Chairholder of the Industrial Research Chair in Peatland Management, Peatland Ecology Research Group (PERG) and Centre for Northern Studies (CEN), Université Laval, Québec, Qc, G1V 0A6.

Contact: kathy.pouliot.2@ulaval.ca

The electric power transmission lines network spans across the territory of the province of Québec on 34 000 kilometers and some of its sections inevitably go through peatlands. For equipment maintenance, mineral roads sometimes have to be constructed. By introducing alkaline material into the acid peaty substrate, the concentration, ionic form and ratio of nutrients are modified and may lead to a transformation of the composition and diversity of the vegetation and favor the propagation of invasive species. Furthermore, these access roads threaten the ecological integrity of bogs by changing the nature of the substrate, which can affect their hydrological connectivity.

A restoration project has been conducted on two peatlands where access roads built from mineral material were constructed under power lines: at Sainte-Eulalie and Chénéville (in the Centre-du-Québec and Outaouais regions, respectively). We examined if burying mineral material in a bog is an effective method to restore the peatland conditions that were prevailing before the disturbance. The restoration by the “Burial Under Peat Layer Method” (BUPLM) consists in excavating and burying the mineral material *in situ*, beneath the underlying peat material. The surface is then mechanically flattened and revegetated using diaspores from the adjacent untouched peatland. In order to allow comparison, half of the road at the Chénéville site has been restored with this method, and the other half by removing the mineral material and replacing it with horticultural peat. In Sainte-Eulalie, only the BULMP was applied. We hypothesize that the BUPLM is efficient at confining the nutrients introduced by the mineral material underneath the peat layer, at recreating a surface elevation similar to the adjacent areas and at re-establishing typical peatland vegetation.

The water pH and electrical conductivity are convenient proxy analysis to characterize a peatland. Water samples have been collected in transect at different distances from the buried road and at different depths in the peat. We noticed an increase of pH values close to the road (within 15 meters) 1 year after the BULPM method was applied. This was probably due to the soil disturbance during the execution of the method (Fig. 1: Chénéville). 3 years post-BULPM, pH decreased to become similar to references values (natural *in situ* comparison ecosystem and literature) (Fig. 1: Sainte-Eulalie). Concerning the electrical conductivity values, the effect of the buried mineral material is perceptible within the first 5 meters from the road after 1 year (Fig. 1: Chénéville) and is limited to the width of the buried road 3 years post BULPM (Fig.1: Sainte-Eulalie).

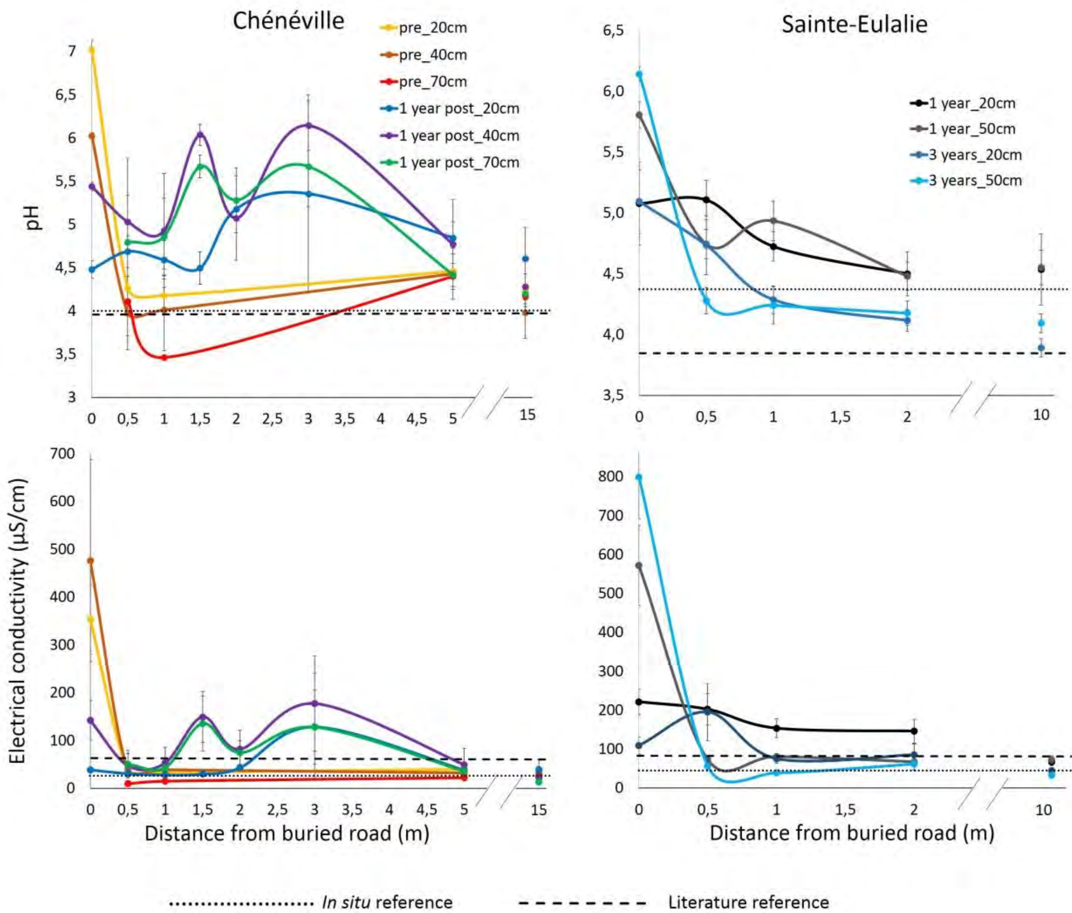


Figure 1 - pH and electrical conductivity values of Chénéville and Sainte-Eulalie.

The plant cover and species surveys conducted in the areas where was the mineral material ,1 to 3 years post-BULPM, and in the reference ecosystem (the adjacent undisturbed right-of-way) showed that transferring donor material is an efficient revegetation technique when the donor site is dominated by typical peatland species. Indeed, Chénéville restored sector presented a mean moss cover of 33% 1 year post-BULPM and Sainte-Eulalie had a total moss cover of around 5% 3 years post-BULPM. Finally, topography was measured with a laser level on transects on both sites, revealing that 1 year (Chénéville) and 3 years (Sainte-Eulalie) post-BULPM, no significant elevation differences between the restored sectors and the adjacent peatland is perceptible (< 13 cm).

In light of the results obtained with the chemistry, vegetation and topography surveys, we consider that the BULPM is comparable to the conventional method of complete mineral material removal. It is even more economical and causes considerably less circulation on the site to restore. Indeed, applying the BUPLM in peatlands involves no organic material supply, less or no transport of mineral material out of the site and less heavy machinery. These elements can be particularly profitable for restoration in remote areas.

DETERMINING BEST PROTOCOLS FOR WILD SEED COLLECTION AND METHODS FOR CALCULATING SEED VALUE

Brittany Rantala-Sykes[1,2] & Daniel Campbell[2,3]

[1]Department of Biology, [2]Vale Living with Lakes Centre and [3]School of the Environment
Laurentian University

Key Words: seed ecology, revegetation, seed collection, native plant, mine reclamation

In northern Canada, mineral extraction is expected to increase 91% from 2011 to 2020 (Rhéaume and Caron-vuotari, 2013). One result of mine development in remote regions is a demand for new research into mine rehabilitation. Revegetation is an important component of restoration. Non-native plants have been commonly used for landscape rehabilitation following disturbance. They are available for a low cost and in large quantity compared to similar native species (Ewel and Putz, 2004). In some cases non-native plants will function superiorly to a comparative native species, such as in rapid growth and establishment (Asay et al., 2001). However, non-native plants may be highly competitive and may spread to invade natural environments nearby, altering their composition and diversity (Flory and Clay, 2009).

Using native plants in revegetation is becoming more common practice (Peppin et al., 2011). Their seeds may be bought commercially or locally collected. Commercially grown native seeds are sometimes available at lower costs compared to locally obtained seed; however, their adaptation to local climate conditions may mean they are not suitable for planting in different growing regions (Belnap, 1995). The distance between genetically distinct populations can be as minimal as 20km (Krauss and Koch, 2004). In some cases, seeds with non-native genotypes are superior to native populations and may become invasive or outcross with local genotypes. Commercial sources for native seed are often difficult to find, especially within a suitable genetic provenance (<100km) (Krauss and Koch, 2004) for remote locations. Local seed collection may be required.

Local seed collection occurs for two main ecological reasons: preservation and rehabilitation. Species that are at risk of becoming extinct, are being preserved through seed collection and stored using ex–situ (seed banks) or in-situ (controlled natural environments) methods (Volis and Blecher, 2010). Plant communities within rare or frequently exploited habitats are collected from, in order to preserve the unique population genetics (Mattner et al., 2002, Volis and Blecher, 2010). More recently remote mines have begun on site seed collection programs for their mine revegetation. Collecting seed from wild sources differs greatly from collecting on uniformly planted, monocultures in a commercial setting. Collection and processing protocols as well as seed economic value are based on species specific plant and seed attributes that are not well described (Ross, 2004). These differences in plant attributes contribute to the effort required in obtaining seed and thereby alter seed value. Do Espirto Santo et al. (2010) examined several plant attributes to assess a value in price/kg of seed for several wild species. The seed value was used to aid in fair seed marketing for local business and individuals. Further knowledge regarding specific storage, collection, and ecological requirements of wild plants is required to ensure a successful seed collection program and promote industry development (Kauth and Perez, 2011).

The purpose of the study is to determine best protocols for collecting, processing, storing, and germinating seed from local, wild sources for approximately 50 species. These are critical aspects in the success of a remote seed collection program. In this study we will also determine how we can evaluate and quantify physical and ecological differences between species to determine a seed value. Seed value can be used for both species prioritizing in mine reclamation and for the development of local businesses.

Field studies will be conducted in 2016 at De Beers Victor mine, located in the Hudson Bay Lowland. We will examine regional plant characteristics such as: habitat and soil preferences, fruit maturation times, and Cree names. We will calculate seed value by evaluating key attributes that affect time and cost of using wild seeds including: regional plant abundances, fruiting characteristics (Fig. 1), requirements for identification, ease of processing and storing seeds, and the species contribution to reclamation. Attributes will be scaled from 1 to 10 and contribute to a relative seed value.

Figure 1.Photo (left) displaying poor ripening synchronicity of star false Solomon's seal (*Maianthemum stellatum*). (right) Diaspore persistence attribute; majority of berries quickly dispersed after ripening for red currant (*Ribes triste*).

Acknowledgements

We wish to acknowledge our funding contributors, DeBeers Victor mine for financial and in-kind support and NSERC. We also wish to thank our many active committee members for their guidance and input on this thesis.

Literature Cited

Asay, K. H., W. H. Horton, K. B. Jensen, and a. J. Palazzo. 2001. Merits of native and introduced Triticeae grasses on semiarid rangelands. Canadian Journal of Plant Science 81:45–52.

Belnap, J. 1995. Genetic Integrity : Why Do We Care ? An Overview of the Issues. Pages 265–266 Proceedings: wildland shrub and arid land resto- ration symposium.

Do Espirto Santo, F. S. E., J. a S. Filho, J. C. F. M. Júnior, E. S. Gervásio, and a. M. B. de Oliveira. 2010. Quanto vale as sementes da caatinga? uma proposta metodológica1. Revista Caatinga 23:137–144.

Ewel, J. J., and F. E. Putz. 2004. A place for alien species in ecosystem restoration. Frontiers in Ecology and the Environment 2:354–360.

Flory, S. L., and K. Clay. 2009. Non-native grass invasion alters native plant composition in experimental communities. Biological Invasions 12:1285–1294.

Kauth, P. J., and H. E. Perez. 2011. Production and Marketing Reports. Hortechnology 21:779–788.

Krauss, S. L., and J. M. Koch. 2004. Rapid genetic delineation of provenance for plant community restoration:1162–1173.

Mattner, J., G. Zawko, M. Rossetto, S. . Krauss, K. . Dixon, and K. Sivasithamparam. 2002. Conservation genetics and implications for restoration of Hemigenia exilis (Lamiaceae), a serpentine endemic from Western Australia. Biological Conservation 107:37–45.

Peppin, D., P. Fule, C. Hull Sieg, J. Beyers, M. Hunter, and P. Robichaud. 2011. Recent trends in post-wildfire seeding in western US forests:costs and seed mixes. International Journal of Wildland Fire 20:702–708.

Rhéaume, G., and M. Caron-vuotari. 2013. The Future of Mining in Canada ' s North.

Ross, C. 2004. NATIVE SEED COLLECTION AND USE IN ARID LAND. Environmental Monitoring and Assessment 99:267–274.

Volis, S., and M. Blecher. 2010. Quasi in situ: a bridge between ex situ and in situ conservation of plants. Biodiversity and Conservation 19:2441–2454.

MOLECULAR IDENTIFICATION AND CULTURE OF FUNGI NATIVE TO HEAVY METAL CONTAMINATED KAM KOTIA MINE

Jacqueline Weber[1], Ayooluwa Adurogbangba[1], Darcy Vaters[1], Jordan O'Reilly[1], Ali Khalvati[1], Matthew Wheeler[2], Mark Priddle[3], Sharon Regan[1]
[1] Department of Biology, Queen's University, Kingston, ON, K7L 3N6
[2] McIntosh Perry, 1-1329 Gardiners Road, Kingston, ON, K7P 0L8
[3]McIntosh Perry, 115 Walgreen Rod RR 3 Carp, ON K0A 1L0

Key Words: Phytoremediation, Bioremediation, Fungi, Symbiosis, *in vitro* root culture, Sequencing, ITS Region

Heavy metal (HM) contamination can be caused by human activity through mining, agricultural applications of fertilizer and manufacturing, and can prove detrimental to the health of humans, plants, animals and soil microbes. As environmental regulations are strengthened throughout the developed and developing world, there has been increased industrial and scientific interest in the remediation and restoration of contaminated sites. Conventional remediation strategies require the excavation of the contaminated soil, which is costly, and further damages the soil's structure and microbial community (Mulligan *et al.* 2001, Jankaite and Vasarevičus 2005). Phytoremediation, the use of plants to immobilize, sequester and/or extract heavy metal contaminants is a relatively inexpensive and non-destructive remediation strategy that could allow for more efficient restoration of contaminated land. Recent research has identified plants such as poplar as being ideal candidates for bioremediation projects, and has begun to describe the genetic and physiological basis for phytoremediatory processes in these plants.

The soil microbiome, a diverse community of bacteria, free-living fungi, and fungi that associate symbiotically with plants must also be considered when phytoremediation strategies are planned. Of particular interest to our laboratory are the Arbuscular Mycorrhizal Fungi (AMF). These fungi, members of the division Glomeromycota, are obligately symbiotic with 80% of land plants (Schüßler *et al*. 2001). AMF have been found to enhance the phytoremediatory abilities of plants (Giasson *et al*. 2006). AMF functionally extend plants' root networks, allowing them greater access to water and nutrients (Hetrick *et al*. 1988), and through the production of organic acids and phosphatases the fungi can mobilize formerly unavailable soil components, including heavy metals, for uptake by their plant symbionts (Marschener 1998). AMF and other heavy-metal adapted fungi can immobilize HMs through the secretion of HM-binding glycoproteins such as glomalin (Gonzalez-Chavez *et al.* 2004) and through hyphal binding, with fungal hyphae having a 2-4 times higher affinity for HMs than plant roots do (Joner *et al.* 2000). Fungi that have adapted to survive in heavy metal contaminated conditions are likely the best candidates for remediation (Raman *et al*. 1993). In addition to their bioremediatory abilities, plants and soil fungi aid in the formation of soil aggregates (Gaur and Adholeya 2004), and help to prevent the erosion of contaminated tailings from the site and into nearby waterways. In order to better understand the bioremediatory abilities of HM-adapted fungi, we have set out to identify indigenous fungi from Kam Kotia, an abandoned zinc and copper mine, and to culture these species *in vivo* and *in vitro*.

Contaminated soil from the Kam Kotia mine site and comparable uncontaminated soil (control treatment) was analyzed by the Queen’s Analytical Services Unit using Inductively Coupled Plasma Atomic Emission Spectroscopy (ICP-AES). The copper, zinc, arsenic and sulfur content of the soil was much higher in contaminated soil than in the control soil analyzed (see Figure 1C). Soil was also collected from the root systems of plants at the Kam Kotia mine in Timmins, ON for use in generating trap cultures to propagate environmental samples of fungi (see Figure 1A). Trap cultures are used to create a microcosm of the soil ecosystem in which environmental conditions are ideal for the growth, and later for the sporulation, of plant-associated fungi (see Figure 1B). Maize (*Zea mays*) was used as the host plant for our Kam Kotia trap cultures. Fungi were cultivated for 3 months with minimal watering to keep the maize alive, but stressed, so that fungal symbiosis would be encouraged. After 3 months of trap culture cultivation, root sections were harvested, DNA was extracted and the polymerase chain reaction (PCR) was performed using primers intended to detect AMF and the resulting fragments were subjected to DNA sequencing. The sequences were then aligned with the MaarjAM and NCBI genomics databases using a Standard Nucleotide BLAST search. Of the ten fragments successfully sequenced, nine were aligned most closely with the fungi *Capnobotryella sp. MA 3612* with a sequence identity of 85% for the Internal Transcribed Spacer region. One of the fragments sequenced aligned most closely with the fungi *Aureobasidium pullulans* with a sequence identity of 91% for the ITS region.

These fungi are known to associate with plants, but are not members of the division Glomeromycota and are not known to associate with plants in a obligately symbiotic fashion. Members of the genus *Capnobotryella* are black fungi that have been found to grow in association with the heavy metal hyperaccumulator *Thlaspi praecox* in heavy metal contaminated soil (Pongrac *et al.* 2009), suggesting their possible involvement in phytoremediatory processes. *Capnobotryella sp.* are thought to form ectomycorrhizae, but this genus and its associations with plants are not well understood. Further research must be carried out before *Capnobotryella* can be optimized for use as a bioremediatory organism. A lack of obligate symbiosis between *Capnobotryella* and plants could prove to be an advantage: if *Capnobotryella sp.* do indeed benefit associated plants in heavy metal contaminated soil, they may prove easier and more practical to culture *in vivo* and *in vitro* than AMF. *Aureobasidium pullulans* is a species of black fungi that is used as a biocontrol agent in agriculture and for the biological synthesis of polysaccharides and antifungal agents. This fungus is not an AMF and is not known to associate with plant roots. It is commonly found in extreme environments, and is able to survive drought and high levels of salt and heavy metal pollution (Gostinč r et al. 2014). Further chraracterization of the soil microbiome at Kam Kotia may allow us to determine whether *A. pullulans* is a primary succession species in the soil ecosystem of heavy metal contaminated sites, or if it is simply one of the only organisms equipped to survive in such an environment.

The sequences generated by this work are an interesting first look into the soil microbial communities of this heavy metal contaminated site. Future sequencing projects will allow us to better understand the diversity of the soil microbial community present at Kam Kotia. Because no AMF were isolated from the Kam Kotia site, a broader sampling approach is necessary to determine which plants at the site form mycorrhizal associations while experiencing heavy metal stress. The next step towards the optimization of these heavy metal tolerant fungi for use in bioremediation is the establishment of *in vivo* and *in vitro* fungal monocultures. *In vivo* monocultures of fungi will be established by inoculating soil with the desired fungal species and planting the seeds of a generalist host plant such as plantain (*Plantago major*) or maize. The creation of *in vitro* monocultures is a more challenging task, and requires the isolation of single viable spores. Spores will then be sterilized and planted alongside *Agrobacterium*-transformed hairy root cultures (see Figure 1D) of varying species to induce their germination, the formation of a symbiosis, and eventually the production of spores by the fungus. Our laboratory maintains a collection of 4 species of hairy root culture (chicory, tomato, carrot and potato). Fungi that form symbioses with plants have been found to exhibit preferences for certain hosts (Khan 2006). This diverse collection is expected to enhance the diversity of fungi that can be maintained in our *in vitro* cultures. These monocultures will be used to inoculate heavy metal tolerant plant species and investigate their impact on phytoremediatory processes. Propagules from these monocultures will also be used to determine how long AMF propagules persist in the soil record, whether they would be harmed by the application of fertilizers or chelating agents, and whether their tolerance for heavy metals is lost when grown on uncontaminated soil.

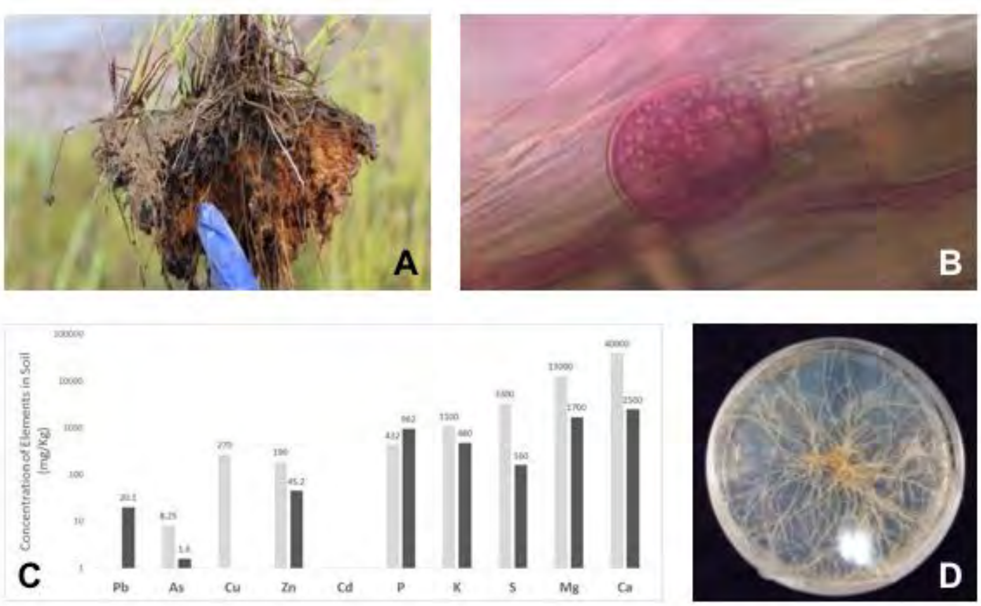


Figure 1. A) The root system of a grass growing in mine tailings at Kam Kotia. **B)** Fungal vesicle and hypha in plant root cortex. Fungal tissue stained with 0.04% fuchsin acid solution. **C)** ICP-AES analysis of soil from the Kam Kotia site. Concentrations in

parts per million (ppm). Light gray columns represent contaminated soil, dark gray columns represent control soil. **D)** *Agrobacterium*-transformed hairy root culture of chicory (*Cichorium intybus*) for use as a host plant in fungal *in vitro* cultures.

Acknowledgements

This research was supported by a Natural Sciences and Engineering Research Council ENGAGE grant awarded to Queen's University and McIntosh Perry.

Literature Cited

Gaur, A. and A. Adholeya. 2004. Prospects of arbuscular mycorrhizal fungi in phytoremediation of heavy metal contaminated soils. Current Science 86:528-534.

Giasson, P., A. Jaouich, P. Cayer, S. Gagné, P. Moutoglis and A. Massicotte. 2006. Enhanced phytoremediation: A study of mycorrhizoremediation of heavy metal–contaminated soil. Remediation Journal 17:97-110.

Gonzalez-Chavez, M. C., R. Carrillo-Gonzalez, S. F. Wright, and K. A. Nichols. 2004. The role of glomalin, a protein produced by arbuscular mycorrhizal fungi, in sequestering potentially toxic elements. Environmental Pollution 130:317-323.

Gostinčr, C., R. A. Ohm, T. Kogej, S. Sonjak, M. Turk, J. Zajc, P. Zalar, M. Grube, H. Sun, J. Han and A. Sharma. 2014. Genome sequencing of four *Aureobasidium pullulans* varieties: biotechnological potential, stress tolerance, and description of new species. BMC Genomics 15:1.

Hetrick, B. D., J. F. Leslie, G. T. Wilson and D. G. Kitt. 1988. Physical and topological assessment of effects of a vesicular–arbuscular mycorrhizal fungus on root architecture of big bluestem. New Phytologist 110:85-96.

Jankaite, A., and S. Vasarevičus. 2005. Remediation technologies for soils contaminated with heavy metals. Journal of Environmental Engineering and Landscape Management 13:109-113.

Joner, E.J., R. Briones, C. Leyval. 2000. Metal-binding capacity of arbuscular mycorrhizal mycelium. Plant Soil 226:227–234.

Khan, A. G. 2006. Mycorrhizoremediation - an enhanced form of phytoremediation. Journal of Zhejiang University Science 7:503-514.

Marschener, H. 1998. Role of root growth, arbuscular mycorrhiza, and root exudates for the efficiency in nutrient acquisition. Field Crops Research 56:203-207.

Mulligan, C., R. Yong, and B. Gibbs. 2001. Remediation technologies for metal-contaminated soils and groundwater: an evaluation. Engineering Geology, 60:193-207.

Pongrac, P., S. Sonjak, K. Vogel-Mikuš, P. Kump, M. Nečemer and M. Regvar. 2009. Roots of metal hyperaccumulating population of *Thlaspi praecox* (Brassicaceae) harbour arbuscular mycorrhizal and other fungi under experimental conditions. International Journal of Phytoremediation 11:347-359.

Raman, N., N. Nagarajan, S. Gopinathan and K. Sambandan. 1993. Mycorrhizal status of plant species colonizing a magnesite mine spoil in India. Biology and Fertility of Soils 16:76-78.

Schüßler, A., D. Schwarzott and C. Walker. 2001. A new fungal phylum, the Glomeromycota: phylogeny and evolution. Mycological Research 105:1413-1421.

THE SUCCESSFUL REHABILITATION OF ABANDONED AGGREGATE SITES ACROSS ONTARIO

Erica Rumbolt, B.Sc.

[1]Management of Abandoned Aggregate Properties Program, The Ontario Aggregate Resources Corporation; 1001 Champlain Avenue, Suite 103, Burlington, ON. L7L 5Z4.

Key words: pits, quarries, aggregate, rehabilitation, habitat restoration, reclamation, agriculture.

The Ontario Aggregate Resources Corporation (TOARC), through the Management of Abandoned Aggregate Properties Program (MAAP) delivers a program for rehabilitating former aggregate properties (deemed to be abandoned; herein referred to as ‘legacy’) across Ontario. Legacy sites qualify for the MAAP program if they have never been licenced following the establishment of the Aggregate Resources Act (ARA) in 1990 and are within designated ARA areas. Most often the legacy sites are relatively small by nature and were created as the result of small-scale operations (municipal wayside pits, private use pits or intermittent commercial operations) and were generally unregulated.

The legacy sites (files) come from an original inventory completed by the Ministry of Natural Resources and Forestry (MNRF) in the early 1990s, which equated to 6,600 qualifying legacy sites. Furthermore, as more areas of the Province are designated under the ARA the number of qualifying sites has grown to 7,900 and expected to grow in the future. The MAAP program is funded by a portion (1/2 cent per tonne) of the annual 11.5-cent/tonne licence fee paid by aggregate producers in Ontario.

Since 1997, all of the legacy sites (all 7,900) have been assessed and it has been determined that 3,200 will require some sort of assistance by the MAAP program. The reality is many of the 7,900 sites have been reverted to other uses since often it has been 40 or more years since these sites have experienced disturbances. Based on the inventories many of the legacy sites files have been ‘closed’ for multiple reasons such as: obtaining re-licence status for aggregate extraction; disappearing under urban expansion (Figure 1); being rehabilitated by the property owner; and/or have naturalized on their own (Figure 2).

Figure 1

Figure 2

In order to successful tackle the volume of sites MAAP created a systematic priority ranking system to evaluate the legacy sites across Ontario. The inventories provide a clear record of the current conditions by documenting three key parameters: safety, environmental and aesthetics factors to provide a composite overall ranking of ‘high’, ‘medium’ or ‘low’ priority. As shown in Figure 3, the legacy site demonstrates unstable slopes, deep water, vertical cliffs with easy public access and high visibility, which triggers a ‘high’ priority status. Figure 4 demonstrates a site that lacks vegetation, susceptibility to erosion and inconsistency with the surrounding agricultural area, but not easily accessible to the public or having safety concerns therefore triggering a ‘medium’ priority. The sites with higher priorities are approached first when organizing the annual work schedule. In general many of these properties exhibit severely degraded soils, steep and eroding slopes, difficult microclimates, unique species and are at various stages of naturalization.

Figure 3: High priority ranking

Figure 4: Medium priority ranking

In the simplest of terms, the MAAP program aims to rehabilitate sites solely using workable material on site to provide a higher level of function (usefulness) over the prevailing condition of the site. Rehabilitation may include grading and stabilizing slopes for safety, grading and seeding sites for agriculture or recreation, and creating and enhancing wildlife habitat by planting native trees, shrubs, wildflowers, and grasses. The appropriate course of rehabilitation is determined following consultation and consent with the landowner and conservation authorities.

Based on recent levels of extraction in Ontario the average amount available for rehabilitation projects ranges from $400,000 to $600,000 each year. This means the MAAP program is capable of rehabilitating 30-40 sites each year. The average cost to rehabilitate a legacy site has been just over $11,500 per hectare. This results in an average cost per site of just under $20,000 with an average site size of 1.58 hectares. To date, approximately $8,000,000 has been spent to reclaim/rehabilitate over 681 hectares of land, on over 435 individual sites.

The rehabilitation construction schedule is divided into a spring and fall work program. To achieve better productivity, the spring and fall sites are targeted within as small a geographical area as possible (usually within a county or regional jurisdiction). By concentrating projects into two annual groupings for work purposes, travel time for staff and contractors is minimized and opportunities are created for tendering a number of small sites together. Counties and regions targeted for work are rotated on a semi-annual basis to ensure that all sectors of the Province are considered for rehabilitation work on as equitable a basis as possible.

Historically, many legacy pits have been returned to agriculture and project site 14-03 located in the Township of Elderslie, Bruce County is an example of the typical conditions and obstacles of rehabilitation. Figures 5 a,b,c show an expansive 10 hectare legacy pit and highly visible to anyone travelling down County Road 19. The landowner was currently using the pit as pastureland but a 10-acre portion was prone to seasonal flooding and multiple pit faces made much of the pasture impractical. Over 40,000m^3 of earth was moved to rehabilitate this site. While it will be awhile before the site will be able to support livestock grazing, it is now well on the way to being able to do so from a relatively barren, unused part of the farm.

Figure 5a: Before construction

Figure 5b: During construction

Figure 5c: Post-construction

Landowner desires for a site are not always accommodated if they are unrealistic or unreasonable. For example, returning a site to agricultural use may not be practical if there is no topsoil or organic materials remaining on site that could be utilized for such purposes. The importation of large quantities of topsoil to achieve such ends may not be practical, possible or economically feasible in certain circumstances. The MAAP program will then try to solve and find other solutions.

The existence of abandoned pits and quarries provides opportunities to re-establish landscapes and ecosystems lost to settlement and urbanization. In addition to the monies being allocated to legacy pit rehabilitation, monies from the fund support research into pit and quarry rehabilitation techniques. As the MAAP program has found in many instances, these sites have reverted to naturally functioning habitat spaces on their own. In others, minimal help from the MAAP program can launch the progress of a site on a trajectory to arrive at a naturalized area in a shorter time frame than if left on its own. Continual research paired with these rehabilitation techniques will lead to the expansion of agriculture and habitat, the enhancement of biodiversity, an overall increase in ecological function and act as demonstration sites for others to replicate in the industry.

Soil amendments to increase vegetative growth on developing Technosols

Jasmine Serré[1]*, Renate Vanderhorst[1], Autumn Watkinson[2], Shane Hayes[3], Graeme Spiers[1] and Peter Beckett[1]

[1]Vale Living with Lakes Centre, Sudbury, Ontario, [2]Department of Renewable Resources, University of Alberta, Edmonton, Alberta, [3]Barrick Hemlo Operations, Marathon, Ontario

Abstract:

Building on previous studies of Technosols manufactured from crushed waste rock and board mill residuals (20:80) for use in mine reclamation, this study investigates the additions of selected soil amendments to improve fertility and soil moisture retention capacity to enhance seedling and transplant survival. This biochamber growth study set for predicted 2030 post-mine closure spring climatic conditions for the Boreal Shield Region north of Lake Superior investigated the suitability of elemental Sulphur (S0), wood boiler ash, N-Rich®, surface humus forms from stockpiled Gleysolic and Podzolic soils as amendments to promote growth of Annual Ryegrass (Lolium multiflorum). Applications of 10% (v/v) Humic Gleysol Humus Form material and wood ash at 80 t ha^{-1} promoted maximum root development. Addition of S^0 Prills (1.5% w/w) produced the greatest increase in both above and below ground biomass. The blending of N-Rich® at 34 t ha^{-1} induced a measured in-situ soil electrical conductivity of 14 $\mu S\ cm^{-1}$ after simulated storm rainfall events (1 cm rainfall hr^{-1}), with bulk through-flow containing high concentrations of dissolved Ca, Mg, K, and low Na. Soil sample analyses illustrated that N-Rich® applications provided the highest enrichment of available soil nutrients to support the growth of ryegrass, with chemical analyses highlighting the overall need for amendment application to the manufactured growth media to support healthy vegetative growth.

Introduction

Site reclamation, a key component in mine closure plans, requires research into cost-effective reclamation methods beneficial to both the mining industry and the post-mining environment. Soil materials which are commonly added to sites as part of reclamation efforts include stockpiled soils removed during mine construction (Gaster, Karst, & Landhäusser, 2015;

Sorenson, Quideau, MacKenzie, Landhäusser, & Oh, 2011), as well as soils constructed from a combination of stockpiled overburden materials and organic materials such as peat (MacKenzie & Quideau, 2010). These soils, often with unusual physical and chemical properties as a result of their diverse origins, follow different development trajectories from those of natural soils (Leguédois et al., 2016). These soils are classified as Technosols by the World Reference Base for Soil Resources due to the strong influence human activities have had on their formation (IUSS Working Group WRB, 2014).

Figure 1. Barrick-Hemlo Operations.

Figure 1. Large mine rock pile produced from open pit mining activities.

Manufacturing a soil for mine reclamation from local industrial waste materials such as mine overburden or lumber mill residuals could have considerable environmental and economic benefits such as the decrease in costs for the transport of large volumes of soil material from diverse locations, leading to a reduction in greenhouse gas emissions. Technosol manufacturing could use waste products which are otherwise valueless. However, long-term studies of Technosol development to determine their utility as a reclamation material, are necessary to provide an understanding of how material maturation may affect plant survival and growth. Ongoing research using a Technosol to revegetate a non-acid generating rock pile from a gold mine in the Boreal Shield has demonstrated that crushed mine rock blended with woody lumber mill residuals are able to support indigenous vegetation such as green alder (*Alnus viridis*), even at early stages in soil development (Vanderhorst *et al*., 2016; Watkinson, 2014). Further study into the improvement of soil properties to enhance their ability to support larger and more diverse communities of native Boreal vegetation with Technosol maturation will provide insight into the use of Technosols created for mine reclamation. One method to promote improvement of soil properties may be through the addition of different amendments. For the current study the amendments added to the Technosol are: N-Rich®, a lime stabilized sewage sludge, as a source of available NPK and trace nutrients; Sulphur prills as a source of sulphur; local forest humus form (LFH horizons) as a source of local propagules, plant nutrients released by decomposition, and also as a pH buffer to local soil conditions; wood ash from the boiler system at the White River mill as a major nutrient source (N, P, K, S, Ca, Mg and trace elements); and stored Orthic Humic Gleysolic soil material as an organic matter, nutrient and local microbial and fungal propagule source. The 80% Organic Matter (OM) Technosol is currently set up in a field experiment on the mine site for four years to mature, while ongoing growth chamber experiments investigate the utility of amendment addition to promote vegetative growth. This current study investigates the addition of the various soil amendments and application rates to Technosols manufactured from 80% OM and 20% finely crushed mine rock to improve fertility and soil moisture retention capacity to enhance grass seedling development.

Materials and Methods

Soil Material and Amendments

The manufactured soil used in this growth chamber study was composed of 80% woody organic material and 20% finely crushed mine rock from the Barrick Williams open-pit mine in Hemlo, ON (Watkinson *et al.*, 2014). The woody residuals, obtained from the (former) Domtar White River Sawmill, contained sawdust, bark, and off-cuttings derived primarily from boreal coniferous trees (Watkinson *et al.*, 2014). A total of ten soil amendments were applied and compared to the productivity of the Technosol (Treatment 1) (Table 1). The resulting eleven treatments consist of: 80% OM and 20% crushed mine rock Technosol, wood Ash obtained from White River, Ontario mixed at rates 40 t ha^{-1} and 80 t ha^{-1}; five cm of a Humus Form (LFH horizons) collected near White River, ON, both being incorporated into the Technosol material to simulate scarification effects and applied at 5 cm surface application layer; Sulphur Prills (S°) broadcasted at 40 kg ha^{-1}, 70 kg ha^{-1} and 100 kg ha^{-1}; N-Rich® applied and blended at 17 t ha^{-1} and 34 t ha^{-1}; stockpiled Humic Gleysol obtained from the mine site area was also blended at

10% soil v/v throughout the soil profile materials. Each treatment was replicated 7 times and the amended soils were randomly distributed over four Styrofoam blocks each containing 24 separate growth wells. The Growth Chamber conditions were set for predicted 2030 post-mine closure spring climatic conditions. The species used for the study was Annual Ryegrass and 32 mL of deionized water was applied twice per week to each soil to moisten evenly throughout the replicates. Soil Monitoring Probes (EC sensors and EM50 loggers from DecagonTM Devices) were installed in one replicate of each treatment for the last four weeks of plant to record soil temperature (C), volumetric water content (m^3/m^3), and the in situ electrical conductivity (mS cm^{-1}) of the soil solution phase. Leachate sampling tubes were attached to each sample's drain hole to collect the bulk throughflow following three simulated rainfall events over a three-day period nearing the end of the experiment immediate prior to harvest at eight weeks of growth. Electrical Conductivity of these solutions was assessed using an Oakton CON Meter with EC probe; the pH of soil leachates was measured with a Fisher Accument AB15 pH meter equipped with an Accumet Combination pH electrode and the concentration of dissolved elements in these solutions was determined by ICP-MS in the ELRFS ISO 17025 accredited laboratory.

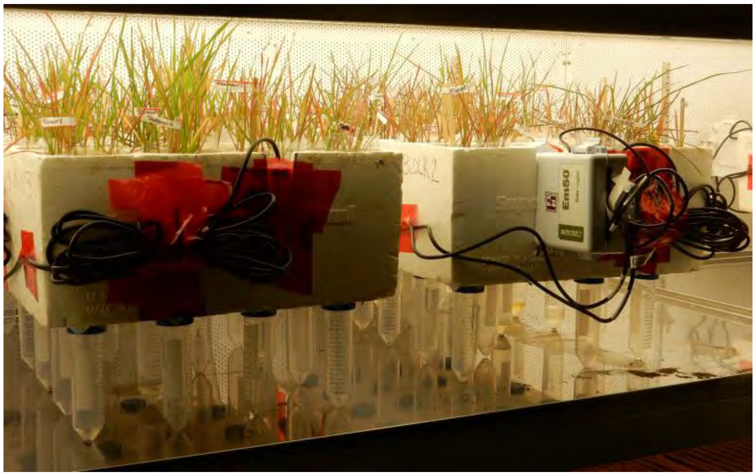


Figure 3. Soil Moisture Monitoring Probes and leachate sampling tubes connected to Styrofoam blocks.

Table 1: Treatments, application rates and method of application to the 80% woody residuals and 20% crushed mine rock Technosol for the 8 week growth study of ryegrass.

Treatment Number	Amendment	Application Rate	Method of Application
Treatment 1:	80% OM Technosol		
Treatment 2	Wood Ash	40 t ha^{-1}	Incorporated
Treatment 3	Wood Ash	80 t ha^{-1}	Incorporated
Treatment 4	LFH	5cm	Incorporated
Treatment 5	LFH	5cm	Surface
Treatment 6	Sulphur Prills (S^o)	40 kg ha^{-1}	Incorporated
Treatment 7	Sulphur Prills (S^o)	70 kg ha^{-1}	Incorporated
Treatment 8	Sulphur Prills (S^o)	100 kg ha^{-1}	Incorporated
Treatment 9	N-Rich®	17 t ha^{-1}	Incorporated
Treatment 10	N-Rich®	34 t ha^{-1}	Incorporated
Treatment 11	Stockpiled Humic Gleysol	10%	Incorporated

Results and Discussion

Ryegrass Biomass

Significant differences were found for shoot dry mass ($P<0.001$). The highest shoot dry mass mean was observed in Treatment 8, Sulphur Prills at 100 kg ha^{-1} (3.20g) and the lowest biomass was obtained by Treatment 3, wood Ash 80 t ha^{-1} (2.61 g). Significant differences were also found in shoot length biomass ($P<0.001$), with Sulphur Prills 100 kg ha^{-1} yielding the highest length mean (12.93 cm). The N-Rich® at 17 t ha^{-1} obtained the lowest shoot length (cm) observed throughout the treatments.

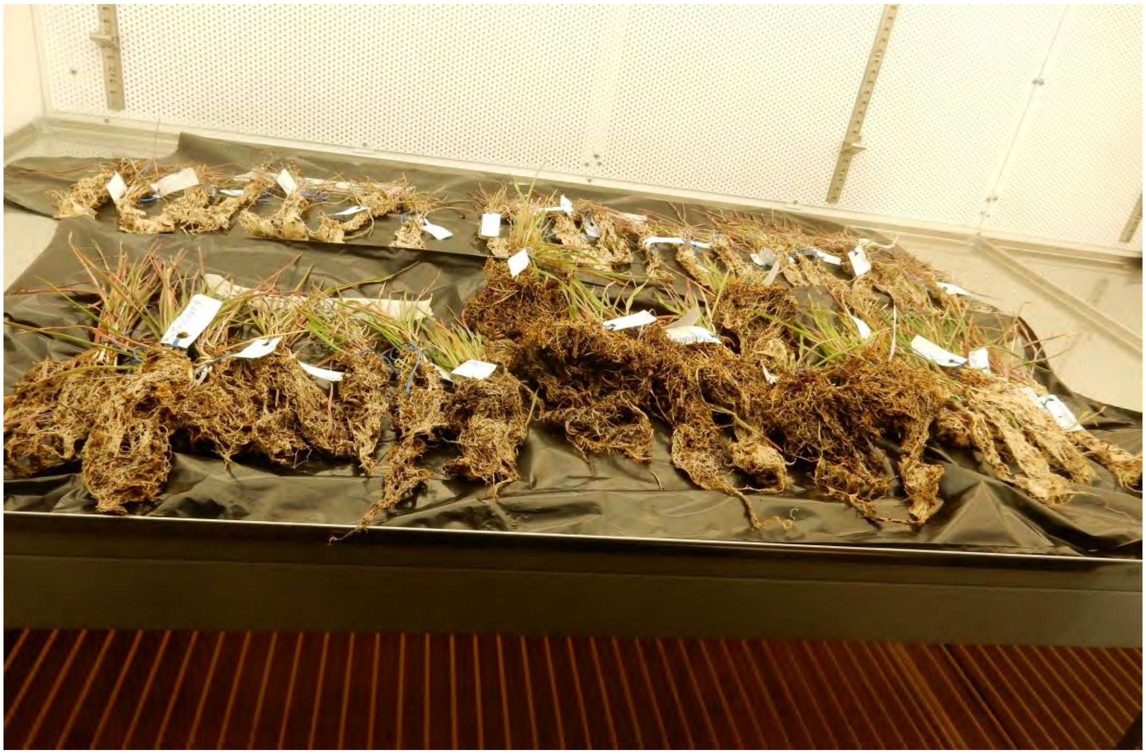


Figure 4. The harvest of Annual Ryegrass at 8 weeks of growth.

Although there was no significant difference (P=0.198) found in root weight (g) when comparing the different treatments, there was an observable difference in the root architecture between the treatments. The root development with the application of Wood Ash 40 t ha^{-1}, Sulphur Prills 100 kg ha^{-1} and stockpiled Humic Gleysol (10 % v/v) have a larger volume root ball, with the roots being thicker and more developed when compared to the rest of the treatments. The LFH treatments were removed from root weight analysis due to contamination of the roots by humus layer. There was no significant difference found in root length (P=0.054) and root:shoot ratio (P=0.606) in this study.

Figure 5. Annual Ryegrass biomass after 8 weeks of growth.

Soil Monitoring Probes

The application of a surface LFH (5 cm thick) induced the highest average Volumetric Water Content (VMC) at 0.164 m^3/m^3, with the application of Sulphur Prills at 40 kg ha^{-1} having lowest VMC at 0.024 m^3/m^3 (Table 4). The soil solution EC (mS/cm) for N-Rich® treatments was the highest of all experimental treatments, with N-Rich® 34 t ha^{-1} application rate being 0.393 mS/cm, and the Sulphur Prills (40 t ha^{-1}) and the 80% OM Technosol control being the lowest in the study at 0.010 mS/cm. Higher electrical conductivity is indicative of higher dissolved nutrient ion content in the soil solution. The maximum temperature throughout the treated Technosols were obtained by the 80% OM Technosol without any vegetation at 27.1°C being °C 9.5°C higher than the growth chamber set point. The treated soil with vegetation to attain the highest increase in temperature was the Sulphur Prills 100 kg ha-1 application at 25.6°C, which is 7.5°C higher than the parent material.

Water Samples

The leachates from the N-Rich® treatments had the highest electrical conductivity, with the lower rate of N-Rich® at 17 t ha^{-1} attaining an extremely high 59.76 mS/cm when comparing to

0.34 mS/cm parent material. The lowest maximum electrical conductivity was observed with the 5cm LFH on the surface of the 80% OM Technosol at 0.16 mS/cm. When comparing the leachate from the treated Technosols to the parent material 80% OM Technosol, the pH remained relatively stable with applications of treatments (7.0-7.4).

Plant Nutrients

A composite soil sample of each treatment was analyzed for routine soil fertility parameters at A&L Laboratories (London, Ontario) in order to measure the impact of the individual amendments on the extractable soil nutrient levels. The two rates of N-Rich® (17 t ha^{-1} and 34 t ha^{-1}) showed the greatest overall available nutrient increase when compared to the rest of the treatments, with the higher rate of N-Rich® (34 t ha^{-1}) obtaining the highest overall increase of nutrient levels and the N-Rich® 17 t ha^{-1} having the second highest increase in overall available nutrient concentrations for 10 out of the 15 tested soil nutrients. N-Rich® 34 t ha^{-1} had the greatest increase in overall available nutrient concentrations in 12 out of the 15 tested nutrients measured in the Technosol samples. Both N-Rich® amendments, however, had a decrease in the extractable Manganese. Overall, the N-Rich® treatments showed the greatest increase in soil nutrients out of the treatments.

Phosphorus (P) and sulphur (S), two key nutrients in soils, were added to the 80% OM Technosol through the N-Rich® and the Sulphur Prill amendments, respectively. Both N-Rich® treatments displayed the greatest increase of phosphorus and sulphur. Copper (Cu) and zinc (Zn) were added at measurable levels in both and Wood Ash treatments, with the largest increase in copper (Cu) being observed with the N-Rich® treatments. The highest increases in extractable zinc was observed for Wood Ash 80 t ha^{-1} and N-Rich® 34 t ha^{-1}, with the greatest increase with the higher Wood Ash application rate. Magnesium (Mg) and calcium (Ca), both being major nutrient elements, were added in higher amounts in the N-Rich® treatments than in the Wood Ash treatments. Both N-Rich® treatments had the greatest increase in both nutrient levels, with the higher rate of N-Rich® 34 t ha^{-1} having the greatest increase. Magnesium (Mg) and Calcium (Ca) were both added in larger amounts in the N-Rich® treatments, as well as in significant levels by the Wood Ash and Sulphur Prill treatments. Magnesium, critical for chlorophyll production in photosynthesis, plays an essential role in carbohydrate metabolism and serves as an activator of the enzymes involved in the synthesis of nucleic acids (Davis and Jamieson, 1983). Calcium is essential for cell division, cell elongation and cell structure (Mahler, 2004).

Nutrient concentrations in vegetation were also analyzed; major and minor nutrients that play key roles in healthy vegetative growth were examined. Boron, Iron, Manganese, Zinc and Copper are minor nutrients that play key roles in plant health. In the shoots, the greatest increase in Zinc was found in Wood Ash 80 t ha^{-1}, the greatest increase in Manganese was found in LFH on surface, Boron had the greatest increase in Sulphur Prills 40 kg ha^{-1}, Iron had the greatest increase with the 10% Natural Soil application. In the roots, the greatest increase in Zinc was found in Wood Ash 80 t ha^{-1} application and the greatest increase in Boron was found in Sulphur Prills 100 kg ha^{-1}. In the shoots, the greatest increase Potassium, Calcium and Phosphorus in the shoots was found in the N-Rich® applications. The greatest increase in Potassium, Calcium, Phosphorus and Magnesium in the roots was also found in the N-Rich® applications.

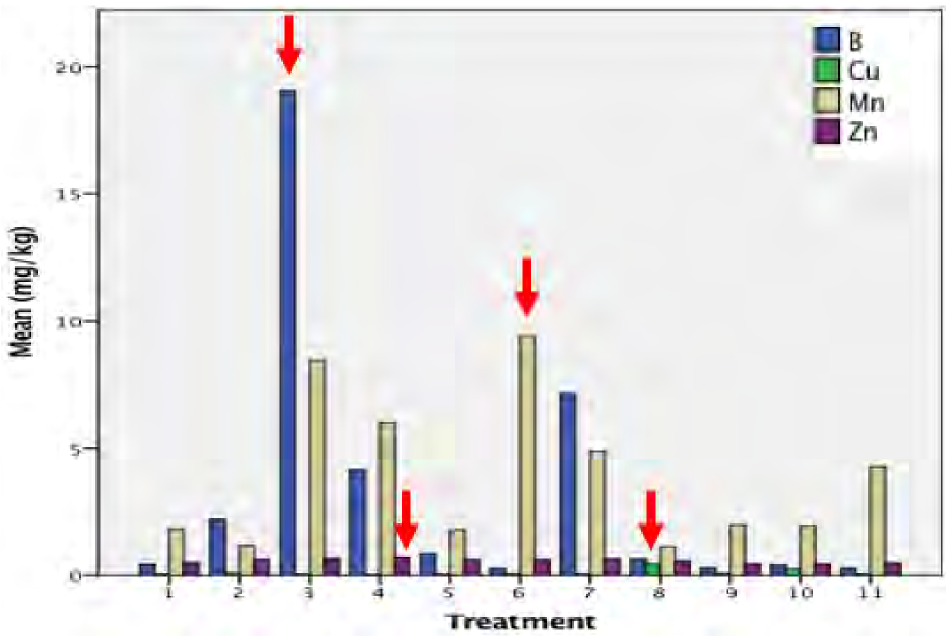


Figure 6. Soil mixtures bioavailable micronutrients

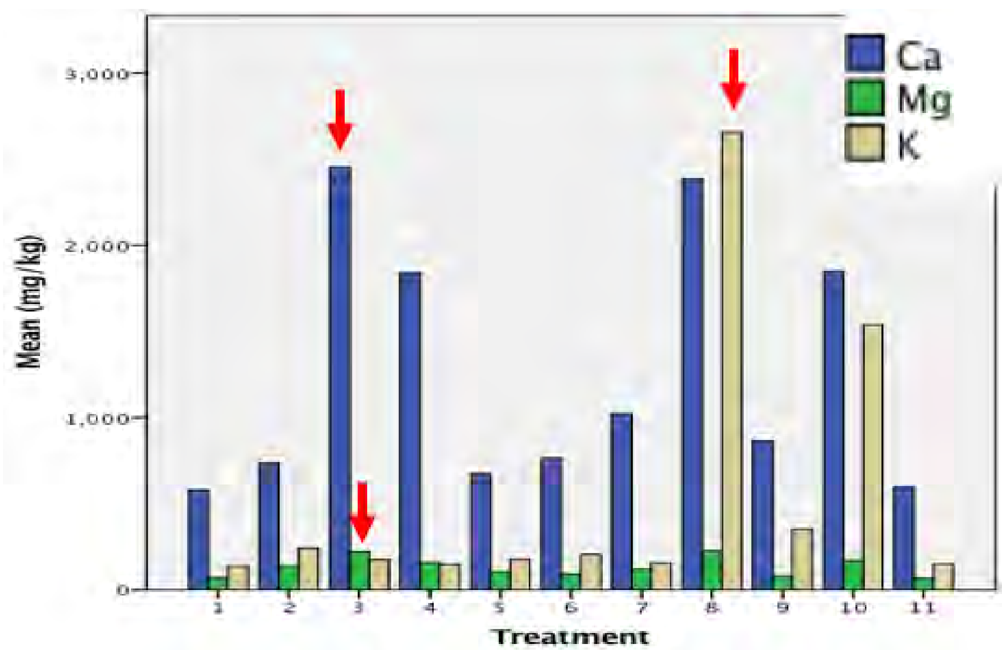


Figure 7. Soil mixtures bioavailable macronutrients.

Discussion

Mine land reclamation is not only environmentally and socially responsible, it is mandated by the government of Ontario (Mining Act, 2012). The Mine Rehabilitation Code (Ontario Mining Act (2012) states that all disturbed sites shall be revegetated and the proponent shall restore their site to its former use or condition (Mining Act, 2012). Annual ryegrass, a short lived grass often used in reclamation, was used in this study to stabilize newly placed soils (Watkinson *et al.*, 2014). Previous studies have used this short lived grass and found successful results for reclamation purposes under different growth conditions (Baker *et al*., 2011)

The increase in shoot biomass and measured root length with the application of Sulphur Prills at 100 kg ha^{-1}) suggests this treatment has a positive effect on plant growth. Thus, the increase in shoot biomass suggests Sulphur Prills 100 kg ha^{-1} is a suitable growth promoting amendment for the 80% OM Technosol manufactured from woody residuals and crushed mine rock to enhance successful plant growth. Even though there was no significant difference found in the root biomass of the ryegrass throughout the treatments, there was an observable difference in root architecture in Wood Ash 40 t ha^{-1}, Sulphur Prills 100 kg ha^{-1} and 10% Stockpiled Humic Gleysol treatments. The root system observed in these specific treatments was a lot thicker, and a lot more developed when compared to the other treated Technosols (see Plate ??).

Additions of specific amendments successfully elevated bioavailable nutrient levels in the 80% OM Technosol to levels more suitable for successful plant growth. Wastes produced by industry, municipalities and households (woody residuals, paper sludge and pulp, sewage sludge, and municipal solid wastes) can contain a substantial amount of organic material that can be used to return essential plant nutrients to soil (Nason *et al.,* 2007). Extractable phosphorus (P) and sulphur (S) are two key components in the amended soils added at measurable levels through different amendments; the different rates of Sulphur prills provided sulphur (S) to the 80% OM Technosol, and the different rates of N-Rich® provided additional phosphorus (P) to the 80% OM Technosol. Both of these nutrients provide important components to the soil for successful plant growth, with P playing an essential role in healthy plant production as a component of key molecules such as nucleic acids, phospholipids and ATP energy requiring processes in plant metabolism (Schachtman *et al.,* 1998; Grant *et al*., 2001). Sulphur (S) also plays an essential role in plant growth, such as in plant defense against biotic and abiotic stress and for the overall quality of the crops. Sulphur (S), a macronutrient for plants, is a component of some amino acids that form proteins, and is a core component of plant protoplasts and enzymes (Mahler, 2004). Although micronutrients are required by plants in smaller amounts, certain elements such as Al, Mn, Cu, Zn and Pb can severely limit plant growth when found in high concentrations in solution (Bradshaw and Chadwick, 1980). Copper (Cu) and zinc (Zn) are nutrients added in measurable amounts in the N-Rich® and Wood Ash treatments, with the greatest increases in Cu being observed in the N-Rich® treatments. Wood ash (80 t ha^{-1}) and N-Rich® (34 t ha^{-1}) promoted the highest increase in extractable zinc (Zn) levels. Micronutrients such as copper (Cu) and zinc (Zn) are required for successful plant growth (Davis and Jamieson, 1983). Copper plays an important role in enzymes (phenolases and ascorbic acid oxidase) and may function in photosynthesis (Davis and Jamieson, 1983) Zinc (Zn) plays a role in protein synthesis; it is also involved in metabolism of plants as an activator of several enzymes (Davis and Jamieson, 1983).

The amendments added to the Technosol enhanced soil nutrient availability, providing components needed to produce a viable soil medium for successful plant growth, with specific amendments having a greater effect on increases of nutrients in the soil. The N-Rich®

applications generally increased available nutrient levels the most in the Technosol being tested in this study. These results suggest that, when handled and mixed correctly, organic wastes could be diverted from the landfill to improve the manufacture soils for use as cover soils or growth media to help restore ecosystems on post-industrial sites (Watkinson *et al.*, 2014).

Conclusion

The blending of mine waste products such as the crushed mine rock produced in open pit mining activities with lumber mill waste products as organic matter (woody residuals) can manufacture a soil for use to promote successful plant growth for the mine site reclamation process. Different treatments provide different positive effects to the Technosol. For example, one amendment can increase available nutrients in the Technosol and another can stabilize the pH. The use of regional waste products eliminates potential disturbances to surrounding healthy ecosystems, thus minimizing the devastation of healthy land through storage of waste products which promotes the leaching of materials potentially harmful to the environment. By obtaining most of the components needed for a manufactured Technosol in close proximity to the mine site, the Carbon Footprint could also be reduced. This research project has provided further information on how to improve manufactured soils in an environmentally sustainable way and also contributed to current information on land reclamation techniques to prepare for future reclamation initiatives.

Literature Cited

Baker, L.R., White, P.M. and Pierzynski, G.M. 2011. Changes in microbial properties after manure, lime, and bentonite application to a heavy metal-contaminated mine waste. Appl. Soil Ecol. 48(1): 1–10.

Barrick Gold Corporation. 2014. Operations, Canada: Hemlo. <http://www.barrick.com/operations/canada/hemlo/default.aspx>

Bradshaw, A.D., and Chadwick, M.J. 1980. The restoration of land: The ecology and reclamation of derelict and degraded land. Blackwell, Oxford, U.K. 28 pages.

Brown, R.L. and Naeth, M.A. 2014. Woody debris amendment enhances reclamation after oil sands mining in Alberta, Canada. Restor. Ecol. 22(1): 40–48.

Canadian Regional Climate Model (CRCM). Environment Canada. Last modified: 2014-03-03. http://www.ec.gc.ca/ccmac-cccma/default.asp?lang=en&n=82DD0FCC-1.

Davis, Fred and Jamieson, Glen. 1983. Micronutrients and Plant Nutrition. Journal American Rhododendron Society. 37(1).

Food and Agriculture Organization of the United Nations. 2014. International soil classification system for naming soils and creating legends for soil maps.

Gaster J, Karst J, Landhäusser SM. 2015. The role of seedling nutrient status on development of ectomycorrhizal fungal communities in two soil types following surface mining disturbance. Pedobiologia 58: 129-135

Grant, C.A., Flaten, D. N., Tomasiewicz, D.J., Sheppard, S.C. 2001. The importance of early season phosphorus nutrition. Canadian Journal of Plant Science. 81(2): 221-224.

IBM Corp. Released 2011. IBM SPSS Statistics for Windows, Version 20.0 Armonk, NY: IBM Corp.

Leguedois, S., Sere, G., Auclerc, A., Cortet, J. 2016. Modelling pedogenesis of Technosols. *Geoderma, 262,* 199-212.

Mackenzie, D.D., and Naeth, M.A. 2010. The role of the forest soil propagule bank in assisted natural recovery after soil sands mining. Restor. Ecol. 18(4): 418-427

MacKenzie, M. D. and S. A. Quideau. 2011. Laboratory Based Nitrogen Mineralization and Biogeochemistry of Two Soils Used in Oilsands Reclamation. Canadian Journal of Soil Science 92: 131-142.

Mahler, Robert L. 2004. Nutrients Plants Require for Growth. University of Idaho: College of Agricultural and Life Sciences.

Sorenson, P.T., S.A. Quideau, M.D. MacKenzie, S. Landhäusser, and S.W. Oh. 2011. Forest floor development and biochemical properties in reconstructed boreal forest soils. Applied Soil Ecology 49: 139-147.

Watkinson, A., Hayes, S, Lock, A., Beckett, P.J., and G.A. Spiers. 2014. Developing manufactured soils for reclamation of mined lands on the Canadian Shield. Canadian Land Reclamation Society Annual Meetings, Timmins, ON. June 26-29, 2016.

Vanderhorst, R., Watkinson, A., Beckett, P.J, and G.A. Spiers. 2016. Technosol evaluation for mine site reclamation on the boreal shield. Canadian Land Reclamation Society Annual Meetings, Timmins, ON. June 26-29, 2016.

A NOVEL APPROACH TO ASSESSING THE CAUSE OF IMPACTS ON BENTHIC MACROINVERTEBRATE COMMUNITIES, AND DISTINGUISHING BETWEEN SEDIMENT VS WATERBORNE EFFECTS DOWNSTREAM OF A METAL MINING DISCHARGE

Helga E. Sonnenberg[1] and Lisa D. Rozon-Ramilo[1]
Stantec Consulting Ltd., Guelph, ON, N1G 4P5

Key Words: Metal Effects Addition Model (MEAM), Biotic Ligand Model (BLM), Sediment and Aquatic Toxicity Testing, *Hyalella azteca,* Investigation of Cause (IOC), metal bioaccumulation, environmental site management

Hudbay has owned and operated a base metal mine and metallurgical complex in the Flin Flon area of northwestern Manitoba for over 90 years. Zinc and copper ores are processed from local mines and concentrates from external sources. The Flin Flon tailings impoundment system (FFTIS) receives input from several processes, such as mill effluent waste, dewatering from underground workings, surface runoff and storm water drainage, and runoff from the town of Creighton. FFTIS effluent is discharged into Flin Flon Creek, where it comprises approximately 70% of the creek volume. The effluent concentration decreases as it flows into the Northwest Arm of Schist Lake, where it is approximately 30%.

Benthic invertebrate community surveys downstream of the FFTIS have shown that benthic invertebrate communities in Flin Flon Creek are severely impacted and dominated by a single pollution-tolerant species: the midge larva *Chironomus* (Stantec 2005, 2007, 2009, 2011). In the NW Arm of Schist Lake, benthic communities are also impacted but not to the same extent.

An integrated assessment of multiple lines of evidence including toxicity testing and modeling approaches was carried out to assess metal bioavailability and causes of toxicity on benthic invertebrate communities. Assessment of Flin Flon Creek and the NW Arm of Schist included extensive water and sediment chemistry, assessment of simultaneously extracted metals/acid volatile sulfides (SEM/AVS), and benthic community composition. A novel approach to toxicity testing was conducted in the field and laboratory using *Hyalella azteca* (*Hyalella*), and bioaccumulation of metals in all surviving test organisms was measured. In addition, wild chronically exposed *Hyalella* were collected for metals analysis from both areas. Finally, the Metal Effects Addition Model (MEAM) and Biotic Ligand Models (BLM) were applied to assess metal speciation and predict toxicity to daphnids and fish.

In the Flin Flon Creek exposure area, aluminum exceeded the CCME WQG 60% of the time and cadmium, copper, iron, selenium and zinc exceeded the CCME WQGs 100%. In the NW Arm of Schist Lake, cadmium, copper, iron and zinc were also above guideline values 26% to 59% of the time, and selenium 100%. Relative to background, selenium concentrations were highest followed by copper and cadmium. For reference, Table 1 shows the water quality guidelines and results.

The biotic ligand model was used to predict the toxicity (LC50) of copper, zinc, and cadmium for fish and daphnids under site-specific conditions in Flin Flon Creek and the NW Arm of Schist Lake. Overall, metal toxicity to fish and daphnids decreased in the order Cu>Zn>Cd. Comparison of predicted LC50 values to site concentrations, along with long term pH values in Flin Flon Creek, indicated that daphnids could be affected by copper, especially at lower pH levels in Flin Flon Creek. Metal speciation confirmed that the observed cadmium and zinc concentrations are unlikely to be the cause of acute toxicity in Flin Flon Creek, whereas it appears that copper could at times be bioavailable, particularly at low pHs. Farther downstream in Schist Lake, the model did not predict Cu, Zn or Cd toxicity to fish or daphnids.

Metals in sediment followed a similar trend to that observed in water. Most of the guideline exceedances occurred in Flin Flon Creek, where CCME Interim Sediment Quality Guideline Probable Effect Levels (ISQG PEL) for As, Cd, Cr, Cu, Pb, Hg, and Zn were exceeded. In the NW Arm of Schist Lake, 90% of the samples exceeded the ISQG PEL for cadmium, and 100% of the samples exceeded the ISQG PEL for zinc. Copper, arsenic, and mercury concentrations were also higher than the ISQG, but not above the PEL, with exceedences occuring 81%, 29%, and 29% of the time, respectively. All metals in the sediment from the reference area were below the ISQG PEL. For reference, Table 2 shows the water quality guidelines and results.

Sediment core samples were collected from exposure and reference areas to determine the SEM/AVS ratios and further assess whether specific free metals (Ag, Cd, Cu, Ni, Pb, and Zn) are bioavailable in pore water at sufficient levels to cause toxicity or be linked to metal bioaccumulation in biota. The SEM/AVS molar ratios can be used to estimate sediment porewater concentrations of divalent metals and provide a better indication of sediment toxicity than total metals analyses on bulk sediment (DeWitt et al. 1996; Hansen et al. 1996a, 1996b).

The $SEM^*_{x,OC}$ values for cadmium, copper, nickel, zinc, and lead was calculated according to Di Toro et al. (2005), and were normalized for organic carbon, which was very high in Flin Flon Creek. A threshold value of >100 µmol/g was used to predict whether metal-toxicity might be expected. The range of $SEM^*_{x,OC}$ in Flin Flon Creek was 430 to 988 µmol/g C-org, and Schist Lake ranged from 13 to 219 µmol/g C-org with three of five samples being greater than the threshold value of 100 µmol/g. Based on SEM/AVS results, zinc was identified as the predominant metal contributing to sediment toxicity in both areas. Redox values in Flin Flon Creek ranged from -153 to -278 mV and from -205 to -236 mV in Schist Lake indicating that the sediment in both areas is anoxic and highly reduced.

To distinguish between sediment and waterborne toxicity, a novel experimental design using slightly modified standard sediment toxicity test methods for *Hyalella azteca* was carried out in the laboratory under controlled conditions and in the field, thereby integrating the complexity of receiving environment. In the laboratory, water only tests were also conducted (in addition to sediment toxicity tests) with receiving water to further help separate sediment vs waterborne toxicity.

Exceedances of water and sediment quality guidelines provided an initial screening tool for evaluating toxicity in the receiving environment, as do models, but they do not conclusively demonstrate that a metal is causing adverse effects in aquatic biota. Guidelines are based on laboratory data, with safety factors applied to the most protective toxicity test results. In contrast, field conditions that affect metal bioavailability and toxicity are far more complex. In the natural environment, metals also exist within a mixture. Table 3 provides and overview of the factorial experimental design that was implemented, showing each treatment (i.e. how different types of overlying water were combined with the various sediment samples collected from the same location). At the end of the toxicity tests, surviving organisms from each treatment (and wild *Hyallela*) were analyzed for metals and used as inputs to the metal effects addition model (MEAM).

Sediment from Flin Flon Creek resulted in complete (100%) mortality of *Hyallela* in lab toxicity test exposures under controlled conditions. *Hyallela* did survive test exposures with reference or lab sediment and overlying Flin Flon Creek water, and actual survival (76 to 94%) was better than predicted (16% to 23%). However, had exposures been prolonged, mortality would have likely increased given that growth was significantly reduced, and both copper and selenium exceeded their LBC25s (Table 3). The *in situ* caged *Hyalella* toxicity tests deployed in Flin Flon Creek resulted in 100% mortality, with or without Flin Flon Creek sediment.

Survival of *Hyalella* in aqueous-only toxicity tests with Flin Flon Creek water was 70%; however, growth was significantly lower than the lab controls, confirming sublethal effects from Flin Flon Creek water. The water-only tests were conducted in conjunction with sediment toxicity tests to further distinguish between historical contamination associated with sediment vs current site water/effluent effects (Environment Canada 2013). Based on results of the biotic ligand model and toxicity tests, it appears that Flin Flon Creek sediment and receiving water can cause toxicity, however; effects are associated moreso with sediment (i.e. historical contamination and site sediment condition).

Sediment from Schist Lake exhibited survival rates of that ranged from 64% to 96% in lab tests, and from 63 to 85% in field toxicity tests. When *Hyalella* was exposed to Schist Lake sediment or surface water alone, metals did not accumulate to levels that exceeded the LBC25s. However, when they were exposed to Schist Lake sediment and water together, in the lab or field, selenium and zinc bioaccumulation was higher than the LBC25s. Wild *Hyalella* accumulated selenium to the LBC25. Actual survival was always better than predicted, indicating that other factors, possibly due to acclimation of the organisms, genetic divergence among *Hyalella azteca* clades and other factors that modify metal bioavailability, uptake and toxicity.

The ability to model the mixture of metals under various scenarios provides insight into metals that may be causing observed benthic community effects. The MEAM (Norwood et al. 2013) was used with one modification, to account for 14 day rather than 28 day exposures. *Hyalella* tissue metal concentrations from the lab and field toxicity test treatments were used as inputs to the model, as well as bioaccumulation data from wild *Hyalella* collections carried out in 2011 and 2014. Results showed that potentially copper or selenium is causing toxicity in Flin Flon Creek. In Schist Lake, metals potentially causing toxicity are selenium and zinc.

Summary

Flin Flon Creek is devoid of resident fish populations and benthic invertebrate communities are severely impacted. Sediment in the creek is anaerobic and conducive to anaerobic microbiological activity as demonstrated by very low redox values. In general, the highly reduced conditions in Flin Flon Creek sediment may contribute to accelerated migration of metals, which supports the SEM/AVS results showing that metals in the sediment of Flin Flon Creek could cause toxicity. Toxicity in Flin Flon Creek to *Hyalella* and potentially other aquatic biota appears to be associated with both sediment and water, But moreso the historical contamination associated with the sediment. The receiving environment is complex but it appears that under certain conditions (especially low pH, high temperature, and low oxygen levels), metals become mobilized and available for uptake. The water column and sediment are in a constant state of flux, a situation that is not fully understood and is being further investigated. However, the weight of evidence to date points to copper, selenium and zinc associated with the sediment and water as being the metals preventing *Hyalella* (or other aquatic biota) from becoming established in Flin Flon Creek.

Benthic invertebrate communities in the NW Arm of Schist Lake are moderately impaired. Results from lab and field based sediment toxicity tests conducted with Schist Lake sediment revealed that survival was > 65% and growth was actually significantly higher compared to the lab controls. Schist Lake water or sediment alone does not appear to cause toxicity. MEAM modeling with bioaccumulation data from lab and field toxicity tests revealed that selenium and zinc accumulated to LBC25 values. However, actual survival was typically better than predicted. Wild Hyalella azteca also accumulated selenium and zinc to LBC25s, indicating that wild populations have acclimated to elevated metal levels, or different local species of *Hyalella* may exhibit different responses.

Table 1: Summary of water quality guideline exceedances for metals in exposure area waterbodies (2004-2014)

Parameter	Fraction	Unit	Guidelines		Total Number of Samples	Sample Values Not Meeting Guidelines				Descriptive Statistics			
						CCME WQG		MB PSOG[1]		Concentrations			
			CCME WQG[a]	MB PSOG[b]		Number Exceeding	Percent	Number Exceeding	Percent	Min.	Max.	Mean	Std. Dev.
Flin Flon Creek (upstream of former Trout Lake Mine discharge)													
Aluminum	T	µg/L	5	5	5	3	60%	3	60%	49	101	71.4	21.1
Arsenic	T	µg/L	5	150	5	1	20%	0	0%	3.1	6	4.6	1.1
Cadmium	T	µg/L	0.37	1.1	5	5	100%	2	40%	0.61	1.88	1.16	0.46
Copper	T	µg/L	4	60	5	5	100%	2	40%	15.9	89	48	34.5
Iron	T	µg/L	300	300	5	5	100%	5	100%	454	640	547	89.7
Selenium	T	µg/L	1	1	5	5	100%	5	100%	6.5	46	30	16
Zinc	T	µg/L	30	750	5	5	100%	0	0%	86	390	231	127
Northwest Arm of Schist Lake													
Cadmium	T	µg/L	0.37	0.83	17	5	29%	0	0%	0.053	0.60	0.22	0.18
Copper	T	µg/L	4	34	17	10	59%	0	0%	2.0	13	5.9	3.4
Iron	T	µg/L	300	300	19	5	26%	5	26%	10	400	120	150
Selenium	T	µg/L	1.0	1.0	19	19	100%	19	100%	2.63	10	4.4	2.0
Zinc	T	µg/L	30	420	19	11	58%	0	0%	27	110	57	29

Table 2: Summary of sediment quality guideline exceedances for metals in exposure area waterbodies (2004-2014)

Parameter	Unit	CCME Guideline		Total Number of Samples	Number of Samples Not Meeting Guidelines				Summary Statistics					
					ISQG		PEL		Concentration					
		ISQG[a]	PEL[b]		Number Exceeding	Percent	Number Exceeding	Percent	Min.	Max.	Mean	Med.	Std. Dev.	Geo. Mean
Flin Flon Creek (upstream of former Trout Lake Mine discharge)														
Arsenic	mg/kg	5.9	17	16	16	100%	14	88%	11	1260	297.5	150	376	133.5
Cadmium	mg/kg	0.6	3.5	16	16	100%	16	100%	5	510	115.4	70.5	148.7	54.9
Chromium	mg/kg	37.3	90	16	10	63%	3	19%	17	180	57.9	43	43.3	46.9
Copper	mg/kg	35.7	197	16	16	100%	16	100%	214	14000	4098.8	2455	4293.2	2192.7
Lead	mg/kg	35	91.3	16	13	81%	11	69%	19	739	309.4	221	277.7	166.6
Mercury	mg/kg	0.17	0.486	16	13	81%	11	69%	0.09	6.9	1.8	0.97	1.91	0.96
Selenium	mg/kg	none	none	16	n/a	n/a	n/a	n/a	19	1050	290	-	338	-
Zinc	mg/kg	123	315	16	16	100%	16	100%	472	33500	9428.3	5490	9721.8	5479.8
Northwest Arm of Schist Lake														
Arsenic	mg/kg	5.9	17	21	6	29%	0	0%	3	12	5.74	4.85	2.81	5.21
Cadmium	mg/kg	0.6	3.5	21	21	100%	19	90%	1.8	25.4	7.81	6.25	5.31	6.53
Copper	mg/kg	35.7	197	21	17	81%	0	0%	29	190	78.7	58.5	47.2	67.1
Mercury	mg/kg	0.17	0.486	21	6	29%	0	0%	0.025	0.27	0.105	0.051	0.083	0.074
Selenium	mg/kg	None	None	21	n/a	n/a	n/a	n/a	2.6	24.4	7.8	-	6.2	-
Zinc	mg/kg	123	315	21	21	100%	21	100%	430	3790	1511.5	1200	986.9	1282.9

Table 3: Results of field and lab based sediment toxicity tests, bioaccumulation and metal effects addition modeling (MEAM) for Flin Flon Creek and Schist lake exposure scenarios.

Test	Sediment	Overlying Water	Background-Corrected Concentration (nmol/g)							
			As	Cd	Cu	Pb	Se	Zn	Hyalella Survival (%)	
		LBC25x24hr	83	585	1850	650	72.4	938	Predicted	Actual
Lab 2011	Flin Flon Creek	Lab	No Survival						n/a	0
		Cran-REF	No Survival						n/a	0
		Flin Flon Creek	No Survival						n/a	0
	Cranberry-REF[a]	Flin Flon Creek	26	0.8	**1992**	5.8	**178**	150	22.5	94.0
	Lab	Flin Flon Creek	4.7	0	**1685**	0.1	**220**	0	16.3	76.0
In situ Caged 2011	Flin Flon Creek	Cran-REF	16.1	0	0	23	20	111	63.4	32.0
		Flin Flon Creek	No Survival						n/a	0
	Cranberry-REF	Flin Flon Creek	No Survival						n/a	0
Wild 2011	Flin Flon Creek		No *Hyalella* in Flin Flon Creek						n/a	0
Lab 2011	Schist Lake	Lab	6.8	0	0	5.5	34.6	337	70.7	86.0
		Cran-REF	8.9	0.2	663	1.5	41.4	45.7	69.5	64.0
		Schist Lake	6.2	7.7	428	1.7	**90.4**	398	53.0	97.8
	Cranberry-REF	Schist Lake	5.3	5.9	0	1.6	35.1	320	72.4	93.3
	Lab	Schist Lake	0	0	0	0	33.8	0	72.7	77.8
In situ Caged 2011	Schist Lake	Cran-REF	14.4	0	140	8.8	6.8	0	80.4	63.0
		Schist Lake	14.3	0	855	16.5	54.7	**1017**	23.6	85.0
	Cranberry-REF	Schist Lake	15	0	180	6.1	29	368	74.1	60.0
Wild	Schist Lake 2011 (n=3)		24.5	8.7	0	16.8	**116**	**556**	34.0	na[b]
	Schist Lake 2014 (n=8)		4.5	0	0	4.8	**63.0**	**249**	54.9	na
	Schist Bay (n=3)		20.9	0	0	5.2	**79.1**	206	58.6	na
	Mirond Lake 2011 (n=1)		4.0	1.9	0	4.5	6.2	104	76.5	na
	Mirond Lake 2014 (n=8)		0	0	0	0.8	2.4	0	77.0	na

Note: All data corrected by Norwood et al (2013) background concentrations

[a]REF – Reference Area

[b] na – not applicable since wild *Hyalella* collected

Bold – indicates mean values exceed the LBC25

Bold – indicates one or more replicates exceeds the LBC25

Acknowledgements

We would like to thank Hudbay Minerals for their ongoing support for this work. We would also like to thank Warren Norwood from CCIW for developing the MEAM model and making it available for us to use; Lesley Novak and Martina Rendas of Aquatox Laboratories for conducting the laboratory toxicity experiments, Malcolm Stephenson for providing expertise and guidance on the *in situ Hyalella* cages; Nikolay Sidenko for running the BLM model and to our many colleagues at Stantec with which we could not have conducted this research Trevor Chandler, Joel Keene, Jess Sosa Campos and Ed Malindzak.

Literature Cited

DeWitt, T.H., Swartz, R.C., Hansen, D.J., McGovern, D., Berry, W.J. 1996. Bioavailability and chronic toxicity of cadmium in sediment to the estuarine amphipod *Leptocheirus plumulosus*. Environmental Toxicology and Chemistry 15: 2095-2101.

Di Toro, D.M., McGrath, J.A., Hansen, D.J., Berry, W.J., Paquin, P.R., Mathew, R., Wu, K.B., Santore, R.C. 2005. Predicting sediment metal toxicity using a sediment biotic ligand model: Methodology and initial application. Environmental Toxicology and Chemistry 24: 2410-2427.

Environment Canada. 2013 (EPS1/RM/33). Biological Test Method: Test for Survival and Growth in Sediment and Water Using the Freshwater Amphipod *Hyalella azteca*. Science and Technology Branch. Communication Services, Environment Canada, Ottawa, Ontario.

Hansen, DJ, Mahony JD, Berry WJ, Benyi SJ, Corbin JM, Pratt SD, Di Toro DM and Able MB. 1996a. Chronic effect of cadmium in sediments on colonization by benthic marine organisms: An evaluation of the role of interstitial cadmium and acid-volatile sulfide in biological availability. Environ. Toxicol. Chem. 15:2126–2137.

Hansen, D.J., Berry, W.J., Mahony, JD., Boothman, W.S., Di Toro, D.M., Robson, D.F., Ankley, G.T., Ma, D., Yan, Q., Pesch, C.E. 1996b. Predicting the toxicity of metal-contaminated field sediments using interstitial concentrations of metals and acid-volatile sulfide normalizations. Environmental Toxicology and Chemistry 15: 2080-2094.

Norwood, W.P., Borgmann, U., Dixon, D.G., 2013. An effects addition model based on bioaccumulation of metals from exposure to mixtures of metals can predict chronic mortality in the aquatic invertebrate hyalella azteca. Environ. Toxicol. Chem. 32, 1672–1681.

Stantec Consulting Ltd. (Stantec). 2004. Metal Mining Environmental Effects Monitoring – FFTIS and Trout Lake Mine Initial Monitoring Program Study Design. Report for Hudson Bay Mining & Smelting Co. Ltd.

Stantec Consulting Ltd. (Stantec). 2005. Metal Mining Environmental Effects Monitoring – FFTIS and Trout Lake Mine Initial Monitoring Program. Report for Hudson Bay Mining & Smelting Co. Ltd. Report No. 162603894.

Stantec Consulting Ltd. (Stantec), 2007. Flin Flon Tailings and Impoundment System and Trout Lake Mine Environmental Effects Monitoring – Periodic Monitoring Study Design 36 pp + Appendices.

Stantec Consulting Ltd. (Stantec). 2008. Metal Mining Environmental Effects Monitoring – FFTIS and Trout Lake Mine Periodic Monitoring Program. Report for Hudson Bay Mining & Smelting Co. Ltd. Report No. 16096034.

Stantec Consulting Ltd. (Stantec). 2010. Flin Flon Tailings Impoundment System and Trout Lake Mine Focused Monitoring Program. Report for Hudson Bay Mining & Smelting Co. Ltd. Report No. 160960453.

Stantec Consulting Ltd. (Stantec). 2011. Flin Flon Tailings Impoundment System and Trout Lake Mine Environmental Effects Monitoring – Investigation of Cause Part 1 Study Design. Report for Hudson Bay Mining & Smelting Co. Ltd. Report No. 160960644.

Stantec Consulting Ltd. (Stantec). 2012. Flin Flon Tailings Impoundment System and Trout Lake Mine Environmental Effects– Investigation of Cause. Report for Hudson Bay Mining & Smelting Co. Ltd.

Stantec Consulting Ltd. (Stantec). 2013. Flin Flon Tailings Impoundment System and Trout Lake Mine Environmental Effects– Investigation of Cause Part 1 Monitoring Program. Report for Hudson Bay Mining & Smelting Co. Ltd. Report No. 160960644.

SURFACE WATER ARSENIC CONCENTRATION AND LOADING TRENDS IN THE HISTORIC COBALT MINING CAMP, NORTHEASTERN ONTARIO

Anthony Story[1], Maria Story[1], Jean Cayouette[2], Jean-François Doyon[2], Josée Noël[2], and Michel Julien[3]

[1]Story Environmental Inc., Haileybury, ON, P0J 1KO; [2]Agnico Eagle Mines Limited, Rouyn-Noranda, QC, J0Y 1C0; [3]Agnico-Eagle Mines Limited, Toronto, ON, M5C 2Y7.

Key Words: Farr Creek, watershed, rehabilitation, reclamation

Introduction

Since 2013, Agnico Eagle Mines Limited (Agnico Eagle) and Story Environmental Inc. (SEI) have monitored surface water in the historic Cobalt/Coleman Mining Area. This study focuses on arsenic concentrations and loadings in the 60 square kilometre Farr Creek watershed. Numerous historic mining properties, not all owned by Agnico Eagle, contribute arsenic loadings to the watershed.

The Cobalt/Coleman Mining Area is considered the birthplace of mining in Ontario, with the first silver veins discovered in 1903. By the 1970s, most mining operations had shut down, with some activity continuing in the Cart Lake and Cobalt Lake sub-watershed into the 1980s. In 1989, all mining and milling in the Farr Creek watershed ceased. Rehabilitation work on Agnico Eagle's properties took place in the 1990s, as part of approved closure plans.

Arsenic Concentration Trend

Multi-decade time series of total arsenic concentrations were compiled from Farr Creek sites near the watershed outlet. Data sources included the SEI monitoring campaigns, previous consulting studies (Beak, 2002 and LGL, 2010), and historic data from the Ontario Ministry of Environment (MOE, 1968 to 1996).

The MOE's historic water sampling site on Farr Creek was located in North Cobalt (Provincial Water Quality Monitoring Network station 18737000102). This is the same location as SEI's site CR-10. Site CR-10 is approximately 2 kilometres downstream of site CR-9, which has been sampled more routinely in recent consulting studies (e.g., Beak, 2002). However, both sampling sites are downstream of all former mining sites in the watershed. Sampling in 2013 indicated similar concentrations at both locations.

Near the outflow of the Farr Creek watershed, average total arsenic concentrations have declined at a rate of about 0.01 mg/L per year since the mid-1970s (Figure 1). Average total arsenic concentrations in the creek declined from 0.66 milligrams per litre (mg/L) in 1976-1980 to <0.3 mg/L in 2013-2015.

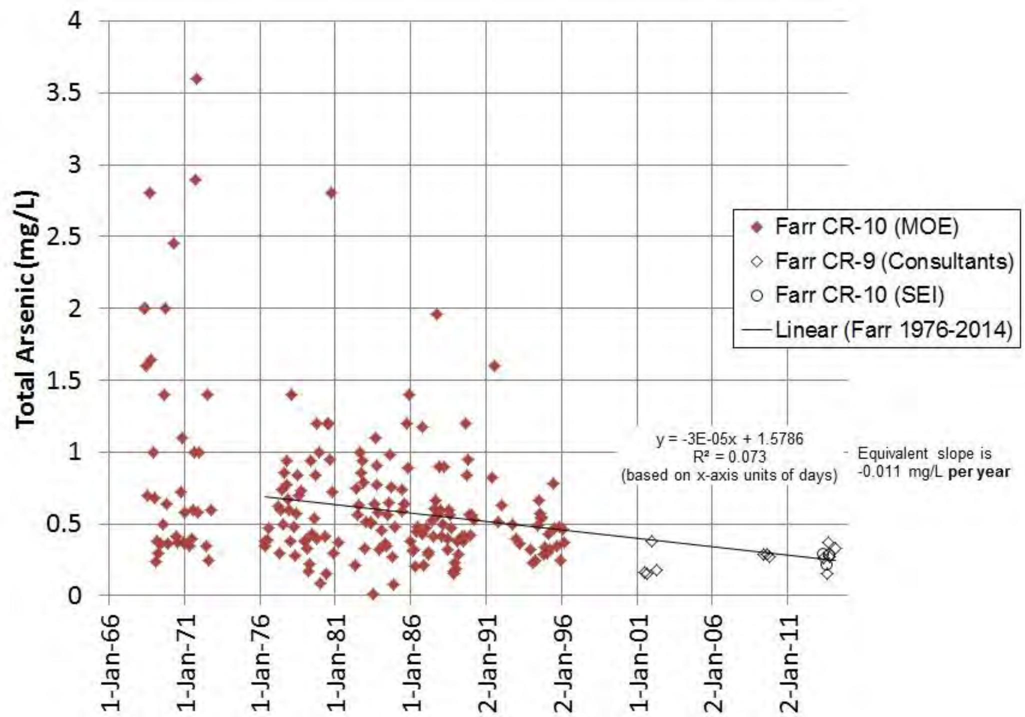


Figure 1 Total Arsenic Concentrations in Farr Creek, 1968-2014

Arsenic Loading

Historical Water Survey of Canada (WSC) flow data for the 1972-1982 period and MOE arsenic data for the 1976-1996 period were used to calculate a flow-weighted historical average annual arsenic loading for Farr Creek (Figure 2). This approach indicated an average total arsenic loading during 1976-1996 of approximately 9700 kg/year (Table 1). Results showed that 48% of the annual arsenic loading occurred in the months of April and May (Figure 2).

Arsenic loadings for the 2013-2015 period were based on 24 data points from site CR-9. Consistent with the historic loading results shown in Figure 2C, 46% of these data (11 of 24) were collected in the important high flow months of April and May. A statistical model of total arsenic concentration based on instantaneous flow in Farr Creek was developed to calculate loadings for 2013-2015. The coefficient of determination (r^2) of the fitted regression model is 0.48.

Applying the regression model to the historic WSC continuous flow records on a daily basis indicated an average annual loading of 4414 kg/year. In other words, assuming that the 11-year record of flow in Farr Creek from 1972-1982 is representative of current flows, a current loading of approximately 4414 kg/year is expected. A plausible upper bound on this mid-range arsenic loading is 5000 kg/year (the 65th percentile of the modelled 2013-2015 annual loadings).

The average total arsenic loading in recent years (2013-2015) has declined by 50% from approximately 9700 kg/year for the 1976-1996 period (Table 1).

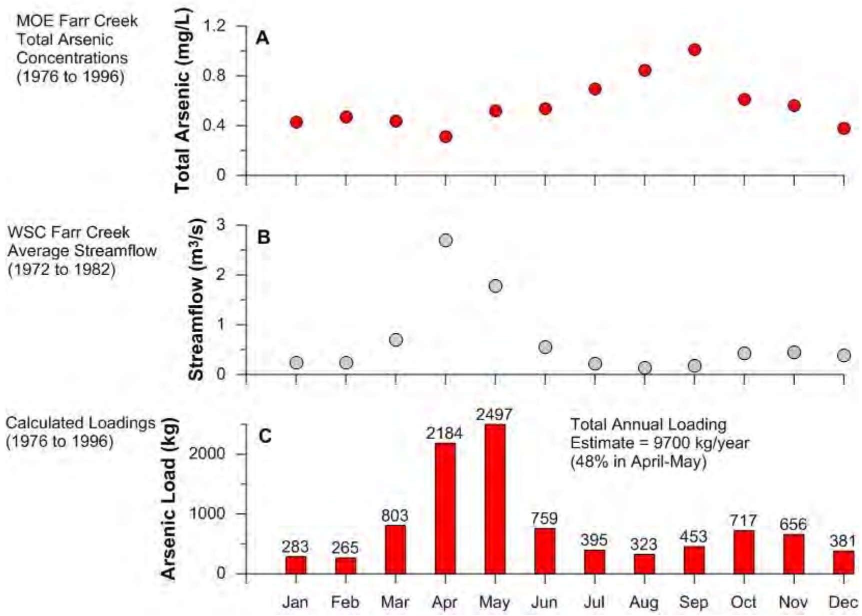


Figure 2 Calculated Monthly Arsenic Loadings in Farr Creek, 1976-1996

Table 1 Farr Creek Annual Arsenic Loadings Estimates

Source	Period	Arsenic Loading Estimate (kg/year)	Notes
SEI (2015)	1976-1996	9700	Flow-weighted load (see Figure 2 above)
SEI (2016)	2013-2015	4400-5000	Flow-weighted load, based on 24 sampling days in 2013-2015, combined with 1972-1982 WSC flow data

Literature Cited

Beak. 2002. Water Quality and In-Stream Loadings – Agnico Eagle Mines Limited Cobalt Area Properties. Report submitted by Beak International Incorporated to Agnico Eagle Mines Limited, Toronto, Ontario.

LGL. 2010. A Chemical and Biological Study to Assess the Receiving Waters in the Cobalt Area, Ontario: LGL Limited Environmental Research Associates. Report submitted to Ontario Ministry of Northern Development, Mines and Forestry.

SEI. 2015. Surface Water Monitoring Report: 2013 and Winter 2014, Cobalt/Coleman Mining Area and Far Field Areas. Prepared by Story Environmental Inc. for Agnico Eagle Mines Ltd., 20 December 2015.

SEI. 2016. Agnico Eagle Mines Limited Consolidation Report: Sources of Arsenic Loadings to Surface Water, Cobalt/Coleman Mining Area. Draft Report prepared for Agnico Eagle Mines Ltd., March 2016.

DEVELOPING THE HOLLINGER OPEN PIT MINE – LEAVING A POSITIVE LEGACY

Thomas Sulatycky[1], Dave Bucar[1], Stephanie Thibeault[1]
[1]Goldcorp Inc., Vancouver, British Columbia, V6C 2X8

Key Words: open pit, rehabilitation, post land use, sustainable value

Introduction

The Porcupine Camp, otherwise referred to as Timmins, Ontario, has a long prosperous history of gold mining. Of all the mines that came and went, the Hollinger Mine was arguably the greatest. Founded in 1909 by Benny Hollinger and his partner Alex Gillies, the mine operated until 1968 producing nearly 20 million ounces of gold. Even when the doors closed, the mine that generated so much prosperity would continue to shine in the eyes of the local community.

In the years that followed however, the site degraded. Other mine operators pillaged any remaining value leaving the property in a state of partial abandonment. As the water levels within the mine rose, subsidence of near surface mine workings occurred creating dangerous public hazards. The property was eventually fenced, locking up 100ha in central Timmins.

Project Sandy

The concept of mining the residual gold left by underground mines is not new within Timmins as it has been done at the Dome Mine and Pamour Mine. Given that the Hollinger Mine was the most prolific mine in Timmins, it was a natural target.

Exploration drilling began as early as 2004 in secret to prevent speculation within the community. Positive results continued to build excitement and, now under the ownership of Goldcorp Canada Ltd. Porcupine Gold Mines (PGM), a pre-feasibility study was launched to look at a potential mining scenario as well as social, environmental, and economic aspects of the project.

PGM began to host a series of open houses and presented different options including keeping the property fenced (current condition), partial remediation to allow some land use, and an open pit mine. Mining would not only “remediate” the hazards within the fence line but also benefit the workforce and the community.

Given the proximity to the community, public engagement became critical to ensure information remained open and transparent. The Hollinger Community Advisory Committee was created and consisted of a diverse group of local stakeholders and worked with PGM to develop strategies that would meet the mutual needs of the community and the company. PGM set up an information centre which was staffed by a full time community liaison coordinator to allow anyone from the community the opportunity to ask questions in person. An online forum was also created through the PGM website to properly address community feedback in a timely manner. A real-time noise and dust monitoring system with data managed by a third party was also implemented to ensure transparency and which can be viewed through the PGM website.

Also of importance was the development of an appropriate subsequent land use plan (SLUP). A series of public consultation meetings were held to gather input from the community on what the property could become once the mine closed.

Unorthodox Project Economics

The project was developing as planned up until 2012. Permitting uncertainty, complexity of remediating voids, and a big downturn in the gold industry created significant challenges and put the project at risk of being shelved. This forced PGM to evaluate the project from a different angle.

If a mine would not go ahead, Goldcorp could not in good faith allow the property to remain fenced in perpetuity and so options to remediate the property were considered, which generally consisted of:

1. Fence property (base case) – annual fencing maintenance, mine hazards remain a risk forever, least acceptable by public.
2. Full/Partial void remediation – very complicated, very expensive, and full of uncertainty. No guarantee that all hazards could be remediated. Option would be more acceptable to public than fencing.
3. Operate to rehabilitate – by mining through the mine workings, the hazard would be eliminated once flooded. Only option that could create a revenue stream to offset mining costs. Most acceptable by the public.

In the fall of 2013, the project was given the green light by the Board of Directors as a rehabilitation project. All permits required were received shortly thereafter and planning to begin construction of the 20m high Environmental Control Berm began which included a highly complex process of evaluating and remediating mine workings which would remain under the weight of the berm. Overburden stripped from the pit would be used to cover the exterior of the berm which would be seeded to create a more aesthetically pleasing view.

As of October 2015, the berm was complete to its final design height which allowed PGM to mine twenty-four hours per day, seven days per week. The regrading, sloping, and seeding of the berm continues throughout the spring of 2016. A final SLUP was presented to the City of Timmins at the end of 2015 and Phase 1 of this plan will also commence in 2016. Based on the current schedule, it is expected that the mine will reach a depth below the predicted flood elevation by mid-2017. The mine is currently scheduled to be in operation until 2019. PGM continues to work with the City of Timmins and the Hollinger Project Community Advisory Committee to work through all phases of the project from concept to reclamation ensuring that this project is successful and meets the long-term needs of the community.

CASE STUDY OF THE REHABILITATION OF THE HISTORICAL CONIAGAS NO. 4 SHAFT IN COBALT, ONTARIO

Jonathon J. L. Taylor[1]

Paul G. Palmer[1]

Sean C. Legassie[1]

Joseé Noël[2]

Jean Cayouette[2]

Jean-Francois Doyon[3]

[1]Golder Associates Ltd., Sudbury, ON, P3C 4Y1

[2]Agnico Eagle Mines Ltd., Val-d'Or, QC, J9P 0E9

[3]Agnico Eagle Mines Ltd., Rouyn-Noranda, QC, J0Y 1C0

Key Words: case study, mine, rehabilitation, shaft, Cobalt, mine opening, concrete, polyurethane foam

The Town of Cobalt, located in northeastern Ontario, Canada was incorporated in 1906 following the construction of the Temiskaming & Northern Ontario Railway and the subsequent discovery of silver in 1903. The Cobalt mining camp produced over 460 million ounces of silver in the half century that followed the discovery. The Coniagas deposit was discovered by W. G. Trethewey in 1904 and sold to Coniagas Mines Limited shortly thereafter. Coniagas Mines operated the claim from 1905 to 1924 when a fire destroyed the concentrator and the No. 2 Shaft headframe. Other small operators mined the Coniagas property sporadically until 1943. The Coniagas property yielded nearly 34 million ounces of silver (Sabina, 1974). The No. 4 Shaft headframe was built in 1914 and the shaft was sunk to a depth of approximately 350 ft. Workings extend as deep as 375 ft near the No. 4 Shaft and are the deepest workings on the Coniagas property.

Following the closure of the No. 4 Shaft, a grocer named Anthony Giachino purchased the building and constructed Giachino's Grocery, enclosing the headframe with the store building in 1926 (Brown, 1999). Figure 1 shows the building in what is estimated to be the 1940's. Giachino used the No. 4 Shaft and the cold air from the mine as a cold storage for produce and meats. Several businesses have since occupied the building including multiple restaurants (Figure 2 and Figure 3), and it is currently in use as a residential apartment and a publisher (Figure 4).

Figure 1: Giachino's Grocery Store – Looking down Prospect Avenue (c.1940's) (Cobalt, 2016)

Figure 2: Coniagas No. 4 Shaft (c.1960's) (Sabina, 1974)

Figure 3: Cornmeals Restaurant (2008)

Figure 4: White Mountain Publishing - Current Tenant (2015)

Investigation

Golder Associates Ltd. (Golder) completed a preliminary investigation with Agnico Eagle Mines Ltd (Agnico Eagle) that consisted of reviewing historical records, and completing a site visit to locate and record the status of the shaft before a detailed investigation was completed. Historical records from previous investigations suggest that the cap was constructed of 18" thick concrete reinforced with 16 lb mine rail and 6-inch square wire mesh, and also indicated that an access port in the top of the cap was left open and covered with planks. The shaft cap was located under the ground level floor of the building inside a storage closet (Figure 5).

Figure 5: Location of shaft cap access port under plywood floor

After removing the plywood floor inside the storage closet, the access port was found to be covered with wood boards. The boards were removed and cool air could be felt coming from the exposed shaft. A light was lowered into the shaft along with a borehole camera. All of the timber shaft guides had deteriorated and fallen into the shaft. At approximately 15 m depth, a significant amount of wood and debris was blocking the shaft (Figure 6). It was not known if the debris was hung up in the shaft with void extending below the debris, or if the shaft was filled from the bottom to the level of the debris.

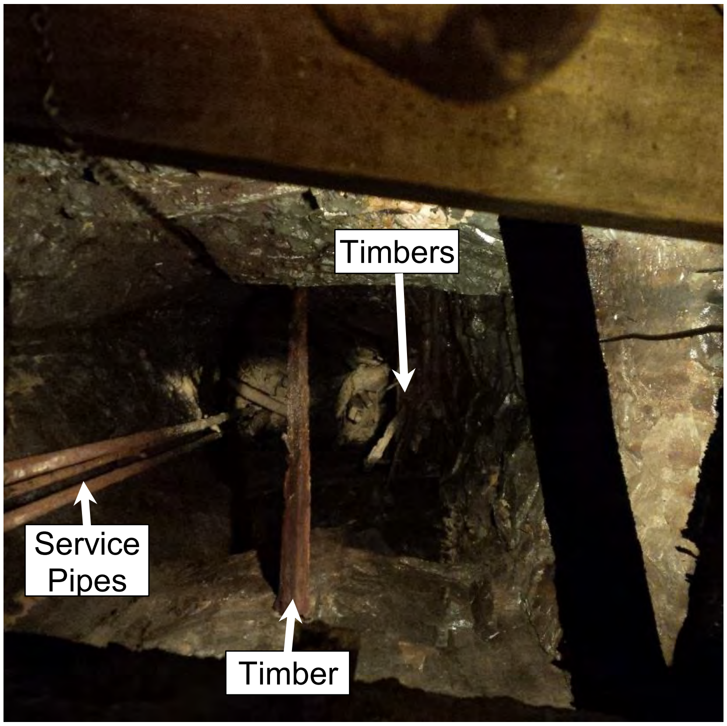


Figure 6: Looking down the shaft from access port. Timbers and other debris are visible at approximately 15 m depth

A cavity monitoring survey (CMS) was completed using a C-ALS borehole deployable laser scanner. The C-ALS probe was lowered 3 m below the building floor into the shaft void on carbon fiber rods designed to keep the probe aligned. The C-ALS creates a point cloud of the void space at assigned degree intervals. Multiple scans were completed of the shaft at different elevations to minimize the portion of the shaft wall which could not be scanned due to blocked line of sight from the scanner. The CMS result confirmed that the shaft was blocked at 15 m depth. The shaft measured approximately 2.5 m by 4.2 m directly below the cap, and narrowed to approximately 2.5 by 3.5 m starting at approximately 1 m depth, from which point, the opening size remained nearly the same to the debris blockage at 15 m. The void space above the obstruction was estimated to be approximately 116 m^3 by creating a wireframe from the point cloud, shown in Figure 7.

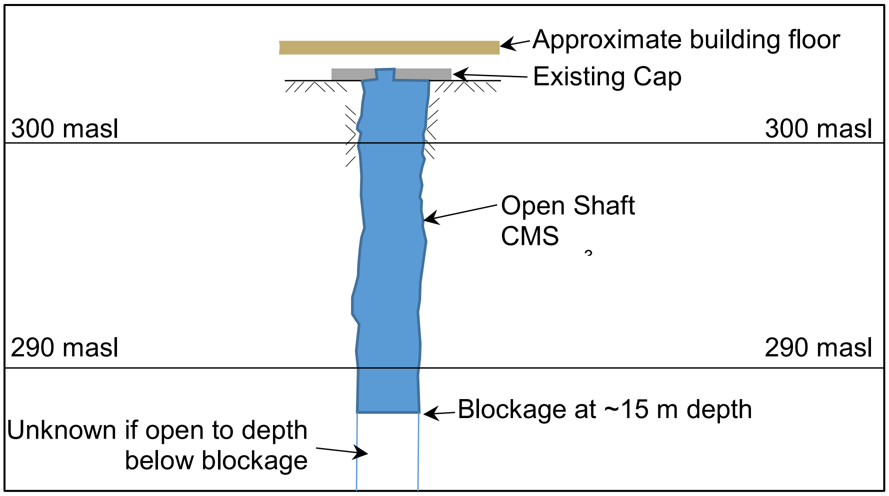


Figure 7: Longitudinal Drawing of Coniagas No. 4 Shaft

Rehabilitation Options

Three rehabilitation options were developed based on the assumption that the shaft void was open below the debris blockage, and are outlined as follows:

Option 1: Spray polyurethane foam on top of the debris blockage to 3 m thick. After the foam cures, pour a 1 to 2 m thick 30 MPa strength concrete layer on top of the foam. Allowing the 30 MPa concrete to gain strength for seven days, fill the remaining void with lower strength concrete designed to be self-supporting

Option 2: Spray 3 m of polyurethane foam on top of a form lowered 6 m into the shaft. After the foam cures, pour 1 m of 30 MPa concrete. Allowing the 30 MPa concrete to gain strength for 7 days, fill the remaining void with lower strength concrete designed to be self-supporting.

Option 3: Place granular backfill on the debris blockage and fill the shaft to 3 m below the existing cap. Pour a self-supporting concrete plug on top of the granular backfill.

The three options vary in their degree of risk versus potential cost savings. Option 1 is the lowest risk as the debris blockage is only used to support the initial spray foam before it cures and serves as the form for the initial concrete plug pour. Option 2 was discussed as a method to reduce the volume and cost of the final concrete plug while maintaining the use of spray foam as the form for the concrete plug. Option 3, identified as the highest risk, would replace the high cost foam with granular backfill and significantly reduce the volume of concrete.

The options were discussed with the Ministry of Northern Development and Mines (MNDM) and the MNDM indicated that all of the proposed options would be acceptable if successful. The high risk of using the debris blockage to support the granular backfill and the logistical problems created by the delivery of backfill into the building and void was not acceptable. It was determined that because of the location of the shaft and marginal cost difference of the rehabilitation between Options 1 and 2, the lowest risk (Option 1) was selected as the rehabilitation.

Rehabilitation Design

The concrete design was completed using the equations outlined in *Analysis and Modelling of Sill Pillars* (Mitchell and Roetteger, 1989) and *Sill Mat Evaluation Using Centrifuge Models* (Mitchell, 1991), referred to as the Mitchell equations. The equations examine several failure mechanisms commonly encountered in the creation of cemented sill pillars in mining. The failure modes considered include flexural (bending), caving, sill shear, and rotational failure.

The failure mechanisms are a function of the geometry of the concrete and void, the weight of the potential load being applied, the strength of the concrete, and other design factors such as friction angle and cohesion. Conservative assumptions were made for parameters that were unknown. The design of the concrete was completed for each of the filling stages; the initial layer of high strength concrete poured on the foam and the lower strength concrete poured on top of the one week old high strength plug. Additionally, the design included consideration of the full column of concrete acting as one unit.

The design of the initial concrete layer was completed to determine the minimum required strength necessary to achieve a minimum Factory of Safety (FOS) of 3 for the potential failure mechanisms considered, and also support the weight of concrete during the second pour. Using the span and width of the shaft obtained from the CMS scans, and assuming a plug height of 1.5 m, the limiting failure mode was identified as rotational failure. The rotational failure mechanism which normally develops in situations with shallow dipping walls was identified as the limiting failure mechanism for the initial plug given the variability in the shaft walls and the relative thickness of the initial plug to the span of the opening. The analysis conservatively assumes that the foam layer will add no support to the initial pour once it has cured and that the initial layer must support the load applied by the concrete during the final stage of backfilling. The analysis showed that the concrete would need to have a minimum required strength of 14 MPa to achieve a FOS of 3. A 35 MPa concrete mix was selected to ensure that the desired strength was met after one week of curing time.

The second concrete layer was designed to consider the strength and thickness of both concrete pour, such that the full column would be self supporting and meet the regulatory requirements for loading. The limiting failure mode was identified as the sliding failure mechanism, which indicated that the cemented backfill would have a FOS of 13 based on a weighted average backfill strength of 10.3 MPa (i.e., 1.5 m of 35 MPa concrete plus 7.1 m of 5 MPa concrete).

Construction of the Rehabilitation

The rehabilitation of the Coniagas No.4 Shaft was completed over three phases; Phase 1 – Polyurethane foam, Phase 2 – High strength concrete plug, and Phase 3 – Final concrete plug.

Phase 1: Polyurethane Foam

The first phase of backfilling consisted of spraying an expanding polyurethane foam on the timber obstruction in the shaft, using the obstruction as a form. The objective of phase one was to build a foam plug in the shaft which would act as a form for the higher strength concrete plug. The foam plug was constructed using three sets of 55 gallon drums of Part A and Part B of 2 lb. medium density closed cell polyurethane foam, which is typically applied as residential insulation. The foam was sprayed from surface through the opening in the shaft using a nozzle that would ensure that the liquid stream could reach the timber obstruction and not hit the walls of the shaft. The foam was applied evenly across obstruction and around the edges of the shaft walls to minimize the risk of concrete leaking around the foam during the next phase. A CMS was completed after the foam had set, and indicated that a foam plug thickness of 7 m had been achieved. This was approximately twice as thick as originally anticipated, likely due to the expansion rate being under estimated during the planning stage. The CMS showed that the surface profile of the foam was not flat (i.e., it was mounded in the center of the shaft). Figure 8 shows the application of the spray foam and the subsequent CMS. The foam was allowed to set for 24 hours before commencing Phase 2.

Figure 8: Application of Polyurethane Foam Plug and Cavity Monitoring Survey Using the C-ALS System

Phase 2 – High Strength Concrete Plug

The second phase of backfilling consisted of pouring a high strength, 35 MPa, fibre reinforced concrete on top of the polyurethane foam. The objective of Phase 2 was to build a concrete layer which could support the load of the final concrete layer before it had cured and became self-supporting.

A total of 12.5 m^3 of 35 MPa concrete was pumped into the shaft void using a positive displacement pump with a slick line through a window in the building into the shaft, as shown on Figure 9. A CMS was completed after the Phase 2 concrete was pumped into the shaft and indicated that high strength plug was approximately 1 m higher than the surface of the foam plug after Phase 1. Based on the shaft area from the CMS and the volume of concrete delivered, it was estimated that the foam had compressed 0.5 m, resulting in a concrete plug that was 1.5 m thick, as planned. It was anticipated that compression of the foam plug would occur between 20% and 30% based on some rudimentary lab testing of foam cores completed at another project location leading up to this work. Observations of the concrete pour showed no evidence of the concrete leaking through the foam plug.

Figure 9: Concrete pump truck and slick line used to pump concrete into shaft void

A sample cylinder of the high strength plug was broken after 6 days curing time in order to confirm that the concrete had achieved the strength necessary to support the second concrete pour. The uniaxial compressive strength test indicated that the initial concrete pour had reached a strength of 22.7 MPa, which represented a FOS of 4.9 based on the Mitchell equations when considering the load of the second pour. Figure 10 shows the shaft after placement of the high strength concrete plug.

Figure 10: Specified 35 MPa fibre reinforced concrete for initial plug

Phase 3 – Final Concrete Plug

The objective of Phase 3 was to fill the remaining void volume and have the final plug be in contact with the historical cap. The minimum concrete strength planned for this phase in the design was 5 MPa; however, 25 MPa concrete was ordered from the supplier to maintain pumping ability through, also considering the differential cost was minimal. Approximately 104 m^3 of 25 MPa strength concrete was pumped into the shaft void. Pumping was ceased when the concrete level had risen into the opening in the historical cap.

Figure 11 shows the delivery of the final concrete plug and the location of the plug surface inside the historical cap.

Figure 11: Pumping specified 25 MPa concrete for final plug

Conclusion

Cemented backfill was placed inside the Coniagas No. 4 Shaft that meets the requirements for rehabilitation of mine openings to surface as outlined in the *Mining Act* O.Reg.240/00. The concrete for the top 7.1 m of the shaft (i.e., Phase 3 pour) had an average 28-day UCS strength of 32.0 MPa (Adbel-Aty, 2014), and the next 1.5 m (i.e., Phase 2 pour) had an average equivalent 28-day UCS strength of 28.4 MPa. The weighted average equivalent 28-day UCS of the concrete backfill placed in the shaft is 31.4 MPa, which represents a FOS of approximately 13 for the limiting failure mechanism using the Mitchell equations.

Figure 12 shows a longitudinal projection of the Coniagas No. 4 shaft with the rehabilitation phases superimposed on the initial void model from the investigation.

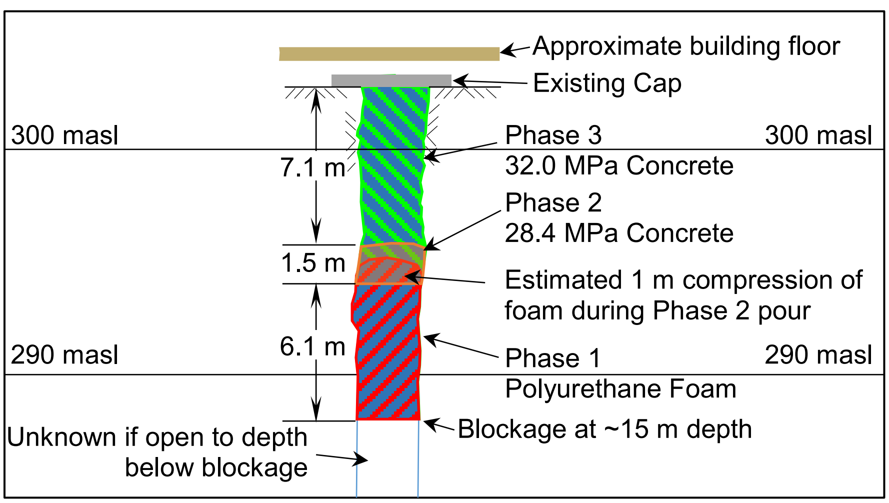


Figure 12: Longitudinal Drawing of Coniagas No. 4 Shaft

Experience on this rehabilitation has shown that cemented backfilling of mine openings to surface using a remotely constructed foam barricade is appropriate when infrastructure is present above the mine opening, or minimal disturbance of the surface/infrastructure is desired. The cost is comparable to constructing a monolithic cap above the mine opening, and concrete backfill should be considered as an alternative where possible. Application of the initial foam barricade is the greatest challenge, and the Coniagas No. 4 Shaft had favorable conditions of a vertical void with an obstruction at a reasonably shallow depth. Alternative methods of constructing a temporary barricade to build the foam plug, or incrementally building the foam plug from off of the rock walls, may be necessary depending on site and void geometry.

Acknowledgements

The authors would like to thank the owner and tenants of 50 Silver St., Cobalt for allowing access to the site to allow for this work to be completed. The authors would also like to thank the Ontario Ministry of Northern Development and Mines for working with us to allow for this innovative approach to be designed and implemented.

Literature Cited

Abdel-Aty, M., 2014. Compressive strength prediction of Portland cement concrete with age using a new model. HBRC Journal, Vol. 10, pp 145-155.

Brown, R. 1999. Ghost Towns of Ontario - Volume 2: Northern Ontario and Cottage Country. Polar Bear Press.

Mitchell R.J., Roettger J.J. 1989. Analysis and modelling of sill pillars. Innovations in mining backfill technology. Balkema, Rotterdam, pp. 53-62.

Mitchell R.J. 1989a. Stability of cemented tailings backfill. Computer and physical modelling in geotechnical engineering. Balkema, Rotterdam, pp. 501-507.

Mitchell R.J. 1989b. Model studies on the stability of confined fills. Canadian Geotechnical Journal, Vol. 26, pp. 210- 216.

Mitchell R.J. 1991. Sill mat evaluation using centrifuge models. Mining Science and Technology Journal, Vol.13, pp. 301-313

Sabina, A.P. 1974. Rocks and minerals for the collector. Cobalt-Belleterre-Timmins; Ontario and Quebec. Canadian Geological Survey, Paper 73-13.

The Corporation of the Town of Cobalt, C.T.C. c2016. Cobalt.ca. Retrieved 9 May, 2016, from http://cobalt.ca/visitors/historic-buildings/

INNOVATIVE SOLUTIONS FOR PHYSICAL MINE HAZARD REHABILITATION

Jonathon J. L. Taylor[1]

Paul G. Palmer[1]

Sue Longo[2]

[1]Golder Associates Ltd., Sudbury, ON, P3C 4Y1

[2]Golder Associates Ltd., Calgary, AB, T2A 7W5

Key Words: mine closure, rehabilitation, crown pillars, backfilling

Closing of mines can be complex, expensive and occur over a long duration often due to location (remote or adjacent a community), era of mining (type of mining and when mining occurred) and availability of reliable information. Successful rehabilitation of mine operations require multidiscipline teams and the integration of these various expertise can result in innovative solutions for physical mine hazard rehabilitation.

Mine sites can consist of several types of physical hazards, such as mine openings to surface (shafts, raises, adits, etc.), near surface mine workings (i.e., crown pillars), waste rock piles, and tailings storage facilities. These physical hazards are often problematic as they can present risks to the general public (i.e., injury, fatality, damage to property) if left un-rehabilitated, and generally speaking they get worse with time. These hazards might have been a low risk situation during operations or directly after closure, but can become a high risk if they are left to degrade over years and/or decades. For example, mine openings to surface can be deep open holes which, if left open and accessible to the public, can result in fatality if a person were to fall into the open feature. In Ontario, preventing inadvertent access and protecting openings are the primary focus of rehabilitation solutions for physical mine hazards that have a connection to surface.

The rehabilitation of physical hazards requires the integration of various disciplines of science and engineering. For example, mine openings to surface and crown pillars are often investigated and assessed by a rock mechanics or mining engineer and the rehabilitation is designed in conjunction with a structural engineer, whereas waste rock pile and tailings facilities are often investigated and the rehabilitation is designed with input from geochemists, hydrologists, and geotechnical and environmental engineers and then typically constructed by third party contractors. In some cases, the stability of the crown pillars on site are not well understood until later in the project while rehabilitation of mine openings to surface (raises and shafts) have progressed. This can lead to a situation where a raise that is connected to a near surface stope is capped before the stability assessment of the crown pillar is completed, and the crown pillar stability assessment may show that the crown pillar is not long term stable and requires rehabilitation. In this example the rehabilitation of the crown pillar could have also included the raise with no additional cost negating the need of building a concrete reinforced cap over the raise.

Recent rehabilitation project experience has shown that developing an overall property rehabilitation strategy that considers all of the physical mine hazards on a site or in an area can have cost saving benefits and technical efficiencies. Optimizing the rehabilitation strategy for all hazards with all disciplines means avoiding duplication of cost and effort as described in the example above. In addition, by assessing and applying innovative rehabilitation methods, such as the use of pre-cast concrete in remote areas or with challenging hazards or the use of newer products, such as foam, the rehabilitation strategy becomes a targeted specific plan for that site and not a one-size-fits-all solution, which can lead to higher costs.

This paper will discuss the process of developing a rehabilitation strategy and options focused on near surface crown pillars and mine openings to surface.

Phases of Closure of Mine Openings

Closure of physical mine hazards generally progresses in five phases. This allows for use of existing data to complete preliminary assessments and plan the investigations, which are then used to develop the rehabilitation plan. The five phases are:

- Phase 1 – collection and review of historical data
- Phase 2 – preliminary desktop studies
- Phase 3 – physical investigation, assessments, and rehabilitation planning
- Phase 4 – implementation of the rehabilitation solution
- Phase 5 – monitoring

Phase 1 – Collection and Review of Historical Data

For older abandoned mines, historical data may be lost, or the amount of information may be insufficient to complete the preliminary desktop studies. Often, the only information available is a report that includes a plan drawing and a simple composite longitudinal section showing many stope areas. These can be difficult to assess in areas where there are several vein systems to the ore body, which would result in several stopes showing in the same area on the longitudinal section, and can be even more complex when the strike and dip of these veins vary from location to location. For this reason, it is important to use as many surface features available on the plan drawings to geo-reference the plan and longitudinal drawings. Establishing surveying control with these features is also important at the onset of the project so that there is more confidence in the crown pillar locations when developing a drilling plan in the later stages of the project.

Identification and collection of geological and geotechnical data is also key as this data has implications in the preliminary desktop study stage. Geological information can come from old exploration drillhole logs, geological plan/section maps, or other regional maps from provincial sources (i.e., Ontario Geological Survey in Ontario). Historical geotechnical information is the most challenging piece of information to collect. Often, geotechnical data was not well documented, or documented in such a way that it is not useful for completing the preliminary desktop study. In many cases, a conservative assumption of geotechnical properties (i.e., rock mass quality) are made during this stage based on some visual observations or assuming a poorer rock mass property than might actually be there. In addition, in some areas recent experience has shown that discussions with the local population may provide some useful direction during investigation. Inevitably someone had a mother/father/grandfather/uncle who worked at the mine and can provide additional information. Sometimes they have old maps and level plans, and sometimes they worked at the mine and can describe the underground situation especially if access is no longer available.

Information about existing rehabilitations (i.e., previously constructed concrete caps, backfilling of openings, etc.) is also useful information to identify at an early stage as these can potentially be used as mitigating measures, or can be integrated as part of the rehabilitation design during the rehabilitation planning stage.

Phase 2 – Preliminary Desktop Studies

Once the process of collecting and reviewing historical data is completed, a desktop assessment of physical mine hazards is completed. This process involves completing preliminary stability assessments of the crown pillars, as well as prioritization and preparation for the physical investigation stage of the project.

The industry standard practice for assessing the stability of crown pillars is to use the Scaled Span method, which was developed by Golder in 1990 under contract with CANMET (Golder, 1990). Golder has subsequently updated and maintained a crown pillar database since that time, which now consists of over 500 crown pillars at 110 mines, and includes 85 failure cases. Updates have also been completed to include estimates of the probability of failure (POF) of the crown pillar, which was used to develop an acceptable crown pillar risk exposure guideline (Carter, 1992; Carter & Miller, 1995; Carter et al., 2008). The method takes into consideration the geometry and properties of the crown pillar (span, strike, thickness, dip, and density) to calculate a 'scaled span' number, which is then graphed against the rock mass quality, Q (Barton, 1974), as shown on Figure 1.

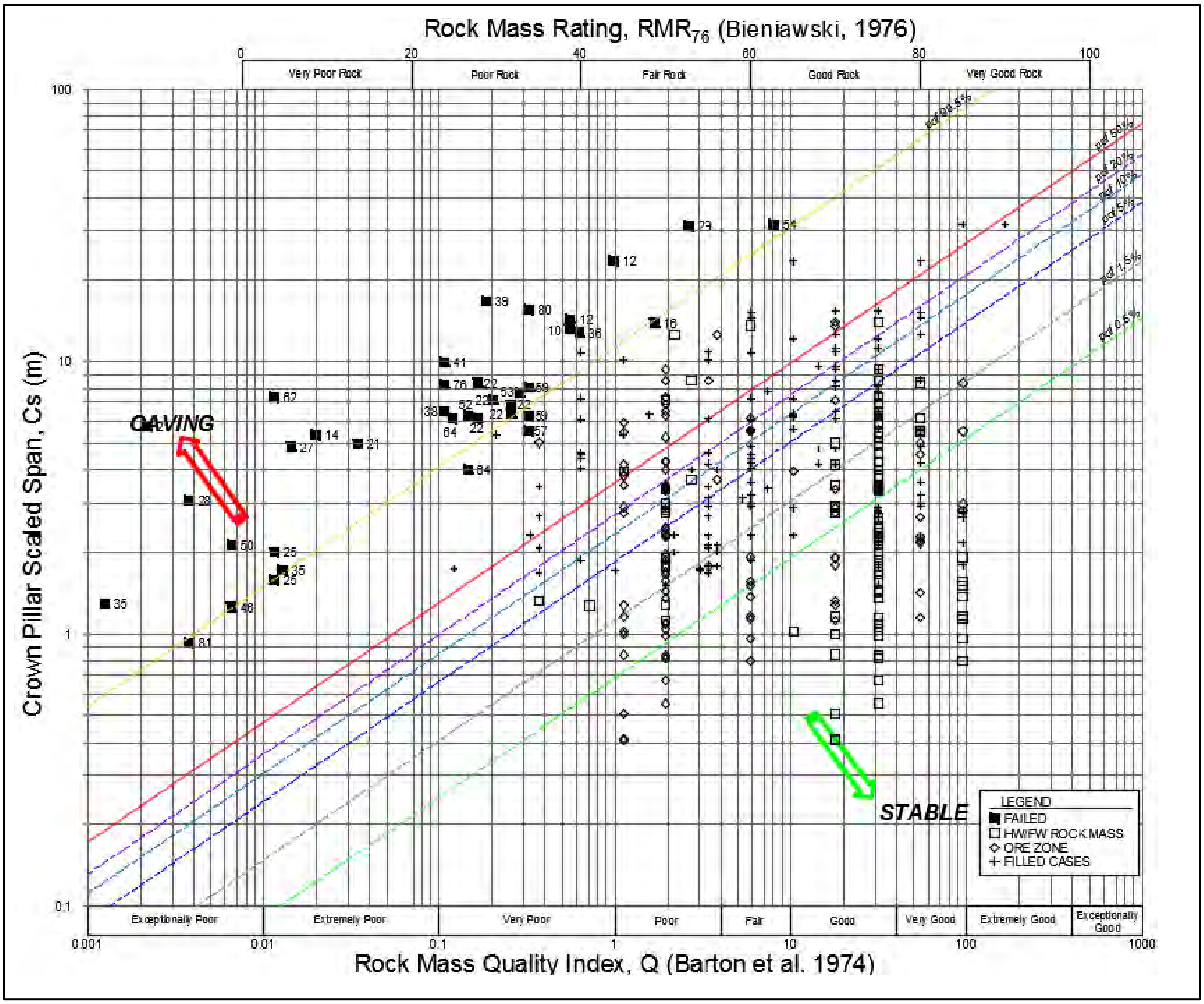


Figure 1: Scaled Span Graph (Golder 1990, Carter et al., 2008)

The Scaled Span method can also be used to calculate the POF percentage, which was developed based on logistic regression of the empirical database (Carter et al., 2008). The POF for several crown pillars can be summarized into ranged bins to estimate the duration of stability of the crown pillar, ranging from effectively zero to very long term stability, as shown in Table 1.

Table 1: Simplified Crown Pillar Risk Exposure Guidelines for Mine Closure Purposes (after Carter et al., 2008)

Class	FOS (%)	Minimum FOS	Serviceable Life	Public Access	Potential Monitoring Requirements
A	50 - 100	<1	Effectively zero	Forbidden	Ineffective
B	20 - 50	1.0	Very, very short-term: temporary mining purposes only	Forcibly prevented	Continuous/regular sophisticated monitoring
C	10 - 20	1.2	Very short-term:	Actively	Continuous/regular

Class	FOS (%)	Minimum FOS	Serviceable Life	Public Access	Potential Monitoring Requirements
			quasi-temporary stope crowns	prevented	monitoring with instruments
D	5 - 10	1.5	Short-term: semi-temporary crowns, e.g. under non-sensitive mine infrastructure	Prevented	Continuous/regular simple monitoring
E	1.5 - 5	1.8	Medium-term: semi-permanent crowns, possibly under structures	Discouraged	Conscious superficial (i.e., visual) monitoring
F	0.5 - 1.5	2	Long-term: quasi-permanent crowns	Allowed	None to Incidental superficial (i.e., visual) monitoring
G	< 0.5	>> 2.0	Very long-term: permanent crowns	Free	No monitoring required

For closure purposes in Ontario, it has generally been accepted that crown pillars with a POF of less than 1.5% (i.e., Class F and G in Table 1) are considered to be long term stable and do not require rehabilitation, while crown pillars with a POF of 1.5% or greater require further investigation that may result in rehabilitation. The POF from the desktop assessment is used to prioritize the investigation and rehabilitation during the subsequent phases of work.

Phase 3 – Physical Investigation, Assessments, and Rehabilitation Planning

After the desktop assessment is completed and is defined as not long term stable, a physical investigation is required to update the stability assessment, and to start planning the rehabilitation strategy. This stage is where the highest degree of engineering input is required, as it involves completing physical investigations, updating stability assessments, completing rehabilitation trade-off studies to determine the most cost effective rehabilitation, and planning the rehabilitation work.

In most cases, crown pillars with a high risk of long term instability (i.e., POF > 20%) are unlikely to be deemed long term stable after a physical investigation unless the estimated parameters used in the desktop assessment were overly conservative. Investigation work for these crown pillars should focus on confirming the geometry, extent, and connectivity of the void space to other workings in preparation for closure rehabilitation planning. Investigation of these crown pillars should proceed cautiously as there may be a high degree of unknowns (i.e., geometry and extent of the void space, backfill conditions, etc.), and there may be a risk of localized instabilities during the investigation.

For crown pillars with a medium risk of long term instability (i.e., POF 1.5 to 20%) have the potential to be reassessed as long term stable after a physical investigation. Often times the rock mass quality is conservatively underestimated, and data collected from drill core can show that the rock mass quality is higher than originally estimated during the desktop assessment. The investigation for these crown pillars should focus on confirming the geometry (span and thickness) of the upper portion of the stope, as well as collecting geotechnical information in the crown pillar rock mass.

Investigation techniques for crown pillars consists of safety/access drilling for high risk crown pillars, geotechnical diamond core drilling, surveying of void space (using in-hole laser scanners or sonar survey tools), and surveying of surface topography. In some instances, digging may also be useful to find reference points (i.e., surface mine openings) to increase the confidence of the georeferenced mine plans. In some cases surface geophysical methods (ground penetrating radar, seismic and electric-resistivity imaging) can be utilized where void sizes are large relative to crown pillar thickness and there is sufficient surface space to complete surveys.

Safety/access drilling consists of drilling a series of holes from a safe location towards the planned geotechnical drilling area. This type of drilling can effectively be completed by using an air-percussive drilling rig (also known as a hydraulic or air-track drill). The holes are drilled at an angle to a specified depth to confirm the thickness of competent rock ahead of the drill. Once safe access is established to the first geotechnical drill area, additional air-percussive drilling can be completed to first locate the void space, or diamond drilling can commence. Safe access drilling can be eliminated for areas once the void space has been found and surveyed, as the survey can be used to adjust the diamond drilling plan to ensure that the drill is set up in a safe location.

Surveying of the void space should be completed on every hole that intersects void space, and then used to modify the investigation plan accordingly. The survey of the void space can also be used to update the geometry of the crown pillar and stability assessment.

Many abandoned sites have previously rehabilitated mine openings to surface, traditionally achieved through concrete capping. Previous concrete capping standards often do not meet current regulatory requirement, and in most cases, these older concrete caps will need to be replaced. Drilling should also be completed through these caps or through bedrock into the void space so that a survey can be completed of the void space. This information is useful in completing the rehabilitation design and planning.

Investigations can often consist of several phases occurring over several years, and can also overlap the rehabilitation phase. For example, it has been common to quickly investigate surface mine openings and start rehabilitation on these features before the investigation of the crown pillars has been completed. In some instances, these surface mine openings are connected to voids below crown pillars. For this reason, it is important to consider a site wide, or area specific, rehabilitation strategy and plan. At several sites in Ontario, capping of the surface mine openings was completed before the connected void space/crown pillars were fully investigated, and then these crown pillars were found to be not long term stable. Recent experience has shown that it is almost always more cost effective to fill void spaces for crown pillar rehabilitation than to construct massive concrete caps over them. In this instance, the incremental cost of backfilling a raise connected to the stope void is often much smaller than to construct a concrete cap over the raise as well.

Many of these mine sites also have waste rock and tailings storage facilities on site, and the closure project can involve rehabilitation of these hazards. In most cases, this is material that is useable for making cemented or paste backfill products for filling underground voids. Using this free source of material is often discounted as the project cost is often judged on a per feature cost basis rather than a site wide basis. For example, digging and transporting tailings material appears expensive in order to rehabilitate the tailings storage facilities, and often times the underground voids that require rehabilitation are not large enough to handle all of the tailings, but when considering that this is material that would otherwise need to be purchased for filling the underground void space, it can quickly become a cost effective solution.

Once the investigation and site characterization work is completed, a trade-off study to determine the best site wide rehabilitation strategy is completed. This includes consideration of different rehabilitation strategies, such as more traditional methods as capping or backfilling described above, or more innovative solutions described later on in this paper.

The schedule of the rehabilitation can be a function of risk mitigation and cost, or may also include complexity of the construction in consideration of other activities on site. For example, a mine owner may only have enough budget in one year to mitigate part or all of the highest risk hazards in one year, leaving other hazards until subsequent years to complete the rehabilitation. In some cases, different regulatory bodies may also have jurisdiction, and additional rules for rehabilitation may apply. One example of this is a uranium mine, where the provincial department of mines regulates rehabilitation standards with respect to the physical hazards on site and a federal government agency, in this case the Canadian Nuclear Safety Commission (CNSC), also regulates the site, more specifically with respect to radon emissions. In this case, elimination of radon emissions can hold precedence for rehabilitation over the physical mine hazards from a regulatory perspective and this will drive the rehabilitation schedule.

Phase 4 – Implementation of the Rehabilitation Solution

Once the trade-off study and rehabilitation planning is completed, the construction can begin. During this stage, most of the engineering work has been already completed, however, there is still a requirement for quality assurance and quality control. This is one of the most important parts of the construction phase as it ensures that the rehabilitation is constructed to the design specifications, and should any change in conditions be noted, they are well documented and become part of the as-built records.

The goal of the investigations and preliminary work is to eliminate uncertainty, but these are dynamic situations and all phases of the rehabilitation strategy should be viewed as an iterative process.

Input can also be sought from the design team should there be a major change in conditions so that designs can be modified if necessary. It is also important to document the as-built information so that certification of the rehabilitation can be completed and any financial assurance can be released or obligation to the regulatory body can be fulfilled.

<u>Phase 5 – Monitoring</u>

Monitoring can involve either monitoring as the rehabilitation solution (i.e., stability monitoring through instrumentation) or monitoring of the rehabilitation to ensure that it remains stable and performs as expected.

Example of monitoring as the rehabilitation solution are installation of multi point borehole extensometers or other monitoring instruments such as tilt metres and time-domain reflectometry cables, radar surveying, LiDAR surveying, and satellite surveying. These methods can be effective for monitoring crown pillars with a low to moderate risk of crown pillar instability (i.e., < 20% POF), but are ineffective for high risk crown pillars.

Monitoring of rehabilitations mostly involves visual inspections on an annual, or multi-annual basis (i.e., every 2 to 5 years). In some cases, rehabilitations can also be monitored using instrumentation. An example of this is a tilt-meter that monitors for movement of an unconsolidated sand fill.

Innovative Rehabilitation Solutions

Recently, advances have been made in technologies and the application of new techniques for rehabilitation of mine workings. These options include the use of sprayed polyurethane foam, precast concrete, stainless steel caps, and paste backfilling with a mobile plant.

Sprayed polyurethane foam has not been accepted in Ontario for use as the only means of rehabilitation as there are concerns about the longevity and strength of the material, however, it has been used recently in three applications in Ontario for mine closure projects as a component of a larger rehabilitation plan. One of the scenarios was the closure of a vertical shaft inside of a building, which involved spraying the foam over a timber blockage in the shaft to a specified minimum thickness, and then pouring concrete in two lifts to backfill the remainder of the shaft. The foam was only used as formwork inside the shaft, and was not used for long term support of the cemented backfill material poured on top of it. The other two scenarios involved spraying the foam remotely through a borehole into an underground stope. In both cases, the intent was to use the foam to block a raise at the bottom of the stope that was connected to other large void spaces that did not require rehabilitation, and then pour paste backfill to tightly fill the void space above the foam, resulting in a long term rehabilitation of the crown pillar.

Precast concrete is a potential solution for capping of surface mine openings for areas with difficult access or site restrictions. An example of this would be for a remote fly-in community where there may not be a local concrete contractor, or the site is such that the mine hazard is not easily reachable by contractor equipment. Another use of precast concrete is for installation of caps during the winter months. In some cases, this may be the only time to complete the work due to schedule restrictions. In this scenario, cast in place concrete would require heating and hoarding, and additional curing techniques, which can significantly increase the cost of the cap. Precast sections can be manufactured to any size or load specification and constructed in ideal curing conditions, and then transported to site and installed. Stainless steel bolts are used to permanently pin the cap to bedrock or a concrete collar. Precast concrete could also be used in a scenario of temporary mine closure, or where protection of the mine opening is required during filling stages and they can be constructed in such a way that they are removable at a later date.

In other provincial jurisdictions, such as Saskatchewan, stainless steel caps have been constructed in remote communities to cover several surface mine openings. These caps have the potential to be a cost effective long term solution in remote, sparsely populated areas where the likelihood of access by the public is low, however, are likely to be cost prohibitive or impractical in areas near populated areas where the likelihood of access by the public is high. It is noted that permanent stainless steel caps have not been constructed in Ontario to date, and there are no current regulatory standards for stainless steel capping in Ontario for final closure.

Paste backfilling is a common practice in operating mines and has been used for over 20 years especially as it is an efficient use of tailings material, but this involves building large paste batch plants that are fed by the output from the mill. Recently, methods have been developed to use mobile plants for abandoned sites. The process involves digging old tailings or other local feed material (i.e., overburden), adding a binder, and in some cases adding other aggregates, using a continuous mobile mixing plant, and pumping the paste into underground stopes through boreholes drilled from surface. The advantage of mobile mixing plant is that it can achieve the same throughput as a batch plant, on the order of approximately 500 m^3/day, without having to build or mobilize large batch plant equipment. Mobile plants can also be moved from location to location, avoiding the requirement of pumping and having a fleet of redi-mix trucks to deliver the paste to the delivery points.

The innovative techniques presented above can be used in conjunction with each other or with more traditional methods of rehabilitation. For example, traditional methods for backfilling of abandoned mines would involve filling part of the stope with non-cemented fill in order to block the connection point of the stope to other workings, filling to a specified height in order to reduce the cost of cemented fill, and then topping up the remainder of void space with cemented fill, such as concrete. This same approach could be used but instead of using concrete, paste could be used to ‘top up’ fill the stope.

In older mines where narrow vein mining techniques were employed, it was common practice to develop a raise through the planned stope area between the first level of mining and surface, and then mine the stope through this raise. In some cases, there is an upper stope and lower stope connected by this raise. Traditional methods for rehabilitating this situation, where the lower stope crown pillar is long term stable but the upper stope crown pillar is not, would be to fill the lower stope and raise with sand or gravel, and then filling the upper stope with cemented fill. Polyurethane foam can be used in this scenario to block the raise in the area of the upper stope, and then filling the upper stope progressively with paste fill. This can result in a cost saving as the lower stope could be a very large volume and could require purchasing a substantial amount of material if locally sourced/free material is not available.

Summary

Using a phased and planned approach to mine hazard rehabilitation allows for the ability for mine site owners to investigate, assess, prioritize, and implement rehabilitation measures in a cost effective and efficient manner. This involves consideration of rehabilitation options for the site as a whole. It has been the authors’ experience that developing a rehabilitation strategy for the entire site that is a targeted specific plan for that site and not a one-size-fits-all solution is the most cost effective method of closing out a mine site. No one solution will fit for the whole site, rather, it is a collection of applications that requires a strategy to implement efficiently.

In most cases, using the “do nothing” approach, or delaying the rehabilitation by a long period of time, is often not acceptable from a regulatory stand point, and can often lead to higher costs over the duration of the project as work tends to happen independently of each other in this scenario. This is another reason why a strategy for rehabilitation needs to be completed for the site as a whole.

Innovation is also always changing and improving the way that rehabilitation is achieved. Consultants, mine site owners, and regulatory bodies need to work together and need to be constantly evolving in their ways of thinking to allow for these improvements and advances.

Acknowledgements
The authors would like to thank their clients and regulatory bodies for their willingness to embrace the innovative approaches presented in this paper. Improvements and new rehabilitation methods require a level of collaboration between consultants, mine owners, and regulatory bodies to research and trial applications. Without the willingness to try new methods in hopes to complete the rehabilitation in a cost effective in and efficient manner, progress could not be made in this field.

Literature Cited

Barton, N., Lien, R., and Lunde, J. 1974. Engineering Design of Tunnel Support. Rock Mech., Vol. 6 (No. 4).

Carter, T. G. 1992. A New Approach to Surface Crown Pillar Design. Proc. 16th Can. Rock Mechanics Symposium, pp. 75-83.

Carter, T. G., Cottrell, B. E., Carvalho, J. L., and Steed, C. M. 2008. Logistic Regression Improvements to the Scaled Span Method for Dimensioning Surface Crown Pillars over Civil or Mining Openings. Proc. 42nd US Rock Mechanics Symposium, ARMA 08-282.

Carter, T. G. and Miller, R. I. 1995. Crown Pillar Risk Assessment - Cost Effective Measures for Mine Closure Remediation Planning. Trans. Inst. Min. Metl, Vol 104, pp.A41-A57.

Golder Associates Ltd. 1990. Crown Pillar Stability Back-Analysis. Canada Centre for Mineral and Energy Technology, Project# 881-1739.

COST EFFECTIVE PLANS FOR SUCCESSFUL MINE CLOSURE – RECENT CASE STUDIES

Marc S. Theisen

Profile Products LLC, Signal Mountain, TN USA 37377

Jin Xu

Maccaferri China, Changsha, Hunan, China 410001

Abstract

Successful rehabilitation, reclamation or closure of massive soil and vegetation disturbances from mining requires a comprehensive and holistic approach. Those overseeing rehabilitation efforts should assimilate and stage several considerations into a working relationship that integrates five fundamentals for successful mine closure. Employing the discipline to work through the discovery sequence of the first three fundamentals – to analyse soils and substrates, pick the right plant materials for the site and select the most cost effective erosion and sediment control techniques, will undoubtedly lead a project in the right direction.

These fundamentals must be followed by the development of clear and comprehensive construction plans and specifications to effectively communicate the project requirements to contractors and installers. Once construction commences, onsite oversight of acceptable installations must be conducted by qualified inspectors knowledgible of the site conditions. Then, active rehabilitation sites must be regularly inspected and maintained after each significant precipitation or other potentially damaging event. Inspections should be conducted by qualified professionals whose expectations are consistent with the installer as well as the owner and regulatory entity(s). Failure to systematically execute on any of these fundamentals can undermine the best laid plans of any mine closure project.

Mined land sites offer unique and unpredictable challenges for successful closure efforts. Published handbooks or manuals can provide general approaches to mined land reclamation, but rarely can they address the specific needs or conditions of unique mining sites. Successful restoration most typically comes from carefully controlled onsite trials and iterative installations to assess efficacy of various treatment combinations. Such treatments must then be refined and customized to develop cost effective closure plans. Exhaustive research on suitable soil amendments, plant materials and erosion control techniques should be planned and budgeted for – as integral steps in the mine closure progression.

Four selected case studies from North America, Asia and Oceania – demonstrating diverse climates with contrasting environmental and site conditions will be offered to illustrate the discovery (required information gathering) and implementation (execution) of the five fundamentals for successful mine closure.

Key Words: Mine closure, erosion control, revegetation, reclamation, rehabilitation

Introduction

Successful mined closure entails an inclusive approach to assess, address, manage and integrate treatments or techniques to overcome the considerable challenges presented in post-mining environments such as poor substrates, large unprotected areas with high erosion potential, difficult access, adverse climatic or seasonal weather conditions and much more. Reclamation or rehabilitation managers must then balance these challenges with other operational concerns such as budgetary constraints, cost of materials, availability of labour, sequencing of earthmoving activities and required timing of completion for mine closure related activities.

Those overseeing rehabilitation efforts must assimilate and stage several considerations into a working relationship that integrates five fundamentals for successful mine closure, supported by proper planning and execution. “Soil poor” sites associated with mining activities offer considerable challenges particularly when topsoil and cover soil sources are scarce or of limited agronomic benefit. Such sites will require innovative thinking and implementation to overcome the absence of favourable growing conditions.

Employing the discipline to work through the discovery sequence of the first three fundamentals – to analyse soils and substrates, pick the right plant materials for the site and select the most cost effective erosion and sediment control techniques, can head a mine closure plan in the right direction.

Following the discovery sequence is development of clear and comprehensive construction plans and specifications to effectively communicate the project requirements to contractors and installers. Once construction commences, capable onsite oversight of acceptable installations must be conducted by qualified inspectors. Then, the active rehabilitation sites must be regularly monitored and maintained after each significant precipitation or other potentially damaging event. Inspections should be conducted by qualified professionals whose expectations are consistent with the installer as well as the owner and regulatory entity(s). Failure to systematically execute on any of these fundamentals can undermine the best laid plans of any mine closure project.

These five fundamentals are by no means novel and have been previously introduced to the mining industry (Theisen 2015). This publication offers a brief overall of five fundamental principles to facilitate successful mined land closure, followed by illustrative case studies where they have been successfully employed in a variety of geographic, climatic and physiographic conditions.

An Overview of the Five Fundamentals

Fundamental #1 – Understand Your Soils or Substrates

The first fundamental is employing creative methodologies to develop suitable growing media – typically from less than desirable soils or substrates. This can only be accomplished by first understanding the make-up of the soil or substrate through comprehensive soil testing for agronomic potential and limitations. Soil testing, interpretation of the test results, and incorporating prescriptive remedies to improve soils should be an essential part of any mine closure plan. Without a proper understanding of soils or substrates considered for use as growing media to establish vegetation, it is difficult to predict or achieve potential project success.

Prior to conducting and interpreting soil tests, it is important to understand testing procedures and analytical methodology that are relevant for mine closure and/or vegetation establishment projects. There are various ways to extract measurable soil characteristics and analyse samples, but rarely do different soil testing methods produce identical results. It is important to properly collect and label soil samples prior to sending them to a reputable laboratory. Two referenced publications offer collection and sampling instructions as well as relevant testing protocol for erosion control projects requiring vegetative establishment (Soil Testing and Interpretation, 2015 and Theisen, 2015).

Fundamental #2 – Proper Species Selection

Equally important to addressing soils is the selection of plant species that exhibit sustainable growth and resulting erosion and sediment control (E&SC) performance. The second fundamental requires an assessment of suitable plant species for achieving both requirements – while meeting the collective post-reclamation needs of regulatory agencies and mine owners. Soil properties, climate, moisture regimes, slope aspect, maintenance, desire for native plant stock or ecotypic progeny, future land use and a host of other considerations contribute to proper species selection.

Perhaps the best resource for obtaining information and availability of suitable plant species are regional growers, collectors and suppliers of locally adapted seeds and plant materials. Experienced botanic professionals are well versed in seasonal pricing, quantities and availability of native or introduced seed sources as well as containerised or bare root shrubs and tree species. Certainly universities, researchers, consultants and agencies can also provide wisdom and guidance.

Fundamental #3 – Select the Most Cost-Effective Erosion Control Techniques

Once soil, agronomic and species selection considerations have been addressed, it is appropriate to begin analysing site conditions or characteristics to assess and select necessary erosion and sediment control measures. Site conditions, such as soils, climate, seasonality, slope lengths, gradients and aspects, ditch and channel flow hydraulics, pond and stream banks, wetlands and more, must be examined and proper controls selected.

A relevant and widely accepted methodology for assessing erosion protection on slopes is the Revised Universal Soil Loss Equation (RUSLE/RUSLE 2) for predicting annual soil loss (Renard et. al., 1997). RUSLE combined with erosion control effectiveness, an international rainfall database, growth establishment ratings, documented functional longevities and factors of safety all help to facilitate product selection for slope protection.

Methodologies from the US Federal Highway Administration's Hydraulic Engineering Circular Number 15 (HEC 15) – Design of Roadside Channels with Flexible Linings (Kilgore and Cotton, 2005) can be used for both unvegetated and vegetated channel designs and selection of techniques. Primary design formulas are Manning's Equation to determine maximum permissible velocity while maximum permissible tractive force is determined using the Shear Stress or Tractive Force Equation (Theisen, 2015).

Erosion control effectiveness, growth establishment (ability to facilitate vegetative growth) and functional longevity (persistence of the technique) are the three pillars of product performance and fundamental to selection criteria for erosion control techniques. Assessing these three key performance attributes in addition to relevant physical or index properties can assist designers in their risk versus reward selection scenarios – balancing costs and ease of implementation versus requisite factors of safety.

Fundamental #4 – Oversee and Insure Proper Installation

Suitable installation practices are critical to the success of any rehabilitation program. Comprehensive and detailed construction specifications with clearly delineated plans and drawings as well as complete mixing/application guidelines and details must be developed and combined with onsite supervision to assure proper installation.

Installation guidelines are readily available from manufacturers, consultants and trade associations for erosion and sediment control techniques, including Hydraulically-applied Erosion Control Products, Rolled Erosion Control Products, and Sediment Retention Fibre Rolls (ECTC, 2014).

All installations should be overseen by qualified and experienced professionals who are intimately immersed in the mine closure requirements. Experience, preferably site specific experience, is always a desired prerequisite!

Fundamental #5 – Coordinate and Conduct Timely Inspection and Maintenance Activities

Once erosion and sediment control measures have been installed, it is important to visually inspect and maintain them on a regular basis. Inspections should be conducted by qualified professionals whose expectations are consistent with the installer as well as owner and regulatory entity(s). Initial inspections should insure that all installations are in accordance with plans and specifications with all material quantities and activities fully documented. Subsequent inspections should be executed at pre-determined time intervals and maintenance activities conducted after each significant precipitation or other potentially damaging weather event.

Obvious examples requiring maintenance would be damaged silt fences or sediment control devices, rills appearing on treated slopes, displacement or movement of rock check dams or slope interruption devices, and excessive sediment being deposited at toes of slopes or near/into receiving water bodies. Timely inspections and maintenance can prevent small problems from turning into major complications.

Regrettably the final inspection and maintenance fundamental is perhaps the most underappreciated and overlooked practice with the “one and done” mentality that often is associated with construction bidding and contracting. Savvy mine reclamation managers should consider incorporating performance and/or maintenance requirements into their construction contracts or internal requirements. Mines with onsite resources should always make inspection and maintenance a standard operating procedure.

Subsequent to the initial “grow in” period, inspections should be conducted to assess agronomic aspects of rehabilitation efforts. Beyond monitoring vegetation vigour, cover and species composition; soils or substrates should be tested to determine if supplemental applications of seed, fertiliser, biological inoculants, other soil amendments or additional erosion covers are warranted. Soil tests may also identify developing problems with soil pH, excessive salts or upward migration of heavy metals or contaminants from underlying substrates. Maintenance related activities should be accounted and budgeted in comprehensive mine closure plans.

Selected mine closure case studies

The following case studies have been selected as examples where the five fundamentals were effectively employed and successfully executed on a variety of sites with contrasting geographic, environmental, climatic and physiographic conditions.

Case History #1 – Canadian Nickel Mine

This project is in the famous Sudbury nickel mining district in the Ontario Province of Canada, one of the world's largest fully-integrated mining, milling, smelting and refining operations. Over the years the operator had produced a mountain of slag, a by-product of the smelting process, which was threatening to overrun parts of the city. As a result, the company studied new ways to accelerate its ongoing reclamation of the slag piles and quickly convert them into green landscape.

An analysis of the slag showed it to be highly acidic and very low in organic matter and nutrients. Thus, it was deemed to be unlikely to sustain vegetation and 61,164 m^3 of a low permeability clay soil cover was specified for placement over the slag material to a 46 cm depth. In October 2006 through February 2007, the slag piles were reshaped to create a series of 3H:1V slopes – each 30 m long and divided from the next by horizontal benches 6 m wide. The clay cover was cat tracked (dozer walked) to create a roughened surface with mini-check dams to reduce sheet flow erosion potential, increase infiltration and provide pockets for water retention and enhanced growth. After placement of the clay soil cover in October 2007; lime, synthetic fertiliser, two biostimulants and a prescribed seed mix were hydraulically applied directly on the clay with a flexible growth medium tracer.

The following seed mix was applied at a rate of 252 kg/ha:

Grasses – *Lolium perenne*, *Poa compressa*, *Phleum pratense*, *Agrostis gigantea*, *Festuca ovina*, *Festuca rubra*, *Festuca elatior*, *Festuca rubra* subsp. *commutata*

Nitrogen-fixing legumes – *Trifolium hybridum*, *Trifolium pratense*, *Trifolium repens*, *Lolium corniculatus*

Due to an anticipated late fall seeding and the long 3H:1V slopes, a flexible growth medium was specified to be applied at the high rate of 5,100 kg/ha in a second application above the seed and soil amendments. The flexible growth medium was applied from two directions and then, fibre filtration tubes were installed at 11 m intervals to serve as slope interruption devices to slow sheet flow from storm events and snow melt. Installations were closely monitored by the owner and there were minimal maintenance activities necessary after the initial installations.

Despite the fall seeding, germination occurred quickly and vegetation was growing in the areas first treated prior to the onset of winter and a snow cover approaching 1 m in depth that persisted throughout the winter season. The following spring vegetation emergence continued and the site achieved exceptional cover with the combination of prescribed soil amendments, seed mix and flexible growth medium.

Figure 1 - Initial installation – Oct 2007

Figure 2 - Successful vegetation – June 2008

Case History #2 – Chinese Limestone Mine

Located in the Huzhu county of Qinghai province near the entrance of Beishan National Forest Park, exceptional rehabilitation of the Huzhu JinYuan was mandated by the Chinese government with a planned budget in excess of 50 M CNY. Excavated rock slopes devoid of organic matter and nutrients in excess of 100 m vertical heights at 65° - 70° slope gradients with exposed loose stone and gravel proved to be very challenging. Moreover, the site is at 3,280 meters in elevation, subject to rapid weather changes with sudden rain and snow, periods of drought and a short growing season running from April through September.

Key problems to be resolved were:

- How to cover and maintain a stable layer of soil on steep rocky slopes?
- How to prevent loose rock from falling down the slopes?
- How to prevent soil and seed to be washed away by rainfall and snowmelt?
- How to insulate and warm the soil while holding moisture?
- How to accelerate growth establishment in a short growing season?

The mine elected to utilize a "high-performance reinforced hydroseeding method" which entailed synergistic components including a hydraulically-applied engineered soil layer, combined turf reinforcement mat and rockfall netting, and a hydraulically-applied erosion control matrix as illustrated in Figure 3.

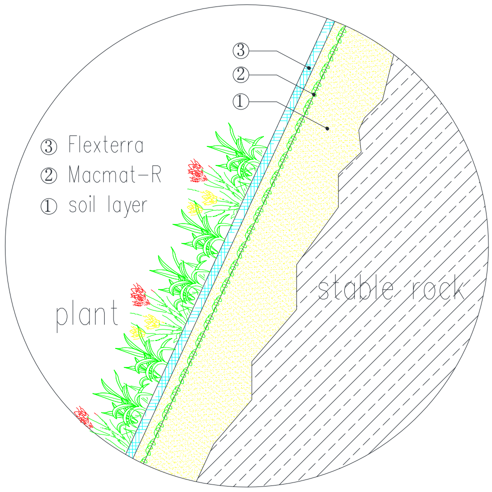


Figure 3 - Schematic of high-performance reinforced hydroseeding method

Soil layer

A combination of screened soil, fertiliser and a soil binding agent were mixed in water using a hydroseeder and the slurry was sprayed on to the rock surface. The binding agent helped to make the soil more stable with a more porous structure while sticking it more firmly to the rock surface.

Turf Reinforcement Mat (TRM) and Rockfall Netting Composite

A permanent TRM and rockfall netting composite was used to hold the soil slurry while also acting as the main structure to reinforce the soil and subsequent developing root mass. Moreover, the TRM/rockfall netting composite was also used to control soil erosion and provide loose rock protection. Due to long periods of limited precipitation, an irrigation system was also incorporated into the project design. For more rapid, consistent and safe deployment, the irrigation water lines and spray head components were pre-assembled within the TRM/rockfall netting at the manufacturing facility. Beyond the single-step installation advantages the irrigation system offered the ability to carefully meter water on an inaccessible, high and steep slope above an engineered soil composite lying on a nearly impervious rock substrate. The gravity fed system running downward from the top of the slopes also led to significant energy savings versus conventional irrigation systems.

High Performance Hydraulically-applied Erosion Control Products

Lastly, a 100% biodegradable, flexible growth medium consisting of wood fibres, biodegradable interlocking fibres, naturally derived biopolymers and wetting agents was hydraulically-infilled into the TRM/Rockfall composite. This material has been rated at 99% effectiveness for erosion control and demonstrates a water holding capacity exceeding 1,700% with a growth improvement factor of 800% over bare soil. The very high erosion control performance, combined with ability to foster vegetative establishment and a proven functional longevity of up to 18 months led to the selection of this component to complete the high-performance reinforced hydroseeding method.

Installation sequencing was as follows:

- Clean the rock surface

- Fix top anchors and steel wire rope
- Hang TRM/Rockfall netting composite in close contact with rock surface of slope
- Spray mixed soil slurry through the open and porous composite
- Spray high performance hydraulically-applied flexible growth medium into the TRM/Rockfall netting composite above the soil slurry
- Install irrigation system

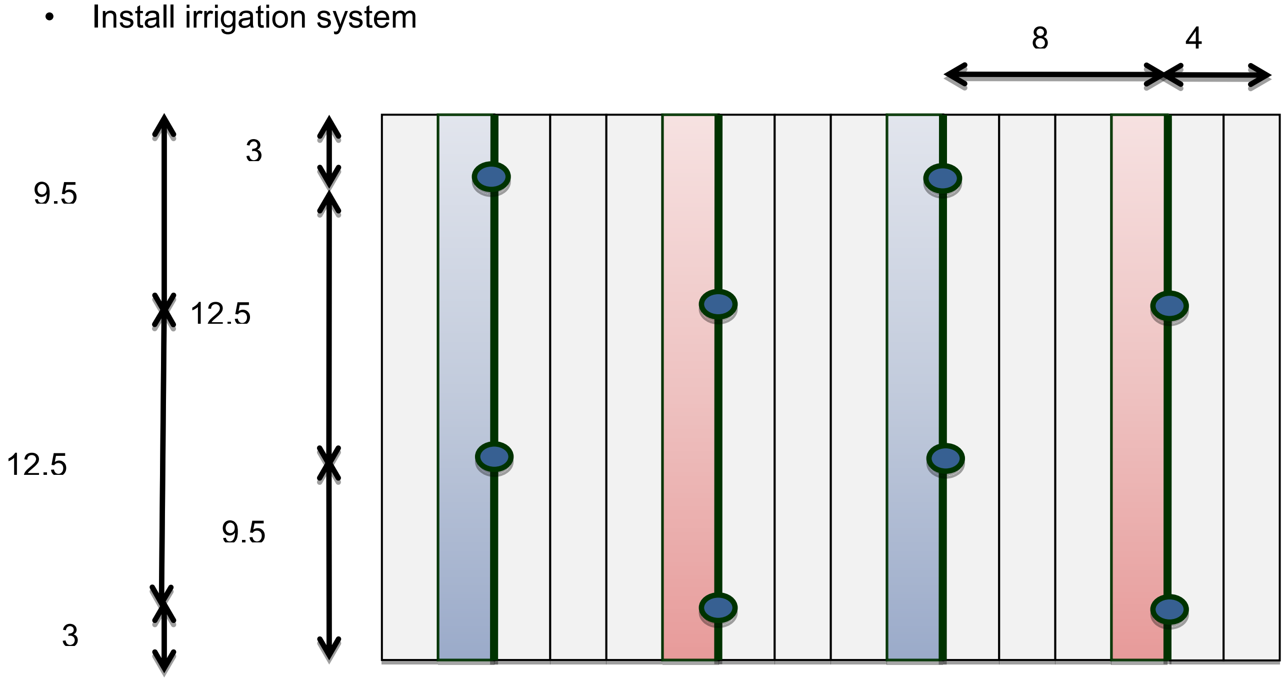


Figure 4 - Schematic of the irrigation system sprinkler head frequency in meters

Planting experiments were conducted with guidance from Qinghai Provincial University to determine locally available plant species adapted to the harsh site conditions. Selected species included *Elymus nutans, Festuca arundinacea, Calendula officinalis* and *Hypoxis sp.* at a rate of 35 kg/ha. Installation of the reinforced hydroseeding method occurred in July and August of 2014. The irrigation system was used when necessary and a reasonable cover of vegetation was established going into the winter season.

Figure 5 - Growth after 10 days

Figure 6 - Growth after 60 days

The winter season commenced and site then was covered with snow for a period of four months. When more moderate temperatures returned in the spring very minimal erosion or sloughing of the system was noted and vegetation establishment persisted through the

2015 growing season. The following photos document the vegetative establishment during the summer of 2015. The Chinese government was very pleased with the system employed and commended the project results. The mine is now employing this technique on other sites and have plans to "fertigate" existing slopes by introducing liquid fertilisers into the gravity feed irrigation system. Efforts in 2015 have continued and the seeding rate has been increased to 50 kg/ha with increased growth and ground cover noted.

Figure 7 - Prior to installation in 2014

Figure 8 - Growth in summer of 2015

Case History #3 – Indonesia Gold and Copper Mine

Situated in very mountainous terrain in the tropical Papua region of eastern Indonesia is a large open pit and tunnelling gold and copper operation reaching elevations of 4,500 m in both sub-alpine and alpine zones. The climate may be characterized as cool and extremely wet with annual precipitation approaching 7,000 mm.

Hydroseeding techniques have been used to revegetate the site's overburden stockpiles since the late 1990s. These techniques have been employed on areas that are not easily accessible, on steep slopes in areas that require quicker vegetation cover, and to introduce additional plant cover in previously revegetated areas. Hydroseeding has promoted the establishment of mosses and grasses such as *Deschampsia klossii* – both of which are locally abundant and indigenous to the area's sub-alpine zone.

Attempts to hydroseed with hand collected *D. klossii* seeds have shown mixed results likely due to low seed germination and viability coupled harsh site conditions. The mine maintains a nursery where *D. klossii* seeds are propagated and seedling transplants have demonstrated good survivability on substrates within favourable pH ranges and with some organic matter and nutrients. Where site conditions allow, results have been encouraging.

More recently the mine experimented with various techniques to stabilise and reclaim overburden piles consisting of waste rock from underground tunnelling operations. The rock dump material is acidic with little organic matter, nutrients or microbial activity. Placement on long 2H:1V slopes precludes usage of *D. klossii* transplants due to the substrate challenges and more importantly – worker safety.

The mine evaluated germination and growth of *Deschampsia caespitosa* in addition to erosion control and growth establishment technologies; including a hydraulically-applied flexible growth medium (FGM), biostimulants, and a fast release micronized lime material.

In April 2013, using a multifactorial design, nine test plots (375 square meters each) were devised to assess erosion control effectiveness of the flexible growth medium, seeding rates of *D. caespitosa*, organic fertilizer rates, and biostimulant rates as shown in Table 1.

Table 1 – Mass of components per 375 m^2 test plot

Plot No.	FGM (kg)	Moss (kg)	*D. caespitosa* (kg)	Fertiliser (kg)	Biostimulant (kg)
1	136.2	50	0.3	0	0
2	136.2	50	0.3	16.8	0
3	136.2	50	0.3	33.6	0
4	136.2	50	0.6	0	0
5	136.2	50	0.6	16.8	0
6	136.2	50	0.6	33.6	0
7	136.2	50	0.6	0	4
8	136.2	50	0.6	16.8	4
9	136.2	50	0.6	33.6	4

Due to the limited amount of test materials, only one replicate of each treatment could be installed. However, the large plot sizes offered significant area to see differentiation in performance from plot to plot and throughout the plot from top to bottom. The plot shapes varied with the lower plot numbers being more rectangular across the slope face, while the higher numbered plots were more rectangular going up and down the slope face. Thus, the higher numbered plots offered longer slopes that could be considered to be more challenging and received the highest amounts of seed, fertiliser and biostimulants.

The application rate of the FGM was held constant at a rate of 3,638 kg/ha – which was slightly less than the recommended application rate of 3,900 kg/ha for the 2H:1V slopes used for the test plots. This was due to a limited supply of the material on this remote site. The rate of hand collected native moss sprigs were also held constant at 1,333 kg/ha per plot. On 10 and 11 April 2013, all materials were hydraulically-applied in one step using a 4,169 L capacity hydromulcher in predominantly wet, cool conditions.

Figure 9 – Hand collected moss sprigs

Figure 10 – Application of slurry to top of slope

Results after two years of monitoring were predictable given the sterile overburden characteristics combined with the challenging site conditions. Germination of the *D. caespitosa* was spotty and the seedlings demonstrated gradual die off from lack of organic matter, soil structure, microbial activity and available nutrients to sustain growth of the grasses. However, the moss sprigs performed reasonably well and have begun to colonize the exposed rock surfaces. There is a distinct correlation to amounts of organic matter and biostimulants applied to the test plots. Those plots with higher amounts of organic matter and biostimulants showed greater establishment and coverage of the moss as shown in Figures 11 and 12.

This is very encouraging as the mosses will colonize the rock and initiate the soil building process creating a more favourable habitat over time for *Deschampsia klossii* and other indigenous species to develop a sustainable ecosystem via the process of natural succession. Older reclaimed sites where soils are more developed demonstrate dense and uniform stands of *Deschampsia klossii.*

Figure 11 – Plots 1-7 – varying amendment rates

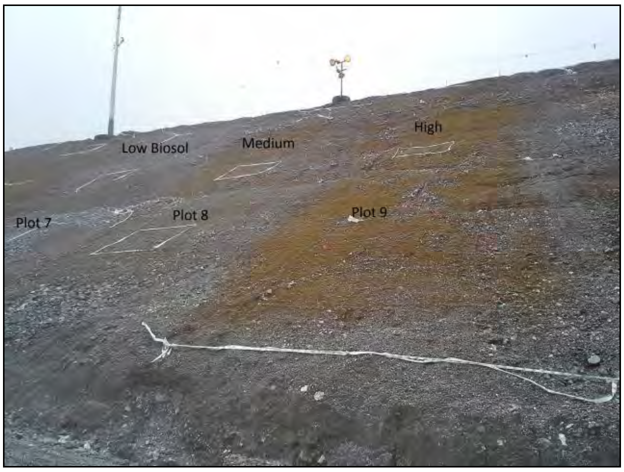


Figure 12 – Plots 8-9 with more amendments

The mine is planning more test plot installations in 2016 to expand research on relationships between types of and application rates of various erosion control and soil building treatments. This is all a part of their enduring commitment to create sustainable biodiversity in a very unique ecosystem.

Case History #4 – New Caledonia Nickel Mine

This large nickel mine is located on the Island of New Caledonia some 1,200 km off the east coast of Australia. The site ranges from 900 m in elevation where the ore is mined and then conveyed down to a port facility at sea level. The natural topography is mountainous while the climate may be characterized as tropical with a hot and humid season from November to March with temperatures between 27 °C and 30 °C, and a cooler, dry season from June to August with temperatures between 20 °C and 23 °C. Annual precipitation on the site ranges from 1,000-1,200 mm. Between December and April, tropical depressions and cyclones can cause wind speeds in excess of 100 km/hr. with gusts of 250 km/hr. and very abundant rainfall.

Initial soil testing revealed the reclamation sites had an abundance of metals and soil pH levels ranging from 5.5 to 8.2. Moreover, there was a lack of organic matter and nutrients present to encourage plant growth. Furthermore, Calcium deficiencies, excessive Magnesium levels and low Cation Exchange Capacities were observed. The surfaces requiring reclamation had been stripped during the mining process, so they lacked healthy topsoil. Slopes were steeper than 60° in places and contained valleys left by the mine excavation process. In areas with little rock the slopes were highly compacted with slick surfaces while other slopes were very rocky.

The combination of challenging site conditions, particularly – acidic substrates lacking organic matter and nutrients, long dry seasons mixed with wet seasons with intense rainfall and steep slopes – led to the selection of a wide-ranging portfolio of materials in the reclamation recipe. Earlier revegetation attempts with open weave jute mattings were unsuccessful due to the rough soil conditions in some areas and highly compacted slopes in others. Due to the challenging site conditions the mine selected a hydraulically-applied flexible growth medium (FGM) that provides a very high level of erosion protection while facilitating vegetative establishment. The FGM also provides up to 18 months of functional longevity which was deemed to be very important with slow vegetative establishment anticipated. In addition to the organic matter afforded by the FGM, the mine used compost produced on site from kitchen waste from their cafeteria and other sources of organics.

Outside of bamboo there are no indigenous graminoid species on the island of New Caledonia. Thus, all grass seeds must be imported. The initial seed mix included *Echinochloa esculenta* for an annual cover crop and *Chloris gayana* and *Cynodon dactylon* as perennial grasses to establish an erosion resistant cover. Over time indigenous forb and shrub species will begin to colonize and then dominate the grassed areas via natural succession and return the land to its natural ecosystems.

The mine utilized hydraulic seeding techniques in a two-step application process. Table 2 offers the prescribed components and application rates:

Table 2 New Caledonia material and application rates

First Pass		Second Pass	
Material	**Application Rate (kg/ha)**	**Material**	**Application Rate (kg/ha)**
FGM	1,680	FGM	2,240
Compost	2,775	Compost	1,100
Soil tackifier	20		

Crusting agent	30		
NPK fertilizer	200	NPK fertilizer	100
Urea fertilizer	100	Urea fertilizer	50
Fish emulsion	100 L/ha		
Annual grass	70		
Perennial grasses	25		

Installations were conducted in July/August of 2012 employing the two-step methodology as described above. Despite very heavy rains soon after installation establishment of the grasses was very good with consistent ground coverage as shown in Figures 13 and 14.

Figure 13 - Second application over compost layer

Figure 14 - Establishment of grasses in first year

Soon after these installations the New Caledonian government instituted regulations prohibiting the importation and use of non-indigenous grasses on mining sites. Thus, all seeds used moving forward must be harvested from the island. While the New Caledonian mines and government are learning which species seeds will be viable for massive reclamation efforts, this mine continues to employ the two-step methodology described, now with a third maintenance step occurring several months following the initial treatments. An additional mulch and organic layer is hydraulically-applied with more native seed, soil amendments and fertilizer to stimulate more germination and biomass from the slow growing shrub and forb species. The mine now has a nursery to better evaluate potential of various hand collected seeds from promising species.

Figure 15 - Indigenous shrub seedling growth

Figure 16 - Native plant nursery

Conclusions

The five fundamentals for mine rehabilitation as described and demonstrated in this publication have been a time-proven model for successful mined land reclamation over six continents working in many of the planet's biomes, climates, and environments; addressing multiple types of mining and substrates. Employing the discipline to work through the discovery sequence of the first three fundamentals – to analyse soils and substrates, pick the right plant materials for the site and select the most cost effective erosion and sediment control techniques, are requisites to head mine closure plans in the right direction. These fundamentals must be followed by the development of clear and comprehensive construction plans and specifications to effectively communicate reclamation or rehabilitation requirements to contractors and installers. Once construction commences, onsite oversight of acceptable installations must be conducted by qualified inspectors and experienced professionals who are intimately immersed in the mine closure requirements. Active rehabilitation sites must then be regularly inspected and maintained after each significant precipitation or other potentially damaging event. Inspections should be conducted by qualified professionals whose expectations are consistent with installer as well as the owner and regulatory entity(s).

Software programs can be used as a platform for coordinating a rehabilitation approach into a cohesive and interconnected framework for the designer as well as other project stakeholders. Processes within the software programs are enhanced with the continued input of environmental conditions, rainfall return frequencies and project information supplied by users around the world.

Case histories from different continents and contrasting environmental conditions were provided to briefly illustrate the discovery (required information gathering) and implementation of the mine rehabilitation fundamentals – leading to successful mined land rehabilitation. There are many more mining, energy, infrastructure, construction and other projects from other market segments around the world to substantiate the utility and legacy of this approach.

References

Erosion Control Technology Council. 2014 Rolled Erosion Control Products (RECPs) General usage and installation guidelines for Slope. Version 1.0. Roxborough, Colorado, USA, viewed http://www.ectc.org/assets/docs/ectcmarch2014_recp%20slope%20usage%20install%20guidelines%20final.pdf

Kilgore, R.T. and Cotton, G.K. 2005. Design of roadside channels with flexible linings. Hydraulic Engineering Circular No. 15 (HEC-15), Publication No. FHWA-NHI-05-114, US Federal Highway Administration, Arlington, Virginia, USA.

Profile Products LLC. 2015. Soil Testing and Interpretation Technical Document. Buffalo Grove, Illinois, USA, viewed http://profilelibrary.info/Files/Soil%20Testing%20Properties%20Tech%20Bulletin.pdf

Renard, K.G., Foster, G.A., Weesies, D.K., and Yoder, D.C. 1997. Predicting Soil erosion by water: a guide to conservation planning with the Revised Universal Soil Loss Equation (RUSLE). US Department of Agriculture, Agriculture Handbook No. 703, 404 pp.

Theisen, M.S. 2015. Five Fundamentals for Successful Mined Land Rehabilitation. *Proceedings of 2015 Mine Closure Conference.* Vancouver, British Columbia, Canada.

BUILDING CANADA'S EMERGING RARE EARTH ELEMENTS AND CHROMITE INDUSTRY THROUGH TECHNOLOGICAL INNOVATION

Bryan Tisch and Janice Zinck,
Natural Resources Canada, CanmetMINING, Ottawa, Ontario, K1A 0G1

In recent years, the steady, reliable, and secure supply of critical metals has become increasingly important to major industrialized economies that seek to sustain their industrial base and develop advanced technologies, such as clean energy. In light of this, Canada, with its significant critical metal reserves, has an opportunity to supply some of the global demand for critical metals. However, to transition from promising mineral deposits to marketable products, investment in fundamental R&D and expertise is needed to address the complex technological challenges around the production, separation and processing of critical metals, and to better understand the global market for these key commodities. For many small and medium enterprises, investing in R&D is extremely challenging with their limited resources; federal investment in R&D would catalyze the development and growth of new businesses and high-value jobs. As a result, the Government of Canada is investing $23 million over six years to accelerate the production of rare earth elements (REE) and chromite/ferrochrome. This industry-led program delivered by CanmetMINING, Natural Resources Canada has the goal to equip these emerging Canadian industries with the technological innovation needed to reach production.

Rare earth elements are critical minerals that represent an opportunity for Canada to enter an emerging and globally strategic market. Canada does not currently produce rare earth elements, but has deposits with significant potential. REE are differentiated into “light” or “heavy” rare earths based on their atomic number. Light REE, whose use is important to many low technology commercial products, typically comprise more than 95% of the available rare earths in any given deposit and are in surplus supply in the global marketplace. Heavy REE typically comprise less than 2% of the recoverable rare earths in a deposit and are considered to be critical to the manufacture of all high technology, clean energy, aerospace, automotive, defence and many other industrial products. A number of Canadian projects have relatively high levels of heavy rare earths. While rare earths are abundant geologically, they are economically recoverable in only a few mineral deposits. The metallurgy for Canadian ores containing rare earth elements involves a complex sequence of individual processing, separation, refinement, alloying and formation stages before they can be used in the production of permanent magnets, consumer electronics and other high value-added high-tech products.

According to the Technology Metal Research Group, there are currently there 53 advanced rare-earth projects located within 16 countries, and 19 of them are located in Canada (Figure 1), primarily in northern Ontario and Quebec, Newfoundland and NWT.

Chromite deposits located in Ontario's Ring of Fire have production potential that could make Canada a significant global producer, processor and supplier of products that contain the metal chromium. Over 90 per cent of global chromite production is used to manufacture stainless steel and other alloys. There is no substitute for this mineral in the production of stainless steel, which has unique corrosion resistance properties. Chromium-based alloys are also used in gas turbines, aircraft engines and other high temperature applications. The Ring of Fire is estimated to hold about 220 million tonnes of chromite. Global demand for stainless steel is forecast to grow at 4-5 per cent annually to 2020. Although major steel mills exist in North America, there is currently no chromite production in North America and has never been mined before in Canada.

The program utilizes both industry-led steering committees as well as numerous technical committees to help define technical work plans and to ensure that the R&D is well-focussed and address industry needs. The research will be strengthen by integrating comprehensive economic analysis of market conditions for rare earths and chromite, in order to increase the Government's understanding of the economic and market dynamics, and their implications for the business case for rare earth and chromite projects in Canada.

A key aspect to the success of this initiative is the involvement of stakeholders throughout the value chain in defining and deploying the technological innovation. Ongoing industry engagement will ensure that the program will de-risk processing challenges to reduce capital and operating costs, develop and commercialize technologies to separate REE and produce ferrochrome and evaluate secondary sources of these commodities as a means of accelerating production and minimizing environmental liability. Stakeholders are further engaged through attendance at annual workshops and through dissemination of reports and research findings online at http://www.reechromite.ca.

External research will complement the federal research by utilizing specialize expertise and addressing research and technological gaps with a multi-directional approach. Post-doctoral fellows, undergraduate and graduate students will work with federal researchers and universities to strengthen and support research teams and at the same time will accelerate the development of HQP for these emerging industries. External contracts will also be offered through standard government of Canada procurement channels such as Requests for Proposals and other Supply Arrangements.

Processing challenges will look primarily at physical (beneficiation) and hydrometallurgical aspects for REE, and pyrometallurgical and efficiencies in rock breakage (comminution) for energy and cost savings. As separation is the most complex but most critical step in rare earth processing a significant R&D investment will be made in this area. Reprocessing examines options to secure a source of critical metals from secondary sources, mainly mining wastes. Environmental research will address a range of concerns from hexavalent chromium, to radioactivity, to aquatic and terrestrial contamination.

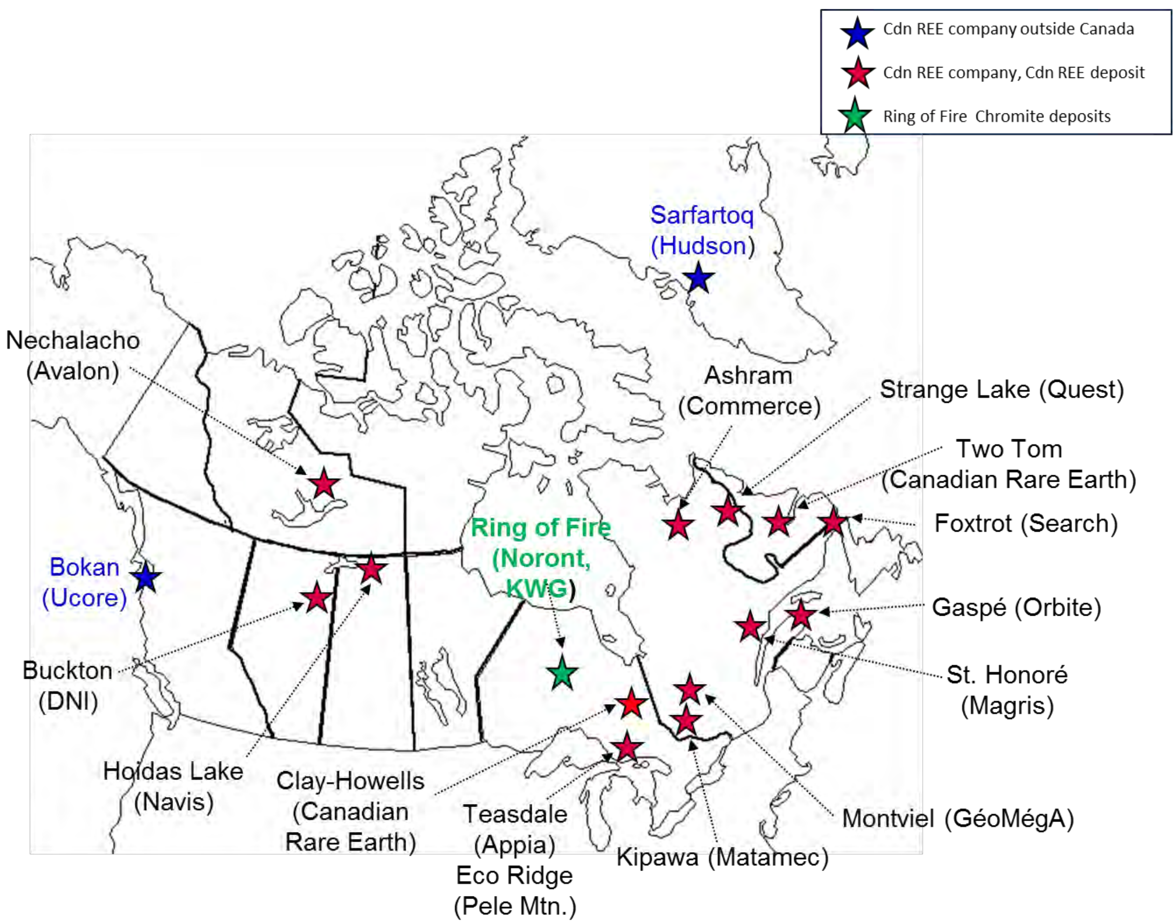


Figure 1: Rare Earth Element and Chromite Deposits in Canada.

Abandoned Mines, NOAMI and Ecotourism

Gilles A. Tremblay[1] and Charlene M. Hogan[1]
Natural Resources Canada, Ottawa, ON, K1A 0G1

Key Words: Orphaned, abandoned, relinquishment, decision-making

Orphaned and abandoned mines are those mines where the owner cannot be found, or is unwilling or unable to remediate the site. They present a number of environmental, health and safety, social and economic problems. The land may be derelict – contaminated, unstable, and/or with dangerous mining infrastructure. They are often expensive to remediate and pose a major public concern in and beyond mining areas.

The National Orphaned/Abandoned Mines Initiative (NOAMI) was created in 2002 at the request of the Canadian Mines Ministers, and based on recommendations put forward at a multistakeholder workshop held in Winnipeg in 2001. The workshop determined the key issues associated with orphaned/abandoned mine sites, and laid down a series of guiding principles and objectives which apply to NOAMI as it exists today.

NOAMI is guided by a multi-stakeholder committee with representatives from government, non-governmental organizations, the mining industry and Indigenous Canadians. They work together to address issues related to remediation of orphaned and abandoned mines in Canada. One of the guiding principles for NOAMI is that "work toward eliminating future abandonment must continue, including the tightening of regulatory approaches." Committee members felt that there is a need for a clear policy framework governing mine closure, long-term liabilities and return (or relinquishment) of mining lands to the Crown.

Over the last six years NOAMI has undertaken various studies to explore these issues and offer recommendations for change. Important tools and guidance documents have been produced that together will make a major contribution towards development of a policy framework to address all aspects of managing abandoned mine liabilities in the long-term, and to prevent future abandonments in Canada.

The initial report, a guidance document for mine closure and management of long-term liabilities, was produced by Cowan Minerals Ltd. (2010). The report examined major components related to mine closure and post-closure site management, including long-term maintenance and monitoring, financial assurance, relinquishment and instititutional care. This has served as a valuable reference tool, and presents a policy framework, together with recommendations for preventing further accrual of abandoned mine hazards. A key message is that jurisdictions should have a managed relinquishment process, that is clear and unfettered, and specific about what will not be accepted The report notes that closure plans are normally "designed for closure", and recommends that a more-forward approach be applied and they should be "designed for relinquishment".

Building and based on the “Cowan report”, a multi-stakeholder workshop exploring the management of long-term liabilities and relinquishment (Tunis, 2011) was held and guidance obtained to assist NOAMI in developing a roadmap for managing long-term liabilities and issues related to mine relinquishment. This led to a further two-part study *Case Studies and Decision-Making Process for the Relinquishment of Closed Mine Sites* (Cowan Minerals Ltd., 2013). The first part examined six case studies from different Canadian jurisdictions that underline “lessons learned” towards their potential relinquishment. The second part, a decision process (Fig. 1), identifies issues to consider when determining if a site could, or should, be returned to the Crown or remain the responsibility of the operator. The process provides a starting point for developing or revising a program for relinquishment; however, each jurisdiction must establish a process that meets its own regulatory regime and policies. The report concludes that mining projects should be designed with the objective of reclaiming the site for possible relinquishment, and future beneficial use.

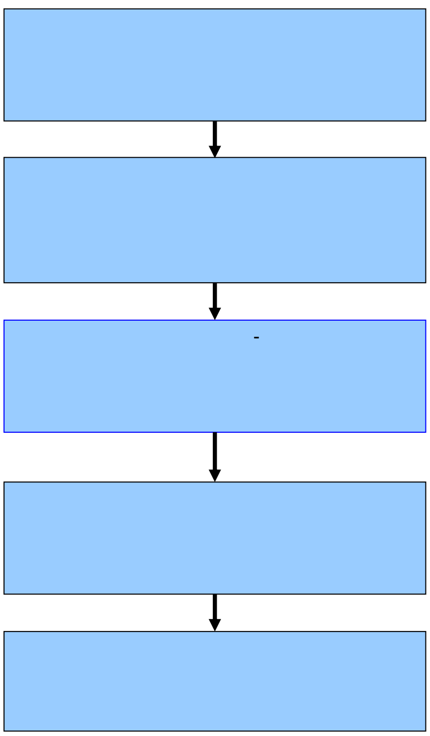


Figure 1: A five-step decision-making process for relinquishment (Cowan Minerals Ltd., 2013)

An effective long-term monitoring and maintenance program at inactive mine sites presents a number of challenges. Kingsmere Resource Services Inc. undertook a study for NOAMI to develop criteria to assess these sites in order to evaluate their condition and provide direction for the planning and delivery of long-term stewardship. As such, the study would address Steps 1 and 2 in the decision-making process. The report entitled *Key Criteria for the Effective Long-term Stewardship of Closed, Orphaned/ Abandoned Mine and Mineral Exploration Sites* (2015) outlines site aspects involved in identifying, analyzing and evaluating potential site hazards, including those that may pose a risk to public health and safety, to the environment, to ecosystem.

A study to address Step 3 *"Mine Closure and Long-term Management: Cost Estimation"* is currently underway by Kingsmere Resource Services. This step considers the various requirements for long-term monitoring, maintenance or capital replacement of rehabilitation works and cost estimation for this work. This work ranges from routine activities such as water quality monitoring or fence inspections to estimated replacement schedules and costs for capital works, such as shaft cap replacement or treatment plant components. The intent would be to develop a methodology to assess the risk of site-specific characteristics, and to develop a cost-estimate methodology for long term monitoring, maintenance and failure remediation financial liabilities.

A mine ends – what happens then!

Post-mining regeneration seeks to convert negative legacies, such as abandoned mines, into positive ones. There are numerous national and international examples of innovative projects that are delivering a variety of social, economic and environmental benefits post-mining. In 2009, the Post-Mining Alliance produced a book "101 Things to do with a Hole in the Ground" that is a snapshot that includes examples of ecotourism, wildlife habitats, educational, sports and leisure facilities and other industrial uses. International examples include the Eden Project in the UK, the Wieliczka Salt Mine in Poland, the Football stadium in Braga, Portugal. In Canada, there is the Butchart Gardens near Victoria, BC, and Dynamic Earth and Science North in Sudbury, ON.

Literature Cited

Cowan Minerals Ltd. 2010. The policy framework in Canada for mine closure and management of long-term liabilities: A guidance document. http://abandoned-mines.org/publications-e.htm

Cowan Minerals Ltd. 2013. Case studies and decision making process for the relinquishment of closed mine sites. http://abandoned-mines.org/publications-e.htm

Kingsmere Resource Services Inc. 2015. Key criteria for the effective long-term stewardship of closed, orphaned/abandoned mine and mineral exploration sites. http://abandoned-mines.org/publications-e.htm

Kingsmere Resource Services Inc. (in progress). Mine Closure and Long-term Management: Cost Estimation.

Tunis, E.S. 2011. NOAMI workshop – Exploring the management of long-term liabilities and the return of mining lands to the Crown in Canada. May 31 – June 1, 2011. Ottawa, ON. http://www.abandoned-mines.org/en/workshops/

Post Mining Alliance. 2009. 101 Things to do with a Hole in the Ground.

TECHNOSOL EVALUATION FOR MINE SITE RECLAMATION ON THE BOREAL SHIELD

Renate Vanderhorst[1], Autumn Watkinson[2], Shane Hayes[3], Peter Beckett[1], Graeme Spiers[1]

[1]Vale Living With Lakes Centre, Laurentian University, Sudbury, ON P3E 2C6
[2]Department of Renewable Resources, University of Alberta, Edmonton, AB T6G 2R3
[3]Barrick-Hemlo, Marathon, ON P0T 2E0

Key Words: Technosol, manufactured soil, soil microclimate, chemical elements, native species

Abstract

The application of Technosols manufactured from industrial waste materials for use in mine reclamation is of interest due to the lower associated environmental and economic costs. Long-term studies of the development of Technosols will provide information on how the changing soil conditions will affect plant survival and growth, information critical for determination of their utility as a site reclamation material. The ability of Technosols to support the growth of vegetation on gold mine sites on the Canadian Shield is being examined in order to determine if they provide a viable reclamation option following mine closure. Technosols were manufactured from crushed mine rock and woody residuals in fixed ratios (60:40 and 20:80) and placed on the surface of mine rock lysimeters at 30 cm or 60 cm depths. Soil microclimate data at various depths was monitored from late 2012 to 2015 and compared with microclimate data from both more traditional reclamation materials, and also from a nearby natural pedon. Technosol pore-water chemical composition and physical changes observed over the same period are also highlighted in this study.

Introduction

Site reclamation, a key component in mine closure plans, requires research into cost-effective reclamation methods beneficial to both the mining industry and the post-mining environment. Mining, particularly surface mining, severely disturbs areas by removing vegetation, topsoil, and geological materials (Turcotte, Quideau, & Oh, 2009); this effectively returns the land to a state for primary successional processes to operate (MacKenzie & Quideau, 2010). Reclamation of these sites therefore requires the addition of soil material to the site (Macdonald et al., 2015).

Soil materials which are commonly added to sites as part of reclamation efforts include stockpiled soils removed during mine construction (Gaster, Karst, & Landhäusser, 2015; Sorenson, Quideau, MacKenzie, Landhäusser, & Oh, 2011), as well as soils constructed from a combination of stockpiled overburden materials and organic materials such as peat (MacKenzie & Quideau, 2010). These soils often have unusual physical and chemical properties as a result of their human origin, and follow development trajectories which differ from those of natural soils (Leguédois et al., 2016). They are classified as Technosols by the World Reference Base for Soil Resources due to the strong influence human activities have had on their formation (IUSS Working Group WRB, 2014).

Manufacturing a soil for mine reclamation from local industrial waste materials such as mine overburden or lumber mill residuals could have considerable environmental and economic benefits. The necessity of transporting large volumes of soil material from other locations is removed, providing cost savings, reducing greenhouse gas emissions, and preventing the disturbance of soil source sites. Technosol manufacture also provides a use for waste products which otherwise would be valueless. Long-term studies of the development of Technosols are necessary to understand how changing soil conditions will affect plant survival and growth, which will determine their utility as a reclamation material.

Previous research into the use of a Technosol to revegetate a non-acid generating rock pile from a gold mine in the Boreal Shield found that crushed mine rock blended with woody residuals from a lumber mill was able to support vegetation such as green alder (*Alnus viridis*) even at early stages in soil development (Watkinson, 2014). Further study into the changes in soil properties and their ability to support larger and more diverse communities of native Boreal vegetation as the Technosol matures will provide increasing insight into the use of Technosols created for mine reclamation.

Methods

Research is being conducted at Barrick-Hemlo (Williams Mine) in Hemlo ON, north of Lake Superior. The combination of an open pit and underground workings has generated a large rock pile which will need to be revegetated upon mine closure. The mine rock is predominantly metasedimentary and intermediate volcanic rock and is non-acid generating; therefore it was determined to be a suitable mineral component for the formation of a Technosol to reclaim the rock pile. The woody residuals organic component of the Technosol consists of sawdust, bark, and off-cuttings of boreal coniferous trees from a lumber mill located about 40 min away in White River.

In the summer of 2012, two Technosols were manufactured from the crushed mine rock and woody residuals in fixed ratios (60:40 and 20:80) to obtain low organic and high organic materials. They were placed on the surface of mine rock lysimeters at 30 cm or 60 cm depths. Each soil-depth combination was replicated four times for a total of twelve plots.

Within each plot 5TM soil temperature/moisture sensors were installed at 10, 30, and, where applicable, 60 cm (Figure 1). Additional sensors were installed at 5 cm July 2014. Each plot also contains one MPS-2 water dielectric potential/temperature sensor at 30 or 60 cm. In July 2014 sensors were installed in the same manner in the successional field behind the plots, an area reclaimed using traditional methods, and in a nearby forest site. These sensors allow the Technosol microclimate to be compared to that of more natural regional soils.

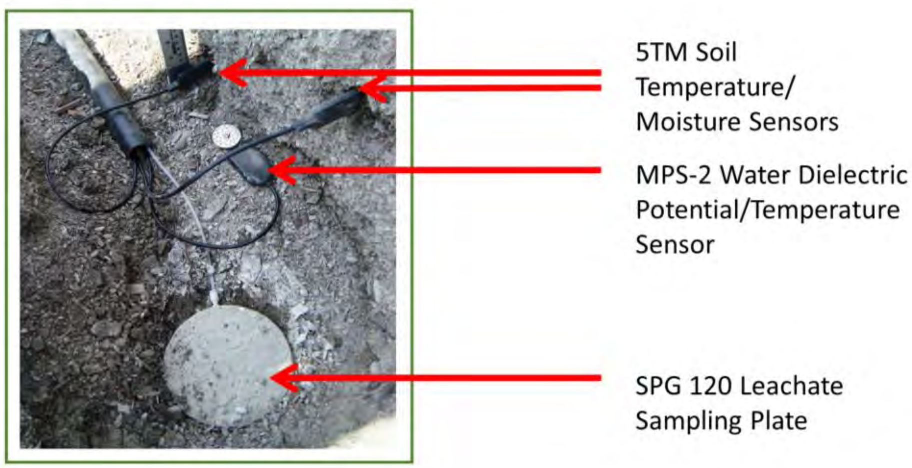


Figure 1. Position of sensors within the Technosol profile. 5TM sensors are located at 5, 10, 30, and (where applicable) 60 cm; MPS-2 sensors and leachate sampling plates are located at base of Technosol layer (30 or 60 cm).

The base of the lysimeters is connected to a large barrel through a drainage tube attached to the geomembrane, allowing the collection of water which has percolated through both the Technosol and mine rock layers (gravity through-flow, Figure 2). There is also a soil leachate sampling plate made of porous borosilicate glass installed in each lysimeter approximately 5 cm above the Technosol/mine rock interface to sample plant root available water held in the Technosol (tension water; Figure 1). These plates draw in water under low vacuum, being pressurized to -0.5 bar with a hand pump to obtain samples. The sample collected represents water held in the soil between field capacity and plant wilting points which is available to plant roots. The water samples are analyzed for pH, EC, Eh, dissolved organic carbon (DOC), anions, and other elements. Uncovered mine rock test cells operated by the mine are also monitored provide information on the natural biogeochemical weathering release from the rocks to percolating waters through time.

Figure 2. Barrel connected to base of Technosol lysimeter for collection of gravity through-flow water.

Green alders (*Alnus viridis*), a native species, were planted in summer 2013 and again in summer 2015 to ensure each plot contained nine plants. Bearberry (*Arctostaphylos uva-ursi*) was also planted in summer 2015, at twelve plants per plot.

Results and Discussion

Temperature data from the end of June 2014 to mid-October 2015 shows large differences in the 10 cm soil temperature variation in the Technosol plots compared to the natural forest soil (Figure 3.) The Technosol plots appear to have more variable temperatures at 10 cm that are closer to the extremes in air temperature than the forest soil. This is particularly noticeable in the winter, where the Technosols drop well below 0°C in late November and remain there until May; in contrast, the temperature in the forest soil remains around 0°C throughout the winter months. Some of the difference in winter temperature may be attributable to the exposure of the Technosol plots, which may prevent the building up of the deeper snow pack which insulates the forest soil. However, as the rock pile itself is also highly exposed, the conditions of the plots are a more accurate simulation of conditions on the pile than the observed adjacent forest soil.

There is also a clear difference between the Technosols and forest soil temperature during summer, particularly in the low-organic Technosol plot where soil temperatures reach air temperature. The temperature at 10 cm in the high organic Technosol, though not as extreme or variable, is still noticeably warmer than the forest soil from the end of May throughout the summer. Again, the exposure of the plots to sun and wind likely contributes to the higher temperatures than those observed in the forest soil pedon. The difference between the high and low organic Technosols may be due to the increased moisture holding capacity of the high organic Technosol.

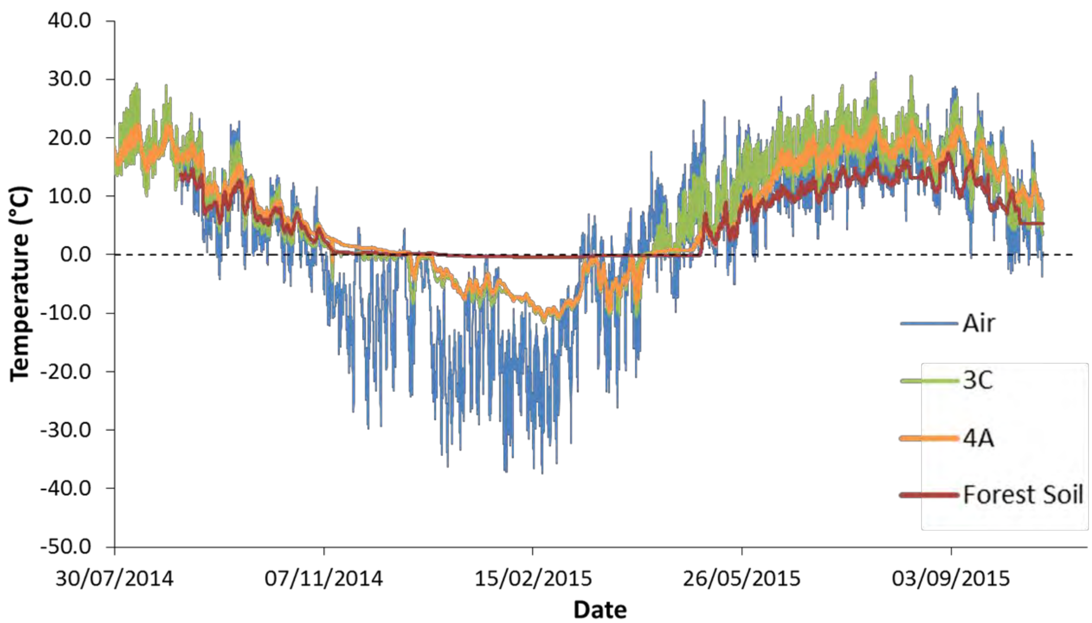


Figure 3. Temperature at 10 cm depth in low (3C) and high (4A) organic Technosol plots and forest soil (June 2014 – Oct 2015). Air temperature at 3 m is also displayed.

The pH of gravity through-flow samples from the Technosol plots from Oct 2012 to Oct 2015 indicates that the water percolating through the Technosol has remained close to neutral (Figure 4) throughout the three-year measurement period. No pH differences between the Technosol treatments were seen. There were also no differences between the tension water and gravity through-flow samples. On average, Technosol waters were slightly more acidic than samples coming from the uncovered mine rock test cells during the same period. The Technosol water samples have a similar pH to the natural waters within the region. These values are likely buffered by the carbonate minerals present within the rock.

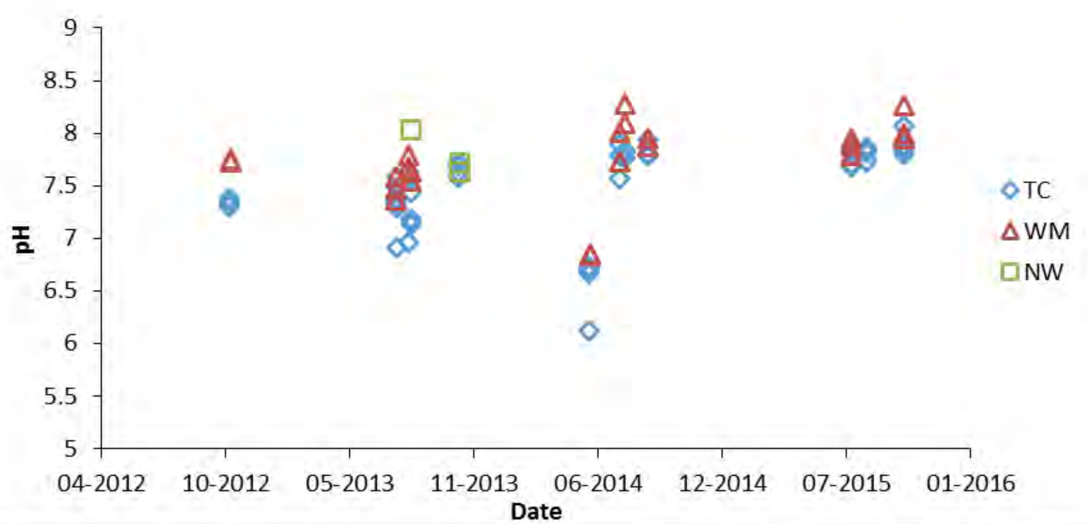


Figure 4. pH of gravity through-flow from Technosol plots (TC), mine rock test cells (MR), and natural waters (NW) from Oct 2012 to Oct 2015.

While there was little difference between the pH of the uncovered test cells and the Technosol plots, there is a clear difference in electrical conductivity (Figure 5), which is a proxy for the total dissolved ions present in the water. The gravity through-flow samples from the Technosol plots clearly have a higher conductivity than the uncovered mine rock plots and natural waters in the region, and this difference appears to be more pronounced with time. Some of the observed differences could be due to the fact that the mine rock in the Technosols is crushed to more surface reactive gravel size, much finer than the rock used in the mine rock test cells. Dissolved substances released from the organic material may also be contributing to greater biogeochemical weathering in the Technosol plots than the mine rock test cells.

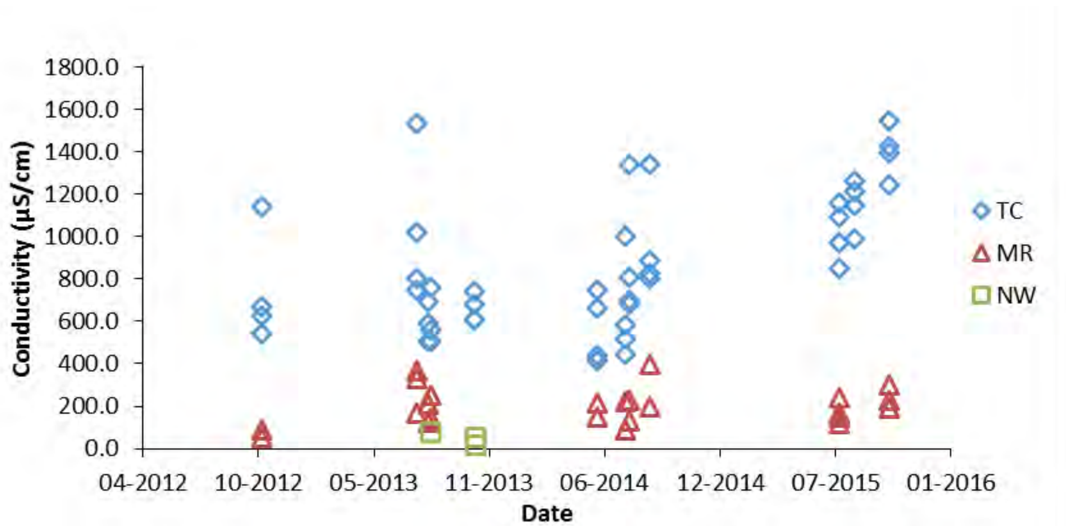


Figure 5. Conductivity (µS/cm) of gravity through-flow from Technosol plots (TC), mine rock test cells (MR), and natural waters (NW) from Oct 2012 to Oct 2015.

The behaviour of select elements within the gravity through-flow and tension water samples from the Technosol plots has been compared to those in the uncovered mine rock test cells. The concentrations of calcium and magnesium divalent cations within Technosol gravity through-flow and tension water samples are consistent higher than concentrations within water samples from the uncovered mine rock plots (Figure 6). Calcium concentration in gravity through-flow is noticeably greater than concentrations in tension water, particularly in 2015. Magnesium concentrations also appear to be greater in gravity through-flow samples, although the pattern is less clear. Samples from the 60 cm low-organic Technosol plots typically have the highest concentrations for both elements, and remain relatively consistent from 2013 to 2015.

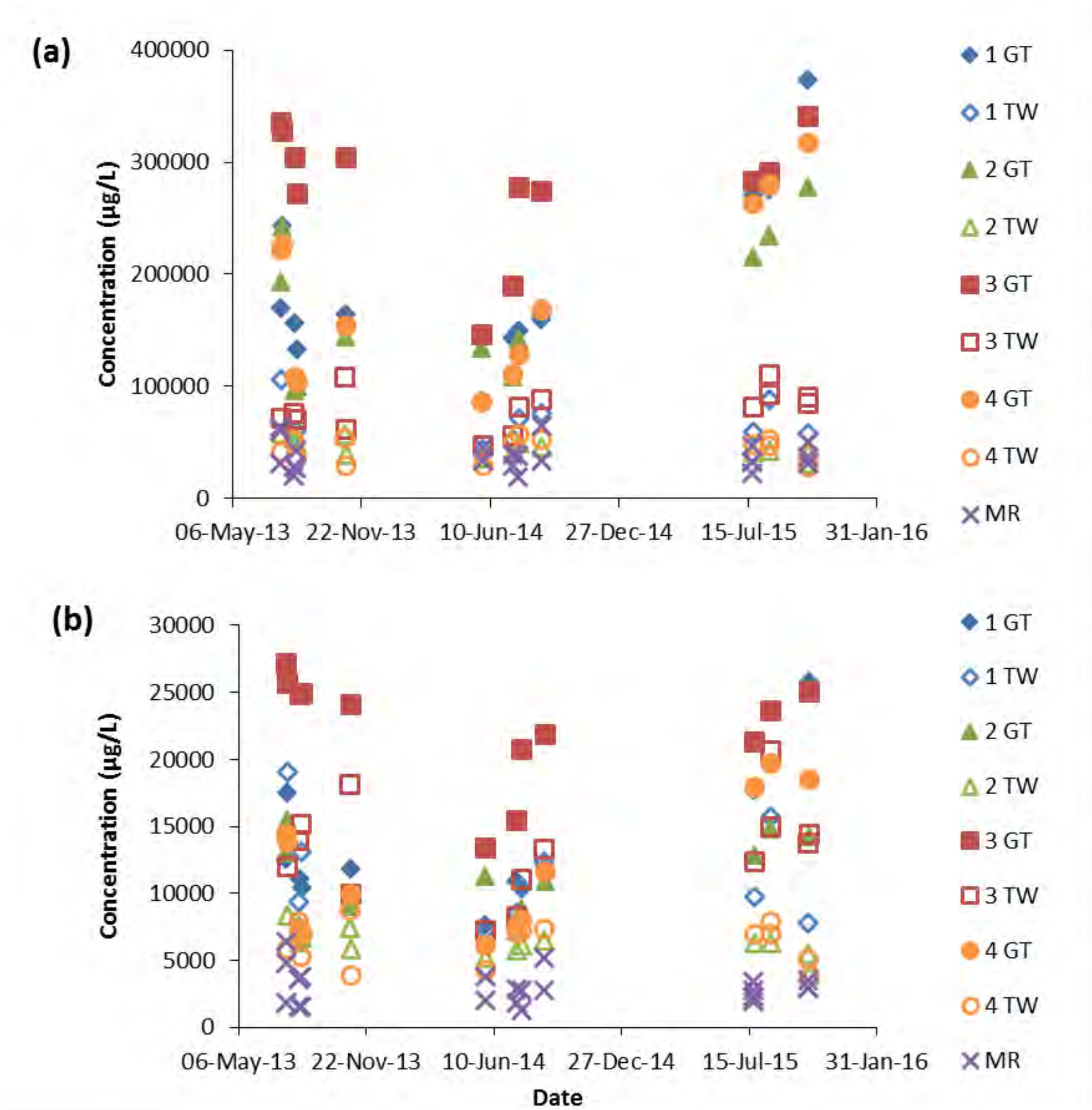


Figure 6. Concentrations of (a) Ca and (b) Mg in Technosol gravity through-flow (GT, solid fill), tension water (TW, hollow), and mine rock test cell (MR) water samples. Treatments are 1: 40% organic, 30 cm; 2: 80% organic, 30 cm; 3: 40% organic, 60 cm; 4: 80%, 60 cm.

The redox-sensitive elements iron and manganese show a similar trend in decreasing concentrations from 2013 to 2015 (Figure 7). Initially the concentrations of both elements in gravity through-flow samples from the Technosol plots were much higher than concentrations in water samples from the uncovered mine rock test cells, while tension water samples had lower concentrations more similar to the mine rock test cell samples. However, there appears to be an exponential decrease in concentration, particularly in iron. By 2015 there is no difference in concentrations of iron and manganese between Technosol gravity through-flow, tension water, or mine rock test cell samples.

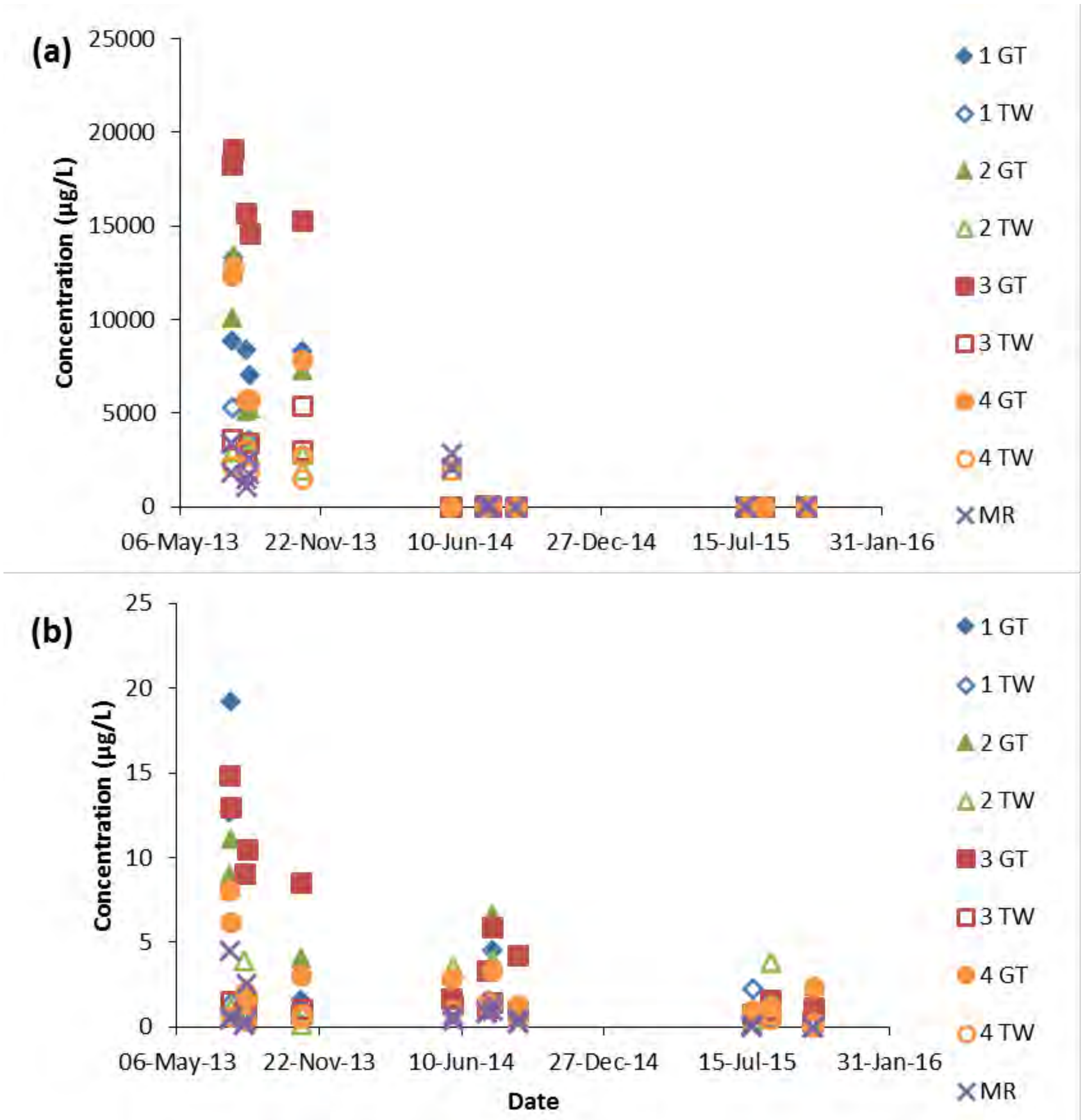


Figure 7. Concentrations of (a) Fe and (b) Mn in Technosol gravity through-flow (GT, solid fill), tension water (TW, hollow), and mine rock test cell (MR) water samples. Treatments are 1: 40% organic, 30 cm; 2: 80% organic, 30 cm; 3: 40% organic, 60 cm; 4: 80%, 60 cm.

The behaviour of copper, an important trace nutrient, and cadmium, an element of environmental concern, within the water samples were very different. Cu shows no trend as the Technosol matures, with no clear difference in concentrations in gravity through-flow and tension water (Figure 8a). However, more copper appears to be released from the Technosol plots than from the mine rock test cells.

There is a clear trend in cadmium concentration of increasing with time (Figure 8b), with gravity through-flow samples having higher cadmium concentrations than tension water samples. This suggests cadmium is more likely to be released into the environment than to be held in the water in the Technosol. Mine rock test cell water samples had highly variable concentrations of Cd, with the cells containing intermediate volcanics and quartz eye muscovite schist having much greater concentrations than in any of the

Technosol plots. Samples of percolating waters from other rock types were comparable to the Technosols (data not shown).

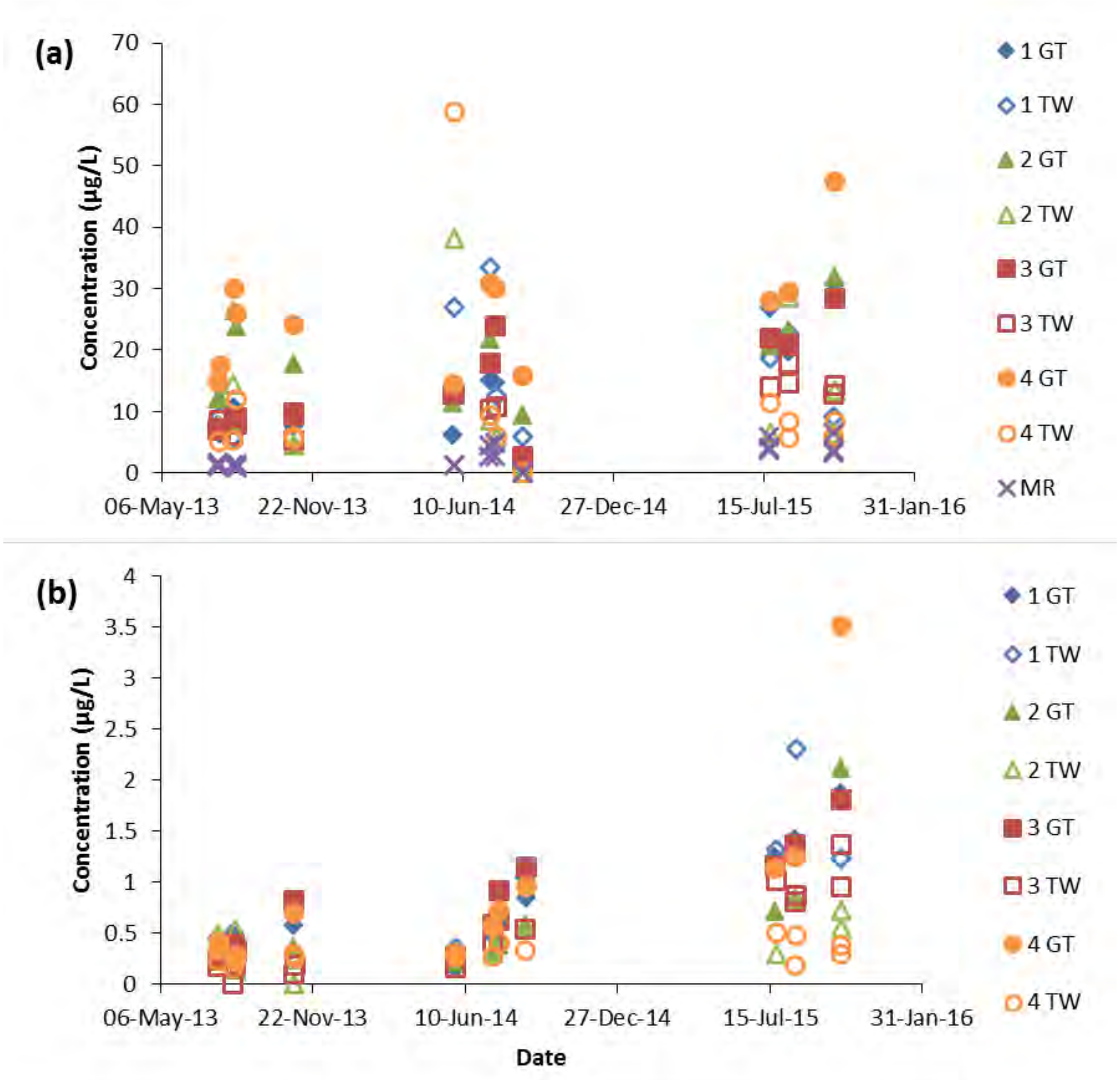


Figure 8. Concentrations of (a) Cu and (b) Cd in Technosol gravity through-flow (GT, solid fill), tension water (TW, hollow), and mine rock test cell (MR) water samples. Treatments are 1: 40% organic, 30 cm; 2: 80% organic, 30 cm; 3: 40% organic, 60 cm; 4: 80%, 60 cm.

Concentrations of molybdenum and phosphorus, both oxyanions, in water samples also showed very different trends (Figure 9). Molybdenum concentrations are generally higher in gravity through-flow samples than in tension water, while there are no clear differences observed for phosphorus. Phosphorus concentrations decrease with time, while molybdenum concentrations remain fairly constant. Phosphorus concentrations in Technosol plots are generally higher than in the uncovered mine rock test cells. Molybdenum has a similar pattern to Cd in terms of differences between the Technosols and the mine rock; lysimeters containing intermediate volcanics and quartz eye muscovite schist have much greater molybdenum concentrations than any of the Technosol plots while other rock types display comparable levels over time.

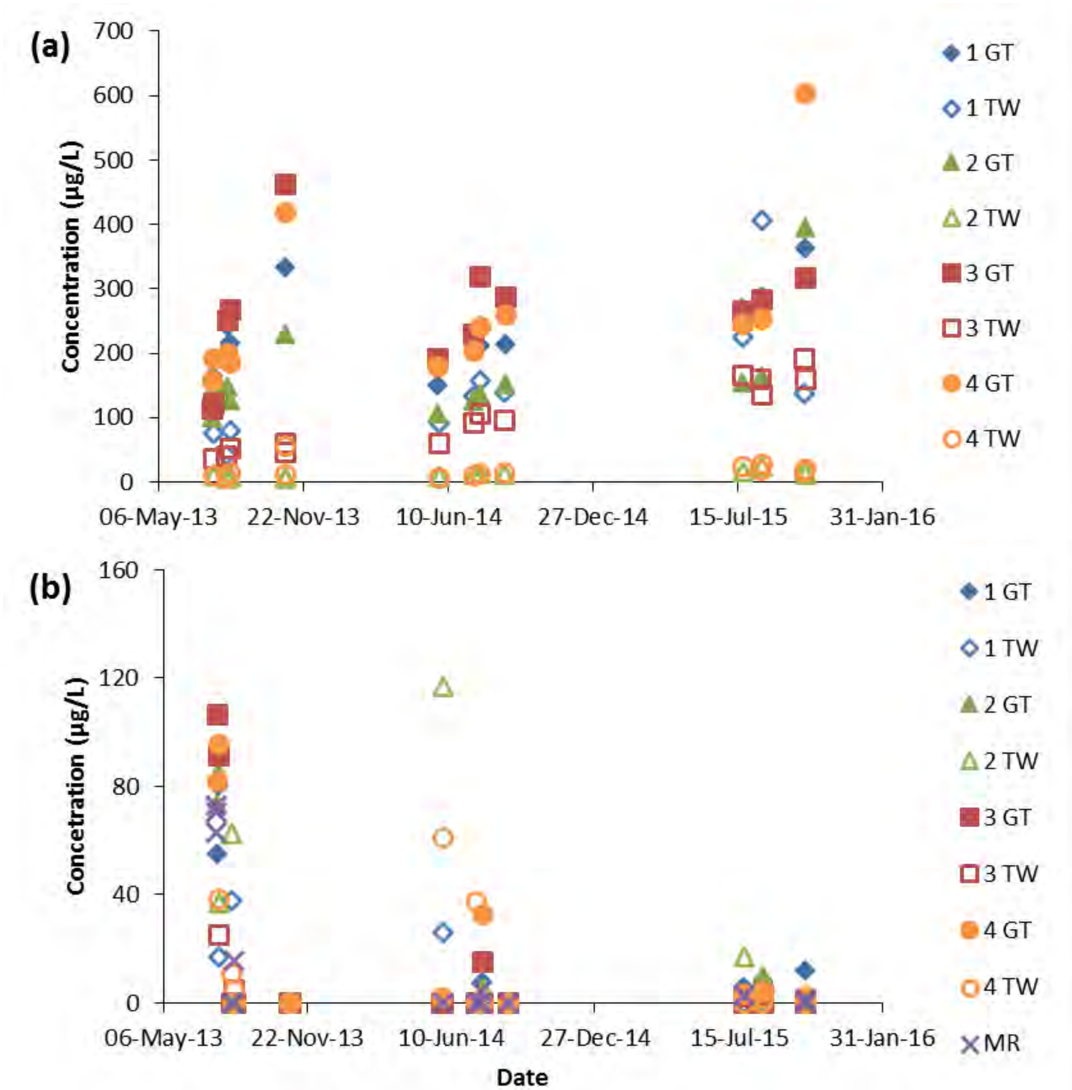


Figure 9. Concentrations of (a) Mo and (b) P in Technosol gravity through-flow (GT, solid fill), tension water (TW, hollow), and mine rock test cell (MR) water samples. Treatments are 1: 40% organic, 30 cm; 2: 80% organic, 30 cm; 3: 40% organic, 60 cm; 4: 80%, 60 cm.

Conclusion

The temperature at 10 cm depth in the Technosol plots was noticeably more variable than in a natural forest soil, displaying greater extremes in both summer and winter. There were no differences between the pH of the Technosol, uncovered mine rock, and natural water samples, although there was a clear increase in Eh in the Technosol samples. Comparison of element concentrations in the Technosol and mine rock water samples revealed the addition of the organic matter increased the concentration of calcium, magnesium, and copper, while decreasing trends were seen for iron, manganese, and phosphorus concentrations, and an increasing trend was seen in cadmium.

Literature Cited

Gaster, J., Karst, J., & Landhäusser, S. M. (2015). The role of seedling nutrient status on development of ectomycorrhizal fungal communities in two soil types following surface mining disturbance. *Pedobiologia*, *58*(4), 129–135. http://doi.org/10.1016/j.pedobi.2015.07.001

IUSS Working Group WRB. (2014). World reference base for soil resources 2014. International soil classification system for naming soils and creating legends for soil maps. *World Soil Resources Reports No. 106*, 1–191. http://doi.org/10.1017/S0014479706394902

Leguédois, S., Séré, G., Auclerc, A., Cortet, J., Huot, H., Ouvrard, S., … Louis, J. (2016). Modelling pedogenesis of Technosols. *Geoderma*, *262*, 199–212.

Macdonald, S. E., Landhäusser, S. M., Skousen, J., Franklin, J., Frouz, J., Hall, S., … Quideau, S. (2015). Forest restoration following surface mining disturbance: challenges and solutions. *New Forests*, *46*(5-6), 703–732. http://doi.org/10.1007/s11056-015-9506-4

MacKenzie, M. D., & Quideau, S. A. (2010). Microbial community structure and nutrient availability in oil sands reclaimed boreal soils. *Applied Soil Ecology*, *44*(1), 32–41. http://doi.org/10.1016/j.apsoil.2009.09.002

Sorenson, P. T., Quideau, S. A., MacKenzie, M. D., Landhäusser, S. M., & Oh, S. W. (2011). Forest floor development and biochemical properties in reconstructed boreal forest soils. *Applied Soil Ecology*, *49*(1), 139–147. http://doi.org/10.1016/j.apsoil.2011.06.006

Turcotte, I., Quideau, S. A., & Oh, S. W. (2009). Organic matter quality in reclaimed boreal forest soils following oil sands mining. *Organic Geochemistry*, *40*(4), 510–519. http://doi.org/10.1016/j.orggeochem.2009.01.003

Watkinson, A. (2014). *Developing manufactured soils for reclamation of mined land in the boreal forest ecosystem*. *MSc Thesis*. Retrieved from https://zone.biblio.laurentian.ca/dspace/handle/10219/2262

FROM GOLDFIELDS TO GREENFIELDS: THE LEGACY OF THE HOLLINGER GOLD MINE

Cindy Welsh
City of Timmins, Timmins, Ontario, P4N 1B3

Don Burke
Goldcorp – Porcupine Gold Mines, South Porcupine, Ontario, P0N 1H0

Bryan Neeley
Goldcorp – Porcupine Gold Mines, South Porcupine, Ontario, P0N 1H0

Introduction

The Hollinger Gold Mine was one of the richest producing mines in the western hemisphere and operated between 1910 and 1968 as an underground operation within the City of Timmins. With the closure of the mine in 1968, numerous mine hazards began to plague the property and were public and safety liabilities located within the City of Timmins, particularly the urban area. In 2007, Goldcorp's Porcupine Gold Mines began to investigate the revitalization of the former Hollinger Gold Mine through the open pit mining process. The question arose of how to develop a best management approach for this resource development using economic development and land use planning principles, for an industry that is located within an urban area of a major City located within Northern Ontario.

This paper will explore the best management approach that was developed to bring this project to fruition. The first section will provide a background and overview of the project. The second section will focus on the process and detail used in the best management approach and plan in terms of economic development and land use planning to permit a traditional resource based "mining" activity to develop within the context of an urban setting. The third section will explore how the land use planning process can be used as an economic development tool to guide the transformation of a historical mining liability into productive public lands that will celebrate the legacy of mining within the City of Timmins. The fourth section concentrates on the lessons learned as part of the development of this best practice approach and plan.

Background

The Hollinger Gold Mine was one of the three original major gold mines developed in Timmins and operated from 1910 to 1968.[1] It yielded 19.5 million ounces of gold which would be equivalent to $2.34 billion at today's gold price of $1,200.[2] The underground workings at the site were developed to a depth of 1,662 metres (5,450 feet) and included almost 600 kilometres (373 miles) of shafts and tunnels.[3] The City of Timmins developed as a result of mining settlements being built to house the influx of people needed to work in the mines, including the Hollinger. Timmins is now one of the largest cities in Northern Ontario with a population of approximately 42,435 as per the 2011 Census and functions as the regional service centre for Northeastern Ontario.

The City of Timmins was built up around the mine with the subsequent central business district being less than one kilometre from the site. A number of commercial enterprises bordering the site also developed with many still located on the boundary of the mine lands. With the closure of the Hollinger mine, the land surface was left in an unproductive and dangerous state with a variety of near surface mine related hazards that plagued the property. Over time a number of subsidence occurrences, referred to as "sink holes" occurred, some of which swallowed parking lots, buildings and even City buses. As a result, fencing was erected around the perimeter of the property to keep the public from entering the site as an interim measure. It was clear, however, that a longer term solution was required for the site and the City of Timmins.

Economic Conditions

In 2007, Goldcorp Canada Ltd., the owner of the former Hollinger Mine property, began to study the feasibility of redeveloping 101 hectares (250 acres) of the site into an open pit gold mine. This was due in part to the site historically being economically unviable due to the lower price of gold. With the increase in the commodity over the past ten years, Goldcorp determined, through a series of studies, that the site could now be developed as a viable open pit gold mine. With Goldcorp's Dome Mine underground operations located within the City of Timmins, approximately five kilometres from the Hollinger Property, the investment needed to bring the Hollinger Open Pit gold mine to fruition would not be on the scale if the development was to be located in an area that was to be considered "remote".

The Hollinger Mine site is located within the urban area of the City of Timmins. As such, much of the necessary infrastructure to develop this mine is in place. An extensive road network is in place along with the electrical supply of power needed for this project. Facilities such as the mine dry and mill are already in existence at the existing Dome Mine owned and operated by Goldcorp, located approximately five kilometres from the Hollinger Property. Ore from the Hollinger Mine is to be hauled and processed at the existing Dome Mine Mill. The tailings and ore processing will be discharged to existing Dome Mine tailing ponds. The project will also include the construction of an Environmental Control Berm around the Hollinger Project Site. The purpose of this berm will be to manage noise and other negative externalities the project may produce, which may, in turn, cause issues on the surrounding urban area. Employees of the Hollinger Mine will not be housed on-site as there are a number of private residential accommodations available within the City of Timmins to rent or purchase. No new company housing is proposed as part of this project.

The Hollinger Gold Mine will create over 200 direct jobs and 400 indirect jobs to support the operation of the mine. Through considerable capital investment and operating costs over the life of the project, the economic benefits and value of the project are significant to the City of Timmins. From an environmental and social benefit perspective, post mining would result in an elimination of hazard lands with a rehabilitated parcel of land that will be 100 hectares in size. The parcel will be returned to the City of Timmins in the form of a lake, parkland and trail system available for the public to use.

In a recent study (commissioned by the Ontario Mining Association with assistance from the Ontario Ministry of Northern Development and Mines) entitled *An An-thentic Opportunity: The Economic Impacts of a New Gold Mine in Ontario*, October 2014, Dungan and Murphy of the University of Toronto, outline the benefits of opening both a new open pit gold mine and a new underground gold mine in a remote location of Northern Ontario. The study did not define the definition of "remote" but for this paper it is considered to be an area not close to an urban centre. Key findings of the study include the following.

The construction of a new open pit gold mine, located in a remote area of the province would take approximately three years to construct with a total capital investment of $750 million, with $300 million of that spent on construction of a new mill and $80 million connecting to existing infrastructure.[4] The study also stated that 996 jobs would be directly created by the construction of an open pit gold mine, including the infrastructure to service it, construction of the mill and site development.[5] Another 440 jobs would be created with regards to production, including the milling of the mine ore.[6]

The scope of this study did not account for developing an open pit gold mine in an urban area of the province. If a comparison was to be made with the Hollinger Project, the following may very well have been reported. The total capital investment over a three-year period could be assumed to be $370 million, taking into account the existing mill and infrastructure to service the mine. With regards to jobs, only 200 direct and 400 sustained indirect jobs in support areas are to be created. These figures do not compare with the above-noted study but still represent a substantial investment by Goldcorp to bring this open pit mine to fruition. As such, a recent report by the Conference Board of Canada (2015) reported that the City of Timmins' economy is forecasted to grow by 2.4% in 2015 and another 2.3% in 2016[7]. This is in part due to the redevelopment of the Hollinger Open Pit Mine. The value of this project to the economy cannot be understated.

Economic Development and Planning Approach

In order to move forward with the redevelopment of this project and being production, a number of approvals and studies were required. The following section outlines the economic development and planning approach taken.

Resource development, including mining activities, typically occur well outside of settlement boundary areas in the rural areas of the municipality. In this case, the open pit mine is located within the urban area of the City, less than one kilometre from the central business district, residential areas and a major commercial corridor. The key challenge for municipal staff was how to develop and implement a process that would effectively permit the re-opening of a historic mining property in close proximity to sensitive land uses in an urbanized area of the community. The mine would not only create jobs, eventually clean-up a hazardous site and bring it back into a productive land use but would also have a range of potential negative externalities on the surrounding built-up area during its operations.

At the same time as Goldcorp began their pre-feasibility review of this former mining site, the City of Timmins was in the process of developing a new Official Plan (OP). Municipalities need to address mine hazards as per the *Provincial Policy Statement* (PPS) in their Official Plan. Consultations were held with the Ministry of Municipal Affairs and Housing (MMAH) as part of the Timmins OP development process. It was clear that a unique policy approach was required in order to address Goldcorp's project. A "Goldfield Area" official plan designation with related policies was derived and references back to the gold mines in Northern Ontario that helped establish many resource development communities in the Province, including Timmins.

As part of this process, consultations were also held with Goldcorp as they continued their open pit mining study. The Ministry concurred and the designation "Goldfield Area" was introduced into the new City of Timmins Official Plan, approved in 2010 by MMAH.

The intent of the "Goldfield Area" is to recognize the potential for a renewed mineral mining operation and the longer term closure and permanent rehabilitation of the subject lands in accordance with the *Mining Act* and broader municipal requirements. Given the proximity of the "Goldfield Area" to the urban centre, there was a strong need to ensure that land use activities are compatible with nearby sensitive land uses, closure and rehabilitation is properly undertaken and consideration is given to subsequent land uses within the designation. The Official Plan also states that mining operations in this designation are subject to a development agreement with the City.

In 2011, the City of Timmins approved a new Zoning By-law which rezoned a portion of the "Goldfield Area" as Mining (EA-IM) and recognized Goldcorp's proposed new open pit mine as a permitted use. In addition to the requirements set out by both the Federal and Provincial Governments in permitting the mine, Goldcorp needed to meet the requirements of the development agreement including: the completion of a best management plan to outline how they intend to manage air, noise, vibration, fly-rock and other nuisances related to mining activity as well as a complaint resolution protocol; phasing of the mine development; buffering and berm placement; fencing; garbage removal; ingress and egress from the site; mine rock stockpiles management; monitoring instruments; rehabilitation; and the completion of a subsequent land use plan. A letter of credit in the amount of $10 million was also required to be submitted to the City, as security, to help ensure that the works required as part of the subsequent land use plan are to be completed.

A key component to bring this project to fruition was the extensive public consultation and community engagement that was undertaken. Goldcorp undertook more than thirty supportive studies and required twelve governmental approvals to move forward with the project. The City of Timmins required the completion of third party reviews of some of the studies to ensure that the recommended mitigative measures were appropriate in light of nearby sensitive land uses. The City and Goldcorp also held more than twenty-one workshops, presentations and key stakeholder meetings (including aboriginal groups). The focus was to address the potential impact of negative externalities of the mine, such as dust, noise and vibration as outlined in the best management plan, as well as introduce the complaint resolution process developed for the project and the subsequent land use plan for the property once mining ceases. Surveys were also used as a way to garner feedback. The result was input from over 500 community members.

Goldcorp also hired a Community Liaison Coordinator to aid in the public consultation process and presently maintains this position. A website dedicated to the project and information centre were also established where the public could view the status of the work. In 2010, the Hollinger Project Community Advisory Committee was established with a mandate to be a liaison between Goldcorp and community members with regards to vetting any concerns, recommendations and future land uses associated with the project. Goldcorp further developed an on-line complaint form and internal resolution process to deal with public comments and concerns. There is also a web enabled live monitoring program instituted for noise, vibration and dust and is available during the life of the project for public review. In addition the company provides quarterly updates to the City of Timmins Council on project status and all complaints received.

In February 2014, the Hollinger Project officially began with construction of the berm which will act as a buffer between the reclamation activity occurring in the former mining areas and the commercial and residential areas. This feature will be progressively rehabilitated and once completed it will be seeded and greened. Based on a wide range of suggestions from the municipality and the public, a vision for the final land use plan was developed. A new trail system will be provided within two years of the berm construction to connect to the City's extensive trail network. The new trail system will follow the landscape of the berm, providing opportunities for residents to walk, jog and cycle as well as providing rest areas, with seating and picnic facilities. In selected areas, the trail will also be paved and lit for evening use. Storyboards will be located along the trail providing information on points of interest and history of the Hollinger Mine.

Upon closure of the mine, the open pit area will be eventually flooded and a portion of the berm will be sloped down to meet the water and allow for safe access to the waterfront. A sand covered beach will be developed. The area adjacent to the beach will become a large park area enhanced with an urban forest, shrubs, hedges and other landscape features including picnic tables and benches. On top of the berm, a public viewing area will be established providing a lookout area of the lake and the City of Timmins. Paved public parking will also be provided.

Lessons Learned

This case study from the City of Timmins has lessons that resonate across much of Northern Ontario wherever mining activity can be found. Cities, towns and villages that rely on this important economic activity need to positively address some of the related issues. This unique project could not have proceeded with the application of traditional economic development and planning approaches. The rejuvenation of a historic mining property within a highly urbanized area of the City has demanded new and innovative approaches to effectively balance economic needs with that of environmental, economic and social expectations.

These can be considered as a best management approach to follow for other similar “urban” mining projects, whether it is an open pit mine or an underground mine. This best management approach is made up of four key elements that have contributed to the success of this project and include pre-consultation, developing an innovative economic development and planning framework, ongoing public consultation and engagement, and importance of visions.

1. In terms of pre-consultation, early consultation with approval authorities, municipal staff, Goldcorp and the public was key in developing an acceptable approach to allow this complex development to occur while effectively balancing competing public interests.

2. The development of an innovative economic development and planning framework helped to demonstrate the important role that municipal staff, including land use planners and economic development practitioners, can play in order to guide a development through an approvals process that could not be achieved through traditional approaches.

3. Ongoing public consultation and engagement at the onset is significant. The importance of early, innovative and ongoing public involvement and engagement in the success of this project cannot be understated and has no doubt resulted in improved public acceptance and support. The consultation process could best be described as a partnership approach between the mining company, approval authorities and the public. The true test of this is the minimal number of complaints received throughout this process.

4. The importance of having a vision with regards to the project. This is perhaps the most significant element of this project from a public perspective. It includes the end vision for the site following the eventual mine closure. During the planning process, stakeholders offered a wide range of elements that they would like to see included in the final land use plan. These were considered and evaluated and many have found their way into the plan. The development of a subsequent land use plan based on accepted and realistic elements for the site and enhanced with progressive rehabilitation, certainly helps to promote trust and buy-in from the community for these types of developments. Continued open and transparent dialogue with the

community during the operational phase of the project remains the focus for the City of Timmins, Goldcorp and the community to ensure that appropriate elements are included in the final land use plan.

In conclusion, the Hollinger Open Pit Mine project will add to the economy of the City of Timmins by offering investment and employment over an eight to ten year life span of the project. It will also leave a portion of the City of Timmins, which was left with numerous mine hazards that were public safety liabilities, with restored lands in the forum of an urban park for all citizens to enjoy. Of utmost importance is the best management process that has been developed as part of this project and that can be utilized worldwide in developing similar projects. It is anticipated that this process will be called upon in similar, future projects.

End Notes

[1] Planning Alliance. 2008. Hollinger Baseline Studies Socio-economic Report. http://www.porcupinegoldmines.ca/en/ouroperations/resources/AppendixJ.pdf, p. 5.

[2] Goldcorp Porcupine Gold Mines. 2015. Hollinger Project. https://www.porcupinegoldmines.ca/en/ouroperations/hollinger.asp.

[3]Ibid.

[4]Dungan, P., & Murphy, S. (2014). An Au-thentic Opportunity: The Economic Impacts of a New Gold Mine in Ontario. Toronto, Ontario: Univesity of Toronto, Rotman School of Management, p.7.

[5] Ibid. p. 8.

[6] Ibid. p. 13.

[7] The Conference Board of Canada. 2015. Economic Insights Into Seven Canadian Mid-sized Cities. Canada, p.14.

Literature Cited

AMEC Earth & Environmental. 2010. Goldcorp Canada Ltd. Hollinger Mine – Mine Closure Plan Amendment. http://www.porcupinegoldmines.ca/en/ouroperations/resources/HollingerClosurePlanMay2011FinalA.pdf.

AMEC Earth & Environmental. 2010. Goldcorp Canada Ltd. Hollinger Mine – Pre-feasibility Environmental Baseline Studies: Surface Water Hydrology. http://www.porcupinegoldmines.ca/en/ouroperations/resources/AppendixE.pdf.

City of Timmins. 2010. City of Timmins Official Plan. http://www.timmins.ca/residents/building-planning/planning-responsibilities/official-plan.

City of Timmins. 2011. City of Timmins Zoning By-law. http://www.timmins.ca/residents/building-planning/planning-responsibilities/zoning-by-law.

City of Timmins. 2015. Our Heritage. http://www.timmins.ca/visitors/explore-timmins/our-heritage.

City of Timmins. 2015. Our History. http://www.timmins.ca/visitors/explore-timmins/our-history.

Dungan, P., & Murphy, S. 2014. An Au-thentic Opportunity: The Economic Impacts of a New Gold Mine in Ontario. Toronto, Ontario: Univesity of Toronto, Rotman School of Management.

Goldcorp Porcupine Gold Mines. 2012. Hollinger Project Best Management Plan. https://www.porcupinegoldmines.ca/en/ouroperations/resources/HollingerBestManagementPlan.pdf.

Goldcorp Porcupine Gold Mines. 2015. Hollinger Project. https://www.porcupinegoldmines.ca/en/ouroperations/hollinger.asp.

Government of Ontario. 2014. Provincial Policy Statement. http://www.mah.gov.on.ca/AssetFactory.aspx?did=10463.

Mining Act 1990 (Ontario) (Canada). http://guides.library.vu.edu.au/content.php?pid=270421&sid=2230821.

PR Associates. 2012. Goldcorp Canada Ltd. Hollinger Project Perception Audit. http://www.porcupinegoldmines.ca/en/resourcesGeneral/HollingerProjectPerceptionAudit.pdf.

Planning Alliance. 2008. Hollinger Baseline Studies Socio-economic Report. http://www.porcupinegoldmines.ca/en/ouroperations/resources/AppendixJ.pdf.

Statistics Canada. 2015. NHS Focus on Geography Series – Timmins. http://www12.statcan.gc.ca/nhs-enm/2011/as-sa/fogs-spg/Pages/FOG.cfm?lang=E&level=4&GeoCode=3556027.

The Conference Board of Canada. 2015. Economic Insights Into Seven Canadian Mid-sized Cities. Canada.

Timmins Economic Development Corporation. 2015. City of Timmins 2011 Economic Report. http://www.timminsedc.com/en/resources/2011%20Economic%20Forecast.pdf.

Timmins Economic Development Corporation. 2015. Community Profile. http://www.timminsedc.com/en/businessclimate/resources/TimminsCommunityProfileEN-May2015.pdf.

Timmins Economic Development Corporation. 2015. Current State of the Economy. http://www.timminsedc.com/en/businessclimate/stateofeconomy.asp.

Timmins Economic Development Corporation. 2015. Mining. http://www.timminsedc.com/en/businessclimate/mining.asp.

PYROLYSIS WITH PURPOSE: SURVEYING RELATIVE BIOCHAR-TAILINGS COMPATIBILITY THROUGH NATIVE SEED GERMINATION TRIALS.

Jasmine M. Williams[1] and Sean C. Thomas[1]
Faculty of Forestry, University of Toronto, Toronto, ON, M5S 3B3

Key Words: biochar, mining, heavy metals, acid mine drainage, tailings, germination, reforestation.

CONTEXT:

Introduction

Canada's diverse geological landscape is foundational to the country's long-standing global leadership in mining. At present, more than 60 minerals and metals are produced from domestic mine sites across Canada, representing a considerable contribution to the national economic portfolio (MAC, 2014). As a result of historical unregulated mining activities, contaminated inactive and abandoned mine sites generate both localized and widespread soil and water quality degradation (NRCAN, 2014). Orphaned mine sites fall under the jurisdiction of cash-strapped government ministries, who oversee an often slow, inadequate and impermanent remediation process (NOAMI, 2014). In recent years, the Canadian mining industry has been exposed to increasingly rigid environmental accountability. All the while, current mining and milling practices remain largely unchanged (NRCAN, 2014). Among the industry's direct operational products, tailings are generally considered the most difficult to manage responsibly and permanently (Bussière, 2007).

Following orebody mining, valuable mineral is separated from the gangue material through a milling process that involves initial size reduction (crushing, grinding) and subsequent chemical separation (floatation, hydrometallurgy) (Bussière, 2007). Mine tailings -the non-metal material produced during processing- is valueless and requires disposal. Precious metal mines are often profitable at very low cut-off grades and generate significant quantities of tailings while isolating very small fractions of valuable metals. These large volumes of tailings are pumped as slurry into a settlement area where they pose long-lasting physical stability risks. Moreover, once deposited into the impoundment area, tailings become geochemically volatile. Sulphides, inherent in most hard rock mine tailings, combine with oxygen in the atmosphere and form acid, which in turn solubilizes remnant trace metals. Heavy metal ions have been reported to infiltrate surrounding water networks and disperse extensively within aquatic and terrestrial ecosystems (Beesley et al., 2011).

Current practices in tailings management

To date, considerable research into environmental mine reclamation has focused on limiting the availability of either oxygen, water, or sulphide minerals within a tailings impoundment in an effort to prevent ionization of metals. Currently, the use of thickened tailings technology and engineered barrier systems are most common in both research and industrial practice (Cabral et al., 2000). Thickened tailings – or densified tailings – are emerging in hard rock mining due to their high physical stability and deposition predictability. Also, densified tailings generally raise the proximate water table and, thus, reduce oxygen flow and acidification within an impoundment (Bussière, 2007). Yet, given that some of the most toxic tailings contaminants, such as arsenic, are soluble at near neutral pH levels, oxygen limitation alone will not effectively mitigate heavy metal contamination. Moreover, lasting vegetation and forestation on thickened tailings impoundments- often a mandatory component within mine closure plans – is difficult if not impossible to achieve (Beesley et al., 2011).

Biochar as a tailings amendment

The introduction of select low-cost biochars at active and legacy mine sites could alter the reclamation scenario dramatically. Biochar is the term given to any pyrolyzed biomass material that is classified for use as a soil additive (Beesley et al., 2011). Biochar has been shown to sorb mine contaminants (heavy metals), encourage substrate water and nutrient retention, and promote fertility; thus could ultimately render reclamation more efficient (Hunt et al., 2010). Furthermore, development of a commercial biochar industry in mining communities would introduce diversity within a volatile, resource-based economic setting. The experimental addition of char onto soils has demonstrated increased crop yields and plant biomass, as well as lasting stress adaptation potential (Keske and Lohman, 2011). Recent findings suggest that the inputs to and conditions of pyrolysis influence a biochar's physical and chemical properties and its ultimate performance as an amendment (Hunt et al., 2010). Less understood, however, is the extent to which biochar behavior varies in different substrate environments, including diverse mining terrains.

Biochar performance determinants

The long-term nature of biochar's heavy metal sorption in addition to its proposed improvement to litter decomposition rates, pH, and nutrient availability in soils, have motivated research into the properties of chars and general char design (Thomas, 2013). Chen et al., Beesley et al., Gell et al., amongst others, have confirmed the positive relationship between pyrolysis temperature and select characteristics of biochar, namely adsorption surface area and carbon content (Beesley et al., 2011)(Chen et al., 2008) (Gell et al., 2011). Limited research has investigated the potential mutable role of biochar in different soil substrates. Laboratory experiments conducted by Yang et al. and Zhang et al. propose that tailored/pyrolysis-defined sorption features of a biochar amendment could react variably in soils with different native organic matter (Yang et al., 2003)(Zhang et al., 2010).

Within the context of land reclamation, arsenic ionization is especially toxic to plants and animals. Arsenic is also unique in its tendency to mobilize more readily within an increasingly alkaline substrate environment (Beesley et al., 2011). In contrast to most other metals, arsenic ions exist in an oxy-anion solution (negative charge) and will, thus, tend to bind to specific heavy metal oxides in soils (or in biochar, for that matter). Amendment through a mixture of biochar and iron oxide should promote arsenic retention. This particularity further highlights the variable amendment role that biochar takes within diverse substrates.

Grass and tree seed germination

The presence of a vegetative layer over soils helps to isolate water-born contaminants and reduce their overall mobility. Re-vegetation and re-forestation atop tailings is prioritized for motivations beyond solely ecosystem regeneration and aesthetics. Indeed, the reintroduction of trees, grasses and shrubs on tailings material (or any substrate) will also assist in stabilizing both physical and geochemical parameters of concern (Ranjan et al., 2015). A rapid rate of forestation on tailings would also encourage greenhouse gas offsets as operations continue or as reclamation work progresses. Biochar has been shown to increase soil water holding capacity and liming effects, and hence encourages vegetation success (Thomas, 2013). Nevertheless, limited research attention has been applied towards matching specific biochars and mine tailings to optimize restoration and regrowth.

RESEARCH QUESTION:

Is a biochar's influence on soil improvement and plant growth substrate-specific? Can the feedstock source and pyrolysis method of 'designed' lignocellulosic biochar be used to predict its ultimate amendment capacity on a particular mine tailing?

HYPOTHESES:

Under controlled conditions, the rate and magnitude of seed germination are expected to depend on the tailing-biochar composition of a substrate, suggesting that char may be customized to encourage optimal growth on a particular tailing. Furthermore, a slightly acidic char (pH = 6.54) is expected to decrease arsenic mobility most when added to an iron-rich tailing and could induce greatest seed fertility in this tailing subclass so long as overall substrate pH remains conducive for incubation. Germination rates and leaf emergence are projected to depend on the species of seed observed, as well, irrespective of the substrate.

METHODOLOGY:

The hypotheses were tested by way of both seed germination and plant experiments performed under the supervision of Dr. Sean C. Thomas, Ph.D., Canada (Tier 1) Research Chair. A series of eight-day germination experiments were conducted in a rooftop glasshouse at the University of Toronto. Three 'designed' biochars with unique feedstock and pyrolysis features and one natural 'high-ash' char sourced from a minesite forest fire were introduced into a variety of tailing-seed settings. The growth media were four distinctive unmodified gold mine tailings from operating sites in Northern Ontario, as well as one granitic sand from Haliburton, Ontario. The tailings selected for experimental application have been sourced exclusively from operational open-pit gold mines within Ontario's Abitibi Greenstone Belt mineral formation. All tailings exhibit various extents of acidification due to inherent sulphide oxidation (St. Andrew Goldfields, 2016). The pH levels and heavy metal effluent concentrations are within legal limits at all tailings source sites, however with no advanced geochemical tailings management in practice, the stability of these levels is unlikely. The four tailings have been produced from similar mineral extraction processes whereby sodium cyanide first leaches gold off ore particles and then activated carbon is used to absorb the gold from solution. All tailings applied contain varying amounts of Cu, Co, and Fe, whereas current mine technicians only identify two of the samples as also carrying trace amounts of arsenic.

In miniature field replicates of 50 cm^3 (n = 500), each biochar has been blended into individual samples of mine tailings or granitic sand at a common dosage of 50 t/ha, as employed in forestry and no-till agriculture (Thomas et al, 2013). Dry bulk density of each tailing was measured to accurately convert the application dose into a substrate-specific percent mass of biochar. Prior to sample preparation, the biochars were be pre-treated by washing with de-ionized water for a 24hr. period on a shaker table at 50 RPM, then subsequently dried for 24hrs. in a batch oven at 60°C, as displayed in Figure 1. Motivations for pre-washing chars stem from the highly porous and absorptive properties of biochar and the release of particles throughout pyrolysis. Any initial biochar pore saturation could influence amendment, overall substrate composition, and the char's ultimate sorption potential. The four tailings applied were fully dried in a batch oven at 60°C and then passed through a coarse screen, followed by standard mechanical analysis of tailings' particle size distribution in five size classes. Both biochar and tailings pH and Electrical Conductivity values were be measured and recorded prior to sample preparation, as well. Moreover, tailings samples was examined using Xray fluorescence technology to determine elemental status of Cu, Fe, Co, and As. Finally, the measured tailings data was cross-referenced to available data collected by each mining company in situ.

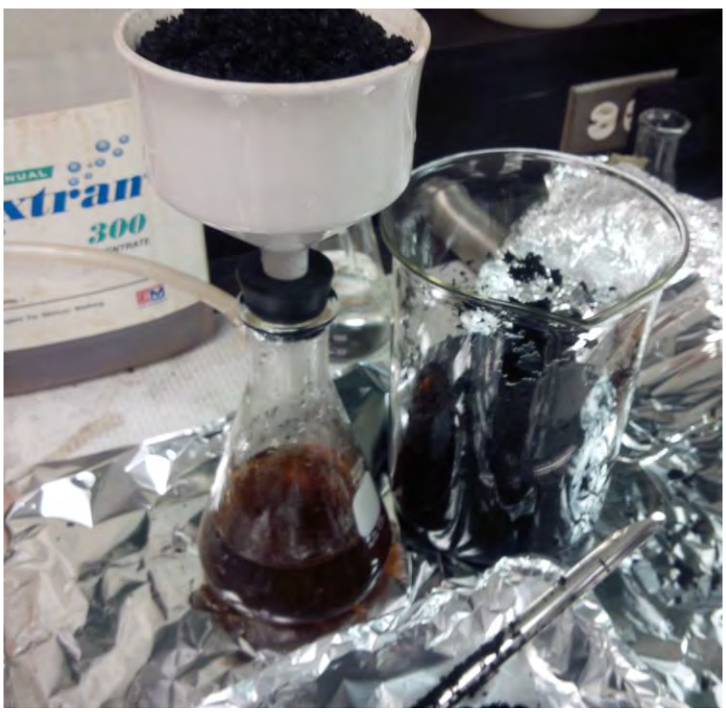


Figure 1: Pre-treatment (draining) of biochar.

Seeds of *Andropogon gerardii*, *Poa palustris*, *Picea mariana*, and *Pinus banksiana* were applied to the media (25 seeds/sample). Both grass and tree seeds were sourced from Natural Resources Canada's (NRCAN) National Seed Centre nurseries in close proximity to the pertinent mine sites, and located within the Boreal Forest Zone. The mine sites fall within the ‘3a’ Plant Hardiness Zone with mean summer daytime temperatures of 23°C and a maximum recent summer temperature near 35°C (Weather2, 2016). Systematic monitoring of % germination on 48 hour intervals as well as cumulative leaf emergence on day 8 and 10, is considered most crucial for judging amendment. One observer performed all data collection over the 8 (grasses) and 10 (trees) day trials in order to eliminate subjective variability in judging seed germination. So as to discern the relationship amongst experiment variables, a multiple regression analysis will be performed. The covariance of properties will likely require an ordination in order to reduce dimensionality for specific considerations. A typical experimental sample of tailings, biochar, and seedlings is depicted in Figure 2.

Figure 2: Substrate sample of tailings-biochar on Day 6 of a germination observation.

PRELIMINARY RESULTS/IMPLICATIONS:

Percentage of total germinated grass and tree seeds was observed to vary across the biochar-tailings substrates, with most dramatic variability observed across the various biochar`s applied to Goldcorp’s Porcupine Mine upper delta tailings. Seed germination success on this aforementioned substrate has been grouped based on biochar type and displayed in Figure 3 below. The natural forest fire charcoal (“M”) produced the second highest germination in all species when applied to the Goldcorp upper delta biochar.

In forecasting possible seed reactions to a substrate combination of natural forest fire charcoal and mine tailings, typical post-fire forest stands should be considered. Indeed, decades of research confirms that revegetation (including reforestation) increases following fire events due in large part to the resulting charcoal deposition, which increases litter decomposition, soil pH, and nitrogen availability (Thomas, 2013).

Substrates with both tailings and natural high-ash forest fire char were, hence, expected to exhibit increased germination rates compared to substrates composed of tailings alone. The natural char's propensity to match pyrolyzed chars in terms of germination response suggests it may be a viable, readily-available amendment. Typically, forest-fire char is produced as a result of complete wood combustion under relatively high oxygen exposure. Such conditions result in an alkaline, nitrogen-depleted substance, which if considered exclusively, should diminish germination potential. In combination with various acidic, contaminated tailings, however, its behavior is less predictable. Moreover, the natural char has been applied to samples in its in situ form and was likely collected in combination with some organics from the underlying terrain. Although the composition of this amendment is not one of "pure" char, the material was left untreated so as to verify the efficiency of a locally sourced amendment option.

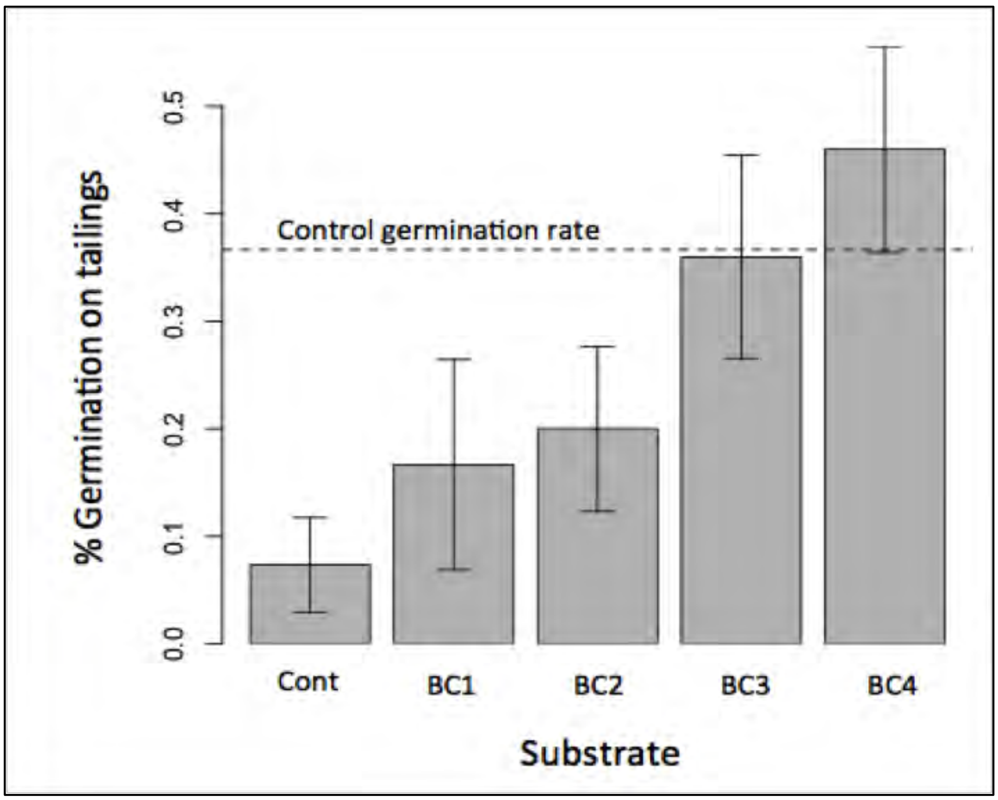


Figure 3: Goldcorp Porcupine Upper Delta Tailings: % Germination in Biochar Types.

Overall, further data analysis is to be conducted and germination rate and leaf emergence is predicted to vary among biochar-substrate groupings with some independent deviation between seed species. Of the biochars applied, that with highest pH and pyrolysis T–designed from mixed construction wood waste – was expected to prompt most rapid and robust germination in all species when combined with most tailings, yet was outperformed by the three other chars tested . Given the distinct toxicity and chemical behavior of arsenic, any biochar-specific germination trends will be of significant interest and likely the subject for further analysis. Since the mobility of arsenic ions is known to increase with increased soil pH, the most alkaline biochar may actually deter germination in this particular scenario. Any arsenic mobility is likely to introduce toxicity within the substrate water particles and, hence reduce seed germination.

CONCLUSIONS/SIGNIFICANCE:

Preliminary results would suggest that biochar might be tailored for use on a particular contaminated substrate, representing a value-added incentive for its application in the industrial context. Furthermore, if a natural mine site char increases seed germination rates in tailings, it may prove to be an internal, low-cost support to reforestation and reclamation. As the substrate-specific behavior of biochar becomes progressively more understood, development of a biochar treatment glossary for Canadian mine tailings could be realistic. Tailored biochar solutions would advance mine remediation and the biochar industry, and reduce the ultimate industrial footprint. Results of preliminary investigations should provide motivation for extended glasshouse experiments and eventual in-situ testing.

ACKNOWLEDGEMENTS

The authors would like to acknowledge the project support from the Ontario Mining Association, as well as the resources provided by Goldcorp Inc., St. Andrew Goldfields, and Haliburton Research Forest. Both authors receive independent research funding from the Natural Sciences and Engineering Research Council. Finally, the authors acknowledge the great opportunity provided by the conference host, the Canadian Land Reclamation Association.

LITERATURE CITED

Mining Association of Canada. 2014. MAC. http://mining.ca/resources/mining-facts

Natural Resources Canada. NRCAN. 2014. http://www.nrcan.gc.ca/mining

National Orphaned/Abandoned Mines Initiative. NOAMI. 2014. http://www.abandoned-mines.org/pdfs/PolicyFrameworkCanforMinClosureandMgmtLiabilities.pdf.

Bussière, B. 2007. Colloquium 2004 : Hydrogeotechnical properties of hard rock tailings from metal mines and emerging geoenvironmental disposal approaches. Canadian Geotechnical Journal. 44: 1019-1052.

Beesley, L., Morno-Jimenez, E., Gomez-Eyles, J.L, Harris, E., Robinson, B., and Sizmur, T. 2011. A review of biochars' potential role in the remediation, revegetation and restoration of contaminated soil. Environmental Pollution.159: 3269-3282.

Cabral, A., Tremblay, P., and Lefebvre, G. 2000. Determination of the Diffusion Coefficient of Oxygen for a Cover System Including a Pulp and Paper By-Product. Geotechnical Testing Journal. 27: 1-14.

Hunt, J., DuPonte, M., Sato, D., & Kawabata, A. 2010. The basics of biochar: a natural soil amendment. Soil and Crop Management. 30: 1-6.

Keske, C.M.H, and Lohman, G.G. 2011. Biochar: An emerging market solution for legacy mine reclamation and the environment. Appalachian Nat. Resources. Hein Online.

Thomas, S.C. 2013. Biochar and its potential in Canadian forestry." *Silviculture Magazine*, January 2013: 4-6.

Chen, B.L., Zhou, D., and Zhu, L. 2008. Transitional adsorption and partition of nonpolar and polar aromatic contaminants by biochars of pine needles with different pyrolytic temperatures. Environmental Science and Technology. (42): 5137-5143.

Gell, K., Van Groenigen, J.W., and Cayuela, M.L. 2011. Residues of bioenergy production chains as soil amendments: immediate and temporal phytotoxicity. Journal of Hazardous Materials. (186): 2017-2025.

Yang, Y. and Sheng, G., 2003. Pesticide absorptivity of aged particulate matter arising from crop residue burns. Journal of Agricultural and Food Chemistry. (51): 5047-5051.

Zhang, H., Lin. K., Wang, H., and Gan, J. 2010. Effect of Pinus radiate derived biochars on soil, sorption and desorption of phenanthrene. Environmental Pollution. (158): 2821-2825.

Ranjan, V., Sen, P., Kumar, D., and Sarsawat, A. 2015. A review on dump slope stabilization by revegetation with reference to indigenous plant. Ecological Processes. (4): 1-11.

St. Andrew Goldfields Ltd. 2016. http://www.sasgoldmines.com/s/History.asp.

Thomas, S.C., Frye, S., Gale, N., Garmon, M., Launchbury, R., Machado, N., Melamed, S., Murray, J., Petroff, A., and Winsborough, C. 2013. Biochar mitigates negative effects of salt additions on two herbaceous plant species. Journal of Environmental Management. (129): 62-68.

Weather2. 2016. http://www.myweather2.com/City-Town/Canada/Ontario/Timmins/climate-profile.aspx.

Biodiversity Offsetting: Opportunities and Challenges

Wynia, Michael J. & Landon W.N. Black

Skelton Brumwell & Associates, Barrie, ON, L4M 5G1

Key Words: policy, aggregate rehabilitation, agricultural afforestation, net gain

Aggregate resources are often located in, or near, areas that have environmental features. The potential conflict between aggregate resources and environmental values is a challenge that many applications inevitably face, particularly as legislation and policy continue to evolve.

In many cases there are opportunities to plan and manage extraction using innovative and creative techniques such as biodiversity offsetting to mitigate potential environmental impacts. Biodiversity offsetting has become a popular topic in the planning and conservation community and there are precedents that prove that this technique can work effectively to achieve results for both the aggregate industry and for environmental protection.

Offsetting, or compensation, may occur under an environmental assessment process as well as through the municipal land use planning process.

This presentation focuses initially on the opportunities and challenges in the context of the land use planning process in Ontario and then focuses on the practical opportunities and challenges as experienced in an aggregate extraction project in the Township of Clearview in the County of Simcoe.

The municipal planning framework for biodiversity offsetting is not explicitly set out in guiding legislation and policy other than for very species matters governed by the federal and provincial government (fisheries and threatened and endangered species). Currently, the opportunity for offsetting arises from the policy language of the Provincial Policy Statement 2014. This may change as the Province continues policy development. For example, Ontario is considering specific legislation and policy which may establish a No-Net-Loss approach to the management of remaining wetlands.

Due to a prohibition on any development within provincially significant wetlands, federal jurisdiction over fisheries and provincial jurisdiction in regard to endangered and threatened species, a municipal planner is limited to consideration of offsetting for a specific set of natural heritage features:

- Components of a municipally identified natural heritage system
- significant wetlands in the Canadian Shield north of Ecoregions 5E, 6E and 7E
- Significant woodlands in Ecoregions 6E and 7E (excluding islands in Lake Huron and the St. Marys River)
- Significant valleylands in Ecoregions 6E and 7E (excluding islands in Lake Huron and the St. Marys River)
- Significant wildlife habitat
- Significant areas of natural and scientific interest; and
- Coastal wetlands in Ecoregions 5E, 6E and 7E that are not subject to policy 2.1.4(b) (Ministry of Municipal Affairs and Housing, 2014)

However, provincial policy sets out the minimum requirements for natural heritage system planning. Municipalities may also elect to broaden those requirements and set out policies for the protection of regionally and locally important features that are not covered by Provincial Policy Statement 2014 (e.g. locally significant wetlands). In these instances, the municipality may also set out policy with regard to the utilization of offsetting in relation to the planning process protecting those local features.

The determination of whether or not to utilize offsetting as a tool is fundamentally impacted by such matters as:

- The policies, programs and interests of other agencies such as Conservation Authorities;
- The nature and sensitivity of the feature or function being impacted;
- The scale and nature of the potential impact proposed to be offset;
- Community interests and acceptance; and,
- The feasibility of offsetting or likelihood of success.

Despite the potential range of situations in which offsetting may be a potential tool, policy should set clear limits. There are situations where the feature or function being impacted is of such significance, or sensitivity, that it should not be subject to impact. There are other situations where the full extent of impacts cannot be mitigated or offset.

A precautionary principle approach suggests that offsetting should not be utilized if there is any significant doubt as to the suitability of offsetting.

Offsetting can be a very useful tool in the management of natural heritage systems. It can offer flexibility and responsiveness to changing conditions. It can provide for significant gains to the quantity and quality of the natural heritage system. However, it is also potentially problematic in regard to effective implementation and achieving the ultimate objectives and goals of natural heritage system protection.

There are some fundamental guiding principles, such as those set out in the Best Practices Guide to Natural Heritage Systems Planning that can assist a municipal planner in making the determination of whether or not to utilize offsetting. These include:

- Only consider offsetting if avoidance and mitigation have been fully explored, are not feasible, and their is a public interest which suggest that proceeding on the basis of offsetting is preferable over denying a development or site alteration approval.
- Due to the risks of the offsetting approach, there should be a compelling reason to utilize it.
- Strong policies guiding the acceptability of offsetting are essential, and the municipality, and other approval authorities, should ideally maintain control over the determination of acceptability of the technique.
- Offsetting should only be utilized on the basis of a significant net gain.

The determination of the suitability of an offsetting proposal should be based on a full understanding of:

- The features and functions which will be lost;
- Their significance in the natural heritage system in terms of quantity, quality and function;
- The proposed replacement features and functions;
- The potential significance of the proposed offsetting features and functions in terms of quantity, quality and function.

An in-depth understanding of what will be lost and what will be gained is essential to decision making.

The approval of a large quarry expansion located on the Niagara Escarpment in the Township of Clearview provides an example of the opportunity to effectively employ offsetting while at the same time managing risks.

This project was subject to intense scrutiny and involved a year long hearing as well as a judicial review during which the issue of the appropriateness and checks and balances of an offsetting proposal were fully explored.

While there were a number of environmental features potentially impacted by the proposal, one of the more significant features in a hardwood forest, part of which could be avoided in the quarry design, but some of which would be lost and could not be subsequently rehabilitated since extraction below the water table was being proposed.

In this case municipal policy specifically permitted consideration of offsetting, or compensation, but only on the basis of a net gain approach.

The loss of portions of a woodland for an aggregate project was offset through the creation of a much larger contiguous woodland area. This included a planned approach to filling gaps in the contiguous woodland to reduce edges and increase interior forest habitat as well as the establishment of enhanced woodland connections or corridors.

The municipality was also provided with a ½ cent per tonne of aggregate extracted from the full quarry site to facilitate tree planting throughout the township as well as another ½ cent per tonne to fund an environmental land acquisition project.

To minimize the risk the municipality utilized performance standards and triggers.

The woodlands to be removed were incorporated into a latter phase of extraction. Prior to actual removal, however, the operator was required to demonstrate that offsetting forest creation efforts were succeeding through meeting specific performance measures regarding tree growth, species diversification and other considerations.

This did not entirely eliminate the risk, but it significantly minimized it.

It was not possible to delay removal until a mature woodland was in place, however phasing did allow for a sufficient period of time to allow a reasonable level of confidence that offsetting works had a high probability of success. This approach also ensured that the planting program would be fully completed and monitoring and maintenance were well under way prior to any woodland loss.

This technique does have limitations and the municipality also therefore approved an adaptive management plans approach which allowed for adjustments to ensure that offsetting objectives were met by:

- Adjusting the offsetting proposal to respond to changes in conditions; and/or
- Adjusting the project to avoid or mitigate impacts where offsetting results are not being achieved.

The aggregate proposal is now fully approved and extraction has commenced. The approval of the project also initiated the offsetting process including the creation of the woodland features. Since extraction within the woodland required that the woodland offsetting work be demonstrated to be effective, there is a significant effort under way to ensure that the offsetting project is successful and will meet the requirements stipulated to allow removal of the existing woodland area. This effort has demonstrated some of its own challenges and opportunities.

The targets that were established for the offset forest pertained specifically to the canopy. Prior to removal of the existing forest the offset forest was required to meet 95% canopy closure at a height of 10m with no more than 15% mortality. Due to the phasing of extraction these targets have a relative timeframe of 10-15 years. In addition, some of the species, included to enhance forest diversity, tend towards slow growth rates.

The limited timeframe to achieve substantial growth meant that typical aggregate rehabilitation models were insufficient. The installation of large quantities of large wire basket or balled and burlapped material was cost prohibitive and difficult to source. This led to the adoption of standard nursery production techniques to expedite growth. This included: vegetative suppression to reduce competition, fertilization, stimulation pruning, irrigation and insect control.

Plant selections included 25mm caliper whips, 40-60cm seedlings and 20-40cm coniferous seedlings. The disparity in sizes was partially to control costs and partially to facilitate different success rates of species regarding rooting habits and structure. Planting was conducted using a three-point hitch shoe plow pulled along straight, 2.5m offset rows. This technique was adopted to take advantage of narrow planting windows and because the area to be planted were agricultural fields. The result was a structured, nursery plantation along one axis to help in growth stimulation and a randomised appearance on the perpendicular axis due to modulation of planting spacing.

The ultimate success of these technique cannot be determined this early in the process, but growth rates in local nurseries are such that material is unsaleable after 10 years due to its excessive size. Some implications are observable at this point nonetheless.

1) Thorough understanding of transplant methods needs consideration during the design phase to account for varying success rates between methods.
2) Expediting growth of any species in an exposed, harsh environment requires intensive management and innovative techniques for integration.
3) Plantation style (uni-dimensional conformity) is absolutely critical to managing the juvenile trees to ensure survival and spur growth.

4) Trees are living organisms and effective supply chain management is imperative to securing quality material. Material needs to be sympathetic to the local conditions and arrive in vigor as no management method is sufficient to expedite growth of sub-optimal material.

Figure 1: Field Planted in Nursery Rows

As land use activities continue to interact with the natural environment, offsetting and compensation will continue to be techniques employed to maintain or ideally enhance the natural environment. Offsetting presents opportunities both at the policy and approvals stage, as well as in the implementation of approved offsetting plans. Continued refinement of policy and techniques, and monitoring of results, will inform the process and ensure that we actually achieve the objective of net gain.

References:

Ontario, Canada. Ministry of Municipal Affairs and Housing (2014) *Provincial Policy Statement*. Kingston: Queens Printer for Ontario.

EVALUATING THE ABILITY OF *ALYSSUM MURALE* TO EXTRACT AGED NICKEL FROM NICKEL-ENRICHED ORGANIC SOILS

Kimberly Zupfer[1], Beverley Hale[1] and Michael Dutton[2]

[1] School of Environmental Sciences, University of Guelph, Guelph, ON, N1G 2W1
[2] Vale Ltd. Toronto, ON, M5J 2K2

Key Words: hyperaccumulator, perennial crop, sequential cropping, STELLA® Professional software, systems modeling, Ni mass transfer

1. Introduction

Port Colborne, Ontario, has an area of nearly 30km^2 that has been contaminated by emissions from a nickel (Ni) refinery that was in operation between 1918 and 1984. Emissions from the process of converting Ni-rich ores to marketable forms of Ni have left elevated Ni concentrations in the surrounding soil. These elevated Ni concentrations are causing phytotoxicity and are suspected of reducing crop yield in some agronomic species. Common remediation techniques (such as 'dig and dump') are not feasible or economically efficient because of the large area that has been contaminated. Phytoremediation would be a cost-effective and sustainable alternative, if its efficacy could be demonstrated on spatial and temporal scales.

Nickel hyperaccumulating plant species are able to accumulate at least 1000 mg kg^{-1} of Ni in their dry biomass without succumbing to toxicity. *Alyssum murale*, a hyperaccumulator of Ni that is native to Ni-rich serpentine soils from Mediterranean Europe, is a species of interest for phytoremediation. Extensive research has been conducted regarding the ability of *A. murale* to extract Ni from Ni-enriched soils (Chaney et al., 1998, 2000, 2005; Li et al., 2003; McNear et al., 2007; Fellet et al., 2009; Centofanti et al., 2012). Research shows that *A. murale* is able to extract Ni from soils with elevated concentrations of Ni without showing signs of toxicity; however the spatial and temporal capacity of this species as a perennial crop to measurably reduce the concentration of Ni in soils has not been demonstrated.

2. Materials and methods

To determine the capability of perennial cropping with *A. murale* as a phytoremediation technology, a mathematical model of soil → plant Ni mass transfer was created. The use of mathematical models to simulate environmental scenarios is efficient as different inputs and outputs can be evaluated without experimenting with the environment. Human rationality, which may promote an error and/or bias, is also eliminated (Canales-Pastrana et al., 2013).

STELLA® Professional, a simulation software used to express changes in a system over time, incorporates specific parameters and process rates to create an interactive output. In 2013, Canales-Pastrana et al. used STELLA® Professional to model a system that expressed the rate at which plants are able to volatilize soil mercury under given circumstances. The model created was able to use specific values to demonstrate the overall extraction capability of the plant. With the use of a STELLA® Professional, a graphical representation of the extraction capability of *A. murale* and a hypothetical timeline for reduction in soil Ni concentration was created. A soil → plant Ni mass transfer model was designed using various assumed values to determine the time required for sequential cropping of *A. murale* to remediate the contaminated soils to (1) an initial target level of 1000 mg kg^{-1} (Dan et al., 2008), and (2) the Ministry of the Environment and Climate Change (MOECC) component value for plants of 100 mg kg^{-1} (Ministry of the Environment, 2011). Variables used in the model include initial soil Ni concentration, an extraction rate, an initial *A. murale* Ni concentration value and a cropping factor that accounts for 100% of A. murale being cropped initially at 1 year, then at 1 year intervals.

3. Preliminary results and discussion

Various scenarios can be created with the use of STELLA® Professional. Preliminary models incorporate values from previous work (Li et al., 2003), and pot study data collected in 2015. Figure 1 shows a base STELLA® Professional model that is used to input values to create a graphical output. Using various assumed values for the rate control steps of STELLA® Professional, preliminary estimates of half-life of soil Ni range between 11 years and 250 years.

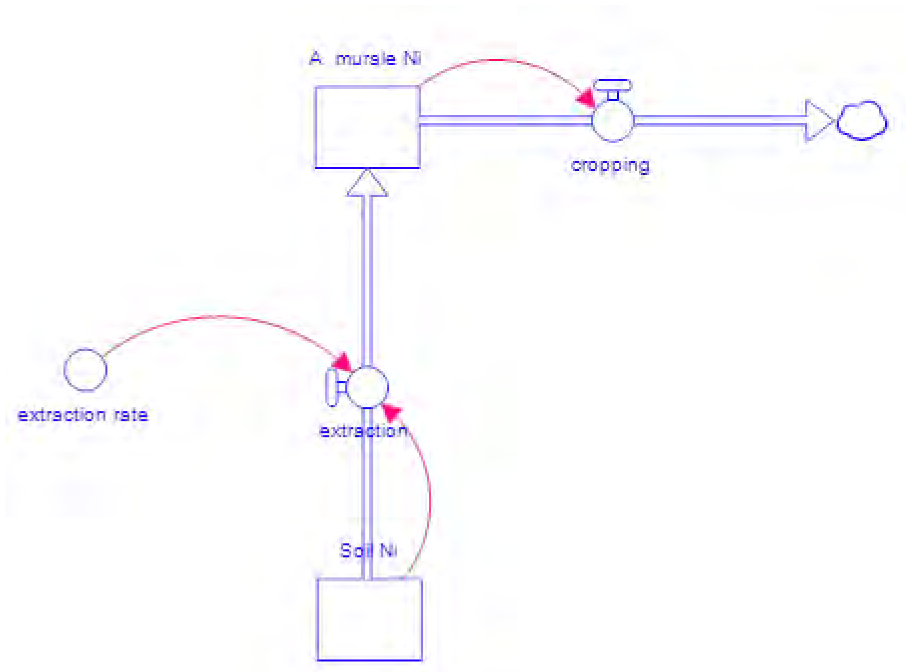


Figure 1: Preliminary soil→ plant mass transfer model created using STELLA® Professional

4. References

Dan, T., Hale, B., Johnson, D., Conard, B., Stiebel, B., & Veska, E. (2008). Toxicity thresholds for oat (Avena sativa L.) grown in Ni-impacted agricultural soils near Port Colborne, Ontario, Canada. *Canadian Journal of Soil Science*,*88*(3), 389-398.

Canales-Pastrana, R. R., & Paredes, M. (2013). Phytoremediation Dynamic Model as an Assessment Tool in the Environmental Management.

Centofanti, T., Siebecker, M. G., Chaney, R. L., Davis, A. P., & Sparks, D. L. (2012). Hyperaccumulation of nickel by Alyssum corsicum is related to solubility of Ni mineral species. *Plant and soil*, *359*(1-2), 71-83.

Chaney, R. L., Angle, J. S., Baker, A. J., & Li, Y. M. (1998). *U.S. Patent No. 5,711,784*. Washington, DC: U.S. Patent and Trademark Office.

Chaney, R. L., Li, Y. M., Brown, S. L., Homer, F. A., Malik, M., Angle, J. S., ... & Chin, M. (2000). Improving metal hyperaccumulator wild plants to develop commercial phytoextraction systems: approaches and progress.*Phytoremediation of contaminated soil and water*, 129-158.

Chaney, R. L., Angle, J. S., McIntosh, M. S., Reeves, R. D., Li, Y. M., Brewer, E. P., ... & Broadhurst, C. L. (2005). Using hyperaccumulator plants to phytoextract soil Ni and Cd. *Z Naturforsch C*, *60*(3-4), 190-198.

Fellet, G., Centofanti, T., Chaney, R. L., & Green, C. E. (2009). NiO (s)(bunsenite) is not available to Alyssum species. *Plant and soil*, *319*(1-2), 219-223.

Li, Y. M., Chaney, R. L., Brewer, E. P., Angle, J. S., & Nelkin, J. (2003). Phytoextraction of nickel and cobalt by hyperaccumulator Alyssum species grown on nickel-contaminated soils. *Environmental science & technology*,*37*(7), 1463-1468.

McNear, D. H., Chaney, R. L., & Sparks, D. L. (2007). The effects of soil type and chemical treatment on nickel speciation in refinery enriched soils: A multi-technique investigation. *Geochimica et Cosmochimica Acta*, *71*(9), 2190-2208.

Ministry of the Environment. (2011). Soil, Ground Water and Sediment Standards for Use under Part XV. 1 of the Environmental Protection Act.