PROCEEDINGS OF THE THIRTY-EIGHTH ANNUAL BRITISH COLUMBIA MINE RECLAMATION SYMPOSIUM

Organized and Published by

BCTRCR
TECHNICAL AND RESEARCH
COMMITTEE ON RECLAMATION

in Cooperation with:

British Columbia Ministry of Energy and Mines
British Columbia Ministry of Environment
Natural Resources Canada
Mining Association of British Columbia
Association for Mineral Exploration, British Columbia
Mining Association of Canada
University of British Columbia
Thompson Rivers University

Editors: **Dirk van Zyl**

Convened at the Prince George Civic Centre
Prince George, British Columbia
September 22 to 25, 2014

PROCEEDINGS OF THE ANNUAL BRITISH COLUMBIA
MINE RECLAMATION SYMPOSIUM

38th ANNUAL
Symposium

THE BRITISH COLUMBIA TECHNICAL AND
RESEARCH COMMITTEE ON RECLAMATION

VICTORIA
BRITISH COLUMBIA
CANADA
SEPTEMBER 2014

ISSN 1199-231X

TABLE OF CONTENTS

THE BRITISH COLUMBIA
TECHNICAL AND RESEARCH COMMITTEE ON RECLAMATION

The Technical and Research Committee on Reclamation (TRCR) originated and first became active in the early 1970s, in response to a demonstrated need in British Columbia mining for a greater government-industry communication in the area of environmental protection and reclamation. Membership is drawn from the corporate sector (several of the large mines are represented); the Ministry of Energy and Mines; the Ministry of Environment; Natural Resources Canada; the Mining Association of British Columbia; Association for Mineral Exploration, British Columbia and British Columbia universities and colleges. The Committee meets four or five times a year to discuss matters of joint concern and interest, exchange experience, plan activities and prioritize research needs.

Since 1977, the TRCR has annually sponsored the British Columbia Mine Reclamation Symposium to foster the exchange of information and ideas on reclamation. Proceedings, published concurrently with the symposia, are a valuable source for anyone interested in this field. Proceedings of the first 26 symposia are now available for purchase on a single CD.

The Acid Mine Drainage Task Force amalgamated with the TRCR to ensure that acid rock drainage issues continued to be fully addressed, and the TRCR now organizes, in partnership with MEND2, the annual Metal Leaching/Acid Rock Drainage workshop held in early December in Vancouver each year.

In addition to the annual symposium dealing with the entire spectrum of reclamation issues, the TRCR has also sponsored symposia and studies focusing on individual issues. For example, in 1985 the Committee sponsored a preliminary study of the practice of resloping waste dump faces, with support from the Canada-British Columbia Mineral Development Agreement. A second study, with support from the Coal Association of Canada, focused on materials handling and cost. In 1986, the Committee sponsored an International Rock Drain Symposium, which drew attendance from the United States, France, Australia, and the Soviet Union, again with support from the Canada-British Columbia Mineral Development Agreement.

In 1990, the Committee established a special project to improve the environmental management of cyanide, and in July 1992 published a “Technical Guide for the Environmental Management of Cyanide in Mining”. As a follow up, the TRCR, in cooperation with MEM, hosted the Cyanide Gold Heap Leach Workshop in Vancouver, May 27 and 28, 1995.

In 1992, a TRCR committee was struck to direct a study to identify environmental and reclamation issues from which research and development priorities could be identified. This initiative resulted in a report entitled “Key Environmental Issues Assessment”, released April 1993. The TRCR co-sponsored a workshop on molybdenum held in conjunction with the symposium in 1999. In 2000, because of increasing concerns of selenium in aquatic environments near many of our coal and metal mines, the TRCR devoted one session to focus on a full discussion of this issue.

To promote excellence in reclamation, the Committee annually presents the Jake McDonald British Columbia Mine Reclamation Award and several citations for excellence in mine reclamation. Citations are provided for the reclamation of exploration areas, metal mines, coal mines, placer operations, sand and gravel operations, and quarry operations.

TONY MILLIGAN BOOK AWARD

To all of us in government, universities, the mining industry and the consulting industry, Tony Milligan was the "grandfather" of mine reclamation in British Columbia. In the early 1970s, there was little experience in mine reclamation, especially in the sub-alpine areas of the Rockies. Tony and his crew at Kaiser Resources led the way in the practical application of reclamation technology, proving to us all that large-scale open-pit coal mining could be reclaimed to a very high standard, and that the large elk and deer populations of the region could coexist with mine development. This pioneering work was undoubtedly responsible for improving the credibility of the mining industry and allowing new mines to develop, pointing to Tony's achievements and successful reclamation. Tony and his crew won the Annual Mine Reclamation Award four of the first seven years it was presented.

Tony was well respected by his colleagues in government, as well as by the mining industry. There are few individuals who, by their consistent honesty, integrity and positive outlook can easily move from industry to government, use the same approach and philosophy to strive for excellent mine reclamation, and be as successful as Tony.

The Tony Milligan Book Award was created in recognition of the many contributions Tony made to mine reclamation in British Columbia, and to the activities of the TRCR. The award is presented each year to the author(s) of the paper from the previous year's symposium which best exemplified Tony's interests in the practical aspects of reclamation and the operational application of sound scientific principles in a cost-effective manner.

The 2012 award was presented to Becky Bravi of Terraforma and Bill Chapman of the Ministry of Forests, Lands and Natural Resource Operations, for their paper "Establishment of a Trial to Test Direct Seeding of Lodgepole Pine at Gibraltar" given at the 2012 symposium.

JAKE MCDONALD SCHOLARSHIP

To encourage student participation in reclamation-related fields, the Committee awards the Jake McDonald Memorial Scholarship, which is given to support a student (or students) in a scientific field related to reclamation. This scholarship was established in honour of Jake McDonald, former Senior Reclamation Inspector in the Ministry of Energy, Mines and Petroleum Resources. The TRCR also provides a $500 award to the Provincial Science Fairs Program.

The 2013 Jake McDonald Scholarship was awarded to:

- Jayba Guy, Simon Fraser University
- Michelle Phillips, Thompson River University
- Jessica Yuszko, Lethbridge College

The TRCR Science Fair Award for 2013 went to Megan Nantel from David Thompson Secondary, in the Greater Vancouver Regional Science Fair, for her project “Wild Diesel Destroyer”. This project involved the use of yeast strains from the Okanagan valley to biodegrade diesel in soil. Megan received a bronze medal at the Canada-Wide Science Fair in Lethbridge, Alberta.

REPORT OF THE AWARDS SUBCOMMITTEE

2012 MINE RECLAMATION AWARDS

Presented at the Reclamation Awards Banquet

September 18th, 2013
Kamloops, British Columbia

Introduction

Because the proceedings of each Mine Reclamation Symposium are published prior to the symposium, and because the recipients of the annual mine reclamation awards are kept confidential until the time of the awards banquet, we publish the awards report from the **previous** year's ceremony. The Technical and Research Committee on Reclamation is, therefore, pleased to include the Awards Subcommittee report from the 2013 Mine Reclamation Symposium.

Thirty-eight years ago, the British Columbia Technical and Research Committee on Reclamation established a Reclamation Award to recognize outstanding achievement in mine reclamation in British Columbia. In addition to this major award, the committee may also recognize reclamation successes through category awards for metal mining, coal mining, sand and gravel, quarries, placer mining and mineral and coal exploration.

The awards may recognize work of major or minor extent, and may be the result of a group of people or a single person's activities. These awards are assessed based on:

- quality in research
- innovation in techniques
- quality of work undertaken
- extent of land reclaimed
- work of a high standard that has been conducted over a number of years

This year the awards committee, comprising Kim Bellefontaine, Ministry of Energy and Mines, Craig Stewart with the BC Ministry of Environment, David Ewing with the Mining Association of BC and Wendy Gardner with Thompson Rivers University, chose to give two awards from the 7 nominations in 3 categories that were received. There continues to be evidence that strong reclamation efforts are being made throughout the province. It is clear that there is substantial planning and innovative thinking and techniques being tested at sites and this will pay dividends in the future both in terms of research that others can learn from and results that are exemplary for industry as a whole. The committee commended the efforts to date and looks forward to seeing future nominations from these and other sites that describe continued efforts and reclamation results.

TERMS OF REFERENCE FOR THE AWARDS

Awards Subcommittee

1. An awards subcommittee under direction of the Technical and Research Committee will decide the winner of the Reclamation Awards.

2. The awards subcommittee shall consist of four members appointed by the Technical and Research Committee, namely:

    1. University representative
    2. Government representative, Ministry of Energy and Mines
    3. Technical and Research Committee on Reclamation representative
    4. Mining Association of BC (industry) representative

3. Each member of the awards subcommittee has an equal vote.

Nominations

1. Nominations will be solicited by the Chair of the Awards Subcommittee from Ministry of Energy and Mines Inspectors, industry, and from committee members.

2. Nominations may be made by companies with respect to their own work, or work done by individuals or organizations familiar with the goals of reclamation.

3. Nominations should be submitted digitally to:

    Kim Bellefontaine Kim.Bellefontaine@gov.bc.ca
    Chair Awards Subcommittee
    Technical and Research Committee on Reclamation

4. In the nomination, documentation of the reclamation achievement must be outlined and reasons proposed why the project or program merits recognition. The award or citation nomination can be made on a nomination form.

5. Deadline for receipt of nominations for the awards is dependent on the date of the symposium in the year the award will be given.

Awards

1. The awards will be presented at the annual Mine Reclamation Symposium.

2. A British Columbia Mine Reclamation Award will be given annually.

3. Awards may also be given in the following categories:

    a) Metal Mine Reclamation
    b) Coal Mine Reclamation

c) Exploration
d) Sand, Gravel and Quarries
e) Placer Mining

4. Special awards may be awarded from time to time.

5. A keeper trophy will be given to the previous year's recipient of the Mine Reclamation Award.

Criteria for Selection

The work on which the reclamation awards are based may be major or minor in extent and may be the result of one person's activities. These awards may recognize the following:

a) Quality in research
b) Innovation in techniques
c) Quality of work undertaken
d) Extent of land reclaimed
e) Work which has been done over a number of years which, when taken together, is of a high standard.

2013 MINE RECLAMATION AWARDS

ANNUAL BRITISH COLUMBIA JAKE MCDONALD MINE RECLAMATION AWARD

The recipient of the 2012 *British Columbia Jake McDonald Mine Reclamation Award* was **Teck Metals Ltd.** for their outstanding reclamation achievements at the historic Pinchi Mine.

The Pinchi Mine is located on the north shore of Pinchi Lake, in north-central British Columbia, approximately 25 km northwest of the community of Fort St James. This historic mercury mine was originally owned and operated by the Consolidated Mining and Smelting Company, which later became Cominco Ltd., a predecessor of the current owner Teck Metals Ltd.

Between 1940 and 1944, cinnabar ore was mined from both surface glory hole and underground workings and ore was coarsely crushed and roasted to recover mercury. Much of the production was used to support the allied efforts of the Second World War. The roaster wastes (called calcines) were deposited on the shoreline and into Pinchi Lake, long before the environmental dangers of mercury were well known. Subsequent to that 1940's phase of operations, all structures were demolished. From 1968 to 1978 the mine was developed for a second time with more modern processing involving crushing, froth flotation circuits, and roasting of concentrate. Coarse tailings were backfilled underground and fine tailings were deposited into an impermeable tailings impoundment. The mine then went into a long dormant phase of care and maintenance.

The early 1990's marked the beginning of Teck's painstaking work to develop a closure plan for the site. Due to the significant environmental and health and safety risks associated with mercury, the planning and decommissioning process for the Pinchi mine was lengthy, challenging, and unique in many ways. Numerous studies were conducted to understand the sources, pathway and fate dynamics of mercury in both the aquatic and terrestrial environments using ecological risk assessments and formalized risk analyses. Aquatic assessment involved regional fish mercury and lake ecology studies. The terrestrial ecological risk assessment involved literature review, extensive field investigations, habitat surveys, food chain modeling, and the development of toxicity reference values for methyl mercury and several other metals. Ecological risks at the site were evaluated for 40 different wildlife species.

As a result, it was determined that only those species that feed primarily on insects or small mammals indicated potential risks, mainly from arsenic, inorganic mercury, and methyl mercury uptake. It was determined that the remediation of the millsite and tailings areas would further reduce the risks to wildlife. Other areas, with elevated inorganic mercury in soils were deemed to pose negligible to moderate risk but were not targeted for further remediation, as the risks to wildlife were considered to be higher if the naturally recovering habitat was disturbed.

An integral part to the development and acceptance of the Closure Plan was a comprehensive consultation process with the Tl'azt'en First Nations, and the Nak'azdli Band which began in 2005 and is still ongoing today. A Technical Working Group was formed and the local First Nations played a vital role in reviewing studies, directing additional studies and ultimately in finalizing the remediation strategies for reclamation and closure of the site.

One of the most challenging aspects of the closure plan related to the mercury contaminated processing facilities. Typical of many sites of that era, the mill contained a range of hazardous chemicals, friable and non-friable asbestos materials, lead paints and PCBs that required special handling and disposal. In addition, the mill buildings and equipment had substantial mercury contamination, which required extensive cleaning and testing prior to dismantling and disposal to a new on-site landfill located in the West Zone Pit.

The presence of low permeability glacial till, combined with the ability to backfill a large portion of the pit and eliminate a large portion of a waste rock dump, made the West Zone Pit the most attractive disposal option. Building debris was hydrologically isolated by placing the materials away from the sidewalls of the pit on top of a 10 metre thick layer of waste rock. The landfill was then capped with a 1 metre till cover, and any seepage drains to the underground workings and discharges out of the reclaimed 750 level portal where it is regularly monitored.

Mercury vapour exposure during the demolition and closure activities posed significant worker health and safety risks. Thus Teck established very strict protective procedures for workers. As well, much of the work was conducted during the colder winter months, when mercury vaporization was reduced due to lower temperatures.

The shipping and disposal of 47,000 kg of mercury contaminated residues posed significant challenges. The disposal had to be done in a manner that was acceptable to Teck's Product Stewardship Committee which requires recycling, life cycle assessment of products, and the minimization of impacts to people and the environment. The selected option involved shipping materials for recovery of mercury to the Clean Harbour facility in Quebec. Cleaned residuals were stabilized and landfilled at the Clean Harbour Lambton facility in Ontario while the liquid mercury recovered was shipped to Bethlehem Apparatus Co. in the US where it was roasted and sold as a marketable product.

Decommissioning of the 22 hectare tailings impoundment area included constructing an active spillway to reduce the phreatic surface and remove ponded water from the facility. This was done to enhance geotechnical stability and to eliminate the significant ecological risks associated with methylation of mercury and uptake through the food chain. The elimination of aquatic habitat in the tailings impoundment removed the exposure pathway for uptake. The subsequent terrestrial risks to browsing and burrowing animals from mercury in dry tailings was further reduced by spreading a layer of stabilizing iron sulphate sludge over the tailings and then capping with 1 metre of till. The capping of the tailings was also done in the winter to facilitate equipment working on top of the low strength tailings. The tailings were then revegetated using mixture of grasses, shrubs and trees.

The Pinchi Mine has now been successfully closed. To ensure that the remaining infrastructure and environmental conditions at the site remain stable over time and continues to meet regulatory requirements and closure plan objectives, Teck is implementing comprehensive geotechnical and environmental monitoring as well as formalized risk management plans

Pinchi Tailings impoundment (top) and Main zone pit area (bottom)

Bruce Donald and Michelle Unger accepted the 2012 British Columbia Jake McDonald Mine Reclamation Award on behalf of Teck Metals Limited.

2012 Award Winner Teck Metals Ltd.

L-R Dave Nikolejsin, Deputy Minister, Ministry of Energy and Mines, Michelle Unger, Teck Metals Ltd, Bruce Donald, Teck Metals Ltd., and Jaimie Dickson, 2013 Chair of TRCR

2012 METAL MINING RECLAMATION AWARD

The award for outstanding achievement for reclamation at a metal mine was presented to the Crown Contaminated Sites Program of the Ministry of Forests, Lands and Natural Resource Operations, as well as AECOM Canada Ltd. for their combined work at the historic Atlin Ruffner Mill and Tailings site.

The Atlin Ruffner Mill and Tailings site is an orphaned mine located on Crown Land near the town of Atlin in northwest British Columbia. The Ruffner mining area was first developed in 1899 and work continued intermittently until 1981. The deposit primarily produced silver, lead and zinc, along with some gold, copper, molybdenum and tin from underground workings. Small by today's standards, approximately 3,500 tonnes of ore was milled at the Ruffner mine site.

The site was brought to the attention of the BC Crown Contaminated Sites Program by the local regional district and ranked as a priority site in 2008. Infrastructure at the site included a mill building with machinery, ore storage bins, two mill pads, two trailers, a shack, an explosives shed, a tailings pond, two sedimentation ponds and an actively discharging adit.

Testing throughout the site determined that concentrations of many metals including arsenic, lead, zinc, antimony, copper, silver and cadmium were found to exceed Contaminated Site Regulation standards and indicated high risk conditions. Samples were also found to exceed the Hazardous Waste Regulation Leachate Quality Standards for various metals. Because of the presence of leachable hazardous waste, the age of the mill and lack of activity since the 1980's, the site was classified as a "historical hazardous waste contaminated site" under the Hazardous Waste Regulation.

The initial site investigations showed that water discharging from the adit, and the local groundwater at the site met standards, with the exception of cadmium and zinc. Concentrations of these parameters decreased substantially within 100 m down gradient from the site suggesting that metals were not being transported by leaching, and further indicating that this aspect of the site did not require specific remediation plans.

Thus the main objective of the remediation plan for the Atlin Ruffner site was to provide a robust, cost-effective, long-term solution to reduce long-term risk related to chemical contaminants. While most projects always aim to be as cost-effective as possible, this is particularly important for the Crown Contaminated Sites Program, as all of their activities are publically funded.

The focus of the remediation plan was to reduce the physical exposure of contaminated soils by placing a one metre thick permeable erosion resistant cap over all contaminated materials. This design allows for the installation of a geomembrane in the future if groundwater monitoring should indicate contamination. Notably, the plan required a change to the Hazardous Waste Regulation to allow for the construction of an in situ management facility, and in 2012 an approval was issued by the Ministry of Environment, the first of its kind in BC.

Upon receipt of approvals, hazardous wastes from the buildings, including fuel, chemicals and asbestos building materials, was removed to an off-site disposal facility. The mill building and equipment were then demolished, compacted and capped within the original footprint and adit water was diverted around the landfill area. The cap also covered the tailings pond, sedimentation pond and high-risk contaminated areas on the mill pads. Local borrow sources were used for capping materials and the area was reclaimed, incorporating the natural topography of the site. Revegetation plans are now currently under development and monitoring of cover performance and water quality will continue to be requirements for the site.

Work at the Atlin Ruffner site resolved contaminated site issues using cost-effective, remediation techniques. All parties involved are commended for their excellent work in addressing historic contamination issues and reducing the associated risk to public health and safety.

Atlin Ruffner Site post rehabilitation

Ryan Mills, AECOM Canada came forward to accept the award for outstanding Metal Mine Reclamation of the Atlin Ruffner Mill and Tailings, on behalf of the BC Crown Contaminated Sites Program of the Ministry of Forests, Lands and Natural Resource Operations and AECOM Canada Ltd.

Metal Mine Award – Atlin Ruffner Mill and Tailings Site

L-R Dave Nikolejsin, Deputy Minister, Ministry of Energy and Mines, Ryan Mills, AECOM and Jaimie Dickson, 2013 Chair of TRCR

LAST YEAR'S WINNER

The keeper trophy for last year's recipient of the Annual British Columbia Jake McDonald Mine Reclamation Award was presented to the **Tsolum River Partnership** for their outstanding efforts to reclaim the historic Mt. Washington mine site. This large-scale reclamation project was the result of a unique partnership that involved many people and agencies working together to mitigate the impacts of acid rock drainage on a large salmon fisheries resources. The installation of a liner significantly reduced copper loadings and has improved water quality to the point that aquatic habitat and salmon resources are beginning to recovery in the lower Tsolum River watershed.

As none of the current members of the Partnership were able to attend, a former participant of the Partnership, Ben Chalmers accepted the keeper trophy for the Tsolum River Partnership.

Ben Chalmers, previous Participant in the Tsolum River Partnership accepting the keeper trophy from Dave Nikolejsin, Deputy Minister, Ministry of Energy and Mines

2013 TRCR MEMBERS

L-R TRCR Committee: Dirk VanZyl, UBC; Carla Fraser, Teck Coal; Darren Cowan, Hillsborough Resources; Jaimie Dickson, Highland Valley Copper and 2013 Chair of TRCR; David Ewing, MABC; Ryan Todd, New Gold and 2014 TRCR Chair of TRCR; Jonathan Buchanon, AMEBC, Kim Bellefontaine, MEM; Wendy Gardner, Thompson Rivers University; Kim Bittman, Taseko Mines, Ben Chalmers, Mining Association of Canada, Tania Demchuk, MEM; and Bill Price, NRCan.

Missing: Nicole Pesonen, Walter Energy and Vice Chair of the TRCR; Craig Stewart, MOE, Todd Wambolt, Gibraltar Mines, and Angela Waterman, MABC.

BCTRCR
TECHNICAL AND RESEARCH
COMMITTEE ON RECLAMATION

Gold Sponsors

Teck

newgold™

Stantec

Silver Sponsor

MABC
MINING ASSOCIATION OF BRITISH COLUMBIA

Bronze Sponsors

AMEBC
Association for Mineral Exploration British Columbia

O'Kane
Consultants Inc.
Integrated Mine Waste Management and Closure Services
Specialists in Geochemistry and Unsaturated Zone Hydrology

ACKNOWLEDGEMENTS

The British Columbia Technical and Research Committee on Reclamation wishes to thank the following individuals and/or organizations for their contributions to the 2014 Reclamation Symposium:

- Ryan Todd (New Gold Inc.), 2014 Chair of the British Columbia Technical and Research Committee on Reclamation;
- Nicole Pesonen (Walter Energy Inc.), 2014 Vice-Chair of the British Columbia Technical and Research Committee on Reclamation;
- Tania Demchuk (Ministry of Energy and Mines), 2014 Secretary/ Treasurer of the British Columbia Technical and Research Committee on Reclamation;
- Mike O'Kane (O'Kane Consultants Inc.) and Justin Straker (Integral Ecology Group Ltd.) for presenting the Design and Assessment of Mine-Waste Cover Systems Workshop
- Michelle Unger (Teck), Bruce Donald (Teck), Steve Hilts (Teck), Randy Baker (Azimuth), Gary Mann (Azimuth), Patrick Allard (Azimuth), John Prezcek (Pryzm Environmental), Rob Marsland (Marsland Associates), Shannon Crwoley (John Prince Research Forest) – Pinchi Lake Mine tour
- Harold Bent (AuRico Gold Inc.) – Kemess Mine tour
- Ryan Todd (New Gold Inc.), Nicole Pesonen (Walter Energy Inc.), Kim Bittman (Taseko Mines Limited) and Tania Demchuk (Ministry of Energy and Mines) – Symposium Sub-committee
- Dirk van Zyl (University of British Columbia) – Technical Paper Sub-committee
- Kim Bittman (Taseko Mines Limited), Wendy Gardner (Thompson Rivers University), Craig Stewart (Ministry of Environment) and Dirk van Zyl (University of British Columbia) – Scholarship Sub-committee
- Bill Price (Natural Resources Canada), Jaimie Dickson (Teck Highland Valley Copper Partnership) – First Nations and Community Sub-committee
- Jonathan Buchanan (Association for Mineral Exploration, British Columbia) – Media Sub-Committee
- Kim Bittman (Taseko Mines Limited– for Website Maintenance; Michelle Donnelly (Mining Association of BC) – for email blasts; and Jonathan Buchanan (AME BC) - for twitter posts.
- Michelle Donnelly (Mining Association of British Columbia) for administrative assistance for the committee
- Cantrav Services Inc for event organization (Missy Preston, Shannon Coutts, Martina Waldkirch)
- Student Volunteers: Kathy Baethke (Thompson Rivers University), Jayda Guy (Simon Fraser University), Geoffrey Lewicki (University of Northern British Columbia/ University of British Columbia), Michelle Phillips (Thompson Rivers University), Anny Shen (University of British Columbia), Linlin Zhang (University of British Columbia)

Sponsors

The BC TRCR is proud to acknowledge the generous contributions of our sponsors for the 38th Annual BC Mine Reclamation Symposium.

- New Gold for a Gold Level Sponsorship
- Stantec for a Gold Level Sponsorship
- Teck for a Gold Level Sponsorship
- Mining Association of British Columbia for a Silver Level Sponsorship
- Association for Mineral Exploration British Columbia for a Bronze Level Sponsorship
- O'Kane Consultants Inc for a Bronze Level Sponsorship.

THANK YOU EVERYONE!

SITE-SPECIFIC WATER QUALITY OBJECTIVES FOR MINE ENVIRONMENTAL MANAGEMENT

Doug Bright, Ph.D., R.P.Bio[1], Debbie Bryant, M.Sc.[1], and Curtis Eickhoff, Ph.D.[2]

[1]Environmental Risk Assessment,
Hemmera, Victoria, BC.

[2]Director - Ecotoxicology,
Maxxam Analytics, Burnaby, BC.

ABSTRACT

Few environmental issues are of greater importance for the planning and approvals of new mines, operational management, and closure/remediation planning than the often complex hydrogeochemical interactions between surface waters that support aquatic life and mine wastes or recently disturbed areas. There is a strong reliance for permitting and approvals within British Columbia (BC) and other Canadian jurisdictions on generic provincial and national water quality guidelines to interpret aquatic ecological risk potential and protect freshwater life. The mining sector supports greater use of site-specific, risk-based water quality benchmarks, which necessarily require a more direct scientific understanding of aquatic ecological risks in watersheds of interest; however, the adoption of site-specific water quality objective (SSWQO) approaches has been limited.

We discuss the methods used to develop generic water quality guidelines, including differences between Canadian (CCME) and BC water quality guideline derivation protocols. The relationship between generic guidelines and SSWQOs are discussed, along with the various practical approaches for development of SSWQOs based on toxicity data such as the water effects ratio (WER) method. Finally, we discuss the factors that may undermine the more widespread adoption of SSWQOs, and of the associated support for science-based decisions for environmental management in the mining sector.

KEY WORDS

Water quality, toxicity testing, risk assessment, site-specific, water effects ratio

INTRODUCTION

Significant progress has been made over the last three decades towards minimizing the impacts of base metal, coal, and other types of mining on ecologically productive aquatic ecosystems. Such progress includes (i) improvements in both mineral extraction and effluent treatment technologies; (ii) an increased percentage of process water re-use; (iii) greater knowledge about the roots of acidic drainage and other forms of weathering/leaching from mine wastes (e.g., non-acidic drainage); (iv) mining design and management at landscape and watershed scales based on an improved understanding of site hydrology (e.g. for tailing management facilities and waste rock deposits); and (v) advances in our understanding of groundwater-surface water interactions, especially in the context of seepage down-gradient from waste deposits. At the same time, the average scale of proposed and new mines worldwide has increased by roughly an order of magnitude with each passing decade, as higher-grade deposits are depleted and as extraction technologies improve in efficiency and cost-effectiveness.

Generic water quality guidelines have historically played a key role in mining environmental management, and will continue to do so in the future in the context of regulatory approvals, discharge permits, monitoring, and closure. According to the BC Ministry of Environment (MOE), an aquatic environmental impact assessment is “the main tool to inform the statutory decision maker of proposed risks of the project to the aquatic environment” (BC MOE 2013a), which in turn is informed by the comparison of data and predictions to the BC generic water quality guidelines or SSWQOs. Generic water quality guidelines, as promulgated by the BC MOE, the Canadian Council of Ministers of the Environment (CCME), the United State Environmental Protection Agency (US EPA), and other agencies are generally intended to “ensure protection of the most sensitive intended water use” (BC MOE 2013b). Use of SSWQOs as a management tool instead of generic water quality guidelines is not intended to alter the desired levels of protection of sensitive uses or aquatic ecosystem health, as defined through public and regulatory policy, but rather allow for greater consideration of the site-specific abiotic and biotic characteristics of lotic, lentic, estuarine, and marine ecosystems that could influence either the sensitivity of biota present or the degree of exposure relative to a measured concentration in water.

A common perception among some members of the regulated community about provincial and national water quality guidelines is that some of the values are overly conservative, leading to predictions of risks to freshwater life at waterborne trace element concentrations that would not impair the intended use of the water body to support the productivity and biodiversity of fish and other taxa of ecological, social, cultural, and economic importance. Perhaps it is for this reason that many mine environmental managers and mining associations such as the Mining Association of BC (MABC) and the Mining Association of Canada (MAC) have advocated for increased regulatory acceptance of the development and use of SSWQOs along with site-specific risk assessments in general.

This paper briefly discusses the derivation procedures used for development of generic BC and CCME water quality guidelines, along with the major reasons that a generic water quality guideline might not be an accurate predictor of a threshold of impairment to the most sensitive intended water use on a site-specific basis. We then summarize the four major approaches that have been prescribed by BC MOE and CCME for derivation of a SSWQO. These approaches are (i) the background concentration approach; (ii) the analytical limit of quantification approach; (iii) the re-calculation (resident species) approach; and (iv) the water effects ratio (WER) approach (CCME 2003, BC MOE 2013b). We provide several relevant examples in which such approaches have been applied, with a focus on the WER approach. Finally, we discuss potential impediments to the use of site-specific approaches for the mining sector in British Columbia, based in part on expectations about the level of effort required to develop a SSWQO relative to potential for (i) shifting environmental management objectives and (ii) regulatory and public acceptance. This brief discussion is also based on the recognition that there is broad acceptance by both the environmental regulator and mining community of the value of environmental monitoring of aquatic effects as an important environmental protection tool, with a focus on field-collected population, community, and other biological data in surface waters that may be affected by current or past mining activities. Generic and site-specific water quality criteria, therefore, are only a subset of tools and approaches that are used for protecting aquatic resources and uses.

ROLE AND LIMITATIONS OF GENERIC WATER QUALITY GUIDELINES

Contemporary derivation protocols for environmental quality guidelines begin with the collation of the existing available toxicological data that relate an adverse effect in an organism of interest and an exposure concentration or dose for a substance of interest. The ability to derive a meaningful and reasonably accurate (relative to narrative objectives) water quality guideline from the meta-analysis of relevant ecotoxicity data is limited by the availability and quality of the existing data. For aquatic ecotoxicity data, considerable effort has gone into developing internet-accessible and other biological effects databases. The fact that the existing primary data have typically been produced as part of various studies with objectives far removed from the identification of toxicological thresholds, however, suggests a need for a reasonable level of skepticism when attempting to define toxicity reference values intended to protect all multicellular species within aquatic ecosystems.

Elements of the derivation procedure that are common to the BC MOE (2012), CCME (2007), and the US EPA (1985) approaches include toxicity data collation, screening, and manipulation, as illustrated in Figure 1. The BC MOE approach departs from the US EPA approach and CCME (2007) revised approach in a few major respects. First and foremost is that the preferred CCME and US EPA option, the availability of data permitting, uses a species sensitivity distribution approach (SSD). The SSD approach amalgamates concentration-based toxicological thresholds for each aquatic species from the lowest to highest concentrations. For example, an estimated concentration from experimental (laboratory toxicity) results associated with a 20% reduction in the growth or reproduction of a species (i.e., an EC_{20}, or concentration associated with a 20% effect size) is tabulated, and the relative ranking of an EC_{20} value for each species in the tabulated data provides an indication of its relative sensitivity. The nomination of a value at or very close to the lower end of the SSD (e.g. the 5th percentile value) serves as a generic threshold that should adequately protect all potentially present aquatic species.

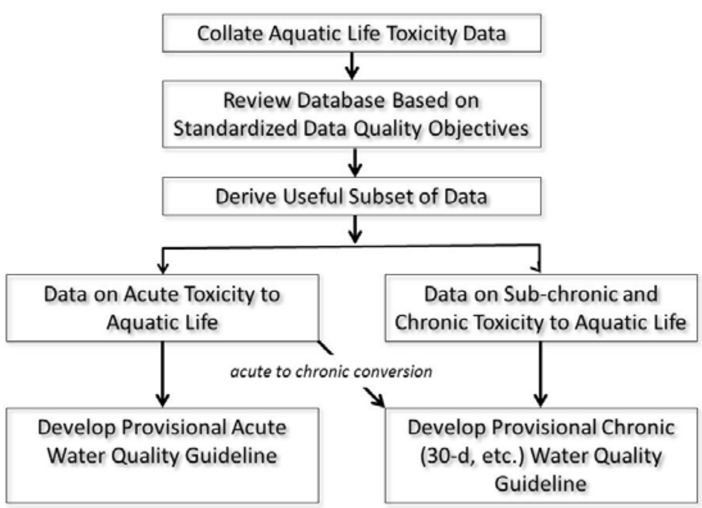


Figure 1: Routine approach for the development of generic water quality criteria

The degree of conservatism in achieving narrative protection goals for an SSD-type approach depends especially on three aspects of data collation and manipulation: (i) the specific types of toxicological endpoints selected to develop the SSD; (ii) whether there are additional uncertainty factors (UFs) applied to the actual individual statistical estimates for each species and toxicological endpoint; and (iii) the point on the SSD that is selected as the generic water quality guideline, allowing for upper and lower confidence intervals in the statistical curve-fitting for the SSD.

CCME (2007) prescribes the use of toxicological data from available studies (or EC_X estimates secondarily calculated from contributing studies) with the following order of preference:

$$EC_{10}/IC_{10} > EC_{11\text{-}23}/IC_{11\text{-}23} > \text{MATC} > \text{NOEC} > \text{LOEC} > EC_{24\text{-}49}/IC_{24\text{-}49} > \text{non-lethal } EC_{50}/IC_{50}$$

IC stands for inhibitory concentration, while the maximum acceptable toxicant concentration (MATC) is calculated as the geometric mean of the no observed effect concentration (NOEC) and lowest observed effect concentration (LOEC).

The preference for EC_X/IC_X data, where x <<50%, and especially ≤10% raises some important issues, not the least of which is that the statistical robustness of the point estimate on the dose-response curve is low in comparison to an EC_{50} value, since it typically involves a statistical interpolation (or rarely an extrapolation) well beyond the observational range of most of the experimental data. Furthermore, an EC_{10} estimate for most aquatic toxicity studies is rarely statistically, significantly different than the EC_0; i.e., the response in the experimental controls. In fact, the CCME Type A approach is specifically intended to preferentially and primarily plot no-effect data for the derivation of SSDs (CCME 2007).

The vast majority of available toxicological data for aquatic life is also expressed as an acute or sub-acute LC_{50} (lethal concentration for 50% of the test organisms) or EC_{50}, which is not a favoured data type in the CCME approach. While it can be argued that an ambient environmental concentration no greater than a chronic LC_{50} value would allow some members of the affected sub-population of the species of interest to survive, and would be commensurate with the persistence of all less sensitive taxa, neither BC MOE nor CCME favour this argument. In fact, narrative statements on aquatic environmental protection goals, as published by BC MOE and CCME, are sufficiently vague that there is no means of exploring what is or is not likely to be a significant aquatic ecological effect (or impairment of use for aquatic life) based on constructs such as ecological productivity, function, and biodiversity. CCME water quality guidelines are intended to "protect all species all the time" (CCME 2007). A key message, therefore, is that the development of SSWQOs will not be useful in circumstances where there is a fundamental disagreement with policies and practices on how the narrative protection goals are translated into generic numerical guidelines.

Under the BC MOE approach (Meays 2012), the preferred ranking of available toxicity data is similar to the CCME approach:

$$EC_X/IC_X \text{ representing low-effects threshold} > EC_{15\text{-}25}/IC_{15\text{-}25} > \text{LOEC} > EC_{26\text{-}49}/IC_{26\text{-}49}$$
$$> \text{non-lethal } EC_{50}/IC_{50} > LC_{50}$$

The development of a generic water quality guideline, however, is not based on an SSD approach. Rather, the lowest relevant toxicity endpoints are scrutinized and an uncertainty factor, typically in the range of 2 to 10, is applied to derive a short-term maximum and long-term average water quality guideline. Not surprisingly, one of the major points of contention for specific BC water quality guidelines is in how professional judgments have been applied in the nomination of uncertainty factors.

Under both the BC and CCME approaches, as formally described in their derivation documents, there appears to be some confusion about whether an EC effect size less than 10% to 15% is a *de facto* no effect or low effect level.

For recent CCME and BC MOE derivations of generic water quality guidelines, the role of exposure and toxicity-modifying factors (ETMFs) (CCME 2007) has received considerable attention. Such factors potentially include pH, alkalinity, hardness, dissolved oxygen (or redox), and temperature. Several generic water quality guidelines include specific predictive regression-based estimates of the threshold of toxicity to freshwater life based on relationships with pH or hardness. For the BC water quality guidelines, these include aluminum, cadmium (in draft), copper, fluoride, lead, silver, sulfate, and zinc. The absence of reference to ETMFs in currently adopted and working generic water quality guidelines does not mean that such modifying factors are not potentially important to an understanding of aquatic ecological risks, but rather that there likely has been inadequate scientific knowledge to incorporate adjustments for ETMFs into the derivation. In addition, the mathematical formulae that have been incorporated in the above-listed guidelines to account for ETMFs should not be viewed as over-arching scientific truths but rather as best approximations of toxicological potential from total waterborne concentrations and important co-variates based on the scientific data available from studies of a very limited subset of species that are of interest (under the premise that all models are wrong; however, some are useful).

The vast majority of derivation protocols for provincial, national, and international water quality guidelines have – by policy – attempted to minimize the probability of committing Type II errors (Figure 2) at the expense of inflating the probability of making Type I errors; i.e., predicting effects to aquatic life in the receiving environment based on reference to generic water quality guidelines when in fact no effects are present. As an environmental quality guideline approaches a value of zero, the probability of a Type II error approaches zero; however, the probability of a Type I error approaches 100%, with commensurate societal implications for the investment of resources in environmental protection beyond any real potential for environmental gains.

For the promulgation of generic environmental quality guidelines in general, the recognized degree of conservatism in the resulting environmental effects thresholds is often rationalized in part based on the fact that generic environmental quality guidelines are necessarily applicable across a broad range of ecosystems, biological species assemblages, habitats, and communities, and must be able to provide adequate protection for more sensitive versus average conditions. The use of site-specific approaches, therefore, should in theory allow a better balancing of the probability of making Type I versus Type II errors.

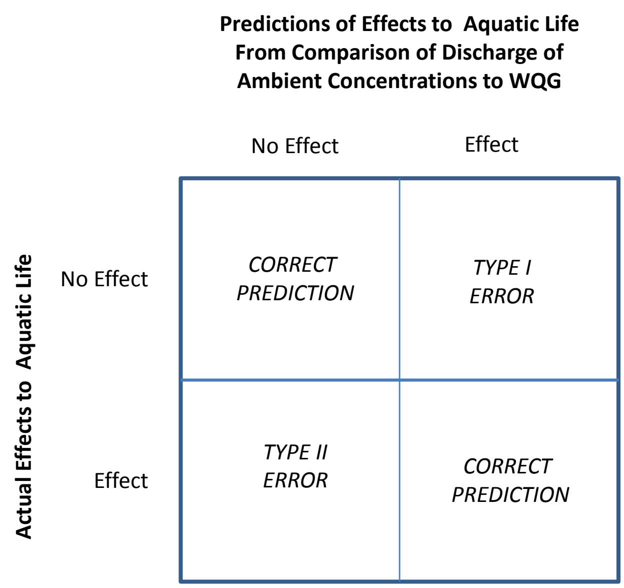


Figure 2: Type II versus type I error in the application of water quality guidelines

PRESCRIBED APPROACHES FOR DEVELOPMENT OF SSWQOS

The supporting rationale and specific methods for development of SSWQOs are documented elsewhere (BC MOE, 2013b; CCME, 2003). Of particular note is that the BC MOE approach for SSWQO is prescriptively constrained, allowing for the application of one or more of four possible approaches. Each of these is briefly described below from the perspective of regulated users.

<u>Background Concentration Approach</u>

It is not reasonable to manage human activities toward the achievement of chemistry-based numerical objectives in the receiving environment that are lower than naturally occurring background concentrations; this principle is broadly recognized in the development of provincial and national regulatory policy and guidance. In the context of mining, it is common to observe the concentrations of some trace elements dissolved in water, in suspended sediments, or in bed sediments at concentrations that exceed the generic guideline for aquatic life protection, For such circumstances, BC MOE prescribes adopting as the preliminary SSWQO an upper limit of natural background concentration based on the 95th percentile concentration.

The Background Concentration Approach is useful in a minority of circumstances in BC for mining environmental management. We have generally found within our studies that the initial perceptions about naturally elevated trace elements are greater than the reality, particularly for downstream areas of watersheds that broadly integrate meteoric water from both highly mineralized areas with surface manifestations and adjacent non-mineralized sub-watersheds. Furthermore, the approach is generally relevant to pristine watersheds, and it becomes more challenging to define through field studies an appropriate upper limit of natural background concentrations in sub-watersheds that may have already been influenced by exploration and extraction activities.

The Background Concentration Approach is sometimes useful as a quick fix for BC working and approved generic water quality guidelines, or CCME guidelines, that were derived using older and different methodologies, and under an era when peer review was not as important in the overall process. The boron BC water quality guideline for protection of marine life, updated in 2003, was set at 1,200 µg/L even though the guideline derivation document states "median values for boron in surface water is about 0.1 mg/L and in Canadian coastal marine water it ranges from 3.7 to 4.3 mg/L"(Moss and Nagpal 2003).

Analytical Limit of Quantification Approach

Rarely, a generic environmental quality guideline has been developed that is lower than the analytical quantification limit based on reasonably accessible laboratory services, methods, and instruments. This occurs for one of the following reasons:

1. The provisional generic water quality guideline was derived through the application of uncertainty factors or other lower-concentration extrapolation procedures that resulted in a calculated threshold for protention of aquatic life protection that is substantially lower than any concentration referenced in any of the underlying laboratory ecotoxicity studies.

2. The critical ecotoxicity study(ies) that strongly influence the provisional water quality guideline did not use a measured exposure concentration, but rather assumed a spiked (nominal) exposure level (e.g., based on serial dilutions), often without due consideration of the associated uncertainty.

3. The researchers that carried out the critical ecotoxicity studies had access to highly specialized, research-quality chemical analytical tools that are generally not readily available beyond specialized research institutions.

4. The water quality guideline is based on back-calculation from a tissue residue concentration, wherein the critical threshold has been based on bioconcentration or biomagnification.

Modern water quality guideline derivations should adequately account for such possibilities; however, water quality guidelines derived more than a decade ago may require adjustments to reflect practical limits of quantification. According to BC MOE (2013b), "A provisional WQO could be set at the achievable analytical level until the improved detection limit can be established."

Recalculation Approach (Resident Species Approach)

The USEPA (2013), CCME (2007), and BC MOE (2013b) allow for the re-calculation of water quality guidelines from the available credible aquatic toxicity data used in the original derivation of the generic water quality guideline to incorporate data for only those taxa that are potentially present in the water bodies of interest. The particulars of the prescriptive guidance for each authority are different but the underlying principles and procedures are the same: The aquatic biota that are present or potentially present within the water bodies of interest need to be defined through references to authoritative sources and based on field surveys. The list of actual potentially present taxa is then cross-referenced against the collated ecotoxicity data that underpin the derivation of the generic environmental quality guideline: Taxa within the ecotoxicity database that are not relevant to waters of interest are omitted and the water quality guideline is re-calculated based on the prescribed procedures for the derivation of generic water quality guidelines.

In practice, this site-specific species selection of recalculation approach has limited utility. Since ecological risks assessments broadly rely on data from surrogate species, it is often difficult to eliminate specific data for one or more potential surrogate species as being irrelevant to the species that might potentially inhabit the site of interest. Furthermore, when assessing trace element, major ion, or nutrient toxicity in surface waters the most sensitive toxicity endpoint in the database is often from a chronic (e.g., reproductive impairment) test on a cladoceran such as *Ceridaphnia dubia* or *Daphnia magna*, for which allied species within the same order and family are very widely distributed in temperate to arctic surface waters. In addition, the elimination of aquatic ecotoxicity data for absent species can have the effect of imposing further data limitations. Checks and balances, as part of the recalculation approach, include the assurance that sufficient ecotoxicity data remain in the reduced database that are of relevance to the site-specific list of aquatic organisms. It may be necessary, therefore, to undertake additional laboratory toxicity studies to meet minimum data requirements for development of a SSWQOs. Under the BC MOE approach, the smaller the available ecotoxicity data set, the larger the uncertainty factor that is likely to be accepted in defining a SSWQO.

Water Effects Ratio Approach

The Water Effect Ratio (WER) procedure is useful in circumstances in which factors related to the physical and chemical characteristics of the mine discharge receiving water are expected to modify the toxicity of the trace element(s) of concern. For example, high levels of total suspended solids (TSS), dissolved organic carbon (DOC), or total organic carbon (TOC) may indicate the presence of high concentrations of organic ligands such as humic and fulvic acids that may interact with metals/metalloids and alter their bioavailability and/or toxicity in the water column. If the receiving water has high concentrations of organics, the WER approach is likely to provide a SSWQO for trace elements such as copper or zinc that is higher than the generic water quality guideline. Adequate knowledge about the aquatic geochemistry of the substances of interest will greatly assist with decisions about whether a WER approach is potentially helpful. For inorganic arsenicals, the scientific literature is somewhat equivocal but nonetheless indicates that organic matter is not a strong sorptive media or complexing agent in comparison with iron and manganese oxyhydroxides. A WER approach, therefore, is likely to have limited value in producing a SSWQO that is substantially different than the generic water quality guideline for inorganic arsenic.

An advantage of the WER approach is that it directly compares the actual toxicity of a substance of concern – across a range of concentrations – in the receiving water and in standardized laboratory test water. Aquatic toxicity bioassays typically are performed with pristine laboratory control or dilution water. Generally, the water is treated by filtration and TOC is normally very low. Many of the aquatic toxicity bioassays such as acute *Daphnia*, and sublethal tests with algae (*Pseudokirschneriella subcapitata*), invertebrates (*Ceriodaphnia dubia*), and aquatic vascular plants (*Lemna minor*) are conducted with reconstituted waters prepared from purified deionised water. These recipes, specified by the test guidelines, generally have very low TOC content; therefore, toxicity tests conducted on an effluent with pristine lab water as a diluent may produce different results than tests conducted using receiving water that is high in TOC.

It is recommended that bioassays are conducted with at least two indicator species using site or receiving water and lab water (see Table A5.2 of BC MOE, 2013b). Typically, an acute and a sublethal or short-term chronic toxicity test will be conducted with organisms that are chosen based on the applicability to the system, availability, and sensitivities to metals and metalloids in freshwater systems. The 96-hour rainbow trout and 48-hour *Daphnia magna* are commonly used to assess acute toxicity. Fathead minnows (*Pimephales promelas*), water fleas (*C.dubia*), freshwater algae, and other species may be used to evaluate longer-term effects to growth and/or reproduction. Other species may be used that are considered resident or particularly sensitive to the substance(s) of concern.

For metals, a stock solution is prepared using a readily soluble salt (e.g. nitrate, chloride, or sulphate salt) that is spiked into both lab water and site water at the same range of concentrations. A range-finding study that investigates a wide range of concentrations is helpful to better narrow the concentration range of interest for subsequent definitive tests (i.e., to narrowly bracket the toxicological endpoint of interest). The concentration of the metal (dissolved and total) should be determined in the test solutions at the beginning and end of the tests to ensure that the measured concentrations are similar to the nominal or expected concentrations. This determination is essential to understanding the results of the tests and to ensuring the quality and accuracy of the test endpoint (LC_{50}) values.

In some instances, obtaining the correct metal concentrations in the test solutions may be challenging if the metals interact with elements of the lab or site waters. For example, carbonate in the recipe of the *Ceriodaphnia dubia* test water may react and precipitate out the test metals, which may reduce the expected concentrations of metal available in laboratory water solutions; this will reduce the concentration of dissolved metal in the tests over time and may result in reduced toxicity in the laboratory test water. In site waters, calcite super-saturation can have a similar influence. Other quality assurance and quality controls are incorporated to identify deviations from the test procedures, and problems associated with temperature, loading rates, feeding, dissolved oxygen, pH, control failures, and other issues that may compromise the reliability of the bioassay results.

The WER value is calculated for each species based on the results of the bioassays conducted using the spiked site/receiving water and laboratory water. Data from the toxicity tests are used to calculate the appropriate statistically derived endpoints such as the LC_{50}, EC_{50}, or IC_{50} values. The WER value is calculated as follows in Equation 1:

$$\text{WER} = \text{Site Water LC50}/\text{Lab Water LC50} \qquad \textbf{[1]}$$

If the WER values for the two species are similar, (within three-fold of each other), then the geometric mean of the two values is calculated and used to modify the generic BC WQG as shown in Equation 2:

$$SSWQO = WQG \times WER \quad \textbf{[2]}$$

If the two WER values are not similar, then an additional set of paired tests is conducted with another relevant species to confirm or refute results of the two initial tests. The SSWQO would then be derived by multiplying the BC generic water quality guideline by the WER calculated as the geometric mean of the two similar lowest WER values. For metals, the final WER is calculated for both total and dissolved concentrations of the metal of interest.

EXAMPLE OF THE WER APPROACH FOR SSWQO ESTIMATION:

Maxxam performed a WER experiment to support development of an SSWQO for copper for a mine project in the BC and Yukon region. Aquatic toxicity tests were performed on two species, using both acute and chronic exposure durations: 96-h rainbow trout acute test (per Environment Canada EPS 1/RM/13); *Ceriodaphnia dubia* 48-h acute test, and 7-d, three-brood reproduction test (per the US EPA 2002 acute method and Environment Canada 2007 chronic method).

Total and dissolved metal concentrations, water hardness, TSS, and TOC were measured in the site water and the hardened laboratory water. The site water total hardness calculated based on the metal analysis was on average 200 mg CaC_O3/L. The TOC was 2.47 mg/L and the TSS was below 1 mg/L. The laboratory water hardness used for the fish tests was adjusted to 204 - 212 mg $CaCO_3$/L to match the site water. The TOC was 0.66 mg/L and the TSS was below 4 mg/L. For the *Ceriodaphnia* tests, the laboratory dilution water was made with Type I deionized water adjusted to 188 mg $CaCO_3$/L hardness with Perrier water, reflecting the hardness of the site water, and addition of vitamin B12 & selenium to maintain the health of the organisms. The culture water used was made with deionized water hardened to 96 mg/L CaCO3 with 20% Perrier water, vitamin B12, and selenium.

Stock solutions were prepared using a highly soluble reagent-grade $CuCl_2$ spiked separately into the laboratory water and the site water. The nominal copper concentration ranges were 0 µg/L to 228 µg/L in laboratory water, and 0 µg/L to 638 µg/L in the site water. Prior to initiating the tests, subsamples were collected from all test concentrations for the analysis of total and dissolved copper. All test solutions were further subsampled for analysis prior to and after a 48-hour water renewal and at the end of the tests.

To calculate the WER value for copper, the laboratory and site water 96-hour LC50 were estimated by probit analysis based on measured copper concentrations of the test solutions. The subsequent WER values, calculated based on measured concentrations of copper in test solutions, are presented in Table 1.

Table 1: Results of Water Effects Ratio Testing

Test	LC50 Lab. Water (Diss. copper µg/L)	LC50 Site water (Diss. Copper µg/L)	WER
Acute rainbow trout	115.7 (97.64-137.1)	188.2 (156.5-225.9)	1.6
Acute *Ceriodaphnia dubia*	3.2 (2.9 - 3.4)	12.0 (9.1 - 16.7)	3.8
Chronic *C. dubia* survival	6.8 (5.7 - 8.1)	33.8 (30.3 - 37.7)	5.0
Chronic *C. dubia* reproduction	3.6 (2.8 - 4.2)	23.7 (17.8 - 28.3)	6.6

The final WER values for copper for the mine site in question were obtained by determining the geometric mean of the comparable WER values obtained from the toxicity tests. In this instance, the WER values obtained from the chronic *Ceriodaphnia* tests were higher than that obtained for the acute trout and *Ceriodaphnia* tests. The guidance for deriving site-specific water quality objectives (BC MOE 2013b) states that when more than two WERs have been determined, the final WER should be calculated as the geometric mean of two lowest WER values. Therefore, in this study, the final WER value was calculated as the geometric mean of the WER values obtained from the acute fish and *Ceriodaphnia* tests as follows:

$$\text{Final WER} = \text{geometric mean } (1.6, 3.8) = 2.5 \qquad \textbf{[3]}$$

In this example, the water quality objective for copper was then calculated based on the water quality guideline for copper. The water quality objectives for the protection of aquatic life for copper obtained from BC MOE are presented in Table 2.

Table 2: Summary of Water Quality Criteria for Copper

Criterion	30-day Average µg/L Total Copper	Maximum µg/L Total Copper
Freshwater Aquatic Life (when average water hardness as $CaCO_3$ is greater than 50 mg/L)	<0.04 (mean hardness) µg/L	(0.094(hardness)+2) µg/L (hardness as mg/L $CaCO_3$)
WQG	8.0	21
SSWQO	20	51

Based on a mean site water hardness of 200 mg$CaCO_3$/L, the water quality guideline for copper would be <8 µg/L Total Copper for the 30-day Average and 21 µg/L Total Copper as a maximum concentration. The SSWQOs were then calculated based on Equation 2 and are presented in Table 2. The WQG values were multiplied by the WER value of 2.5, which resulted in SSWQO values of 20 µg/L and 51 µg/L total copper for the 30-day average and maximum concentrations respectively.

CONCLUSIONS

The SSWQO derivation approaches prescribed by BC MOE and CCME are of practical value to mine environmental planners and managers under specific circumstances. As discussed above, three of the four prescribed approaches (background concentration approach, limit of quantification approach, recalculation approach) are likely to be relevant to only a very limited number of substances and site-specific receiving environments. The water effects ratio approach has proven to have good practical value for sites where cadmium, copper, lead, silver, and zinc are substances of interest (Nautilus Environmental 2009; Diamond et al. 1997; Jop et al. 1995). The WER approach has been used for copper more than any other trace element. An obvious challenge with the WER approach is that it can provide only limited compensation for a generic water quality guideline that is based on potentially unrepresentative primary data, or is unduly influenced by conservatism in the derivation approach, since the SSWQO is a product of the generic water quality objective and the WER.

The available approaches for development of SSWQOs do not address several aspects associated with application of generic water quality guidelines. Among these are cases where the existing approved and working generic water quality guideline do not provide an accurate estimate of the threshold of toxicity based on (i) non-reproducible primary data that drive the lower end of the SSD or are among the lowest values available at the time of derivation; (ii) use of uncertainty factors or other compensatory approaches in the face of data limitations; (iii) estimation of EC_{10} or similar endpoints from dose-response data with the attendant degree of statistical uncertainty; and/or (iv) the chemical form of the substance of concern in the site receiving environment relative to the readily soluble metal salts that are typically used in WER studies. Often, the best approach for dealing with scientific or technical limitations in the existing generic water quality guidelines is to complete focused scientific studies to address knowledge gaps (e.g., to better understand modifying factors for exposure and toxicity) that are not accommodated within the existing prescribed SSWQO approaches. Furthermore, the prescribed SSWQO approaches cannot reconcile philosophical differences in professional opinion about the translation of vague narrative protection goals into concrete management objectives, or resolve perceptions regarding the effectiveness of aquatic environmental protection on a case-by-case basis as a result of interpretations of water chemistry relative to water quality guidelines versus biological survey data on resident species, communities, biodiversity, and productivity.

REFERENCES

British Columbia Ministry of Environment (BC MOE). 2013a. The effluent permitting process under the *Environmental Management Act:* An overview for mine project applicants. 19 pp. http://www.env.gov.bc.ca/epd/industrial/mining/pdf/effluent_permitting_guidance_doc_mining_proponents_apr2013.pdf (accessed July 19, 2014).

BC MOE. 2013b. Guidance for the derivation and application of water quality objectives in British Columbia. 147 pp. http://www.env.gov.bc.ca/wat/wq/pdf/wqo_2013.pdf (accessed July 19, 2014).

Canadian Council of Ministers of the Environment (CCME). 2003. Guidance on the site-specific application of water quality guidelines in Canada: Procedures for deriving numerical water quality objectives.

http://www.google.ca/url?sa=t&rct=j&q=&esrc=s&source=web&cd=1&ved=0CBwQFjAA&url=http%3A%2F%2Fceqg-rcqe.ccme.ca%2Fdownload%2Fen%2F221%2F&ei=8bXNU8S5NeGmigKIvYAI&usg=AFQjCNFoXvp9_8mibOLd9sia_zloMAUp7Q&bvm=bv.71198958,d.cGE (accessed July 19, 2014).

CCME. 2007. A protocol for the derivation of water quality guidelines for the protection of aquatic life 2007. 37 pp. http://www.google.ca/url?sa=t&rct=j&q=&esrc=s&source=web&cd=1&ved=0CBwQFjAA&url=http%3A%2F%2Fdocuments.ccme.ca%2Fdownload%2Fen%2F220%2F&ei=RFrNU7r7Ja7UigK04oCwBg&usg=AFQjCNFg94OOl4Ydlcf909LVxBMg19ewiw&bvm=bv.71198958,d.cGE (accessed July 19, 2014).

Diamond, J.M., Koplish, D.E. McMahon J. III, and Rost R. 1997. Evaluation of the W\water-effect ratio procedure for metals in a riverine system. Environmental Toxicology and Chemistry 3:509–520.

Environment Canada. 2000. Biological test method: Reference method for determining acute lethality of effluents to rainbow trout. Report EPS 1/RM/13. Ottawa, ON.

Environment Canada. 2007 Biological test method: Test of reproduction and survival using the cladoceran *Ceriodaphnia dubia*. Report EPS1/RM/21 Ottawa, ON.

Johnson T. R., 1987. Water quality criteria for copper-Overview report. BC Ministry of Environment, Environmental Protection Division. http://www.env.gov.bc.ca/wat/wq/BCguidelines/copper/copper.html (accessed July 19, 2014).

Jop, K.M., Askew, A.M., Foster, R.B., 1995. Development of a water-effect ratio for copper, cadmium, and lead for the Great Works River in Maine using *Ceriodaphnia dubia* and *Salvelinus fontinalis.* Bulletin of Environmental Contamination and Toxicology 54: 29-35.

Meays, C., 2012. Derivation of water quality guidelines to protect aquatic life in British Columbia. 34 pp. http://www.env.gov.bc.ca/wat/wq/pdf/wq-derivation.pdf (accessed July 19, 2014).

Moss, S.A and N. Nagpal, 2003. Ambient water quality guidelines for boron: Overview. http://www.env.gov.bc.ca/wat/wq/BCguidelines/boron/boron.html (ccessed July 20, 2014).

Nautilus Environmental, 2009. Water effects ratio for Minto Mine–copper. 16 pp. http://www.yukonwaterboard.ca/registers/quartz/QZ09-094/Volume%202/1.4/2-1-4.pdf (accessed July 20, 2014).

TEMPORAL CHANGES OF FISH MERCURY CONCENTRATIONS IN MINING-AFFECTED PINCHI LAKE, BC.

R.F. Baker, M.Sc., R.P.Bio.
G.S. Mann, M.Sc., R.P. Bio.
P.J. Allard, M.Sc. R.P. Bio.

Azimuth Consulting Group Partnership
218-2902 W. Broadway Ave.
Vancouver B.C. V6K 2G8

ABSTRACT

The Pinchi Mine produced metallic mercury from 1940 to 1944 (historical) and from 1968 to 1975 (modern operation). From 2010 – 2012, the mine underwent decommissioning and reclamation to ensure that terrestrial areas affected by the mine do not pose unacceptable risks to ecological resources. Historical operations included placement of roasted ores (calcines) in the lake opposite the old mill. This resulted in highly elevated mercury concentrations in nearshore sediments. This source, as well as broad aerial deposition of elemental mercury during the roasting process in both operations, increased the mercury load to sediments throughout the lake. Prior to 2000, there were limited data on mercury in fish from Pinchi Lake. Detailed fish mercury studies were conducted in 2000, 2006 and 2011, focusing on mercury-size relationships of lake trout (*Salvelinus namaycush*), whitefish (*Coregonus* sp.) and rainbow trout (*Oncorhynchus mykiss*). While fish mercury concentrations have declined by nearly an order of magnitude from their peak in 1974, the rate of decrease in mercury concentrations has slowed since 2000. The focus of temporal comparisons is based on length-standardized fish size, because of the positive correlation between mercury concentration and size. Overall, mercury concentration in fish have declined from their peaks in the late 1970s (>5 ppm), but remain elevated in Pinchi relative to nearby Stuart and Tezzeron lakes in lake trout (0.53 ppm), whitefish (0.25 ppm) and rainbow trout (0.18 ppm). Given the longevity and size of lake trout and slow burial rate of historically contaminated sediments, the timeframe for recovery of the lake to 'regional' fish mercury concentrations is unknown. Risk assessments have evaluated potential implications of fish consumption on local wildlife species (eagle, grebe, otter), as well as on human health. Continued monitoring of fish mercury concentrations is part of the risk management plan for the site.

Key Words: Pinchi Mine, Mercury, Fish, Temporal Trends

INTRODUCTION AND BACKGROUND

This paper is one of two papers prepared in 2014 on closure activities at the Pinchi Mine. The other paper is entitled "A Risk Assessment / Risk Management Perspective on Mercury Contaminated Sediments in Mining Affected Pinchi Lake, BC, by G.S. Mann, R.F. Baker and P.J. Allard. (Mann et al. 2014).

Pinchi Mine is located in a heavily wooded wildland setting in central British Columbia (BC) on the north shore of Pinchi Lake, northwest of Fort St. James in the headwaters of the Fraser River system (**Figure 1**). Pinchi Lake is 25 km long and relatively narrow (1.2 km to 3.7 km) with a surface area of 55 km^2. The Ocock and Tsilcoh Rivers are the only named tributary streams entering Pinchi Lake. Pinchi Creek, situated at the southwest end of the lake is the only outflow from Pinchi Lake, discharging to Stuart Lake, and ultimately, the Fraser River.

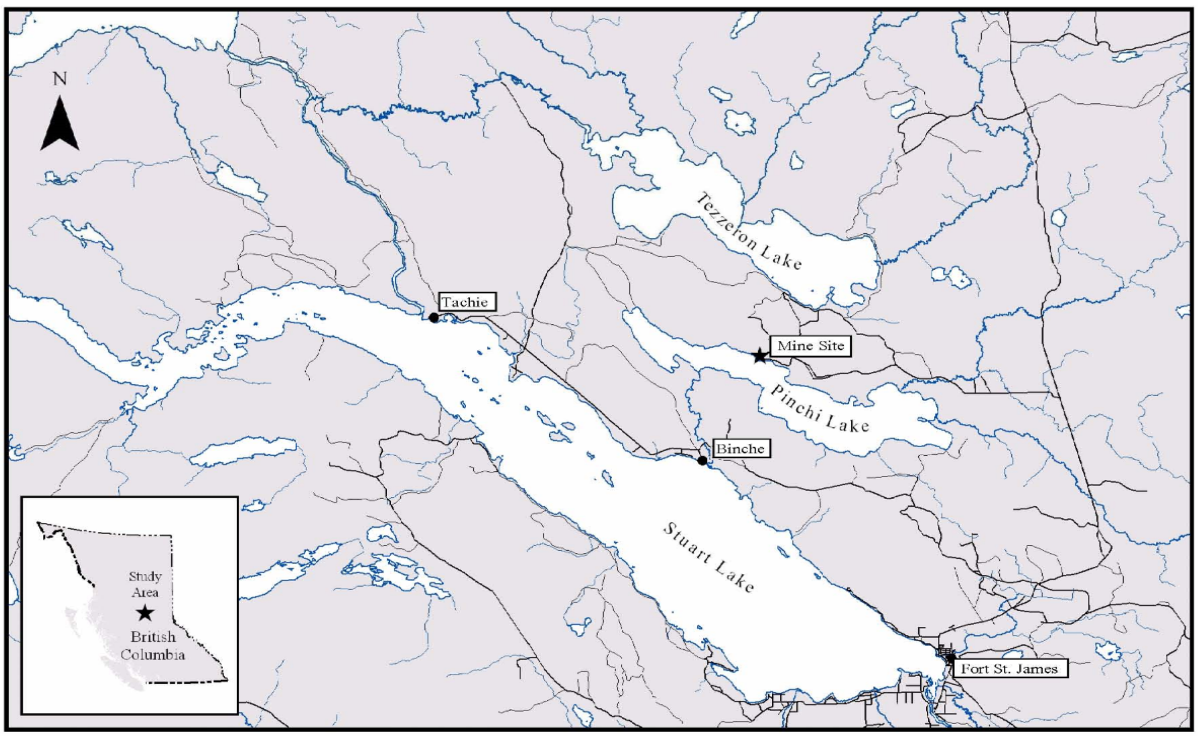


Figure 1. Map of Pinchi Lake region

The Pinchi fault region, located in central BC, is naturally enriched in mercury because of the abundance of cinnabar mineralization in bedrock along several portions of the fault (Plouffe 1995). Predecessor companies (now Teck Metals Ltd.) produced metallic mercury from the Pinchi Mercury Mine from 1940 to 1944 (historic operation) and from 1968 to 1975 (modern operation). During the historic operation, cinnabar ore was roasted without concentration and both the waste rock and the residue from roasting (calcine) were stockpiled or pushed into the lake opposite the old mill. During the modern operation, using a new mill, ore was concentrated and then processed to recover metallic mercury. Residues from this operation were slurried to a large on-site Tailings Impoundment Area. During both operations there was loss of elemental, atmospheric mercury to the environment during the roasting process.

Prior to remediation, the lake foreshore and small on-site ponds that gathered waste products from the mill had already been capped with clean fill and vegetated. Further details on mine operations can be found in Donald and Unger (2013) and in the Mine Closure Plan (Marsland Environmental Associates, 2009). After mining ceased in 1975, the Pinchi Mine was placed under long-term care and maintenance and has been the subject of numerous aquatic and terrestrial investigations. Aquatic studies focused on mercury and methylmercury in surface water, plankton, and sediment (EVS et al. 1999), sediment toxicity and benthic community health (Baker and Mann 2002) and fish (Baker et al. 2001; Azimuth 2008; Azimuth 2013).

Mercury has no known biological function and is a known toxicant to wildlife and to humans. Unlike other metals, mercury is accumulated in the tissues of animals at concentrations higher than in the environment (bioaccumulation) and these concentrations become progressively higher moving up the food chain from invertebrates, to peak in fish and fish-eating animals (biomagnification). Methylmercury, the dominant form of mercury that is present in fish (95%; Bloom 1992), is highest in species that consume other fish (e.g., lake trout, burbot) compared to those fish that eat primarily plankton (e.g., lake whitefish, kokanee). Methylmercury is acquired almost exclusively via dietary means (Hall et al. 1997) and exposure via other pathways such as through water, is very small.

Mercury is very persistent in the environment and can represent a potential human health risk in the event of long-term exposure to elevated concentrations. The process of methylation of inorganic mercury occurs principally in aquatic environments, in sediments of wetlands, marshes and lakes. In Pinchi Lake, the greater abundance of inorganic mercury has resulted in greater methylmercury production and bioaccumulation by fish. While fish is a very healthy and nutritious food source, consuming too many fish of certain species can pose a potential risk. However, as with any substance, the ‘dose’ determines the level of exposure. A human health risk assessment (Wilson 2010) did not identify health risks from consumption of most Pinchi Lake fish. The only exception was for large lake trout and burbot, where frequent and routine consumption would potentially pose a risk.

OBJECTIVES

Fish is the major source of methylmercury exposure to humans as well as to fish-eating wildlife such as mink, otter, eagles and other predators. The amount of mercury in the top predator fish, such as lake trout or burbot, is very much dependent on mercury concentrations in their diet, especially other fish species like whitefish, suckers, or kokanee. Consequently, the study design for fish mercury in more recent investigations conducted by Azimuth (2008, 2013) has focused on measuring meristics (length, weight, age), other life history information (growth, diet, maturity, gender) and stable isotopes, to better understand food web relationships from a wide range of fish species, from the top of the food chain to the bottom.

Key objectives of the 2006 (Azimuth 2008) and 2011 (Azimuth 2013) studies were as follows:

1. Characterize basic limnology, water and sediment quality of Pinchi Lake and place into context relative to two nearby lakes: Stuart and Tezzeron lakes.
2. Characterize key components of the Pinchi Lake aquatic ecosystem (phytoplankton, zooplankton, benthos and fish).
3. Determine mercury accumulation patterns and food web relationships using stable isotopes across fish species in Pinchi Lake and place into context with Stuart and Tezzeron lakes.
4. Determine historical changes and temporal trends in fish tissue mercury concentrations in Pinchi Lake.

This paper focuses on the last objective, how fish mercury concentrations in Pinchi Lake have changed over time. While there has been a significant reduction in mercury concentrations in Pinchi Lake fish since active mining ceased, concentrations remain elevated relative to nearby lakes; this paper speculates on the time course to recovery and puts fish mercury concentrations from Pinchi Lake in perspective other lakes with Tezzeron and Stuart lakes and other lakes in Canada (DePew et al. 2013).

METHODS

During fish mercury surveys conducted since 2000, mercury concentrations have been measured in up to 10 fish species from Pinchi, Stuart and Tezzeron lakes, including lake trout (*Salvelinus namaycush*), lake whitefish (*Coregonus clupeaformis*), mountain whitefish (*Prosopium williamsoni*), rainbow trout (*Oncorhynchus mykiss*), burbot (ling; *Lota lota*) and five other species. Prior to this, surveys were opportunistic and did not target specific species or investigate size-mercury relationships. In each of the 2000, 2006 and 2011 fish surveys (Azimuth 2013), fish were captured using short-set gill nets or by angling with a focus on three key species: lake trout a top level predator and a targeted sport fishing species; lake whitefish, benthic feeders and key food chain species; and rainbow trout, an insectivore and sport species. Tissue samples were harvested non-destructively from live lake and rainbow trout using biopsy tools (Baker et al. 2004) or destructively from whitefish and analysed for total mercury concentration by ALS Laboratories, Burnaby.

There is a well-known positive relationship between increasing mercury concentration and fish length (or weight or age) (Bodaly et al. 1984; Somers and Jackson 1993), as larger, older fish tend to have higher mercury concentrations than smaller, younger fish. This is partly due to differences in diet and the length of time of exposure. This positive relationship is typically seen for strongly carnivorous species (e.g., bull trout, lake trout, walleye) and sometimes for whitefish but seldom for suckers, forage fish or fish that consume terrestrial insects such as rainbow trout. Consequently, it is necessary to collect tissue from target species over a wide size range to determine the species-specific, size-mercury relationship. To accurately represent mercury concentrations across the size range typically observed, up to 30 – 35 fish are required, stratified (5 – 7 fish) within several discrete size intervals (e.g., 201 – 300 mm; 301 – 400 mm). Using linear regression on log-transformed data, a 'size-adjusted' mercury concentration is derived for lake trout (550 mm), lake whitefish (350 mm) and rainbow trout (300 mm) at a standardized length, specific to each species. The standardized size approximates the mean length of fish typically targeted by fishermen and allows for comparisons of mercury concentrations over time or between waterbodies for the same species that are unbiased by differences in fish size of the population sampled. Appropriate statistical methods are used (e.g., analysis of covariance, Tukey's test) to determine whether significant changes in mercury concentration have occurred over time, within or between lakes.

RESULTS AND DISCUSSION

The earliest documented collection of fish from Pinchi Lake for mercury analysis occurred in 1969 (Peterson et al. 1970), shortly after the Pinchi Mine resumed operations. Several further surveys were conducted but did not target specific species, use standardized methods, or attempt to determine size-mercury relationships or changes in fish mercury concentrations over time in a systematic way. In 2000, the first dedicated study of fish mercury concentrations in key fish species of Pinchi Lake using size-standardized techniques and analytical procedures was conducted (Baker et al. 2001). Following is a brief chronology of fish mercury investigations of Pinchi Lake.

- BC Fish and Wildlife (Peterson et al. 1970) surveyed heavy metals in sportfish from BC lakes in 1969/70, including several from the Pinchi Fault region. Lake trout from Pinchi Lake stood out, with elevated muscle tissue mercury concentrations, (> 4 mg/kg or ppm wet weight). These fish were collected three decades after cessation of historic operations and shortly after commencement of the modern operation.
- BC Fish and Wildlife (Reid and Morley 1975) followed up this study in 1974, several years later. The arithmetic mean mercury concentration of 11 lake trout was 5.2 ppm, with a concentration of >8 ppm in a large fish.
- BC Ministry of Environment (Martin et al. 1995) collected 16 lake trout in 1986, a decade after the termination of modern operations. They reported a substantial decrease in mean lake trout mercury tissue concentration (1.1 ppm).
- EVS (1996) collected 10 lake trout in 1995, approximately two decades after termination of mine operations. The arithmetic mean concentration was 0.96 ppm. The trend towards decreased mercury concentrations in lake trout that was observed between 1974 and 1986 was not detected in 1995.
- Baker et al. (2001) undertook a survey of fish mercury of six regional lakes in 2000 (Pinchi, Stuart, Tezzeron, Trembleur, Tchentlo, Francois) using biopsy techniques and a size-standardized approach. The objective was to understand spatial trends in fish mercury concentrations, explore potential "recovery concentrations" and to put the influence of Pinchi Fault geology into perspective. Pinchi Lake fish mercury concentrations were

elevated (~2-3x) relative to other Pinchi Fault lakes, which were not significantly more elevated than fish from Francois Lake, a reference lake.

- In 2006, Azimuth (2008) conducted a survey of Pinchi, Tezzeron and Stuart lakes to understand mercury dynamics and trophic relationships (based on stable isotope analysis), targeting phytoplankton, zooplankton, benthic invertebrates and fish. Similar to previous studies, tissue mercury concentrations in lake trout and other species from Pinchi Lake were elevated relative to Tezzeron and Stuart lakes. It was determined that elevated mercury tissue concentrations in Pinchi Lake are due to mine-related contamination and not to regional (i.e., Pinchi Fault background mercury) or ecological (i.e., longer food chain) factors. While lake trout continued the slow reductions observed since 1986, mercury concentrations in whitefish appeared to drop considerably. This was the first year that mercury in rainbow trout was examined. A positive size-mercury relationship for rainbow trout was established with a size-adjusted mean of 0.14 ppm for a 350 mm fish.
- A Pinchi-only fish survey was conducted in 2011 by Azimuth (2013) with a focus on mercury concentrations and stable isotopes in tissue samples from lake trout, whitefish and rainbow trout. Size-standardized mercury concentrations of lake trout were significantly lower, while mercury in lake whitefish was significantly higher than in 2006, the opposite trend as observed in 2006. Mean mercury in rainbow trout in 2011 (0.19 ppm) was slightly higher than in 2006.

This section describes our understanding of temporal changes in mercury concentrations of lake trout, lake whitefish and rainbow trout in Pinchi Lake since detailed, size-adjusted investigations of mercury concentrations began in 2000, in perspective with historic (pre-2000) studies. **Figure 2** illustrates all mercury (mg/kg ww or ppm) versus length (mm) data for lake trout (LKTR), lake (LKWH) and mountain whitefish (MNWH) and rainbow trout (RNTR) collected between 1974 and 2011. In 2006 there was a clear positive correlation between increasing fish size (length) and mercury concentration for lake trout and whitefish, as well as in rainbow trout – an atypical result for this species in most lakes. There has also been a clear reduction in mercury concentration in lake trout and lake whitefish over time, especially since 1974. There were too few data from 2006 and 2011 to make clear inferences for mountain whitefish and rainbow trout.

Although not illustrated here, mercury concentrations of these species from Pinchi Lake were significantly higher than their counterparts from Tezzeron and Stuart lakes, on a size-adjusted basis, from the most recent (2006) data (Azimuth 2008). Mercury concentration for a 550 mm lake trout from Pinchi Lake (0.91 ppm) was significantly higher ($p<0.01$) than mercury in lake trout from Stuart (0.39 ppm) and Tezzeron lakes (0.26 ppm). Mercury in 350 mm lake whitefish from Pinchi Lake (0.26 ppm) was also significantly higher than in whitefish from Stuart (0.10 ppm) and Tezzeron (0.09 ppm) lakes, although the magnitude of difference was lower. The highest mercury concentration measured in Pinchi Lake was from burbot with a concentration of 1.16 ppm for a 600 mm fish. This was much higher than for burbot from Tezzeron Lake (0.08 ppm) and Stuart Lake (0.17 ppm) on a size-adjusted basis.

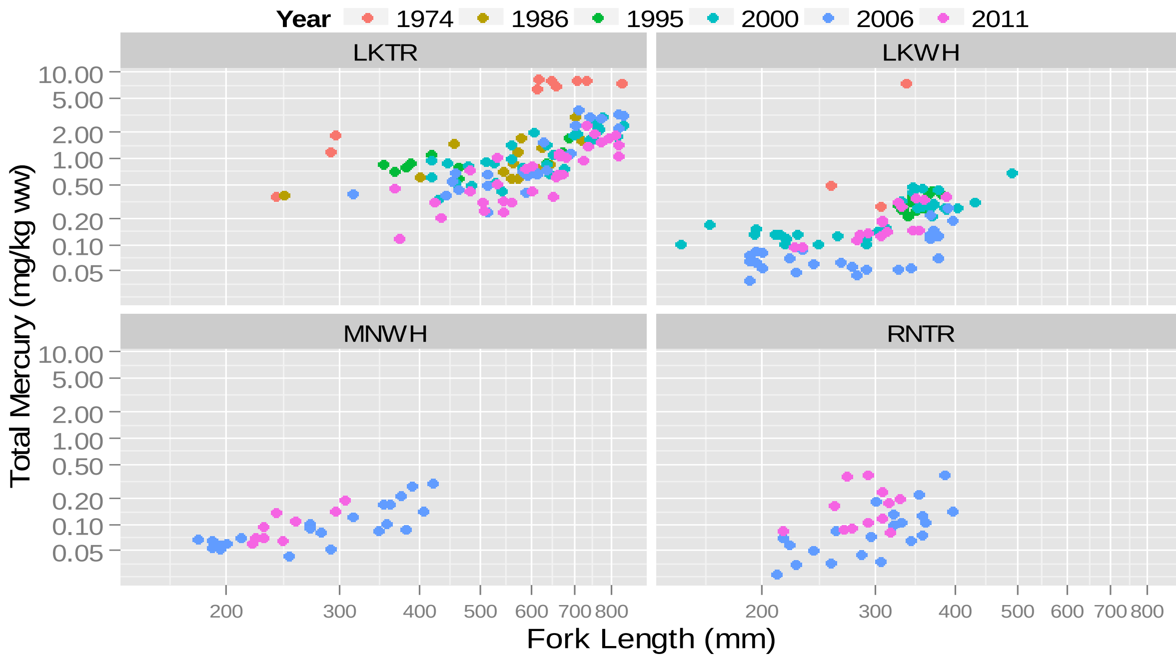


Figure 2. Log_{10} mercury (ppm) on length (mm) relationships for Pinchi Lake fish, 1974 - 2011.

Results from the most recent 2011 survey (Azimuth 2013) are illustrated in **Table 1** for the above species as well as kokanee, burbot, longnose sucker and white sucker. The relationship between increasing fish size and mercury concentration was only significant for lake trout and the whitefish species and not for rainbow trout, which was consistent with other years and lakes (Azimuth, 2008). White sucker mercury concentration also increased significantly with increasing fish size. This table also illustrates the size-adjusted mercury concentration for each species, based on the regression equations in this table. The mercury concentration for a 550 mm lake trout was 0.53 ppm, 0.25 ppm for a 350 mm lake whitefish and 0.19 ppm rainbow trout (independent of fish size).

Table 1. Fish mercury (Hg) on length, age, and weight regression results and standardized mercury concentrations for Pinchi Lake, 2011.

Species	Sample Size	R^2	p-Value	Significant Relationship	Mercury (Hg) on Length (L), Age (A), or Weight (W) Relationship	Standardized Measures[1] Length or Weight	[Mercury] (mg/kg ww)
Hg-Length Relationships						(mm)	
Lake trout	32	0.65	<0.001	yes	$Log_{10}Hg = -7.65 + 2.69 (Log_{10}L)$	550	0.53
Lake whitefish	16	0.62	<0.001	yes	$Log_{10}Hg = -6.94 + 2.49 (Log_{10}L)$	350	0.25
Mountain whitefish	9	0.68	0.006	yes	$Log_{10}Hg = -7.69 + 2.80 (Log_{10}L)$	300	0.18
Rainbow trout	12	0.07	0.412	no	$Log_{10}Hg = -3.87 + 1.24 (Log_{10}L)$	350	0.19
Kokanee	3	0.89	0.216	no	$Log_{10}Hg = -4.38 + 1.58 (Log_{10}L)$	250	0.26
Burbot	2	1	IS	IS	IS	600	IS
Longnose sucker	1	0	IS	IS	IS	350	IS
White sucker	10	0.55	0.014	yes	$Log_{10}Hg = -5.45 + 1.92 (Log_{10}L)$	350	0.27

1 Using standardized length; IS = insignificant relationship

Figure 3 and **Figure 4** illustrate all mercury-length data collected since 1974 for lake trout and lake whitefish respectively, but with statistically derived log-transformed regressions for 2000 (green line), 2006 (blue line) and 2011 (pink line) based on analysis of covariance, where a common slope between years was derived. For lake trout (**Figure 3**), the magnitude of difference in the intercept of the slopes between 2000 and 2006 was small and not significant ($p>0.05$). However, the 2006 distribution and size-standardized mean (0.53 ppm) was significantly lower than in 2000 (0.91 ppm) and 2006 (0.81 ppm) (**Table 2**).

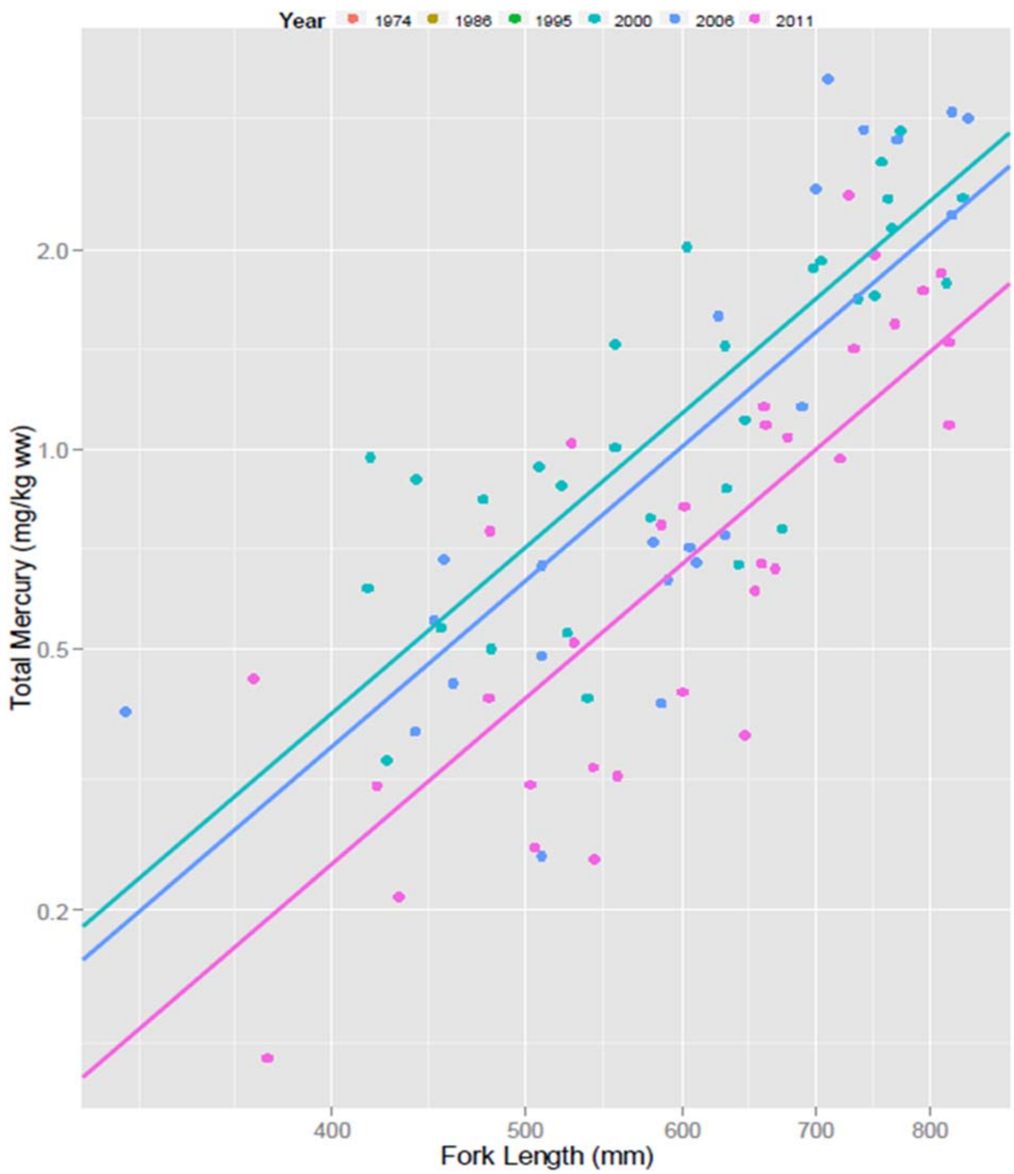


Figure 3. Lake trout ANCOVA-based (assuming equal slopes) on $\log_{10}$(mercury) vs $\log_{10}$(length) model for Pinchi Lake for 2000, 2006, and 2011.

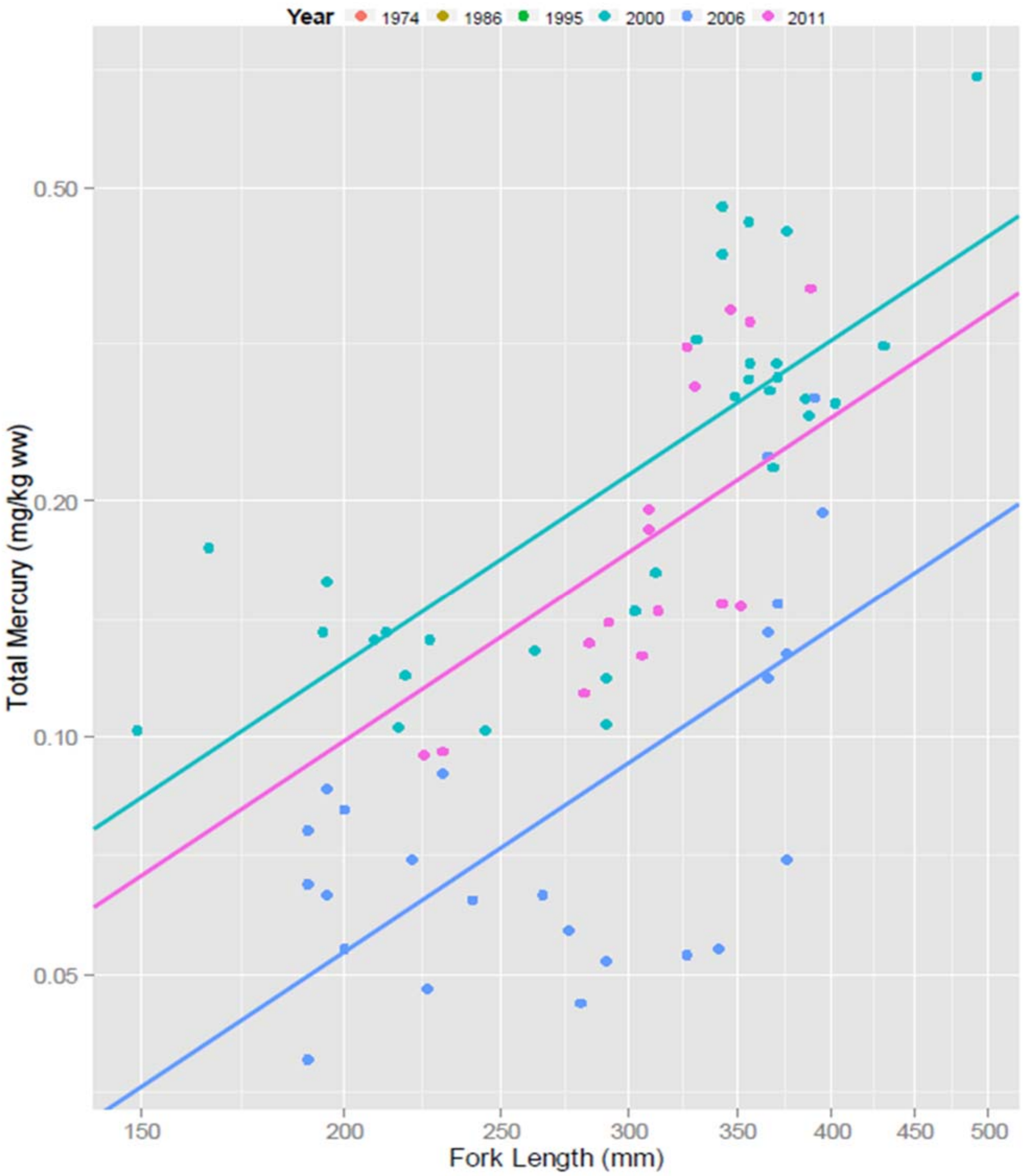


Figure 4. Lake whitefish ANCOVA-based (assuming equal slopes) on $\log_{10}$(mercury) vs $\log_{10}$(length) model for Pinchi Lake for 2000, 2006, and 2011.

The trend for lake whitefish was much more variable over time (**Table 2**). Size-standardized mean mercury concentration (350 mm) in 2000 was 0.26 ppm which was significantly higher than in 2006 (0.11 ppm) but not 2011 (0.25 ppm). The difference between 2006 and 2011 data might be related to small sample size in 2011, or other factors that might have contributed to a real increase in mercury in this species, as described below.

Table 2. Comparison of results of standardized tissue mercury (Hg) concentrations for lake trout and lake whitefish from Pinchi Lake.

Year	Sample Size	Mercury (Hg) on Length (L) Relationship $Log_{10}Hg = a + b\ (Log_{10}L)$	R^2	p-Value	Significant Relationship	Standardized Measures[1]	
						Length (mm)	[Mercury] (mg/kg ww)
Lake trout							
1974	9	$Log_{10}Hg = -4.25 + 1.81\ (Log_{10}L)$	0.92	<0.001	yes	550	5.13
1986	16	$Log_{10}Hg = -3.31 + 1.19\ (Log_{10}L)$	0.29	0.030	yes	550	0.89
1995	10	$Log_{10}Hg = -1.90 + 0.70\ (Log_{10}L)$	0.37	0.064	no	550	1.04
2000	31	$Log_{10}Hg = -6.04 + 2.19\ (Log_{10}L)$	0.62	<0.001	yes	550	0.91
2006	23	$Log_{10}Hg = -7.90 + 2.85\ (Log_{10}L)$	0.68	<0.001	yes	550	0.81
2011	32	$Log_{10}Hg = -7.65 + 2.69\ (Log_{10}L)$	0.65	<0.001	yes	550	0.53
Lake whitefish							
1986	6	$Log_{10}Hg = -7.99 + 2.99\ (Log_{10}L)$	0.79	0.018	yes	350	0.41
1995	12	$Log_{10}Hg = -8.47 + 3.13\ (Log_{10}L)$	0.49	0.011	yes	350	0.31
2000	32	$Log_{10}Hg = -4.09 + 1.38\ (Log_{10}L)$	0.60	<0.001	yes	350	0.26
2006	25	$Log_{10}Hg = -3.89 + 1.15\ (Log_{10}L)$	0.37	0.001	yes	350	0.11
2011	16	$Log_{10}Hg = -6.94 + 2.49\ (Log_{10}L)$	0.62	<0.001	yes	350	0.25

SUMMARY AND CONCLUSIONS

The historical trend observed, especially for lake trout, suggests a two-phased recovery response in fish mercury concentration. The initial phase, between 1974 and 1986 would have seen a significant reduction in mine-related mercury-containing discharges to the lake. Subsequent reductions in total and methylmercury concentrations in the water column would have likely occurred rapidly and cascaded up to phytoplankton and zooplankton, as well as to any fish feeding on these organisms (e.g., lake whitefish).

The secondary phase, which continues today, is characterized by more modest reductions in tissue mercury concentrations likely due to population attrition and deposition of mercury into deeper sediments (EVS et al. 1999; Mann et al. 2014). Over the decades following mine shutdown, older, larger lake trout would have naturally died and been replaced in the population by fish exposed to progressively lower mercury during their lifetime. Concurrently, sediment coring programs (EVS et al. 1999) documented a slow burial process where deeper sediments generally act as a mercury sink. The slow reductions in mercury concentrations in biologically active surface sediments would drive lower tissue mercury concentrations in benthic organisms, which would gradually be reflected in the food chain (e.g., to juvenile lake trout and other fish relying heavily on aquatic insect larvae and ultimately to adult lake trout).

The lack of significant inputs of relatively cleaner sediments to Pinchi Lake (i.e., no large tributaries) is probably the greatest rate-limiting factor now affecting the secondary phase. However, the reduction in tissue mercury concentrations observed in smaller lake trout between 2000 and 2006 foreshadowed a future trend in the larger lake trout. The apparent drop in zooplankton mercury concentrations between 1997 and 2006 also supports this hypothesis. While these results suggest continued, but slow, reductions in the amount of mercury actively cycling through the ecosystem, verification of this hypothesis will require continued monitoring.

The 2011 results for both lake trout and lake whitefish were unexpected. For lake trout, the magnitude of the reduction in tissue concentrations relative to 2006 was greater than expected, even considering the foreshadowing we speculated about. One possible difference between the 2011 and 2006 studies was the use of non-lethal biopsy sampling procedures for total metals in 2011 (i.e., not just limited to mercury). However, this method has been shown to be accurate (Baker et al., 2004) and there was no apparent bias in the 2011 data due to sampling type. Lake whitefish on the other hand (based on a sample size of only 16 fish) showed the opposite pattern, with a reasonably large increase in tissue mercury concentrations in 2011 relative to 2006. These results suggest that the small sample size of fish did not represent the true underlying population, there are spatial differences within the lake that confound results, and/or that there was a real increase in mercury for this species, such as can be caused by logging and forest fires (Garcia and Carnignan 2000), activities that have been common in this area recently. Logging, fires and death of forests from pine beetle are known to cause soils to erode and become deposited in lakes as sediment, transporting organic material and mercury into lakes, where it is methylated. As well, readers should be cautious about over-interpreting the results of a single survey and should instead focus on broader temporal trends.

Notwithstanding the uncertainty in some data driven by small and uneven sample size distribution between years, mercury concentrations in key fish species in Pinchi Lake have been slowly declining since mining ceased in 1976. Despite the fact that mercury concentrations in fish are elevated relative to other BC lakes and reservoirs (Baker 1999), these concentrations are similar to what is observed in many hundreds of lakes elsewhere in Canada (DePew et al. 2014). This is likely why ecological effects to local fish-eating wildlife species have not been observed. For example, Weech et al. (2006) found that reproductive success and average productivity of bald eagles over the 3-year period (2000 – 2002) were 62% and 0.98 chicks/territory on Pinchi Lake compared to 64% and 1.17 chicks/territory on all other study lakes combined. While exposure to mercury was higher on Pinchi Lake (blood, feathers, eggs), birds were in excellent condition and successfully raised eaglets. An ecological risk assessment conducted by Azimuth (2009) did not predict unacceptable risks to fish-eating wildlife such as loons, grebes and otter.

In summary, given the strongly bioaccumulative nature of methylmercury, long lifespan (>30 y) and large size of lake trout, and slow deposition and burial rate of historically contaminated lake sediments (Mann et al. 2014), it is expected that recovery of the fish mercury concentrations in Pinchi Lake to ‘regional’ (i.e., lakes in close proximity to the mercury-rich Pinchi Fault) fish mercury concentrations, may take several decades, but may also remain higher than regional lakes, given possible influence of the Pinchi Fault and lake-wide elevations in mercury concentrations from historic mining operations (Mann et al. 2014).

REFERENCES

Azimuth Consulting Group (Azimuth). 2006 Regional fish mercury and Pinchi Lake ecology report. Prepared for Teck Cominco Metals, Kimberley BC by Azimuth Consulting Group Inc., Vancouver BC. June 2008.

Azimuth. 2009. Pinchi Mine Ecological Risk Assessment. Prepared for: Teck Metals Ltd. Kimberley BC by Azimuth Consulting Group, Vancouver. June 2009.

Azimuth. 2013. Pinchi Lake Mine Fish Mercury Study 2011. A report prepared for Teck Metals Ltd. Kimberley BC by Azimuth Consulting Group Partnership, Vancouver BC. March 2013. 112 p.

Baker, R.F. and G.S. Mann. 2002. Assessment of Sediment Mercury Contamination, Pinchi Lake, BC. Report prepared by EVS Environment Consultants and Azimuth Consulting Group for Teck Cominco Metals Ltd. November 2002.

Baker, R.F., F. Landry and D. Gass. 2001. Regional Survey of fish mercury concentrations, Pinchi Lake, BC. Report prepared by EVS Environment Consultants for Cominco Ltd. April 2001.

Baker, R.F., P.A. Blanchfield, M.J. Paterson, R.J. Fudge and L. Wesson. 2004. Evaluation of non-lethal methods for the analysis of mercury in fish tissue. Trans. Am. Fish. Soc. 133: 568 – 576.Bloom, N. 1992. On the chemical form of mercury in edible fish and marine invertebrate tissue. CJFAS 49:1010 – 1017.

Bodaly, R.A., R.E. Hecky and R. Fudge. 1984. Increases in fish mercury levels in lakes flooded by the Churchill River Diversion, northern Manitoba. Can. J. Fish. Aquat. Sci. 41: 682 – 691.

Depew, D.C., N.M. Burgess, M., Anderson, R.F. Baker, S.P. Bhavsar, R.A. Bodaly, C.S. Eckley, M.S. Evans, N. Gantner, J.A. Graydon, K. Jacobs, J.E. LeBlanc, V.L. St. Louis, and L.M. Campbell. 2013. An overview of mercury concentrations in freshwater fish species: A national fish mercury dataset for Canada. Canadian Journal of Fisheries and Aquatic Sciences 70: 436 – 451.

Donald, B., M. Unger and R. Marsland. 2013. Remediation of the Pinchi Lake Mine. Teck Metals Ltd. Kimberley BC and Marsland Environmental. Prepared for the Technical and Research Committee on Reclamation, Kamloops BC. September, 2013.

EVS, NDM, and Frontier Geosciences (EVS Environment Consultants, Norecol Dames and Moore, and Frontier Geosciences). 1999. Pinchi Lake Mine: Mine site and Lake Investigation –

Environmental Assessment Report. EVS Environment Consultants, Norecol Dames and Moore, and Frontier Geosciences for Cominco Ltd. Vancouver BC.

EVS Environment Consultants. 1996. 1995 Pinchi Lake receiving environment investigation. Prepared by EVS Environment Consultants, North Vancouver for Cominco Ltd., Vancouver, BC. January 1996. 23 pp.

Garcia, E. and R. Carnignan. 2000. Mercury concentrations in northern pike (*Esox lucius*) from boreal lakes with logged, burned, or undisturbed catchments. Canadian Journal of Fisheries and Aquatic Sciences, 2000 57: 129 – 135.

Hall, B.D., R.A., Bodaly, R.J.P. Fudge, J.W.M. Rudd and D.M. Rosenberg. 1997. Food as the dominant pathway of methylmercury uptake by fish. Water, Air & Soil Poll. 100:13 – 24.

Marsland Environmental Associates. 2009. Pinchi Lake Decommissioning and Reclamation Plan. Prepared for Teck Cominco Metals Ltd. by MEA. April 2009.

Martin, J.W., W.F.A. Duncan and M.N.M. Masselink. 1995. Mercury concentrations in fish tissue and sediment from Pinchi Lake, BC. Prepared by BC Ministry of Environment, Environmental Protection Branch, Prince George, BC. *Unpublished*.

Mann, G.S., R.F. Baker and P.J. Allard. 2014. A risk assessment / risk management perspective on mercury contaminated sediments in mining affected Pinchi Lake BC. Prepared for the Technical and Research Committee on Reclamation, Prince George BC. September 22 – 24, 2014.

Peterson, G.R., H.V. Warren, R.E. Delavault and K. Fletcher. 1970. Heavy metal content of some fresh water fishes in BC. Fisheries Technical Circular. No. 2. British Columbia Fish and Wildlife Branch. 33 pp.

Plouffe, A. 1995. Glacial dispersal of mercury from bedrock mineralization along Pinchi Fault, north central British Columbia. Water, Air and Soil Poll. 80: 1109 – 1112.

Reid D.S. and R.L. Morley. 1975. Mercury contamination of fish from Pinchi Lake. BC Habitat Protection Service, Fish and Wildlife Branch, Department of Recreation and Conservation, Victoria BC.

Somers, K.M. and D.A. Jackson. 1993. Adjusting mercury concentration for fish-size covariation: A multivariate alternative to bivariate regression. CJFAS 59: 2388 – 2396.

Weech, S.A., A.M. Scheuhammer and J.E. Elliot. 2006. Mercury exposure and reproduction in fish-eating birds in the Pinchi Lake region, British Columbia, Canada. Environ. Toxicol. and Chemistry 25: 1433 – 1440.

Wilson, R. 2014. Human Health Risk Assessment of Pinchi Mercury Mine. Prepared by Wilson Scientific Ltd. Vancouver BC for Teck Metals Ltd., Kimberley BC. February 2010. 210 p.

A RISK ASSESSMENT / RISK MANAGEMENT PERSPECTIVE ON MERCURY CONTAMINATED SEDIMENTS IN MINING AFFECTED PINCHI LAKE, BC.

G.S. Mann, M.Sc. R.P.Bio., R.F. Baker, M.Sc., R.P. Bio., P.J. Allard, M.Sc., R.P. Bio.

Azimuth Consulting Group Partnership
218-2902 West Broadway
Vancouver, BC V6K 2G8

ABSTRACT

The Pinchi Mine, located on the merciferous Pinchi Fault region of BC, produced metallic mercury from 1940 to 1944 (historical operation) and from 1968 to 1975 (modern operation). Between 2010 and 2012, the mine underwent decommissioning and reclamation to ensure that the terrestrial areas affected by the mine do not pose unacceptable risks to ecological resources. The historical operations included placement of roasted ores (calcines) in Pinchi Lake adjacent to the site, resulting in highly elevated mercury concentrations in nearshore sediments (subsurface calcines). This source, as well as aerial deposition of elemental Hg during the roasting process during both operations, broadly elevated sediment mercury concentrations throughout the lake. In 1997, inorganic and methylmercury concentrations were measured in pore water and sediment at different depths in sediment cores and showed that subsurface calcine sediment was a significant contributor of inorganic mercury to the lake, notwithstanding slow burial by cleaner sediments. In 2001, a sediment quality triad (chemistry, toxicity, benthos) study showed no correlation between sediment inorganic mercury concentration, toxicity or benthic community structure. However, benthic organisms living in subsurface calcine sediment were elevated in total and methylmercury concentrations relative to benthos elsewhere in the lake. This paper reviews and integrates historical sediment investigations using a risk assessment and risk management framework to guide further investigations and support long-term decision-making regarding Pinchi Lake sediment.

KEY WORDS
Mercury, methylmercury, sediment quality triad, Pinchi Mine, Pinchi Lake, risk assessment, risk management.

OVERVIEW

The Pinchi Mine, located on the merciferous Pinchi Fault region of BC (Figure 1), produced metallic mercury from 1940 to 1944 (historical operation) and from 1968 to 1975 (modern operation). The historical operations included deposition of roasted ores (calcines) into Pinchi Lake adjacent to the site, resulting in highly elevated mercury concentrations in nearshore sediments (subsurface calcines) and beyond. A series of investigations have been conducted over the years to characterize the environmental significance of the subsurface calcine area and Pinchi Lake in general. This paper[1] summarizes key findings of previous investigations relevant to the subsurface calcine area and presents the results within a risk assessment/risk management framework to support informed management.

---

[1] Note that a companion paper by Baker et al. in these proceedings addresses temporal changes in fish mercury concentrations in Pinchi Lake.

Figure 1. Pinchi Mine Site, BC, with inset showing general location of subsurface calcine sediments in Pinchi Lake.

BACKGROUND

Mercury (inorganic and organic [methylmercury]) are the main contaminants of potential concern (COPC) for this site. Methylmercury (MHg) is important because this is the most toxic form of mercury. Although it normally comprises only a small percentage of the total mercury in environmental media, it is readily taken up by organisms, biomagnifies up the food chain, and can be toxic at low concentrations. Total mercury (THg; sum of inorganic and organic mercury species) is the most commonly measured mercury analysis. Secondary COPCs in subsurface calcine sediments were arsenic, antimony, chromium, and nickel. The location of the subsurface calcine area is shown in Figure 1 (lower left inset) relative to the upland portion of the site.

EARLIER STUDIES

Advancements in the late 1960s in the understanding of mercury's toxicity in aquatic systems prompted the first sampling at Pinchi Lake. Peterson et al. (1970) conducted a regional survey of heavy metals in sport fish in lakes along the Pinchi Fault in 1970. Lake trout from Pinchi Lake contained dramatically elevated mercury concentrations (over 4 mg/kg wet weight). Monitoring conducted over the next four decades showed substantial decreases through 2000, then more modest reductions in fish tissue mercury concentrations since then (see Baker et al. 2014 for details).

The first quantitative survey of mercury in sediment in Pinchi Lake took place in 1976-1978. Ableson and Gustavson (1979) measured total mercury concentrations in surface sediments offshore of the mine site to determine the spatial extent of mercury in subsurface calcine sediments. Concentrations were highest directly offshore of the mine site (i.e., subsurface calcine area) and diminished east and west of this peak zone with increasing depth and distance offshore. Cores collected in 1978 showed maximum mercury concentrations occurring in the top 5 cm. Martin et al. (1995 Draft Report) collected sediment cores again in 1986, showing lower concentrations in both the 0 to 5 cm and 5 to 10 cm horizons.

LATER STUDIES

From 1995 through 2001, Teck Cominco commissioned detailed investigations to examine the mine site and its relationship with Pinchi Lake with respect to environmental contamination (primarily mercury, but also other metals) in lake water, tributary streams, sediment, pore water, groundwater, lower trophic level biota (i.e., zooplankton and benthos), and fish (EVS 1996, EVS et al. 1999a, b, EVS 2001, EVS and Azimuth 2002). Subsequently, follow-up monitoring was conducted in 2006 (fish and lake ecology in Pinchi, Tezzeron and Stuart Lakes; Azimuth 2008) and 2011 (fish in Pinchi Lake; Azimuth 2013) to (1) continue tracking temporal trends in fish mercury concentrations in Pinchi Lake and (2) serve as the "benchmark" for describing the basic trophic structure of Pinchi Lake to help interpret any future changes in fish mercury concentrations. This work was conducted by a multidisciplinary team that included mercury specialists. Key results from these investigations relevant to assessing the environmental significance of contaminated sediments in the subsurface calcine area (both locally and to the lake as a whole) are described within a risk-based framework.

CONCEPTUAL MODEL/RISK ASSESSMENT

Conceptual models are commonly used in risk assessments to depict important contaminant-related processes such as sources, release mechanisms, transport pathways, exposure routes, and receptors. While a formal risk assessment, *per se*, of mine-related contamination in Pinchi Lake has not been conducted, the aforementioned environmental investigations characterizing conditions in Pinchi Lake (including the subsurface calcines) were guided by a systematic risk-based approach that ultimately integrated information pertaining to the physical, chemical, biological, ecological, and toxicological nature of the site to help inform management decisions (Figure 2).

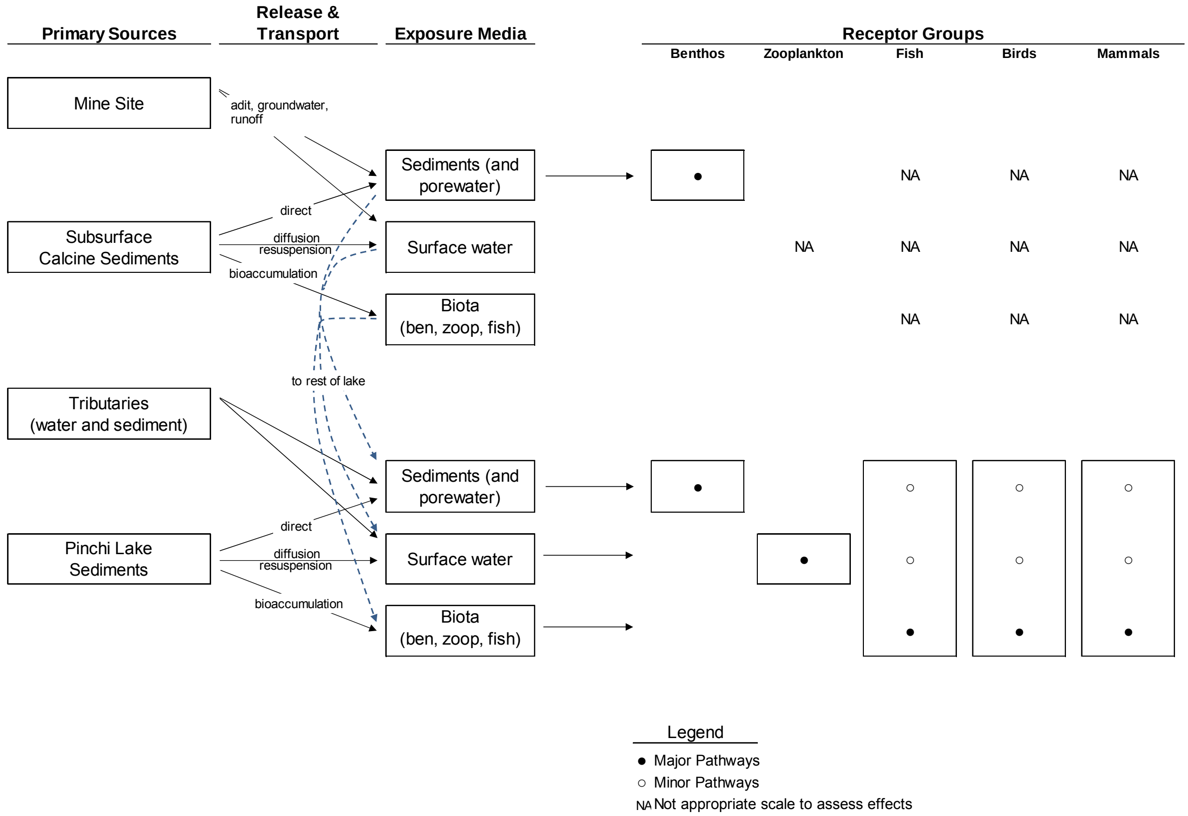


Figure 2. Conceptual model showing subsurface calcine sediments relative to other mercury sources to Pinchi Lake receptors.

Available data for key aspects of the conceptual model are summarized below (with emphasis on subsurface calcines and lake sediments; the relative importance of all sources to Pinchi Lake is discussed in the last bullet below):

- *Primary Sources* – Surface sediment mercury[2] concentrations (Figure 3) are elevated throughout Pinchi Lake, but particularly so in the subsurface calcine area. Sediment core data from 1997 (Figure 4) shows the strong signal of the historical operations (e.g., as seen in the peaks in the deep portions of the east and west basins; dating confirmed with lead isotopes) and more recent deposition of cleaner sediments at the surface; the recovery pattern is weaker in other portions of the lake, including the subsurface calcines (possibly due to higher biologically-induced mixing). Total mercury and methylmercury concentrations in sediments and porewater were substantially higher in the subsurface calcines than the rest of the lake. Core data show that the thickness of the subsurface calcines is approximately 10 cm.

[2] Secondary contaminants present in the calcines include arsenic and antimony, among others; the magnitude of contamination is far less for these metals.

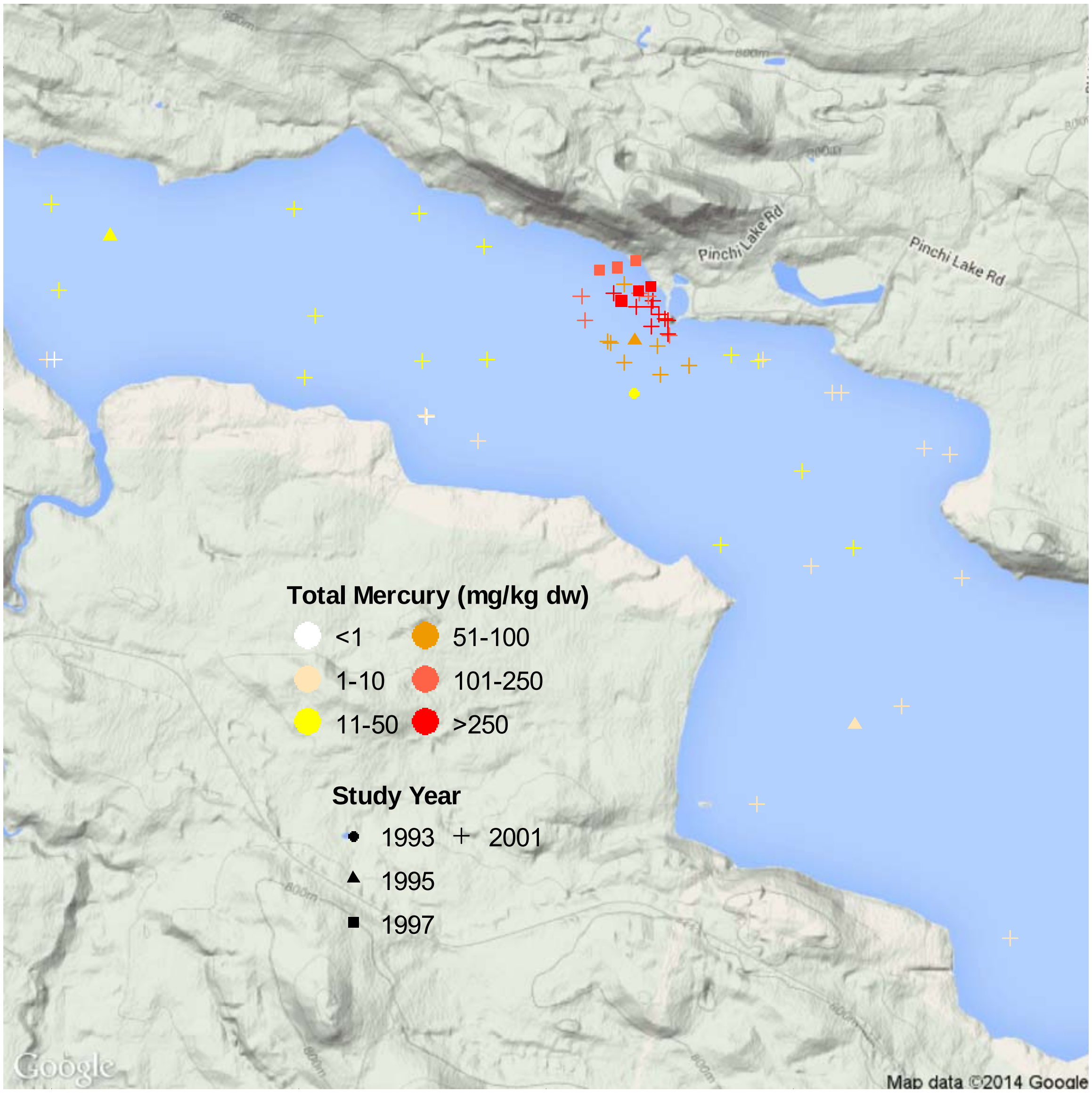


Figure 3. Horizontal extent of total mercury in surface sediments based on compilation of data.

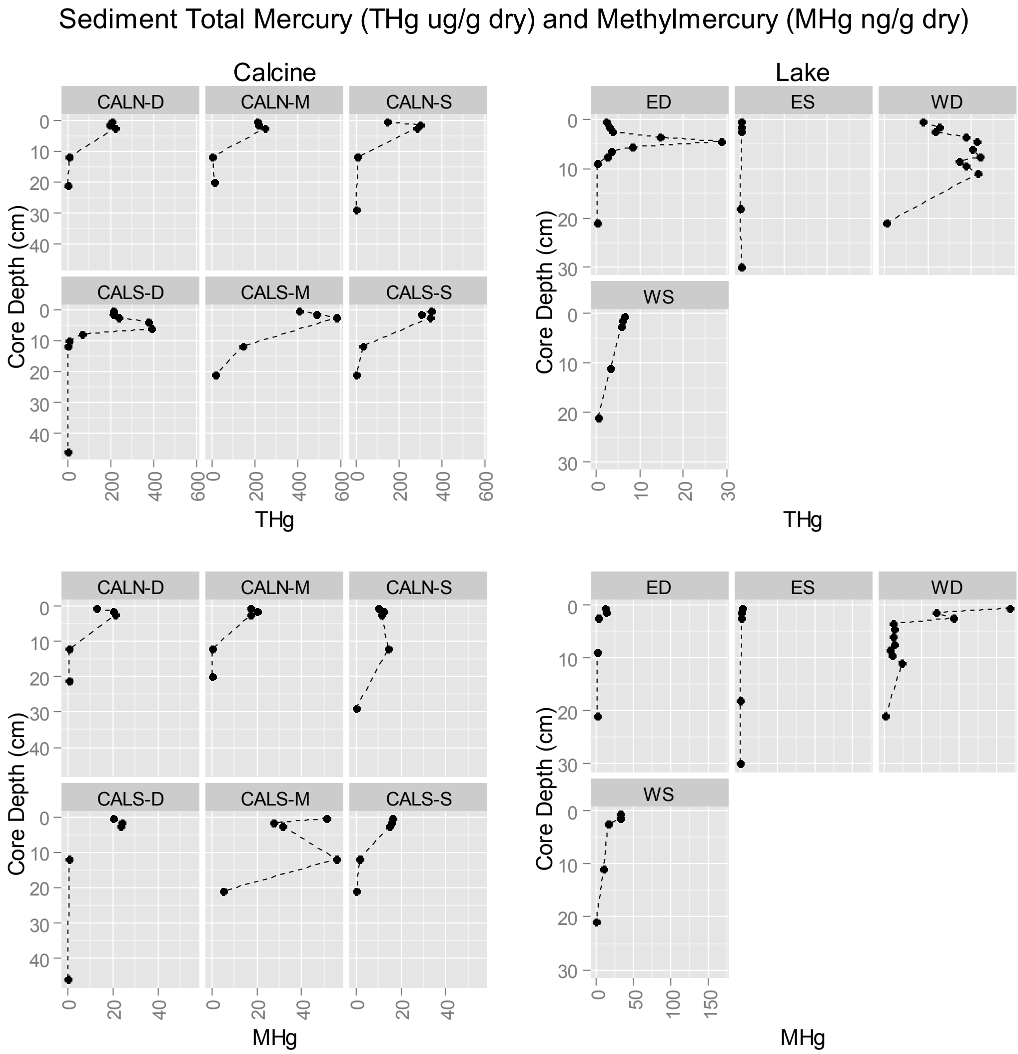


Figure 4. Vertical extent of total (top) and methylmercury (bottom) in subsurface calcines (CAL north [N] and south [S] transects; D = deep, M = medium, S = shallow) and lake sediments (East Basin deep [ES] and shallow [ES]; West Basin deep [WD] and shallow [WS]) from 1997 study.

- *Release/Transport and Exposure Media* – Three primary pathways are present from sediments: direct contact by biota with contaminants in sediment and/or porewater, transport via diffusion/re-suspension to the water column and biota bioaccumulation. Direct exposure is higher in the subsurface calcine area (see *Primary Sources* above) relative to the rest of lake. Diffusion modelling based on porewater gradients (from the 1997 sediment cores) close to the sediment/water interface showed the subsurface calcines as a source for total mercury, but not for methylmercury (see below for more discussion). Surface water mercury concentrations (Figure 5) were variable, but were generally higher in unfiltered samples from the subsurface calcine area. Biota tissue concentrations (Figure 6) showed consistently higher total and methylmercury in sediment-associated biota (i.e., benthos [chironomids] and bivalve mollusks) in the subsurface calcines; mercury concentrations in zooplankton were much lower than the other biota and patterns more variable.

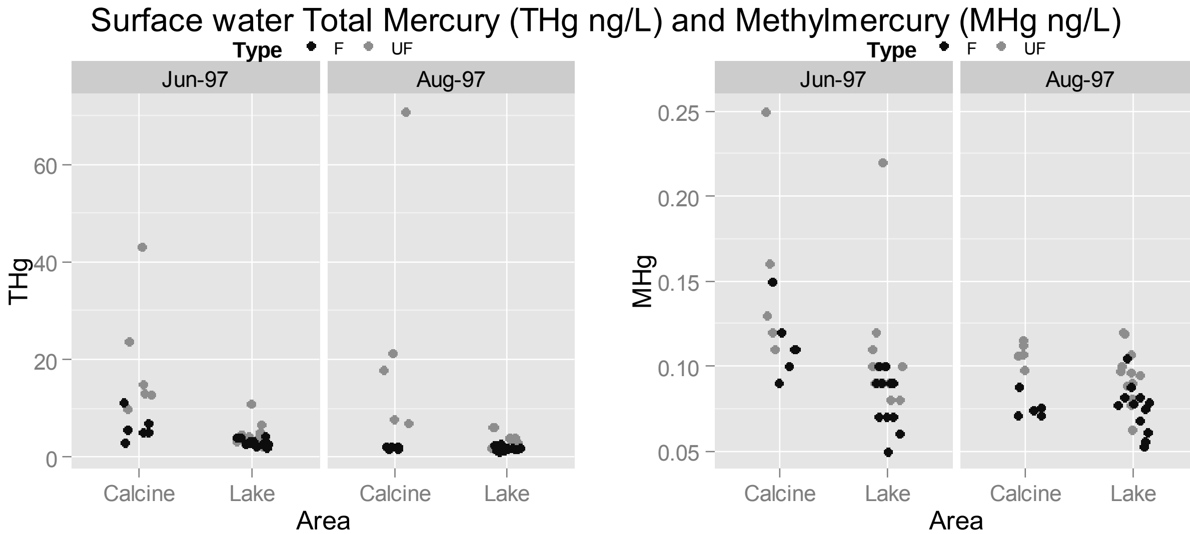


Figure 5. Surface water total and methylmercury concentrations for filtered (F) and unfiltered (UF) water collected above the subsurface calcine area and other portions of Pinchi Lake in June and August 1997.

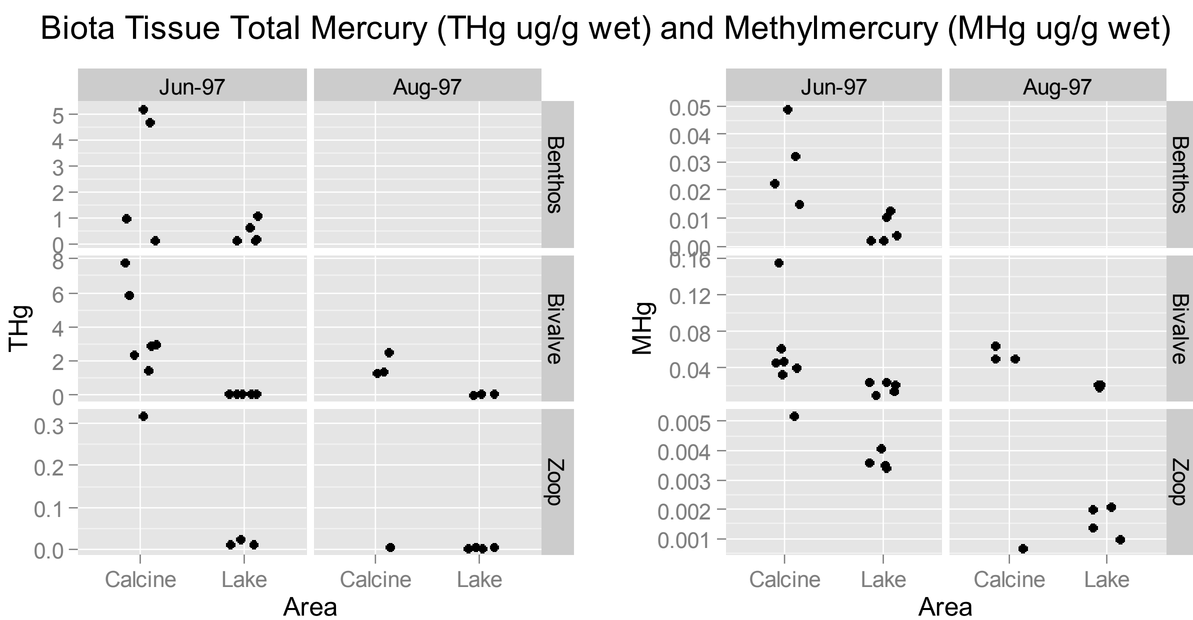
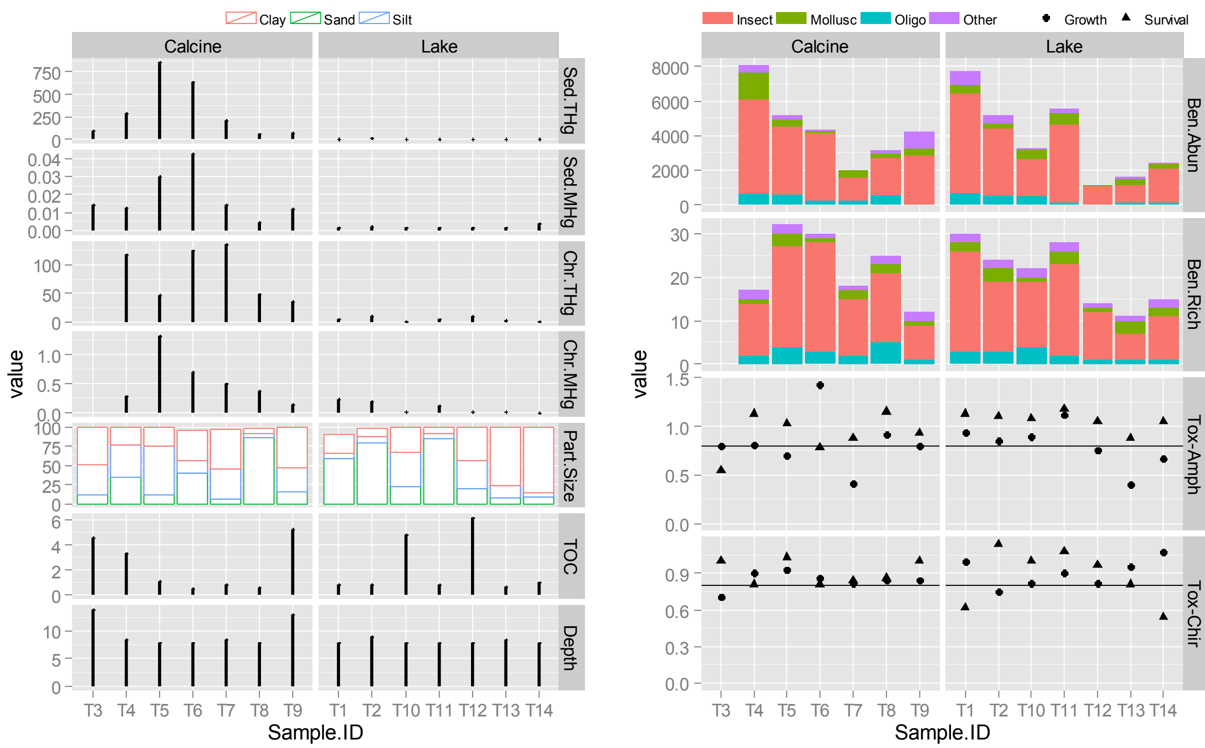


Figure 6. Biota tissue total and methylmercury concentrations samples collected in the subsurface calcine area and other portions of Pinchi Lake in June and August 1997.

- *Receptors* – Spatial scale is important is making inferences regarding effects to receptors. Apart from benthic invertebrates, which live directly in the sediment and are relatively immobile, the whole lake is a more ecologically relevant spatial scale for assessing effects than the subsurface calcine area for the other receptors. However, the relative importance of the subsurface calcine area to overall exposure can still be estimated for these receptors (e.g., fish, birds and mammals; see last bullet in this section for more discussion). Results are discussed by receptor group:

- *Benthos* – The potential for direct effects and bioaccumulation by benthos in the subsurface calcines was investigated in 2001 (EVS and Azimuth 2002). Benthic community (abundance and richness), sediment toxicity (midge and amphipod survival and growth), bioaccumulation (midge tissue concentrations), and sediment chemistry was assessed at 14 stations spanning a wide exposure gradient within and outside the subsurface calcine area. While exposure metrics (Figure 7 [left]) showed substantial differences between areas, the pattern was not evident in any response metric (Figure 7 [right]). Correlations between exposure and response metrics (Table 1) point to sediment grain size explaining the observed differences in response metrics. Thus, despite a clear gradient in mercury exposure, the most obvious patterns in benthos were related to physical conditions.



Note: Sed.THg = sediment total mercury (ug/g dw); Sed.MHg = sediment methylmercury (ug/g dw); Chr.THg = chironomid tissue total mercury (ug/g dw); Chr.MHg = chironomid tissue methylmercury (ug/g dw); Part.Size = particle size (%); TOC = total organic carbon (%); Depth (m); Ben.Abun = benthos total abundance (#/m^2); Ben.Rich = benthos taxa richness (# taxa/grab); Tox-Amph = amphipod toxicity (% of control); Tox-Chir = chironomids toxicity (% of control); the two toxicity plots show the 20% response (i.e., 0.8 of control) line.

Figure 7. Exposure (left) and response (right) metrics from 2001 sediment toxicity/bioaccumulation study of the subsurface calcine area at the Pinchi Mine Site, BC.

Table 1. Correlations (Spearman) between exposure and response metrics from 2001 sediment toxicity/bioaccumulation study of the subsurface calcine area at the Pinchi Mine Site, BC.

	Benthic Community		Sediment Toxicity			
	Total	Total	*Chironomus*		*Hyalella*	
	Abundance	Richness	Survival	Growth	Survival	Growth
Sediment Mercury						
Total Mercury	0.434	0.49	0.113	-0.091	-0.179	0.176
Methylmercury	0.044	0.187	0.003	-0.097	-0.45	-0.11
Tissue (Chironomid) Mercury						
Total Mercury	0.148	0.248	0.047	-0.299	-0.234	0.077
Methylmercury	0.407	**0.682**	0.116	-0.133	-0.105	0.286
Other Sediment COPCs						
Antimony	0.024	0.066	0.003	-0.151	-0.51	-0.275
Arsenic	-0.077	-0.165	-0.366	0.144	**-0.677**	-0.506
Chromium	0.16	0.03	0.329	-0.186	-0.123	-0.146
Nickel	0.016	-0.179	0.304	-0.227	-0.24	-0.352
Biophysical						
Sand	**0.571**	0.525	0.288	-0.172	**0.739**	**0.879**
Silt	0.011	-0.017	0.321	-0.346	-0.353	-0.253
Clay	**-0.637**	**-0.63**	-0.437	0.211	**-0.781**	**-0.676**
TOC	-0.116	-0.42	0.194	-0.236	-0.029	-0.377
Depth.m	0.038	-0.534	0.176	-0.315	-0.258	-0.388

Bold values indicate statistically significant spearman correlations (critical rs = 0.560 for $\alpha[2]=0.05$, $p<0.05$, $n=13$).

- *Zooplankton* – from a direct toxicity perspective, mercury concentrations in surface waters in Pinchi Lake are typically much lower than guidelines developed for direct toxicity (e.g., CCME's water quality guideline of 26 ng/L for total and 4 ng/L for methylmercury), so no effects would be expected. This conclusion is corroborated by the 2006 lake ecology study (Azimuth 2008), which documented a community typical of boreal lakes.
- *Fish* – see Baker et al. (this issue) for detailed discussion of fish mercury concentrations; fish populations appear healthy and there are no systematic differences compared to Stuart and Tezzeron Lakes.
- *Birds and mammals* – two main lines of evidence have been used to assess wildlife: food chain modelling and direct ecological studies. The Pinchi Mine Site ecological risk assessment (Azimuth 2009) concluded that predicted exposures to birds and mammals feeding in Pinchi Lake did not pose unacceptable risks. For birds, this result is consistent with a field study on fish-eating birds documented higher mercury concentrations in Pinchi Lake birds, but showed no apparent effects to egg production, fledging success or

growth (Weech et al. 2006; see Baker et al. 2014 for more detail). A similar study is underway for river otter.

- *Relative Importance of Mercury Sources to Pinchi Lake* – One of the goals of the 1996-2001 studies was to quantify the relative contributions of a range of sources to Pinchi Lake. The 1997 data was used to estimate contributions from each major source to Pinchi Lake water using a mass-balance approach (Figure 8 [left]). Mine Site sources (consisting of contributions from the 750 Adit and groundwater inputs through the foreshore calcines and former lagoons) were low for both total and methylmercury. While the subsurface calcine area was estimated to contribute 25% of total mercury loadings to Pinchi Lake water, it was not an important source for methylmercury. The 1997 and 2001 chironomid (midge larvae) tissue methylmercury concentrations and the spatial boundaries of the various sampling locations were used to estimate the relative contributions to the Pinchi Lake benthic-based food chain (Figure 8 [right]). The deep and shallow zones of the subsurface calcine area combined to contribute 5% of the methylmercury loading associated with chironomids in the benthic food chain of Pinchi Lake.

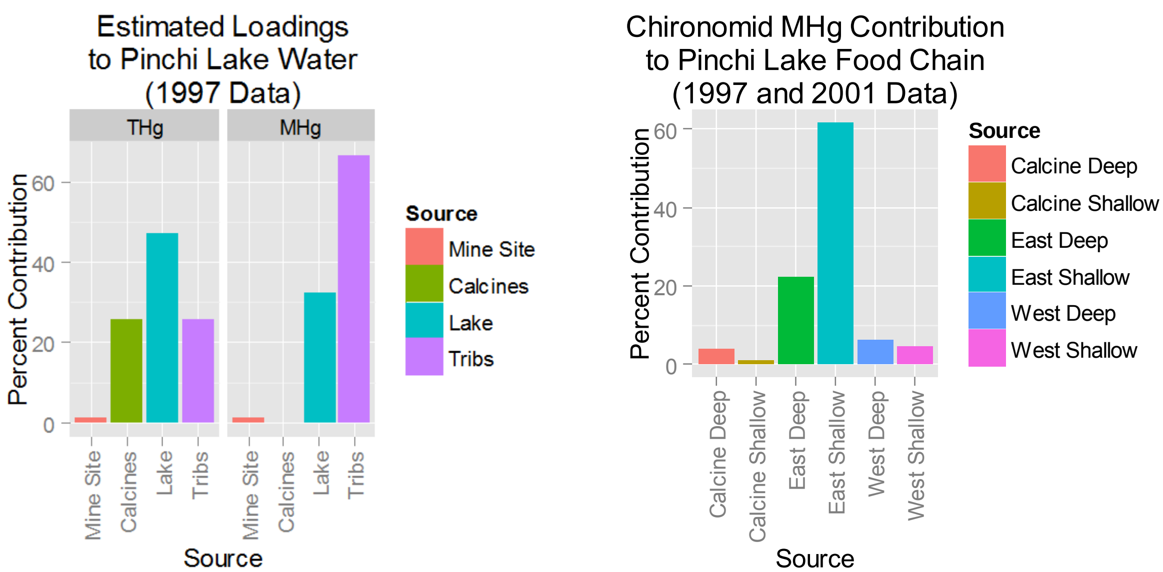


Figure 8. Relative importance of key mercury sources to Pinchi Lake water (left) and benthic food chain (right).

IMPLICATIONS FOR RISK MANAGEMENT

The subsurface calcine area, a legacy of historical wartime mine operations, contains substantially elevated total and methylmercury concentrations in sediment. While vertical profiling conducted in 1997 showed some recovery (i.e., lowered concentrations due to burial with cleaner material) in most subsurface calcine cores, residual surface sediment concentrations remained high and were mirrored in benthos and bivalve tissue concentrations relative to other areas in Pinchi Lake (EVS et al. 1999a,b; EVS and Azimuth 2002). Notwithstanding, there is no evidence of adverse effects to benthos, zooplankton, fish, birds, and mammals (EVS and Azimuth 2002; Azimuth 2008). Furthermore, available data suggests that the relative contribution of the subsurface calcine area to Pinchi Lake biota mercury exposure is relatively low.

The risk management strategy for mine-related contamination in Pinchi Lake (including subsurface calcines) employed since 2001 has been dedicated towards monitoring of natural attenuation, based primarily on documenting temporal trends in fish mercury concentrations. Lake trout (Salvelinus namaycush) and lake whitefish (Coregonus clupeaformis) are the prime target species: lake trout is the top aquatic predator, integrating mercury exposure over the entire food chain (i.e., making this endpoint perhaps the most important indicator of mercury dynamics in Pinchi Lake); whitefish are lower in the food web and are the preferred food source for lake trout (i.e., changes to this endpoint should ultimately be reflected in lake trout tissue mercury). Baker et al. (2014) present the fish monitoring results, which have shown continued reductions in fish mercury concentrations, albeit at slower rates since 2000.

The Pinchi Lake monitoring program was expanded in 2006 to include a full suite of lake ecology components (e.g., limnology, water chemistry, sediment chemistry, phytoplankton community, zooplankton mercury and community, benthic invertebrate mercury and community, more fish species, and stable isotope analysis to document food web relationships) and extended to include Tezzeron and Stuart lakes as regional reference areas. The rationale of this change was to better support the interpretation of fish tissue mercury concentrations by having a better understanding of lake ecology and associated mercury dynamics. This program, to be conducted every decade, is part of the proposed long-term monitoring program for the Pinchi Lake Mine (Teck 2012 DRAFT); the next planned event is 2016.

REFERENCES

Ableson, D.H.G. and E. Gustavson. 1979. Water quality and sediment analysis for heavy metals in selected lakes in the Pinchi Fault area of north central B.C. Omineca Peace Region, B.C. Ministry of Environment.

Azimuth Consulting Group (Azimuth). 2008. 2006 Regional fish mercury and Pinchi Lake ecology report. Prepared for Teck Cominco Metals, Kimberley BC by Azimuth Consulting Group Inc., Vancouver BC. June 2008.

Azimuth. 2009. Pinchi Mine Ecological Risk Assessment. Prepared for: Teck Metals Ltd. Kimberley BC by Azimuth Consulting Group, Vancouver. June 2009.

Azimuth. 2013. Pinchi Lake Mine Fish Mercury Study 2011. A report prepared for Teck Metals Ltd. Kimberley BC by Azimuth Consulting Group Partnership, Vancouver BC. March 2013. 112 p.

Baker, R.F., G.S. Mann and P.J. Allard. 2014. Temporal changes in fish mercury concentrations in mining-affected Pinchi Lake, BC. Prepared for the Technical and Research Committee on Reclamation, Prince George BC. September 22 – 24, 2014.

Environment Canada. 2002. Metal Mining Guidance Document for Aquatic Environmental Effects Monitoring. June 2002. Ottawa, ON.

EVS (EVS Environment Consultants Ltd.). 1996. Pinchi Lake receiving environment investigations. Report prepared by EVS Environment Consultants for Cominco Ltd. March 1996.

EVS. 2001. Regional survey of fish mercury concentrations, Pinchi Lake, BC. Report prepared by EVS Environment Consultants for Cominco Ltd. April 2001.

EVS and Azimuth. 2002. Assessment of sediment mercury contamination, Pinchi Lake, BC. Report prepared by EVS Environment Consultants and Azimuth Consulting Group for Teck Cominco Metals Ltd. November 2002.

EVS, NDM, and Frontier Geosciences (EVS Environment Consultants, Norecol Dames and Moore, and Frontier Geosciences). 1999a. Pinchi Lake Mine: Mine site and lake investigation – data report. Prepared by EVS Environment Consultants, Norecol Dames and Moore, and Frontier Geosciences for Cominco Ltd. August 1999.

EVS, NDM, and Frontier Geosciences (EVS Environment Consultants, Norecol Dames and Moore, and Frontier Geosciences). 1999b. Pinchi Lake Mine: Mine site and lake investigation – environmental assessment report. Prepared by EVS Environment Consultants, Norecol Dames and Moore, and Frontier Geosciences for Cominco Ltd. August 1999.

Martin, J.W., W.F.A. Duncan, and M.N.M. Masselink. 1995. Mercury concentrations in fish tissue and sediment from Pinchi Lake, B.C. Draft. B.C. Environment, Environmental Protection, Prince George, B.C. May 1995.

Petersen, G.R., H.V. Warren, R.E. Delavault, and K. Fletcher. 1970. Heavy metal content of some fresh water fishes in British Columbia. Fisheries Technical Circular No. 2. British Columbia Fish and Wildlife Branch. 33 pp.

Reid, D.S. and R.L. Morley. 1975. Mercury contamination of fish from Pinchi Lake. BC Habitat Protection Section, Fish and Wildlife Branch, Department of Recreation and Conservation. Victoria, B.C.

Teck (Teck Metals Ltd.). 2012. DRAFT Pinchi Lake Mine Long-term Monitoring Plan. June 15, 2012.

Weech, S.A., A.M. Scheuhammer and J.E. Elliot. 2006. Mercury exposure and reproduction in fish-eating birds in the Pinchi Lake region, British Columbia, Canada. *J. Environ. Toxicol*. 25: 1433 – 1440.

DEVELOPING AN EFFECTIVE NATIVE TREE AND SHRUB PLANTING PROGRAM AT TECK COAL LTD.'s ELKVIEW OPERATIONS

John Przeczek, R.P.F.[1]
Lanny Amos, R.P.Bio.[2]

[1]PRYZM Environmental Ltd.
Cranbrook, B.C. V1C 5B4

[2]Teck Coal Ltd., Elkview Operations
Sparwood, B.C. V0B 2G0

ABSTRACT
The "20 Year Conceptual Reclamation Plan" (Przeczek 2003) for Teck Coal Ltd., Elkview Operations provides direction for promoting a range of habitats that will support a range of wildlife species over time. Creating these habitats requires consistently successful revegetation treatments including the establishment of native trees and shrubs. Tree and shrub establishment is critical to meeting target biological diversity objectives for species, spatial distribution and vertical structure set for the reclaimed environment at the mine.

Successful implementation of planting programs requires optimizing seedling quality and carefully managing any situations that can negatively impact seedling physiology. Starting in 2008, Elkview Operations has made a concerted effort to identify and control as many of the links in the "reclamation chain" as possible. We have specifically focussed on:

1. Appropriate species selection;
2. High quality seed collection;
3. High quality planting stock (regular communication with nursery staff);
4. Effective site preparation;
5. High quality planting (close communication with tree planters);
6. Rigorous seedling handling procedures from the cold storage facility to the planting hole;
7. Planting as early in the spring as possible;
8. Fertilization at the time of planting; and,
9. Monitoring success through formal survival plots and regular walk-through assessments of past plantations to inform adaptive management programs.

Plantation success has improved from 2008 – 2013 so that we are now comfortable prescribing the establishment of specific tree and shrub species plant communities. We currently focus on creating eight broad plant community types. Our program has expanded to include 4 coniferous tree species, 3 deciduous tree species, and 13 shrub species. We will continue to expand the list of species, minimize seedling physiological stress, and monitor success to ensure that the mosaic of planted vegetation types, in combination with areas seeded with agronomic grasses and legumes, will promote site level biological diversity objectives.

Key Words: Conceptual Reclamation Plan, Mine Reclamation

BACKGROUND

The 20 year conceptual reclamation plan for the Elkview property was updated in 2003 (Przeczek, John 2003, Przeczek, John and Dave Ryder 2004) and again in 2014 (Przeczek, John 2014). The plan recognizes the wildlife end land use specified in Reclamation Permit C-2 and it was prepared to ensure that additional site level biological diversity objectives are achieved and to provide direction for reclamation planning and implementation into the future. Key components of the plan include a vision for the horizontal distribution, vertical structure, and species composition of the plant communities that will be established, the recognition of Reclamation Treatment Units (RTU's) as the basis for developing reclamation treatment plans, and a commitment to develop and implement monitoring programs that can inform adaptive management needs (Przeczek, John 2003). The Conceptual Reclamation Plan is based on the concept of ecological replacement which allows us to focus on creating post-mining landscapes that have similar functioning to pre-mining conditions but does not imply that they will be the same (Cooke and Johnson 2002). The 2014 Conceptual Plan update is also consistent with Teck's "Biodiversity Management Planning" initiative (Gullison 2014).

Prior to 2007, revegetation efforts focussed on establishing self-sustaining agronomic grass and forb dominated plant communities. Engelmann spruce (*Picea engelmannii* Parry ex Engelm.) and lodgepole pine (*Pinus contorta* Douglas ex Loudon var. *latifolia* Engelm. ex S. Watson) plantations were also established in a few areas with wolf-willow (*Elaeagnus commutata* Bernh. ex Rydb.) patches established in one of the spruce plantations. Black cottonwood (*Populus balsamifera* L. ssp. *trichocarpa* (T &. G) Brayshaw) commonly volunteers throughout the reclaimed areas. By 2007 it was clear that establishing tree and shrub dominated plant communities with a larger diversity of species would be required to meet end land use.

Varying levels of plantation success through 2007 were limiting our ability to meet reclamation objectives. The goal of consistently establishing plant communities with specific species composition and vertical structure required us to focus on the factors that affect plantation success. These factors have been compared to the links in a chain; if any one of them is weak, the chain will break and success will not be possible (Helgerson 2004). In 2008 we began to focus on the components of the planting program that we could control to improve the probability of success. All of our efforts have been designed using the extensive experience and research results that guide the reforestation programs of the Forest Industry. Everything we promote in the tree and shrub establishment program is aimed at acquiring the best planting stock possible, minimize seedling physiological stress, and establishing diverse plant communities.

PLANTING PROGRAM CONSIDERATIONS

Species selection is probably the most important decision we make in the reclamation process. The 2014 Conceptual Reclamation Plan identifies the species that we work with at Elkview Operations; this list will continue to evolve as we move forward (Table 1). A detailed seed needs analysis was conducted to allow for the purchase of a 20 year supply of desired conifer species[1] and to determine the amount of seed that would be required to maintain a 2- 4 year supply of other tree and shrub species. “The capability of stored seeds to produce vigorous seedlings is affected by seed quality at the time of collection and the care with which seeds are handled during collection, transport, processing, and storage” (Leadem, C.L., R.D. Eremko and I.H. Davis 1990, p.193).

Conifer seed is purchased from certified registered seedlots and it is stored in the BC Forest Service Tree Seed Centre in Surrey, BC. Seed collections for other species are coordinated and conducted by consultants with extensive native tree and shrub seed collection experience. Collected material is stored in refrigerators prior to shipping to extraction facilities, temperature is controlled with freezer packs during shipping, seed extraction is completed by qualified facilities with a proven record of success, seed is dried down to 8 – 10% moisture content and then kept refrigerated at 2 – 4 degrees until needed for a reclaimed site. Each seed collection for a species is given a unique number (Seedlot Number) and seed is used on a first-in first-out basis. It takes approximately 16 months to order and grow a seedling for spring planting. Having high quality seed available provides the flexibility required to plan effective planting programs at the range of elevations that we work with at Elkview Operations (1150 – 2200 meters). We started to experiment with freezing aspen, cottonwood and willow seed in 2013 and the results look promising which will provide more flexibility and more consistent seedling quality for these species in the future.

We grow our seedlings at private nurseries that have a proven ability to consistently provide high quality seedlings at requested levels. Our planting program is spring-based and we have adopted the “forestry” production regime which includes fall/winter seed stratification, spring sowing, fall lifting, freezing seedlings for storage over the winter and a short (10 day) controlled thaw regime just prior to planting. We believe that it is important to develop a strong working relationship with an experienced grower; it is not good enough just to order your plants in the fall and plant them 18 months later. Your knowledge of your sites in combination with your grower’s knowledge of nursery culture generates a powerful contribution to plantation success. Nurserymen should be able to provide guidance regarding stocktype options and growing regimes based on the requirements of your different sites. They should also be responsive to your requests for boxing, labeling, shipping and storage.

We continue to have good plantation success on simple resloping treatments that create relatively continuous 26 degree slopes. However, our post-resloping site preparation treatments now focus on incorporating diversity without compromising slope stability. Micro-topographic features that reflect conditions that we find in adjacent undisturbed sites are created through twist and turn treatments with the crawler-tractor or through creating small mounds, up to 50 cm high, with an excavator. We will also continue to use ripping treatments to promote seedling establishment where the topography is relatively subdued. All treatments are completed before snowfall to ensure that the maximum potential soil moisture is available during spring planting operations. We have experimented with spring site preparation treatments but they create conditions that become excessively dry due to the windy conditions at the mine site.

[1] Purchases will be made periodically to maintain a 20 year conifer seed supply.

Table 1. Native tree and shrub species used in the reclamation program at Teck Coal Ltd., Elkview Operations.

Scientific Name	Common Name
Plant species regularly used in the reclamation program at Elkview Operations	
Alnus viridis (Chaix) DC, ssp.*sinuata* (Regel) A. & D. Love	Sitka alder
Amelanchier alnifolia Nutt.	saskatoon
Betula papyrifera Marsh.	white birch
Cornus stolonifera Michx.	red-osier dogwood
Pentaphylloides floribunda (Pursh) A. Love	shrubby cinquefoil
Elaeagnus commutata Bernh. *ex* Rydb.	wolf-willow
Larix occidentalis Nutt.	western larch
Picea engelmanni Parry *ex* Engelm.	Engelmann spruce
Pinus contorta Dougl. *ex* Loud.	lodgepole pine
Populus balsamifera L. ssp. *trichocarpa* (T &. G) Brayshaw	black cottonwood
Populus tremuloides Michx.	trembling aspen
Prunus virginiana L.	chokecherry
Rosa acicularis Lindl. ssp. *sayi* (Schwein.) W.H. Lewis	prickly rose
Rubus idaeus L. var. *strigosus* (Michx.) Maxim	red raspberry
Salix sp.	upland willow
Sambucus racemosa L. ssp. *puben*s (Michx.) House var. melancocarpa (A. Gray) McMinn	black elderberry
Vaccinium membranaceum Douglas *ex* Hook.	black huckleberry
Plant species used in the reclamation program at Elkview Operations (limited data)	
**Acer glabrum* Torr. var. *douglasii* (Hook.) Dippel	Douglas maple
**Rubus parviflorus* Nutt. var. *parviflorus*	thimbleberry
**Shepherdia canadensis* (L.) Nutt.	Canadian buffalo-berry
Plant species with potential for use in the reclamation program at Elkview Operations	
Arctostaphylos uva-ursi (L.) Spreng.	kinnikinnick
Ceanothus sanguineus Pursh	redstem ceanothus
Ceanothus velutinus Douglas *ex* Hook.	snowbrush
Dryas integrifolia Vahl	entire-leaved mountain-avens
Juniperis communis L.	common juniper
Pseudotsuga menziesii (Mirb.) Franco var. *glauca* (Beissn.) Franco	Douglas-fir
Salix sp.	riparian willow
Symphoricarpos albus (L.) Blake	common snowberry
Sorbus sitchensis M. Roem.	Sitka mountain-ash

As the implementation of a planting program approaches it is important to remember that frozen storage facilities typically require a minimum of 10 days to execute a thaw request and you need to build some flexibility into your timing to allow for planter scheduling, days off and taking advantage of optimal soil moisture conditions. We have found that planting high elevation sites (over 1600 meters) requires us to accept that a proportion of the planting site will be covered with snow patches because of differential melting at the site. Our general rule of thumb is to start planting when approximately 75% – 80 % of the planting area is snow-free. This also provides for a higher level of horizontal diversity in plant communities because the areas affected by slower snow melt will become grass and legume dominated when the planting area is seeded after two or three growing seasons.

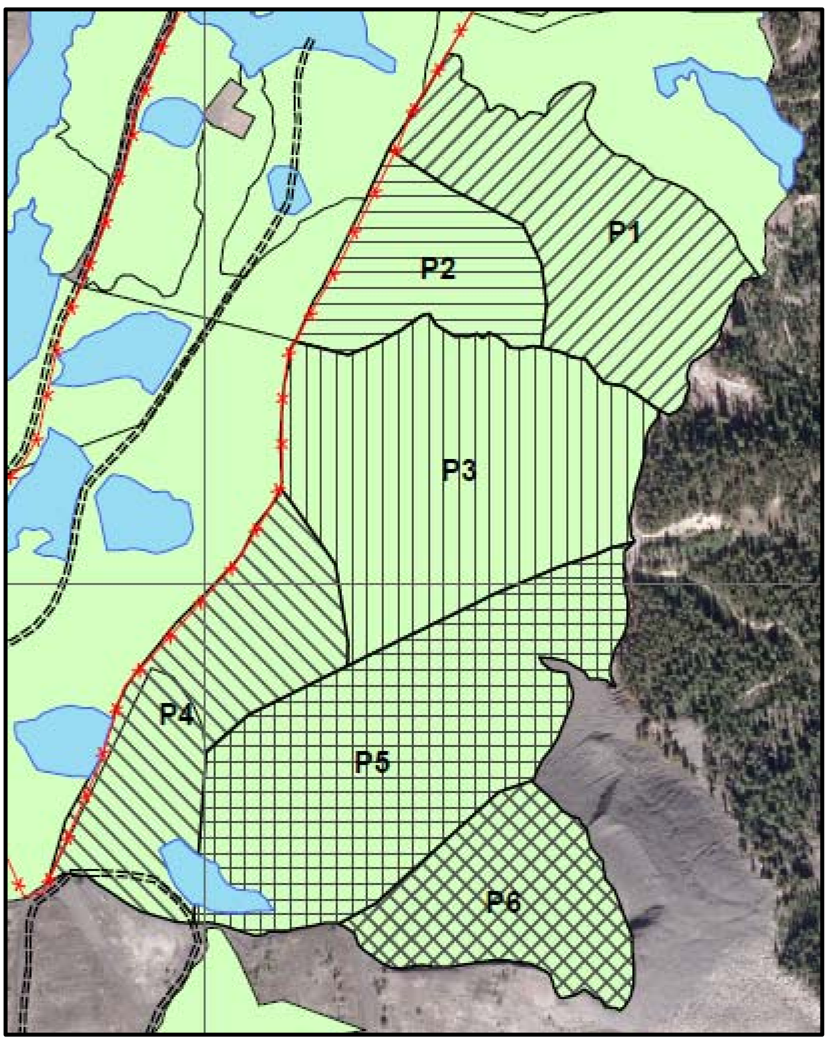
It is important to thoroughly inspect seedlings upon receipt from the seedling storage facility. Check foliage, stems and roots for signs of mechanical damage, desiccation, freezing, or storage molds. Some deciduous species respond quickly to increased temperatures during thawing and may start flushing in the boxes; plant them as quickly as possible if this occurs. A small, shallow scrape of the stem should reveal a healthy green cambium layer. After a brief inspection, re-seal the vapour barrier and close the box. Make detailed notes of observed issues with specific species or seedlots so that you can follow plantation results through time with the information required to isolate factors that may affect plantation survival and performance.

We have implemented rigorous stock handling procedures that are designed to reduce physiological stress and maximize seedling performance potential (Lavender 1990). Stock handling procedures focus on minimizing physical damage, keeping seedlings cool and moist, and avoiding the development of storage molds. We use snow caches for the main on-site storage to control inbox temperatures. We strictly enforce the use of insulated seedling canopies on planter’s trucks, a maximum half-day seedling supply for removal from the main cache, and use of therma-cool tarps for any seedlings stored on the site for a maximum of 3 hours. Seedling boxes need to be handled carefully. Tabbush (1986) found that dropping seedling boxes one to fifteen times, from a height of three meters, reduced seedling survival and performance significantly. “Similar effects may be seen if boxes and seedlings are crushed by compaction, therefore boxes should never be stocked more than three high. (Mitchell et al 1990, p 239). It is important to include planters in stock handling discussions so that they understand how their behavior can influence seedling survival and performance.

Maintaining high levels of planter moral is critical in promoting plantation establishment and growth. This is particularly important at Elkview Operations because planters are generally astonished that planting trees and shrubs would be considered viable on resloped coal spoils. We always show planters and planting contractors the success levels of past planting projects so that they will understand that their efforts and our requirements will result in the establishment of impressive plantations with diverse species mixes that they are generally not used to seeing. Planting conditions are difficult and our stock handling requirements are demanding; happy planters make for happy plantations. All seedlings are also planted with a fertilizer tea-bag to promote establishment and early growth in the nutrient poor environment typical of resloped coal spoils.

RESULTS AND FUTURE EXPECTATIONS

The period from 2008 through 2010 was spent getting all of the players in line with what we were trying to accomplish with our comprehensive seed to planting hole approach to improving plantation success. Our expectations were communicated and issues with nurseries and cold storage facilities were ironed out. We were not sure which species would be most reliable at this point but experience from other mine sites provided some guidance. Seedling survival study plots were established each year to provide localized data and they will continue being established annually. By 2011 we were comfortable that we could successfully implement a more complex planting regime and that we had sufficient local survival information to support our decisions. In 2012 we started allocating specific species mixes to various planting units and in 2013 the whole planting program was implemented around the idea of creating specific plant communities (Figure 1).



TU	Area (ha)	Plant Community Type
P1	1.88	Decid-Conif/Tall Shrub
P2	1.02	Short Shrub/Mixed Tree
P3	2.85	Decid-Conif/Tall Shrub
P4	1.74	Mixed Tree/Mixed Shrub
P5	2.90	Conif-Decid/Tall Shrub
P6	1.22	Mixed Shrub/Mixed Tree

Figure 1. Planting units from the 2013 planting program at Teck Coal Ltd., Elkview Operations.

Survival plot data collected since 2008 shows that survival generally exceeds 70% for species that are regularly used in the reclamation program (Figure 2). There have been cases where planting stock quality, very rocky planting medium or unknown factors have caused higher levels of seedling mortality. The survival plot monitoring system has been implemented to allow us to isolate survival problems that may be nursery/seedling storage versus site/planting related. This allows us the opportunity to follow seedling survival over time and explain the reasons behind failures and take corrective action in future planting projects. We maintain close communication with the nurseries that grow our stock and with the frozen storage facilities that we use and everyone in the program strives for continual improvement.

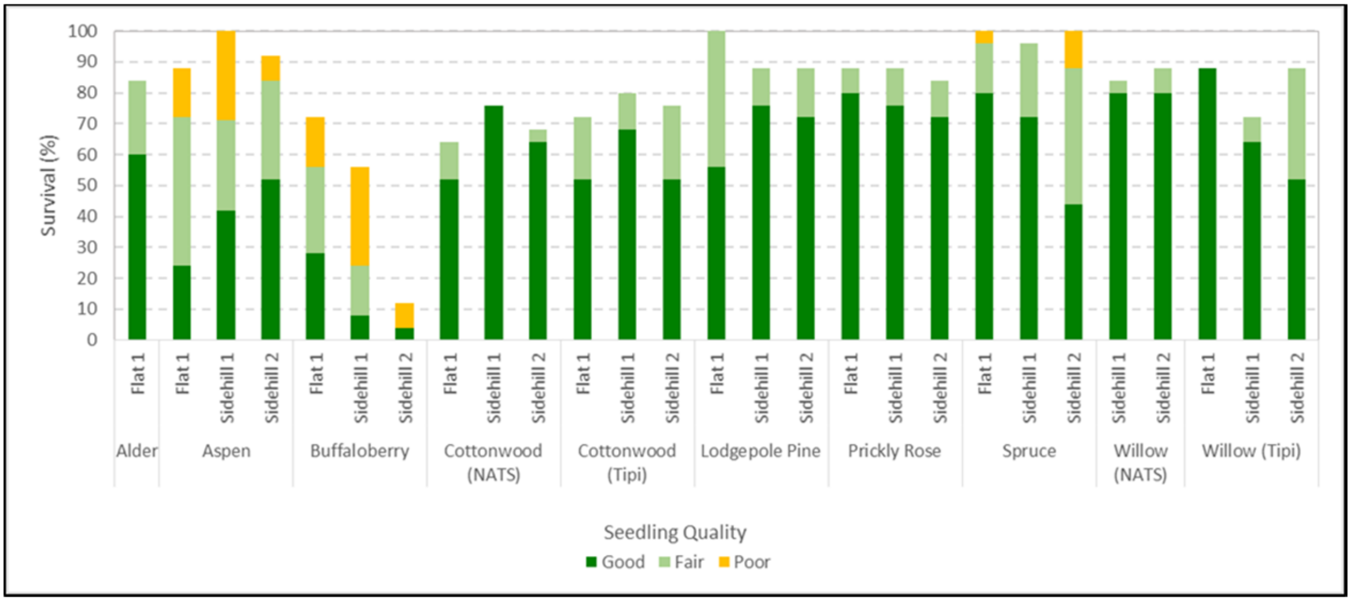


Figure 2. Third growing season seedling survival on plots established in 2011, Harmer East, Elkview Operations.

The planting program at Elkview Operations will continue to concentrate on the factors that appear to have the most profound influence on plantation success. The species list of trees and shrubs that are regularly used in the reclamation program will continue to expand as we test new species and generate the data required to support their use. We also recognize that some species with intermediate levels of seedling survival may be used in the reclamation program because they are important to specific Communities of Interest. For the next 5 - 10 years we will continue to promote the establishment of the plant community types that are shown in Table 2. Diversity in each type will occur on the ground by using the range of tree and shrub component targets and using different species mixes in the planting units that are identified prior to planting (Figure 3).

We believe that focussing on all of the links in the “reclamation chain” will continue to yield positive outcomes in the tree and shrub establishment program at Elkview Operations. This will feed directly into current Biodiversity Management Planning initiates at Teck and will help to ensure that end land use objectives designated for the Elkview Property are realized.

Table 2. Plant community types that will be established through the reclamation program at Teck Coal Ltd., Elkview Operations.

Name	Tree Component (%)		Shrub Component (%)		Comments
	Conif.	Decid.	Tall	Short	
Coniferous	85 - 95	5 - 15	0 - 10	0 - 10	
Deciduous	5 - 15	85 - 95	0 - 10	0 - 10	
Coniferous - Deciduous	50 - 80	30 - 40	0 - 15	0 - 15	Tree component will be at least 85%
Deciduous - Coniferous	30 - 40	50 - 80	0 - 15	0 - 15	Tree component will be at least 85%
Tall Shrub	0 - 15	0 - 15	85 - 100	0 - 15	
Short Shrub	0 - 15	0 - 15	0 - 15	85 - 100	
Mixed Tall and Short Shrub	0 - 15	0 - 15	30 - 50	30 - 50	Shrub component will be at least 85%
Mixed Tree and Shrub	15 - 40	15 - 40	15 - 40	15 - 40	

Figure 3. Example of a 6 year old Deciduous-Coniferous plant community type on the Harmer West spoil, Elkview Operations.

REFERENCES

Cooke, J.A. and M.S. Johnson.2002. *Ecological restoration of land with particular reference to the mining of metals and industrial minerals: a review of theory and practice*. Environ. Rev.10:41 – 71.

Gullison, Ted. 2014. *Biodiversity management planning guidance manual*. Unpulbished Consultant’s Report prepared for Teck Coal Ltd., Sparwood, BC.

Helgerson, Ole T. 2004. *Successful reforestation planning to get the forest you want*. In: Northwest Woodlands Magazine, Winter 2004, Oregon Small Woodlands Association, Washington Farm Forestry Association, Idaho Forest Owners Association and Montana Forest Owners Association Publications, Portland, OR, USA. 12 p.

Lavender, D.P. 1990. *Physiological principles of regeneration*. In: “Regenerating British Columbia’s Forests”. UBC Press, Vancouver, BC, pp 30 – 43.

Leadem, C.L., R.D. Eremko and I.H. Davis. 1990. *Seed biology, collection and post-harvest handling*. In: “Regenerating British Columbia’s Forests”. UBC Press, Vancouver, BC, pp 193-205.

Mitchell, W.K., G. Dunsworth, D.G. Simpson, and A. Vyse. 1990. *PlantiThe period from 2008 – 2009 was spent ng and seeding*. In: “Regenerating British Columbia’s Forests”. UBC Press, Vancouver, BC, pp 235-253.

Przeczek, John. 2003. *Conceptual reclamation plan for the Elkview Coal Property at Sparwood, BC*. Unpubl. Consultant’s Report, Elkview Coal Corp. 35 pp + append.

Przeczek, John. 2014. *Conceptual reclamation plan for the Elkview Coal Property at Sparwood, BC*. Unpubl. Consultant’s Report, Elkview Coal Corp. 35 pp + append.

Przeczek, John and Dave Ryder. 2004. *Conceptual reclamation planning and the development of results-based standards at Elk Valley Coal’s Elkview Operations*. In: “BC Mine Reclamation Symposium, 2004”. BC Technical and Research Committee on Reclamation. 10 pp.

Tabbush, P.M. 1986. *Rough handling, soil temperature and root development in outplanted Sitka spruce and Douglas-fir*. Can. J. For. Res. 16:1385-1388.

THE EFFECT OF SITE PREPARATION ON THE ESTABLISHMENT OF NATIVE GRASSLAND SPECIES IN SOUTHERN INTERIOR, BC

K. Baethke
L. Fraser

Thompson Rivers University
900 McGill Road
Kamloops, BC V2C 5N3

ABSTRACT

New Gold's New Afton Mine, located near Kamloops in the BC southern interior grasslands, is committed to ecosystem restoration through stakeholder consultation and research, including restoration of native grassland. Our objective was to study restoration potential of 24 selected native species from the BC interior grasslands, including those of cultural significance to First Nations. We applied a controlled experimental manipulation on the top of a flattened top soil stockpile to test germination and establishment of 12 native forb and 12 native graminoid species seeded on plots that were raked, hydroseeded or seeded with no manipulation. Raking resulted in greater establishment and count data for both native forb and graminoid species; however, it also increased the number of 'volunteer' non-native forbs. Hydroseeding had a negative effect on native graminoid species overall and on non-native forb species. Plant diversity, calculated as Shannon's diversity index, was greatest in the seeded plots that had been hydroseeded and hydroseeded + raked. As well, species richness was highest in the raked + seeded treatments. In terms of management, raking appears to result in positive seed germination for many native species; whereas hydroseeding was less effective.

KEY WORDS: Diversity, species richness, hydroseeding, raking, grassland restoration, mine reclamation

INTRODUCTION

Mining is an important economic driver of British Columbia's economy. The gross mining revenue in British Columbia in 2011 amounted to $9.9 billion and spin offs from the mining industry provided a further $3 billion in direct industry expenditures ("The Mining Association of BC" 2013). However, mining causes disturbances on the landscape through the building of infrastructure and the extraction of minerals and ores (Vickers et al. 2012). To offset these ecological disturbances, the BC mining industry has made environmental protection a priority during the development, operation and closure of mines. Mining regulations in BC are governed by both federal and provincial governments, and regulatory and monitoring processes occur throughout the life of the mine, including closure. Legislation requires mines to either reclaim or restore these disturbed sites to pre-disturbance productivity levels.

Reclamation and restoration are two different objectives for managing disturbed sites. Reclamation is the process of returning a disturbed site back to its pre-disturbance productivity level using a combination of species which may include non-native agronomics. Restoration, is the process of returning a site back to its natural, self-sustainable state (Alday et al. 2011). To attain either of these goals, seeding is often undertaken to mitigate erosion from wind and water and to assist site recovery in terms of the types of desirable species (Baasch et al. 2012). Historically, industries used agronomic species for reclamation because of their ability to colonize quickly thus reducing erosion and making the area aesthetically pleasing (Carrick and Krüger 2007, Prach and Hobbs 2008). However, agronomic species can become invasive and thus reduce species richness and diversity within nearby native communities (Christian and Wilson 1999). With restoration, the addition of native seed has been shown to affect the path and speed of succession (Martin and Wilsey 2006, Baasch et al. 2012). Research in this area is important as ecosystems differ and methods of restoration must be altered to match the biotic and abiotic factors and stresses of the area (e.g. arid grassland versus forested wetland). As degraded sites are in an altered state, the impact restoration will have on the path of succession is unknown and hard to predict (Suding et al. 2004), thus research should be an integral part of a mine's restoration strategy.

Grasslands present a unique set of problems when it comes to restoration. In BC, grasslands are predominantly found in the 'rain-shadow' east of the Coast and Cascade Mountains, where the climate is dry and summers are hot ("Grasslands Conservation Council" 2012). Restoring grasslands in semi-arid systems is difficult due to environmental and economic limitations (Prach and Hobbs 2008). Low precipitation rates in grasslands can reduce germination, establishment and growth (Josa et al. 2012). Adding a disturbance stress to the system can reduce the availability of nutrients (Brady and Weil 2012), disrupt the microbial community (Wanner and Dunger 2001), increase compaction where heavy equipment is used thus decreasing water and root penetration (Brady and Weil 2012), and increase the potential for invasion by exotic species (Yurkonis et al. 2008). Microclimates created by topography as well as plant biomass, including litter, play an important role in the amount of sun or shade an area receives, the intensity of wind experienced, the amount of precipitation that can be sequestered and the amount of evaporation and transpiration that occurs at a site. Economically the lack of or low availability of and high cost of seed (Rowe 2010) can make restoration goals difficult to obtain. Together these factors make finding techniques to ensure successful restoration challenging.

Raking and tilling have been used in an effort to increase the success of grassland restoration (Wilson and Gerry 1995, Pywell et al. 2002, Carrick and Krüger 2007). Wilson & Gerry (1995) found that low disturbance and high disturbance tilling resulted in higher densities of native species. Although tilling can reduce soil compaction (Burke 2003) and increase microclimates by roughening the soil surface (*An Integrated Approach to Establishing Native Plants* 2007), whether it decreases or increases invasion by exotics is controversial as studies have shown both outcomes (Wilson and Gerry 1995, Montalvo et al. 2002, Kiehl et al. 2010). Litter and hydroslurry have been used to reduce erosion, maintain moisture levels, add nutrients and increase germination and seedling establishment (Matesanz et al. 2006, Dunifon et al. 2011, Oliveira et al. 2012). A meta-analysis on the effects of litter on seedling emergence by Loydi et al. (2013), revealed seedling emergence in dry grasslands was affected by the amount of litter. Hydroslurry which may create a similar environment to plant litter, is often used for re-vegetating slopes in an attempt to reduce erosion (Matesanz et al. 2006), increase moisture availability and increase germination. As some studies have found little value to hydroseeding, the writer feels the benefit to hydroseeding is somewhat controversial.

Hypotheses

1) Hydroseeding will result in increased germination and establishment rates for native species while suppressing non-native forb species.
2) Site preparation, such as raking, will increase germination and establishment of native flora indigenous to the interior of BC, but will also increase invasion by exotics.
3) Seeding native grassland species will result in higher species richness and diversity.

Many studies have shown planting native seeds results in germination and establishment as well as increased species richness and diversity over one or more years (Montalvo et al. 2002, Martin and Wilsey 2006, Martínez-Ruiz et al. 2007, Yurkonis et al. 2008, Dornbush and Wilsey 2010, Kiehl et al. 2010). As the BC Interior grasslands are water limited, using species adapted to the area should result in native vegetative cover. Hydroseeding has been successful at re-establishing road banks with native seed mixes in countries like Spain (Tormo et al. 2007), but has been less successful in re-vegetating slopes in the Mediterranean where it is semiarid (Bochet and Garcıa-Fayos 2004), we expect to see increased germination on a non-sloped site. We also expect to see increased establishment on sites that have been raked as has been seen in previous studies which looked at tilling (Wilson and Gerry 1995, Montalvo et al. 2002).

MATERIALS AND METHODS

Site Description

The study took place at New Gold's New Afton Mine (NAM) site west of Kamloops, British Columbia (50°38'54.92" N 120°29'59.67" W, elevation 775m). The NAM is an underground working, copper-gold mine situated on a historical open pit. The mine is located in the Ponderosa Pine and Interior Douglas-fir biogeoclimatic zone and the surrounding grasslands are a northern extension of the Pacific Northwest Bunchgrass grassland (*The ecology of the Interior Douglas-fir zone* 2014). This area has a short, warm summer season (May – September) with average temperatures ranging from 8°C to 29°C respectively. Winter mean annual temperatures range from -6°C to 5.6°C. The average yearly precipitation is ~278mm with 81% of the moisture coming as rainfall and 19% coming as snowfall ("Cimate Data, Environment Canada" 2014).

Experimental Design

Plant species selected for sowing were chosen based on their presence in the interior grasslands of British Columbia and their cultural importance to the local First Nations. Seeds were either handpicked or sourced from local seed companies. For those species which were handpicked, the populations were followed through the 2012 summer season and harvested once seeds had set and matured. Seeds were collected from a number of populations whenever possible. Exceptions were *Mentzelia laevicaulis* and *Oxytropis campestris*. *M. laevicaulis* was collected from a single population found on the New Afton Mine site and *O. campestris* was picked from a single population found at Teck's Highland Valley Copper Mine site. After collection, all seeds were sealed in plastic ziploc bags and stored in a chest freezer.

A topsoil stockpile located north of the tailings pond at NAM was levelled and a grid of 80 plots was created on the east end in October, 2012. Each plot measured 2 m^2 with a half meter between each plot to allow for movement between the treatments for watering and assessment. Each row (rep) had 8 treatments; four soil preparation treatments: raking, hydro-slurry, hydro-slurry and raking, and a control where no soil preparation was completed; and each of these treatments had a seeded and unseeded component (4 x 2 x 10; n=80). Treatments were randomized within each rep using a computer generated randomization plan (random.org) ("Random.org" 2014).

In the fall of 2012 seed packets were prepared for fall planting. Coin envelopes were labelled and filled with 45 ml of sand, a carrying agent, for each treatment (80 in total). The study plan was to seed each plot with 1200 seeds/m^2 (Fraser and Grime 1999). However some of the species collected by hand did not have enough seeds to obtain the 4800 seeds per plot needed and so the number was reduced to 200 seeds per species or 600 seeds/m^2 which is considered a high density planting by Carter & Blair (2012). In October 2012 when the grid was set up it was determined the stockpile in the area chosen had areas that were compacted. To reduce the effect of compaction on raked treatment plots, a shovel was used to loosen the dirt on the plots and dirt from the edge of the stockpile was added to help simulate a tilled/raked treatment. Three 12L buckets of soil were added to each of the raking treatments to maintain consistency throughout the study grid.

Seeding took place in November 2012 when temperatures were low enough to ensure early germination would not occur (Martínez-Ruiz et al. 2007). Seeds were hand-seeded on all seeded treatment plots except those designated for hydro-slurry. Hand-seeding was conducted by only two people to reduce bias. For hydro-seeded plots, seeds were mixed into 2 – 12L buckets. Half of the seed mixture was mixed into each bucket. Control packages which contained just the sand were also mixed into the hydro-slurry, ½ into each bucket to stay consistent with the seeded packages. The hydro-seed mixture was then spread over the plots by pouring the two buckets evenly over the treatment area. All control plots were hydro-slurried first to ensure no contamination occurred between control sites and seeded treatment sites. In the spring of 2013, plots were watered with 4mm of water twice a week beginning in May using watering cans with a disperser spouts. Watering continued until the June rains began and then was only completed when no rain occurred on watering days. Plots were watered until the water pooled on the soil surface. This moisture was allowed to seep through the soil before the remaining water was added. Plot assessments were carried out the second week of July 2013. A 1m^2 grid was placed in the center of each treatment plot and all species within the grid were counted and recorded.

<u>Statistical Analysis</u>

Statistical analysis of vegetation count data was conducted using R Studio (R version 2.15.3; 2012 R Foundation for Statistical Computing). A Filgner test for homogeneity of variances was completed for all data sets. Data was square root transformed to satisfy assumptions of a normal distribution, and a two-way analysis of variance (ANOVA) was used to determine differences between treatments and a Tukey post hoc analysis was conducted when there was a significant statistical difference ($p<0.05$).

RESULTS

At the species level, there were effects of site management and seeding on the number of individuals (Table 1). With respect to native forbs, there was a greater number of *Achillea millefolium* plants in sites that were raked or undisturbed compared to the hydro-slurry. *Balsamorhiza sagittata* had greater numbers on sites that had been both raked and hydro-slurried. *Gaillardia aristata* had the greatest number of individuals on sites that had either been raked or hydro-slurried and raked. Non-native forbs showed very little preference to treatments, however *Salsola tragus* had greater numbers of individuals in the raked plots, while hydro-slurry caused a negative response when the plots were not raked. Looking at the graminoid species, *Elymus trachycaulus* and *Pseudoroegnaria spicata* exhibited a positive response to raking as a soil preparation and a negative response to hydro-seeding resulting in fewer seedlings on these sites. Non-native graminoids, *Agropyron. cristatum* and *Bromus tectorum*, showed no significant differences between soil treatments, but *A. cristatum* did show a positive trend in the hydro-seeded + raked sites.

Some non-native species responded to both the seeding treatments and the soil preparation treatments. *Kochia scoparia* responded positively towards raking and negatively towards the hydro-slurry. The hydro-slurry reduced the number of plants by approximately 85% on the seeded sites and 75% on the non-seeded sites. *Sisymbrium loeselii* also responded positively to raking, but was suppressed by the hydro-slurry. For the sites that were hydroseeded, *S. loeselii* was suppressed by approximately 83% and the hydro-slurry with no seed reduced *S. loeselii* by 41%. A slight trend was seen for *Salsola tragus* where the hydro-slurry showed some negative impact on the number of plants counted.

Both non-native and native species as groups reacted similarly to the raking and hydroseeding treatments (Table 1; Figure 3). Raking significantly increased the number of seedlings counted, while the hydroslurry suppressed the number of seedlings counted overall. When raking and hydroseeding were integrated the number of non-native seedlings was not significantly different between the raked and raked + hydroseeded sites. However, the native species count overall was still significantly depressed by the hydroslurry even when raking was added to the treatment.

The Shannon Diversity Index revealed significance between the seeded and non-seeded treatment plots (Table 1), such that seeded plots had a higher diversity (Figure 1). Site preparation as a main effect did not influence diversity; however, the interaction between seeding and site preparation was significant. Seeded hydroslurry and seeded hydroslurry plus raking treatments had higher diversity than non-seeded raked treatments (Figure 1).

Species richness (*S*) was also highest in the seeded treatments (Table 1; Figure 2). The raked and raked + hydro-seeded sites had higher *S* than the control sites. Whereas the seeded and hydro-seeded treatments were not significantly different from the control treatments.

Table 1: Results from 2-way ANOVAs, soil preparation (Soil Prep) and seeding (Seed/No Seed) on numbers of native and non-native species on stockpiled topsoil at New Gold's New Afton Mine Site. Soil preparation included control (no manipulation), raking, and hydro-slurry, while seeding treatments were either seeded or unseeded.

Species	**Soil Prep**		**Seed/No Seed**		**Soil Prep + Seed**	
	F-value	**P-value**	**F-value**	**P-value**	**F-value**	**P-value**
Native forbs (seeded)						
Achillia millefolium	**9.772**	**1.74e-05**	**41.308**	**1.24e-08**	**9.772**	**1.74e-05**
Balsamorhiza sagittata	**6.075**	**9.59e-04**	**20.764**	**2.07e-05**	**6.075**	**9.59e-04**
Erigeron filifolius	1.411	0.247	1.244	0.268	1.244	0.300
Fritillaria pudica	0.667	0.575	2.000	0.162	0.667	0.575
Gaillardia aristata	**4.313**	**0.007**	**28.738**	**9.54e-07**	**2.806**	**0.046**
Mentzelia laevicaulis	0.668	0.575	**4.475**	**0.038**	0.668	0.575
Native forbs (volunteer)						
Artemisia tridentate	1.320	0.275	3.240	0.076*	1.320	0.275
Artemisia frigidata	2.364	0.078*	0.896	0.347	0.299	0.826
Astragalus tenellus	0.333	0.801	0.333	0.566	1.222	0.308
Calocortus macrocarpus						
Non-native forbs						
Berteroa incana	0.667	0.575	2.000	0.162	0.667	0.575
Camelina microcarpa	0.667	0.575	1.000	0.321	0.333	0.801
Centaurea stoebe	1.000	0.398	1.000	0.321	1.000	0.398
Chenopodium album	0.736	0.534	0.093	0.762	1.058	0.372
Descurainia sophia	0	1.000	**4.000**	**0.049**	0	1.000
Kochia scoparia	**14.345**	**2.02e-07**	0.093	0.761	1.595	0.198
Lactuca serriola	1.000	0.398	1.000	0.321	1.000	0.398
Lepidium densiflorum	1.222	0.308	0.333	0.566	0.333	0.801
Medicago lupulina	1.112	0.350	0.135	0.714	1.453	0.235
Melilotus alba	**3.565**	**0.018**	1.870	0.176	0.178	0.911

Species	Soil Prep		Seed/No Seed		Soil Prep + Seed	
	F-value	**P-value**	**F-value**	**P-value**	**F-value**	**P-value**
Myosotis sp.	1.222	0.308	0.333	0.566	0.333	0.801
Polygonum aviculare L	0.901	0.445	0.025	0.875	1.100	0.355
Rumex acetosella	1.000	0.398	1.000	0.321	1.000	0.398
Salsola tragus	**4.194**	**0.009**	0.094	0.760	0.636	0.594
Sisymbrium loeselii	**8.666**	**5.53e-05**	0.113	0.738	1.807	0.154
Taraxacum officinale	0.667	0.575	0.000	1.000	1.333	0.270
Thlaspi arvense	1.000	0.398	1.000	0.321	1.000	0.398
Native grasses						
Elymus glaucus	0.708	0.550	**5.287**	**0.0244**	0.708	0.550
Elymus trachycaulus	**3.980**	**0.011**	3.578	0.0626*	**5.971**	**0.001**
Festuca sp	1.134	0.341	14.148	**2.95e-04**	1.134	0.341
Poa sandbergii	1.000	0.398	1.000	0.321	1.000	0.398
Pseudoroegneria spicata	**13.190**	**5.90e-07**	**23.140**	**8.02e-06**	**10.280**	**1.03e-05**
Sporobolus cryptandrus	2.250	0.090*	2.250	0.138	2.250	0.090*
Non-native grasses						
Agropyron cristatum	0.792	0.502	1.499	0.225	2.649	0.055*
Bromus squarrosus	0.667	0.575	2.000	0.162	0.667	0.575
Bromus tectorum	1.086	0.360	0.125	0.725	0.391	0.760
Elymus repens	1.000	0.398	1.000	0.321	1.000	0.398
Poa sp.	1.222	0.308	0.333	0.566	0.333	0.801
Poa compressa	1.000	0.398	1.000	0.321	1.000	0.398
Triticum sp	**3.240**	**0.027**	0.360	0.550	0.360	0.782
Shannon diversity index	**2.518**	**0.065**	**11.593**	**0.001**	1.909	0.136
Species richness	2.060	0.113	**31.272**	**3.79e-07**	**2.877**	**0.0419**
Native plants	**7.457**	**0.000**	**66.797**	**7.29e-12**	**4.990**	**0.003**
Non-native plants	**11.552**	**2.88e-06**	0.276	0.601	1.466	0.231

Species	Soil Prep		Seed/No Seed		Soil Prep + Seed	
	F-value	**P-value**	**F-value**	**P-value**	**F-value**	**P-value**
Native graminoid	**5.630**	**0.002**	**15.902**	**0.000**	**6.522**	**0.001**
Non-native graminoid	0.699	0.556	0.139	0.710	1.765	0.161
Native forb	**4.758**	**0.004**	**24.229**	**5.25e-06**	**3.480**	**0.0202**
Non-native forb	**9.395**	**2.56e-05**	0.098*	0.755	1.765	0.161

* trend

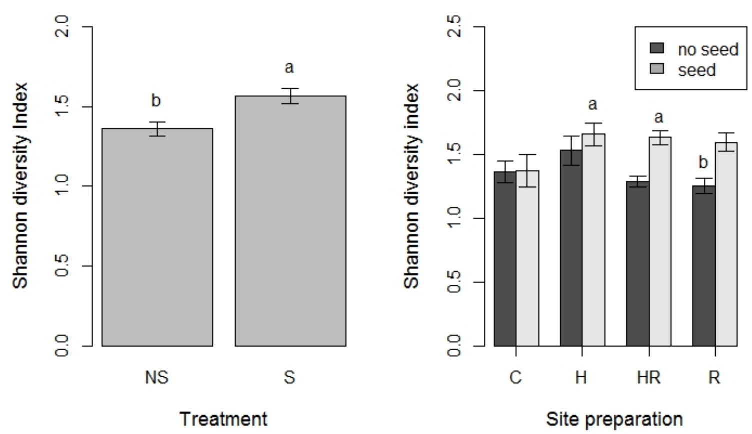


Figure 1: Analysis of Shannon diversity on stockpiled topsoil in the summer of 2013 at New Gold's New Afton mine site near Kamloops, BC. Treatments included seed (S), no seed (NS), raking (R), hydroslurry (H), and no manipulation (control (C)). Treatments labelled with different letters are significantly different (p<0.05).

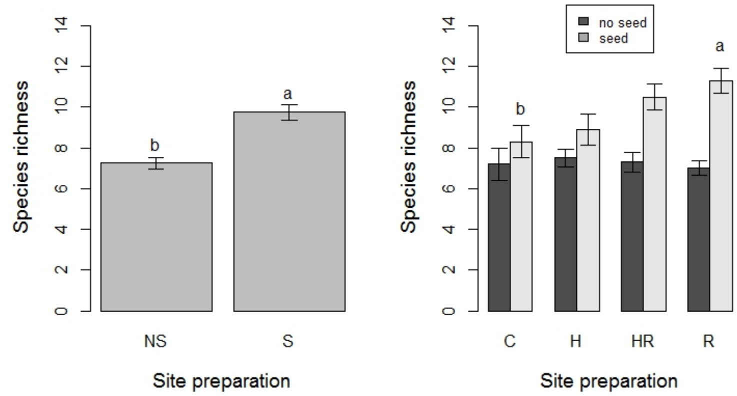


Figure 2: Species Richness based on seeded (S) and unseeded (NS) treatments as well as treatments that consisted of hydro-slurry (H) and/or raking (R). All treatments were carried out within a grid block located on stockpiled topsoil at New Gold's New Afton Mine site. Treatments labelled with different letters are significantly different (p<0.05).

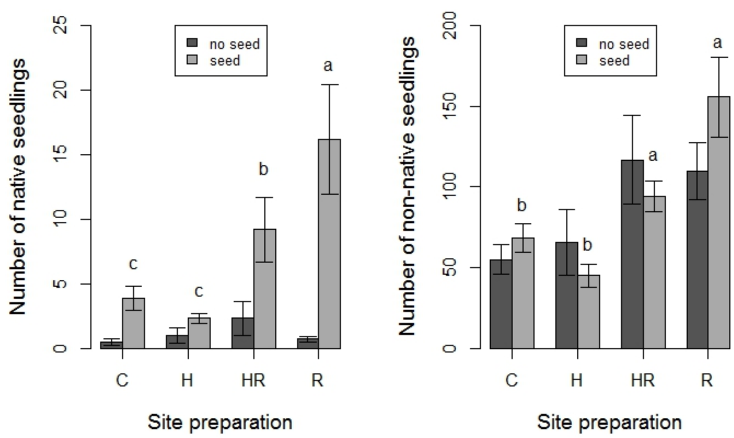


Figure 3: Results of seeding and soil preparation on stockpile topsoil at New Gold's New Afton Mine in the summer of 2013. Overall seedling count on plots treated with either raking (R), hydroslurry (H), both raking and hydroslurry (HR), control or no manipulation (C) and each of these treatments was either seeded or not seeded. Treatments labelled with different letters are significantly different (p<0.05).

Looking at the treatments from a group functional level (native forbs, native graminoids, non-native forbs and non-native graminoids), there were treatment effects for all groups except for the non-native graminoids (Table 1). Neither hydro-slurry nor raking had significant positive or negative effects on the germination of non-native grasses. However, there appears to be a slight increase in the number of non-native seedlings found on hydro-slurried sites when comparing it to raked sites. Raking did significantly increase the establishment of native graminoids, while adding hydro-slurry decreased the number of established native graminoid seedlings (Figure 4). The addition of raking to hydro-slurry did not result in a significant increase in the number of seedlings established. The non-native and native functional grouping of forbs revealed a positive response to raking for both groups and hydro-slurry resulted in a significant negative response for both native and non-native forbs. When hydro-slurry and raking were combined the result was a slight decrease in the number of seedlings counted for both non-native and native forbs as compared to the raked only sites (Figure 5). The control sights showed a slightly higher seedling count than the hydro-slurry treatment, but there was no significant difference between these two soil preparation methods.

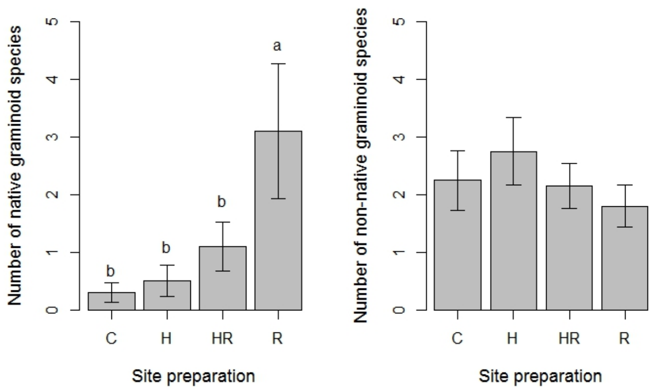


Figure 4: Native (left) and non-native (right) graminoids are depicted above showing the response to soil preparation carried out in a study looking at seedling establishment on stockpiled topsoil. C = control, H = Hydro-slurry, R = Raking. Treatments labelled with different letters are significantly different (p<0.05).

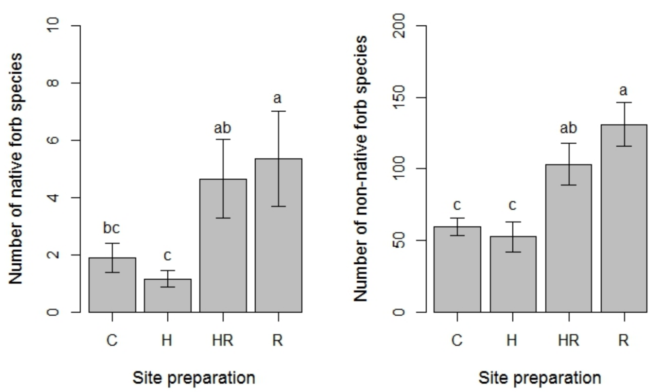


Figure 5: Response of native (left) and non-native (right) forb species to different soil preparations on stockpiled topsoil (C = control, H = hydro-slurry, R = Raking). Note the scale for the non-native forbs is 20x greater than for the native forb species. Treatments labelled with different letters are significantly different (p<0.05).

DISCUSSION

We hypothesized that raking would increase the number of native seedlings established on a site and results did show a positive response for both native graminoids and forbs. The positive effect of raking was most likely due to seed soil contact, which is enhanced by roughing up the surface of the soil (*An*

Integrated Approach to Establishing Native Plants 2007). As well, the crevices created by raking create areas of higher moisture, humidity, wind protection and shade from the sun (Montalvo et al. 2002). However, our method of raking only created ridges approximately two centimeters in height. Even though we attempted to loosen the soil with shovels the effect of roughing and loosening the soil to enhance seeding establishment, may diminish as the seedling roots reach below this loosened area into more compact soil. Certainly some of our plots were more compact than others. This was indicated by the difficulty in putting in our grid stakes as well as the difficulty we had in penetrating our shovels into the topsoil to loosen and roughen the raked sites.

Our objective with the hydro-slurry was to suppress unwanted/non-native species. In our study the hydro-slurry did not have the intended negative effect on non-native graminoids, but there was a negative effect on the native graminoids. Both native and non-native forbs were suppressed by the addition of hydro-slurry. Forbs overall, however, responded positively to the raking treatment. As one of the objectives to hydro-seeding was to suppress non-native species; the hydro-slurry treatment did have the intended effect. Unfortunately, the hydro-slurry also suppressed native forbs. In terms of management, hydro-slurry will make it difficult to re-establish native graminoids and forbs while controlling the non-native species.

At a species level, we found responses to the different treatments were selective. *A. millefolium* responded positively to either no soil management or raking, while hydro-slurry resulted in a negative response. *B. sagittata* and *G. aristata* shared a positive response to the double treatment of hydro-seeding and raking together. This could possibly be due to their larger seed size. *A. millefolium* seeds are small (~0.14 mg) and it's possible they get trapped in the hydro-slurry which may result in desiccation when the slurry dries out. It's also possible that as the hydro-slurry dries it shrinks inhibiting newly formed roots from making contact with the soil. The fibre and tackifying agent within the slurry may also create a mat which allows little light penetration thus reducing germination. A review by Moles and Westoby (2006) found studies indicated the increased survival rate of large seeded species through such conditions as shade, drought, competition and defoliation. Large seeds have the capacity to withstand short-term environmental stresses due to their stored reserves (Westoby et al. 1996). However, smaller seeded species like *A. millefolium* may not have the carbon reserves needed in times of short-term stress such as being able to penetrate the hydroslurry and access both the light above and soil below the slurry. Being of larger size the *B. sagittata* and *G. aristata* seeds may have the carbon stores needed to penetrate the hydro-slurry.

Unlike the native forbs, native graminoids responded positively only to raking. Raking resulted in a significantly positive response from *E. trachycaulus* and *P. spicata*. Hydroseeding native grasses has a negative effect and the non-native graminoids *A. cristatum* and *B. tetorum* showed no significant response to any of the treatments. As the hydro-slurry does not significantly inhibit the growth of non-native grasses and does not benefit native grasses, it is the researcher's opinion that hydroslurry is not feasible in terms of the restoration of native grasses in the BC Interior.

Some non-native forb species that appear to be primary successional species showed some negative response to hydro-slurry, e.g. *K. scoparia,* a Eurasian species. Mustards also appear in the first year after disturbance. Both of these species appear to be ephemeral and as the site ages they become less common. These non-native species hold little threat to the native species trying to re-establish themselves. In fact, these ephemeral species may, in some ways, help native species take hold by creating areas of shade and reducing wind and water erosion. They may also play an important role in changing the soil parameters to better suit native species. For example, *K. scoparia* is a halophyte and may remove salinity from the disturbed soils. These factors may outweigh the resources these plants use in the short-term. Other species, such as *S. tragus,* are considered invasive, noxious species and stay around long term making it difficult for native species to establish.

Raking may create micro-climates on the soil surface. The loosening and roughing up of soil may form pockets thus protecting seeds from wind and water erosion that may otherwise remove them from the site. These microclimates may create areas of more or less moisture, warm and cool zones as well as increase the boundary layer between the soil and the atmosphere lowering the effects of wind reducing the amount of evaporation from the soil surface. Seed radicles may benefit from the loosening of soil creating better seed soil contact. The loosened soil may improve root penetration thus helping to anchor the young seedlings as well as allowing them better access to water, nutrients and oxygen below the soil surface.

CONCLUSION/MANAGEMENT

This study showed that hydroslurry can reduce the number of non-native species, however raking will somewhat negate this effect. Hydroslurry was also found to inhibit native species. *B. sagittata* benefitted from the hydro-slurry only when raking was also done. These results indicate that the use of hydroslurry in the restoration of BC interior grasslands is not feasible in terms of successful reintroduction of native species and therefore it is also not economically feasible. Our study supported similar studies in which raking or roughening up the soil, before seeding native species, resulted in increased diversity and species richness. However, it also increased the number of non-native species. It would be valuable to further study this effect in terms of deep tilling. The hollows created may allow for the accumulation of water, organic matter and seeds. It may also be beneficial to set up shade netting vertically in rows to reduce wind erosion and thus slow water evaporation from newly establishing sites (Carrick and Krüger 2007). Further research should be carried out in this area. Transplanting native woody species may also help with shading, soil stability, wind erosion, and the capture of precipitation especially snow in the winter months. A phased approach to planting (Burke 2003) may also be warranted as some species like *B. sagittata* appear to need litter in order to successfully germinate. Restoration may be more successful if seeding were to take place in intervals, thereby creating a facilitated succession. It is recommended that further studies be done in this area.

REFERENCES

Alday, J. G., R. H. Marrs, and C. Martínez-Ruiz. 2011. Vegetation succession on reclaimed coal wastes in Spain: the influence of soil and environmental factors. Applied Vegetation Science 14:84–94.

An Integrated Approach to Establishing Native Plants. 2007. Vancouver, Washington.

Baasch, A., A. Kirmer, and S. Tischew. 2012. Nine years of vegetation development in a postmining site: effects of spontaneous and assisted site recovery. Journal of Applied Ecology 49:251–260.

Bochet, E., and P. Garcıa-Fayos. 2004. Factors Controlling Vegetation Establishment and Water Erosion on Motorway Slopes in Valencia ,. Restoration Ecology 12:166–174.

Brady, N. C., and R. R. Weil. 2012. The nature and properties of soils. 14 Edition. Pearson Education, Inc., publishing as Prentice Hall, Upper Saddle River, New Jersey.

Burke, A. 2003. Practical measures in arid land restoration after mining - a reveiw.pdf. South African Journal of Science 99:413–417.

Carrick, P. J., and R. Krüger. 2007. Restoring degraded landscapes in lowland Namaqualand: Lessons from the mining experience and from regional ecological dynamics. Journal of Arid Environments 70:767–781.

Carter, D. L., and J. M. Blair. 2012. High richness and dense seeding enhance grassland restoration establishment but have little effect on drought response. Ecological applications : a publication of the Ecological Society of America 22:1308–19.

Christian, J. M., and S. D. Wilson. 1999. Long-Term Ecosystem Impacts of an Introduced Grass in the Northern Great Plains. Ecology 80:2397–2407.

Cimate Data, Environment Canada. 2014.
http://climate.weather.gc.ca/climate_normals/station_metadata_e.html?StnId=1275.

Cosgriff, R., V. J. Anderson, and S. Monson. 2004. Restoration of communities dominated by false hellebore. Journal of Range Management 57:365–370.

Dornbush, M. E., and B. J. Wilsey. 2010. Experimental manipulation of soil depth alters species richness and co-occurrence in restored tallgrass prairie. Journal of Ecology 98:117–125.

Dunifon, S. N., G. K. Evanylo, R. O. Maguire, and J. M. Goatley. 2011. Soil nutrient and fescue (Festuca spp .) responses to compost and hydroseed on a disturbed roadside soils. Compost Science & Utilization 19:147–151.

Fraser, L. H., and J. P. Grime. 1999. Interacting effects of herbivory and fertility on a synthesized plant community. Journal of Ecology 87:514–525.

Grasslands Conservation Council. 2012. http://www.bcgrasslands.org/learn-more/grasslands-of-bc.

Johnson, D. B., J. C. Williamson, and A. J. Bailey. 1991. Microbiology of soils at opencast coal sites. I. Short- and long-term transformations in stockpiled soils. Journal of Soil Science 42:1–8.

Josa, R., M. Jorba, and V. R. Vallejo. 2012. Opencast mine restoration in a Mediterranean semi-arid environment: Failure of some common practices. Ecological Engineering 42:183–191.

Kiehl, K., A. Kirmer, T. W. Donath, L. Rasran, and N. Hölzel. 2010. Species introduction in restoration projects – Evaluation of different techniques for the establishment of semi-natural grasslands in Central and Northwestern Europe. Basic and Applied Ecology 11:285–299.

Loydi, A., R. L. Eckstein, A. Otte, and T. W. Donath. 2013. Effects of litter on seedling establishment in natural and semi-natural grasslands: a meta-analysis. Journal of Ecology 101:454–464.

Martin, L. M., and B. J. Wilsey. 2006. Assessing grassland restoration success: relative roles of seed additions and native ungulate activities. Journal of Applied Ecology 43:1098–1109.

Martínez-Ruiz, C., B. Fernández-Santos, P. D. Putwain, and M. J. Fernández-Gómez. 2007. Natural and man-induced revegetation on mining wastes: Changes in the floristic composition during early succession. Ecological Engineering 30:286–294.

Matesanz, S., F. Valladares, D. Tena, M. Costa-Tenorio, and D. Bote. 2006. Early Dynamics of Plant Communities on Revegetated Motorway Slopes from Southern Spain: Is Hydroseeding Always Needed? Restoration Ecology 14:297–307.

Moles, A. T., and M. Westoby. 2006. Seed size and plant strategy across the whole life cycle 113:91–105.

Montalvo, A. M., P. a. McMillan, and E. B. Allen. 2002. The Relative Importance of Seeding Method, Soil Ripping, and Soil Variables on Seeding Success. Restoration Ecology 10:52–67.

Oliveira, G., A. Nunes, A. Clemente, and O. Correia. 2012. Testing Germination of Species for Hydroseeding Degraded Mediterranean Areas. Restoration Ecology 20:623–630.

Prach, K., and R. J. Hobbs. 2008. Spontaneous Succession versus Technical Reclamation in the Restoration of Disturbed Sites. Restoration Ecology 16:363–366.

Pywell, R. F., J. M. Bullock, A. Hopkins, K. J. Walker, H. Tim, M. J. W. Burke, S. Peel, and T. I. M. H. Sparks. 2002. Restoration of species-rich grassland on arable land : assessing the limiting processes using a multi-site experiment. Journal of Applied Ecology 39:294–309.

Random.org. 2014. www.random.org.

Rowe, H. I. 2010. Tricks of the trade: techniques and opinions from 38 experts in tallgrass prairie restoration. Restoration Ecology 18:253–262.

Suding, K. N., K. L. Gross, and G. R. Houseman. 2004. Alternative states and positive feedbacks in restoration ecology. Trends in ecology & evolution 19:46–53.

The ecology of the Interior Douglas-fir zone. 2014. Victoria, British Columbia.

The Mining Association of BC. 2013. www.mining.bc.ca.

Tormo, J., E. Bochet, and P. Garcıa-Fayos. 2007. Roadfill revegetation in semiarid mediterranean environments . Part II : Topsoiling , species selection , and hydroseeding. Restoration Ecology 15:97–102.

Vickers, H., M. Gillespie, and a. Gravina. 2012. Assessing the development of rehabilitated grasslands on post-mined landforms in north west Queensland, Australia. Agriculture, Ecosystems & Environment 163:72–84.

Wanner, M., and W. Dunger. 2001. Biological activity of soils from reclaimed open-cast coal mining areas in Upper Lusatia using testate amoebae (protists) as indicators. Ecological Engineering 17:323–330.

Westoby, M., M. Leishman, J. Lord, H. Poorter, and D. J. Schoen. 1996. Comparative ecology of seed size and dispersal. Philosophical Transactions: Biological Sciences 351:1309–1318.

Wilson, S. D., and A. K. Gerry. 1995. Strategies for mixed-grass prairie restoration- herbicide, tilling and nitrogen manipulation.pdf. Restoration Ecology 3:290–298.

Wilson, S. D., and D. Tilman. 1993. Plant Competition and Resource Availability in Response to Disturbance and Fertilization. Ecology 74:599–611.

Yurkonis, K. a., B. J. Wilsey, K. a. Moloney, and A. G. Van Der Valk. 2008. The Impact of Seeding Method on Diversity and Plant Distribution in Two Restored Grasslands. Restoration Ecology 18:311–321.

ASSESSMENT OF VEGETATION CHANGE AFTER BIOSOLIDS TREATMENT: USE OF REMOTELY SENSED VEGETATION TIME SERIES

Mar Martínez de Saavedra Álvarez[1]
Leslie N. Brown[1]
José B. Lim[1]
Kaan Ersahin[1]
Gary A. Borstad[1]
Jaimie Dickson[2]
Peter Martell[2]

[1]ASL Environmental Sciences Inc.[*], Victoria, BC

[2] Teck Highland Valley Copper Partnership, Logan Lake, B.C.

ABSTRACT

Highland Valley Copper has run an experimental treatment program for many years, to use treated, de-watered sewage sludge (biosolids) as a supplement to the capping of waste rock and tailings materials before revegetation. This is an attempt to improve water retention on coarse-grained materials, as well as provide a source of nutrients. In 2012, we used the 12-year remotely sensed vegetation time series compiled by Teck and ASL to evaluate the effects of biosolids applications at selected tailings and waste rock sites. The vegetation maps allowed us to compare rates of vegetation change at a number of these treated areas with nearby untreated sites. Though our findings were based on a rather small number of sites, we concluded that while the short-term effects of biosolids on vegetation growth were site dependent, over the longer term (~10 years or more) there was a small, positive effect at all sites, in the form of increased growth rates at biosolids treated sites relative to nearby untreated sites.

Qualitative assessment of community composition showed that, like the short-term effects on growth rate, the effects of biosolids varied by site. In some areas, the application of biosolids appeared to have little influence on community composition. In these locations, capping with overburden had a greater effect. In other areas, treatment with biosolids appeared to promote communities dominated by grasses. This is in agreement with an earlier study that also showed a response to biosolids treatment by grasses.

Keywords: Highland Valley Copper, reclamation, vegetation, remote sensing, waste rock, tailings.

INTRODUCTION

Highland Valley Copper (HVC) near Kamloops, British Columbia has had a major mine reclamation program underway since 1983, in which the establishment and development of self-sustaining

---

[*] Borstad Associates Ltd. merged with ASL Environmental Sciences Inc. in 2009, and has since operated as the remote sensing division (ASL Borstad Remote Sensing). ASL is a corporate member of the Canadian Land Reclamation Association (CLRA).

vegetative cover are important elements. The success of the program has been monitored since its inception using standard biological sampling techniques. Aerial multispectral surveys were initiated in 2001 to monitor the success of the vegetation establishment over the entire mine, and have been acquired annually since then. In addition, high resolution satellite imagery has been acquired since 2011. Previous reports on this work (Borstad Associates, 2006; ASL Borstad Remote Sensing, 2009-2013) have demonstrated the ability of the remote sensing surveys to provide synoptic, quantitative and thematic maps of the vegetation cover on the mine site. The continuous, 2-dimensional coverage they provide can be used to extrapolate from and between the more detailed observations at ground sampling sites, with the additional value of information on temporal changes over large areas, some of which are inaccessible from the ground.

As part of their reclamation program, HVC began in 1996 an experimental treatment program using treated, de-watered sewage sludge (biosolids) to supplement the capping of waste rock and tailings materials before revegetation (Bloodgood *et al*., 1998). It was hoped that biosolids could improve water retention on coarse-grained materials, as well as provide a source of nutrients. The program became operational in 1998. A 2003 study examined the effects of the biosolids on soil and foliar chemistry, soil water retention, and the biomass and composition of subsequent vegetation development (Straker *et al.*, 2003). This document reports the results of a 2012 study of the effects of biosolids applications on subsequent vegetation growth at experimental sites, using the 2001-2012 remote sensing time series.

METHODS

Remote sensing data

Aerial multispectral surveys over revegetated areas have been performed every year between 2001 and 2012 (except for 2004) using a Compact Airborne Spectrographic Imager (CASI) (Richards *et al.*, 2003; Borstad *et al.*, 2009; Martínez *et al.*, 2013). In an endeavour to move from airborne to satellite technology, in 2011 and 2012 we also acquired a multispectral image data from the Worldview-2 (WV2) and Quickbird-2 (QB2) satellites, respectively. The satellite and the airborne data were acquired within one day of each other. . The acquisition of both airborne and satellite data allowed us to cross-calibrate the two datasets before integrating the satellite data into the existing time series.

Vegetation indices and remote sensing biomass

Two vegetation indices were calculated from the satellite and airborne reflectance data. The ‘Greenness Index’, also called the Normalized Difference Vegetation Index (NDVI), is a well-known index commonly used in remote sensing to serve as a proxy for green vegetative cover or biomass (Lyon *et al*., 1998; Peñuelas and Filella, 1998; Rouse *et al*., 1974). Our ‘Normalized Yellow Index’ provides an index of desiccation and/or senescence to assist with interpretation of low NDVI values (Borstad Associates, 2006). As the CASI, WV2, and QB2 spectral configurations are different, the vegetation indices were calculated using slightly different wavelengths. Before including the WV2 and QB2 NDVI in the time series, cross-calibration was performed by pairwise comparison of NDVI calculated from both WV2 – CASI and QB2 - CASI sensors. For application to Highland Valley, we calibrated NDVI to ‘remote sensing biomass’ (RSB) based on a comparison of CASI and *in situ* biomass measurements made over a wide range of conditions between 2001 and 2005 (Borstad Associates, 2006).

Study sites

Study sites included tailings impoundments (Trojan and Highmont) and waste rock dumps (Bethlehem, Table 1). Among the tailings sites, some had been capped with overburden and others had not. All waste rock sites were capped. In general, capping is applied at HVC where deposits are unsuitable to permit direct revegetation (Straker *et al*., 2003). For each class of site (tailings or waste rock, capped or not), untreated sites located close to biosolids treated sites and similar in age were studied as controls. A total of 16 biosolids treated sites and 14 untreated controls were used.

Table 1. Summary of sites selected for biosolids analysis, showing dates of seeding, capping and biosolids application. Blue shading indicates biosolids treated sites.

Area and Material		Site ID	Capped	Biosolids	Seeded/Planted
Trojan Tailings (not capped)		2	no	1996	1996
		3	no	2000	2000
		4	no	2001	2001
		9, 10	no	2001	2001, 2004 (partial)
		11	no	2001	2001
		5, 8	no	no	1991, 1998
		6	no	no	1993, 1998
		7	no	no	1998
Trojan Tailings (capped)		16	2003	2005	2005
		1, 12	2003	no	2004, 2005
		15	2003	no	2004
Highmont Tailings (not capped)		13	no	2008	2008
		14	no	2000	2000
		12	no	no	1996
Bethlehem Waste Rock Dumps (capped)	North	D, E	pre-2002	1999	1999
		F, G	pre-2002	no	1994, 1997
		H	pre-2002	no	1992, 1994 (partial)
	West	M	pre-2002	1997	1998
		N	pre-2002	1997	1984, 1987, 1998
		O	pre-2002	no	1987
		P	pre-2002	no	1984
	South	J	pre-2002	1997	1997
		K, L	2007	1997, 2007	1983, 1997
		I	pre-2002	no	1984

The dates given in Table 1 show that the sites ranged in age from 30 years since their first seeding or planting to 5 years for the most recent site at Highmont. Waste rock sites tended to be older than tailings sites, all (with the exception of sites K and L) having been seeded and (where applicable) treated with biosolids at least 3 years before the beginning of the remote sensing time series. Many of the tailings sites were seeded and/or treated during the course of the remote sensing time series (since 2001), which was useful for the assessment of shorter-term effects of biosolids. Treatments applied prior to the most recent disturbance at each site are not listed.

A variable that was not taken into account during this study was the *application rate* of the biosolids, and whether they were top-dressed or incorporated. Any effect was expected to be minor, since, according to the Straker *et al*. (2003), biosolids applications of 50-200 dry tonnes per hectare yielded similar vegetation biomass after 2-3 years; as well, top-dressing has not been used since the initial trials in 1996 and 1997. We note that HVC sites that do not receive biosolids are typically treated annually with chemical fertilizers for the first 3 (for waste rock) to 4 (for tailings) years after seeding.

Field observations

Vegetation community composition was qualitatively estimated from site photographs acquired in August 2012.

RESULTS

Figures 1 and 2 show mean NDVI time series for all sites included in the biosolids study. In general, the NDVI trajectories for sites with similar histories were similar, with the possible exception of site 15, which had greener vegetation than other similar sites in 2005 (Figure 1D).

Short-term effects

Of the 16 biosolids treated sites, eight (tailings sites Trojan 4, 9-11, 16, Highmont 13 and waste rock sites Bethlehem K and L) received biosolids during the years for which NDVI data are available. The timing of biosolids additions is indicated by the coloured arrows in Figure 1A, C, E and Figure 2E. In each case, NDVI dropped to zero (or less) in the year in which biosolids were applied, indicating coverage of the existing vegetation by the biosolids.

At Highmont Tailings (Figure 1E) and Bethlehem (Figure 2E), biosolids treated sites demonstrated clear increases in growth in the years following their 2007-08 treatment, relative to untreated sites or sites that had been treated before 2001 (Figures 1F and 2F). However, at Trojan the trends were different: Trojan sites displayed little or no change in growth rates in the years following the 2001 biosolids application (Figure 1A) – in fact their early growth patterns were similar to untreated sites (Figure 1B). On the other hand, capped sites displayed rapid growth in the years following the 2003 capping, whether they were treated with biosolids or not (Figure 1C and D).

It appears that at Trojan, capping with overburden provided growth-promoting conditions in a way that biosolids did not. Although we do not know the physical properties of the two types of materials, it may be that overburden is better at retaining moisture, for example, than biosolids. In a previous study, we showed that vegetation at Trojan Dam was highly responsive to precipitation, whereas at Highmont Tailings it was not (Borstad Associates, 2009; Martínez *et al*., 2011). There are, of course, other differences between these sites that make it hard to be sure of the underlying mechanisms. The uncapped Trojan sites, for example, were treated earlier than the other sites; perhaps there were differences in biosolids composition, application rates or application methods that changed over time.

To summarize, our analysis suggests that short-term biosolids effects vary by site. At Highmont, biosolids treatment resulted in increased growth, whereas at Trojan, capping with overburden appeared to be more important for growth promotion. Since Bethlehem sites received both biosolids and capping in the same year, we cannot resolve the separate effects of these treatments at this site. This interpretation is made on the basis of only a handful of sites; an examination of more sites is warranted to draw more definite conclusions.

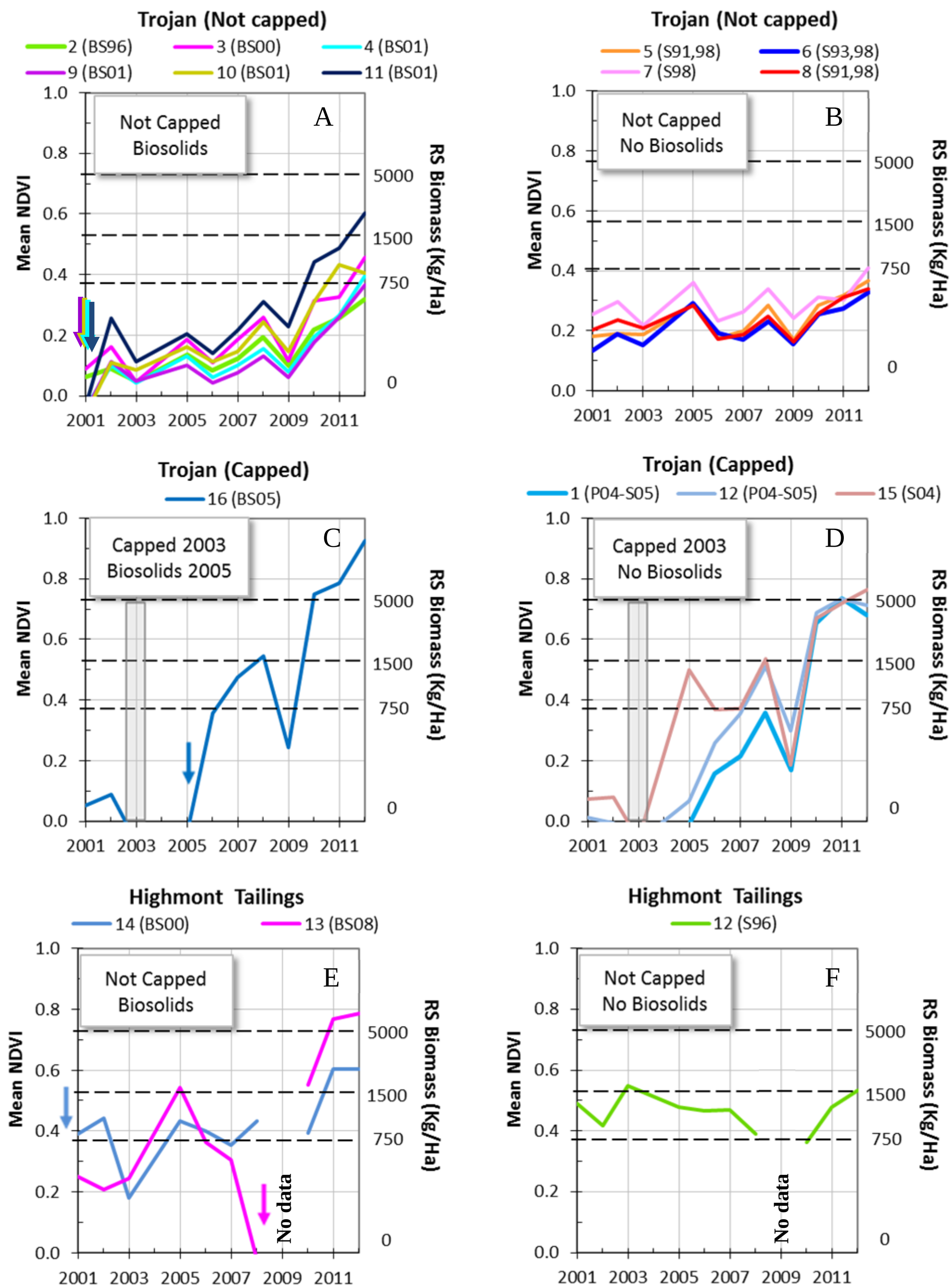


Figure 1. Summary of NDVI time series at tailings sites.

The horizontal dashed lines represent NDVI corresponding to reclamation thresholds[‡] at 750, 1,500 and 5,000 kg/ ha. Coloured arrows indicate the timing of biosolids treatments. Grey bar indicate years when sites were capped with overburden. (BSyy) = year of biosolids application, (Syy) = year of seeding, (Pyy) = year of planting.

[†] When this work was undertaken in 2012, biomass thresholds were used to determine reclamation success. However, Teck is currently in the process of developing new performance indicators that will include biodiversity.

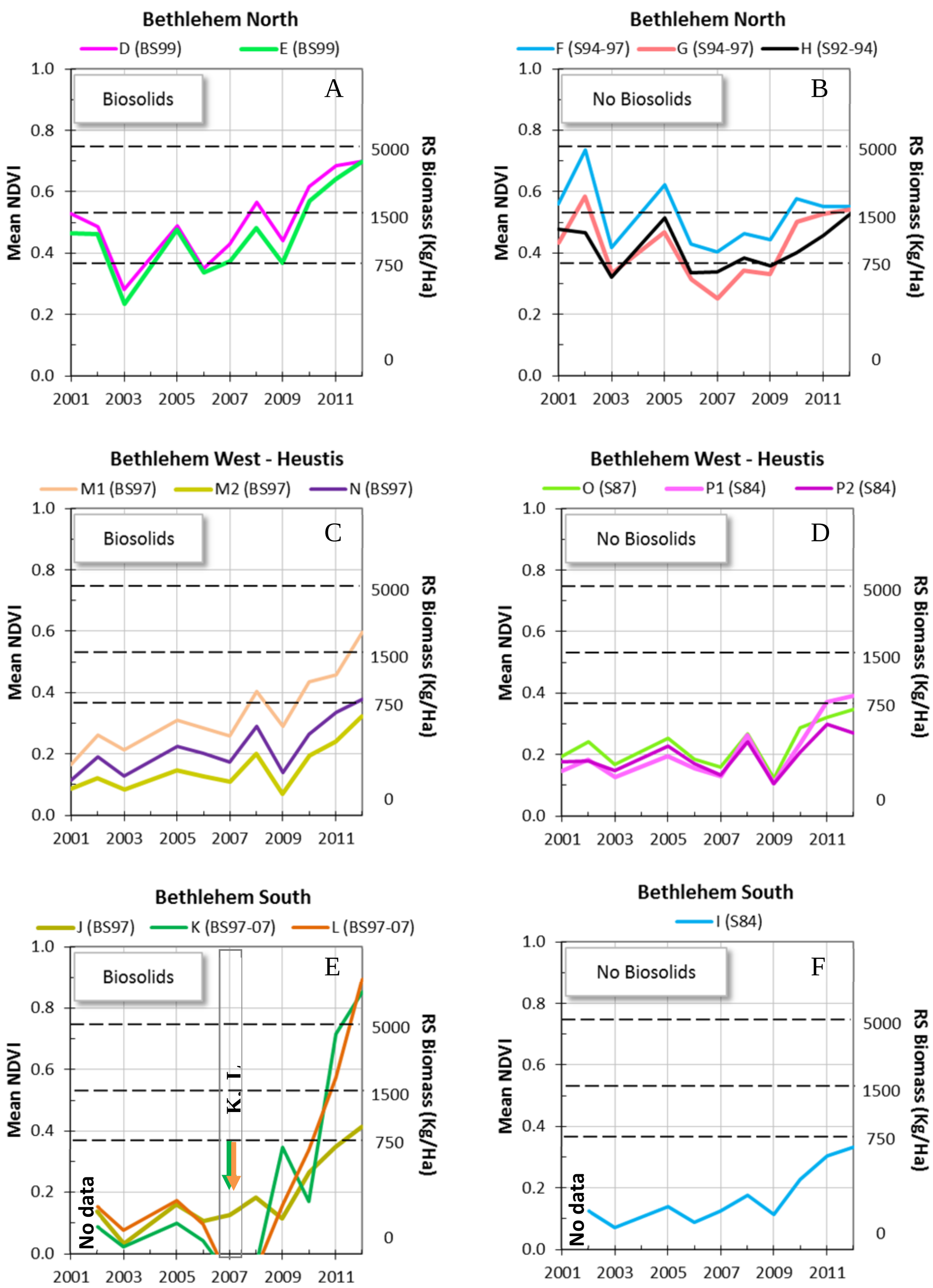


Figure 2. Summary of NDVI time series at waste rock sites. For an explanation of annotations see Figure 1. With the exception of sites K and L, biosolids treatments were applied prior to the beginning of the remote sensing time series.

Longer term effects

An interesting facet of the time series at all sites is a rise in NDVI in the last 3 years of the record following a low year in 2009 for the majority of sites. The reason for the drop in 2009, which at some sites was quite strong, is unclear – 2009 was not unusually hot or dry in the weeks preceding the remote sensing survey – however, the effect was apparently transitory, as sites that experienced a decrease in 2009 all appeared to have fully recovered in 2010.

In general, biosolids sites displayed steeper post-2009 increases than non-biosolids sites at the same location even when biosolids treatment occurred before 2002. This phenomenon, summarized in Figure 3 for all sites treated prior to 2002, was statistically significant for the 2010-12 rates of change ($p=0.0027$ for a 1-tailed t-test), and also for the 2001-09 rates of change (p (1-tailed) = 0.0005). These findings suggest that the addition of biosolids benefits vegetation growth in the medium and long term.

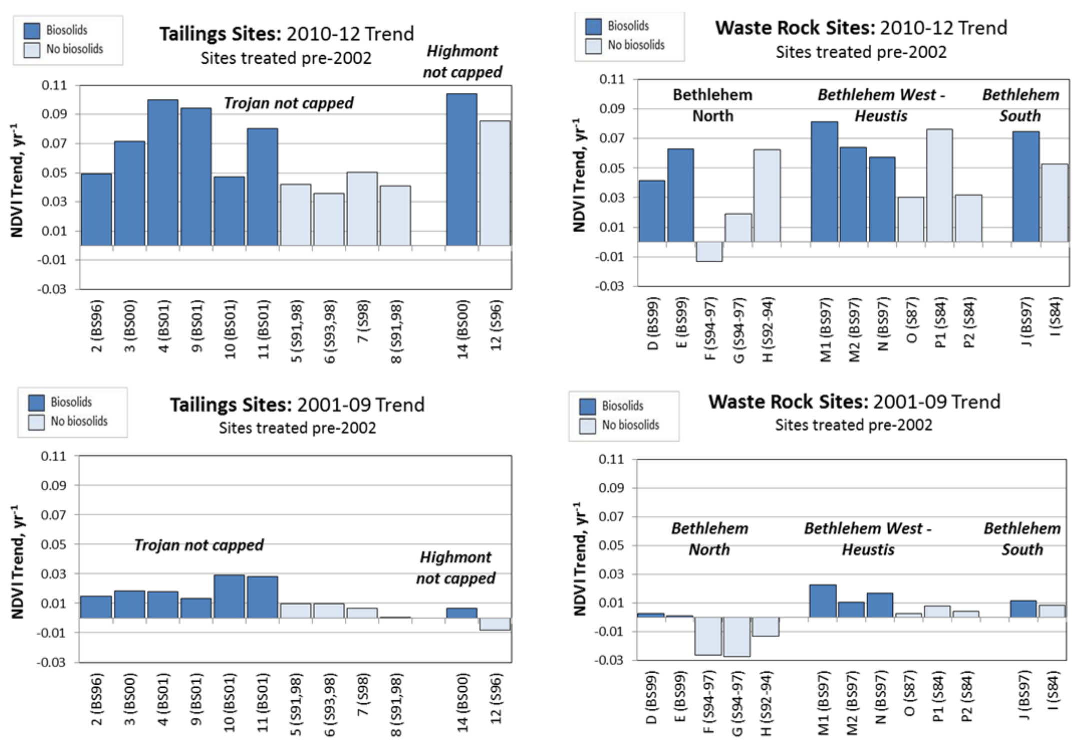


Figure 3. NDVI rates of change between 2010 and 2012 (left) and 2001-2009 (right), for sites treated with biosolids and/or capped with overburden before 2002, and corresponding untreated sites. Dark blue bars indicate biosolids treated sites; light blue bars untreated sites.

As observed for the shorter term, there also appear to be effects of capping in the medium and long term. However, the effects were reversed, such that capped sites grew more slowly than non-capped (Figure 4, $p=0.02$ for 2010-12 and 0.09 for 2001-09 growth rates). On the other hand, of the sites that were treated before 2002, all capped sites were waste rock and all non-capped sites were tailings, so we cannot be certain that these differences were due to capping and not differences in substrates.

Our conclusion is that in the long term there are beneficial effects that can be attributed to biosolids. The long-term effects of capping are less clear in that they cannot be separated from differences due to substrate material.

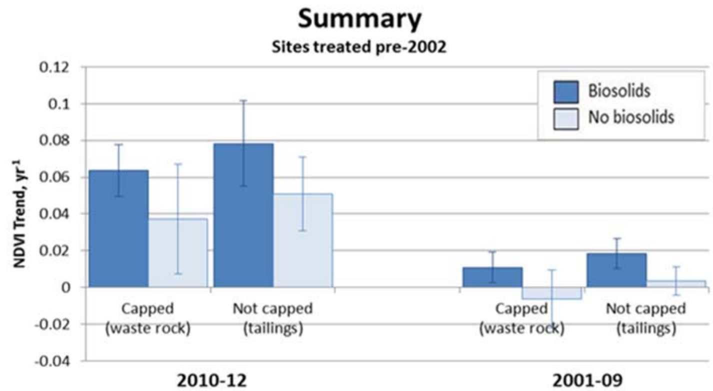


Figure 4. Summary of the effects of biosolids treatment and capping on NDVI rates of change between 2010-12 and 2001-09. Plot shows the means and standard deviations for sites shown in previous figures.

Qualitative effects

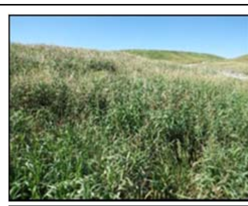
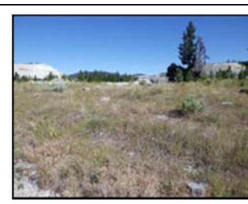
Photos in Table 2 illustrate the nature of the plant communities at biosolids and non-biosolids test sites.

For non-capped sites at Trojan Tailings, we observed little difference between sites with and without biosolids treatment: all were relatively sparsely vegetated with heterogeneous plant communities. An exception was site 11 (treated with biosolids in 2001) which was more densely vegetated. This site is in area affected by floods in early spring, so the denser vegetation may reflect greater water availability there. Trojan capped sites were also characterized by similar vegetation at biosolids and non-biosolids sites, but in this case it was dense grasses. We concluded that at Trojan the strongest influences on vegetation density and composition appear to be capping and possibly water availability, not biosolids treatment. This is consistent with the findings from the NDVI time series which indicated an effect of capping.

At Highmont Tailings (not capped), the non-biosolids site 12 was highly diverse, with heterogeneous plant communities varying from very sparse to dense. Biosolids sites (13 and 14) were more densely vegetated with a higher percentage of grasses, except for the north end of site 13 which was sparsely vegetated, chiefly with non-grass species. However, in this area exposed tailings were visible suggesting it may have received only limited quantities of biosolids. It should also be kept in mind that at Highmont the non-biosolids site was more mature (seeded 1996) than the two biosolids sites (seeded 2000, 2008).

Waste rock sites were less well documented than the tailings sites, but for most of Bethlehem, biosolids appeared to be promoting strong growth of grasses, for both recently planted (K) and more mature sites (D). We only have one photo of a non-biosolids site (I) in these areas, but there the vegetation was heterogeneous and sparse. This is consistent with the findings at Highmont Tailings.

Table 2. August 2012 field photographs from tailings and waste rock sites.

Area and Material	**Biosolids**		**No Biosolids**	
Trojan Tailings Not Capped	 *Site 3*: very sparse vegetation, native species, subject to desiccation	 ***Site 11***: heterogeneous dense cover of grasses, mustard, natives, some alfalfa	 *Site 7*: heterogeneous, moderately sparse cover	 ***Site 8:*** heterogeneous, sparser cover
Capped	 ***Site 16***: homogeneous dense grasses		 *Site15:* homogeneous, dense grasses	
Highmont Tailings Not Capped	 *Site 13 north*: sparse cover of yarrow, alfalfa & grasses. Exposed tailings.	 *Site 13 south*: dense grasses with some alfalfa, yarrow	 *Site 12 east:* dense cover of grasses, legumes, and some mustard	 *Site 12 central*: heterogeneous, very sparse cover of grasses, yarrow, and cryptogamic crust
Bethlehem Waste rock Capped	 *Site E*: Homogenous dense grasses	 *Site K*: Homogenous dense grasses	 *Site I*: heterogeneous sparse cover, grasses, natives, some trees and shrubs	

The western portion of Bethlehem differed from the other areas at Bethlehem, in that the vegetation consisted of homogeneous, sparse grasses subject to desiccation at both biosolids and non-biosolids sites, probably due to limited water availability.

In summary, growth at Trojan appears to be limited by a factor that is not ameliorated by the addition of biosolids, but is improved by capping: from the discussion of short-term effects on growth rates, that factor may be related to water retention. In other areas (both Highmont Tailings and waste rock sites) biosolids appear to be promoting vigorous growth of grasses. This is consistent with the report by Straker *et al*. (2003) that "grass species account for almost all of the measured response to [biosolids] treatment on the trial sites." However, Straker's group reported declines in grasses following an initial (3-4 years) increase, which was not evident from our study. An exception to this trend is the western portion of Bethlehem (a waste rock area), where vegetation was limited at all sites, again perhaps by water.

CONCLUSIONS

We hope that the biosolids study will provide valuable information regarding the effects of biosolids applications, and particularly, *where* they have been useful and *where* other factors could be limiting reclamation success. We caution that our findings are based on a rather small number of sites, especially given the variety of site types. To increase the confidence in the findings, a follow-up study would be advisable. Other variables, such as biosolids application rate, source and class of biosolids, could be examined. Also, maps of NDVI change rates could be used for higher resolution analysis of variability in vegetation growth rates at biosolids-treated sites and surrounding areas, examining the patterns of growth rates corresponding to other factors such as slope or substrate. Spatial analysis of this type can provide additional levels of information beyond that achieved from examinations of specific sites.

ACKNOWLEDGEMENTS

We thank Randy C. Kerr and Peter Willis, members of the remote sensing group at ASL, for their valuable contributions during data acquisition and processing; and Richard Doucette of Teck Highland Valley Copper Partnership for providing input during the review of this paper.

REFERENCES

ASL Borstad Remote Sensing, Inc. (2013). Remote sensing of reclaimed areas: 2012 analysis. Teck-Highland Valley Copper 2012 Annual Reclamation Report vol II report 8.

ASL Borstad Remote Sensing, Inc. (2012). Multispectral mapping of reclaimed areas: 2011 aerial acquisition, integration of WV2 in the time series, and analysis of 2001-2011 vegetation changes. Teck-Highland Valley Copper 2011 Annual Reclamation Report vol II report 9.

ASL Borstad Remote Sensing, Inc. (2011). Multispectral mapping of reclaimed areas: 2010 aerial acquisition and analysis of 2001-2010 vegetation changes. Teck-Highland Valley Copper 2010 Annual Reclamation Report vol II report 10.

ASL Borstad Remote Sensing, Inc. (2010). Multispectral mapping of reclaimed areas: 2009 aerial acquisition. Teck-Highland Valley Copper 2009 Annual Reclamation Report vol II report 10.

Bloodgood, M.A., K. G. Gizikoff, C.E. Jones. 1998. Biosolids amendment of waste rock and tailings as reclamation treatment at Highland Valley Copper. Presented at the *22nd British Columbia Mine Reclamation Symposium* 98, Penticton.

Borstad Associates Ltd. 2006. *Multispectral mapping of reclaimed areas: extended analysis of the 2005 imagery*. Highland Valley Copper 2005 Annual Reclamation Report vol II report 8.

Borstad, G. A., L. Brown, M. Martínez, B. Hamaguchi, J. Dickson, and M. Freberg. 2009. Airborne mapping of vegetation changes in reclaimed areas at Highland Valley between 2001 and 2008. Presented at the *33rd British Columbia Mine Reclamation Symposium*, 14-17 September 2009, Cranbrook, BC, 10 p.

Jones & Associates Ltd. 2005. *Reclamation Permit No. M-11*, Volume 1. Unpublished report prepared for Highland Valley Copper by C. E. Jones & Associates Ltd., Victoria, B.C., 86 p.

Lyon, J.G., D. Yuan, R.S. Lunetta, and C.D. Elvidge. 1998. A change detection experiment using vegetation indices. *Photogrammetic Engineering & Remote Sensing, 64*(2), 143-150.

Martínez de S. Álvarez, M., L. Brown, G. A. Borstad, P. Martell, J. Dickson and M. Freberg. 2011. Assessment of reclamation status and identification of water stress using airborne remote sensing. Presented at *the 35th Annual TRCR Symposium and 6th International Mine Closure Conference*, 18-21 September 2001, Lake Louise, Alberta, 8p.

Martínez, M., G. Borstad, L. Brown, K. Ersahin, M. Henley. 2013. Remote sensing in vegetation monitoring: More than just a pretty picture. Presented at the *38th CLRA/ACRSD Annual Meeting and Northern Latitudes Mining Reclamation Workshop "Overcoming Northern Challenges,*" Whitehorse, Yukon, September 9-12, 2013.

Peñuelas, J. and I. Filella. 1998. Visible and near-infrared reflectance techniques for diagnosing plant physiological status. *Trends in Plant Science*, *3*(4), 151-156.

Richards, M., M. Martínez de Saavedra Álvarez, and G. A. Borstad. 2003. The use of multispectral remote sensing to map reclaimed areas at Highland Valley Copper. Presented at the *105th CIM Annual General Meeting*, 4-7 May 2003, Montréal, QC, 9 p.

Rouse, J.J., R.H. Haas, J.A. Schell, and D.W. Deering. 1974. Monitoring vegetation systems in the Great Plains with ERTS. Proceedings of the *Third Earth resources Satellite-1 Symposium 1* (Sect. A), 309-317, Maryland, NASA SP-351.

Straker, J., R. Hamaguchi, C. Jones, and M. Freberg. 2003. Seven years of research on reclamation using biosolids at Highland Valley Copper. Presented at the *27th Annual BC Mine Reclamation Symposium*, Kamloops, BC.

CAPILLARY BREAK COVER DESIGN AT LORADO MILL TAILINGS SITE

Alvin Tong, PEng[1]
Iozsef Miskolczi, PEng1
Dianne Allen[2]
Ian Wilson[2]
Maritz Rykaart, PhD, PEng[1]

SRK Consulting (Canada) Inc.,Vancouver BC

Saskatoon Research Council
Saskatoon, SK

ABSTRACT

The historical Lorado mill tailings site deposited uranium tailings within and adjacent to nearby Nero Lake. The remediation plans called for a capillary break tailings cover with in-lake water treatment to reduce radiation exposure from the tailings, limit tailings dust inhalation and acid generation, prevent efflorescent salts formation, and block tailings pore fluid uptake from vegetation. Detailed designs, including water and load balances were developed to establish appropriate in-lake water treatment design criteria and to set measurable load reduction targets for the cover efficiency. A borrow source investigation was carried out to confirm whether appropriate volumes of suitable cover materials were available for the proposed design. These studies showed original remediation plan assumptions were no longer valid. This paper will describe the engineering process that was followed to evaluate and ultimately adopt appropriate design criteria to ensure that realistic, site-wide closure objectives could be met. The authors will emphasize the importance of linking design criteria to measurable and achievable performance criteria that can be properly evaluated over the intended life of the design.

INTRODUCTION

The historical Lorado mill tailings site is located in Saskatchewan near Uranium City. The operation deposited uranium tailings within and adjacent to Nero Lake. As part of the overall site remediation plan, a capillary break cover for the tailings in conjunction with in-lake water treatment was identified to reduce the negative environmental effects of the site. The original objectives for the cover, as identified during the environmental assessment stage, were to reduce gamma and radon exposure, limiting inhalation of tailings dust, reduce tailings acid generation by preventing formation of efflorescent salts, and block uptake of tailings pore fluid into vegetation.

As part of the detailed engineering phase for the cover, a water and load balance was developed and a comprehensive borrow source investigation was carried out. The water and load balance was required to establish appropriate in-lake water treatment design criteria and to set measurable load reduction targets for the cover efficiency. The borrow source investigation was done to confirm whether appropriate volumes of suitable cover materials were available for the proposed design, specifically the coarse materials required for the capillary break. The outcomes of these studies concluded that some of the original design assumptions that led to the proposed cover design and associated design criteria were no longer valid and more appropriate cover design criteria were warranted.

This paper describes the engineering assessment to develop quantifiable criteria for design. Each of the criteria is supported by detailed analyses, risk evaluation and reasoning. One of the main purposes of the detailed assessment is to address and quantify identifiable variables by reducing the amount of assumptions and information gaps. The resulting parameters from the assessment would then become design criteria of the remediation design. The key engineering assessments included in this paper are water quality prediction, capillary break modeling, radiation attenuation, and water-cover assessment. There are other assessments completed for the cover design but are not described in this paper.

The authors will emphasize the importance of linking quantifiable design criteria to remediation objectives such that the success and performance of the cover can be properly measured over its intended design life.

BACKGROUND

In the 1950s uranium exploration and development boomed in the remote wilderness of northern Saskatchewan, resulting in the construction of three uranium mills and numerous uranium mines. The town of Uranium City, and a few other towns that no longer exist, developed in response to this activity. Interest in the area's resources dropped off within a few years as the markets changed and richer deposits were found elsewhere. The mine and mill sites were abandoned with little or no decommissioning. It wasn't until the 2000s that the provincial and federal governments took stock of the safety and environmental liabilities at these sites and made a commitment to deal with the risks they presented.

The historical Lorado uranium mill tailings site is one of these legacy sites. It is located 8 kilometers southwest of Uranium City, on the north shore of Lake Athabasca. The mill operated from 1957 to 1960, processing 305,000 tonnes of low-grade uranium ore and producing somewhere between 190,000 to 344,000 cubic meters of tailings deposited near the shore of and in the western part of Nero Lake. Originally tailings were deposited on land upstream of Nero Lake and over time the tailings have reached land capacity and spilled out into the western portion of the lake. A natural isthmus southeast of the Nero Lake was built up with an earth berm (also known as the land bridge) to artificially raise the lake level during mill operation. The land bridge now separates Nero Lake with approximately 3 m of head above Beaverlodge Lake.

Although not abandoned, the Lorado site was neglected as ownership changed hands, ending up as property of Conwest Exploration Company Ltd. This company oversaw the demolition of the mill buildings in 1990. When the parent company of Encana Corporation bought Conwest, the Lorado site was transferred as well. Encana contracted Golder Associates in 2004 to characterize the site. Some measures were taken at that time to reduce safety and environmental risks. In 2007, the property was transferred to the Saskatchewan Government along with funds for the site remediation.

The Saskatchewan Research Council (SRC) was contracted to manage the remediation of the former Lorado uranium mill site as well as the Gunnar uranium mine and mill site and 35 legacy uranium mine and exploration sites in northern Saskatchewan. SRC submitted a proposal to remediate the Lorado site to the Saskatchewan Ministry of Environment and Canadian Nuclear Safety Commission (CNSC) in 2009, where it was determined that an environmental assessment was required under both the *Saskatchewan Environmental Assessment Act* and the *Canadian Environmental Assessment Act*.

SRC managed the development of a Risk Reduction Plan (RRP) and an Environmental Impact Statement (EIS) for the site. The project endpoints were developed with input from community members of

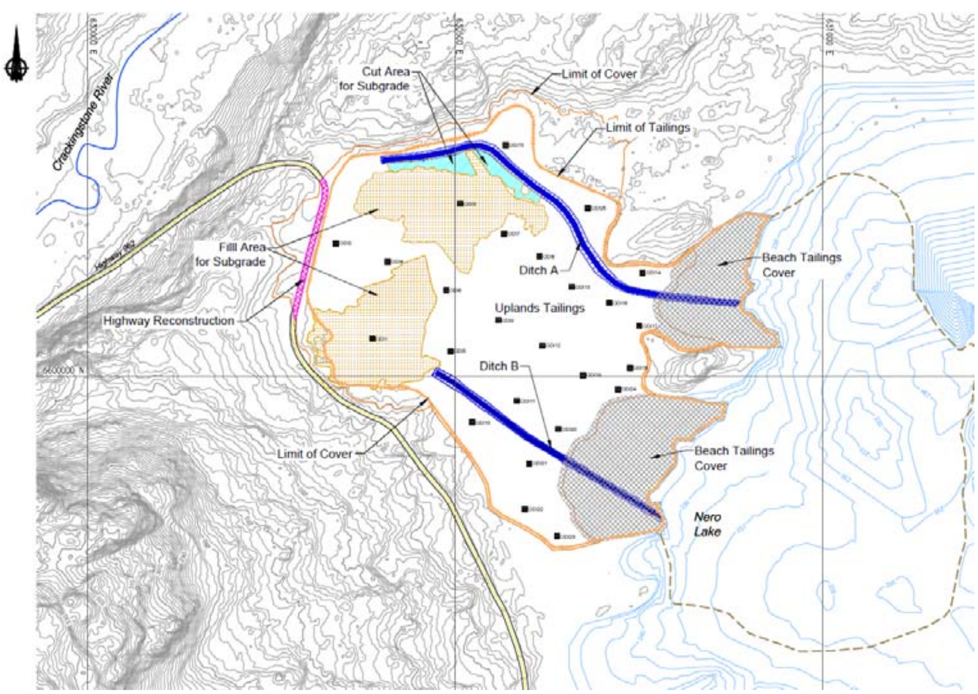
Uranium City and government regulators. The overall risk management goal identified for the Lorado site was that "risk reduction at the site has achieved an acceptable level of residual risk to human health, and to terrestrial ecological populations, and to the aquatic population of Hanson Bay and Beaverlodge Lake". The specific objectives for a tailings cover were to reduce gamma and radon exposure, while limiting inhalation of tailings dust, reduce tailings acid generation and prevent formation of efflorescent salts, and block uptake of tailings pore fluid into vegetation (Golder 2013).

SRC obtained a Waste Nuclear Substance License from the CNSC to manage the Lorado site in 2013. SRK Consulting (Canada) Inc. was contracted in 2013 to begin the detailed design for the remediation. The detailed design included a borrow investigation, water quality prediction and treatment, radiation attenuation analyses, and hydrologic and hydraulic evaluations (SRK 2014). The project to remediate the historical Lorado site was approved by the provincial government in February 2014.

TAILINGS COVER DESIGN

General

The Lorado Tailings Cover is approximately 14 ha in size with tailings upwards of 6 m deep. The cover is divided into upland and beach cover. The upland is defined as the cover portion that is above normal water level (NWL), and beach cover is the portion that covers the tailings below the NWL. The cover extends beyond the exposed tailings, tying into the natural terrain to prevent ponding and to facilitate runoff. A small section of provincial Highway 962 would be raised to match the cover and organic debris islands would be constructed at strategic locations to create microclimate and moisture traps to encourage natural vegetation growth. Figure **1** shows the site general arrangement of Lorado Tailings Cover.



Source: SRK, 2014

Figure 1: Lorado Tailings Cover General Arrangement.

Following the objectives described in of the RRP and EIS, the preferred option must be technically feasible, practically constructible, and sustainable. The chosen option is to cover the surface tailings in situ with an engineered cover system that incorporates a capillary break type soil cover, including tailings within the immediate beach area and the adjacent affected soil and vegetation. Within this context, the specific mitigation objectives for the cover were to limit, as much as practical, formation of efflorescent salts, potential for inhalation and intake of tailings dusts and salts, limit surface water infiltration into the tailings to not greater than the uncovered state, and reduce gamma and radon exposure to background levels.

To ensure these objectives were met over the life of the cover, a set of quantifiable and measureable criteria for long-term monitoring was adopted. The comparison between the monitoring data and the criteria determines if the cover is performing as intended and is meeting the remediation objectives. Hence, a set of practical and measurable criteria are crucial to the Lorado—and for that matter any—remediation project.

Design Criteria

Specific decommissioning and closure objectives for the remediation were not explicitly measureable as they were descriptions of desired achievements. The CNSC typically requires environmental effects to be predicted with a 10,000 year time frame. While it was recognized that the Lorado tailings cover will remain in place for a very long period, its performance could not be credibly and practically measured in geological time lines. A 100 year design criterion was adopted that was consistent with industry best practices and provided a realistic time frame to monitor the cover performance and effectiveness.

The original intention of the RRP was to prevent formation of efflorescent salts to reduce the acidity loading into Nero Lake and potential of inhalation (Golder 2013). Nonetheless, further analysis (SRK 2013) demonstrated that the acidity load from the salts will not significantly affect the water quality in Nero Lake. With this information, the cover criteria was to practically minimize formation of efflorescent salts on the cover to no more than 10% of the original tailings surface at any time to satisfy reduction in inhalations. Details of the water quality prediction are discussed in the following section.

As a spot reading, gamma radiation and radon gas exposure limit measured 1 m above cover must be no greater than 2.5 μSv/hr (micro Sievert per hour) above background and no higher than 1.0 μSv/hr above background as an average measure over 1 ha. A zero-reading is not a practical target as the Uranium City areas natural environment background is above 0 μSv/hr average.

The effect of differential settlement on the cover could have an effect on the cover surface drainage pattern and a secondary effect on infiltration regime that may influence vegetation diversity. None of these effects are detrimental to the stated objectives of the cover, resulting in the tolerance of up to 0.3 m in the upland cover and 0.6 m in the beach cover.

The RRP stipulates that any tailings in water less than 3 m deep within Nero Lake should be covered. The rational is that in case of a land bridge failure where the water level between Nero and Beaverlodge Lake equalized, the tailings would not be exposed. The risk of failure could be mitigated with a regular geotechnical inspection. It was recommended that the in-water tailings cover be designed based on best practices for tailings water cover design theory that stipulates a water cover must be thick enough to prevent re-suspension of tailings resulting from wave action and ice scour. The resulting evaluation concluded conservatively that only tailings in water depth less than 1.3 m, as measured from the normal water level (NWL), needed to be covered.

Water Quality Prediction

Previous studies (Golder 2013) confirm that runoff from the tailings area dissolves the efflorescent salt precipitate containing metals, acidity, and total dissolved solids and dissolves them into Nero Lake. This surface flow contributes towards 80% of the mass loading to Nero Lake, and there is enough acid inventory remaining in the tailings deposit to contribute acidity and metal loadings to Nero Lake for the long-term. An additional 16% of the mass loading is from surface runoff carrying material from the upper, oxidized tailings and 4% is from seepage through the deeper unoxidized tailings and from diffusion from lakebed tailings. The oxidized tailings contain very high acidity and metals; however, these metals are not as concentrated as in the efflorescent salts. The unoxidized tailings are potentially acid-generating, and have low pH and elevated contaminant concentrations.
Further analysis (SRK 2013) demonstrated that once Nero Lake is neutralized through in-situ treatment, the acidity load contributed by the uncovered tailings would not be sufficient to result in re-acidification, due to excess natural alkaline inflows to the lake. A sensitivity analysis confirmed construction of a tailings cover that results in a load reduction of 50 to 95% would have no discernible effect on the long-term Nero Lake water quality. As a result, there was no real requirement to specify either an infiltration or load reduction criteria for the cover system to ensure suitable water quality in Nero Lake over the long-term.

Preventing the unoxidized tailings from oxidizing by limiting oxygen ingress will not make a discernible difference in the acidity of the pore water and therefore constructing an oxygen limiting cover was not warranted.

As demonstrated (SRK 2013), reduction of surficial efflorescent salts formed through evapo-concentration will not result in measurable water quality benefits to Nero Lake; however, it will provide environmental benefits including, but not limited to, improving visual impact, providing an improved substrate for re-establishment of vegetation, and limiting exposure to terrestrial animals, birds, and aquatic life.

Capillary Break Cover Design

The initial screening of the soil cover options (Golder 2013) established that store-and-release or infiltration-reducing type covers could be suitable for the site based on the climate analysis. However, the objective of reducing pore water from the tailings that wick to the surface and precipitate contaminant salts was best mitigated by a capillary break cover.

A capillary break cover consists of a relatively coarse material adjacent to a layer of relatively fine grained material. The contrast in hydraulic properties between the coarse and the fine layers ensures that, under average conditions, water is retained in the fine layer, thus reducing the transfer of water between layers. In the case of Lorado, the cover objective was to prevent tailings pore water from rising through the cover. The coarse layer was in this case placed directly over the fine-grained tailings. In effect, this is an inverted capillary break cover where suction developing in the coarse layer above should remain low enough to prevent water from being wicked up from the tailings. A top layer of well-graded till was also included in the design to provide a suitable substrate for vegetation.

Numerical modelling was carried out by Golder to confirm the suitability of the capillary break. The hydraulic properties of the sand used in the model were, however, not based on laboratory testing of samples collected from the actual available borrow sources at Lorado, but through empirical estimations using particle size distribution data (Golder 2013).

In 2013, SRK carried out a comprehensive borrow investigation (SRK 2014) to determine the quantity and location of the cover materials available on site. This included laboratory characterization of the physical and hydraulic properties of the soils to be used to construct the capillary break cover.

Although the gradation of the sand used by Golder (2013) was similar to the gradation obtained by SRK (2014), the estimated soil water characteristic curve (SWCC) was more favorable in terms of capillary break effect than the SWCC measured in the laboratory. Following SRK's review of the obtained data, it became clear that the soils available on site may not be ideally suited for a capillary break type cover.

As a result, a more comprehensive performance prediction of the proposed cover concept was then carried out by SRK (2014), with focus on estimating the capillary rise afforded by the sands. The idea was that although the capillary break effect was marginal, the water retaining properties of the available sands were such that capillary rise to the surface (hence wicking of tailings pore water) could be minimized by a sufficiently thick sand layer.
Two distinct methods (i.e. analytical and numerical) were used to estimate the magnitude of capillary rise for the sand materials considered for covers. The methods are vastly different in terms of the concepts used to estimate capillary rise, but they yielded comparable results.

Analytical Approach
The analytical method used relied on a theoretical formulation using the surface tension of pure water and clean glass tubes as an analogue for interconnected pores in a soil column (Holtz and Kovacs, 1981). The advantage of this method is its flexibility in studying changes in capillary rise by adjusting the effective particle size of the material corresponding to the D_{10} particle fraction.

The theoretical capillary rise of the coarse and medium sand were in the range of 0.65 to 0.75 m, whereas the fine sand exhibited a much higher capillary rise at 2.5 m. Increasing the effective grain size of the fine sand by applying a fines cutoff (D_{10}) at 0.13 mm reduced the theoretical capillary rise to 1.15 m. This method is excessively conservative because it does not take into account the complexity of the internal soil structure.

Numerical Approach
The numerical model makes use of the Richards equation to predict the capillary rise in a state of equilibrium with the suction forces present in the soil. The mechanics of the SVFlux model relied on a specified SWCC fitted to a smooth curve using one of the built-in fitting methods. This method utilized

hydraulic properties obtained from laboratory testing; thus it provided representative actual behavior of the soils in the area.

The capillary rise predicted by the numerical model was approximately 0.5, 0.85, and 1.4 m for the coarse, medium, and fine sand, respectively.

Correlation of Analytical and Numerical Results

The theoretical capillary rise values using the analytical methods were compared with those predicted by the numerical model (Table **1**).

The reasonably good correlation (R^2 = 0.91) between the estimated and predicted values allowed the approximation of the capillary rise predicted by the numerical model (considered more accurate than the analytical method) as a response to changes in the D_{10} particle fraction applied in the analytical model. Based on this correlation, changing the fines cutoff in the sand materials to obtain a D_{10} of 0.13 mm would result in a theoretical capillary rise of approximately 1.15 m in the analytical model that would in turn correspond to a capillary rise of about 1 m as predicted by the numerical model.

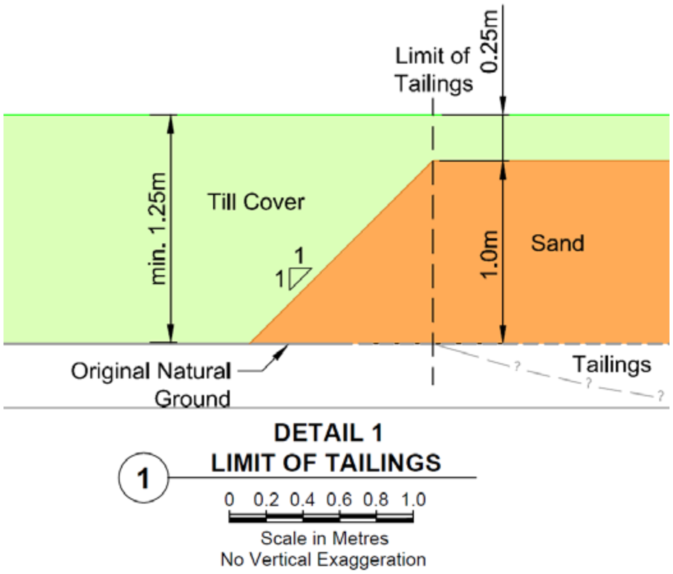
Table 1 Comparison of Capillary Rise Values

Material	**Estimated Value (Holtz and Kovacs 1981)**	**Predicted Value (SV Flux)**
Fine Sand (SRK, WP-64)[1]	2.50 m	1.40 m
Medium Sand (SRK, BB-88)[1]	0.65 m	0.85 m
Coarse Sand (SRK, B1-15)[1]	0.75 m	0.50 m
Modified Fine Sand (D_{10} = 0.13 mm)	1.15 m	1.00[1] m

1. Predicted based on the correlation graph

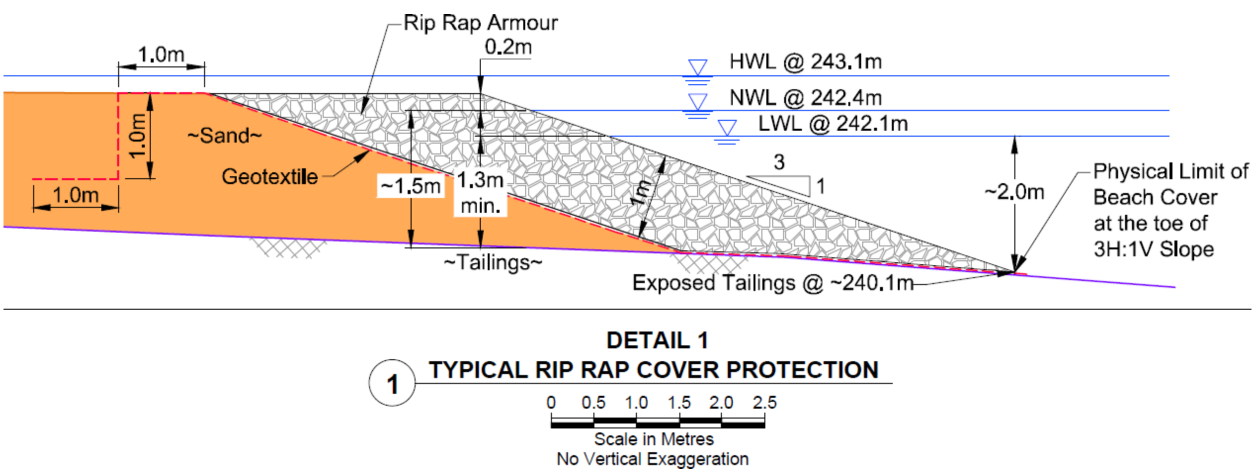
Final Cover Design

As an outcome of the model results, the initial upland cover design proposed by Golder (2013) consisting of 0.5 m of sand (capillary break layer) overlain by 0.5 m till (infiltration reduction layer) was refined to include a 1 m thick sand layer (capillary break) overlain by 0.25 m of till (vegetation substrate). A capillary break sand layer is only required over the upland dry tailings. The extended cover that was outside of tailings did not require the sand for capillary break and consisted of 1.25 m of till. Figure 2 shows the typical configuration of the Lorado tailings upland cover. The beach covers for the submerged tailings under NWL would be saturated at all times. The beach covers would be built to 0.2 m above NWL from sand for its constructability in water. Figure **3** shows the typical configuration of the Lorado tailings beach cover.



Source: SRK, 2014

Figure 2: Typical Section of Lorado Tailings Upland Cover



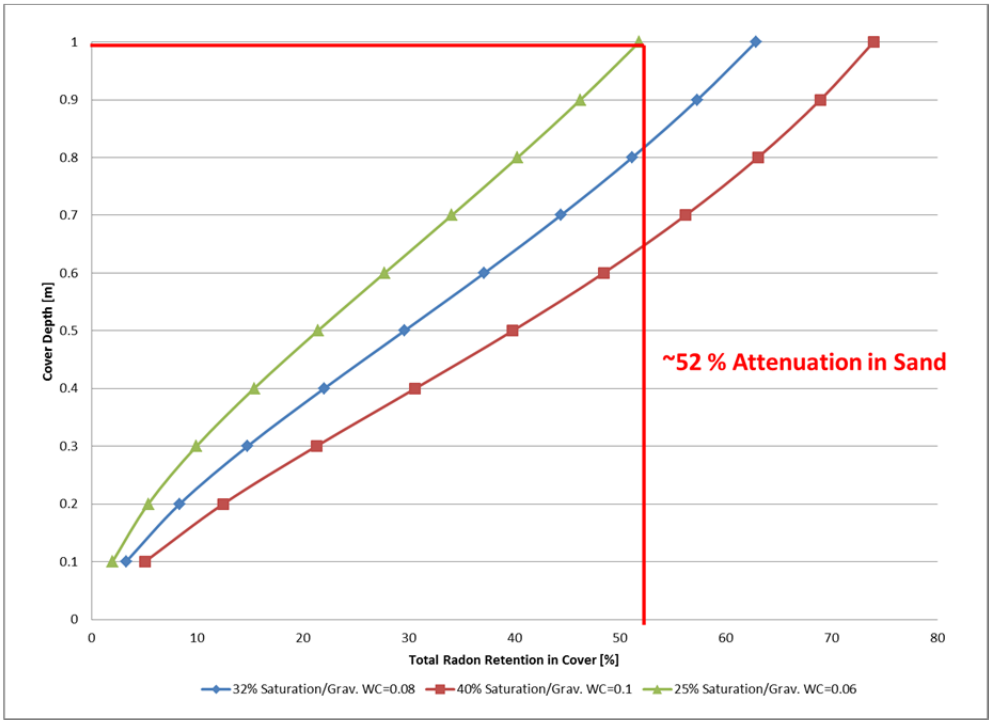
Source: SRK, 2014

Figure 3 Typical Section of Lorado Beach Cover

Radon and Radiation Attenuation Design

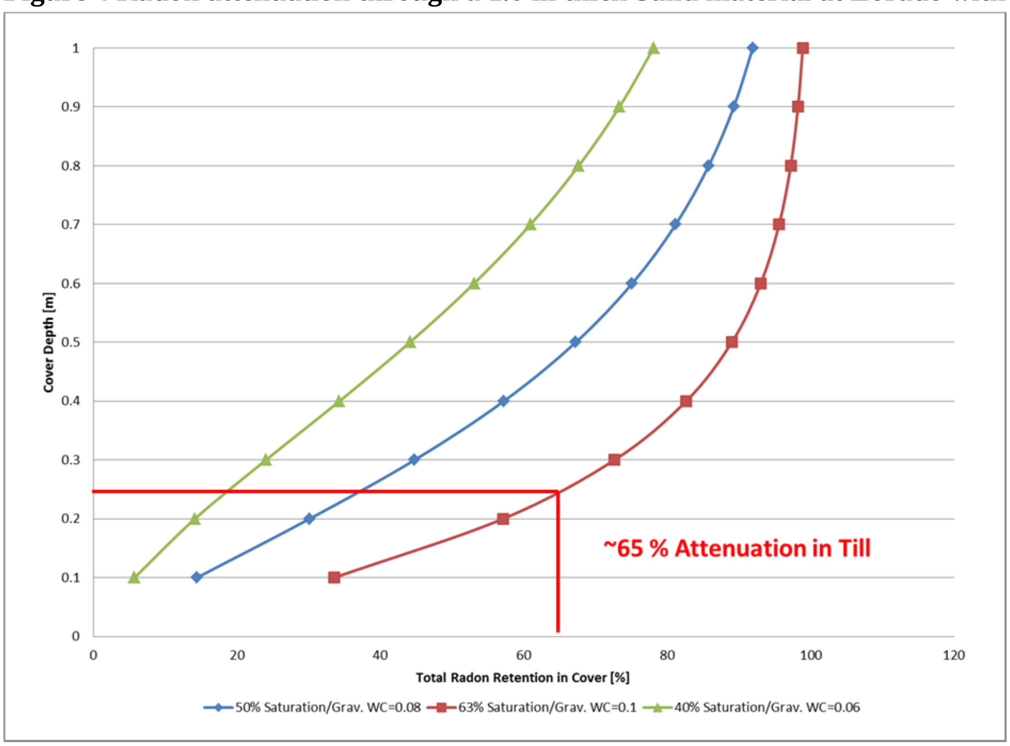
The radon attenuation calculation was completed using the empirical formulation developed in the RAECOM model (Rogers and Nielson 1984). The model utilized effective diffusion coefficient and physical properties of the cover material (e.g., porosity and gravimetric water content (GWC)). Sensitivity analyses were completed to determine the effect on radon attenuation due to variations in physical material properties.

The results of the attenuation model are illustrated in Figure **4** and Figure **5**. At assumed natural moisture contents, a 0.25 m thick till cover attenuated 65% radon and 1.0 m of sand attenuated 52%. Using these material attenuations and the high average radiation source of 14.4 µSv/hr recorded on site as conservative approach, the sand attenuated 52% (7.28 µSv/hr) of the initial source and the till then attenuated 65% of the remaining radon (4.4 µSv/hr) to release 2.35 µSv/hr of radon immediately above the cover. The combined 1.25 m thick soil cover attenuated 83% of source radon. This meets the criterion of less than 2.5 µSv/hr of single spot reading as described in Section 0.



Source: SRK, 2014

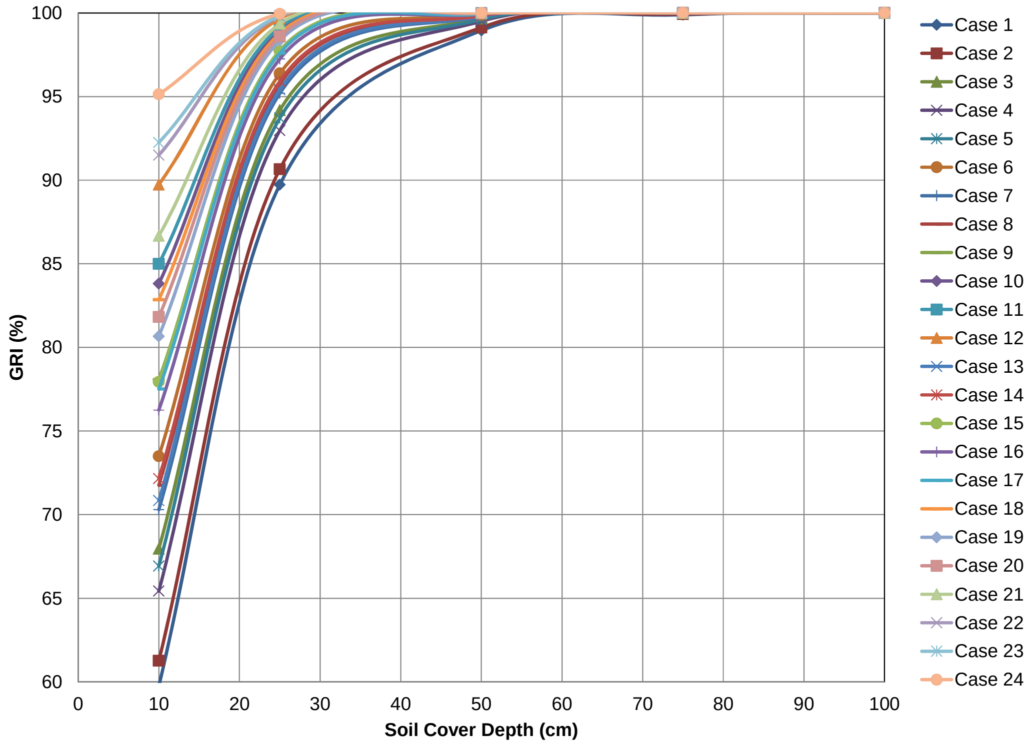
Figure 4 Radon attenuation through a 1.0 m thick Sand material at Lorado with a porosity of 0.4.



Source: SRK, 2014

Figure 5 Radon attenuation through a 0.25 m thick Till material at Lorado with a porosity of 0.3.

The gamma ray exposure at the site was a result of radium-226 decay. Gamma ray attenuation was estimated using the empirical formulation based on an exponential decay function presented by Inyang et al. (2005). Sensitivity analysis was done on various material properties to determine their effect on gamma ray attenuation as shown in Figure **6**. The results indicated that while gamma ray attenuation was sensitive to material properties, the attenuation was raised very rapidly with an increase in cover thickness; hence, the thickness of the cover is the governing factor. At 0.6 m cover thickness, 100% attenuation was achieved irrespective of cover properties.



Source: SRK, 2014

Figure 6 Gamma ray reduction as a function of soil material thickness.

Water Cover Design

The current accepted best practice in water cover design is documented in MEND (1998). According to this guideline, there are five processes that affect waterbed stability: seiche, seasonal lake turnover, currents, wave action, ice entrainment and scouring. For small tailings impoundments with less than 500 ha water surface area and water depth less than 10 m, only wave action, and ice entrainment and scouring should be considered. Nero Lake falls into the small impoundment category, with surface area of approximately 160 ha and depth less than 9 m.

For tailings re-suspension due to wave action, the Lorado design compared the MEND (1998) and Lawrence et al. (1991) methods using the US Army Corps of Engineer, Coastal Engineering Research Center (CERC 1984) theory to substitute the minimum water cover requirements. The wave action evaluation methods took into account the wind speed and direction, water surface fetch length, water

depth, median particle size of sediment (MEND 1998), and threshold velocity (Lawrence et al. 1991). The MEND (1998) and Lawrence et al. (1991) methods are valid in deep water waves (ratio of water depth over wave length is less than 0.5) that do not apply to shallow Nero Lake conditions. To overcome this, the design substituted the appropriate shallow wave theory (CERC 1984) results into MEND (1998) and Lawrence et al. (1991) formulations to suit shallow conditions at Nero Lake. The water cover thicknesses calculated using modified shallow wave action theory are shown in
Table **2**. Since the two empirical methods used were considered equally valid, the average of water thickness was used and resulted in a design cover thickness of 1.3 m.

Table 2 Water cover thickness comparison using deep and shallow water wave theory.

Method	**Water cover thickness using shallow water wave results**
Lawrence et al. (1991)	1.5 m
MEND (1998)	1.1 m
Final Lorado Water Cover Design	1.3 m

Ice entrainment occurs when the ice layer is sufficiently thick that it freezes to the lake bed, also known as grounded ice. As the ice thaws, sediment entrained in the ice is released into the water column. Ice scouring occurs where the underwater velocities increase in the restricted flow area due to ice grounding. MEND (1998) recommends the minimum water cover should be the ice thickness plus a 1.1 factor of safety. Nero Lake has no record of ice thickness so an ice growth model (US Army Corps of Engineers 2005) was used to predict the ice thickness. The model used daily average air temperature, a calibrated empirical factor (α) using Environment Canada data from four nearby sites (Fort Reliance, Fort Chipewyan, Cree Lake and Brochet). Using the climatic data set and statistical analysis, a maximum ice thickness associated with the 100 year recurrence interval was determined to be 1.06 m and with a 1.1 factor of safety equated to a thickness of 1.17 m.

Since the wave action requirement was greater than the ice entrainment, the final design adopted the 1.3 m water cover.

Other Cover Design Components

As far as practical, ponding on the dry tailings surface was avoided since it would result in saturation of the capillary break layer rendering it ineffective. The tailings surface would be re-graded and landscaped to allow overland sheet to flow toward two drainage channels and then to flow into Nero Lake. The channels were integrated into the tailings surface with appropriate thicknesses of cover materials to be placed on top. The final channel configuration had a 6 m wide base and side slopes of 3H:1V (horizontal to vertical). The overall drainage grade was variable between 0.5 to 1%. The channels were designed to convey a 24-hour, 1-in-200 year precipitation event at 1.57 m^3/s.

Organic debris islands were designed as landform features to create a micro-climate to promote vegetation growth over the cover. The islands provide windbreaks, retain moisture from precipitation, trap seeds, and offer nutrients through decomposition. The organic debris is expected to decompose over time so they are not considered permanent features. Natural, local vegetation is expected to grow in place over the long-term. Organic debris (e.g., tree slashing, tree stumps, and topsoil from borrow developments) would be placed in strategic configurations. A total of twenty-five organic debris islands configured approximately 1.5 m high and 5 m by 5 m at the base would be placed on the completed cover.

CONCLUSION

A post-construction geotechnical monitoring program will be implemented to confirm that the Lorado tailings cover is performing in accordance to design criteria and meeting remediation objective. These inspections are expected to be done by qualified personnel annually for three years following construction completion and every five years thereafter. A set of inspection parameters are laid out to measure cover physical properties and water quality.

It is acknowledged that while the tailings cover is a permanent structure and has a 100 year design life, there will be some degree of damage over time that will have to be address through planned maintenance activities. While regularly scheduled maintenance is not likely required, it is unrealistic to expect an earthen structure that supports natural vegetation to be maintenance free for its design duration. Over time, nature will reclaim the land as the design intended. Until such time, appropriate maintenance, repair work, and monitoring schedule will be prepared and subsequently executed by the appropriate qualified professionals. Areas requiring maintenance and repair will be identified from the monitoring program.

Every project has its unique criteria that influence the project's acceptable risks, allowable ongoing maintenance, and overall remediation costs. While many remediation ideologies desire an end product with low remediation project costs, little or no risks and little or no post-construction management, it is unrealistic to expect all projects to meet all such criteria. Furthermore, with a post-construction monitoring program and on-demand maintenance requirements, the design criteria becomes more important to be realistically defined and kept as record for the life of the structure. A set of realistic, measureable design criteria can turn often descriptive remediation objectives into specific achievable targets by defining the functional limits of the structure, identifying the monitoring parameters needed to measure ongoing performance, and establishing clear maintenance repair targets.

REFERENCE

Inyang H., Wachsmuth P. et al, 2005, Simplified Design of Georadiological Barriers for Contaminated Sites. Practice Periodical of Hazardous, Toxic, and Radioactive Waste Management. 9, pp 253-262

Golder Associates, 2013, Environmental Impact Statement – Risk Reduction Plan for the Inactive Lorado Uranium Tailings Site, prepared for the Saskatchewan Research Council.

Holtz, R.D., and Kovacs, W.D., 1981, An Introduction to Geotechnical Engineering, Prentice-Hall Civil Engineering and Engineering Mechanics Series, Inglewood Cliffs, NJ, 733 p.

Rogers V., and Nielson K, 1984, Radon Attenuation Handbook for Uranium Mill Tailings Cover Design. NUREG/CR-3533, U.S. Nuclear Regulatory Commission, Washington, DC., 75 p + Appendices.

SRK Consulting (Canada) Inc., 2013, Nero Lake Acidity and Alkalinity Load Balance – Draft. SRK Project Number 4CS008.003, 21p + Appendices

SRK Consulting (Canada) Inc, 2014, Detailed Cover Design Report for the Lorado Mill Site Tailings and Peripheral Areas. SRK Project Number 4CS008.003, 33p + Appendices.

Komar, P.D. and Miller, M.C. 1975. On the comparison between the threshold of sediment motion under waves and unidirectional currents with a discussion of the practical evaluation of the threshold. Journal of Sedimentary Petrology, Vol.45, pp. 362-367.

Lawrence, G.A., et al., 1991. Wind-Wave-Induced Suspension of Mine Tailings in Disposal Ponds – A Case Study. Canadian Journal of Civil Engineering, Vol. 18, pp.1047-1053.

MEND 1998. Design guide for the subaqueous disposal of reactive tailings in constructed impoundments. Project 2.11.9, 87 p + Appendices.

U.S. Army Corps of Engineers. 2005. Review of Ice Processes and Properties. In: U.A. Engineers, Engineering and Design: Ice Engineering, pp. 2-1, 2-17.

U.S. Army Corps of Engineer, Coastal Engineering Research Center. 1984. Shore Protection Manual, Volume 1. Department of the Army Waterways Experiment Station, Corps of Engineers, Vicksburg, Mississippi. 652 p.

RECLAIMED CLOSURE LANDSCAPES: THE IMPORTANCE AND BENEFITS OF OPERATIONS MAINTENANCE AND MONITORING

J.M. Fair, MSc., PAg.
J. Pollard, MSc., Geol.I.T. (AB)
G.T. McKenna, PhD., Peng., PGeol.

BGC Engineering Inc. Edmonton, AB

ABSTRACT

The process of landform design of closure landscapes and watersheds can be divided into a seven step process; conceptual closure planning design, individual landform design, individual landform construction, final closure plan design, final closure plan construction, monitoring and operation, and closure.

As part of recent work on landform design and construction for open pit mines, several constructed projects have progressed into the monitoring and operations phase. This phase is important for assessing the performance of the landform to compare to the proposed landform design goals and objectives, for preparation for final closure, and potentially for closure certification. This paper presents a call to action for undertaking this phase of the landform design process with skill, creativity, and effective organization. We provide some useful tools and some lessons learned which highlight where this aspect of landform design is being undertaken successfully and where it could be improved upon.

There are three separate components to the monitoring and operation step of landform design; operation, maintenance, and monitoring. The scope of each of these will depend on the individual landform design and what purpose it is designed for. For example, for an instrumented research watershed the level of effort will be high with respect to managing water levels, maintaining access, fixing outlets, and undertaking research monitoring to assess performance. At the other end of the scale, for a small opportunistic wetland or a small waste dump, operation and maintenance may be negligible and monitoring may be limited to an annual inspection.

As part of this paper we provide a table of important operation, maintenance, and monitoring activities as well as roles and schedules for both an upland and a lowland setting which can be used as a tool for this phase of design. We also discuss the intricacies involved in some typical monitoring and operating activities such as outlet operation, water balances, post construction landscape fixes, and methods to simplify the entire process.

KEY WORDS: Closure, Operation, monitoring, maintenance, OMM Manual, certification, early planning

INTRODUCTION

Design of mine closure landscapes and their landforms and watersheds involves numerous steps: conceptual closure planning design, individual landform design, individual landform construction, final closure plan design, final closure plan construction, operation, maintenance and monitoring, and final closure (McKenna and Cullen, 2008; CEMA in prep). As part of recent work on landform design and construction for open pit mines, several projects the authors have worked on have progressed into the operations, maintenance, and monitoring (OMM) phase. This phase is important for assessing the performance of the landform to compare to the original landform design goals and objectives. It is also important for preparation for final closure, and potentially for closure certification / bond release, yet this phase is often overlooked. There are opportunities to enhance this aspect of mine reclamation. This paper describes the current state of practice of planning for this OMM phase, proposes a new way forward, provides some example of success, and lists a few lessons learned thus far, mainly from recent experience in oil sands reclamation and from examples elsewhere from the past two decades.

CURRENT STATE OF PRACTICE

Most mines begin with an overarching goal that mine reclamation will allow a “walk away” / self-sustaining landscape at closure or soon after (McKenna 2002) and will be entitled to release of financial assurance obligations. Much of the early reclamation may involve re-sloping, cover soiling, and planting of upland terrestrial vegetation and any required post reclamation maintenance and monitoring is done in an informal / ad hoc way. Issues related to more challenging reclamation (e.g. stabilizing dykes for abandonment, dealing with acid rock drainage, creation of wetlands) often arise much closer to closure. Dealing with these issues highlights the need for higher levels of maintenance and monitoring during the operational phase.

In reality, most mines find that they will need to monitor and maintain much of the reclaimed landscape for a considerable period after reclamation, through closure, and in many cases, in perpetuity (e.g. Goldcorp 2006). A typical goal is to conduct the minimum levels of OMM to protect the reclamation investment and develop enough information for successful relinquishment (Cowen et al 2013). Thus the overall challenge is to design and carry out an optimal level of OMM activities, in an organized fashion, with an eye to steering each landform (and the entire reclaimed mine landscape) towards meeting regulations, achieving declared goals, and ultimately leading to timely relinquishment to a willing custodian (usually the Crown).

Reclamation specialists do monitor their reclaimed land, keeping an informal eye on the growth of vegetation, ingress of volunteer species, land use by wildlife, evidence of erosion or deposition, zones of seepage, etc. But documentation is sparse, budgets for fix-ups difficult to justify, and monitoring is generally done in the absence of a plan or schedule. Lacking a repository for observations or data collection, knowledge of the land is usually lost when the specialist moves to another job or retires. Given that the time between reclamation and relinquishment may span decades, such losses are common.

Unfortunately, the current state of practice is not leading to timely relinquishment of mining properties – the mines are usually either abandoned, often partially reclaimed, or remain under long-term control of the mining company or subsequent owners (McKenna 2002, Cowen et al 2013). There are numerous reasons behind this situation, many of them routed in uncertainty of how well the reclaimed landscapes will perform into the future. Making the case for good long-term performance usually requires reliable performance data over years or decades. Typically such information is only available if a timely, formal monitoring program is designed and implemented at the mine.

In an ideal world, the landform design and OMM plan would be developed in tandem, and monitoring would start the start of mine operations. In practice planning for the OMM stage is generally overlooked until long after reclamation is complete. The OMM program and a manual are an afterthought, and the program can suffer from lack of budget, motivation, sometimes from lack of access to the site.

Thus, there is a major opportunity to develop and implement the OMM plan in a timely manner to help prove to the company, stakeholders, and regulators that the investment in reclamation is being protected, the reclaimed landscape is performing as intended, that reclamation technologies are working, and that the land is on a trajectory to successful relinquishment.

A NEW WAY FORWARD

As an initial attempt to solve this problem, we propose that writing the OMM manual in a conceptual form during the design stage could help with being more successful in the later OMM stage and ultimately achieving the landform goals, with the documentation to demonstrate it. Planning for OMM during the design stage will encourage the project team to think about how to fit in the final steps of landform design into the overall project. Developing the OMM plan concurrently with design will also boost the importance of these vital steps in the minds of the designers and operators.

There are some examples of mines that have adopted a take on this approach, and are having success with it. These mines are ahead on the road to relinquishment. Adopting some of their ideas and processes could help bring the industry forward to a place where the OMM steps are not left to the end of construction, could save time and money, and could streamline the relinquishment process.

Specifically, it is proposed that mines develop an OMM Manual as part of the landform design, prior to construction of each landform. The manual would be developed in consultation with regulators and stakeholders, and be updated periodically as new information becomes available. The monitoring is focused on helping to ensure that the reclaimed landscape is on a trajectory towards relinquishment. Care is taken to make the OMM program sustainable within the corporate environment: that it is useful, can be carried out efficiently, has corporate support, and is supported by a good data management program. Ideally the mining landforms and landscapes are designed to be easy to monitor and maintain, and that there is a process to take learnings from the OMM program and use them to adapt future landform design and reclamation activities on the site as part of an overall adaptive management program (e.g. Holling 1978; CEMA 2012).

DEVELOPING DESIGN AND OMM PLANS IN TANDEM

Ideally, landform designs are done prior to construction, and updated as needed through the construction, regrading, and reclamation process. Often, formal design waits until the landform is largely constructed. Either way, the design includes documentation of design goals and usually includes a description of potential failure modes as part of an engineering risk assessment. As part of engineering risk management, the monitoring plan is developed. The OMM provides the information to assess the landscape performance against these goals and indicates when and where any intervention is needed.

The level of required monitoring may vary considerably based on the goals. For example, a reclaimed landform built with the goal of obtaining research information (e.g. an instrumented watershed for reclamation research) will require significant data collection over a number of years and a high level of monitoring (e.g. Daly et al 2010). Similarly, wetland establishment on reclaimed land may initially require daily or weekly adjustment of water levels or ongoing short-term control of invasive plants (CEMA in preparation). At the other end of the scale, an upland area may only require minimal OMM – perhaps limited to a brief annual visual inspection and a final vegetation surveys.

Figure 1. Developing design and OMM plans in tandem

EXAMPLE OMM PLANNING FOR WETLANDS AND UPLANDS

There are a few notable examples of companies that are benefitting from OMM planning early in the landform design phase. Teams working on the constructed fens at Syncrude Canada and Suncor Energy oil sands operations both had OMM plans in mind from the beginning (Pollard et al. 2012). The main goal of both fens was to gain knowledge about construction and performance of reclaimed wetlands, so monitoring began early (even before reclamation), and the data were analyzed and used to make operational decisions at a later date. Water flow and chemistry data, weather data, records of construction and materials, photos, and satellite imagery collected during construction were invaluable in understanding the functioning of the landform and informing the future construction of fens. These fens had the goal of operating as research landforms and were therefore operated, maintained, and monitored at a much higher level of effort than a run-of-the mill reclamation landform could be. However, a lot of knowledge can be taken from these two fens and applied as needed to design, construction and operation of commercial scale reclamation landforms in the future (Pollard et al. 2012). Ultimately data from the

OMM will be used in support of an application for reclamation certification to the Alberta Government, a key step in relinquishment of reclaimed mine lands.

As part of our work in post closure and studying mines that have started to adopt parts of the early OMM development, we, with guidance from CEMA's 2005 Landscape Design Checklist, have identified the following common areas to consider for operations and monitoring of a post closure landscape:

- Access and security
- Geotechnical stability
- Groundwater / seepage
- Surface water control – quality and quantity
- Erosion
- Vegetation
- Wildlife
- Dust / air quality
- Safety – public and on-site.

A worked example of an OMM table which can be produced during the closure design phase for surface water quantity and quality in a lowland setting and a vegetative cover in an upland setting are shown in Tables 1 and 2. We have focused on the key activities for that area, and who is responsible for what, where, and when. Developing this table at the closure design stage then sets up for the production of an OMM manual and identifies key areas where the design may need to accommodate some operational or monitoring activities.

Table 1 shows an example of a marsh that is in its first season of operation since construction was completed. Such a recently constructed landform will require the maximum amount of monitoring, maintenance, and operation effort that a landform would require, as the early stages are when most adjustments and optimizations must be made.

Table 1. Example Operation, Maintenance, and Monitoring Summary Table for a Recently Constructed Marsh

	Activity	**Action**	**Personnel Responsible**	**Schedule (Maximum Required)**
MONITORING	Water levels	Datalogger	Landscape Operator or other assigned personnel	Collect and analyze bi-weekly, May to October
	Water quality	Grab sample	Landscape Operator or other assigned personnel	Collected and analyzed monthly, May to October
	Specific research monitoring	As determined by researchers	University Researchers and students	As determined by researchers

MAINTENANCE	Repairing outlets	Fix any beaver damage/ damage from high flow events etc.	Area owner or other assigned personnel	As needed
	Maintain access	Repair boardwalk / walkway	Area owner or other assigned personnel	As needed
OPERATION	Maintain Water Balance	Adjust stop logs and pumping activity	Landscape Operator or other assigned personnel	Weekly, May to October, or as needed.

Table 2 shows an example OMM plan for an upland forest that was planted 8 years prior. The forest requires a minimum of monitoring, maintenance, and operation. In the forest's first few years, it received much more attention in order to put it on the correct trajectory towards becoming an upland mixed-wood forest, and needs less attention now to keep it on that trajectory.

Table 2. Example Operation, Maintenance, and Monitoring Summary Table for an Upland Forest

	Activity	**Action**	**Personnel Responsible**	**Schedule (Minimum Required)**
MONITORING	Check coverage percentage	Field survey	Landscape Operator or other assigned personnel	Collected and analyzed once yearly during summer season
	List of vegetative species present	Field Survey	Landscape Operator or other assigned personnel	Collected and analyzed once yearly during summer season
	Soil Quality	Dig soil quality pit	Landscape Operator or other assigned personnel	Collected and analyzed once yearly during summer season
	Erosion	Field Survey	Landscape Operator or other assigned personnel	Once yearly and after severe rain storms
MAINTENANCE	Weed control	Pulling weeds, applying control chemicals, etc.	Area owner or other assigned personnel	Once yearly, if needed or more often if required by specific weed species

	Protection from wildlife	Add fences to keep wildlife out, wrap base of seedlings, etc.	Area owner or other assigned personnel	As needed
OPERATION	Maintain vegetative cover	Seed and plant	Landscape Operator or other assigned personnel	Re-seed or plant if needed as shown by monitoring
	Prevent erosion	Add erosion mats or other means to prevent erosion	Landscape Operator or other assigned personnel	As needed according to monitoring

LESSONS LEARNED DURING THE OMM PHASE

To conclude, we present some key lessons learned from various mining projects during the OMM phase which may benefit future projects:

- An OMM Manual for a post closure landscape is similar to that of any operating facility such as a tailings facility, which are commonplace in the industry. (Crossely et al. 2011).
- Methods to heighten the chance of the landform's success in operations include:
    - Operations and maintenance team should work together with the construction team, and have some involvement at the design and construction stage to see that achievable operation goals are clearly set (McKenna 2002).
    - Including in the design contracts and budget that operation and maintenance visits will required.
    - Assigning a dedicated monitoring team with clear goals, procedures and data recording responsibilities.
    - Keeping naming conventions consistent throughout the entire design, construction, and operation process. When giving names to different areas or sampling spots within the project site (for example, BH19 or South Outlet), set the names in the design and do not change or duplicate them throughout the process.
- Once construction has been completed, our experience is that fixes to the site are more difficult than anticipated, even for a relatively simple issue.
    - For example, if more material is needed after construction is completed, site and material access can become major barriers.
    - If larger fixes are required, such as retro-fitting pipes or pumps or re-contouring, there is a risk that some of the reclamation that has been completed will have to be destroyed in order for equipment to re-enter the site and carry out the fixes.
    - It is more efficient time-wise and budget-wise to ensure that if these types of fixes may be required, access is maintained and it is planned for during the construction process. Access can be designed to be easily reclaimed in the future when no longer needed.
- A date to cease monitoring must be chosen based on the type of landform and its goals and should be included within the design and conceptual OMM process. Monitoring typically will decrease with time and cease at relinquishment unless the new custodian wishes to do long-term monitoring in support of a regional adaptive management program.

- Data from monitoring must be mindfully managed to ensure that once it is collected, it is being analyzed and used as the basis for operation and maintenance decisions. The data should be analyzed on a schedule so the real conclusion about the landform can be developed and management decisions made. For example, if data analysis for a landscape shows poor water quality, but no action has been carried out to assess the problems and mitigate them, it is likely that this condition will continue to worsen.
- Data and data management are important aspects of operations, monitoring, and maintenance, and some specific data lessons learned include:
    - Data collection and analysis should be undertaken early in the design phase to make adaptive management decisions.
    - Data should be recorded in an organized, project-specific, easy to navigate database.
    - The database should be backed up in another location.
    - Specific people or groups should be assigned specific tasks and methods documented for consistency.

NEW INNOVATIONS

There are numerous tools and emerging technologies available to enhance monitoring capabilities. Along with the current state of practice monitoring, such as water quality and level monitoring, soil surveys, vegetation surveys, wildlife surveys, etc., some recent innovative examples that have proved useful on constructed closure landforms include:

- Static cameras
- Video cameras with remote feed
- Multispectral satellite and aircraft mounted imagery
- LiDAR survey and radar satellite imagery for settlement
- Instruments and gauges – remote feed, manual visits.

These innovations may allow less expensive, more expansive data collection that is easy to organize. Development of methods of remote monitoring of reclaimed mine sites lags behind some other uses of these technologies.

Other bright lights in the industry where new innovations are being used for OMM are:

- Dominion Diamond Corporation/Rio Tinto Grizzly bear monitoring in the NWT, where bear hair samples are collected and submitted for DNA analysis to build up a database to follow bear population over time (MAC 2014).
- Vale’s Voisey’s Bay operation in Newfoundland and Labrador, where they track water quality data in real-time and the data is available to the public (Yukon Government 2012).

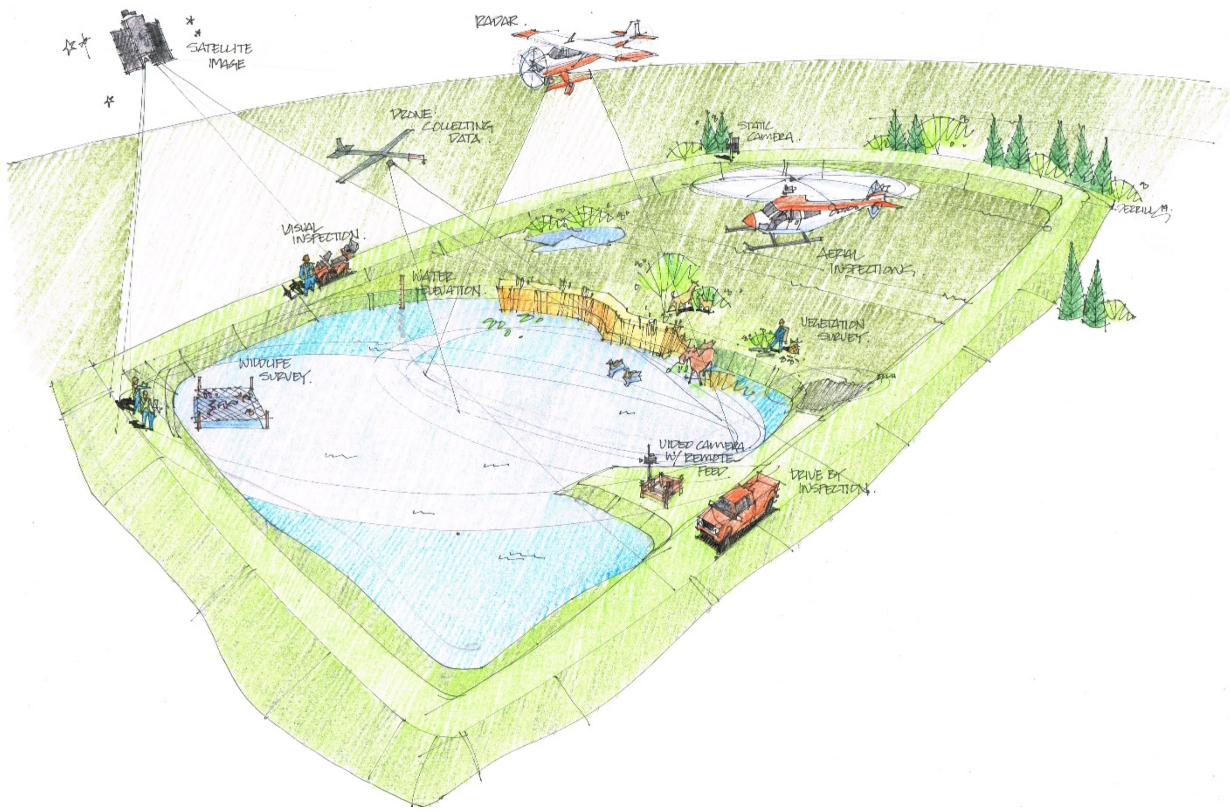


Figure 2. New Innovations in Landscape Monitoring

It is likely in the future that there will be less of a "boots on the ground" approach, and more use of remote sensing (although the importance of field visits must not be forgotten). Though the use of drones to collect information has not yet been tested, there is room for such innovation in the future. All of these new possible technologies still require good data management to be effective.

CONCLUSION

The benefits of developing an OMM manual in tandem with the construction design will reduce the risk of retro-fitting, fixes post-construction, access and accessibility issues, etc. Developing an OMM plan and doing so early in design will benefit mines on the path to closure and relinquishment by saving money, frustration, and time, and levels of effort. The operations, maintenance, and monitoring phases of a landform reclamation project carry just as much importance as the earlier steps of reclamation, and cannot be ignored until the last minute.

REFERENCES

CEMA (Cumulative Environmental Management Association) (CEMA). 2012. End Pit Lakes Guidance Document. Fort McMurray, AB. 360p.

CEMA (Cumulative Environmental Management Association). In preparation. Guidelines for Wetland Establishment on Reclaimed Oil Sands Leases (3rd Edition). Fort McMurray, AB.

CEMA RWG Landscape Design Subgroup. 2005. Landscape Design Checklist. Revised RSDS Government Regulator Version. Cumulative Environmental Management Association. Fort McMurray, AB. 8p.

Cowan WR, Mackasey WO and Robertson JGA. 2013. Case studies and decision making process for the relinquishment of closed mine sites. Consultant's report prepared for National Orphaned / Abandoned Mines Initiative (NAOMI) by Cowan Minerals Ltd. Sudbury, Ontario. July 2013. 92p.

Crossely, C., Russell, B., van Zyl, D., and McKenna, G. 2011. Developing an operation, maintenance and surveillance manual for the post-closure management of tailings facilities. Mine Closure 2011 — A.B. Fourie, M. Tibbett and A. Beersing (eds). Australian Centre for Geomechanics, Perth. 10p.

Daly CA, Price J, Rochefort L, Graf M, Russell B. 2010. Innovative wetland reclamation design case studies: the Suncor Fen and Pond 1 Marsh. Symposium on the reclamation and restoration of boreal peatland and forest ecosystems : towards a sustainable future, Edmonton, AB (Canada), 25-27 Mar 2010.

PEATNET Peatland Ecosystem Analysis and Training Network. Southern Illinois University, Carbondale, IL. p1-31.

Goldcorp Inc. 2006. Goldcorp Inc Closed Sites in Canada: 2006 Sustainability Report. Goldcorp Inc. Vancouver, BC, Canada. 16p.

Holling CS. 1978. Adaptive environmental assessment and management. International Institute for Applied Systems Analysis. John Wiley & Sons, Chichester. 377p.

MAC (The Mining Association of Canada). 2014. Towards Sustainable Mining Awards. Environmental Excellence Award – 2014 Finalists. http://mining.ca/towards-sustainable-mining/tsm-awards.

McKenna G, Cullen V. 2008. Landscape design for soft tailings deposit. Tailings and Mine Waste 08 Conference. Vail, Colorado.

McKenna GT. 2002. Landscape engineering and sustainable mine reclamation. PhD Thesis, Department of Civil and Environmental Engineering, University of Alberta, Edmonton. 660p.

Pollard J, McKenna GT, Fair J, Daly C, Wytrykush C, Clark J. 2012. Design aspects of two fen wetlands constructed for reclamation research in the Athabasca oil sands. Mine Closure 2012 Symposium. Brisbane. Australian Centre for Geomechanics p815-829.

Yukon Government. 2012. 2012 Leckie Award Recipients. http://www.emr.gov.yk.ca/mining/478.html.

DEVELOPMENT OF A GLOBAL COVER SYSTEM DESIGN TECHNICAL GUIDANCE DOCUMENT

M. O'Kane, MSc.[1]
A. Baisley, MSc.[2]

[1]Senior Geotechnical Engineer,
O'Kane Consultants Inc, Calgary, Canada.

[2]Hydrologist,
O'Kane Consultants Inc, Calgary, Canada.

ABSTRACT

The International Network for Acid Prevention (INAP) is an organization of international mining companies dedicated to reducing liabilities associated with sulphide mine materials. Liabilities associated with mine closure span the globe, occurring across a climate, hydrogeological and material spectrum. Cover systems are a practical tool in managing closure liability surrounding reactive wastes, and thus are required to perform under a gamut of site specific conditions. Previous guidance specifically addressed cover system design as it pertains to cold regions but to date, no further material exists as it specifically pertains to cover system design on a global scale. This paper describes the technical guidance approach for cover systems globally being developed by O'Kane Consultants Inc. (OKC) with the assistance of a Technical Advisory Group (TAG), funded through INAP. The focus is a conceptual filter framework philosophy used to rapidly refine cover system alternatives through a set of site specific filters, to more efficiently determine cover systems for further evaluation. Site specific filters employ the Köppen−Geiger climate classification system to incorporate seasonality into design. A process based approach is used through each refinement to better understand mechanisms and controls that can be exploited, enhanced and combined to achieve reductions in acidity generation and transport. The management of oxygen and net percolation based on each site specific filter is discussed in detail along with timing of reclamation and performance value. Evaluation of cover system performance and risk is not possible or complete without discussion of performance assessment period and design life. Existing guidance have briefly covered landform design, whereas this document highlights the importance of developing covers and landforms in concert. Furthermore, this guide advances cover system design of integrating the mine waste landform into the site wide water management strategy. The document's second focus surrounds generic case studies and real case studies to further demonstrate the utility of the proposed framework.

KEY WORDS

Cover systems, design framework, landform integration, mine reclamation.

INTRODUCTION

Performance expectations of mine waste management plans, mined waste landforms, and their associated cover systems are high. The key issue is management of chemically reactive mine waste, which is typically associated with sulfide oxidation, and the concomitant release of constituents of concern. Effluent from sulphidic mine waste during operations and closure can report as basal seepage, groundwater toe seepage or drainage from mine openings or pit walls, which can impact both surface water and groundwater. This process is referred to as metal leaching (ML) and acid rock drainage (ARD), or ML/ARD. The result is often seepage from reactive mine waste with elevated sulfate salinity and/or acidity and dissolved metals.

The guidance document presented builds upon previous work completed in the Global Acid Rock Drainage Guide (INAP, 2009) on cover system design, construction, and performance monitoring. This guidance seeks to further subdivide the applicability of each cover system strategy on a site by site basis by recognizing a continuum of cover system function exists for managing net percolation (NP) oxygen ingress (O_2) and erosion potential.

The first half of the guidance document includes a holistic framework for management of reactive/ non-reactive materials during operations and at closure. The framework for cover system design is presented at a high level, suitable for readers with a minimal technical background. The framework is presented to form a conceptual basis using a hierarchy of, climate, materials / geology, and topography, allowing the formation of a conceptual model to understand the tendencies for water movement on a site specific landscape. Ultimately, climate, materials /geology, and topography will govern how cover systems perform, and it is up to cover system designers to manipulate these components to achieve desired performance. Application of the conceptual framework presented is not entirely attainable at the current time; however, the advocates that it is the direction the industry should be headed to better resolve problems facing them. Stakeholders and mine planners are able to use the framework to evaluate cover system design proposals in a hierarchical fashion to ensure important design concepts have been considered.

Appendices will be referenced in the main body of the guidance document, and require readers to have a better understanding of water migration in soils, in particular unsaturated soils, in addition to mass transfer knowledge. Information presented in the technical appendices represents the best state of knowledge available as of the preparation of this guidance document and targeted for cover system design practitioners. The technical material is largely excluded from the body text into appendices so it can be updated as new science and state of understanding evolves pertaining to cover systems, becoming a living document.

OVERVIEW OF GENERIC COVER DESIGN APPROACH

The formation of ML/AMD is most generally controlled through limiting water flux into reactive mine wastes and gas flux, primarily oxygen to reactive mine wastes. It is important to recognize that each site will be able to manage NP, O_2 and erosion to a certain extent based on climate, hydrogeology, and materials, as well as the type of cover system employed. The two most common purposes for utilizing a cover system include management of oxygen and water to the underlying reactive materials. Additionally, cover systems must also be a component of developing a stable landform in the context of managing erosion (i.e. geomorphic stability). A continuum of cover system functionality is discussed for these three aspects (i.e. net percolation, oxygen ingress, and erosion), and is presented conceptually within Figure 1 in terms "very low", "low", "moderate" and "high" NP, O_2, and erosion rates.

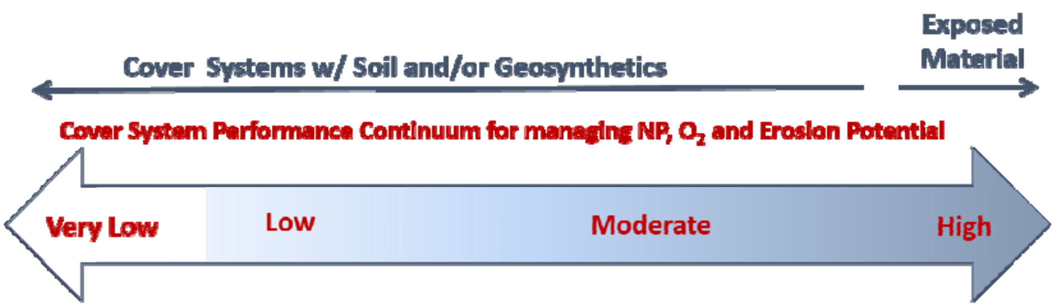


Figure 1 Qualitative cover system functionality continuum.

<u>Continuum of NP Control</u>

Generally, covers have been thought to limit NP into waste is by one of two methods:

1) **Diversion –** a layer of the cover system may be constructed from materials with a sufficiently low hydraulic conductivity so as to limit downward percolation of rainfall or snowmelt and 'release' water as surface runoff and/or interflow.
2) **Store-and-release –** infiltrating water is stored within the rooting zone of the cover system so that it can be subsequently released via evapotranspiration (ET). In these types of covers, the objective is to minimize net percolation by returning most of the infiltrating waters from storage to the atmosphere via ET.

To achieve 100% of the function for each one of the end members is practical. Rather, a continuum exists, which is a function of climate, hydrogeology, materials and vegetation. It is important to understand that all cover systems provide store and release components as well as diversion functionality under a specific set of conditions. Therefore, cover systems functionality occurs as part of a continuum with a focus on a store and release end member focusing on enhancing storage and ET components of the water balance and a diversion end member relying on runoff and interflow being dominant components of the water balance (Figure 2). Because cover systems exist as a point on a functionality continuum, Store-and-release covers still possess water-shedding abilities when infiltration excess runoff occurs during high intensity precipitation events. Likewise, cover systems with a focus on water diversion, often possess vegetation or small amounts of storage that contribute to evapotranspiration. Therefore, in using climate as a primary filter to refine cover system design alternatives, one can identify on a site-specific basis the dominant mechanisms of the water and energy balances to be enhanced and used in conjunction with subsequent filters (i.e. materials) to best achieve the required performance.

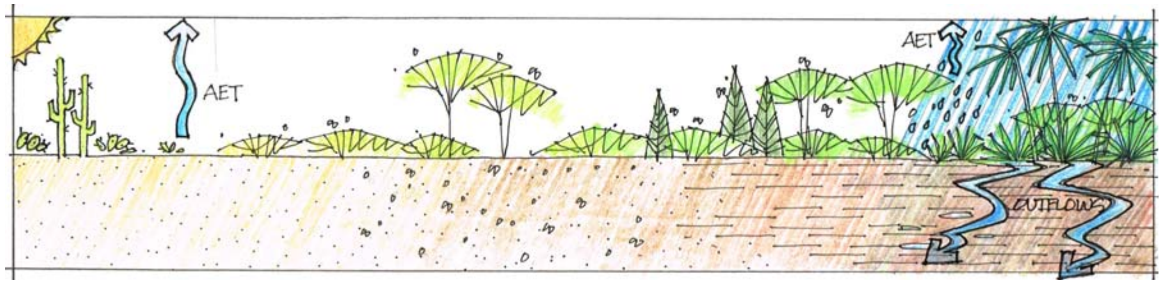


Figure 2 Cover system function continuum for managing NP with dominant water balance components identified based on climate (arid sites from left, to temperate center, and tropical far right of the continuum).

Continuum of Oxygen Ingress Control

Alternatively, or if managing net percolation is also feasible through the use of an engineered cover system design, limiting the flux of oxygen to underlying waste presents another control mechanism for managing production of ML/AMD. Management of gas flux requires addressing diffusion and advection gas fluxes. Therefore, understanding the physical controls on each one of the mechanisms is a key in cover system design.

Control of gas flux resulting from diffusion is done through managing the diffusion coefficient of waste or cover system material. Diffusion coefficients can be decreased through the use of finer textured material, or by decreasing the air filled porosity of the material by increasing water contents. Using this control mechanism often requires near saturated conditions be maintained throughout the year, spanning all seasons (Figure 3). Figure 3 represents a generalized understanding of O_2 control mechanisms based in particular due to soil water conditions to control advection. Further refinement using material filters will identify if controls can be implemented. If there are seasons where evaporation grossly outweighs precipitation, sufficient storage capacity is an engineering requirement that needs to balance water requirements for plants and maintaining cover system saturation. Alternatively, diffusive flux can be controlled by decreasing the diffusion gradient by altering path length or concentration differences. Altering the path length will be largely managed through geometrical constraints of landforms, while controlling differences in concentration will largely depend on internal waste geochemistry as atmospheric concentrations remain static.

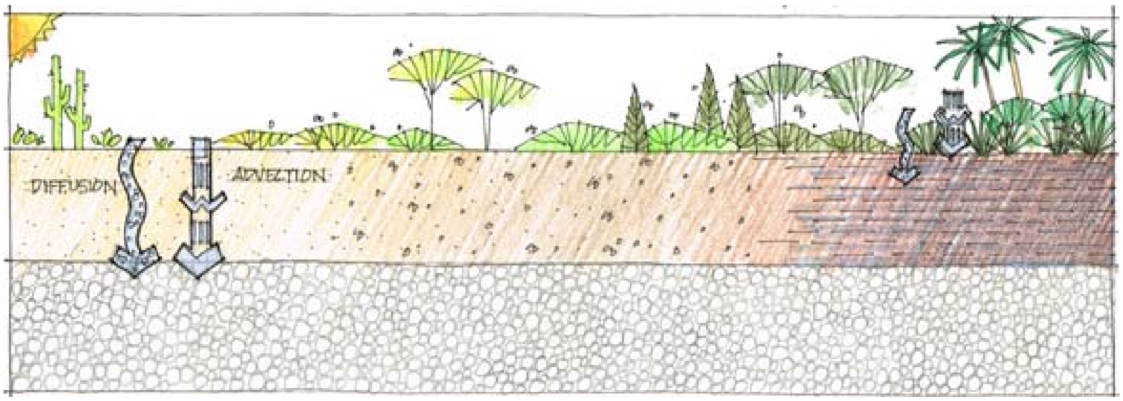


Figure 3 Cover system function continuum for managing O_2 with dominant gas transport mechanisms generally identified based on climate. Arid sites from left to temperate center and tropical far right of continuum.

Continuum of Erosion Control

Conceptually, like NP and O_2, erosion potential occurs as part of a cover system continuum of functionality. Generally, cover system erosion potential is controlled by preventing raindrop erosion and slow surface water velocity in bare areas. This can be achieved by managing: material texture, slope length, slope angle, and vegetation; that will affect erosion simultaneously on cover systems, producing a unique potential erosion risk. Erosion management should aim to achieve the same erosion potential across the entire landform. Due to differences in water balance, energy regime and topography across a landform, multiple strategies may need to be employed to achieve equal erosion rates.

Immediately following construction, or if rapid establishment of vegetation is difficult (i.e. cold climates) more emphasis on material selection and landform design becomes important. A reliance on vegetation can be used to provide long term erosion protection, other aspects of the cover system design must be manipulated to create a stable landform with acceptable mass loss. Erosion potential generally increases with greater slope length or angle, coarser textured material and decreasing vegetation cover. Therefore, by manipulating these parameters in cover system design, cover systems can be constructed with increased physical stability even in the absence of vegetation.

General Cover System Design Alternatives

In order to meet the chemical, physical and land-use objectives described in Section 2, various cover system designs may be employed. Cover system design alternatives are described early in the document at a high level to provide context for future chapters. For the purposes of describing the appropriate cover systems, the designs have been divided into the following six categories: erosion -protection cover systems, store-and-release cover systems, enhanced store-and-release cover systems, barrier-type cover systems, and saturated soil or rock cover systems.

FILTER FRAMEWORK FOR COVER SYSTEM DESIGN

This guidance utilizes a filter framework for addressing cover system design globally. The premise is based on a filter set used to inform cover system design. Filters, as defined for this framework represent an attribute of the site (climate, hydrogeology, materials, etc.) that further constrain design alternatives to achieve the desired performance (Figure 4).

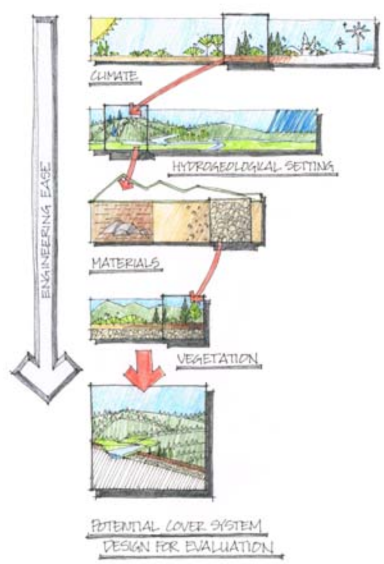


Figure 4 Filter framework for cover system design. Climate represent filter with largest impact on cover system design, while also representing a site attribute not easily modified through engineering design compared to vegetation.

Before any information is provided regarding climate, unlimited cover system design alternatives exist. As soon as climate is specified and seasonality is understood, particular designs may no longer be capable of meeting performance criteria. For example, Figure 5 below demonstrates that on a conceptual basis, tropical sites typified by large magnitude precipitation events, will have a more difficult time in managing NP as compared to an arid regions with similar cover design.

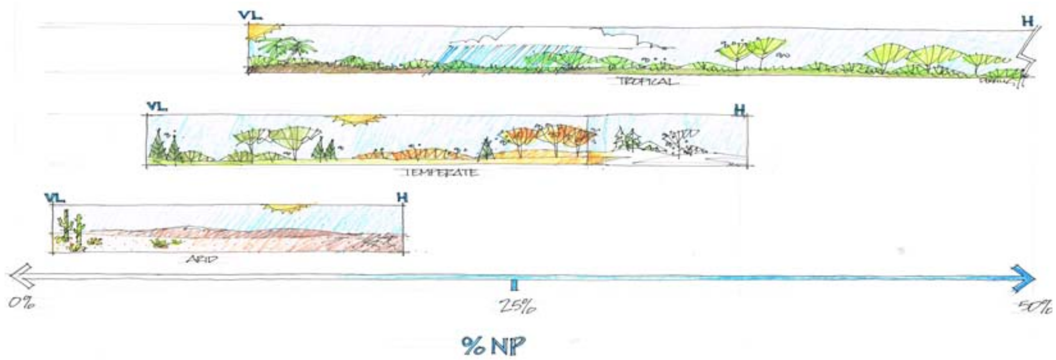


Figure 5 Conceptual net percolation management based solely climate generalizations of the Köppen classification system.

The broadest filter is used to refine gross climatic understanding. With regions of the world divided according to the Köppen-Geiger classification system (Peel et al., 2007), one is able to express the seasonal and annual tendencies of a region based on precipitation inputs and temperature. Precipitation and temperature are integral parameters in understanding the physical processes which govern gas and water transport within mine waste material and geochemical reaction rates. Although covers systems represent a continuum of functionality, the climate filter allows the designer to rapidly refine conceptual designs and identify dominant physical processes that can be exploited and/or enhanced to achieve performance criteria. This first filter also represents an aspect that cannot be modified through engineering efforts, or at least to any significant affect; therefore, the filter forms the base of the conceptual design.

CLIMATE CLASSIFICATION

Major climate regions were selected based on the occurrence of past, current and future exploration efforts as indicated with the Map Mine Mapper tool (InfoMine, 2013). Composite maps were used to overlay point data of mining activity with each major climate type (Figure 6). Although not all climate regions possess wide spread mining activity, each climate type serves to capture major concentrations of mining. Although the location of mining operations is determined by geology, it is evident that although the majority of mining in the world is generally concentrated based largely on geology, mining activity is wide spread, spanning a full spectrum of climates. Thus, the framework must include cover system guidance applicable to all climate types, using climate type to quickly focus suitable cover system alternatives.

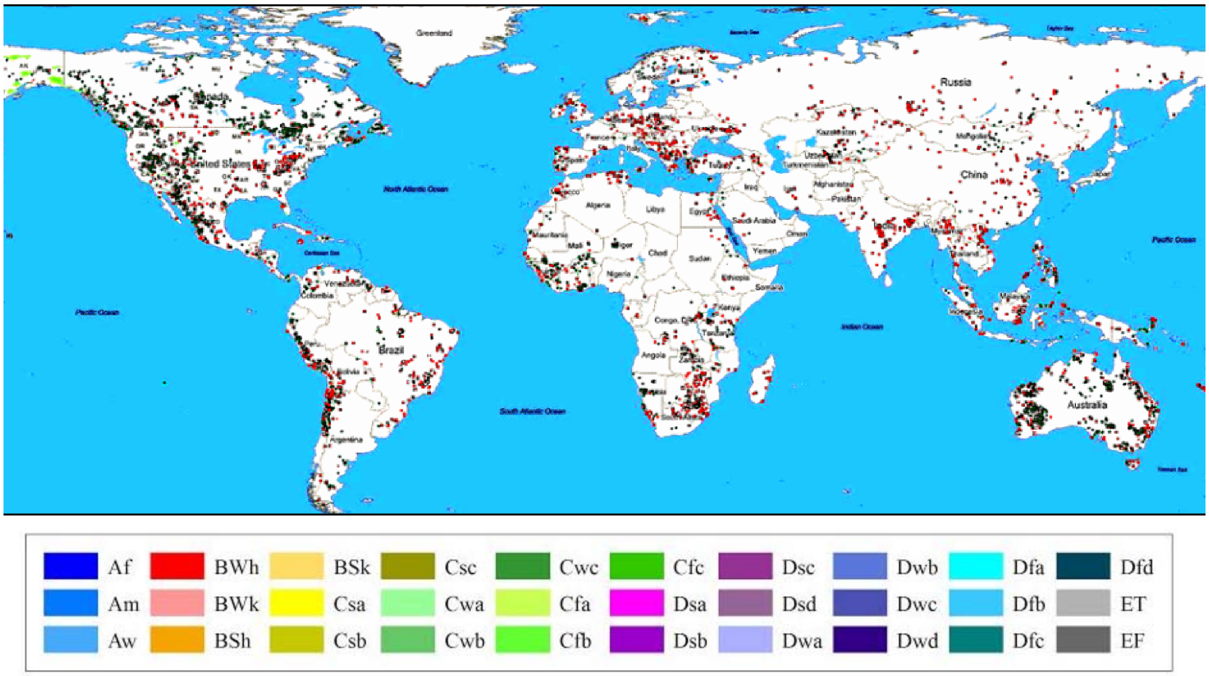


Figure 6 Major regions focused in the guidance will include all climates. Existing mines (red dots) and proposed future mining (green dots) occur throughout the climate spectrum.

Regions of the world are divided based on a large global data set of long-term monthly precipitation and temperature station time series. These climatic thresholds were developed in part due to field observations using landscapes signals such as vegetation. Due to its strong ties to landscape signals such as vegetation and related soil development, the Köppen system is attractive for the framework put forth in this guidance document, which includes filters for climate, materials and vegetation. By combining the major climate region with each sub classification based on PPT and temperature seasonality, a more refined site-specific picture exists compared to conventional annual averages where many critical elements are lost or hidden in the average

The climate classification system then allows designers to quickly assess climatic inputs and the major elements that can be exploited and used for cover systems depending on the climatic setting. For each sub classification, the guidance aims to highlight cover system components to be enhanced, exploited or combined to achieve performance, without further setting, material and vegetation inputs.

Although climate represents the broadest filter and most largely refines design, other filters exist and require site-specific data to further focus design alternatives. Climate variability, climate change and climate controls on NP, O_2 and erosion is discussed in greater detail within the document. Following climate, the next filter for cover system alternative refinement is for materials which includes general discussion on availability of cover system materials, characterization and material evolution. More technical information is provided in appendices as to allow the conceptual framework from being lost in technical detail. Furthermore, micro-climate attributes resulting from topography are explored as potential tools to alter landform water and energy balances

ACID PRODUCTION

ML/ARD is usually a critical aspect within mine closure plans, including the design of cover systems. Demonstrating the benefits, or rather the magnitude of benefits, provided by cover systems to prevent/mitigate adverse impacts to receiving environments including the implications for water management (quantity and quality) and water collection/treatment systems is often challenging.

For this guidance, the key geochemical to be addressed is the oxidation of sulphidic material, recognizing that many other geochemical issues may be present on a specific site. Identifying if / what geochemical problems exist for a specific site should be a critical first step in the cover system design process such as outlined in the GARD Guide. Identifying the geochemical processes will highlight mechanisms the cover system design should seek to control. In most situations, the chemical loading principals and conceptual models presented may still be largely applicable. For other reactive materials, oxidation is a major issue as is managing NP to prevent mobilization. NP usually leads to the flushing of first pore volumes, in which the majority of COCs will be presented. Therefore, identifying the geochemical problem is a critical first step in understanding how it can be managed.

<u>Two Conceptual Models for Acidity Generation</u>
The guidance document discusses two competing models for acidity generation from a theoretical standpoint. The first assumes acidity load increases linearly with increasing NP, while the second model assumes acidity load remains constant with changes in NP. Practically, the acidity load must be zero when the NP rate is zero for model 2 and inherently assumes that there is a 'jump' to a high acidity load with very little net percolation.

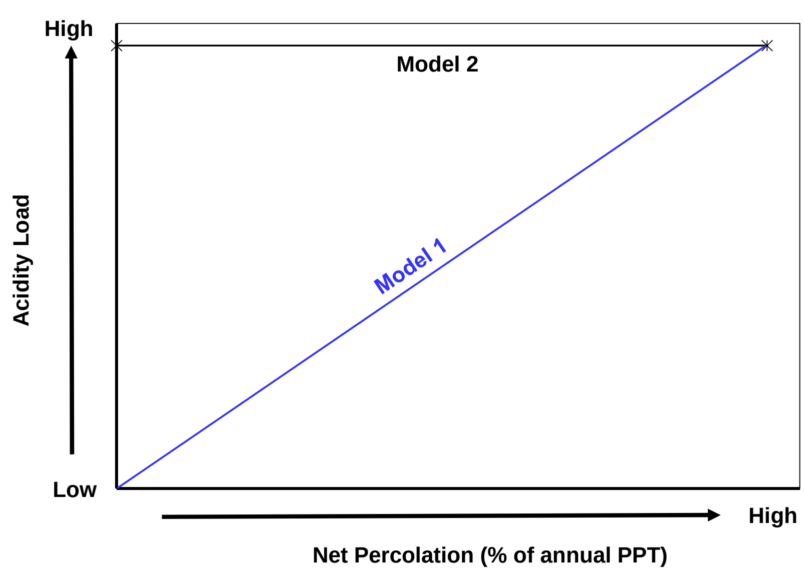


Figure 7 Conventional conceptual models showing relationships between acidity load and NP.

Given that: Acidity Load = [Solute Concentration] x Flow (i.e. NP), Model 1 results in constant ML/ARD solute concentration, and independent of NP (Concentration 1 in Figure 7). Model 2 assumes very low NP will result in very high acidity concentration, while high values of NP, will contribute to very low acidity concentrations due to dilution (Concentration 2 in Figure 7).

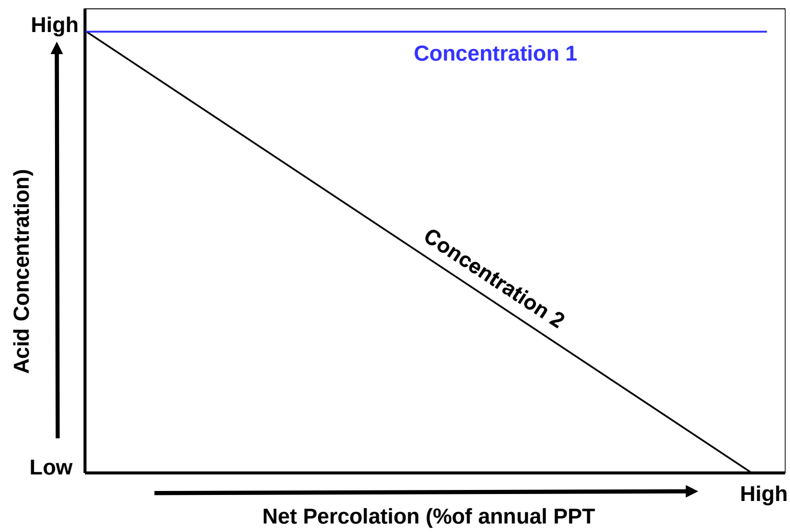


Figure 8 Acid concentration and NP relationship from conceptual models.

Model 1 (acidity load increases linearly with NP) can represent two situations where the waste storage facility contains stored acidic oxidation products that are flushed out with increasing NP where ongoing sulfide oxidation generates secondary acidic oxidation

It is likely that the mass loading from a waste rock dump will be due to some combination of solubility control (Model 1) and reaction control (Model 2). For example, early mass release may be dominated by the release of pre-existing weathering products (Model 1) followed by a longer term reaction controlled release (Model 2). Understanding the models or any transition between the models is important for cover system design and understanding acidity loads. In light of the limitations of the described conceptual models, this guidance document considers the utility of an alternate conceptual model for acidity loading, which takes into consideration the role that both oxygen ingress and net percolation can have on acidity load generation in waste storage facilities.

The conceptual basis for this combined model is that chemical load production (Load) is a function of both the rate of oxygen ingress and the rate of net percolation. The first function makes loading reaction limited while the second, solubility limited. Loading will always be zero if NP is zero; however, even if there is no oxygen ingress there could be potential loading from NP if there was an initial chemical load present due to pre-dump placement weathering. A conceptual 3D model that describes geochemical behavior is presented within the guidance that could be populated with real data to determine the best cover system design strategy. The timing of cover system employment will also determine the design as will the performance value; two aspects discussed within the geochemical load production chapter.

SITE WIDE WATER MANAGEMENT

Water occurs as a resource on the landscape, and requires a management strategy that aligns with existing or proposed closure objectives. Water resources could be in abundance to the level of becoming difficult to manage or in such scarcity requiring finite management. Generally, management of water on a specific mine feature can be divided based on three management scenarios:

1. To isolate reactive materials and diverting all water from the landform or site;
2. To capture and contain all water on a landform/ site (i.e. to prevent release to the receiving environment) and/or;
3. To utilize a portion of the water for onsite process (i.e. maintain saturated barriers or sustain vegetation), while redirecting surplus water off the landform.

In reality, a mine closure landscape may utilize all three forms of water management concurrently or in sequence throughout the closure landscape depending on specified closure objectives for each reclaimed area. Additionally, following spatially considerations for water resources, temporal considerations can be equally important. The timing and magnitude of water resources may be critical for sustaining reclamation features such as ephemeral creeks, wetlands and vegetation onsite throughout succession, and needs to be considered outside of water quantities.

Sections within the site wide water management chapter include but are not limited to landform – landscape integration and site-wide water management integration opportunities. Lack of thorough site wide water management is a common cause of permit violations, is an impediment to sustainable mine reclamation, and an added cost to mine operators. Often neglected in mine planning, integrated site wide water management planning is recommended to avoid expensive solutions and for maximizing the productive capability of the reclaimed mine landscape. One goal of integrated site wide water management planning is to provide abundant lead-time so that the reclamation landscape can be shaped concurrently with mine operations, and at lower cost.

COVER SYSTEM AND LANDFORM FIELD PERFORMANCE MONITORING

Long-term performance monitoring is critical for evaluating performance of cover systems. It is impossible to develop a single rule for how long monitoring should occur that would apply to every cover system. Instead, a cover system requires developing a monitoring strategy integrated within the design of the cover system, regulatory requirements, the needs of the mine operation and the stakeholders, and most importantly be conducted within the context of the mine closure plan.

Direct field performance measurement as part of a cover system monitoring program is the state-of-practice methodology for measuring performance of a cover system. Field performance monitoring can be implemented during the design stage with cover system field trials, or following construction of the full-scale cover (e.g. Ayres *et al.*, 2007). Direct measurement of field performance of a cover system is the best method for demonstrating that the cover system will perform as designed

CASE STUDIES

The second half of the guidance will include case studies, used to convey the use of the proposed framework for design. Case studies are divided into two categories: 1) generic, hypothetical case studies and, 2) factual case studies.

The generic case study includes a hypothetical mine site of average to large-scale design. Reactive wastes on site are contained in a large-scale tailings storage facilities as well as a large-scale WRD. Both the TSF and WRD measure approximately 700 ha, and are representative of current large-scale facilities. To highlight the importance of climate, hydrogeological setting and microclimate, the generic mine facility components will be examined under different scenarios to determine how cover system performance is expected to be affected. Secondly, a synthesis of factual studies will aim to provide a cross section of mine sites globally in varying climates, hydrogeological setting and unique micro-climates. A one to two page fact sheet will briefly characterize each site and summarize results of the cover system to date. A lessons learned section is also included, and does not aim to highlight any failure specifically. Rather, the objective is to provide an opportunity to disseminate lessons learned on specific sites for the benefit of other sites, and propagate a knowledge base that may not have been available during the design or construction process for another site.

ACKNOWLEGEMENTS

OKC wishes to thank INAP and its members for generous support for the Global Cover System Design Technical Guidance Document. Also, a thanks to members of the TAG; Dr. Lee Barbour, Dr. Ross Hardie, Mr. Mark Logsdon, Dr. Gord McKenna, Dr. Andy Robertson, Mr. Justin Straker, Dr. Jeff Taylor and Dr. Rens Verburg for providing their technical expertise throughout the development of the guidance. Additional reviewers for their comments from Dr. Laurent Ameglio, Mr. Mike Aziz, Dr. David Jones, Dr. Andrew Garvie, Ms. Tania Laurencont, Dr. Ron Nicholson, Mr. Mathew Orr and Dr. Steve Slater were also appreciated.

REFERENCES

Ayres, B., Lanteigne, L., Smith, Q., and O'Kane, M. 2007. Closure planning and implementation at CVRD Inco’s Whistle Mine, Ontario, Canada. In Proceedings of the II International Seminar on Mine Closure, Santiago, Chile, October 16-19.

GARD, 2011. Global Acid Rock Drainage Guide. Property of the International Network for Acid Prevention (INAP). Online. www.gardguide.com. September 2011.

InfoMine. n.d. MapMine Mapper. 2013. http://mapper.infomine.com/

Peel, Murray C., Brian L. Finlayson, and Thomas A. McMahon. Updated world map of the Köppen-Geiger climate classification. Hydrology and earth system sciences discussions 4.2 (2007): 439-473.

DECOMMISSIONING AND RECLAMATION OF THE SANTO ANTÔNIO TAILINGS MANAGEMENT FACILITY

Débora do Vale[1], Giani Aragão[1], Joaquim Pimenta de Ávila[1], Alexandre Matos[2]

[1]Pimenta de Ávila Consultoria Brazil

[2]Kinross Brasil Mineração Brazil

ABSTRACT

The Santo Antônio Tailings Management Facility (TMF) consist of an earth dam built for the purpose of forming a reservoir for the storage of tailings from the Kinross gold mining processing plant from the Morro do Ouro Mine located in the city of Paracatu, Minas Gerais, Brazil. The dam area is about 950 hectares, the highest section reaches 105 meters in height, and the crest length is 5.2km. This TMF was constructed in 1987 and currently is the biggest tailings dam in operation in Brazil. The Normative Resolution COPAM n.º 127/2008, applicable in Minas Gerais, establishes guidelines and procedures for environmental assessment during the mine closure phase and institutes the mandatory development of a closure plan. To satisfy the regulations Kinross started the development of the Santo Antônio Closure Plan with Pimenta de Ávila in 2013 and it was completed at a conceptual level in April 2014. The main purpose of this Plan was to identify, evaluate and propose sustainable closure solutions to promote physical, chemical and biological stabilization of the whole site, making it appropriate for future uses considering acceptable risk levels. The main environmental issues at the site were found to be related to the potential acid drainage generation from the tailings, the geotechnical challenges considering the tailings and downstream slope, and the integration of the closure process with the mine operations. The closure plan was segmented in two specific phases and the TMF was divided into separated components, and specific closure actions were developed for each one. In summary the closure solutions consist of reshaping the downstream slope, implementing a cover system, proposing a surface drainage system and revegetating the covered areas. Finally the closure solution components were integrated to provide a consistent plan for the overall site, and a post-closure monitoring program was developed. The closure plan will be presented to the Brazilian environmental agencies as well as to the local communities.

KEY WORDS

Tailings, dam, cover system, revegetation, future land use, long term stability.

INTRODUCTION AND SITE CHARACTERIZATION

This paper presents the conceptual plan for decommissioning and reclamation of the Santo Antônio Tailings Dam located at Morro do Ouro Mine owned by Kinross in the Municipality of Paracatu, Minas Gerais, Brazil. The Santo Antônio Dam is located to the north of the pit and beneficiation plant; it was constructed for the disposal of tailings from the flotation circuit of the ore processing plant and to store water; it has a total area of 950 hectares (Figure 1). The tailings solids content ranges from 35 to 40%. Tailings are discharged in the reservoir from one disposal channel located in an upstream basin. Tailings flow from the discharge point along the tailings surface and around a dividing berm into the pool area (Figure 1(a)). Fresh water from a well field is also stored in the pool and a barge pumping system returns the water to the ore processing plants, refer to Figure 1(a). Currently the Santo

Antônio Dam is 105m high, has a crest length of 5 km, contains 382 Mm^3 of deposited tailings and 9 Mm^3 of stored free water.

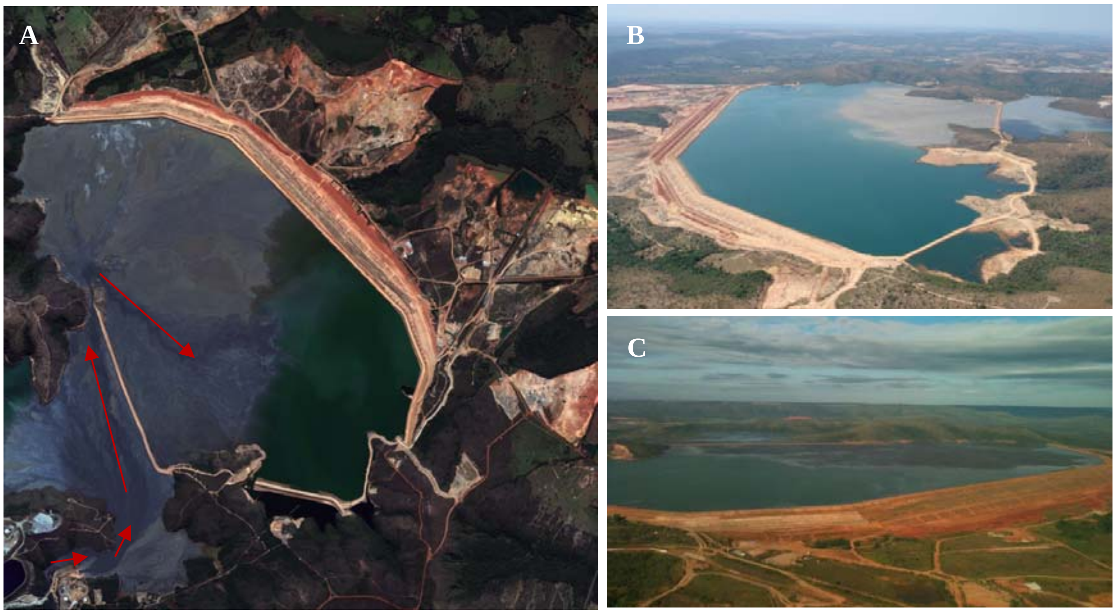


Figure 1. a) Tailings distribution in Santo Antônio Dam; b) View of Santo Antonio Dam in 2009; c) View of Santo Antônio Dam in 2014

The Dam started operating in 1987 and disposal of tailings is scheduled to end in the next 5 years. Until the end of the mine lifecycle forecast, in principle for 15 years from now, part of the reservoir would remain in operation for impounding and storing fresh water to provide processing water demand.

In fulfillment of legal obligations imposed by the environmental regulatory agency and in compliance with company internal procedures, Kinross prepared, with the support of Pimenta de Ávila Consultoria, the Conceptual Plan of Decommissioning and Reclamation for the Santo Antônio Dam. The main purpose of this project was contributing towards the sustainable development of the region through identification, assessment and proposal of closure solutions which would promote the environmental integrity (physical, chemical and biological stability of the entire site) and which are technically and economically acceptable to the Company and community enabling a future use of the area; the corporate and regulatory policies were also considered in the development of the plan. The major challenges associated with this project were related to the characteristics of the tailings, geotechnical issues, compatibility of closure of the dam with the mine operations and ultimately the large dimensions presented by the structure which is now the largest tailings dam in Brazil.

ALTERNATIVES FOR FUTURE USE

Options were developed for the post-closure use of the Santo Antônio Dam considering the attributes of the area, intentions of post-operational use, landscape adequacy and possibility of transfer of custody. It is also important to maintain physical, chemical and biological stability in the long term. The assessments focused mainly on three options:

- Environmental conservation,

- Environmental conservation associated with scientific research oriented to local college institutions, and
- Agricultural use.

All alternatives were assessed through a qualitative classification methodology envisaging technical, environmental, economic and socioeconomic variables. A point score was assigned to each aspect and the alternatives were compared by obtaining a sum of the scores for all aspects per alternative. The highest score reflects the most preferred alternative. Table 1 shows the scores that each alternative received for each aspect evaluated, considering the implementation phase of the closure solution and during future land use.

Considering the risks and restrictions associated with the structure and the results shown in Table 1 the future preferred use for the area is mixed use of environmental conservation and scientific research aiming to boost the research and learning infrastructure of local college centers by promoting technical and academic knowledge applied to mining activities and establishing the township as a regional academic pole. Subsequent to the implementation of closure and post-closure monitoring the area will be destined to environmental conservation which is in alignment with economic zoning established for Paracatu Municipality; the actual involvement of the community will be part of the next stage of the closure project, referred to as the basic project development stage.

GENERAL DECOMMISSIONING GUIDELINES

Considering the challenges associated with the structure the following strategy was adopted: the structure was divided in two different areas, namely the reservoir and downstream slope. The closing process was also divided into two stages: Stage 1 occurring during the Mine operation and considering the partial closing of the dam, and Stage 2 considering the full closure of the dam concurrent with the decommissioning of the entire mine site.

Guidelines for Closure of Downstream Slope

The downstream slope of the structure at closure should have a slope and geometry that will provide stability for long-term conditions. Long-term stability considers physical stability, control of erosion processes and restoration of vegetal cover. Thus it was essential to assess, in an integrated way, the geotechnical aspects, surface drainage and re-vegetation.

Table 1: Comparative Analyses between Future Use Alternatives

CLOSURE PLAN - FUTURE LAND USE - SANTO ANTÔNIO TAILINGS MANAGEMENT FACILITY - MORRO DO OURO MINE					
PERIOD: CLOSURE					
VARIABLE	FACTOR	ASPECTS ANALYSED	Alternative A	Alternative B	Alternative C
			Environmental Conservation	Environmental Conservation and Scientific Research	Agricultural Use
TECHNICAL	Physical Stability	Effort in geotechnical works for ensuring physical stability	5	5	1
		Effort in drainage works for ensuring physical stability	5	5	1
		Effort in revegetation works for ensuring physical stability	5	5	9
	Chemical Stability	Effort in works for ensuring chemical stability (Surface and Groundwater, Soil and	5	5	5
	Flood of the Project	The structures attend the flood for closure condition	9	9	9
ENVIRONMENTAL	Impact	Degree of impact resulting from implementation	9	9	1
ECONOMIC	Implantation	Agility in implementation	5	5	1
	Period of Implantation	At the end of useful life/During operation/Progressive	9	9	1
SOCIOECONOMIC	Access	Existing accesses are positive for the future use	9	9	9
	Transfer of Custody	Agility in the process of transfer of custody	5	7	1
SUBTOTAL FOR IMPLANTATION			**66**	**68**	**38**
PERIOD: POS CLOSURE					
VARIABLE	FACTOR	ASPECTS ANALYSED	Alternative A	Alternative B	Alternative C
			Environmental Conservation	Environmental Conservation and Scientific Research	Agricultural Use
TECHNICAL	Physical Stability	Promote physical stability in long term	9	9	1
	Chemical Stability	Promote chemical stability in long term	9	9	1
ENVIRONMENTAL	Biological Stability	Promote biological stability in long term	9	9	1
	Impact	Degree of impact during regular use	9	9	1
	Landscaping Adequacy	Promote landscape integration	9	9	1
ECONOMIC	Monitoring and Maintenance	Condition: Walk way/Passive care/Active care	5	5	1
SOCIOECONOMIC	Employment	Promote employment	1	5	9
	Community	Aim to attend a community necessity	1	9	5
		Encourage an specific community potential	1	9	5
	Income	Provides Income for economy and community	1	5	9
	Health and Human Well-Being	Promote well-being and health for the community	5	5	1
SUBTOTAL FOR POS CLOSURE			**59**	**83**	**35**
Alternative A	Environmental Conservation			**125**	
Alternative B	Environmental Conservation and Scientific Research			**151**	
Alternative C	Agricultural Use			**73**	
				BEST ALTERNATIVE	**B**

Several alternative downstream slope configurations were evaluated and the major challenge was minimizing the large volumes of material required for re-sloping. The criteria for selection of the preferred alternative focused on long-term physical stability and the feasibility of constructing this alternative; the following geometry for the closure of the downstream slope was selected:

- Slopes between berms with an inclination of 1V:1.7H corresponding to an overall slope equal to an overall slope angle of 1V:3.2H;
- Horizontal berms parallel to the crest, 6m wide with longitudinal slope of 0.5% and cross slope of 5.0%. These berms are spaced at 7.0m vertical height.
- Surface drainage system on the berm consists of compacted laterite (30 cm layer) flowing to downslope concrete channels spaced at 400m along the entire slope. The drainage system design was based on a return period of 1:1000 years;
- Slope stability safety factors must be above 1.5 as required by Brazilian standards. The safety factors were assessed for three sections of the dam: right side abutment, central section and left side abutment. In all analyses carried out the values found were above the required by the Brazilian standards (See Figure 2);

Figure 2: Sections of Analyses of Stability and safety factors found for the closure condition.

- The erosional loss of soil for the proposed configuration was calculated as 4.52 ton/ha/year using the RUSLE methodology and considering vegetal cover around 60-80% (C=0.03);
- The volumes of cut, fill, material handling, implementation of drainage systems and re-vegetation of the slope face were estimated for purposes of cost comparisons.

The alternatives considered for geometry and drainage at closure of the downstream slope of the Santo Antônio Dam aimed at maintaining physical stability and also to control erosion in the long term. The re-vegetation plays a critical role in reaching these goals and also results in landscape integration. In the short term the re-vegetation of the slope aimed at the slope stability and control of erosion processes. In the medium and long term it is expected that the area will be integrated into the native vegetal appearance of the area.

To maintain low levels of soil loss the selection of species was based on the following criteria:

- Low vegetation height or herbaceous size;
- Medium depth root penetration;
- Fast growth and considerable production of biomass;
- Adapted to conditions of local water availability (precipitation);
- Providing good rates of ground cover favorable to erosion control and slope stability.

Based on these criteria and the proposed embankment geometry different options of grass, small size leguminous and some shrubs were assessed to form a 60-80% cover of the downstream slope with grass and 25% with shrubs. Field experiments in the area were carried out for 10 species and the ones presenting the best performance, Bermuda and Pensacola grass, were recommended for utilization at the closure stage (Figure 3).

Figure 3: Result of experiment carried out in the field with Bermuda Grass

The full closure of the downstream slope is scheduled to be implemented and completed during Stage 1 of Decommissioning.

Guidelines for Closure of the Reservoir

The concepts for closure of the dam beach and reservoir area required the assessment of storage of water in the pool area, the unit water demand, the chemical stability of tailings, implications of conventional and multilayer cover systems and the geotechnical characteristics of tailings. As the Santo Antônio Dam is a multi-use structure for disposal of tailings and impounding of water, the technical closure solution adopted considered the storage volumes available in the reservoir, the volume require to regulate flow rates and prevention of discharging tailings liquid into the environment. The volume available in the reservoir should be sufficient to store the water volume required to meet the water demand, the volume of tailings to be deposited until the end of the structure life cycle and still maintain a volume available for the tailings surface decommissioning measures.

As to the chemical stability of tailings the data provided showed that tailings with different characteristics have been disposed of in the dam throughout the operation. The composition of the tailings requires that the area should be covered in order to result in environmental conditions satisfying the applicable legislation and standards.

A cover system must isolate the tailings from the environment by controlling the ingress of oxygen and must also limit the water infiltration to minimize the generation of leachate. Due to the climatic conditions in the region and the large extent of the area, dry cover options were assessed using multi-layered cover systems that include a conventional hydraulic barrier and store and release components.

Conventional hydraulic barriers use layers of clayey material with low hydraulic conductivity to minimize infiltration and maximize surface runoff and evaporation. Despite being a very simple type of cover it requires strict quality control during construction to establish a uniform low permeability layer on the whole area. Still in the long term degradation of the layer may occur with increase of permeability and loss of efficiency chiefly associated with occurrence of cracks.

The store and release type of cover is a multilayer system with specific functions. The upper layer formed by vegetation (top soil) is used to enhance evapotranspiration; the intermediate layer formed by lightly compacted silty soil, called “store and release”, accumulates infiltration water during the rainy period and releases it by evapotranspiration during the dry period. A third layer of clayey soil was included to act as a hydraulic barrier thus reducing entrance of oxygen.

Simulations of the behavior of the two types of cover were carried out assessing infiltration and maintenance of saturation of materials using the Vadose/W program considering climate, vegetation and geotechnical parameters. Figure 4 shows the variation of the degree of saturation in the profile used for the simulations in Vadose /W over one year. During this interval, the degree of saturation in the store-and-release layer ranged between 35 and 100% and the hydraulic barrier layer remained saturated during the entire period of the simulation, showing its efficiency. At the layer of tailings the saturation remained above 85%, which prevents the inflow of oxygen.

The higher the thickness of the store-and-release layer the lower was the infiltration rate. Considering the results obtained for the store-and-release cover associated with a hydraulic barrier this became an alternative for use in the decommissioning of the Santo Antônio Dam. Despite being more robust, this type of cover presents higher physical stability over time and it is efficient in reducing water infiltration and minimizing oxygen entrance.

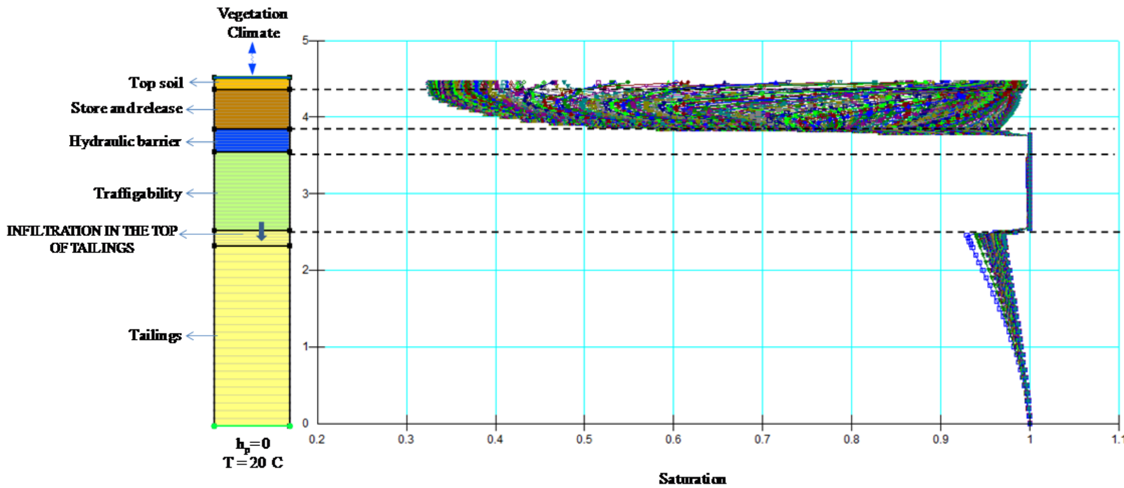


Figure 4: Result of Modeling of Store and Release Cover

For Stage 1 Closure (ending with disposal of tailings until the decommissioning of the Mine) the suggestion was, conceptually, to construct only one trafficability layer of variable thickness and minimum value of 1m in the upstream region and near the left side abutment on top of tailings. The implementation of such a trafficability layer would allow equipment traffic on top of the tailings surface and promote the re-sloping of the surface to direct surface water flow. This cover will only partially cover the tailings surface as the pool will be maintained during this period for the supply of the processing plant water demand. The presence of the lake inevitably implies variation of the water level which could cause damage to the cover system if it were fully implemented. Stage 2 considers the emptying of the reservoir lake by pumping the water to the exhausted mine pit; placement of a full cover system consisting of a store-and-release layer (formed by saprolites) over a hydraulic barrier layer (formed by low permeability clayey material). A topsoil layer will be placed on top of the multi-layered cover.

The numerical modeling used to design the cover system presents limitations related to the simulation of certain field conditions such as, for instance, occurrence of cracks, heterogeneity of materials and their permeability. Thus further studies, such as filed trials, will be required for the development of a final closure plan.

The tailings are discharged at a solids content between 35 and 40%, followed by sedimentation and subsequent self-weight consolidation as the successive layers are deposited over the years the years. A one-dimensional consolidation analysis was developed for the numerical modeling of filling of the reservoir, sedimentation and self-weight consolidation plus consolidation caused by the surcharge of the cover system. This provided a first approach to the effects of the partial and full closure of the future surface elevations and where material will be required to cover depressions which will be formed due to consolidation of the tailings over time to avoid the formation of ponding.

The limiting factors for the revegetation of the area are the instability associated with the consolidation of the tailings, the tailings substrate composition, which is a poor soil due to the absence of organic materials and the possible residual presence of arsenic. The selection of species was based on the assessment of several studies carried out in the field by Kinross with research institutions which addressed the local chemical characteristics and their influence on the establishment of species, the influence of the presence of one soil layer in the cover system and a survey of species that presented good adaptation to the area. The implementation of a cover system is beneficial to the selection of species as it avoids the migration of possible contaminants to the soil surface by providing better conditions for the development of vegetation and maintaining the arsenic levels within the limits set forth by the applicable legislation. Table 2 below shows a summary of species which presented positive results in the assessed studies with good growth rates and good adaptation to local conditions.

Table 2: Summary Table of Species assessed by Previous Studies

Size			Melo 2006- Species for Phytoremediation	Neri 2011-Species of occurrence by natural regeneration	Assis 2006-Species well adapted to the coverage systems
Herbaceous	Gramineous	Low	*Lolium multiflorum*	*Axonopus marginatus* *Andropogon bicornis* *Aristida ekmaniana* *Digitaria ciliaris*	-
		Shrub	-	*Sabicia brasiliensis*	-
	Leguminous	Low	*Stilozobium aterrinum* *Arachis pintoii*	-	-
		Shrub	*Stylosanthes humilis* *Crotalaria spectabilis* *Sesbania virgata* *Leucaena leucocephala*	*Stylosanthes viscosa*	*Acacia holosericea*
Arboreal			*Eucaliptus grandis* *Corymbia citriodora*	*Simarouba amara* *Maprounea guianensis*	-

The recommendation arising out of such assessments was that the re-vegetation of the Santo Antônio Dam reservoir should occur only in Stage 2 of Closure after the implementation of the final cover system and a layer of organic soil (top soil) of approximately 20 cm. Considering the results of the field studies, the short-term goals and the restrictions existing in the area, the recommendation was to plant only small sized herbaceous species. Subsequently when the reservoir area physical stability is assured arboreal size species may be added. It is expected that, along with the development of the vegetation and gradual increase of diversity of species the return of local fauna and fauna may also occur so that little by little the closed area will be integrated with the local landscape and biological diversity will be reestablished.

Guidelines for Closure Stages

Figure 6 illustrates the design for closure expected for the 1st and 2nd Stage of Closure. Figure A shows the 1st Closure Stage consisting of the partial closure of the dam with the implementation of the trafficability layer on the tailings surface, the presence of the lake and the reshaping of the downstream slope. Figure B shows Stage 2 closure consisting of the total closure of the dam with the implementation of a full cover system and re-vegetation.

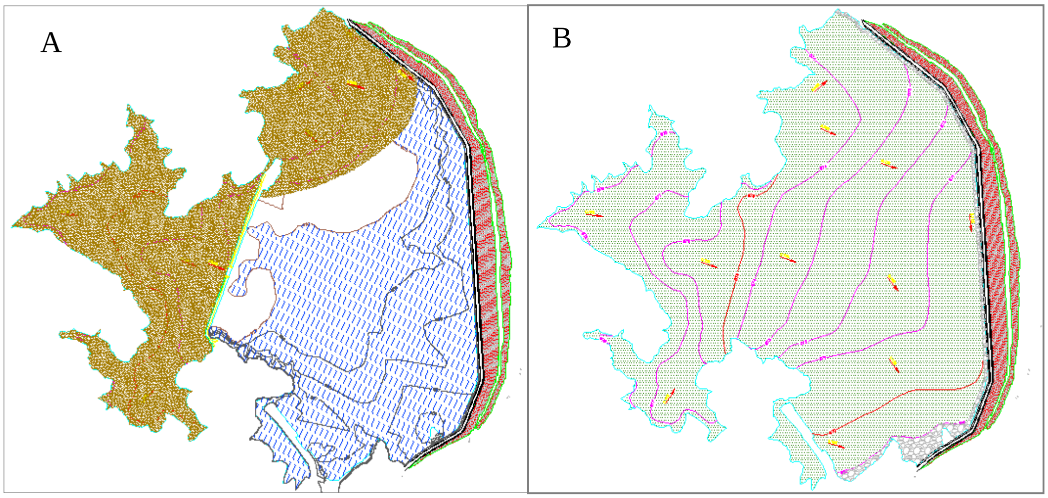


Figure 6: Schematic illustration of conceptual closure stages of Santo Antônio Dam:
A) Stage 1: Partial Closure: Presence of lake, placement of trafficability layer and reclamation of downstream slope
B) Stage 2: Full Closure: Implementation of full cover system and revegetation

MONITORING

Monitoring will be used to obtain information before the implementation of closure components as well as the success of the implemented closure actions and the need (or not) for maintenance. Initial monitoring was planned for a 5-year period however it should also continue after that until acceptable conditions set forth by the legislation and required for closure will be reached. The Santo Antônio Dam Monitoring and Maintenance Plan considered:

- Monitoring the physical stability (by visual inspections and instrumentation),
- Monitoring of chemical stability (assessment of quality of ground and surface waters and quality of soil), and
- Monitoring of biological stability (assessment of reestablishment of species and evolution of vegetal cover).

During the proposed monitoring the need for maintenance should be assessed although no major problems are expected. Corrective actions will be implemented as necessary. Monitoring should be more frequent just after the implementation of closure actions and may become more sporadic as the conditions stabilize and the requirements agreed upon with the supervising agency are approached.

SUCCESS INDICATORS

Success indicators for closure of Santo Antônio Dam were established that are aimed at verifying the performance of the proposed decommissioning and reclamation measures. Such indicators will enable Kinross to perform maintenance and other actions to maintain stability and ongoing performance of the components. Some of the suggested indicators are:

- ✓ Reduction of soil loss on the downstream slope;

- ✓ Surface and groundwater quality;
- ✓ Increase diversity of species of flora on site, mainly native species; sssessment of integration of reclaimed areas with the local landscape.

FINAL CONSIDERATIONS

The concept of continuous planning for closure is in alignment with the best practices adopted by developed countries. In Brazil the requirement of the company's commitment to closure of a mine is a relatively new development due to the recent legislation on closure.

The Santo Antônio Dam Closure Plan was developed in a conceptual (initial) form and its main purpose was the identification of applicable closure alternatives while seeking the balance between the technical, environmental and socioeconomic variables. It was done based on information available at the time of preparation.

For the next phase of the project new information will be produced and will be used in relation to the adopted cover system, the proposed timetable and actual community involvement. Therefore some of the proposed approaches may be adjusted and changed.

ACKNOWLEDGEMENTS

The Pimenta de Ávila team developed the closure plan covered by this paper and the authors would like to thank all personnel directly involved with this project. The authors also would like to thank Kinross for the support and their consent for the publication of this paper.

BIBLIOGRAPHIC REFERENCES

AMORIM, N. R. (2008). *“Evaluation of Field Performance os Cover Systems at Rio Paracatu Mine (RPM)”*, Doctor's Thesis, Universidade Federal de Viçosa, Viçosa, Brazil.

ASSIS, I. R. (2006). *“Mitigation of arsenic, iron and sulfur leachates and revegetation from mined substrates in acid mine drainage occurrence area.”*, Master's Thesis, Universidade Federal de Viçosa, Viçosa, Brazil.

BRASIL. Constitution (1988). *Constituição da República Federativa do Brasil.*

MELO, R. F. (2006). *"Potential of vegetal species for phytoremediation of soil one contaminate by arsenic."*, Doctor's Thesis, Universidade Federal de Viçosa, Viçosa, Brazil.

MINAS GERAIS. (2008) *Deliberação Normativa COPAM nº 127* of November 27, 2008: It sets forth guidelines and procedures for environmental assessment of the mine decommissioning stage.

NERI, A. V.; SOARES, M. P.; NETO, J. M.; DIAS, L.E. (2011). *"Cerrado Species with Potential for Recovery of Degraded Areas for Gold Mining, Paracatu-MG"* *Revista Árvore*, Viçosa-MG, v.35, n.4, p.907-918.

ORTIGÃO, J.A.R; PAIVA, P.; FAHEL, A.; LANDI, A.; SOUTO, R. (2013) "Trial Embankment on Gold Mine Tailings". Soils and Rocks, São Paulo, 36(1): 3-19, January-April, 2013, p 3-19.

PIMENTA DE AVILA (2014). Report KR-120-RL-29815 – Concept Plan of Decommissioning and Reclamation of Santo Antônio Dam.

TOWARDS CLOSURE OF THE FIRE ROAD ARD SITE IN NEW BRUNSWICK

K.D. Phinney, P.Eng.[1]
M. Coleman, P.Geo, P.Eng.[2]

[1]Consulting Chemist/Chemical Engineer, Halifax, NS
[2]Manager of Environmental Services, Mine Reclamation Inc. (subsidiary of NB Power), Fredericton, NB

ABSTRACT

The approximately 120 ha backfilled Fire Road coal mine cut, located near Fredericton New Brunswick, has been a source of acid rock drainage since the mid 1980's. Lime neutralization treatment of drainage has been continuously ongoing. The cut, with depths to approximately 20 m, is located in sandstones containing iron sulphides, principally pyrite, in the range of 1 to 2 wt%.

Site water chemistry has been intensively monitored over the years using a series of groundwater wells within and bordering the disturbed areas. This has provided a considerable inventory of water chemistry data which has enabled interpretation of the characteristics of acid generation at the site; the effectiveness of "in situ" neutralization in reducing the ultimate acidity of the drainage, and trends in acidity over the years. During the past number of years a definite trend of decreasing acidity has been observed leading to the conclusion that the site should exhibit "zero lime demand" within the next 10 years. At present, plans are being developed for final closure of the site.

KEY WORDS

acid mine drainage, reclamation, mine water chemistry, groundwater monitoring

LOCATION AND BRIEF HISTORY

The former NB Coal Limited Fire Road coal mine, located near the village of Minto, approximately 40 km north east of Fredericton, New Brunswick, produced approximately 50000 tonnes per year of thermal coal until shutdown in 1987. The general extent of the approximately 120 ha site is shown in Figure 1. Photos 1, 2 and 3 provide an overview of the site at present.

The mine had been in operation since 1983 and was shutdown primarily due to an acid mine generation problem first observed in 1985. Cut and fill mining methods were used to recover a thin seam of coal overlain by as much as 20 m of sandstone bedrock. Both the coal and sandstones contain low concentrations of iron sulphides (< 2 wt% pyrite). The remaining mine cut was backfilled with waste rock to the limit formed by the west highwall after cessation of mining. Surface and ground water which flow into the site are collected in a sump at the southern limit of the site and have been treated by lime neutralization since early 1987. The sump is maintained at a water level to ensure that all water contacting the site drains to the sump. Treatment sludge is returned from the sedimentation ponds to the backfilled waste rock area (Coleman, Whalen, and Landva, 1997).

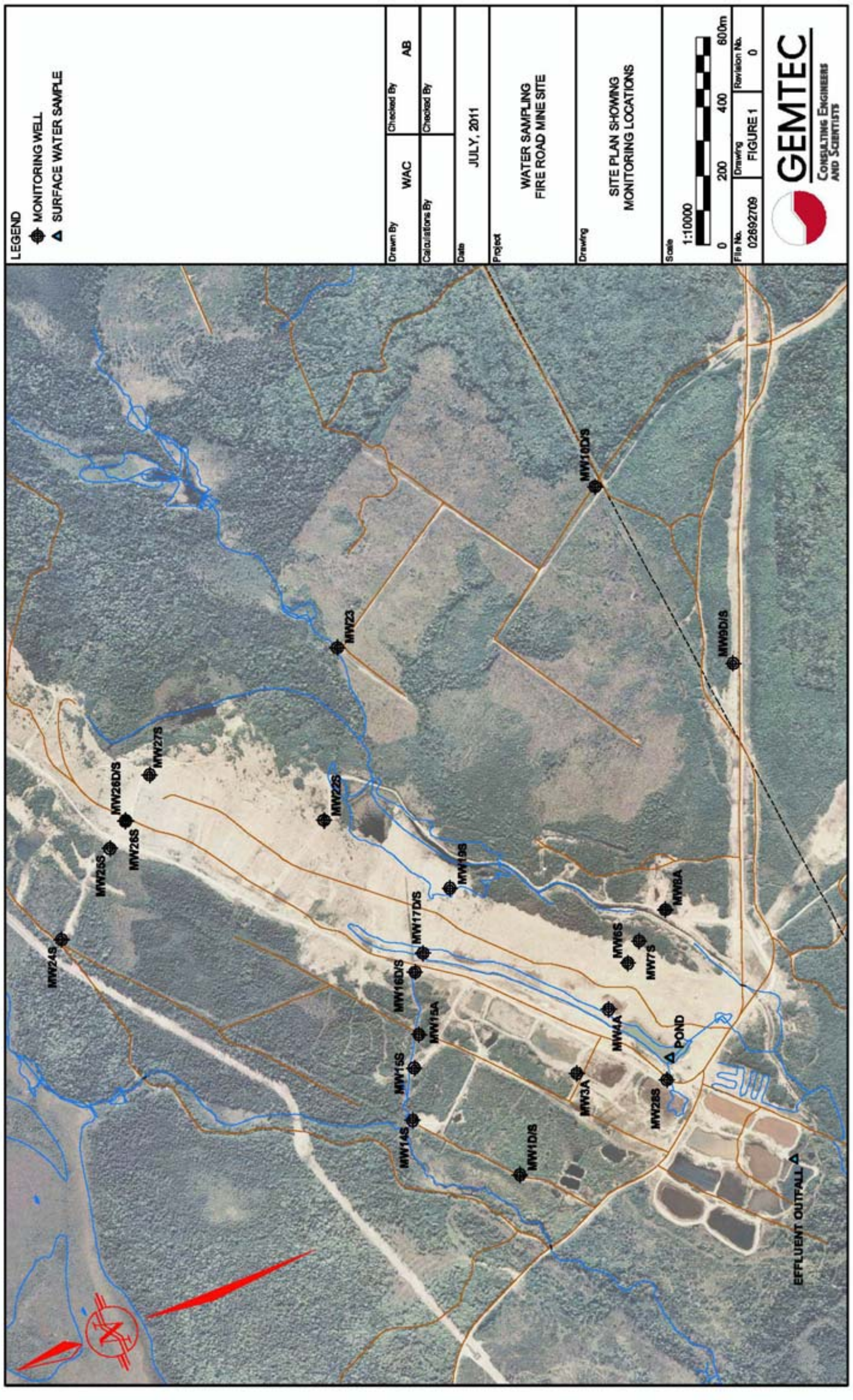


Figure 1. Site Layout

Photo 1. Treatment Plant

Photo 2. Natural Re-vegetation of Backfilled Area

Photo 3 – Sump

WATER CHEMISTRY OVER THE YEARS

Water chemistry data are available for a series of groundwater monitoring wells and the minewater collection sump at the south end of the site. Annual sampling data are available from approximately 1990 for a number of the sampling locations and for a number of years for the remaining locations.

Characteristic water chemistry, as measured at the sump in June 2013, is given in Table 1. Aluminum, iron, and manganese are the main species of concern. An assessment of the data clearly indicates a trend toward decreasing acidity at the sump and at a number of the sampling wells within the mined (disturbed) area. A summary of calculated acidity values is given in Table 2. The trend towards decreasing acidity is indicated by data for a number of the sampling wells: 4A, 19S, 27S, 26S, and 26D. Other sampling wells within the disturbed area do not indicate such a trend, 17S and 17D, and are more indicative of continuing strong acid generation.

The decrease in lime demand, as shown in Table 3, is consistent with the trend towards decreasing acidity of the acid rock drainage.

The rock matrix continues to neutralize a substantial portion of generated acidity, resulting in the "residual" acidity which is measured as part of the water chemistry analyses or which can be calculated based on pH of the sample and the metals content, particularly aluminum, iron, and manganese. Other trace metals such as copper and zinc also contribute to the acidity of the water. An assessment of "initially generated" to "residual acidity" indicates that in the order of 30 to 70% of the generated acidity is neutralized within the rock matrix. Water from the sump indicates a slightly higher degree of in situ neutralization, 50 to 70%, which may be due to the effect of surface runoff collected in the sump and entry of alkalinity from the adjacent sludge ponds. The degrees of "in situ" neutralization have not changed over the years.

In situ neutralization of acid rock drainage by reaction with calcium and magnesium silicates in the host rock has and continues to be the major factor in the residual acidity of the drainage. The significance of such neutralization reactions is indicated by the relatively high concentrations of calcium and magnesium in the drainage prior and subsequent to disposal of treatment sludge onto the mine site. Recent data, given in Table 4, are consistent with prior-to-1992 data, given in Table 5. These "weathering" reactions can be expected to continue until the acidity decreases to an "acceptable" level for direct discharge into the environment, due to the high proportion of silicate mineralization relative to the content of reactive sulphides, such as pyrite.

The decrease in acidity over the years is expected and will continue into the future. The decrease can be attributed to a decrease in exposed sulphide minerals as oxidation proceeds; the effectiveness of the sludge placement within and on the waste rock surface in reducing the rate of transfer of atmospheric oxygen to the sulphide minerals, and possible leaching of any excess alkalinity in the sludge into the groundwater zone, thereby neutralizing acidity. Passage of local groundwater through the disturbed zone would be expected to have little neutralizing effect due to the limited alkalinity of groundwater from the undisturbed areas. The water chemistry data for sampling locations from undisturbed areas (1D, 14S, 8A, 23, 24, 9D, 9S, 10D, and 10S) indicate limited alkalinity (order of 8 to 50 mg/L with a median of 20 mg/L as $CaCO_3$).

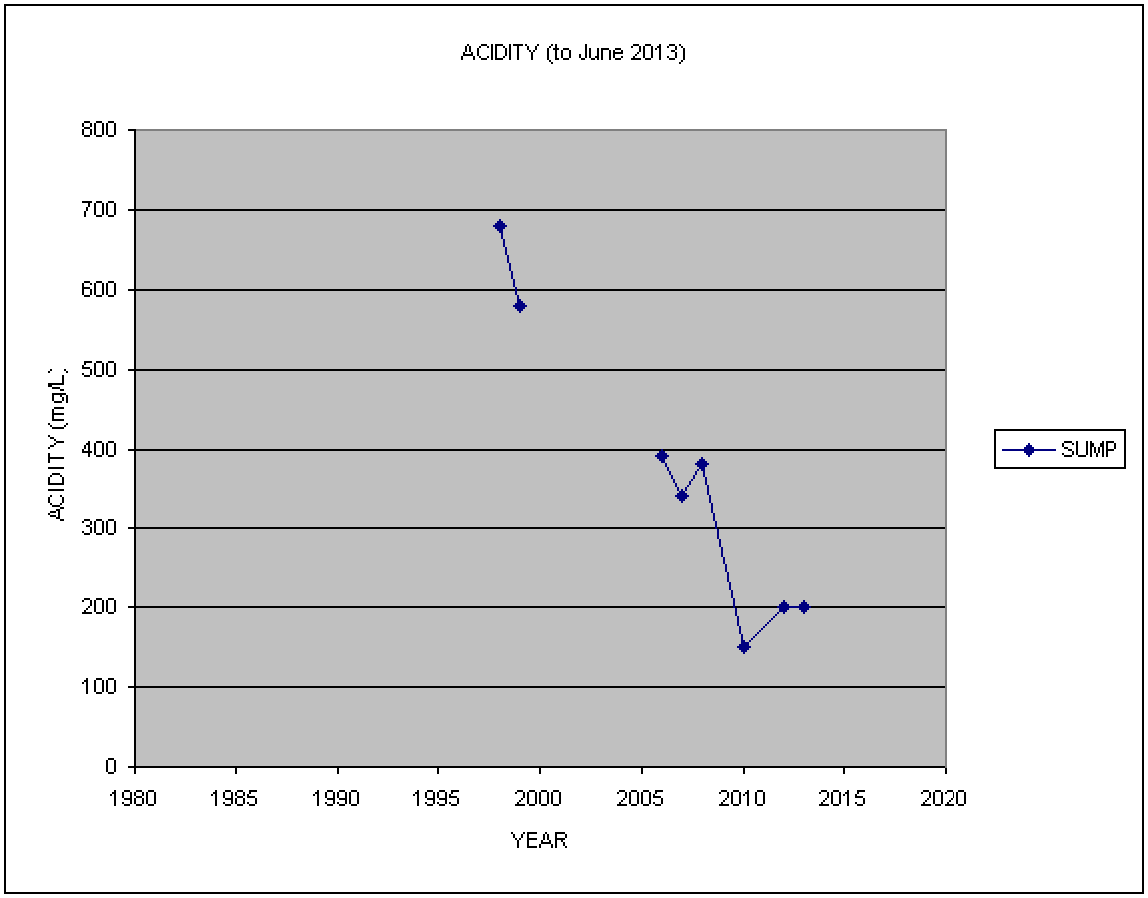
As previously noted, the key contributors to acidity include iron, aluminum, and manganese. To achieve regulatory requirements, sufficient alkalinity must be added to essentially remove iron and aluminum, which are highly insoluble at the regulatory lower pH limit of 6.5.

Table 1. Minewater Chemistry at Sump

Groundwater sampling period	June 2012	July 2013
Turbidity (NTU)	3.8	4.1
pH	4.1	4.10
Total acidity (mg/L as $CaCO_3$)	200	205
Calcium (mg/L)	178	197
Magnesium (mg/L)	28.1	29.4
Sodium (mg/L)	4.02	4.10
Potassium (mg/L)	1.26	1.34
Aluminum (mg/L)	28.2	28.0
Iron (mg/L)	6.10	6.05
Manganese (mg/L)	12.4	12.1
Copper (mg/L)	0.017	0.016
Zinc (mg/L)	0.471	0.465
Alkalinity (mg/L)	0.0	<2
Chloride (mg/L)	1.5	1.4
Sulphate (mg/L)	680	740

Table 2. Fire Road water chemistry data;
summary of acidity values (calculated)

YEAR	SUMP	4A	19S	17S	17D	27S	26S	26D
1988			391	1500	4700		3800	800
1990			217	2500	6400		2900	2900
1992						2800	4100	1700
1993		1300	750	1500	1400	2800	3500	1800
1994		1200	731	2800	4100	2200	2200	1700
1995		1300	615	2600	2700	1700	470	1300
1996		900	881	3700	1300	446	1600	1300
1998	680	800	1170	3800	1100		1100	
1999	580							
2000		600	900	5700	1200	1500		1000
2004		500	530	3200	2200	431	634	700
2006	390	480	594	2900	2600	700	440	584
2007	340	450		260	600	210	224	530
2008	380	450	516	2800	1700	145		600
2010	150	360	370	2700	1000	231	100	470
2012	200	260	260	2000	1100	250		370
2013	200	192	211	561	540	240	67	345



Data are for Sump and selected monitoring wells, 4A, etc.
Calculated acidity values are mg/L as calcium carbonate.
Data are for annual groundwater sampling program (summer period).

Table 3. Annual Hydrated Lime Consumption

Year	Water pumped	Purchased lime	Average lime demand
	m^3/a	t/a	mg/L
1992	Not recorded	2417	
1993		2113	
1994		1785	
1995		1492	
1996		1662	
1997		1009	
1998		1197	
1999		700	
2000		623	
2001		534	
2002		736	
2003		685	
2004		591	
2005		620	
2006		671	
2007	1 502 001	406	270
2008	1 863 564	628	337
2009	1 822 626	557	306
2010	1 745 537	370	212
2011	2 022 194	435	215
2012	1 600 650	275	172
2013	2 243 248	347	155

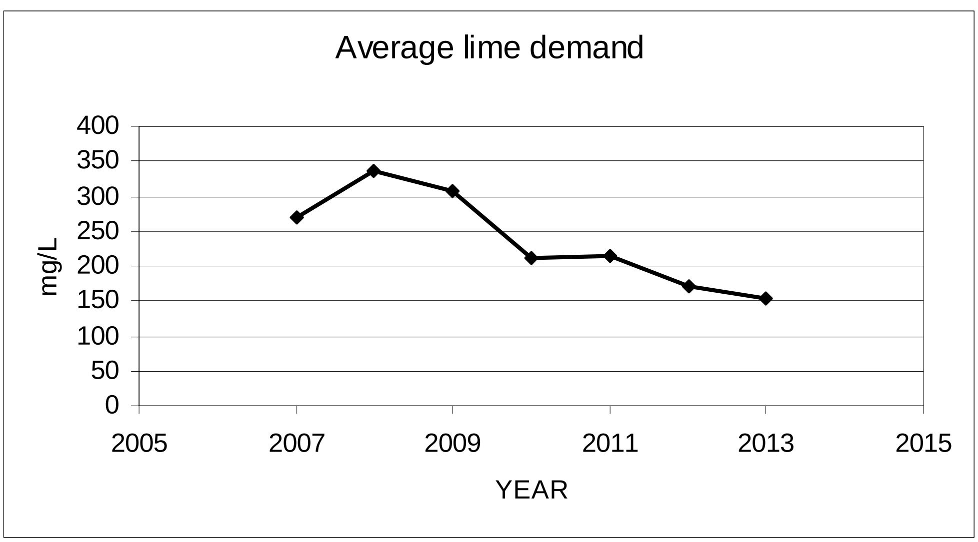


Table 4. (Ca + Mg) for Mined Area Sample Locations (2012 data)

Sample	pH	SO_4 (mg/L)	Acidity (mg/L as $CaCO_3$)	In situ Neutralization (%)	Ca + Mg (mg/L as Ca)
Sump	4.1	680	200	72	224
28S	4.1	1230	74	94	482
4A	4.1	930	260	73	279
7S	5.9	92	70	28	29
6S	5.8	122	100	23	37
19S	5.7	300	260	18	60
17D	4.3	1630	1100	38	333
17S	4.0	2800	2400	19	551
16D	7.2	800	31	96	336
16S	4.1	560	140	76	184
22S	4.8	210	60	73	64
27S	4.1	370	250	35	52
26D	3.6	1030	370	66	221

Table 5. (Ca + Mg) for Mined Area Sample Locations (Pre-1990 data)

Sample	pH	SO_4 (mg/L)	Acidity (mg/L as $CaCO_3$)	In situ Neutralization (%)	Ca + Mg (mg/L as Ca)
Sump (1986)	3.05	790	380	54	143
19S (1988)	3.00	460	280	41	79
17D (1988)	2.85	6190	4140	36	923
17S (1988)	2.85	2480	1343	48	500
16S (1988)	3.75	516	90	83	179
26D (1988)	2.75	1390	793	45	262
26S (1988)	3.00	5940	3750	39	975

3.0 STABILIZATION OPTIONS

A number of reclamation options were considered, during the years immediately following the onset of acid generation, to stabilize the site so as to allow for closure without the continuing need for treatment of acid mine drainage. The objective of stabilization methods is to minimize the rate of transfer of atmospheric oxygen into those areas of waste rock at which oxidation of mineral sulphides is occurring. This objective has been achieved to a certain degree by maintaining the highest practical ground water level within the waste rock area.

The following stabilization methods were considered:

- excavation and transfer of waste rock to ensure permanent disposal under water;
- construction of ponds over the waste rock to form a "wet" seal;
- construction of an engineered high capillary head earthen cover over the waste rock;
- construction of a simple earthen cover over the waste rock, and
- vegetation of the waste rock area.

Studies concluded that the cost for stabilization would be in the range of $3 M to $6 M, with the likely risk that treatment operations would still be required. Thus, treatment operations have continued over the years since the onset of acid generation.

TREATMENT OPERATIONS

The lime addition treatment system is effective in meeting the requirements of the Province's Approval to Operate. Hydrated lime slurry is added to water, pumped from the sump, in neutralization tanks which overflow to sedimentation ponds for separation of the aluminum/iron sludge from the water phase. Overflow from the ponds is discharged into the South Branch of East Brook which drains towards the Little River. The ponds are periodically dredged, with the sludge discharged onto the disturbed mining area north of the sump. The inert sludge assists in sealing the area and may provide some alkalinity due to leaching from the sludge.

The decreasing acidity of the mine drainage may enable consideration of passive treatment methods at some time in the future. Passive methods are attractive as they offer the potential for minimal chemical and maintenance costs into the future. At present, the substantial flow of minewater, in the order of 1.8 Mm^3/a (206 m^3/h), and the degree of total acidity (order of 200 to 400 mg/L as $CaCO_3$) would be challenging for the application of passive methods, especially where regulatory requirements must be consistently achieved.

Passive methods can be expected to become of interest as the acidity of the drainage continues to decline and, if eventually, pumping can be stopped to allow natural water levels to re-establish, resulting in one or more low-flow discharges at minimal acidity.

CLOSURE PLANS

The site is predicted to stabilize with respect to water chemistry, as monitored at the sump, within approximately the next 10 years. That is, water chemistry as monitored at the sump can be expected to approach similarity to the baseline water chemistry of undisturbed groundwater in the area. If the trend towards "zero" lime demand continues, it may be possible to discharge water from the sump without treatment.

The more reactive areas along the west highwall may remain as sources of acid generation for some period beyond the next 10 years. At present, mixing of drainage from this area with other water in the cut and contact with the backfill materials, as it flows towards the sump, results in substantial in situ neutralization. Acidity generated along areas of the west high wall will also gradually decrease towards a “zero” lime demand but this process will likely extend beyond the approximately 10 year period. Continuing acid generation in this area will be of concern if the level of water in the cut is allowed to re-establish at natural levels. Thus, to meet regulatory requirements, it may be necessary to continue maintaining the lowered water level in the sump, resulting in continued pumping beyond the 10 year period (without lime treatment). Alternatively, it may be feasible to discharge overflow(s) from the highwall area, especially if these are limited in quantity and mixed with local surface and ground waters.

Major close-out reclamation items will include restoration of the stream across the cut and filling-in/grading of the sludge ponds to the north and south of the Fire Road. The former mine area has generally re-vegetated in a very satisfactory manner and requires no further reclamation efforts.

The ongoing surface and ground water monitoring program is very valuable in following the progress at the site. This program will be continued. In addition, a periodic test program is recommended to assess the quantity and water chemistry of overflows from the west highwall area. Such tests, possibly carried out on an annual basis, would involve allowing the sump water level to rise with careful observation at the East Brook area, downstream from the highwall area. Such observations will enable the effects on water chemistry of the proposed stream restoration to be estimated.

CONCLUDING COMMENTS

The monitoring of water chemistry over the years has proven to be very valuable in understanding acid generation at the site and in planning for eventual closure with a high degree of confidence.

REFERENCES

Coleman, M.M., P.J. Whalen and A. Landva. May 30 - June 6, 1997. Investigation on the placement of lime neutralization sludge on acid generating waste rock. Proceedings of the 4th International Conference on Acid Rock Drainage: Application Technology. Vancouver.

RESTORING ACCESS TO FISH HABITAT THROUGH THE DESIGN AND CONSTRUCTION OF TWO FISHWAYS AT HUCKLEBERRY MINE

T.L.S. Poulton, R.P.Bio.

Senior Environmental Specialist,
Hatfield Consultants, North Vancouver, Canada.

ABSTRACT

Creek M is a short, high-gradient creek located on the south side of Tahtsa Reach in Ootsa Lake (Nechako Reservoir) near Houston, British Columbia. This creek was barren of fish until 1996, when Huckleberry Mines Ltd. implemented a habitat-compensation plan to offset prospective losses of fish habitat associated with mine development. The goal of this plan was to allow upstream passage of fish from Tahtsa Reach throughout the creek to a headwater pond located approximately 425 metres (m) upstream. Compensation structures within the creek consisted of three fishways comprised of log-steps (i.e., weirs) lined with geotextile and anchored with rock fill. Annual fish surveys documented the failure over time of many log-step structures, precluding fish passage. In 2012 and 2013, remedial works were undertaken to re-establish upstream fish passage for juvenile and adult rainbow trout in Creek M. New fishways were designed to be more robust, to function over a range of flows, and to meet specific requirements of migrating rainbow trout.

KEY WORDS

Fishway, fish passage, rainbow trout, habitat-compensation, migration.

INTRODUCTION

Creek M is a short, high-gradient creek located on the south side of Tahtsa Reach in Ootsa Lake (Nechako Reservoir) near Houston, B.C. Huckleberry Mines Ltd. (HML) implemented the Creek M compensation plan in 1996 to offset prospective losses of fish habitat in Creeks 2 and 4 as a result of mine development and operation. The goal of the compensation plan was to allow upstream passage of fish from Tahtsa Reach to a headwater pond located approximately 425 metres (m) upstream. Prior to the implementation of the compensation plan Creek M did not support fish populations as a result of two steep gradient reaches (10% to 20% slopes) which precluded fish passage. Compensation structures within Creek M consist of three fishways. Two of the fishways are located approximately 50 m and 300 m upstream the creek mouth while a third very small fishway is located immediately downstream the headwater pond (Figure 1). The original fishways were comprised of a series of log-steps (i.e., weirs) lined with geotextile and anchored with rock fill. Annual surveys conducted by Hatfield Consultants

Partnership (Hatfield) from 2005 to 2011 documented the failure of several of the log-step structures whereby fish passage was precluded.

Annual surveys from 2001 to 2011 documented juvenile rainbow trout (*Oncorhynchus mykiss*) throughout the creek and adult rainbow trout within the headwater pond. The presence of juvenile fish may have indicated the downstream movement of juvenile trout, resulting from pond-resident trout entering the creek to spawn in spring, rather than fish moving upstream from Tahtsa Reach into upper Creek M. Juvenile fish numbers generally declined from 2008 to 2011, with record low numbers below the middle fishway in 2011. No other fish species have been observed or captured in this creek.

The decrease in juvenile trout is likely attributable to the failure of weirs in the lower and middle fishways. The uppermost fishway (comprised of two weirs) is still functioning as intended. Migrating adult rainbow trout either from Tahatsa Reach or the headwater pond were likely precluded from preferred spawning areas between the lower and middle fishways, while juveniles became stranded in isolated pools during the late summer and winter low flow seasons. The original weir design for each step-pool included a single log weir complete with low flow notch anchored into the banks of the creek. The weirs were installed perpendicular to the flow of the creek and lined with a geotextile fabric on the upstream face. The geotextile was anchored with native substrate at a depth suitable to form a pool above each weir. Weirs that extend across the stream at right angles to the flow tend to create short pools that extend across the channel making them less stable (Slaney and Zaldokas 1997). The majority of the weirs became undermined or blown out completely while the pools between each weir in-filled with gravel and cobble substrate. As a result, the majority of flow was conveyed subsurface and the original channel gradient reestablished (Figure 2). HML proposed to remediate the failed fishways during the summer low-flow period (i.e., August) in 2012 and 2013.

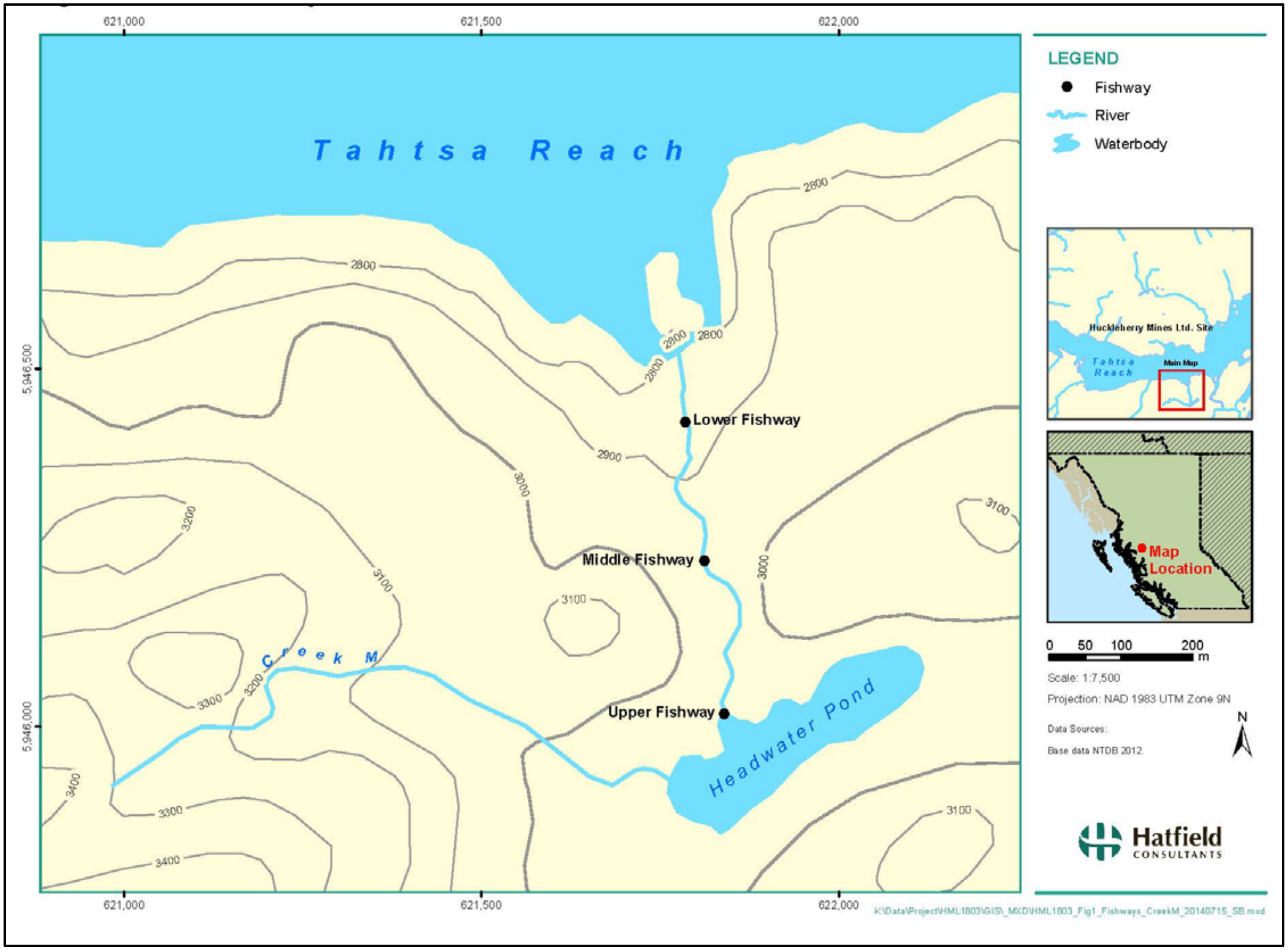


Figure 1. Creek M Fishways

Figure 2. Creek M lower fishway prior to remedial works.

FISH PASSAGE DESIGN RATIONAL

Riffles, pools and steps may be added to channels to change the local state of flow, increase access to floodplains, provide fish passage, reduce bed and bank erosion and create new habitats for fish (Newbury 2011). The objectives of the fishway remedial works in this case were to re-establish upstream fish passage for juvenile and adult rainbow trout between Tahtsa Reach and the Creek M headwater pond across a range of flows and extend the design life of the fishways. Design options were driven by the remoteness of the site, seasonal hydrology (i.e., spring freshet followed by an extended period of low flow), target species life-history and physiological capabilities, construction feasibility and budget.

Remedial works included the replacement of degrading and failed weirs (Figure 2) with new milled cedar timber and/or rock and mortar weirs. The new weirs were configured in an upstream-V orientation, reinforced with rebar and lined with a nonwoven geotextile fabric on the upstream face. The milled cedar weirs were constructed two timbers thick whereby one timber was completely buried in the substrate to prevent undermining. The geotextile was anchored to the weirs with large wood screws and washers and along the channel bottom upstream of the weir with native cobble substrate below the invert of the upstream pool. Timber weirs were substituted with rock weirs (of similar configuration) when bedrock substrate prevented anchoring of the timber and non-woven geotextile. Rock interstices were filled with mortar to increase stability and provide a seal to the channel bottom. The modified design is more stable than the existing weirs. The upstream-V configuration also promotes scour of the downstream plunge pool during both high and low flow conditions (Figure 3). The upstream V has the strength inherent in an arch design and tends to concentrate the current and subsequent scour in the middle of the channel and away from adjacent stream banks (Slaney and Zaldokas 1997). The weirs were anchored into the stream banks with rock ballast or fastened with rock anchor bolts and aircraft cable. Each weir crest was designed to slope down from the stream bank to the upstream point of the weir to confine the main flow to the middle 1/3 of the stream creating a type of low flow notch (Figure 3). These timber or log type weirs should only be used in small streams with an average channel width of less than 5 m. Larger streams with average gradients exceeding 0.4% typically have stream power or energy during floods that result in structural failure or burial of structures by excess bed load (Slaney and Zaldokas 1997).

Each weir was replaced starting from the downstream end of the fishway to ensure channel grade control and pool-jump specifications were achieved for the target life-stages of rainbow trout. For each weir the depth of the plunge pool should be a minimum of 1.25 times the height of the jump (Adams and Whyte 1990) and the height of the jump should not exceed 0.3 m to permit passage of juvenile rainbow trout (Slaney and Zaldokas 1997).

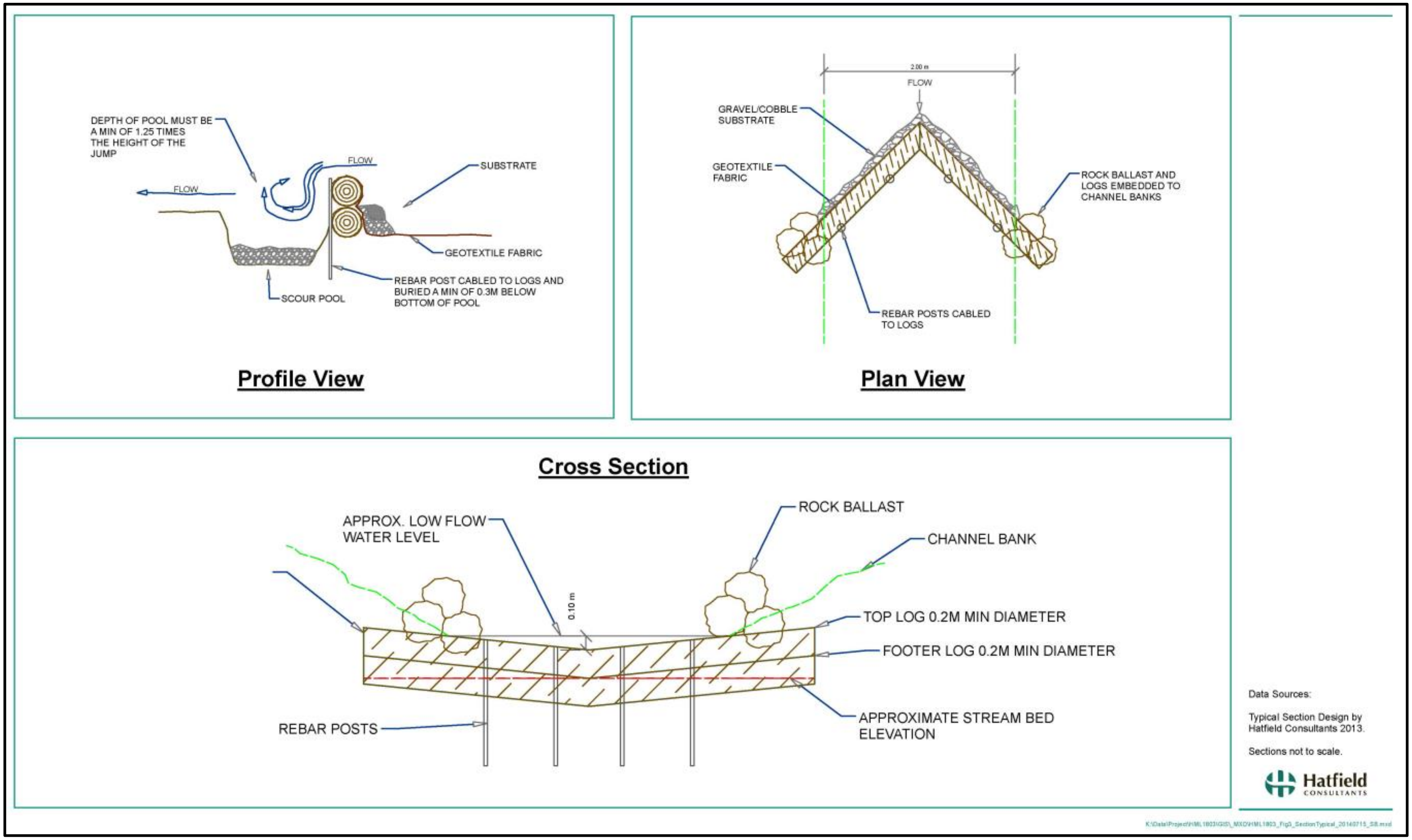


Figure 3. Typical weir sections for the Creek M fishway

CONSTRUCTION METHODOLOGY

Given the remoteness of the site (i.e., boat access only) cedar timbers and geotextile were imported via helicopter to a predetermined drop zone outside the creek riparian area; remaining construction materials were imported by boat. A fish salvage was conducted prior to the commencement of works within the work area using a backpack electrofisher. Full-span stopnets were installed at the upstream and downstream ends of the work area prior to commencing the fish salvage. The electrofishing crew fished wetted portions of the work area until fish were no longer captured. All captured fish were enumerated, weighed and measured for fork length then relocated to a pool upstream the work area. Flows were conveyed around the work area with a sandbag headwall and four-inch pipe such that the work area was isolated from flowing water. Installation of the weirs occurred from the downstream extent of each fishway and progressed upstream. The stream banks of each fishway were armored with rock salvaged from the removal of the original fishway and channel excavation (completed by hand). Large construction machinery (e.g., excavator) was not used during construction due to access restrictions.

Creek M, downstream view of lower fishway construction (August 2012).

Figure 4. Creek M fishway construction photographs

Creek M, downstream view of lower fishway construction (August 2012).

Creek M, downstream view of middle fishway (August 2013)

Figure 5. Creek M fishway construction photographs (Cont.)

.

MONITORING

Site inspections of the fishways were completed in June 2013 and 2014 to evaluate the pool-depth and jump-heights of each step-pool within the fishways during spring freshet and rainbow trout spawning season. Rainbow trout are spring spawners and typically migrate from overwintering habitat to spawning grounds in the spring when temperatures reach between 10.0 and 15.5 °C (Scott and Crossman 1979) depending of geographic location. Assessments of hydraulic function and physical stability were also assessed during the spring inspections. With the exception of three weirs in the lower fishway all pool-depth to jump-height ratios have increased from what was observed during the as-built survey and meet the design criteria for migrating adult and juvenile rainbow trout. The structural integrity of the fishways was upheld and evidence of erosion, weir undermining or pool in-filling was not observed. As part of the mines environmental effects monitoring (EEM) program fish sampling in Creek M is conducted annually in mid-August. In 2013 a total of 39 rainbow trout were captured in Creek M, all of which were young-of-year (YOY) fish. This is the largest number of YOY fish observed in Creek M since 2008. All fish were captured between the lower fishway and the middle fishway within isolated pools as a result of low water levels. It should be noted that remedial works within the middle fishway were underway at the time of the 2013 fish sampling program. A comprehensive assessment of both remediated fishways in the context of fish distribution and relative abundance will be completed in August 2014. Monitoring and remediation if required will continue for five years post-construction.

Upstream view of the lower fishway
(October 2012).

Downstream view of the lower fishway
(October 2012).

Figure 5. Photographs of the Creek M fishways in operation

Downstream view of the lower fishway (June 2014).

View of lower fishway from top of canyon bank (June 2014).

Figure 5. Photographs of the Creek M fishways in operation (Cont)

Downstream view of the middle fishway; note rock and mortar weir at the bottom of the photograph (June 2014).

Figure 5. Photographs of the Creek M fishways in operation (Cont)

ACKNOWLEGEMENTS

The author would like to thank Ron Robichaud and Marke Wong at Huckleberry Mine for their support in this project. Also the Wet'suwet'en First Nation for assisting with monitoring fishway performance.

REFERENCES

Adams, M.A. and I.W. Whyte. 1990. Fish habitat enhancement: a manual for freshwater, estuarine, and marine habitats. Department of Fisheries and Oceans Canada. DFO 4474. 330p.

Scott, W.B. and E.J. Crossman. 1973. Freshwater fishes of Canada. Bull. Fish. Res. Board Can. 184:1-966.

Newbury, R.W. 2010. Stream Restoration Hydraulics. Project Casebook. Newbury Hydraulics and the Canadian Rivers Institute. 240p.

Slaney, P.A. and D. Zaldokas. 1997. Fish Habitat Rehabilitation Procedures. Watershed Restoration Technical Circular No. 9. BC Ministry of Environment, Lands and Parks, Vancouver BC.

THE POTENTIAL EFFECTS OF ALKALINE MILL EFFLUENT ON TAILINGS MANAGEMENT FACILITY WATER QUALITY AND IMPLICATIONS FOR CLOSURE

S. Nowicki, BSc.[1]
S. Jensen, PEng[2]

[1]Staff Geochemistry Consultant,
SRK Consulting, Vancouver, Canada.

[2]Senior Environmental Engineering Consultant,
SRK Consulting, Vancouver, Canada.

ABSTRACT

Lime is commonly used as a depressant for flotation of non-ferrous ore such as molybdenum and copper. Addition of lime during the flotation process increases the alkalinity and pH of mill effluent, consequently reducing dissolved concentrations of some metals such as cadmium, copper, manganese, nickel, lead, and zinc by metal hydroxide precipitation and co-precipitation. This process is also the basis of lime treatment, which is the most common type of treatment for mine-influenced water. Bench scale tests showed that lime addition in the mill circuit can reduce concentrations of cadmium in tailings slurry and supernatant. The tests indicate that co-precipitation and adsorption interactions are critically important in this system. This is further supported by water quality data from an operating mine site. Neglecting to account for metal hydroxide formation, co-precipitation, and adsorption interactions enabled by cycling water through the mill can cause model predictions for the operations period to overestimate dissolved concentrations of some constituents in tailings management facilities (TMFs). Similarly, using operations phase TMF water quality to inform post-closure predictions could result in underestimation of dissolved concentrations since improvements in TMF water quality due to the mill process are unlikely to persist after closure.

KEY WORDS

tailings management facility, water quality, post-closure mine water quality, water treatment, acid generation, mine closure, mineral processing, flotation, tailings

INTRODUCTION

Most base metal, precious metal, coal, uranium, and diamond mines excavate large quantities of geologic materials, the drainage from which often has concentrations of metals and other elements exceeding permissible effluent standards (Price 2009). Predicting the chemistry of this drainage enables proactive mitigation measures and environmentally sound, cost-effective management of mine water. Successful mine water quality predictions must account for all relevant mine components. Tailings management facilities (TMFs) are particularly important because, in addition to storing tailings and mill effluent, they are often used to consolidate drainage from other mine components such as waste rock dumps and open pits (Figure 1). Often water from the TMF is reclaimed as process water for the mill.

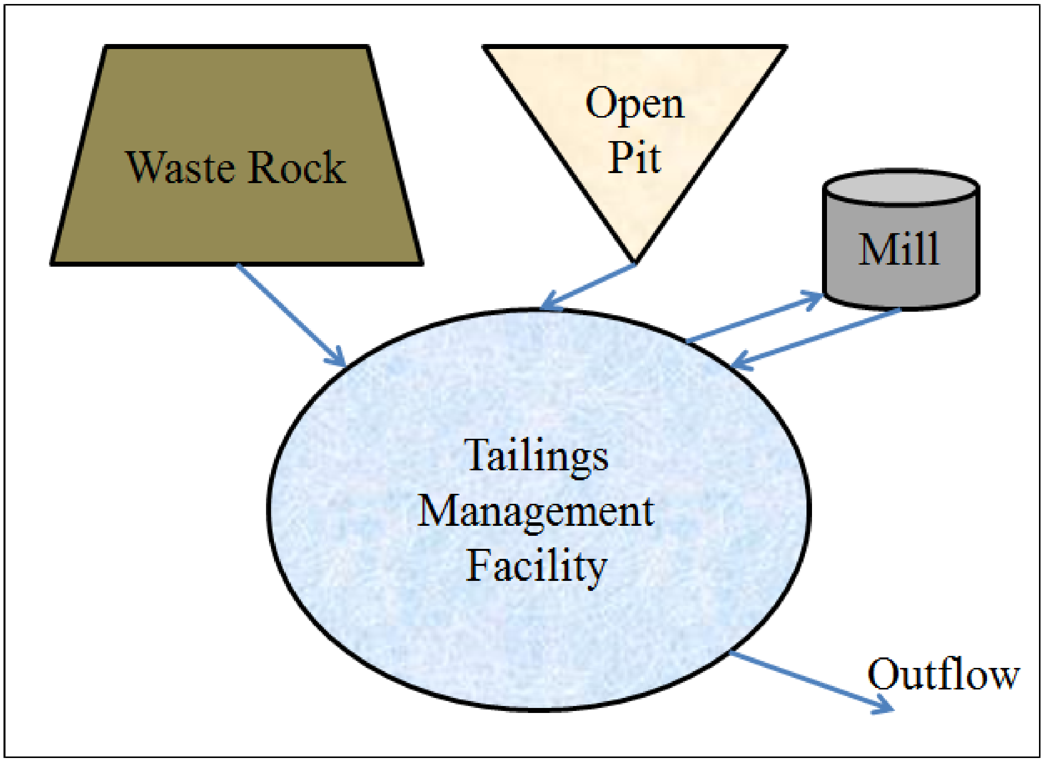


Figure 1 Typical role of a tailings management facility in relation to other mine components.

In flotation circuits for non-ferrous ores such as molybdenum and copper, lime is commonly used as a depressant (settling aid) to separate the valuable minerals from the gangue. Lime addition increases alkalinity and pH, consequently reducing concentrations of pH sensitive multivalent metals ions such as cadmium, copper, manganese, nickel, lead, and zinc through hydroxide precipitation and co-precipitation. This effect is well-known and is the basis of lime treatment, commonly used for mine-influenced water. In addition to precipitation and co-precipitation of hydroxides, adsorption onto mineral surfaces may also reduce dissolved concentrations of some constituents in mill effluent.

This paper discusses bench scale laboratory tests as well as a mine-site case study that indicate how lime addition in the mill flotation circuit substantially impacts TMF water quality. The chosen examples demonstrate the importance of considering the mill process when predicting TMF water quality. Neglecting to account for the effect of cycling TMF water through the mill can cause model predictions for the operations period to overestimate dissolved concentrations of constituents in the TMF. However, the improvement in TMF water quality is unlikely to persist after closure. Inflow to the TMF from various site components will continue and, without continuous cycling of water through the mill, operations phase water quality will not be maintained. The change in water quality will be strongly dependent on the characteristics of the inflow as well as the long term geochemical stability of constituents within the TMF.

COMPARING HIGH DENSITY SLUDGE AND MINERAL FLOTATION PROCESSES

The solubility of many metal ions that are present in acidic, mine-influenced waters (e.g. cadmium, copper, iron, nickel, lead, and zinc) is a function of pH. These ions precipitate out of solution when pH increases in various optimum ranges (US EPA 1983). Figure 2 shows how solubility of these common metals varies with changing pH. Dissolved cadmium concentrations, for example, decrease as pH increases from 8 towards 11.

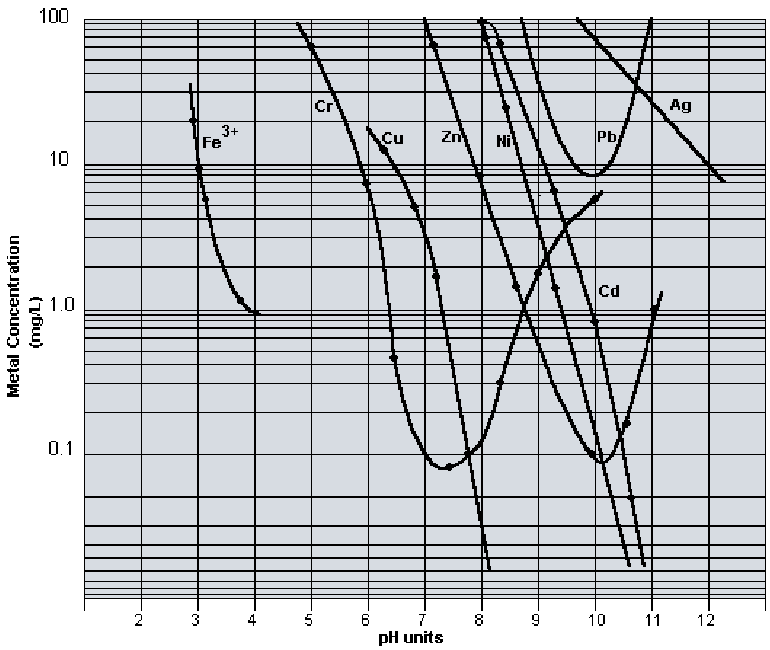


Figure 2 Solubility of common metal ions as a function of solution pH (Hoffland 2012).

These relationships between pH and solubility of various metal ions are very useful for water treatment applications. Based on consideration of effectiveness, cost, and ease of operation, the most common treatment method for mine waters is addition of lime to increase pH and concurrently decrease dissolved concentrations of a broad spectrum of metals by hydroxide precipitation (Lou et al. 1992; Tunay and Kabdasli 1994). A broad spectrum of metals can be removed because co-precipitation can result in residual metal solubilities lower than what could be achieved by precipitating each metal individually at its optimum pH (US EPA 1980).

There are five basic steps in lime addition treatment, as outlined in the EPA Design Manual for Neutralization of Acid Mine Drainage (1983):

1) Equalization – large holding basins are used to equalize flow and quality of influent to the treatment system with the objective of simplifying controls and operator attendance.

2) Neutralization – lime is mixed into the water to raise the pH.

3) Aeration – ferrous iron (Fe^{2+}) is oxidized to ferric iron (Fe^{3+}), which is the less soluble form. Minimum solubility for Fe^{2+} is between pH 9 and 12 whereas Fe^{3+} begins to precipitate as iron hydroxide around pH 4 with minimum solubility around pH 8. The key reactions are:

$$Fe^{2+} + \frac{1}{4}O_2 + H^+ \leftrightarrow Fe^{3+} + \frac{1}{2}H_2O$$
$$Fe^{3+} + 3H_2O \leftrightarrow Fe(OH)_3 + 3H^+$$

4) Sedimentation – holding basins are used to settle out precipitated metal hydroxides and other suspended particles.

5) Sludge recirculation and disposal – a proportion of the treatment sludge is recirculated and the remainder is disposed of in various different ways depending on the site.

The recirculation of sludge from Step 5 is a variation on conventional lime treatment that was developed in 1970 and is referred to as the High Density Sludge (HDS) process. Recirculating the sludge achieves better reactivity of lime and produces smaller volumes of sludge with higher solid metal hydroxide content. This is because the recirculated solids act as nucleation sites for precipitation reactions. Continuous recirculation creates increasingly large metal hydroxide particles. The ideal ratio of sludge recirculated to disposed is between 20:1 and 30:1 (US EPA 1983). The recirculated sludge is returned to the reactor where the lime is added (Figure 3).

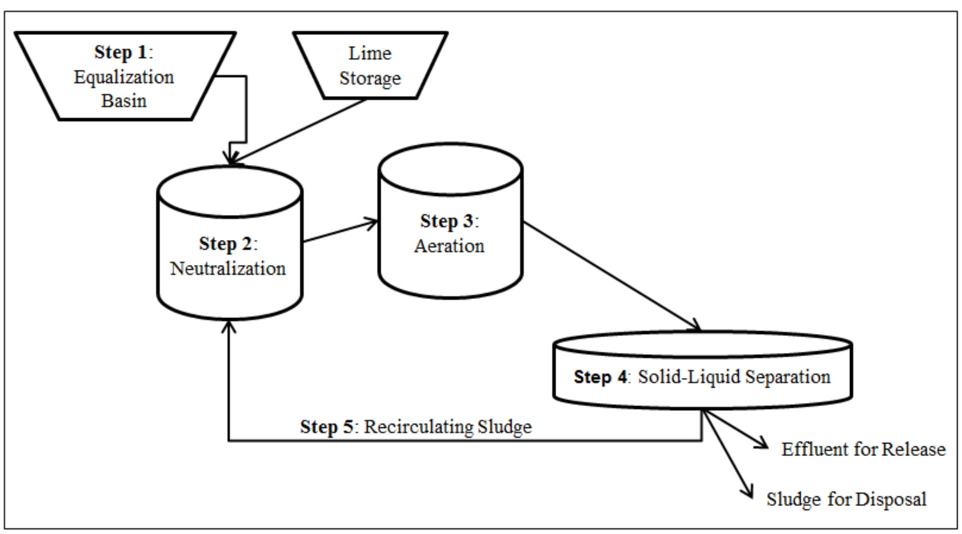


Figure 3 High Density Sludge (HDS) lime addition treatment process.

There are several functional similarities between the HDS process and the mineral flotation process that is used to extract valuable non-ferrous minerals from ore. Lime can be added to mill process water as a modifier. It has the obvious effect of increasing pH and can be useful for dispersion and depression of certain gangue minerals including pyrite. Of course, as in the HDS process, addition of lime in the flotation circuit also causes precipitation of multivalent metal ions as hydroxides (SME 2007).

The third step of the HDS process, aeration, also occurs in the mineral flotation process – the ore slurry is vigorously agitated after lime addition when pH is elevated. The primary solids separation step in both processes is gravity settling. Although sludge is not recirculated in the same way in the mineral flotation process, the mineral solids present in the ore slurry may act as nucleation sites for metal precipitation; essentially serving the same purpose as the recirculated sludge in the HDS process.

BENCH SCALE TESTING

Bench scale laboratory tests from the Kitsault molybdenum project in British Columbia provide an example of how lime addition in the mill flotation circuit can substantially impact mill effluent water quality. The tests were conducted on tailings slurry and supernatant samples obtained from Kitsault's metallurgical pilot plant. The purpose of the testing was to establish (1) how effectively lime addition in the mill flotation circuit removes dissolved metals, and (2) whether use of sulphide reagent to further reduce dissolved concentrations by precipitating metal sulphides is necessary as an additional treatment step to meet effluent water quality standards. The focus of the bench-scale testing was dissolved cadmium concentrations, which are required to be very low in order to meet water quality standards downstream of the TMF.

<u>Testing Methodology</u>

The testing program consisted of four sequential steps with an aliquot sampled at each step to follow the behavior of cadmium:

1) A total of 17 samples (7 slurry and 10 supernatant) were spiked with approximately 2 μg/L of dissolved cadmium. After approximately 2 hours, measurements of cadmium concentration in each sample were taken.

2) Hydrated lime ($Ca[OH]_2$) was added to a selection of samples to reach target pHs of 9.5 or 10.5, and sodium hydrosulphide (NaHS) was added to a selection of samples (as detailed in Table 1). The samples were left to settle for approximately 2 hours and a measurement of cadmium concentration in the overlying water was taken.

3) All of the samples were passed through a 0.45 μm filter (the cut-off size for dissolved metals), which targets cadmium hydroxide ($Cd[OH]_2$) particles, and a measurement of cadmium concentration in the filtered water was taken.

4) Finally, all of the samples were passed through a 0.1 μm filter (captures some colloidal-sized solids), which targets cadmium sulphide (CdS) particles, and a measurement of cadmium concentration in the filtered water was taken.

The pH of each sample was measured before and after cadmium was added, and after hydrated lime and/or sodium hydrosulphide was added (Table 1).

Table 1 pH Measurements from Kitsault In-Plant Water Treatment Testing.

Test No.	Feed Type	Cadmium Addition		Hydrated Lime Addition			Sodium Hydrosulphide Addition	
		pH before	pH after	Target pH	$Ca(OH)_2$ Added (mL)	pH after	NaHS Added (mg/L)	pH after NaHS / Final pH
1	Slurry	6.8	7.0	-	None	7.1	0	7.1
2		8.1	8.0	9.5	29.9	9.6	0	9.6
3		8.0	8.0	10.5	40.0	10.7	0	10.7
4		8.0	8.0	-	None	8.0	5	8.0
5		8.0	8.0	9.5	15.0	9.5	5	9.4
6		7.9	7.9	-	None	7.9	5	8.0
7		8.1	8.1	9.5	22.5	9.4	10	9.4
8	Supernatant	7.3	7.4	-	None	7.4	0	7.4
9		8.2	8.2	9.5	6.8	9.5	0	9.5
10		8.1	8.1	10.5	17.6	10.5	0	10.5
11		8.3	8.3	9.5	7.1	9.5	5	9.5
12		8.2	8.2	9.5	7.6	9.5	5	9.5
13		8.3	8.3	9.5	11.0	9.5	5	9.5
14		8.2	8.2	9.5	7.1	9.5	10	9.3
15		8.1	8.1	-	None	8.1	5	8.2
16		8.2	8.2	-	None	8.2	10	8.4
17		8.3	8.3	9.5	8.9	9.5	10	9.5

Cadmium Hydroxide Formation Results

Generally, in oxic freshwater systems with circumneutral pH, cadmium occurs predominantly as dissolved cadmium (Cd^{2+}) cations. At high pH, cadmium hydroxide forms and cadmium cation concentrations decrease (McGreer et al. 2012). The simple precipitation reaction for cadmium hydroxide is:

$$Cd^{2+} + 2OH^{-} \leftrightarrow Cd(OH)_{2\,(s)}$$

and the minimum solubility of $Cd(OH)_2$ occurs around pH 10.5 (Patterson et al. 1977; Luo et al. 1992). After passing through the 0.45 μm filter, the two samples with the highest final pHs (sample Slurry 3 had pH 10.7 and sample Supernatant 10 had pH 10.5) had the lowest concentrations of cadmium (both 0.03 μg/L) (Figure 4). The other 15 samples did not reach pHs above 10, but cadmium concentrations decreased in most of them between step 1 and step 3.

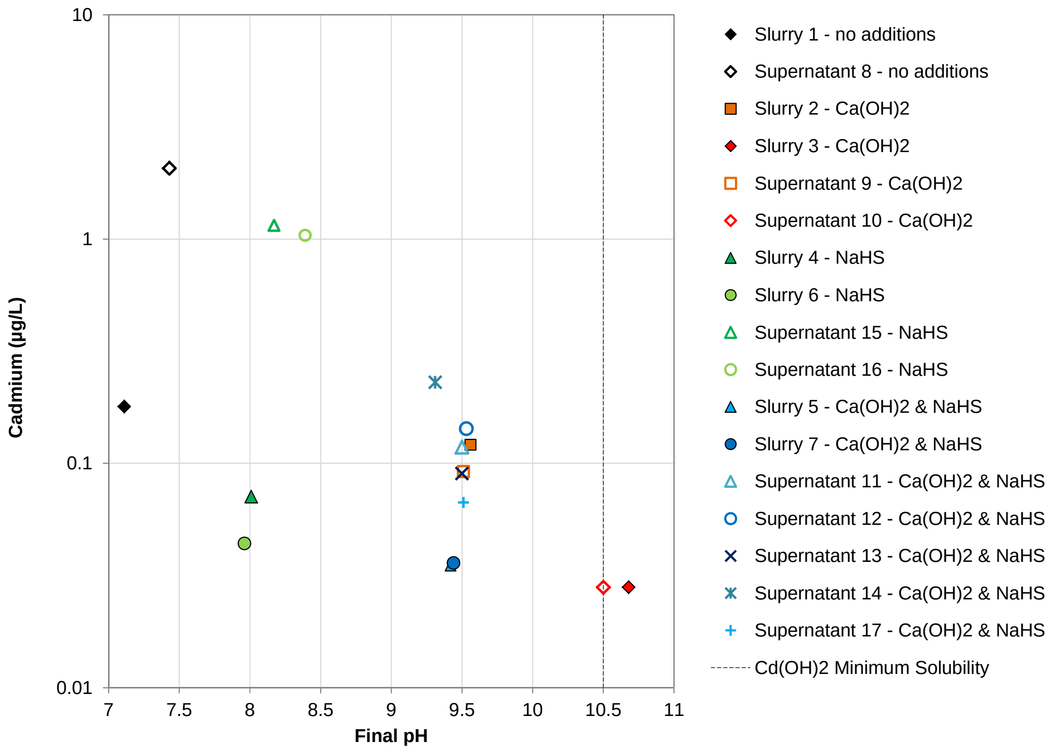


Figure 4 Cadmium concentrations after 0.45 µm filtration (Step 3) versus final pH.

As shown in Figure 5, there were two main differences between the slurry and supernatant samples:

1) The majority of dissolved cadmium in the slurry samples was removed from solution in step 1, *prior* to addition of lime and / or sodium hydrosulphide. The pH values of the samples were between 7.0 and 8.1 and the cadmium concentrations ranged from 0.09 to 0.17 µg/L. These values are low compared to the supernatant samples which ranged from 1.8 to 3.5 µg/L. Dissolved cadmium concentrations are known to decrease as a result of co-precipitation and / or adsorption onto mineral surfaces (Hem 1985; McGreer et al. 2012), which are readily available in tailings slurry.

2) For the supernatant, the final cadmium concentrations in the samples that had lime added to them were approximately an order of magnitude lower than in the samples that did not. For the slurry samples, lime addition had less of an effect on cadmium concentrations. This may be due in part to the slurry samples having notably lower cadmium concentrations than the supernatant samples after step 1. Dissolved cadmium did decrease in three of the four slurry samples in steps 2 and 3 (Figure 5). The exception was sample 2, which had higher concentrations after step 3 than after step 1 – likely due to experimental error or analytical inaccuracy, which is common when concentrations are close (within a factor of 10) to the analytical detection limit (in this case 0.01 µg/L).

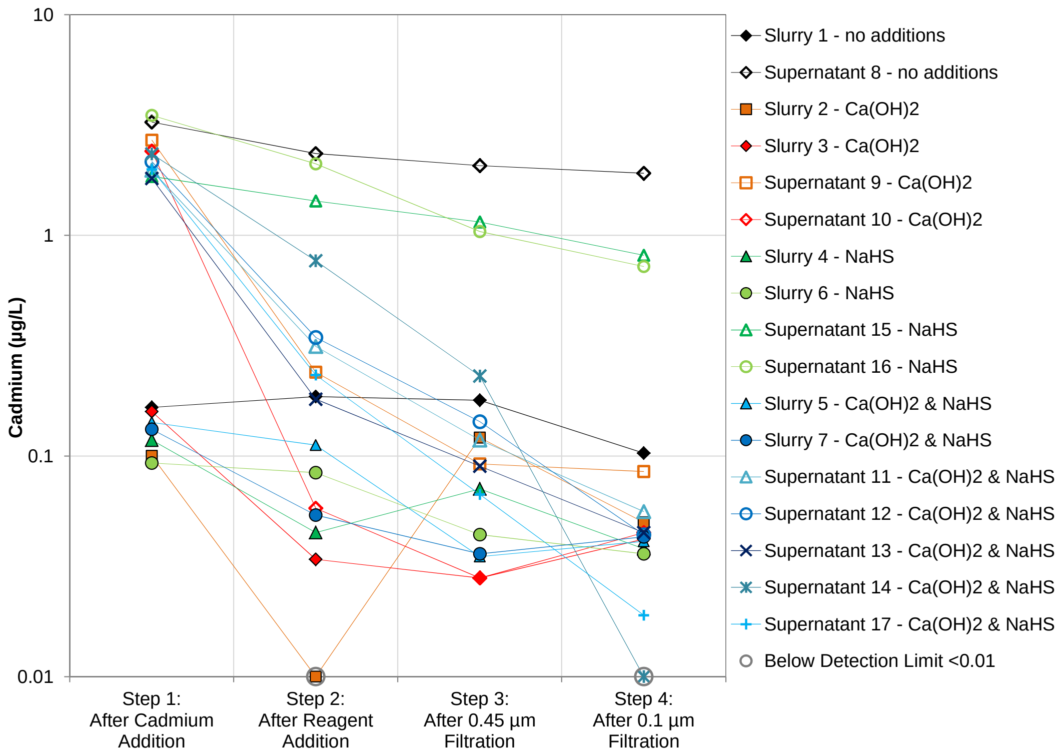


Figure 5 Line-plot showing slurry and supernatant sample concentrations throughout testing.

Cadmium Sulphide Formation

In addition to hydrated lime, sodium hydrosulphide (NaHS) was added to some of the samples in an attempt to precipitate cadmium as cadmium sulphide (CdS), which is many orders of magnitude less soluble than cadmium hydroxide ($Cd[OH]_2$) over a broad range of pH. Adding sulphide reagent re-solubilizes metal hydroxides by reducing dissolved metal concentrations below the equilibrium level predicted by hydroxide solubility, essentially converting metal hydroxides to metal sulphides. Metal sulphides generally form by discrete precipitation (rather than nucleation – precipitating onto already existing particles) as small particle fines and hydrated colloidal particles (US EPA 1980). Consequently, cadmium sulphides are more effectively removed by a 0.1 µm filter as opposed to a 0.45 µm filter.

For the supernatant samples that had sodium hydrosulphide added, cadmium concentrations were lower after 0.1 µm filtration, indicating that sulphide precipitation contributed substantially to removal of cadmium from solution. In the slurry samples, however, the effect of sulphide addition on dissolved cadmium concentrations was less clear. In the two slurry samples that had lime and sodium hydrosulphide added, cadmium actually increased after the sample was passed through the finer filter – likely because of analytical inaccuracy. In the two slurry samples that had only sodium hydrosulphide added, cadmium concentrations decreased overall between steps 1 and 4 but once again displayed variability indicative of analytical inaccuracy.

MINE SITE CASE STUDY

This case study uses water quality data from a copper mine to exemplify the effect of cycling TMF water through the mill. The primary mineral in the ore that is processed at the mine is chalcopyrite ($CuFeS_2$). The ore is crushed and ground before it enters the mill flotation circuit. After the first round of flotation, copper concentrate is reground and floated again to produce a final copper concentrate. All tailings and tailings supernatant are discharged into a TMF. Water from the TMF is reclaimed as process water for the mill or discharged to the environment provided that it meets discharge criteria.

The main constituent of concern for water quality at this site is copper, which occurs predominantly as cupric cations (Cu^{2+}) in oxygenated waters. At above neutral pH in waters that contain dissolved carbon dioxide (CO_2) species, formation of copper hydroxides ($Cu[OH]_2$) and copper carbonates ($CuCO_3$) can decrease concentrations of Cu^{2+} to around 0.01 mg/L or less. Additionally, dissolved copper concentrations can be further reduced by co-precipitation and / or adsorption interactions (Hem 1985).

On the site, approximately 2.0 million m^3 of water was allowed to accumulate in an open pit. Dissolved copper concentration in the pit water were generally above 0.01 mg/L and as high as 0.05 mg/L. In just over 4 months the accumulated water was pumped into the TMF, which contained approximately 1.8 million m^3 of free water at the beginning of the 4 month period. Historically, copper concentrations in the TMF were generally between 0.001 and 0.01 mg/L.

A simple mixing model was developed to calculate the concentrations of dissolved constituents in the TMF after mixing with water from the open pit. The model assumed that the inflowing pit water was perfectly mixed with the existing free water in the TMF. Loss of free water and associated loads from the TMF, both to the underlying tailings and to the receiving environment, was taken into account. Figure 6 shows the results of the mixing model for dissolved copper and strontium along with actual measured concentrations in the TMF.

Predicted strontium concentrations were similar to actual measured strontium concentrations. Strontium was chosen because it is a conservative constituent that isn't altered in the mill flotation circuit. It is generally unreactive and stable in dissolved form (Sr^{2+}) when it occurs at low concentrations (below 10 mg/L) as it does in both the pit and TMF water (HEM 1985). At the start of the pumping period, strontium concentrations were just below 3 mg/L in the pit water and almost 5 mg/L in the TMF, accordingly the model predicted that over the pumping period concentrations in the TMF would gradually decrease to around 3.7 mg/L. The modelled concentrations closely approximated the actual concentrations measured in the TMF during the entire pumping period.

Dissolved copper concentrations were modelled in the same way that strontium was, but the actual copper concentrations measured in the TMF during the pumping period were substantially lower than what the model predicted (Figure 6). The model predicted that concentrations would increase to just around 0.012 mg/L by the end of the 4 month pumping period, but actual concentrations remained below 0.002 mg/L.

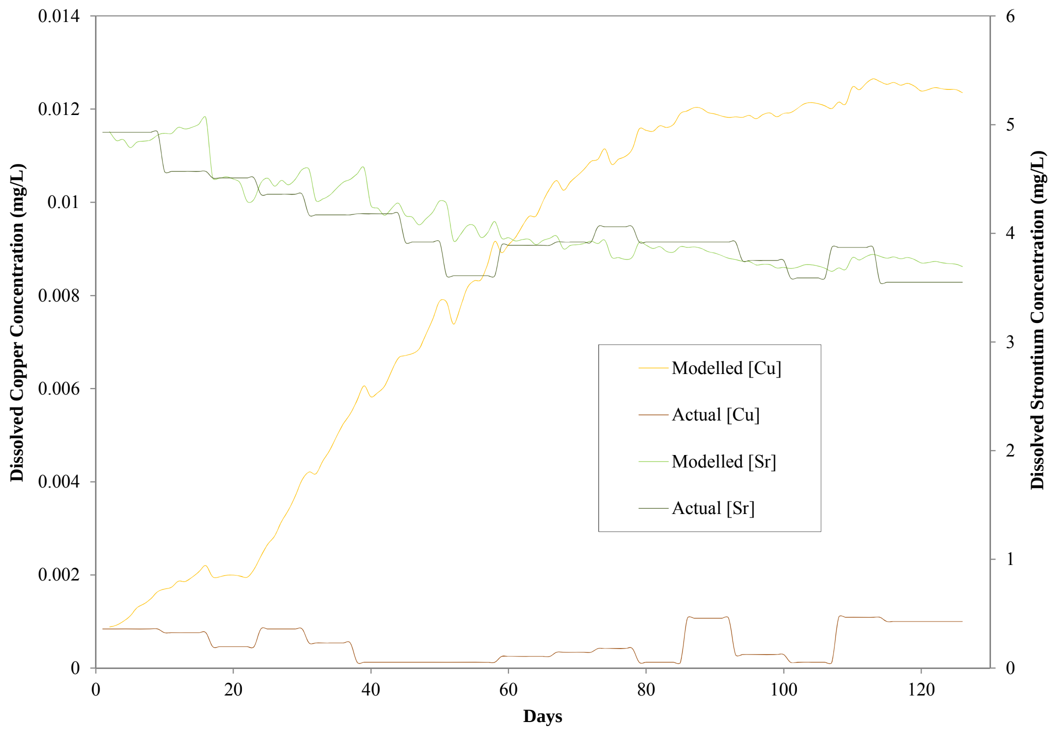


Figure 6 Modelled and actual dissolved copper and strontium concentrations in the TMF.

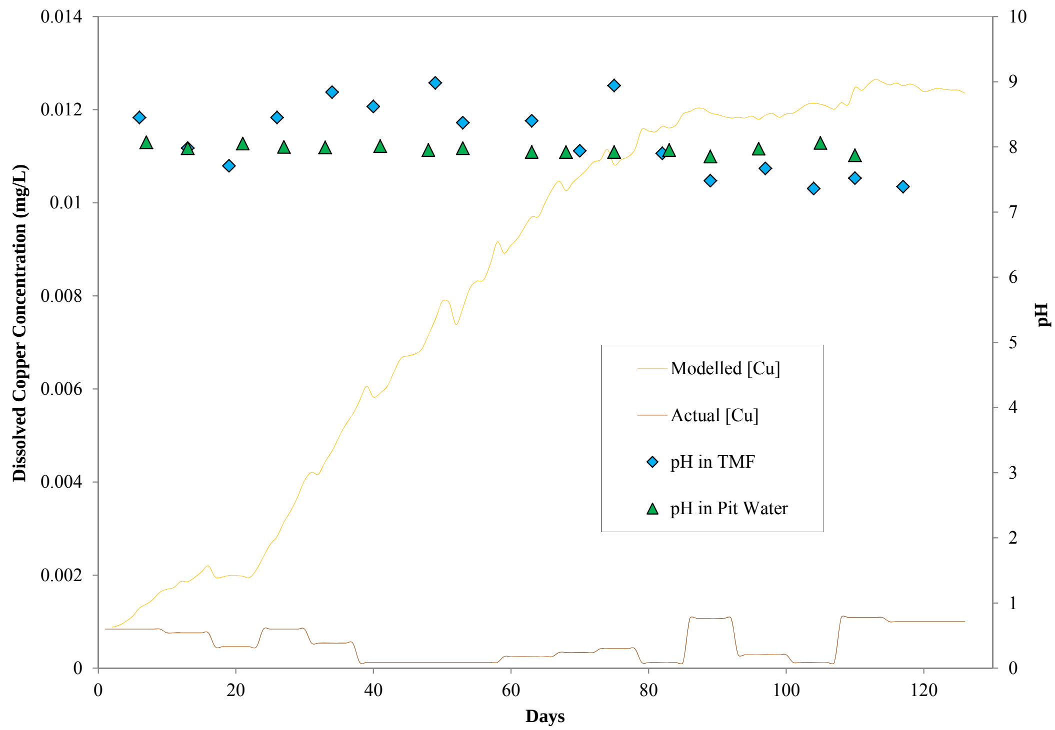


Figure 8 pH levels in the TMF and pit water.

When the pit water mixed with the water in the TMF, dissolved copper was removed from solution. This was not a result of simple pH-controlled copper hydroxide or copper carbonate precipitation. The optimal pH for precipitating copper out of solution is around 8 (Hoffland 2012). In the pit water, pH levels were consistently between 7.5 and 8, and the difference in pH between the pit and TMF was not substantial enough to explain the persistence of low copper concentrations in the TMF (Figure 8).

As was the case for dissolved cadmium in the slurry samples from the Kitsault bench scale testing, a likely explanation for the persistence of low copper in the TMF is co-precipitation and/or adsorption onto mineral surfaces within the tailings slurry. This process is made possible by cycling the TMF water through the mill. Additionally, the solids in the tailings slurry may have facilitated precipitation of copper and other metal hydroxides by acting as nucleation sites, as is the case with recycled sludge in HDS treatment processes.

Copper and strontium were chosen to demonstrate two different effects that a milling process may have on reclaim water – *i.e.* near-complete removal and "no effect". The effect is not as clear for other dissolved constituents. There are other factors that affect constituent loadings and concentrations in TMF water such as ore characteristics, seasonal conditions, and operational/process changes on the mine site.

CONCLUSION

The Kitsault bench scale laboratory tests and the mine site case study discussed in this paper demonstrate that lime addition in mill flotation circuits reduces dissolved concentrations of multivalent cations, like cadmium and copper, by hydroxide precipitation and co-precipitation. It was also shown that the solids within the tailings slurry have an important role, acting as nucleation sites for precipitation and providing surfaces for adsorption.

Caution should be applied when using operational water quality data to infer post-closure water quality in a TMF. Improvement in water quality that may result from precipitation, co-precipitation, and adsorption processes facilitated by cycling TMF water through the mill, may not persist after closure even if the pH regime is not expected to change. The change in water quality will be strongly dependent on the characteristics of the inflow as well as the long term geochemical stability of constituents within the TMF. Conversely, water quality predictions for the operations phase of a mine that do not account for the effect of mill effluent may be overly conservative.

ACKNOWLEGEMENTS

The authors wish to thank Avanti Mining Ltd. for their approval to describe the bench scale test work that was completed for their Kitsault operation.

REFERENCES

Hem, J.D. 1985. Study and Interpretation of Chemical Characteristics of Natural Water, Third Edition. U.S Geological Survey Water-Supply Paper 2254.

Hoffland Environmental Inc. 2012. Hydroxide Precipitation. http://www.hofflandenv.com/hydroxide-precipitation/

Luo, B., Patterson, J.W., & Anderson, P.R. 1992. Kinetics of cadmium hydroxide precipitation. Water Research, 26: 6, 745-751.

McGreer, J.C., Niyogi, S., & Smith, D.S. 2012. Cadmium. Chapter 3 of Homeostasis and Toxicology of Non-essential Metals, Volume 31, Part 2. Eds. Wood, C.M., Farrell, A.P., & Brauner, C.J.

Price, W.A. 2009. Prediction Manual for Drainage Chemistry from Sulphidic Geologic Materials, MEND Report 1.20.1. Prepared for Mine Environmental Neutral Drainage Program.

Patterson, J.W., Allen, H.E., & Scala, J.J. 1977. Carbonate precipitation for heavy metal pollutants. Research Journal of the Water Pollution Control Federation, 49: 12, 2397-2410.

Society for Mining, Metallurgy, and Exploration, Inc. (SME). 2007. Froth flotation: a century of innovation. Eds. Fuerstenau, M.C., Jameson, G.J., & Yoon, R-H.

Tünay, O. & Kabdaşli, N.I. 1994. Hydroxide precipitation of complexed metals. Water Research, 28: 10, 2117-2124.

United States Environmental Protection Agency. 1980. Control and Treatment Technology for the Metal Finishing Industry, Sulfide Precipitation. EPA 625/8-80-003.

United States Environmental Protection Agency. 1983. Design Manual: Neutralization of Acid Mine Drainage. EPA 600/2-83-001.

LEONARDITE AND BIOCHAR FOR MINE IMPACTED WATER AND SOILS

K.J Stewart, PhD.[1]
A. Janin, PhD.[2]

[1]Research Associate,
[2]Industrial Research Chair,
Yukon Research Centre, Yukon College,Whitehorse, YT

ABSTRACT

Immobilization of metals using soil amendment processes is increasingly being considered as an effective and low cost remediation alternative in the mining industry. Both leonardite, a carbon-rich material rich in humic acids and biochar, an organic material that has undergone pyrolysis, have shown to adsorb heavy metals, such as Cd and Zn and promote plant growth. We examined the potential to use leonardite and biochar for metal sequestration in mining impacted water and soils, by determining their capacity to adsorb metals in water, sequester metals in tailings and promote plant growth. Biochar removed up to 95% Cd and 90% Zn from synthetic water and resulted in a 74% reduction of Cd and 18% of Zn leached from columns containing tailings. Whereas, leonardite only adsorbed 38% Cd and 29% Zn from synthetic water and resulted in column leachate with higher concentrations of heavy metals. Leonardite amendments caused decreases in pH and mobilization of metals from tailings may be due to acidification. Above and belowground growth of 2 different northern native herb species (*Lupinus arcticus* and *Hedysarum alpinum*) in amended tailings were examined. Amendments had little influence on growth with only the leonardite and lime treatment showing increased belowground biomass. This initial trial demonstrates that both amendments show potential for on-going management of contaminated waters and tailings, however, additional liming agents are likely necessary with leonardite.

KEYWORDS

Leonardite, biochar, remediation, heavy metals, tailings, sequestration

INTRODUTION

In situ immobilization of metals using soil amendment processes is increasingly being considered as an effective and low cost remediation alternative (Mench et al. 2007, Kumpiene et al. 2008, Fellet et al. 2011). Leonardite is a carbon-rich material derived from the oxidation of Lignite and is rich in humic acid whereas biochar is a product that results from the oxygen limited, pyrolysis of various biological ingredients, such as wood, fish or animal bone. Several studies have found biochar amendments result in significant decreases in the bioavailability of heavy metals associated with mining impacted soils (Namgay et al. 2006, Fellet et al. 2011, Beesley et al. 2010) and simultaneously improve physical, chemical and biological soil properties (Laird et al. 2010). Leonardite is known to improve soil conditions (Lao et al. 2005, Zeledón- Torunõ et al. 2005, Madejón et al. 2010) and has potential to significantly reduce metal bioavailability due to high metal adsorption capacity (Lao et al. 2005, Zeledón-Torunõ et al. 2005).

The study of solubility and bioavailability of metals in contaminated soils or water is important in remediation activities because they represent the most labile fractions subject to leaching and to being uptaken by plants and microorganisms (Adriano 2001). Both leonardite and biochar have shown to adsorb heavy metals, such as Zn, Pb and Cd from contaminated waters (Lao et al. 2005, Chen et al. 2011, Zeledón- Torunõ et al. 2005, Kolodynska et al. 2012, Regmi et al. 2012). The removal of metals from both water and soils is highly pH dependent (Zeledón- Torunõ et al. 2005). While metal precipitation requires alkaline pH, metal adsorption is less pH dependent. Alteration of the pH by liming is a frequent remediation practice for trace element polluted systems (Adriano 2001, Madejón et al. 2009) however the effects of liming gradually reduce over time due to the dissolution and leaching of the liming agent (Ruttens et al. 2010). Biochars and leonardite are both highly recalcitrant and their effects may persist over long time periods (Steiner et al. 2007).

Phytostabilization of mine tailings is highly difficult, not only due to phytotoxic effects of elevated heavy metal concentrations, but also due to extreme pH values, low fertility, low water-holding capacity and unfavorable substrate structure (Fellet et al. 2011). In highly degraded soils amendments can not only assist in trace element stabilization, but may also directly impact plant cover establishment and the long-term improvement of soil quality (Pérez-de-Mora et al. 2005, Pérez-de-Mora et al. 2006). Both leonardite and biochar are known to promote seed germination and viability, increase plant biomass and the rate of root development (Chen and Aviad 1990, Nardia et al. 2002, Jones et al. 2012). In addition, these materials can help reduce metals toxicity and revegetation contaminated areas (Pérez-de-Mora et al. 2007). We examined the potential to use leonardite and biochar for metal sequestration in mining impacted water and soils, by determining the capacity of leonardite and biochar to adsorb metals in water, sequester metals in tailings and promote plant growth.

METHODS

Metal adsorption capacity

Leonardite and biochar ranging from 10g/L to 50g/L were exposed for 24 hours to synthetic contaminated water containing either Cd or Zn and pH was adjusted at 7. Water metal concentrations before and after exposure to the treatments were measured and compared. The data was used to determine the adsorption capacity of each treatment.

Metal sequestration property

Tailings contaminated with Cd and Zn from Keno Hill Mining District (KHMD) were mixed with 4 treatments: 1) Lenoardite (6%v/v), 2) Biochar (6% v/v), 3) Leonardite and Biochar (each at 6% v/v) and 4) Leonardite (6%v/v) and dolomite lime (54.6% $CaCO_3$, 41.5% $MgCO_3$ at 484 g/m2). The mixture was contained in PVC pipes (4 columns + 1 unamended control). Water was passed through the mixture continuously at a low flow rate for two months to simulate water infiltration through mine tailings. Data was compared to assess the capacity of the biochar and leonardite to sequester metals leached from tailings.

Plant growth

Tailings contaminated with Cd and Zn from KHMD were mixed with the same 4 treatments as above. All treatments were also fertilized with a 19:19:19 fertilizer at a rate of 110 kg/ha. There were 10 replicates (i.e. individual containers with two seedlings) for each treatment. The and above and belowground growth of two different northern native herb species (i.e. Arctic Lupin (*Lupinus arcticus*) and Alpine Sweetvetch *(Hedysarum alpinum*)) were examined across each of the 4 treatments over a two month greenhouse experiment. The greenhouse conditions and watering (6 ml DI per replicate every second day) were controlled to reflect typical summer growing conditions in the Keno area.

RESULTS

Both biochar and leonardite showed metal adsorption capacity. Up to 95% of Cd initially present in solution is removed by biochar after 24 hours of contact versus 38% by leonardite. Adsorption of Zn showed a similar trend with up to 90% of Zn removed by biochar after 24 hours of contact versus 29% by leonardite. With 10 g/L of leonardite or biochar, Cd loading was 49.2 mg Cd/g leonardite and 110.6 mg Cd/g biochar (Figure 1). Therefore, biochar was able to adsorb more than 10% of its mass in Cd.

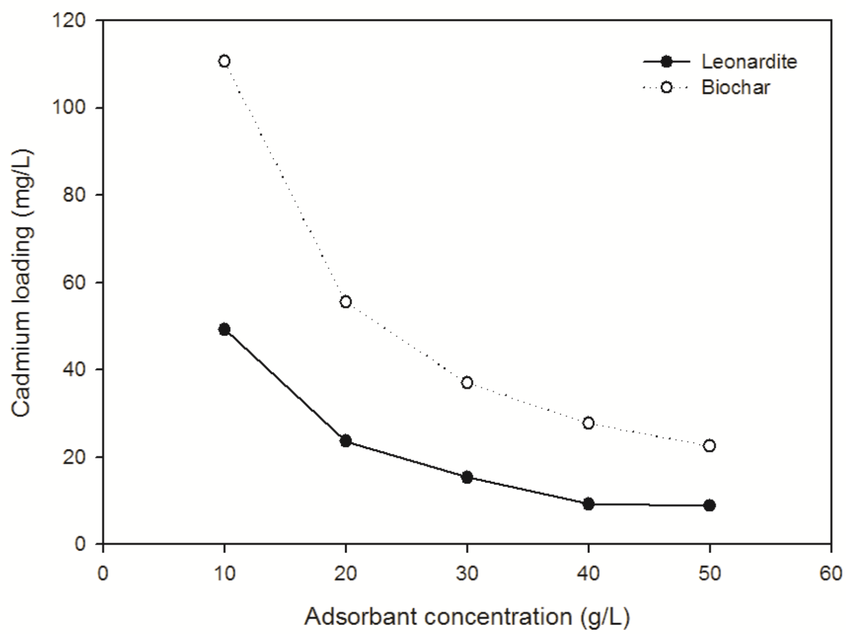


Figure 1. Cadmium loading on leonardite and biochar after 24 hrs with leonardite and biochar concentrations ranging from 10-50 g/L.

With 10 g/L of leonardite or biochar, Zn loading was up to 19.6 g Zn/g leonardite and 51.9 g Zn/g biochar (Figure 2). We did not observe any increase in metal removal by increasing the amount of leonardite or biochar above 10g/L (Figures 1 and 2).

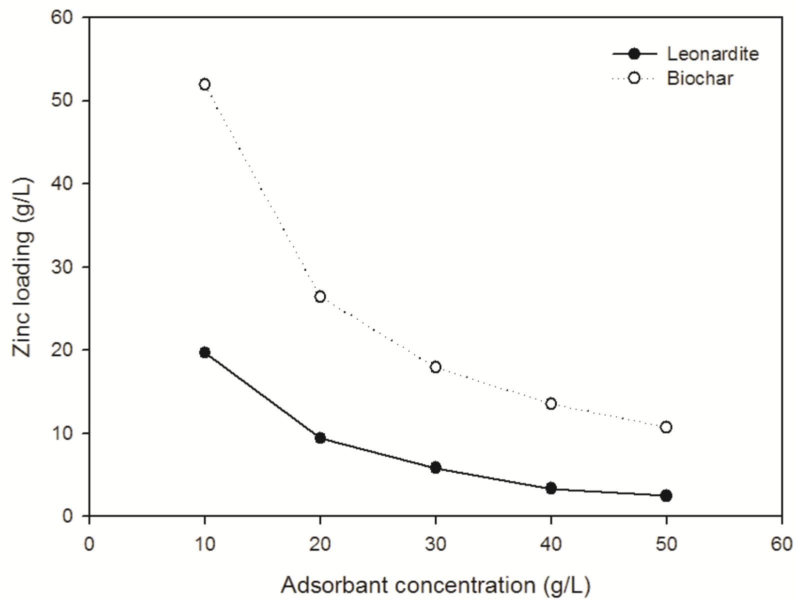


Figure 2. Zinc loading on leonardite and biochar after 24 hrs with leonardite and biochar concentrations ranging from 10-50 g/L.

Solutions with an initial pH of 7 containing leonardite and biochar ranging in a solid to liquid ratio of 10-50% were held with 24 hours of contact between the water and each adsorbent. Both leonardite and biochar lowered the pH, however, leonardite lowered the pH to 2.9-3.4, while biochar lowered the pH less to 5.6-6.4 (Figure 3).

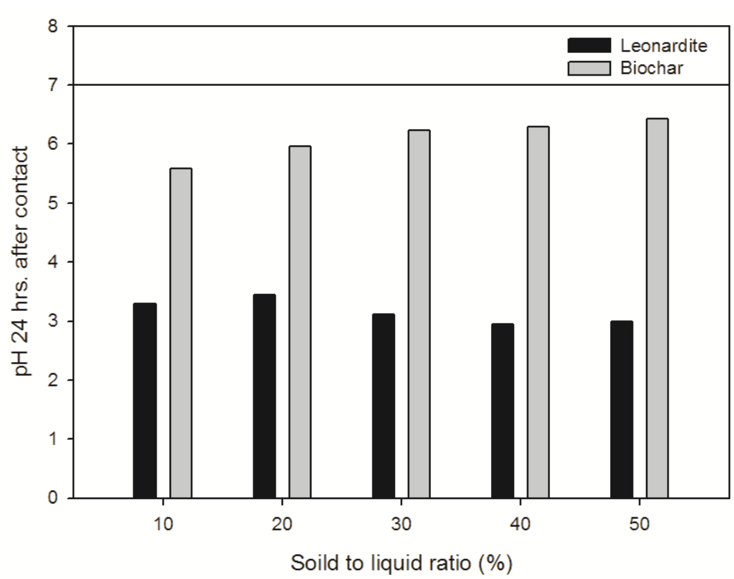


Figure 3. Change in pH after 24 hours of contact with leonardite or biochar with solid to liquid ratios ranging from 10 to 50%. Initial pH of 7 is shown as a reference line.

We observed a similar trend in pH change with leachate from the metal sequestration column experiment. Columns amended with leonardite resulted in a leachate with a much lower pH compared to the unamended control in the first month of column operation (Figure 4). Biochar had very little effect on the pH with leachate having a similar pH to the unamended control. Addition of lime did not raised the pH in the first month of operation. However, after 45 days, leachate from the column amended with lime showed an increase in pH towards that of the control.

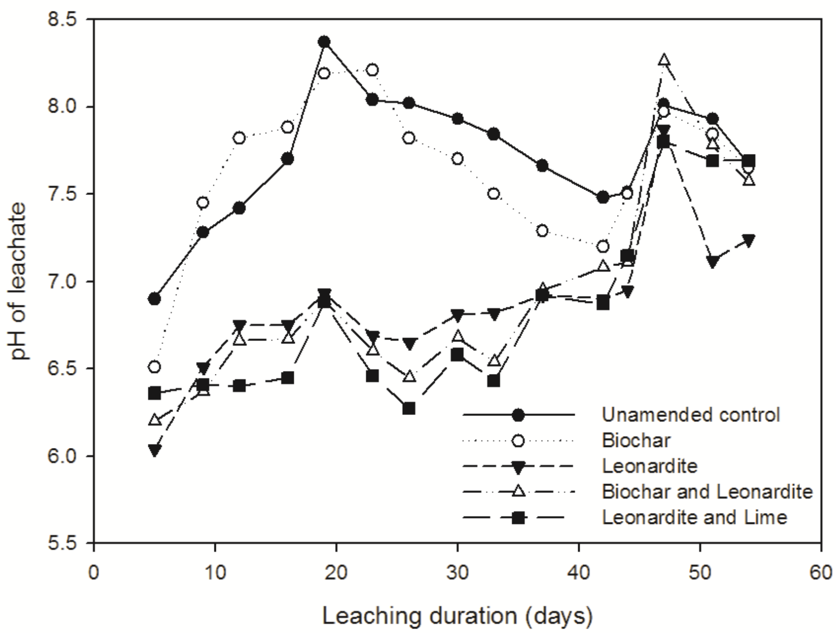


Figure 4. pH of leachate from tailings columns amended with biochar, lenoardite, biochar and leonardite and leonardite and lime over 60 days.

The column amended with leonardite had leachate with higher Cd and Zn compared with both the column amended with biochar and the unamended column (Figure 5). This trend may suggest that Cd and Zn are mobilized from the tailings by the leonardite. Biochar appears to reduce the amount of Cd and Zn leached from the tailings. Compared to the unamended control, leachate from the column with biochar had a 73.8% reduction in Cd and a 18.27% reduction in Zn.

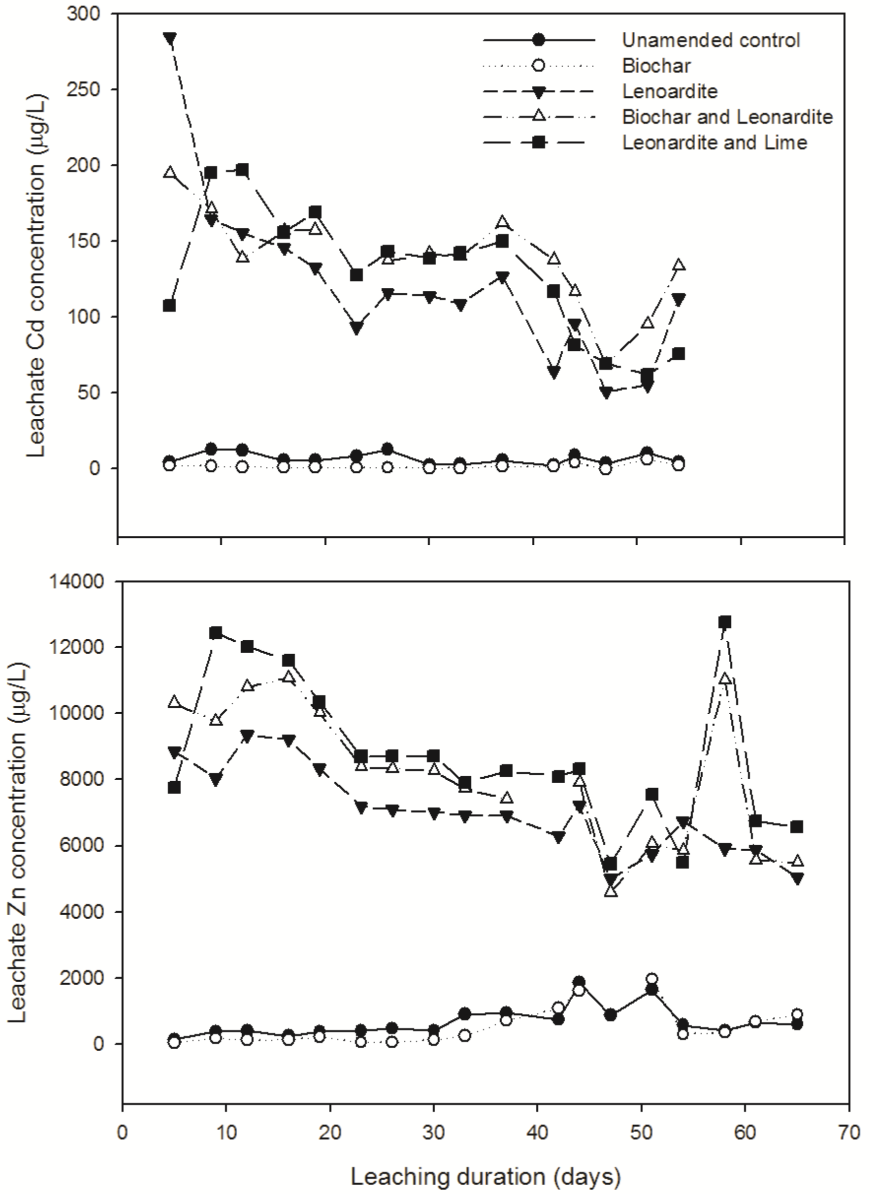


Figure 5. Cadmium and Zinc concentrations of leachate from tailings columns amended with biochar, lenoardite, biochar and leonardite and leonardite and lime over 70 days.

Both northern native herb species showed poor growth after two months in the greenhouse with higher belowground biomass than aboveground biomass. We did not detect any influence of the treatments on the below and aboveground biomass accumulation for *L. arcticus*, but found significantly higher belowgound biomass for *H. aplinum* grown in tailings amended with leonardite and lime (Figure 6).

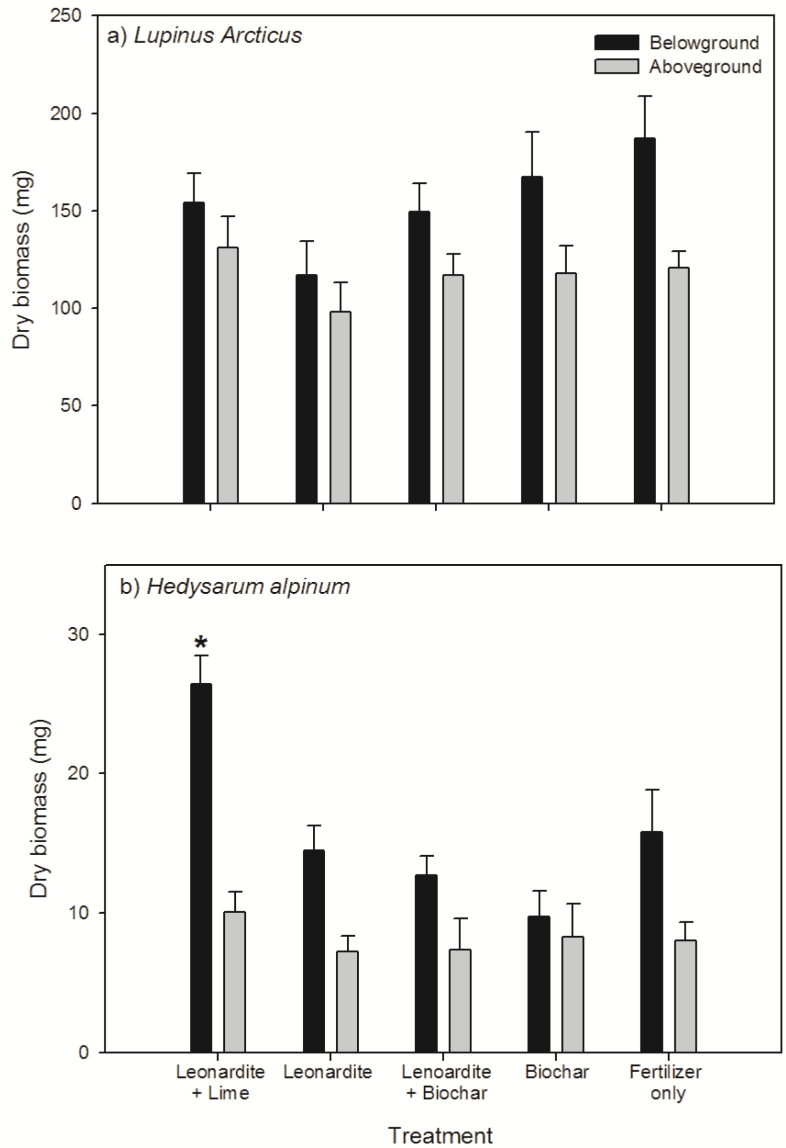


Figure 6. Belowground and aboveground biomass of *Lupinus Arcticus* (Arctic Lupin) and *Hedysarum alpinum* (Alpine Sweetvetch) following two months of growth in a greenhouse in tailings treated with leonardite and lime, leonardite, leonardite and biochar, biochar and fertilizer only. Bars are means with standard error. There were no significant differences in above or belowground biomass with treatment for *L. arcticus or* aboveground biomass for *H. aplinum* (ANOVA, $p>0.05$ for all comparisons). Belowground biomass of *H. aplinum* was significantly higher in tailings treated with leonardite and lime compared with all other treatments (ANOVA, TukeyHSD, $p<0.01$ for all comparisons).

Both species showed signs of potential heavy metal toxicity with chlorosis and a purpling and reddening of leaves after approximately 30 days of growth in the tailings (Figure 7). After 2 months of growth stem death was evident for some replicates of both herbs.

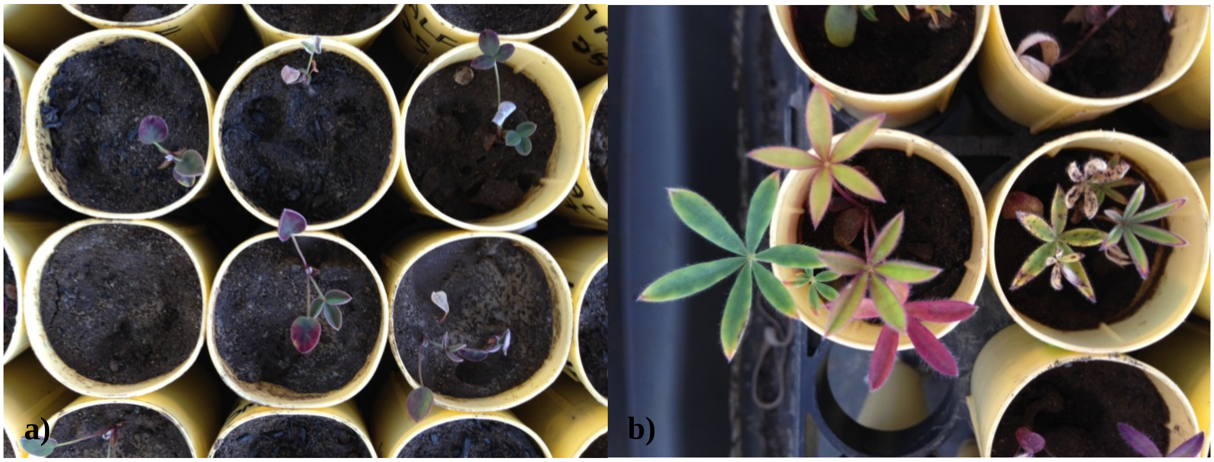


Figure 7. *Hedysarum alpinum* (a) and *Lupinus arcticus* (b) showing signs of chlorosis (i.e. yellowing and purpling of leaves) after approximately 30 days of growth in tailings likely due to heavy metal toxicity.

DISCUSSION

Both biochar and leonardite demonstrated metal adsorption of Cd and Zn. However, adsorption rates for biochar were considerably higher in synthetic water and Cd and Zn concentrations in leachate from tailings with biochar were reduced compared to the unamended control. Leonardite amended tailings resulted in the mobilization of Cd and Zn from the tailings, which was likely due to acidification of the water flowing within the tailings. We also observed that both biochar and leonardite lowered pH of mining contaminated water and tailings. However, leonardite lowered the pH to a much greater extent and likely accounts for the potential acidification and subsequent mobilization of Cd and Zn from leonardite amended tailings columns.

The poor growth of both northern native herb species on the tailings is most likely due to heavy metal toxicity. Other studies that have examined vegetation establishment of the UKHM tailings have also found chlorosis in species such as, *Arctostaphylos uva-ursi*, *Carex aquatilis*, and *Equisetum arvense* (Clark & Hutchinson, 2005). Zinc toxicity symptoms are known to include chlorosis and a reddening of younger leaves (Reichmann, 2002). However, the increased belowground biomass for *H. aplinum* in the leonardite and lime treatment suggests that leonardite may improve root development. Leonardite is known to stimulate root growth with both increased root length and development of secondary roots (Chen and Aviad 1990) and can reduce soil pH around roots helping to convert unavailable nutrients to plant accessible forms (Vaughan and Donald, 1976). However, in the context of remediation and phytostabilization of mine tailings decreases in pH associated with leonardite are undesirable and a liming agent may need to be considered. In the tailings amended with both leonardite and lime, pH appeared to equilibrate after 45 days and further studies are needed to examine the longer-term influence of a combined leonardite and lime treatment. Biochar shows strong potential for metal removal in both mining impacted water and tailings. This initial trial demonstrates that both leonardite and biochar have potential as amendment technologies for remediation and restoration in northern Canada.

ACKNOWLEGEMENTS

We would like to thank Raymond Potié of Wapaw Bay Resources for his collaboration on this project. We would also like to thank Hiromi, Isobel Ness, Annie-Claude Letendre, Justine Viros and Carrie Boles for laboratory and greenhouse assistance. The project was funded by an NSERC Applied Research and Development Grant.

REFERENCES

Adriano, D.C. 2001. Trace Elements in Terrestrial Environments: Biogeochem, Bioavailability and Risks of Metals, 2nd ed., Springer, New York.

Beesley, L., Marmiroli, M. 2011. The immobilisation and retention of soluble arsenic, cadmium and zinc by biochar. Environ Pollut, 159:474-480.

Chen, Y., Aviad, T. 1990. Effects of humic substances on plant growth. In: P. MacCarthy, C.E. Clapp, R.L. Malcolm and P.R. Bloom (Eds.) Humic Substances in Soil and Crop Sciences: Selected Reading, p. 161-186. American Society of Agronomy, Madison, WI.

Chen, X., Chen, G., Chen, L., Chen, Y., Lehmann, J., McBride, M.B., Hay, A.G. 2011. Adsorption of copper and zinc by biochars produced from pyrolysis of hardwood and corn straw in aqueous solution. Bioresource Technol 102:8877-8884.

Clark, A., and Hutchinson, T. 2005. Enhancing natural succession on Yukon mine tailings sites: A low-input management approach. MPERG Report 2005-3. Whitehorse, Yukon.

Fellet, G., Marchiol, L., Delle Vedove, G., Peressotti, A. 2011. Application of biochar on mine tailings: Effects and perspectives for land reclamation. Chemosphere, 83:1262-1267.

Jones, D.L., Rousk, J., Edwards-Jones, G., DeLuca, T.H., and Murphy, D.V. 2012. Biochar-mediated changes in soil quality and plant growth in a three year field trial. Soil Biol Biochemi 45:113-124.

Kolodynska, D., Wnetrzak, R., Leahy, J.J., Hayes, M.H.B., Kwapinski, W., Hubicki, Z. 2012. Kinetic and adsorptive characterization of biochar in metal ions removal. J Chem Eng 197:295-305.
Regmi, P., Moscoso, J.L.G., Kumar, S., Cao, X., Mao, J., Schafran, G. 2012. J Environ Manage 109:61-69.

Kumpiene, J., Lagerkvist, A., Maurice, C., 2008. Stabilization of As, Cr, Cu, Pb and Zn in soil using amendments – A review. Waste Manage 28:215–225.

Laird, D.A., Fleming, P., Davis, D.D., Horton, R., Wang, B., Karlen, D.L. 2010. Impact of biochar amendments on the quality of a typical Midwestern agricultural soil. Geoderma, 158:443-449.

Lao, C., Zeledon, Z., Gamisans, X., Sole, M. 2005. Sorption of Cd(II) and Pb(II) from aqueous solutions by a low-rank coal (leonardite). Sep Purif Technol 45: 79-85.

Madejón, E., Pérez-de-Mora, A., Burgos, P., Cabrera, F., Lepp, N.W., and Madejón, P. 2010. Do amended, polluted soils require re-treatment for sustainablerisk reduction? — Evidence from field experiments. Geoderma 159:174-181.

Madejón, E., Madejón, P., Burgos, P., Pérez-de-Mora, A., Cabrera, F., 2009. Trace elements, pH and organic matter evolution in contaminated soils under assisted natural remediation: A 4-year field study. J Hazard Mat 162:931-938.

Mench, M., Vangronsveld, J., Lepp, N.W., Ruttens, A., Bleeker, P., Geebelen, W., 2007. Use of soil amendments to attenuate trace element exposure: sustainability, side effects, and failures. In: Hamon, R.E., McLaughlin, M., Lombi, E. (Eds.), Natural Attenuation of Trace Element Availability in Soils. SETAC Press, Pensacola, Florida, pp. 197–228.

Namgay, T., Singh, B., Singh, B.P. 2006. Plant availability of arsenic and cadmium as influenced by biochar application to soil. World Congress of Soil Science: Soil Solutions for a Changing World. August 1-6, 2010, Brisbane Australia.

Nardia, S., Pizzeghelloa, D., Muscolob, A., Vianelloc, A. 2002. Physiological effects of humic substances on higher plants. Soil Biol Biochem 34:1527-1536.

Pérez-de-Mora, A., Burgos, P., Madejón, E., Cabrera, F., Jaeckel, P., and Schloter, M., 2006. Microbial structure and function in a heavy metal contaminated soil: effects of plant growth and different amendments. Soil Biol Biochem 38:327–341.

Pérez-de-Mora, A., Madrid, F., Cabrera, F., and Madejón, E. 2007. Amendments and plant cover influence on trace element pools in a contaminated soil. Geoderma 139:1-10.

Pérez-de-Mora, A., Ortega-Calvo, J.J., Cabrera, F., Madejón, E., 2005. Changes in enzyme activities and microbial biomass after "in situ" remediation of a heavy metal-contaminated soil. Appl Soil Ecol 28:125–137.

Reichmann, S. 2002. The responses of plants to metal toxicity: A review focusing on Copper, Manganese and Zinc. The Australian Minerals and Energy Envrionment Foundation, Occasional Paper No. 14 ISBN 1-876205-13-X

Ruttens, A., Adriansen, K., Meers, E., De Vocht, A., Geebelen, W., Carleer, R., Mench, M., Vangrosveld, J., 2010. Long-term sustainability of metal immobilization by soil amendments: cyclonic ashes versus lime addition. Environ Pollut 158:1428–1434.

Steiner, C., Teixeira, W.G., Lehmann, J., Nehls, T., Macedo, J.L.V.D., Blum, W.E.H., Zech, W., 2007. Long term effects of manure, charcoal and mineral fertilization n crop production and fertility on a highly weathered Central Amazonian upland soil. Plant Soil 291:275–290.

Vaughan, D., MacDonald, I.R. 1976. Some effects of humic acid on cation uptake by parenchyma tissue. Soil Biol. Biochem. 8:415-421.

Zeledón-Torunõ, Z., Lao-Luque, C., Sole´-Sardans, M. 2005. Nickel and copper removal from aqueous solution by an immature coal (leonardite): Effect of pH, contact time and water hardness. J Chem Technol Biotechnol 80:649-656.

ENVIRONMENTAL, SOCIAL, AND ECONOMIC BENEFITS OF BIOCHAR APPLICATION FOR LAND RECLAMATION PURPOSES

Elizaveta Petelina, B.Sc., MSEM[1]
David Sanscartier, PhD, P.Eng.[2]
Susan MacWilliam, B.Sc.[3]
Reanne Ridsdale, M.Sc. candidate[4]

[1]Remediation Specialist
[2]Research Engineer
[3]Life Cycle Assessment Analyst
Saskatchewan Research Council, Saskatoon, Canada

[4]M.A. candidate, Department of Geography & Planning
University of Saskatchewan, Saskatoon, Canada

ABSTRACT

Biochar is a solid material produced by pyrolysis of biomass, which was shown to improve soil properties. On the other hand, there are a number of risks and uncertainties associated with its use in land reclamation. This case study is aimed to assess environmental, social, and economical benefits and limitations of biochar use for revegetation projects in northern Saskatchewan. Four revegetation options were examined, i.e. natural restoration, revegetation with peat application, and revegetation with application of commercially or locally produced biochar. The assessment methods included option screening by the expert panel, stakeholder opinion survey, and quantitative assessment (i.e. screening life cycle assessment and life cycle costing analysis).

The study results suggest that biochar provides a number of environmental benefits and its on-site production can also provide social benefits and economic opportunities. On the other hand, biochar production and application is expensive and associated with technical risks, which can undermine overall project success. Nevertheless, positive trends in biochar production industry suggest that in the near future this material may serve as an affordable and technically reliable alternative to conventional soil amendments for land reclamation.

KEY WORDS

Revegetation, Multi Criteria Decision Analysis, option screening, life cycle assessment, life cycle costing analysis, soil amendment

INTRODUCTION

Biochar is a solid material obtained from any biomass (e.g. wood, organic wastes) through pyrolysis. It has been shown that adding biochar to the soil can improve its biological, chemical and physical properties and consequently increase its fertility (Lehman and Joseph 2009; Verheijen et al. 2009). In addition, biochar can be produced on-site using local labor and feedstock (e.g. waste wood), which may reduce reclamation cost, create local jobs, and improve waste management. On the other hand, biochar use as a soil amendment may face challenges, such as occupational risks (e.g. dust exposure or fire hazards) associated with its production, transportation, and storage, high cost, limited commercial availability, and lack of standardization of this product. (Lehman and Joseph 2009; Verheijen et al. 2009;

Petelina et al., 2013). With all the above, overall sustainability of using biochar as a soil amendment for land reclamation purposes is still questioned. This research is to get a better understanding of biochar benefits and limitations and to assess environmental, social, and economic aspects of biochar use in revegetation. The Gunnar Site Remediation Project (the Gunnar Project) managed by Saskatchewan Research Council (SRC) was selected as a case study for this research.

The Gunnar Project is aimed to clean-up an abandoned uranium mine site in northern Saskatchewan. The site includes unconfined uranium tailings, which gamma radiation exceeds acceptable levels and thus poses a risk to human health and environment. To reduce gamma radiation to acceptable levels and decrease contaminant loading to the environment, some 1 m engineered vegetative cover is to be installed on the top of the tailing deposits (SRC, 2014). The tailings engineer cover is to be constructed using borrow material, which is mostly composed of mineral silt and sand and almost devoid of organic matter and nutrients. The vegetation cover is to be formed by a grass-legume community with 60-80% ground cover. This plant community shall be self-sustaining and perform a number of ecosystem functions, such as impeding water infiltration into the buried tailings, provision of erosion control, prevention of establishment of prohibited weeds, exotic species, and deep rooting vegetation (i.e. trees and shrubs, which can facilitate contaminant transport to the soil surface). Under Gunnar conditions, achieving the above goals may require different time and effort depending on a revegetation approach; therefore, a thorough consideration of potential revegetation options is essential for the project success.

Four options were proposed for the Gunnar tailings revegetation as follows:

- <u>Option 1 (Natural Restoration)</u> suggested a passive restoration approach, i.e. creating favorable conditions for native recovery of the site. Wind erosion was found to be a main concern at the site, which could significantly impede plant establishment (SRC, 2013; Petelina, 2014). Therefore, wind controls (e.g., wind breakers) can both protect engineered covers from blowing and create conditions favorable for natural vegetation recovery. Due to harsh environmental conditions and poor growing media, it may take a few decades to reach revegetation goals. As natural recovery is driven not only by native herbaceous species, but also woody species and exotic species, tree suppression and active weed management will be required until a dense herbaceous cover is developed. Thus, this option included installation of wind breaks and site natural recovery accompanied with tree suppression and active weed management.
- <u>Option 2 (Peat)</u> suggested a conventional, proactive revegetation methodology including soil amended with peat (added to achieve 2% of the organic matter in the soil) followed by fertilizer application and native plant seeding. Results of SRC field trials at Gunnar showed that with this approach the revegetation goals could be achieved within two or three years (Petelina, 2013b).
- <u>Option 3 (Commercial Biochar)</u> was similar to Option 2, with the only difference that peat was substituted by biochar obtained from a commercial supplier. It was assumed that biochar was very similar to peat in its ability to promote plant growth, so all other features of Option 3 are the same as Option 2.
- <u>Option 4 (Local Biochar)</u> was similar to Option 3, but biochar was assumed to be produced with a mobile unit installed on-site utilizing local wood wastes as feedstock. Rough estimation of wood wastes from remediation activities showed that there would be enough wood wastes to produce sufficient amount of biochar. The other features of Option 4 are the same as Option 3.

The assessment of environmental, social, and economical benefits of the above revegetation options was completed through the following stages:

- Stage 1 – Expert Panel Review to identify the most preferred option on basis of available information
- Stage 2 – Stakeholder Opinion Survey to study stakeholders' preferences for revegetation approach
- Stage 3 – Quantitative Assessment of environmental impacts and revegetation cost through screening life cycle assessment and life cycle costing analysis

The overall approach is based on the NICOLE's and SURF-UK sustainable remediation decision-support and sustainability appraisal frameworks (NICOLE, 2010; SURF-UK, 2010). More details on each stage are provided in corresponding sections. Each section presents methodological details, as well as results and discussion for individual stage.

OPTION SCREENING BY EXPERT PANEL

Methods

SRC appointed an internal expert panel to screen the proposed revegetation options and identify a viable revegetation option with an optimal balance between benefits for environment, society, and economy. The panel included two engineers experienced in assessment of sustainability aspects, a socio-economic specialist, and a revegetation specialist. The team composition ensured a balance of technical, environmental, social, and economic expertise during the option screening.

The panel reviewed documents related to the Gunnar Project (including technical and socio-economic data, as well as public concerns), biochar and peat production and use, and site restoration techniques (including their perception by public) and also carried out interviews with interested parties (e.g. Gunnar Project management team, pyrolysis engineers, biochar producers, environmental consultants, representatives of local shipping companies). The collected data were used to identify benefits and limitations for each option. Upon discussion of option benefits and limitations each option was screened through a set of environmental, social, and economic criteria (Table 1). The criteria were developed on basis of sustainability criteria used for assessment of environmental, social, and economic benefits (Becker and Vanclay, 2003). For screening, the expert panel used a four level scoring system with the highest score of four (4) assigned to the most desirable benefit and the lowest score of one (1) assigned to the least desirable impact. First, all the options were rated against each criterion, then each option was ranked in relation to environmental, social, or economic aspect, and finally the most preferred option was identified. This approach allowed to select the most preferred option and identify trade-offs associated with each option.

Results

In general, the panel review provided the following summary of benefits and limitations for each option:

- Option 1 (Natural Restoration): in spite of some benefits (i.e. low cost, low occupational risks, low emissions), this option required a long time to achieve the project goals and had long-term risks, which could also have negative impact on local ecosystems and project perception by local communities.
- Option 2 (Peat): this option was more costly and complicated than Option 1, yet more reliable and allowed achievement of the desired results within a shorter period of time.
- Option 3 (Commercial Biochar): this option was more costly than the previous two and was also associated with significant challenges and numerous uncertainties. On the other hand, successful completion of this project would not only have a positive impact on local ecosystems and project perception by local communities, but also would create a precedent for biochar use in large scale land reclamation.
- Option 4 (Local Biochar): this option might be more cost effective than Options 2 (Peat) and 3 (Commercial Biochar), but bears significant challenges, risks, and uncertainties. In addition, successful completion of the project would not only have a positive impact on local ecosystems and project perception by local communities, but also create a precedent for alternative approaches to land reclamation providing long term opportunities for local communities.

Results of option screening against environmental, social, and economic criteria (Table 1) showed that both Option 1 (Natural Restoration) and Option 3 (Commercial Biochar) provided the least preferred approaches, while Option 4 (Local Biochar) was the most preferred option. However, biochar production and application for Option 4 was less proven than peat harvesting and application, and thus had an increased risk of failure. As such, Option 2 (Peat) could be considered a more practical solution, though opportunities for economic development and community involvement would be lost.

It should be noted that the panel review approach had limitations, such as lack of stakeholder participation in review, no criteria prioritization, the qualitative nature of the method used, and limited data for assessment of some criteria (e.g. cost). These factors affected the results of the assessment; therefore, it was decided to carry out multi-criteria decision analysis, which includes increased involvement of stakeholders and criteria prioritization based on stakeholders' values and needs.

STAKEHOLDER OPINION SURVEY

Methods

Stakeholder opinion was studied using multiple criteria decision analysis (MCDA) approach which is often used to examine a variety of options against a wide range of environmental, social, and economic criteria (e.g., White and Noble, 2013). The MCDA was carried out by evaluating the four revegetation options based on nine sustainability evaluation criteria (Table 2), selected from the 20 criteria (i.e., criteria listed in Table 1). Selection was made by the expert panel based on the criteria perceived relative importance.

Table 1. Results of option screening against environmental, social, and economic criteria

	Criterion	Option 1: Natural Restoration[1]	Option 2: Revegetation with Peat[1]	Option 3: Revegetation with Commercial Biochar[1]	Option 4: Revegetation with Local Biochar[1]
Environmental	Biodiversity Footprint (abundance and diversity of native wildlife and plants at local and regional level)	**2**	**3**	**2**	**3**
	Air Quality (emissions from combustion engines (e.g. trucks, tractors, or bio-char mobile unit), as well as noise and dust pollution)	**3**	**2**	**2**	**2**
	Energy Consumption (consumption of energy or power for the option implementation (including consideration of non-renewable energy versus renewable energy))	**3**	**2**	**2**	**2**
	Greenhouse Gases (greenhouse gases emissions from use of fossil fuel and use of fertilizers)	**3**	**2**	**1**	**2**
	Carbon Sinks (carbon storage opportunities to decrease amount of carbon dioxide in the atmosphere)	**2**	**2**	**3**	**3**
	Waste Generation (amount of wastes to be produced during a revegetation option implementation)	**2**	**3**	**3**	**3**
	Environmental Benefits Overall Outcome	the most preferred option	the second-preferred option	the least preferred option	the most preferred option
Social	Occupational Risks (risks associated with carrying out the project)	**2**	**2**	**2**	**3**
	Site Aesthetic (time required for desired vegetation cover establishment)	**2**	**4**	**3**	**3**
	Land Use (benefits that the reclaimed land can provide to the community members both locally and regionally)	**2**	**4**	**3**	**3**
	Public Safety (meeting the reclamation goals in terms of the public safety)	**1**	**4**	**3**	**3**
	Community Perception (perception of the revegetation results by the community)	**2**	**3**	**3**	**4**
	Community Involvement (opportunities for local communities to benefit from the project)	**2**	**3**	**3**	**4**
	Social Benefits Overall Outcome	the least preferred option	the second-preferred option	less preferred option	the most preferred option
Economic	Project cost (investments in the revegetation option implementation)	**3**	**1**	**1**	**2**
	Project risks (likelihood for successfully meeting the project goals, which will reduce risk of additional investments in the project)	**1**	**4**	**3**	**2**
	Economic Opportunities (future economic opportunities which can raise from a proposed option implementation including both for local community and Province)	**1**	**2**	**3**	**3**
	Province Revenue (provincial tax revenue due to increased economic activities)	**2**	**3**	**3**	**2**
	Job Opportunities (creation of job opportunities for local communities)	**2**	**2**	**2**	**3**
	Job Diversity (creation of different job opportunities for local communities)	**2**	**2**	**2**	**3**
	Technical Feasibility (reliability of proposed technology and availability of corresponding resources)	**4**	**4**	**3**	**2**
	Economic Benefits Overall Outcome	the least preferred option	the most preferred option	the second-preferred option	the second-preferred option

Note: [1] 4 - the most desirable benefit and 1 - the least desirable impact

As the study was run as an internal SRC project, the survey participants were selected from SRC employees not directly involved in the project. The selected employees acted as stakeholders and reflected different perspectives of five interested parties: aboriginal communities (a local employee from a northern Saskatchewan community), project operator (Business Unit Manager), environmental consultants (Distinguished Scientist, expert in forest ecosystems), technical specialists (Research Engineer, expert in biochar production), and a finance specialist (Financial Analyst). Prior to participating in the survey, stakeholders were provided with supportive documentation outlining the project objectives, the four revegetation options, and the survey methodology.

Expert Choice, a web-based MCDA software, was used for the data collection and processing. Expert Choice was developed on basis of an analytical hierarchy process, which allowed to prioritize multiple criterions with different project options and get accurate results even when complex sustainability issues were under consideration (White and Noble, 2012; Hreljac, 2013). The pairwise comparison method was applied for this survey. First, stakeholders were asked to compare the sustainability criteria one against the other. Results from this comparison were used for factors weighing to be used for the determination of the preferred revegetation options. Then, stakeholders were asked to compare the revegetation options one against the other for all nine evaluation criteria. Expert Choice automatically processed the survey data and generated results.

Results

Table 2 provides the results of criterion weighting, option ranking against each criteria, and overall outcome from the stakeholder survey. Greenhouse gases, biodiversity, and project risks were top three criteria indicated by stakeholders as the most important ones (their weighting factor was 19, 16, and 15%, respectively). All social criteria (i.e. occupational risks, community involvement, and land use) received the lowest weighting score (4, 6, and 6%, respectively), which could be explained by low representation of local communities among the survey participants (i.e. only one of five participants represented local community interests).

In general, the results of stakeholder survey were in line with the panel review results, i.e. Option 4 (Local Biochar) was selected by stakeholders as the preferred option. Option 1 (Natural Restoration) and Option 3 (Commercial Biochar) were identified as the least desirable options, which also support the validity of the panel review outcome. On the other hand, the results of option ranking against specific criteria by stakeholders in some cases differed from the expert panel ratings. For example, stakeholders indicated Option 4 as the preferred option in relation to project risks, but the expert panel assigned quite a low scoring for this option against project risks criteria (score of 2 – Table 1). This discrepancy between Stage 1 and Stage 2 results was likely due to unequal amount of information provided to the expert panel and stakeholders. It should be noted that upon survey completion some participants indicated that they did not have enough information to make meaningful choices during pairwise option comparisons. For example, they would have liked to get more information on overall option cost estimates and estimates of greenhouse gas emissions, which was in line with data gaps indicated by the expert panel. Thus, it was decided to carry out quantitative analysis of revegetation options as discussed in the next section.

Table 2. Results of the multiple criteria decision analysis of stakeholder opinion survey

Criterion	Weighting Factor (%)	Option 1: Natural Restoration[1]	Option 2: Revegetation with Peat[1]	Option 3: Revegetation with Commercial Biochar[1]	Option 4: Revegetation with Local Biochar[1]
Environmental: Biodiversity	16	27	89	100	85
Environmental: Air Quality	12	100	46	42	39
Environmental: Greenhouse Gases	19	100	66	69	97
Social: Occupational Risks	4	100	78	63	67
Social: Community Involvement	6	25	25	19	100
Social: Land Use	6	24	100	50	89
Economic: Project cost	11	100	64	20	59
Economic: Project risks	15	25	98	77	100
Economic: Economic Opportunities	11	18	23	30	100
Overall Stakeholder's Opinion	**n/a**	**72**	**81**	**70**	**100**

Note: [1] Results are presented as a percent of the maximum score, i.e. the highest choice was given 100%, and the following choices were valued relatively to the maximum score.

QUANTITATIVE ASSESSMENTS

Methods

Quantitative assessment included screening Life Cycle Analysis (LCA) and Life Cycle Costing (LCC) analysis. LCA is a method of examining the potential environmental effects of a product or process across its life span. LCC is employed to examine the cradle-to-grave cost of products, processes or systems. Due to budget and timing constraints, LCA and LCC screening studies were conducted for only Option 2 (Peat), Option 3 (Commercial Biochar), and Option 4 (Local Biochar). Screening studies were limited in scope and detail when compared to full-scale analyses and were intended to provide information for use internally to an organization, or to direct future research. The functional unit of analysis was 40-60% vegetation cover and less than 60% bare ground of the largest Gunnar tailing (53 ha) within three years of initiating revegetation activities. Amount of peat and biochar required for reaching the revegetation goal was assessed on basis of SRC revegetation trials and equaled to 8,480 and 5,300 tonnes, respectively (Petelina, 2013).

Only readily available data were used in screening studies. The data were selected to be as specific as possible to the case study, but constraints required the use of several assumptions. The following main activities were considered: acquiring/producing organic soil amendment, application of organic soil amendment, and transport of personnel and materials. Shipment estimation of commercial materials and equipment was based on delivery of the materials from Saskatoon to the Gunnar Mine Site. It was decided to assess only relative impact or cost of the options, so activities similar between the revegetation options (e.g. seeding and fertilizing) were excluded from LCA and LCC analysis on the that these activities are associated with similar impacts. In other words, only the environmental and economic effects different between the options were considered.

Assessment of biochar production slow pyrolysis systems (i.e. commercial and mobile) were based on a LCA study examining the energetic, economic, and climate change potential of biochar from corn stover (Roberts et al., 2010). The biochar product yield was 30%, with the remaining products in the forms of oil and gas. The oil and gas products were combusted to generate energy for use elsewhere in the commercial facility (i.e. for producing products other than biochar), thus offsetting a portion of the facility's natural gas requirements. In the mobile unit, the syngas product was combusted and released to the atmosphere as it could not be successfully used in other operations. The capacity of the mobile pyrolysis unit was 50 dry tonnes of feedstock per day. The feedstock would be transported to the pyrolysis unit (approximately 5 km) where it would be grinded prior to producessing into biochar. It was assumed that biochar production would only occur during the six warm months of the year (from mid-April to mid-October), due to harsh winter conditions in Northern Saskatchewan. It was assumed that SRC would purchase and operate the mobile unit. Upon revegetation completion, the mobile unit would be used for other projects.

For the LCA, peat, biochar, diesel, propane, and woody biomass were the material inputs. While electricity and energy required for production and transportation of the material inputs, as well as energy required for revegetation implementation were the energy inputs. LCA outputs comprised revegetated land and emissions to air, soil, and water. SimaPro (version 7.3.2, Pré) LCA modeling software was used for the analysis. The IMPACT 2002+ midpoint method was employed (Jolliet et al., 2003). The following five environmental metrics were examined:

- Respiratory inorganics (kg $PM_{2.5}$-eq) – air pollutants such as sulphur oxides and volatile organic compounds
- Terrestrial acidification (kg SO_2-eq) – potential proton and/or chemical nutrient release to soil
- Global Warming (kg CO_2-eq) – potential greenhouse gases emitted over the course of the system life cycle
- Non-renewable energy (MJ primary energy) – non-renewable energy consumption
- Mineral extraction (MJ additional energy) – resource intensiveness of the options examined

One of the main benefits of biochar is its ability to sequester atmospheric carbon. Biomass takes up carbon dioxide from the atmosphere as is grows. This carbon dioxide is then released back to the atmosphere when the biomass decomposes. In the case of biochar from pyrolysis, however, a portion of the carbon is locked in the biochar where it remains for many years. As such, biochar is considered to sequester carbon. For the purpose of this LCA, three tonnes of carbon dioxide were assumed to be sequestered by every tonne of biochar applied to land (Galinato et al., 2008).

For LCC, life expectancy of the mobile slow pyrolysis unit was approximately 20 years. The capital cost and operation and maintenance cost of the mobile unit were based on data listed in Shackley et al. (2011) which provided costs for a range of scales and resulted in $8.37 million for capital cost and $0.43 million for operational cost for the scale of the machine assessed (i.e., 50 dry t feedstock/day). It was assumed that it would be used only for five years for this project (i.e. mobilization and commissioning (Year 1), production of biochar during Year 2 and 3 (6 months/yr), demobilization (Year 4), and one year idle to find another contract (Year 5)) and then used for other project. Therefore, only a portion of the capital cost was allocated to the cost of Option 4 (Local Biochar). This portion was estimated by applying the annual capital cost equation (Equation 1 from Roberts et al., 2010) and represented $4.03 million.

The price of peat was based on Saskatchewan retail price of bailed peat and was equivalent to $300 /dry tonne. The cost of production of biochar was based on average US retail price and was equivalent to $1,000/ dry tonne (Brunjes, 2012). All cost data were reported in 2011 $Cdn. Data from the literature were converted using relevant exchange rates, construction price indices, and consumer price indices (for transportation) (Statistic Canada, 2013a and 2013b). Transport of material was calculated based on data in Ghafoori and Flynn (2009) (i.e. fixed cost of $6.79/tonne, variable cost: $0.12/t*km)

Results

LCA screening results are shown in Figure 1. Option 2 (Peat) resulted in the greatest potential environmental impacts in all impact categories when compared to the other options. Option 3 (Commercial Biochar) resulted in the lowest potential greenhouse gas impacts (i.e. global warming). The impacts to global warming are related to the ability of biochar to sequester atmospheric carbon (3t CO2eq/t biochar assumed in this study). This option also was the most preferred option in terms of non-renewable energy consumption. These results were based on the assumed usage of the biochar oil and gas co-products as a heat source in other operations at the facility, thus offsetting a portion of the natural gas requirements. Thus, the commercial biochar production process resulted in a greater amount of energy than was required for pyrolysis and resulted in an energy credit. Option 4 (Local Biochar) showed the best balance among all environmental impacts, i.e. it had relatively low potential impact on respiratory inorganics, terrestrial acidification/nutrification, non-renewable energy, and mineral extraction, and also resulted in a reduction of the greenhouse gases. Figure 1 shows the global warming impacts of Option 4 with (3t CO2eq/t biochar) and without (0t CO2eq/t biochar) carbon sequestration to demonstrate the effects of the assumed rate of sequestration.

LCC screening results are shown in Table 4. The total cost of Option 3 (Commercial Biochar) was about $ 20 million, which was much higher than total costs of Option 2 (Peat) and Option 4 (Local Biochar), which were approximately equal to $4 and $5 million, respectively. For Option 4, the highest cost was the capital cost of the mobile unit allocated to the project. This value also had the highest level of uncertainty and variability as it was estimated based on best fit of empirical data. It was also sensitive to the analysis assumptions. An important assumption was that after this project, new contracts for biochar production would be obtained and the unit would be used, thus allowing the allocation of the capital cost of the mobile unit to other projects. The cost to the project would differ from the estimate if contracts were not obtained. For example, if the unit remained idle for 5 years after the project, the capital cost allocated to the project increases from $4 million to $8 million, thereby increasing the cost of the project to $9 million. This cost was still lower than that of the cost of Option 3, but it was twice that of Option 2.

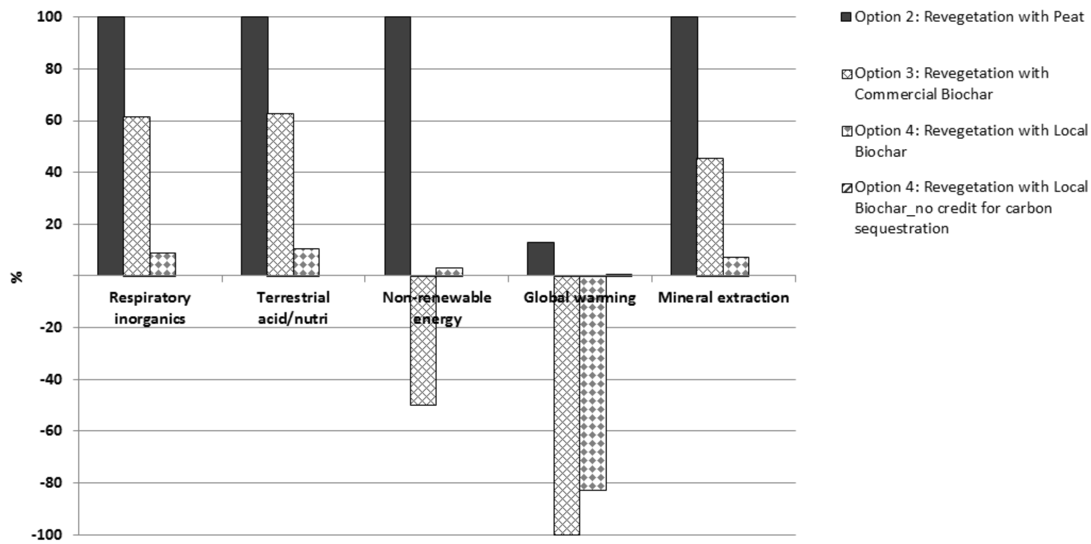


Figure 1. Life Cycle Assessment results for potential environmental impact from three revegetation options. Results are presented relative to highest absolute score (i.e. Option 2 for respiratory inorganics, terrestrial acidification/nutrification, non-renewable energy and mineral extraction and Option 3 for global warming).

As the mobile unit capital cost was sensitive to a number of years allocated to the project, reduction of Option 4 (Local Biochar) cost could be made by operating the unit all year around. At a rate of 15 dry t biochar produced per day and assuming 24h/d operation with 20% down time for maintenance, it would take approximately 400 days to produce 5,300 t of biochar. Reducing the number of years from 5 to 3 by operating all year brings the cost of Option 4 in line with that of Option 2 ($4.3 million). But this might require additional capital investments, such as storage for the feedstock and equipment, which were not considered. Other scenarios might also improve the cost attractiveness of Option 4, such as selling the mobile unit after the project to recover a portion of the initial investments.

Table 3. Life Cycle Cost Analysis Results for three revegetation options.

Life cycle activities	Option 2: Revegetation with Peat	Option 3: Revegetation with Commercial Biochar	Option 4: Revegetation with Local Biochar	Notes
Acquisition of soil amendment	$2,544,000	$19,451,000	NA	8,480 dry tonne for peat and 5,300 dry tonne for biochar
Application of organic soil amendment	$67,302	$42,063	$42,063	The same method of application is assumed
Capital cost of mobile unit associated with this project	NA	NA	$4,031,000	Total capital cost $8.37 million for a unit. Number of year allocated to the project = 5 years. Discount rate = 5%.
Transport of mobile unit to Gunnar	NA	NA	$22,300	Assumed transport on 10 flat beds
Transport of biomass to mobile unit	NA	NA	$196,000	Assumed 5 km transportation. Input biomass moisture content: 40%

Life cycle activities	Option 2: Revegetation with Peat	Option 3: Revegetation with Commercial Biochar	Option 4: Revegetation with Local Biochar	Notes
Biomass processing for biochar production	NA	NA	$132,000	Based on cost of delivered diesel in Gunnar of $1.86 - based on invoice from Ucity Bulk Fuels (Jansen et al., 2012)
Operation and maintenance of mobile unit	NA	NA	$425,000	Based on data listed in Shackey 2011
Transportation soil amendments and equipment	$1,405,000	$878,000	NA	
Travel to site of staff for set up of biochar production	NA	NA	$143,100	
Total	**$4,016,000**	**$20,371,000**	**$4,991,000**	

To sum up, it is not immediately clear from the quantitative analysis results which option is preferred. For example, Option 4 (Local Biochar) was found more preferable from the environmental point of view and Option 2 (Peat) was the most cost efficient option. However, the information could support decision making for reaching consensus on option selection and provides information on trade off amongst options. For example, higher cost options (e.g. Option 4 vs Option 2) should not be disregarded as it could provide environmental benefits. In line with the purpose of this case study (i.e., identify a viable revegetation option with an optimal balance between benefits for the environment, society, and economy), on-site biochar appeared to strike a balance between life cycle cost and environmental performance.

CONCLUSION

This case study provides suggested a comprehensive analysis of environmental, social, and economic benefits associated with options biochar application for land reclamation in northern Saskatchewan. The study demonstrated the potential environmental impacts associated with It has been shown that biochar production and transportation is associated with air pollution, greenhouse emission, energy consumption, terrestrial acidification, and waste production. On the other hand, this soil amendment appears more environmentally friendly than the common alternative (i.e. peat). Therefore, its use for revegetation purposes can help to achieve environmental goals such as reduction of greenhouse gases, air pollutants, terrestrial acidification, and energy consumption, as well as protection of local and regional biodiversity. On-site biochar production can also provide social benefits, such as local community involvement and better public perception of the project. On the other hand, biochar production can be associated with occupational risks and its use instead of peat can be less effective in reaching other social goals such as improved site aesthetic, increased land use benefits and assured public safety due to remediation success.

Besides environmental and social benefits, biochar application for land reclamation practices can provide economic benefits through providing new opportunities for local and regional businesses, rise in province revenue, and employment increase. On the other hand, high cost of biochar may be a challenge. Also, the relatively new technology may be associated with higher technological and project risks. This could result in less application by reclamation practitioners and lack of faith by decision makers.

It should be noted that biochar industry develops fast, which results not only in increased production capacity of biochar facilities but also improve biochar quality and, subsequently, reduces biochar cost. For example, SRC biochar supplier have increased their production by a factor of 20 and reduced the prices twofold for the last four years (Levine, 2014). They also increased biochar quality by improving its physical and chemical properties. This positive trend suggests that in the nearest future, biochar can become more affordable and a technically reliable alternative to conventional land reclamation techniques.

REFERENCES

Becker, A. and F. Vanclay, 2003 The International Handbook of Social Impact Assessment: Conceptual and Methodological Advances. Edward Elgar Publishing Limited, Cheltenham.

Brunjes, L. 2012. Biochar Industry Status Update. 2012 US Biochar Conference.

Galinato, S.P, Yoder, J.K., and Granaststein, D. 2008. The economic value of biochar in crop production and carbon sequestration. Energy Policy 39:6344-6350

Ghafoori, E. and Flynn, P.C. 2007. Optimizing the size of anaerobic digesters. Transaction of the 2007 American Society of Agricultural and Biological Engineers, 50: 1029-1036.

Hrelijac, E. (Expert Choice Technical Staff). 2013. Email correspondence with D. R. Ridsdale.

[IPCC] Intergovernmental Panel on Climate Change. 2007. Chapter 2: Changes in Atmospheric Constituents and in Radiative Forcing. IPCC Fourth Assessment Report. Working Group I Report “The Physical Science Basis”. pp.212

Jansen R., Hemingston H., James C., Sanscartier D. 2012. Evaluation of Diesel Fuel Use Reduction and Prefeasibility Study for Efficiency Measures at the Gunnar Mine Site. SRC Internal Report (July 20, 2012).

Jolliet, O., M. Margni, et al. 2003. "IMPACT 2002+: A New Life Cycle Impact Assessment Methodology." International Journal of Life Cycle Assessment, 83: 324-330.

Lehmann, J. and S. Joseph. 2009. Biochar for environmental management: an introduction. IN: Biochar for Environmental Management (Lehmann J. and S. Joseph eds). Earthscan, London. pp. 1–12.

Levine, J. 2014. Update on the Biochar Company progress from 2011 to 2014. Email correspondence with E. Petelina.

NICOLE, 2010. SR Road Map. http://www.nicole.org/documents/DocumentList.aspx?w=SR

Petelina E. 2013. Gunnar Revegetation Research, Field Trials, Stage II, Interim report. SRC Publication No. 12194-460-10B12.

Petelina E. 2014. Restoration of Native Plant Cover after Uranium Mining: a Case Study from Northern Saskatchewan, Canada. (Master's report).

Petelina E., A. Klyashtorin, and T. Yankovich. 2013. Biochar application for revegetation purposes in northern Saskatchewan. Proceedings of the 2013 Northern Latitudes Mining Reclamation Workshop.

Roberts, K.G., Gloy, B.A., Joseph, S., Scott, N.R. and Lehmann, J. 2010. Life Cycle Assessment of Biochar Systems: Estimating the Energetic Economic, and Climate Change Potential. Journal of Environmental Science and Technology, 44: 827-833.

Shackley S., Hammond J., Gaunt J., Ibarrola R. 2011. The feasibility and costs of biochar deployment in UK. Carbon Management, 2(3): 335-356.

SRC (Saskatchewan Research Council). 2014. Gunnar site remediation project, environmental impact statement. SRC publication, Saskatoon, SK. 920 pp.

Statistics Canada. Consumer Price Index, Transportation, Canada. 2013a. http://www.statcan.gc.ca/tables-tableaux/sum-som/l01/cst01/econ160a-eng.htm Accessed Dec 6, 2013.

Statistics Canada. Non-residential building construction price indexes. 2013b. http://www40.statcan.ca /l01/cst01/econ144e-eng.htm, Accessed Dec 6, 2013

SURF-UK] Sustainable Remediation Forum UK 2010. *A framework for assessing the sustainability of soil and groundwater remediation.* Contaminated Land: Applications in Real Environments (CL:AIRE), London.

Verheijen, F., S. Jeffery. A.C. Bastos, M. van der Velde, and I. Diafas. 2009. Biochar Application to Soils - A Critical Scientific Review of Effects on Soil Properties, Processes and Functions. EUR 24099 EN, Office for the Official Publications of the European Communities, Luxembourg, 149pp.

White L., and B. Noble. 2012. Strategic environmental assessment in the electricity sector: an application to electricity supply planning, Saskatchewan, Canada. Impact Assessment and Project Appraisal, 30(4): 284-295.

ASSESSING GROUNDWATER DISCHARGE TO STREAMS WITH DISTRIBUTED TEMPERATURE SENSING TECHNOLOGY

T. Birkham[1]
S.L. Barbour[2]
A. Goodbrand[1]
L. Tallon[3]
J. Szmigielski[3]
R. Klein[4]

[1]O'Kane Consultants Inc., Cranbrook, BC.
[2]University of Saskatchewan, Saskatoon, SK.
[3]O'Kane Consultants Inc., Saskatoon, SK.
Teck Resources Ltd., Sparwood, BC.

ABSTRACT

Characterization of the interactions of groundwater with surface stream water is fundamental to understanding and managing stream water quality in natural and altered watersheds. The spatial and temporal variability in the exchange of water and solutes across the streambed affects stream water quality evolution along the channel length, especially in coarse-grained alluvial sediments. Identifying and delineating areas of groundwater discharge has the potential to inform water quality management strategies including the location and design of intake structures, groundwater cut-off systems, or permeable reactive barriers. Distributed temperature sensing (DTS) is an emerging technology that has found wide application in stream hydrology. A DTS system integrates a fibre-optic (FO) cable to measure continuous temperature profiles (manufacture reported accuracy of ±1 °C and resolution of ±0.01 °C) with spatial resolutions of 1 m or better over distances of up to several kilometres. In September 2013, a DTS cable was installed in a predominantly coarse-textured stream channel at a coal mining operation in the Elk Valley of British Columbia. The objective was to evaluate the utility of a FO DTS system in a location where identification of groundwater discharges to a stream was being studied as part of Teck Resources Limited (Teck) applied research and development program focused on managing water quality in mine watersheds. This study was intended as a verification of methods for in-stream DTS installation and measurements with recommendations made for improved calibration procedures. Temperature was measured along a 160-m stretch of stream and streambed at 20 minute sampling intervals using a 5 minute signal integration time over six days. In-stream DTS measurements were successful in delineating a localized area of lower temperature identified as a groundwater discharge zone. The measurement of cooler groundwater discharge in the streambed relative to the stream was corroborated by the presence of well-established riparian vegetation and a streambank seep.

INTRODUCTION

Characterization of the interactions of groundwater with surface stream water is fundamental to understanding and managing stream water quality in natural and altered watersheds. The spatial and temporal variability in the exchange of water and solutes across the streambed affects stream water quality evolution along the channel length, especially in coarse-grained alluvial sediments. Identifying and delineating areas of groundwater discharge has the potential to inform water quality management strategies including the location and design of intake structures, groundwater cut-off systems, or permeable reactive barriers.

Distributed temperature sensing (DTS) is an emerging technology that utilizes fibre optic (FO) cable to measure continuous temperature profiles with manufacturer stated temperature resolution as high as 0.01 °C over long distances of up to several thousands of metres. Distributed temperature sensing has found wide application in stream hydrology and has strong potential for application in mine-affected watersheds. Fibre optic DTS systems use a laser signal with temperature dependent backscatter properties along the length of the cable (Selker et al., 2006 a,b; Tyler et al., 2009). The difference in Raman backscatter of the Stokes (temperature independent) and anti-Stokes (temperature dependent) wavelengths are used to determine the location of temperature measurements.

Three processes control streambed heat transfer: 1) advective heat transfer; 2) conductive heat transfer; and, 3) radiative heat transfer (Constantz, 2008). Research has shown that patterns in streambed temperatures are dominantly controlled by advective heat fluxes from upwelling groundwater. At most times of the year there is a significant difference between groundwater and stream temperatures, as stream temperatures are altered by daily and seasonal climatic conditions while groundwater temperatures remain relatively constant. This difference in temperatures provides a conservative tracer to help identify and delineate areas of groundwater upwelling (Selker et al., 2006a), and has been used to quantify localized groundwater discharge using mixing models (e.g. Westhoff et al., 2007; Lauer et al., 2013).

The objective of this study was to evaluate the utility of a FO DTS system in a location where groundwater discharge to a stream was being studied. The study was part of a Teck Resources Limited (Teck) applied research and development program focused on managing water quality in mine watersheds.

METHODS

Study Area

The study was conducted in a small watercourse called Line Creek (Figure 1; Photo 1), located within Teck's Line Creek coal mine operations (11 U 659421 E 5533087 N) in the Elk Valley of British Columbia. The climate is humid continental with a mean annual precipitation of 600 to 800 mm and a potential evapotranspiration of approximately 630 mm. Line Creek emerges from a rock drain upstream of the study area and flows in a southwesterly direction. Flow data measured at the LC_3 station (Figure 1) during 2013 indicated an annual mean daily discharge of 1.31 m^3/s, peak flows of 6.14 m^3/s during spring freshet and flows as low as 0.30 m^3/s during winter. West Line Creek (WLC) forms a tributary with Line Creek ~100 m upstream of LC_3. At the time of the study, WLC was diverted through a culvert and into Line Creek south of historic settling ponds approximately 160 m downstream of the natural confluence. A 160 m reach starting at approximately 460 m from the LC rock drain was selected for installing the DTS system (Figure 1). The study reach has been influenced by mining operations resulting in steep banks and varying bank widths from 5 to 15 m along the length of the study reach. The streambed gradient in the study reach is relatively flat. The stream changes from a wider, braided channel with approximately 0.2 – 0.5 m water depth (at time of study) to a narrower channel (~0.6 – 0.8 m water depth) with more turbulent flows downstream. The streambanks support mixed grass species with a few conifer species. At ~105 m along the study reach, vegetation is located below the toe of the streambank. A small seep was observed at this location (Photo 2). The geology of the site can be described as fluvial and glacial alluvium underlain by marine shale bedrock (Jakub Szmigielski, personal communication). The streambed is located in coarse-textured alluvium sediments and range from a mixture of pebbles and gravel with a greater percentage of cobbles and boulders further downstream in the study reach.

Field Measurements

Line Creek streamflow was measured at an established stream gauging station (LC_3; ~1430 m asl; 11 U 660090 E 5532022 N) approximately 170 m upstream of the study section (Figure 1). Discharge from the WLC diversion culvert during the study period was estimated using the Manning's n equation for a corrugated culvert (n = 0.0245) and depth of water. WLC streamflow was also measured upstream of the diversion culvert through a monitored culvert. Climate data consisting of net radiation (Net Radiation Kipp & Zonen model NR-LITE2 Net Radiometer), air temperature and relative humidity (Air Temperature and RH HC2-S3 Probe), wind speed (R.M. Young Model 05103AP-10 Wind Monitor) and rainfall (CS700 Tipping Bucket Rain Gauge) were collected from the closest weather station located 1.3 km away on the plateau of a waste rock deposit (LC_RME; 1800 m asl; 11 U 659421 E 5533087 N) north of the study reach. The temperature of four possible sources of water to Line Creek (Figure 1) were measured using thermistors integrated in the water level pressure transducers and measured at 30 min intervals (Solinst Gold Levelogger) from: a) below the West Line Creek diversion culvert (T1); b) streambank groundwater standpipe (T2); c) Line Creek (T3); and, d) a deep groundwater standpipe screened between 22 – 25 m below ground surface (Jakub Szmigielski, personal communication).

DTS Installation

A fibre-optic DTS network was installed to measure stream and streambed temperature patterns in response to aquifer-stream exchange fluxes. An Sensornet Oryx (Sensornet, U.K.) field deployable DTS with manufacturer stated temperature resolution of 0.01°C and 1 m measurement resolution was installed downstream of the Line Creek rock drain (Figure 1). The DTS system used a 320 m long BruSens 150 (Brugg, Switzerland) fibre-optic cable to measure the stream temperature over a six day period.

The cable was laid from the downstream to upstream end with the cable length reading 0 m at the DTS unit, 160 m at the reach end before looping back, and 320 m at the downstream reach starting point. The first section (hereafter termed 'bed temperature'), stretching downstream to upstream, was installed at an average depth of 5 cm within the streambed and about 0.5 m from the right-bank edge to directly measure streambed temperature. The second section (hereafter termed 'stream temperature'), looping back from upstream to downstream, was installed approximately 10 cm below the surface of the water column near the middle of the stream to measure stream temperature through the water column and reduce the influence of solar heating on the cable. The cable was positioned in the most turbulent section of flow to measure a more composite stream temperature and was secured by rebar driven into the streambed approximately every 5 – 10 m depending on the morphology of the stream. Rebar spacing was as close as 5 m in riffles or more turbulent sections, and as great as 10 m in pools. The cable was attached to the rebar with plastic cable ties to avoid preferential heat conduction along the metal rebar. Prior to stream installation, the cable was rolled out and positioned along the streambank to reduce any high tensile stress. Temperature measurements from the initial 9 m and 14 m from the bed and stream downstream cable sections, respectively, were not considered since several meters were exposed directly to the atmosphere. The initial 10 m from the upstream section was also excluded as a result of having several exposed wound cable coils (Photo 3). The DTS system was powered using AC power from the electrical grid. At the end of data collection, the cable was inspected along its length to check for any movement or areas where the cable may be above the water surface.

Calibration Procedure

Single-ended or double-ended measurements can be collected with the DTS instrument. Double-ended measurement was chosen for this study as this method only requires a temperature offset for calibration. During calibration, the built-in DTS unit routines were used to quantify differential attenuation. Calibration was conducted following the manufacturer's specifications by placing a 45 m section of wound fiber-optic cable in a calibration bath comprised of a circular container of 1 m diameter and 0.4 m height filled with a water and ice slurry, and measuring four 10 minute integration data sets. In an effort to maintain a constant temperature, the calibration bath was set up in an enclosed building during the procedure. Post-processing of calibration data indicated that thermal stratification in the ice bath likely occurred due to limited mixing during the calibration period. A revised calibration technique is recommended that includes better mixing of the calibration bath and longer integration times.

In order to compensate for the issues encountered in the laboratory calibration bath procedure, a field calibration was conducted based on a spatial and temporal repeatability analysis of collected field DTS temperature measurements. For spatial repeatability, or how well all the points at one time represent the true value of a constant temperature system, the mean of all measurements from 140 m to 150 m (n = 12) at minimum temperature (i.e. no influence of solar heating; 07:00 hr) were calculated for the bed (5.10 ± 0.01 °C; ± one standard error, SE) and stream (5.11 ± 0.01 °C; ± SE). For temporal repeatability, or how well the instrument measures a constant temperature at successive and consecutive times, temperatures in the early morning were assumed to be constant. The mean temperature at 150 m on the FO cable (closest to reference logger temperature T1) was determined from 04:00 hr to 07:00 hr (n = 10) for the bed (5.14 ± 0.01 °C; ± SE) and stream (5.16 ± 0.01 °C; ± SE). This exercise provided confidence in the certainty of the DTS field measurements. The resolution of temperature measurements depend on the DTS installation, integration time, and calibration method (Selker et al., 2006b; Tyler et al., 2009). Although the standard error of 0.01°C from field data provided confidence in the precision of the DTS data, a more detailed assessment of the precision and accuracy of the DTS measurements could be completed by applying improved calibration procedures described above.

Data analysis

Preliminary DTS data analysis focused on determining anomalies in temperature along the study reach as outlined in Kraus et al. (2012). The strength of the temperature anomaly (A) provided a measure of spatial variability in lower temperature areas via:

$$A(x_i) = T(x_i) - \overline{T(x_i)} \tag{1}$$

where x_i is measurement locations along the cable. Temporal variability of stream temperature measurements was indicated by their standard deviation:

$$SD\ A(x_i) = \sqrt{\frac{1}{n}\sum_{t=1}^{n}\left(T(x_i) - \overline{T(x_i)}\right)^2} \tag{2}$$

It is expected that the temperature of groundwater upwelling would be lower than in surface water during fall (i.e. warm daytime surface water temperatures) meaning negative temperature anomalies would be expected as a result of cooler groundwater inputs. In addition, Krause et al. (2012) found that groundwater upwelling was associated with increased temporal variability meaning a greater standard deviation was associated with an area of lower temperature.

RESULTS AND DISCUSSION

Hydrology and Meteorological Conditions

Air temperatures measured at the meteorological station indicated a period of warming, then cooling, during the DTS study within an overall monthly cooling trend in September (Figure 2). Diurnal air temperature varied with maximum day and night temperatures ranging from 14 to 2 °C during the study period. A total of 15 mm of rain fell during the experiment (Figure 3). Streamflow measured upstream of the study area installation generally increased from 0.55 to 0.58 m^3/s (Figure 3). Flow entering Line

Creek from the historic settling ponds through the diversion culvert made up ~5-10% of Line Creek streamflow during the study period (Photo 4).

DTS Dataset

Stream temperature measurements measured over a six day period in September, 2013 show that FO DTS was successful in identifying areas of cooler water temperature. The placement of the cable within the streambed sediments allowed for detection of localized cooling compared to the well-mixed stream temperature zone of the stream cable (Figure 4). Bed and stream temperatures were measurably different in some locations, with maximum differences of 0.12 °C and 0.23 °C at 104 m during the night (07:00 hr) and day (15:00 hr), respectively (Figure 5 and 6). The temperature differences measured at 104 m coincided with the location of vegetation on the streambank and an observed seep (Photo 5). Site personnel confirmed that this section of stream does not ice over in the winter. The area of distinctly cooler bed temperatures is more apparent in Figure 7, which compares minimum and maximum values standardized to the spatial mean along the study reach. The input of the relatively cool WLC diversion flow between 130 and 140 m is apparent in the decreased stream and bed temperatures. The diverted WLC inflow is assumed to be well mixed with the Line Creek flow shortly after entering Line Creek and to be independent of the cooling trend measured approximately 30 m downstream at 104 m.

Step-like changes occurred in the maximum day (15:00 hr) streambed temperatures from approximately 115 m to 95 m (Figure 6). A warming trend from 115 m to 110 m changed to a cooling trend from 110 m to 104 m, followed by a warming trend from 104 m to 95 m. The warming trend in streambed temperature from 115 m to 110 m was attributed to solar heating during the day, as this warming trend was not observed in the minimum (07:00 hr) streambed temperatures (Figure 5). The cooling trend from 110 m to 104 m was attributed to upwelling of relatively cool groundwater. This cooling trend was not likely the result of radiative or conductive heat loss given the short distance over which it occurred. Additionally, the cooling trend was not likely the result of shading or differences in streambed cable depth as streambank vegetation, channel morphology, and water depth were similar in the areas of both cooling and warming. The absence of the cooling trend from 110 to 104 m in the minimum nighttime streambed temperatures was attributed to the relative similarity of streambank groundwater and streambed temperatures at night compared to the late afternoon. Similar step-like changes in streambed temperatures have been recorded in the literature and were associated with groundwater upwelling (e.g. Selker et al., 2006a).

Stream and bed temperatures gradually increased downstream of the identified seep likely due to energy input from solar radiation. The effect of solar heating on downstream temperatures may result in signal reduction of other upwelling areas (Lautz et al., 2013). Standardized bed temperature values indicate small negative temperature anomalies at approximately 65 and 80 m (Figure 7), which may be other potential zones of groundwater discharge; however, a more detailed energy balance taking into account solar radiation (including shading effects), longwave radiation, streambed conduction, latent heat, and sensible heat would be required to confirm this. In addition, installation of streambank piezometers to measure vertical hydraulic gradients and to sample groundwater for chemical analyses would be required to confidently support or refute this hypothesis.

Localized groundwater discharge to a stream has been estimated in the literature with a simple mixing model using temperature as a conservative tracer given the significant difference between groundwater and stream temperatures (e.g. Lauer et al., 2013). Groundwater inflows dampen the diurnal cycles of stream temperature as perennial groundwater temperatures remain relatively stable (Constanz, 2008). In September, differences between groundwater and stream water may be less distinct than would be expected during more pronounced diurnal surface water temperatures from June to August. The opposite effect would occur during the winter, meaning that groundwater inflows would appear as areas of warmer temperature as surface temperatures are typically cooler. During the experiment, temperature differences measured between groundwater from the nearby standpipe piezometer and from the stream ranged from 1.1 – 2.2°C. However, less difference (0.1 – 1.1 °C) was measured at the groundwater-stream interface when water from the streambank (obtained from a groundwater standpipe installed at the assumed interface) was compared to the stream water temperatures. DTS studies should be completed in late summer when the difference between streambank groundwater and stream temperatures is expected to be largest and more pronounced cooling trends in areas of groundwater upwelling would also be expected, resulting in improved ability to identify fluxes.

CONCLUSIONS

A DTS study was successful in determining the presence of a groundwater discharge zone into an alpine stream. The measurement of cooler groundwater discharge in the stream bed relative to the stream was corroborated by the presence of well-established riparian vegetation, a streambank seep and observations that ice does not form in that area in the winter. This study was intended as a verification of methods for in-stream DTS installation and measurements. The methods employed are recommended for DTS streamflow work with recommendations for improved calibration procedures that include longer integration time and better mixing of the calibration bath. Late summer is also recommended for DTS studies when the difference between streambank groundwater and stream temperatures is expected to be largest and more pronounced cooling trends in areas of groundwater upwelling would be expected, resulting in improved ability to identify fluxes.

REFERENCES

Constantz, J. (2008) Heat as a tracer to determine streambed water exchanges. Water Resour. Res. 44: W00D10.

Lauer, F., H-G. Frede, and L. Breuer (2013). Uncertainty assessment of quantifying spatially concentrated groundwater discharge to small streams by distributed temperature sensing. Water Resour. Res. 49: 400-407.

Kraus, S., T. Blume, and N.J. Cassidy (2012). Investigating patterns and controls of groundwater up-welling in a lowland river by combing fibre-optic distributed temperature sensing with observations of vertical hydraulic gradients. Hydrol. Earth System. Sci.16: 1775-1992.

Neilson, B.T., C.E. Hatch, H. Ban, and S.W. Tyler (2010). Solar radiative heating of fiber-optic cables used to monitor temperatures in water. Water Resour. Res. 46: W08540.

Selker, J., N. Van de Giesen, M. Westhoff, W. Luxemburg, and M.B. Parlange (2006a). Fiber optics opens window on stream dynamics, Geophys. Res. Lett. 33: L24401.

Selker, J., L. Thevanaz, H. Huwald, A. Mallet, W.Luxemburg, N. Van de Giesen, M. Stejskal, J. Zeman, M. Westhoff, and M.B. Parlange (2006b). Distributed fiber-optic temperature sensing for hydrologic systems. Water Resour. Res. 42: W12202.

Szmigielski, Jakub. 2014. Personal communication. M.Sc. Thesis Candidate. University of Saskatchewan, Saskatoon, Saskatchewan, Canada.

Tyler, S.W., J.S. Selker, M.B. Hausner, C.E. Hatch, T.Torgersen, C.E. Thodal, and S. G. Schladow (2009). Enviromental temperature sensing using Raman spectra DTS fiber-optic methods. Water Resour. Res. 45: W00D23.

Westhoff,M, H.H.G. Savenije, W.M. Luxemburg, G.S. Stelling, N.C. Van de Giesen, J.S. Selker, L. Pfister, and S. Uhlenbrook (2007). A distributed stream temperature model using high resolution temperature observations. Hydrol. Earth Syst. Sci. 11(4): 1469-1480.

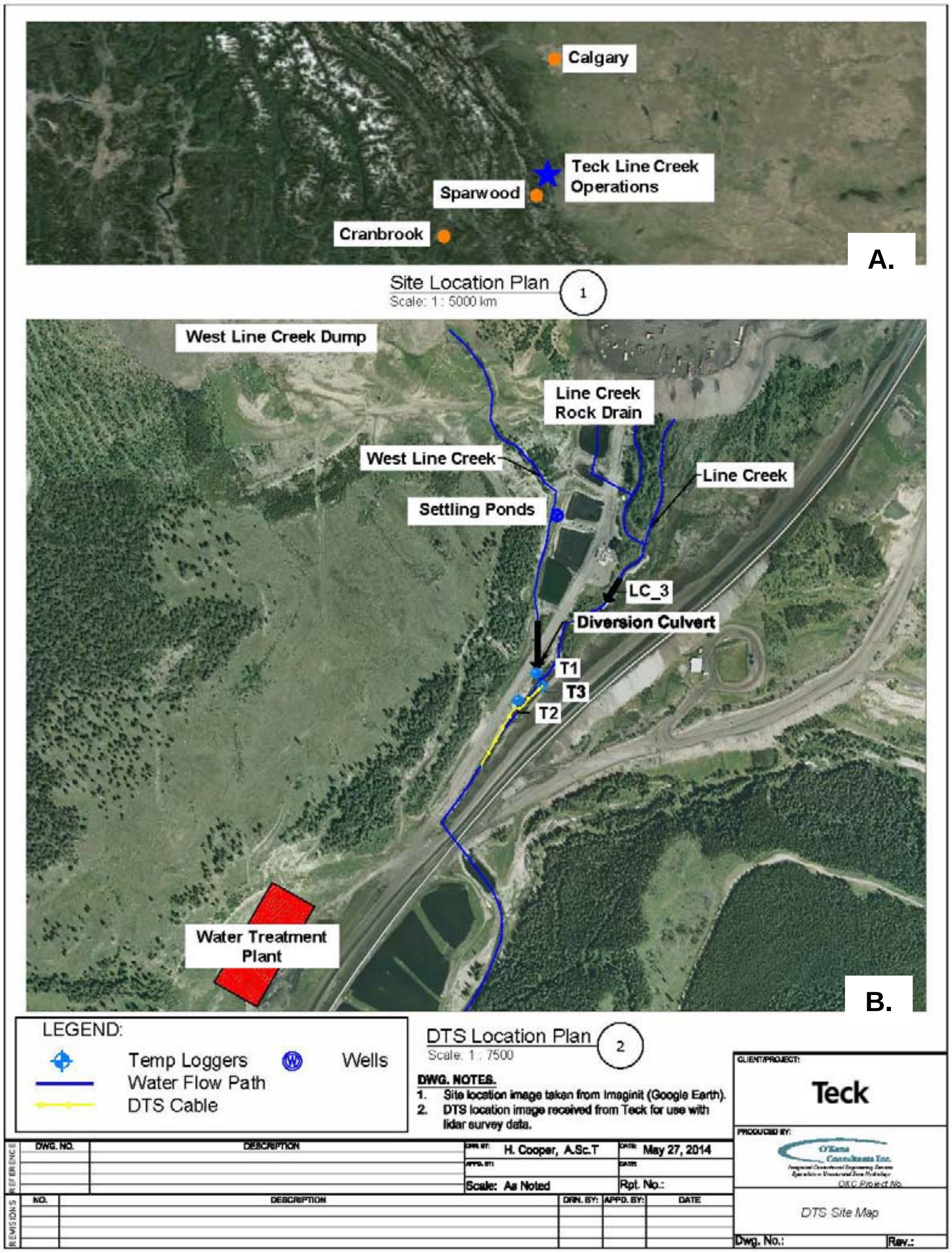


Figure 1 Maps (Google Earth) of Teck Resources Ltd Line Creek Operations (A.) showing Line Creek study reach, DTS installation, as well as other field instrumentation (B.). Regional groundwater flows in a southeasterly direction toward Line Creek from the West Line Creek dump area (Jakub Szmigielski, personal communication).

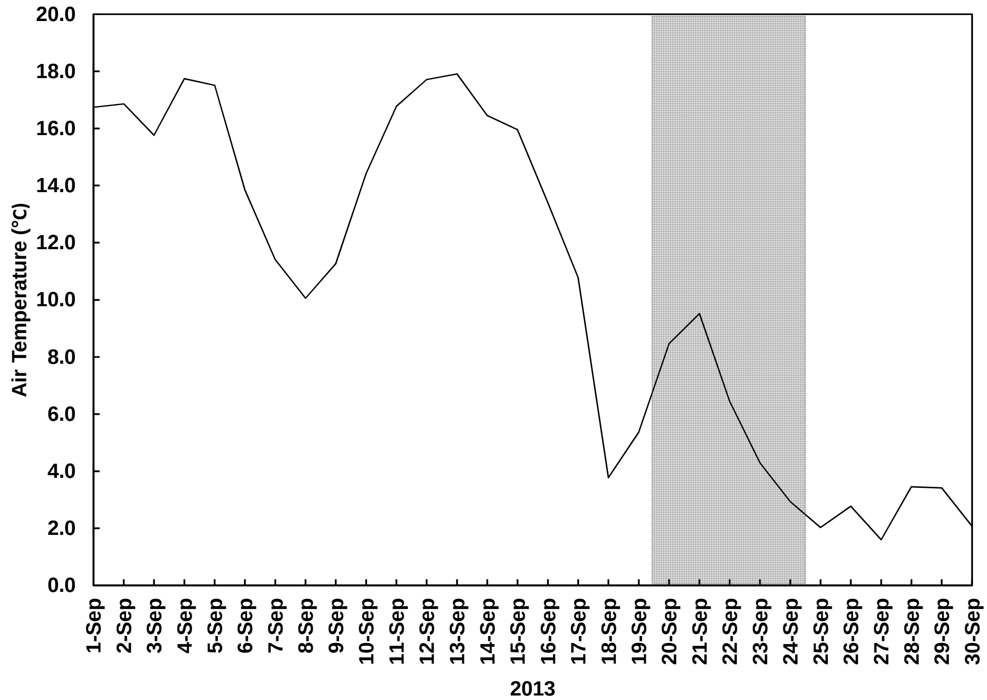


Figure 2 Mean daily air temperature measured at meteorological station for September, 2013. Grey bar indicates study period.

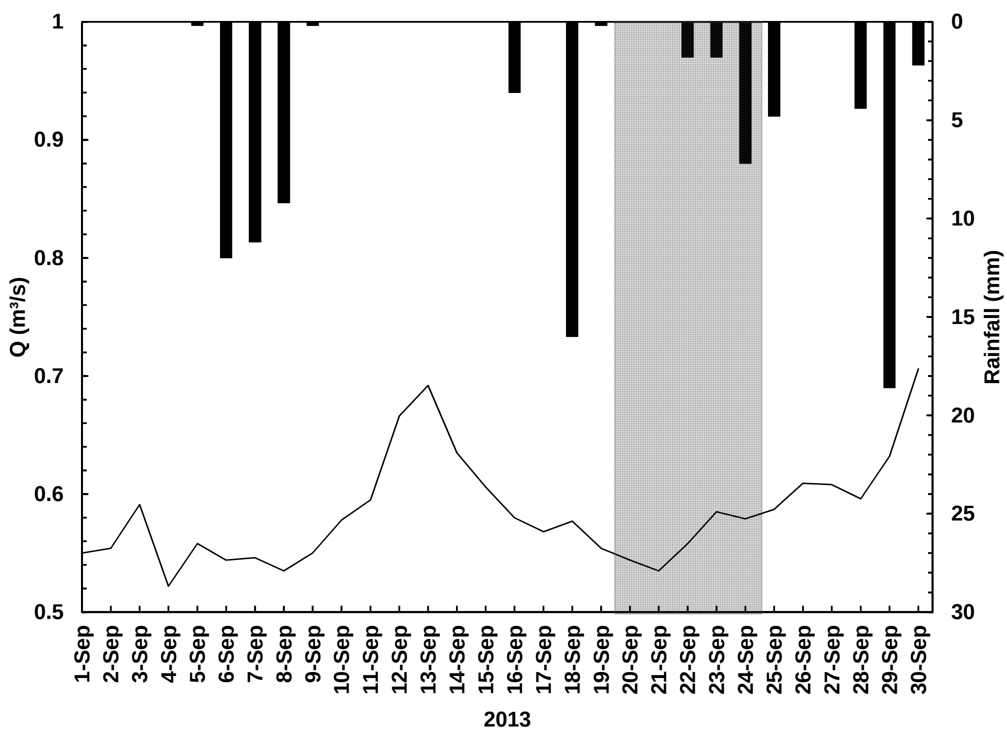


Figure 3 Daily mean discharge (Q) measured at LC_3 stream gauging station located upstream of the study reach and total daily rainfall measured at the meteorological station during September, 2013. Grey bar indicates study period.

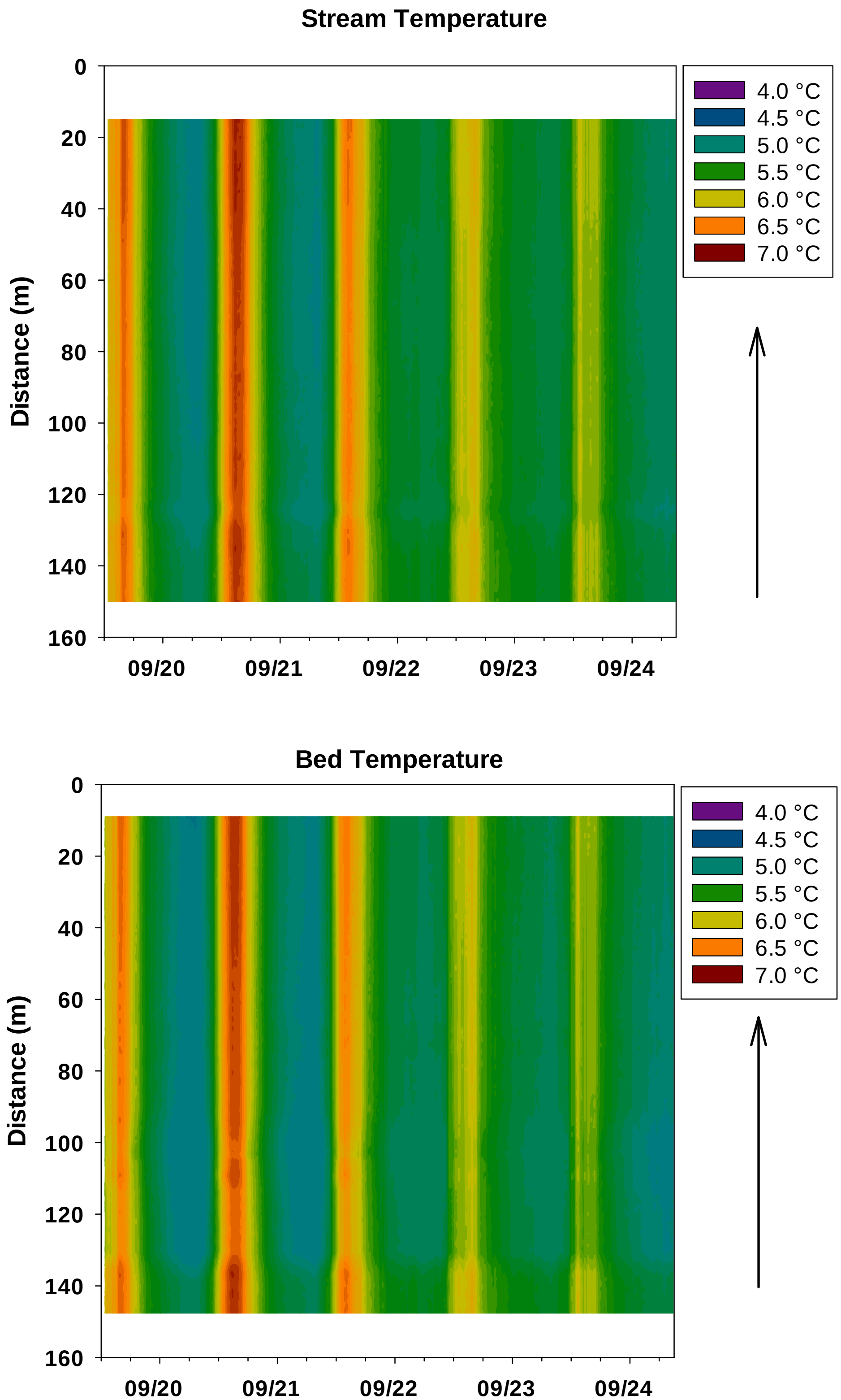


Figure 4 Stream and bed temperature (°C) measured at 1 m spatial resolution and 20 min sampling intervals over 24 hours along the length of the study reach starting September 19th 12:50 to September 24th 08:50, 2013. Black arrows indicate direction of stream flow.

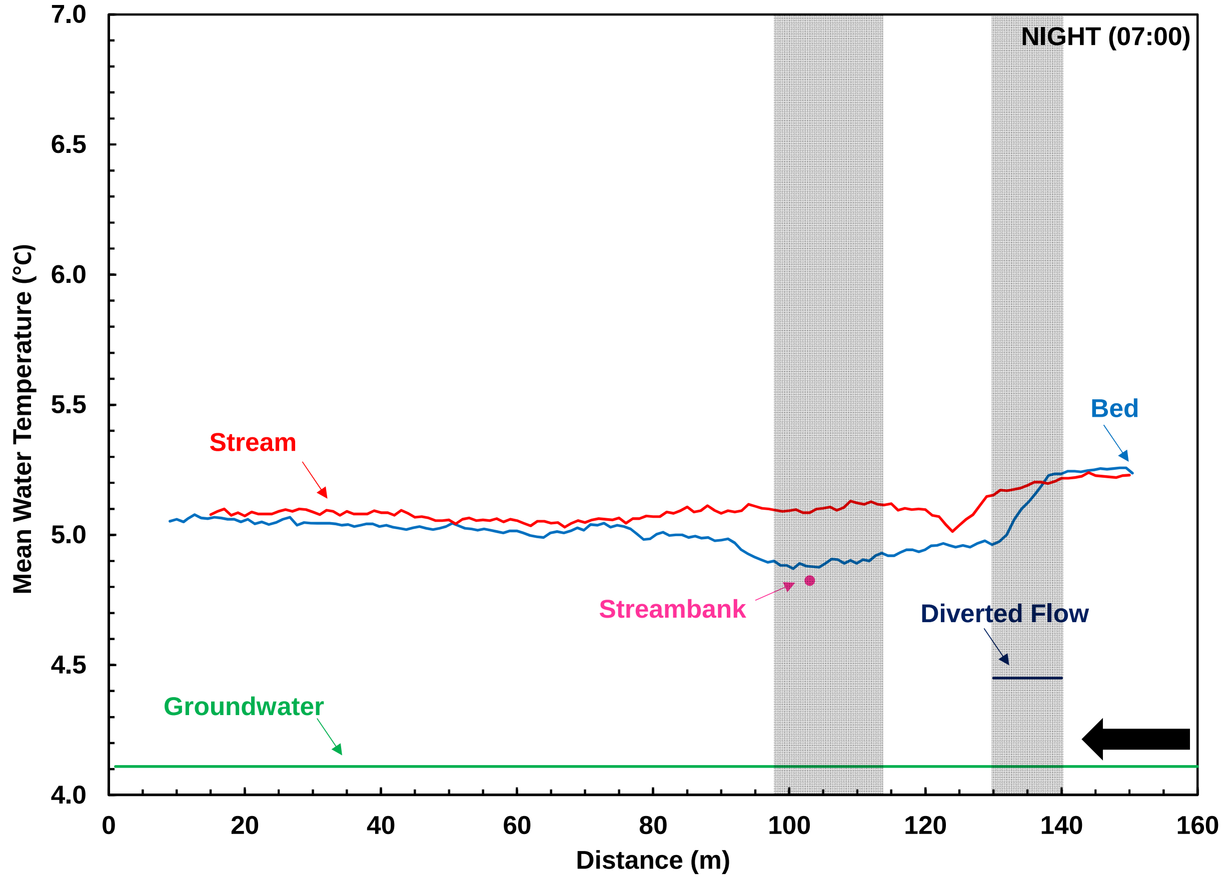


Figure 5 Average minimum stream temperature over four days (September 20th to 23rd, 2013) measured with DTS in the water column (stream) and streambed. Water temperature of groundwater (WLC 12_12 standpipe), streambank groundwater (standpipe), and the WLC diversion flowing into Line Creek were measured with dataloggers. The left vertical grey bar is centered on a cooling trend in streambed temperature (moving in the downstream direction). The right vertical grey bar marks the zone where diverted WLC flows into Line Creek. The black arrow indicates direction of stream flow in Line Creek.

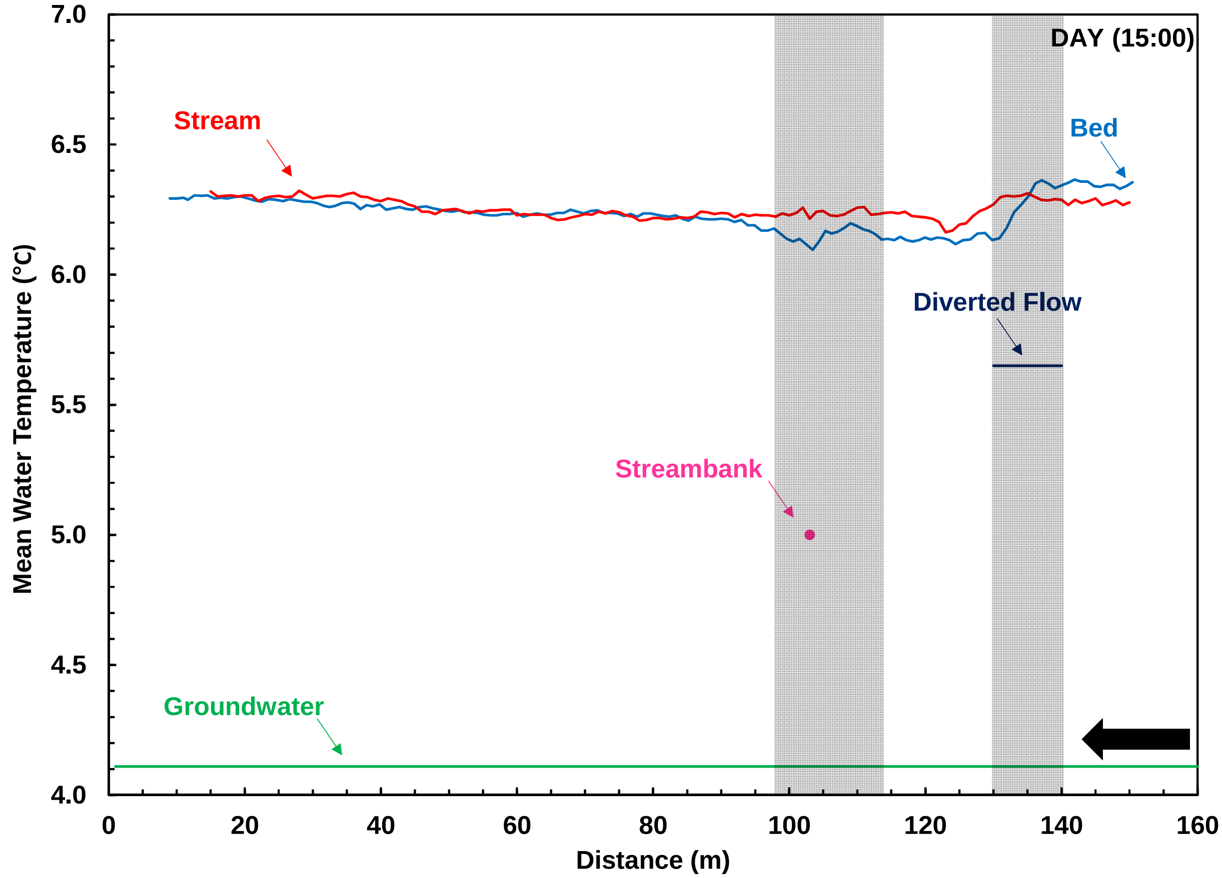


Figure 6 Average maximum stream temperature over four days (September 20th to 23rd, 2013) measured with DTS in the water column (stream) and streambed. Water temperature of groundwater (WLC 12_12 standpipe), streambank groundwater (standpipe), and the WLC diversion flowing into Line Creek were measured with dataloggers. The left vertical grey bar is centered on a cooling trend in streambed temperature (moving in the downstream direction). The right vertical grey bar marks the zone where diverted WLC flows into Line Creek. The black arrow indicates direction of stream flow in Line Creek.

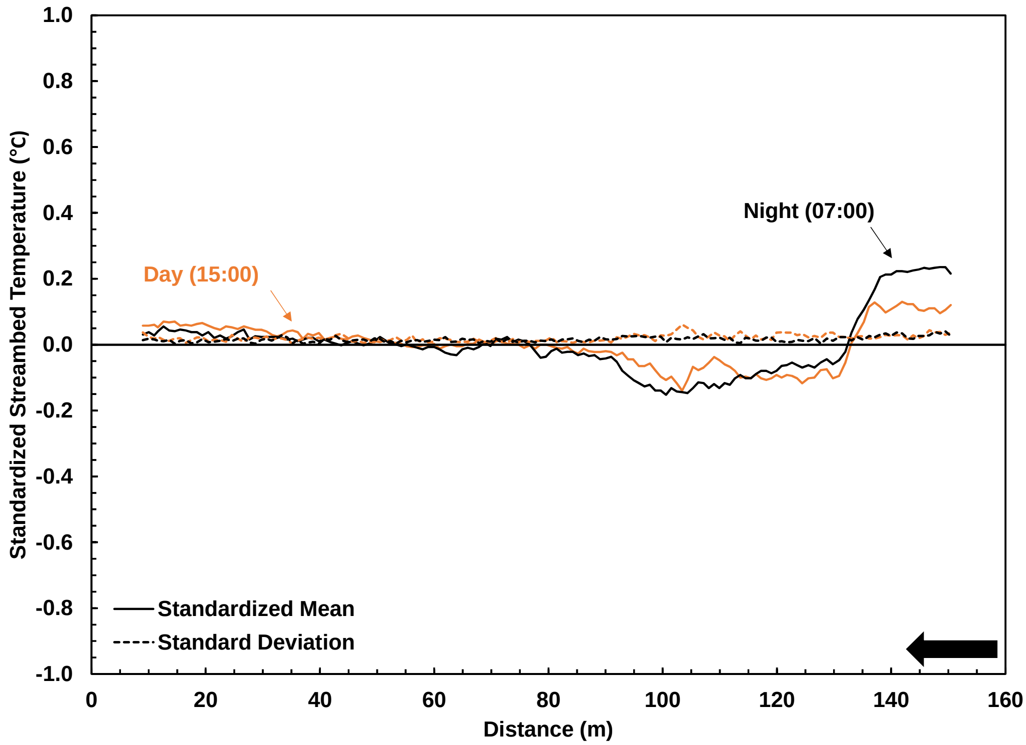


Figure 7 Temporal and spatial variability of DTS streambed temperatures for four days (September 20th to 23rd, 2013) during minimum (07:00 hr) and maximum (15:00 hr) periods showing mean (solid line) and standard deviation (dotted line). Values were standardized by subtracting the spatial mean at each sampling date. The black arrow indicates direction of streamflow.

Photo 1 Looking downstream at coarse-textured streambed and steep banks of Line Creek.

Photo 2 Location of streambank standpipe piezometer upstream of observed streambank seep and adjacent to the well-established riparian vegetation at the toe of the streambank.

Photo 3 Coils of DTS cable exposed to the atmosphere.

Photo 4 Flow entering Line Creek from the West Line Creek settling ponds through the diversion culvert.

Photo 5 Area of identified cooler groundwater discharge zone with the in-stream DTS measurements (~104 m). The photo is centered at approximately 105 m. Stream flow is from right to left.

DESIGN OF *IN-SITU* WATER TREATMENT OF ACID CONTAMINATED LAKE

Soren Jensen, P.Eng.[1]
Tom Sharp, P.Eng.[1]
Diane Allen, P.Eng.[2]
Ian Wilson.[2]

[1]SRK Consulting (Canada) Inc.,
Vancouver, Canada

[2]Saskatchewan Research Council,
Saskatoon, Canada

ABSTRACT

An in-situ treatment system was designed for Nero Lake, an acid contaminated lake in northern Saskatchewan. The purpose of the treatment system was to neutralize the lake water, which would be accomplished by adding lime to the entire lake. The process design was based on a pilot trial and a water and load balance model study. The pilot trial showed that lime could be delivered to the lake as a dilute lime solution to 10% to 20% of the lake water without significant loss of lime utilization and allowing the natural seasonal turnover of the lake to mix the lime throughout the lake. In-line injection of lime resulted in similar performance as mixing lime slurry and lake water in a mix tank.

A water and load balance model was developed to determine whether addition of excess alkalinity (lime) was warranted as a way of mitigating acidity that may report to the lake in the future. The model was developed as a simple spreadsheet based model. The modelling results showed that since 1978, the annual load of alkalinity to Nero Lake has been greater than the annual load of acidity. This means that the lake will not become acidic following neutralization. The model also showed that any reduction in the current acidity loadings reporting to the lake is unlikely to significantly affect the long-term water quality of Nero Lake.

KEY WORDS
In-situ treatment, lime treatment, lake mixing, load balance, water balance

INTRODUCTION

The Lorado site is an abandoned uranium milling operation located on the western shore of Nero Lake south of Uranium City in northern Saskatchewan. The mill was constructed in 1957 and operated until 1960. Tailings and acidic waste produced by the milling operation were deposited near the western shore of Nero Lake and some tailings were submerged in the Lake (Figure 1). Deposition of milling waste into the lake acidified the water and elevated concentrations of dissolved metals. The mine and mill sites were vacated in 1960 with little or no decommissioning. It wasn't until the 2000s that the provincial and federal governments took stock of the safety and environmental liabilities at the site and made a commitment to deal with the potential human health and environmental risks (Golder 2013).

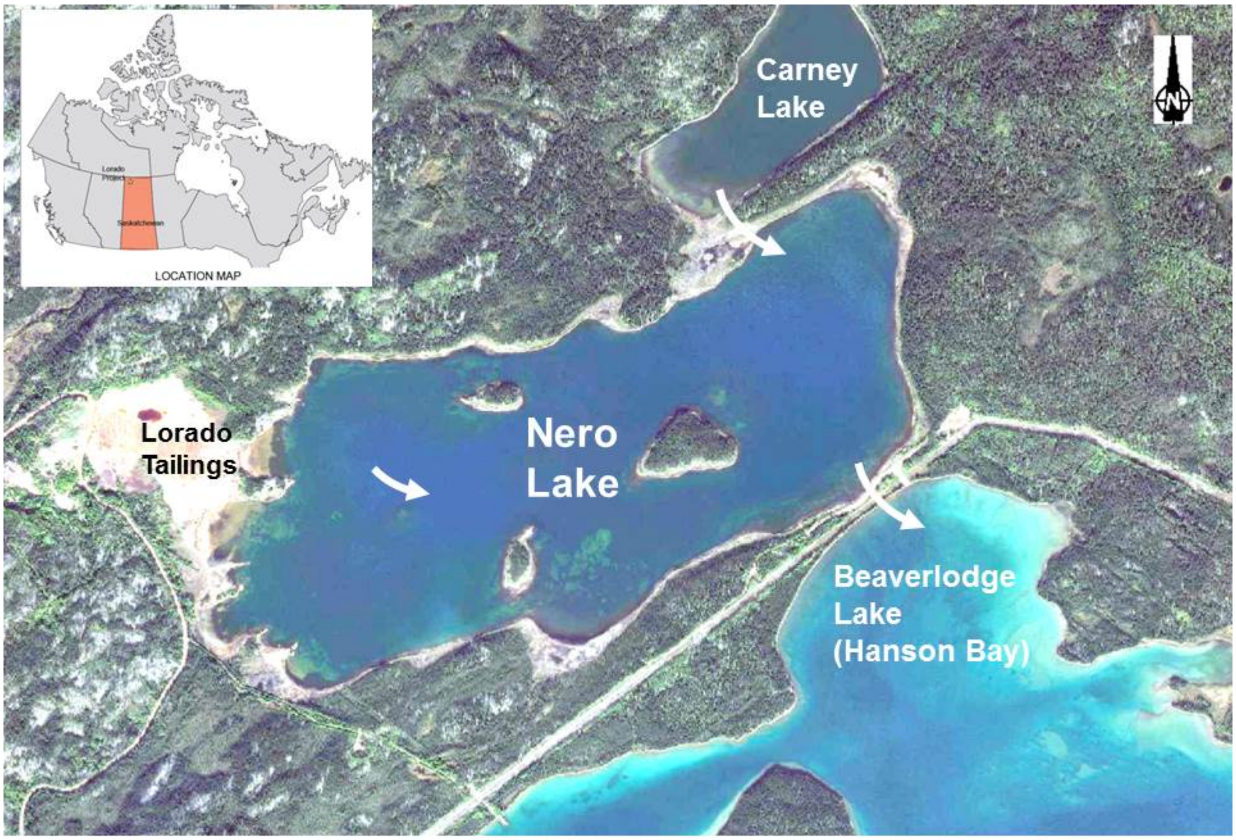


Figure 1 Nero Lake flows to Beaverlodge Lake (Hanson Bay). Carney Lake is located upstream of Nero Lake.

Although not abandoned, the Lorado site was neglected as ownership changed hands, ending up as property of Conwest Exploration Company Ltd. This company oversaw the demolition of the mill buildings in 1990. When the parent company of Encana Corporation bought Conwest, the Lorado site was transferred as well. In 2007, the responsibility for the Lorado site was transferred to the Saskatchewan Ministry of Economy along with funds for site remediation.

The ministry retained the Saskatchewan Research Council (SRC) to manage the development of a Risk Reduction Plan and an Environmental Impact Statement (EIS) for the Lorado Site. The EIS for the Lorado Site were completed in 2013 (Golder 2013). The overall risk management goal identified for the Lorado site was that “risk reduction at the site has achieved an acceptable level of residual risk to human health, and to terrestrial ecological populations, and to the aquatic population of Hanson Bay and Beaverlodge Lake”. The main reclamation activities proposed under the risk reduction plan included:

- Covering of surface tailings in-place, and
- In-situ batch treatment (lime neutralization) of Nero Lake (Golder 2011a).

In 2013, SRC retained SRK Consulting to complete the detailed design of a capillary break cover for the surface tailings and process design for an in-situ batch treatment system for Nero Lake. This paper describes how the results of the batch treatment pilot trial were used along with a water and load balance to develop a design basis and ultimately the process design for water treatment of Nero Lake.

TREATMENT OBJECTIVE AND DESIGN BASIS

Nero Lake has a volume of approximately 11 million m^3. The lake is relatively shallow with an average depth of 6.4 m. The total catchment area, including the Carney Lake sub-catchment located north-east of Nero Lake, is approximately 5 km^2. The deposition of tailings and acidic waste to Nero Lake during the operation of the Lorado milling operation left the lake with acidic water with relatively high concentrations of dissolved metals and sulfate. In 2013, the pH of the lake water was approximately 4.2 and the average acidity concentration was 18 mg/L as $CaCO_3$. Dissolved metals of concern included aluminum and uranium.

The risk reduction measure identified in the EIS was to neutralize the water in Nero Lake using lime (calcium oxide). Accordingly, the treatment objectives for Nero Lake were to neutralize the lake water and minimize or eliminate requirements for long-term water treatment. Long-term treatment would be required if future acidity loadings from the tailings area were sufficiently greater than alkalinity loads to lake from runoff.

The design basis for the in-situ treatment process was based on a water treatment pilot trial completed in July of 2013 and the results of a water and load balance model. The purpose of the pilot trial was to determine the lime dose required to neutralize the lake and to provide a basis for sizing equipment and defining other process parameters. The purpose of the modelling study was to determine if (and if so, when) future acidity loadings from the tailings area would once again acidify the lake and therefore necessitate another treatment campaign and to determine the natural state of the Nero Lake and downstream environment.

PILOT TRIAL METHODOLOGY

A schematic of the initial treatment concept for Nero Lake is shown in Figure . Quick lime is fed to a lime slaker that hydrates the lime and produces lime slurry. The lime slurry is transferred to a storage tank. Untreated lake water is pumped to the shore where it is mixed with lime slurry in a mix tank. Neutralized water is then returned to the lake.

The purpose of the pilot trial was to inform the full scale design by:

- Determining the lime dose required to neutralize the lake water;
- Assessing the effect of neutralization of Nero Lake water quality;
- Evaluating the long-term stability of the precipitated solids, and
- Evaluating the effect of mixing on lime utilization.

The pilot trial used limnocorrals submerged alongside a temporary dock as a pilot-scale analog for Nero Lake (Figure 2). Limnocorrals are large cylindrical plastic bags made of impermeable high density polyethylene (HDPE) liner (Figure 3). Floats are attached to the top of the liner to provide buoyancy and structural support and chains are attached along the bottom to anchor the corral. The limnocorrals used for the Nero Lake pilot trial had closed bottoms to permit the collection of precipitates formed in the

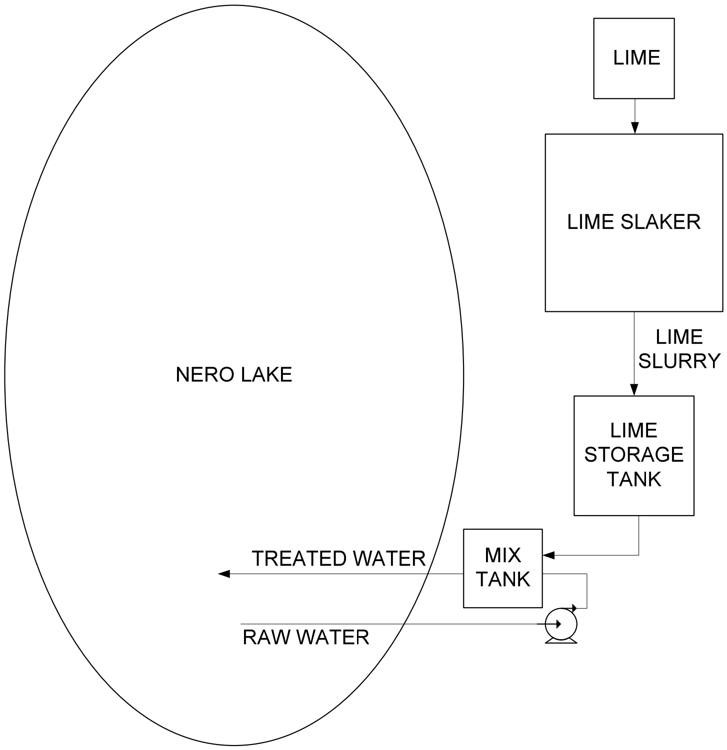


Figure 2 Schematic of Nero Lake Treatment Concept

Figure 3 Limnocorrals Used for the Nero Lake Water Treatment Pilot Trial

treatment process. The use of limnocorrals for piloting in-situ water treatment ensured that temperature, depth and operating conditions are reasonably representative of full-scale conditions. Figure 4 shows a schematic of the pilot trial configuration. For each of the trial runs conducted, the limnocorrals were initially filled with untreated lake water. The untreated water was then pumped to a mix tank on shore

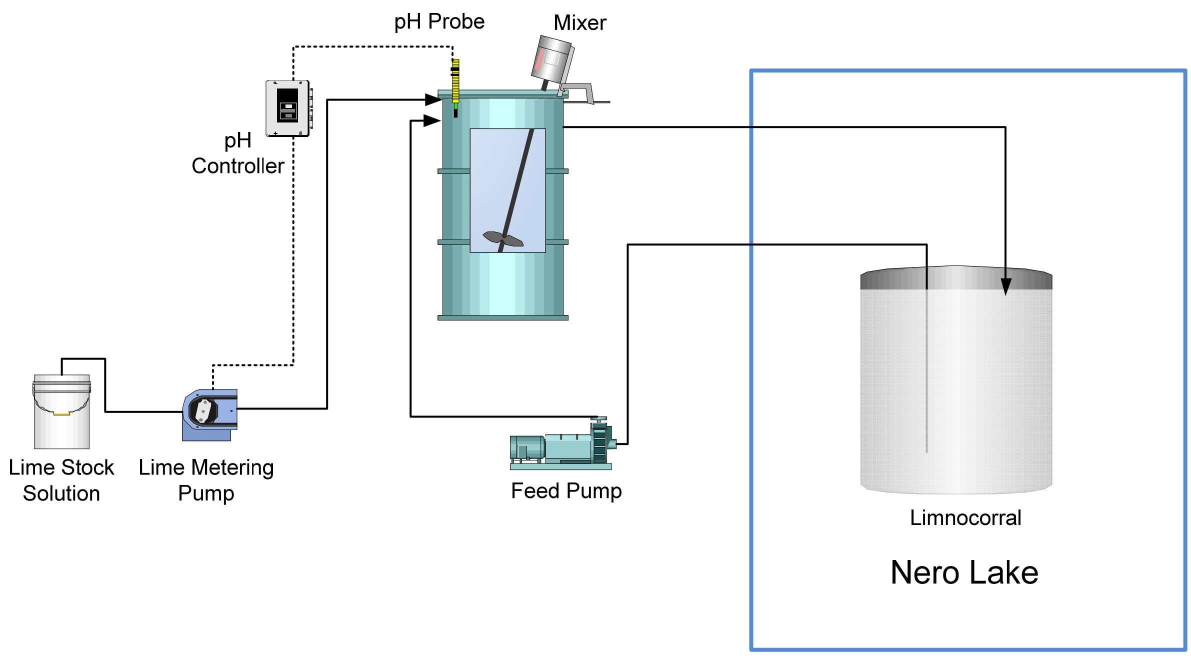


Figure 4 Pilot Trial Schematic

where lime was added to neutralize the water. The neutralized water then flowed back to the limnocorrals where it would mix with untreated water. The nominal capacity of the pilot plant was 60 L/min (16 gpm), which corresponded to a hydraulic retention time of 8 hours for one limnocorral.

WATER AND LOAD BALANCE MODELLING APPROACH

The purpose of the water and load balance model was to determine if acidity loadings from the tailings area were likely to acidify the lake after the treatment campaign. The model relied on information from past studies (Golder 2011b). Inputs included historical water quality data, catchment delineations and hydrology and meteorology baseline data. The modelling approach focused on quantifying annual acidity and alkalinity loadings flowing to Nero Lake. If loadings of acidity consistently are greater than loading of alkalinity, the lake will become acidic over time even if the water is treated/neutralized. Conversely, if the annual alkalinity load to the lake exceeds the annual acidity load the lake will not become acidic after neutralization.

A mass loading model was developed to assess alkalinity and acidity loads to the lake. Figure 5 shows a schematic of the conceptual load balance model. Inflows of alkalinity to the lake include local surface runoff, runoff from Carney Lake and groundwater reporting to the lake.

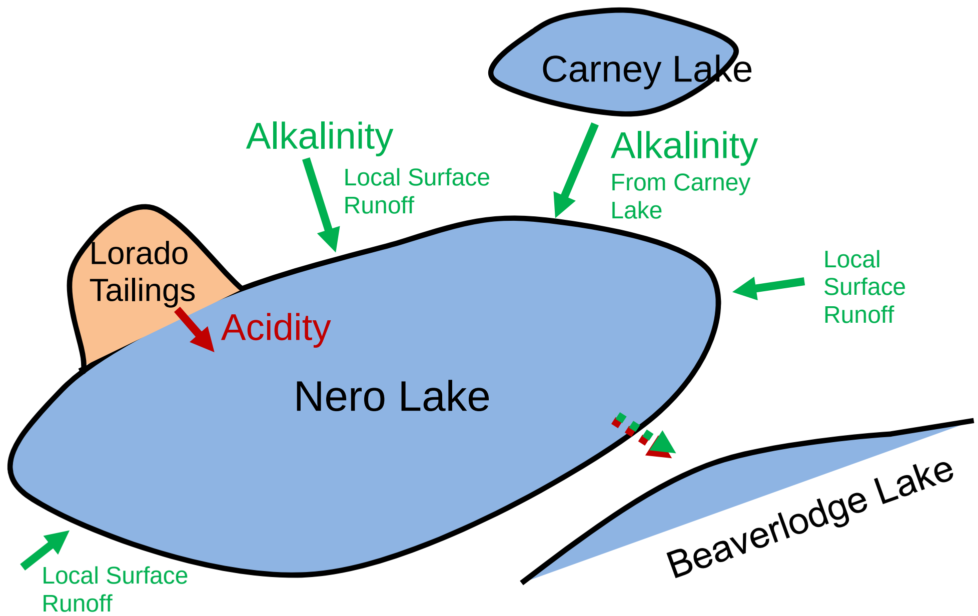


Figure 5 Schematic of the Conceptual Acidity and Alkalinity Load Model for Nero Lake

For simplicity, the surface runoff and groundwater inflow was combined into a single inflow term. Acidity loadings were attributed to flow from the Lorado tailings. Acidity loadings were treated as a single bulk term. No attempt was made to partition loadings into surface loadings from efflorescent salts on the tailings surface and acidity transported with groundwater moving through the tailings beach. Although this distinction is important when planning and designing a cover, it is not important for evaluating the effects of the total loadings on the quality of water in Nero Lake.

Water in Nero Lake flows south though a shallow land bridge to Hanson Bay in Beaverlodge Lake. Historically, the acidic water from Nero Lake has affected the water and sediment quality in Hanson Bay. As the acidic water was neutralized by alkalinity in Beaverlodge Lake, many of the metals dissolved in the Nero Lake water precipitated and settled in Hanson Bay. The elimination of acidity and metal loading to Hanson Bay was one of the risk reduction measures identified in the 2013 EIS.

RESULTS

Pilot Trial Results

Figure 6 shows the results of bench-scale titrations of Nero Lake water using lime. The results demonstrate that approximately 10 mg/L of lime (as CaO) is required to reach a pH of 7 and that about 40 mg/L is required to reach a pH of 10. The bench-scale titration results were used to define the lime dose used for the limnocorral tests.

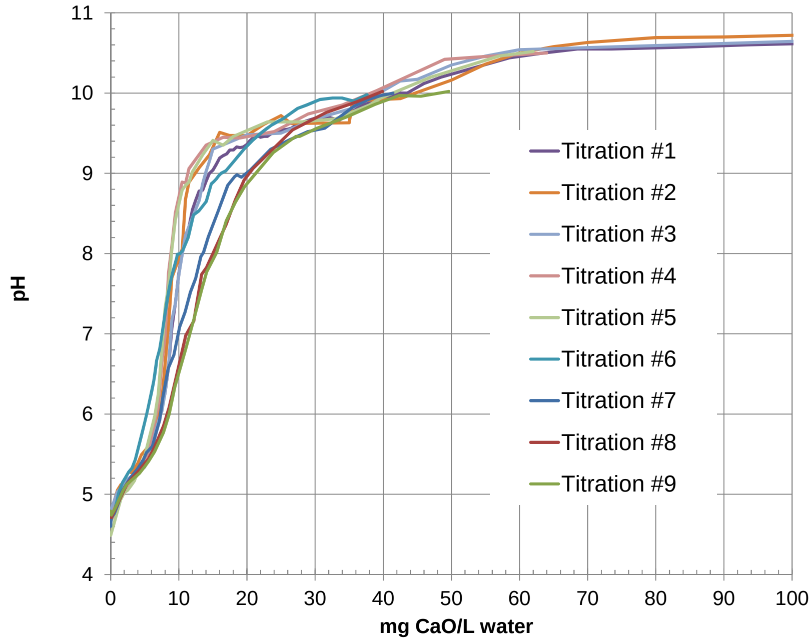


Figure 6 Titration of Nero Lake Water with Lime

Table 1 shows a summary of Nero Lake water quality (untreated) and corresponding Canadian Environmental Quality Guidelines for the Protection of Aquatic Life (CWQG). The comparison of water quality parameter concentrations to the CWQG are included here only as a reference. Treatment/neutralization of water in Nero Lake was intended to improve the lake water quality but was not specifically targeting the CWQG values. In 2013, the water in Nero Lake did not meet the CWQG for pH, aluminum, uranium, zinc and mercury. No other exceedances were noted.

Table 1 Raw Nero Lake Water Quality, July 2013

Parameter	Units	Untreated Water	CWQG
pH		4.3	**6.5 - 9.0**
Aluminum	mg/L	2.1	**0.10**
Uranium	µg/L	20	**15**
Zinc	mg/L	0.043	**0.030**
Mercury	µg/L	0.074	**0.026**

Besides pH, the effect of lime addition on the lake water quality was evaluated by measuring concentrations of the metals listed in Table 1. As lime is added to the lake water and pH increases, the metals listed in Table 1 precipitate as, or co-precipitate with, metal hydroxide. Figure 7 shows the effect of lime dose on concentrations of dissolved aluminum and uranium (log scale) in the limnocorrals 24 hours after the end of treatment. The results show that concentrations of dissolved uranium were reduced significantly at lime doses greater than approximately 20 mg/L. Dissolved aluminum decreased by approximately a factor of 10 at a lime dose of 50 mg/L and was below the CWQG when the lime dose was greater than approximately 75 mg/L.

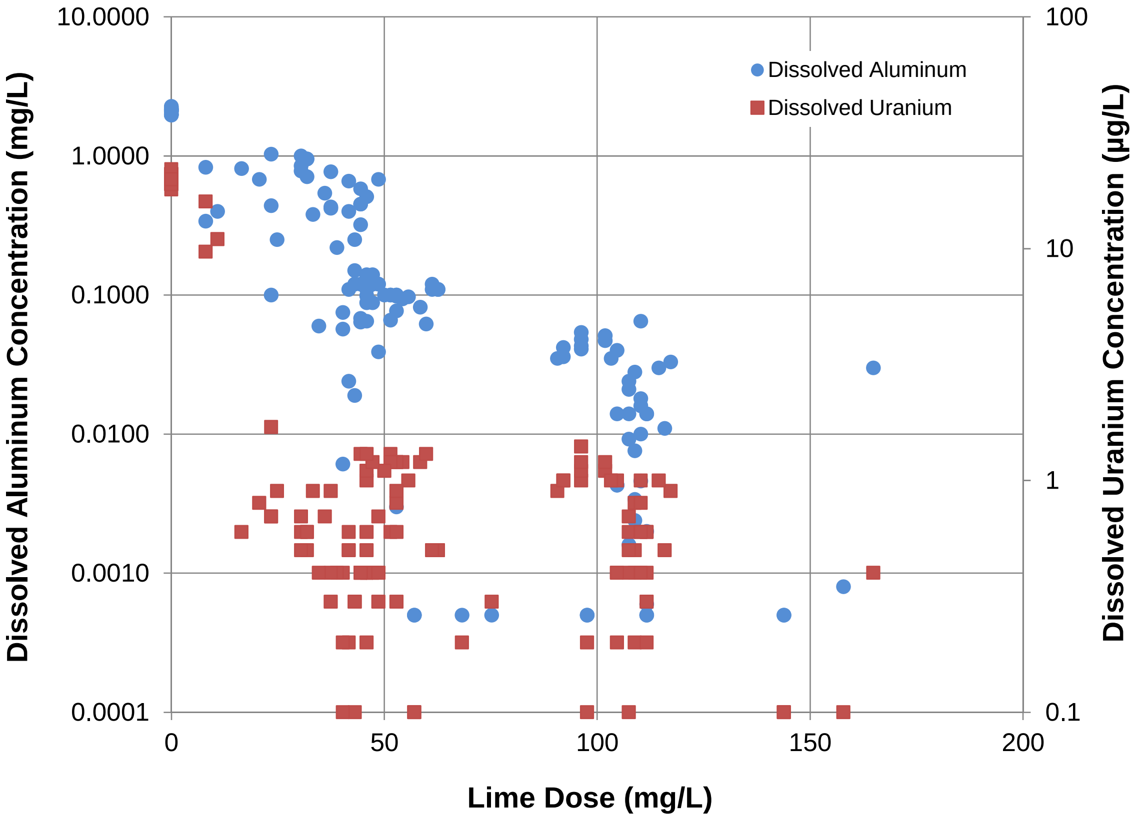


Figure 7 Concentrations of Dissolved Aluminum in Limnocorrals vs. Lime Dose

One objective of the pilot trial was to determine the requirement for mechanical mixing of lime in the lake. If dry lime or concentrated lime slurry were added directly to the lake a significant portion of the lime would likely settle unreacted to the bottom of the lake and would not be immediately available for neutralization. On the other hand, pumping and neutralizing the entire volume of the lake would require large pumps and treatment equipment with correspondingly high energy demand and cost. Therefore, the approach was to add the required quantity of lime to a fraction of the lake water volume, which would then mix with and neutralize the untreated volume.

The question of lime mixing was addressed in the pilot trial by adding the amount of lime required to treat the entire limnocorral to only a portion of the water, and then relying on passive mixing to uniformly mix the lime to the untreated water. Figure 8 shows a summary of the pilot trial results in which 5% to 25% of the limnocorral volume was treated. The results are expressed as dissolved aluminum and uranium concentrations in limnocorral water (post-treatment) vs. lime dose. In untreated lake water, the dissolved aluminum concentration was 2.1 mg/L and the dissolved uranium concentration was 20 µg/L. The results showed that treatment of 10% and 25% of the limnocorral volume resulted in similar performance as

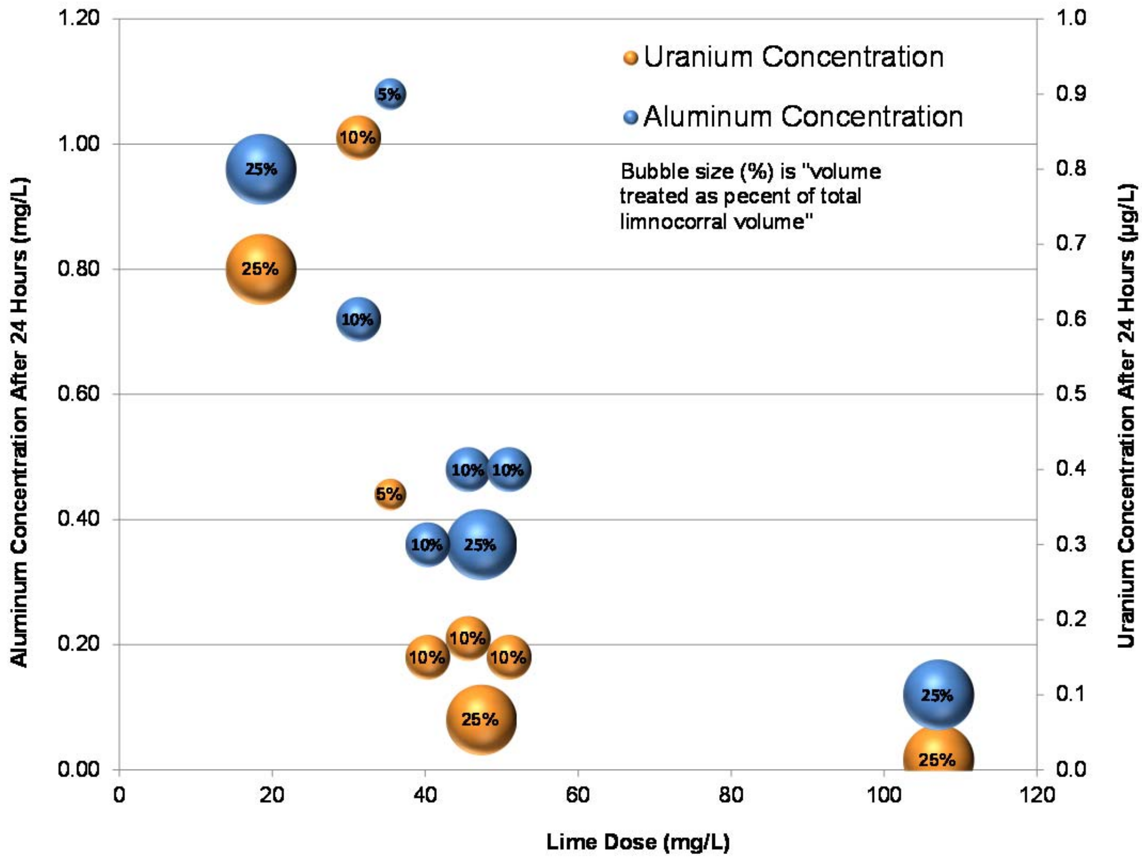


Figure 8 Aluminum and Uranium Concentrations in Limnocorrals vs. Lime Dose and Lake Fraction Treated

treating the entire limnocorral volume, and by inference the lime utilization[1] was similar. However, when the lime dose was added to only 5% of the limnocorral volume, the lime utilization appeared to be reduced. This was corroborated by the presence of settled lime in the bottom of the limnocorral after the trial run. Because of the large difference in scale, it was not possible to evaluate or predict the intensity of full-scale mixing in Nero Lake based on the pilot trial results. The mixing trials were intended to be a one-dimensional (i.e., depth) evaluation of the tendency of lime to settle unreacted to the bottom of the lake. However, depth profiles of temperature, conductivity, pH and Oxidation-Reduction Potential collected at several stations throughout the lake indicated that Nero Lake was well mixed, both laterally and vertically in the summer.

The effect of reactor (mix tank) residence time was also evaluated in the pilot trial. The results of the tests showed that lime utilization was similar when lake water was mixed with lime slurry in a reactor with 20 minutes residence time and when lime slurry simply was injected directly inline (data not shown).

[1] Here, lime utilization refers to the ratio of lime that dissolved into the lake water over the total quantity of lime added.

Water and Load Balance Results

Results of the water balance component for Nero Lake showed that the outflow from Nero Lake in a year with average annual precipitation and evaporation is approximately 360,000 m^3. With a volume of 11,000,000 m^3 the average hydrological residence time of the lake is approximately 30 years.

In the load balance model, inflows to Nero Lake (local runoff/groundwater and flow from Carney Lake) were assigned alkalinity concentrations that were typical of measured baseline concentrations. Historical estimates of acidity loadings from the tailings area were highly uncertain. Fortunately, historical lake water quality data dating back to 1978 were available. Because of this, the input of acidity to the lake could be established by varying acidity loading rates and comparing the model water quality predictions to the water quality measured over time.

Figure 9 shows the results of the acidity loadings sensitivity analysis that were completed for Nero Lake along with historical concentrations of acidity. Four different scenarios are shown here:

1. Acidity Input = Alkalinity Input. This scenario is the threshold situation where the lake would be neutral but would have no or little excess of alkalinity or acidity. If acidity loadings flowing to the lake are greater than alkalinity loadings, then the lake would become acidic over time.
2. Acidity input is half of the alkalinity input. In this scenario the lake would become neutral over time and remain neutral in the future.
3. Annual loadings of acidity to Nero Lake of 3.5 tonnes/year (best fit based on sensitivity analysis).
4. Decaying rates of acidity input to Nero Lake. This scenario assumed that annual loadings of acidity in 1978 were approximately 30 tonnes (as $CaCO_3$) but then gradually decreasing over time.

Comparing the measured acidity concentration to the modelling results shows that the annual loadings of acidity to Nero Lake likely are less than half of the alkalinity loadings reporting to the lake. Therefore, the observed reduction in acidity since 1978 is only possible if the lake receives a net surplus of alkalinity. Furthermore, the model results suggest that the effect on acidity concentration in Nero Lake is similar whether the acidity loadings have been constant or whether loading rates were relatively high in 1978 and decayed over time. The significance of this result is that the lake is unlikely to become acidic once the water has been neutralized.

Figure 10 shows the predicted effect of neutralizing the water in Nero Lake with varying lime doses. The results show that the addition of 150 tonnes of lime is sufficient to neutralize the lake, while addition of 800 tonnes of lime would bring the alkalinity concentration near to the expected long-term steady state concentration of about 75 mg/L of alkalinity (as $CaCO_3$).

Figure 11 shows the predicted effect of reducing acidity loadings by 0% to 95% by placing a tailings cover on the tailings beach or by other mitigation measures. The results show that a reduction in the current loadings of acidity from the tailings are likely insignificant in terms of effects to the water quality in Nero Lake.

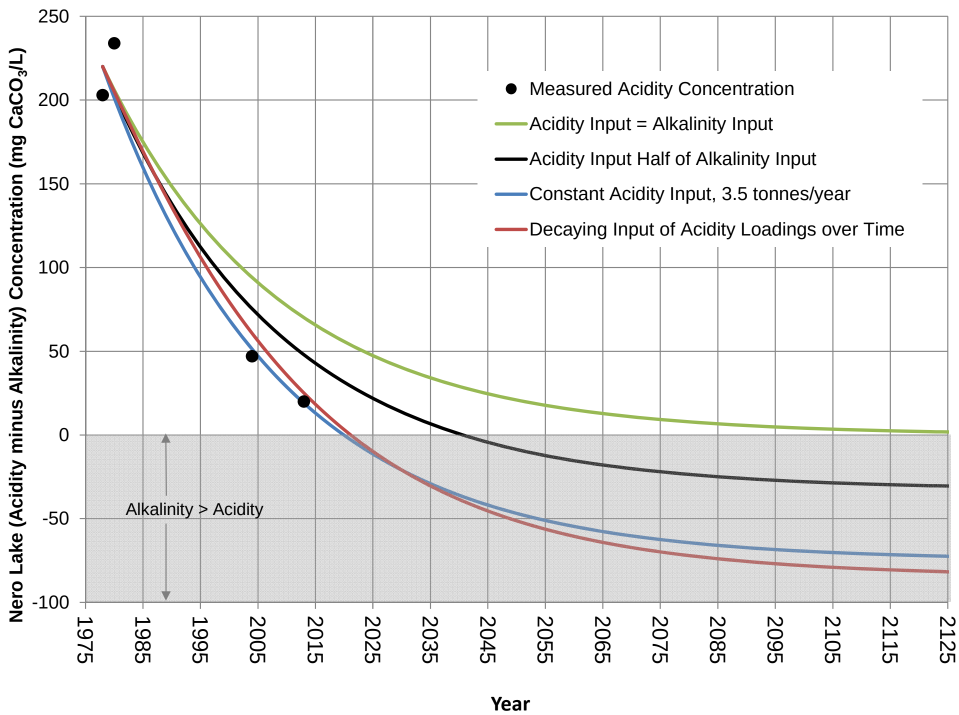


Figure 9 Sensitivity Analysis for the Acidity and Alkalinity Load Balance for Nero

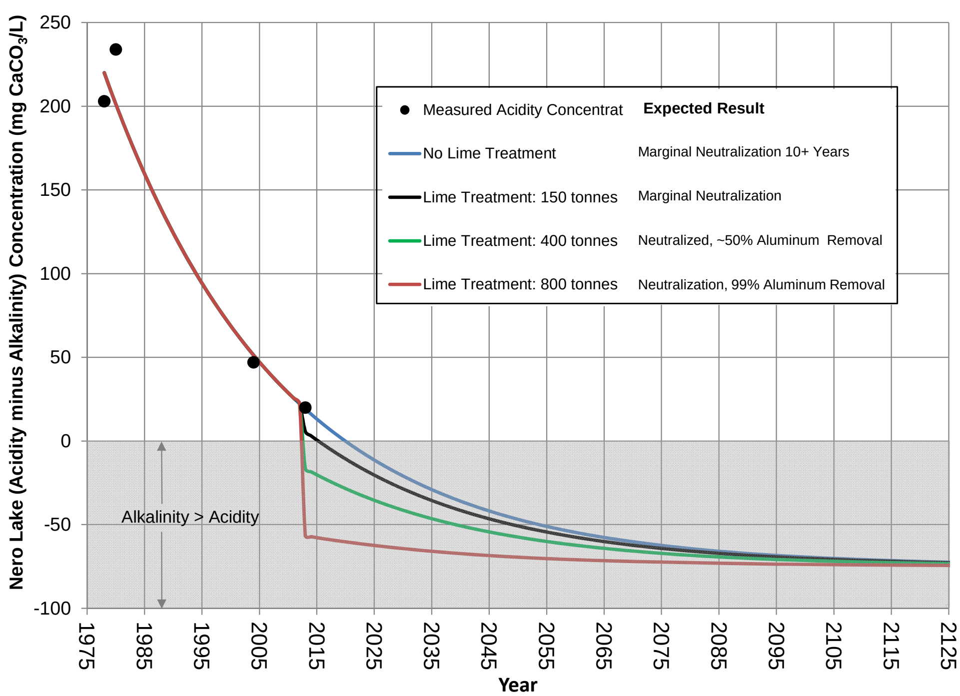


Figure 10 Predicted Effect of Lime Treatment on the Acidity and Alkalinity Load Balance for Nero Lake

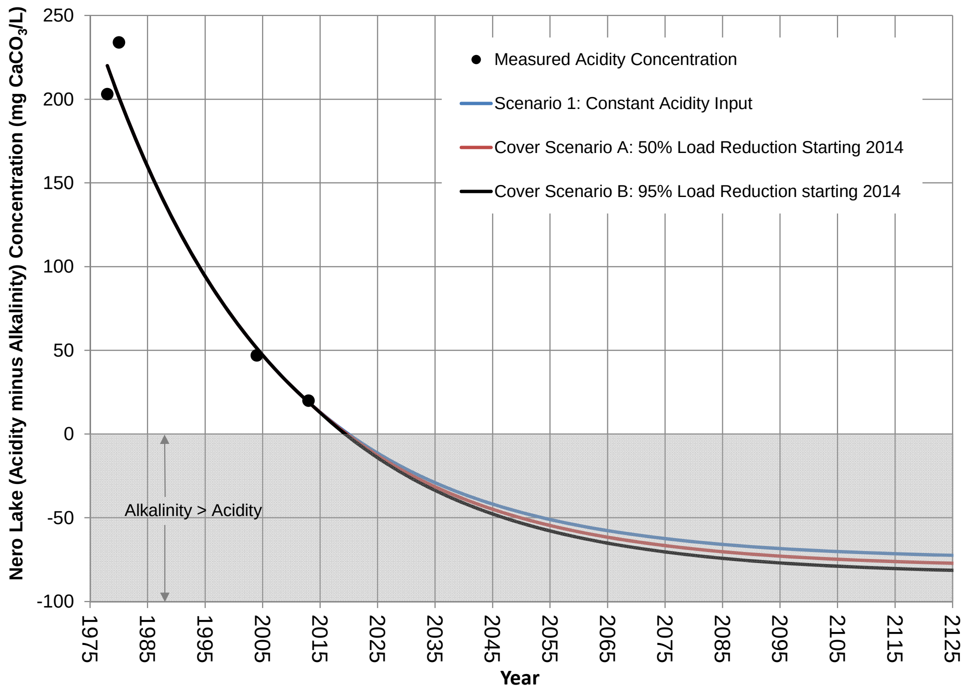


Figure 11 Predicted Effect of Tailings Cover on Nero Lake Water Quality

CONCLUSIONS

A pilot trial and a modelling study were used to develop the design basis for an in-situ treatment system for Nero Lake. Results from the pilot trial showed that:

- A lime dose of 20 mg/L to 75 mg/L would be required to treat the lake water depending on the desired treatment endpoint. A lime dose of 37 mg/L was recommended for the final design.

- The required lime dose can be delivered to the lake by circulating and dosing 10% to 20% of the lake volume without significant reduction in lime utilization. The final design recommendation was for 20% of the lake water to be circulated and dosed.

- A reactor for mixing lake water and lime slurry would not be required: in-line injection of lime slurry resulted in similar lime utilization as mixing slurry and lake water in a reactor tank.

The acidity and alkalinity load balance showed that:

- Since 1978, the annual load of alkalinity to Nero Lake has been greater than the annual load of acidity. This means that the lake will not become acidic following neutralization, assuming that no new and unanticipated sources of acidity appear.

- Any reduction in the current acidity loadings reporting to the lake is unlikely to significantly affect the long-term water quality of Nero Lake.

REFERENCES

Golder Associates (Golder) 2013. Environmental Impact Statement: Risk Reduction Plan for the Inactive Lorado Uranium Tailings Site. Report submitted to Saskatchewan Ministry of Environment and the Canadian Nuclear Safety Commission. March 2013.

Golder Associates (Golder) 2011a. Lorado Uranium Tailings Site: Water Treatment Alternatives Evaluation for Nero Lake. Annex F to 2013 Environmental Impact Statement: Risk Reduction Plan for the Inactive Lorado Uranium Tailings Site. October 2011.

Golder Associates (Golder) 2011b. Long-Term Water Quality in Nero Lake. Annex E to 2013 Environmental Impact Statement: Risk Reduction Plan for the Inactive Lorado Uranium Tailings Site. Report submitted to Saskatchewan Research Council. November 2011.

MEASURING SUCCESS TO ACHIEVE REGULATORY SIGN-OFF

Ron Sparrow, B.SC., RPF, CPESC
Senior Professional Forester
Trace Associates Inc., Calgary, Alberta, Canada

ABSTRACT

For simple sites such as wellsites, the requirements to achieve regulatory sign-off are clearly defined. For more complex sites such as mines (coal, mineral, aggregate pits, and quarries) the legislative requirements are not as clearly defined. The differences between legislative requirements are often a result of the complexity of the sites, which can be amplified by the sheer size of some locations. Using geographical information systems (GIS) with available spatial datasets (wet area mapping, land capability modeling, vegetation cover type, soil classification, surficial geology, wildfires, flooding, wet areas, wildlife habitat, etc.) allows the opportunity to analyze key site characteristics both pre- and post-disturbance and define realistic end targets for a site. The use of GIS spatial datasets during the planning stage makes it possible to relate the site conditions (progressive and final reclamation) at time of monitoring, back to the desired target for a site or sub-component of a site to determine if a site is on target, and to make interventions (weed control, supplement revegetation, wildlife preclusion, etc.), as necessary. Monitoring results over time contributes to continual improvement and creates a legacy GIS database to be used when planning reclamation programs, creating targets for a site, and applying for final reclamation.

KEY WORDS

Monitoring, reporting, GIS, data, progressive reclamation, reclamation.

INTRODUCTION

The transition of reclaimed lands to the crown is often complicated by legislation, reluctance of regulators to certify portions or entire sites prior to full closure, and hesitancy by mines (coal, mineral, aggregate pits and quarries), since they will lose access to the certified lands. The progression towards regulatory signoff is simple in concept but complex in practice, due to stakeholder requirements, legislation, the long-life typical of mines, and the ability to link pre-disturbance conditions to reclamation objectives and outcomes. Key to measuring success to achieve regulatory signoff is the development of a comprehensive reclamation monitoring program coupled with robust data management and reporting. This paper touches on all these aspects, incorporating learning's from projects and literature.

LEGISLATION

In British Columbia, legislation requires all mining operations to carry out a program of environmental protection and reclamation to ensure that upon termination of mining, land, watercourses, and cultural heritage resources will be returned to a safe and environmentally sound state and to an acceptable end land use (MEM, n.d.a.); or once operations cease, mine site lands are returned to a useful and productive state (MEM, n.d.b.).

In Alberta, legislation under the Environmental Protection and Enhancement Act (EPEA) states that "equivalent land capability" must be met, which means that the ability of the land to support various land uses after conservation and reclamation is similar to the ability that existed prior to an activity being conducted on the land, but that the individual land uses will not necessarily be identical; (GOA, 2013). The Government of Alberta defines reclamation as: "…the stabilization, contouring, maintenance, conditioning, reconstruction, and revegetation of the surface of the land to a state that permanently returns the plant to a land capability equivalent to its predisturbed state" (AESRD, 2012). If the inspector is satisfied that the conservation and reclamation have been completed in accordance with appropriate sections of the act and a reclamation certificate, then the inspector may issue a reclamation certificate to all or only a part of the specified land (GOA, 2014).

OBJECTIVE AND CRITERIA DEVELOPMENT

At the start of the process, identify and engage local stakeholders including communities, First Nations, overlapping industries, and regulators. Review with stakeholders the overall goal of the reclamation program as it relates to approval conditions and current legislation. State the goal concisely and in plain language by reducing the ambiguity and subjective interpretation of the approval conditions and legislation.

Establish the objectives of the reclamation program with the focus to meet the goal of achieving regulatory sign-off (reclamation certificate). The objectives should be clear, consider stakeholder needs, current values, economics, technical capability, and meet company requirements.

Identify criteria (critical measures of success) to measure if a site is on track to meeting the goal and prescribed objectives. Adapt as conditions change and use criteria and protocols that provide flexibility, while still being rigorous enough to be defensible. Establish criteria that are acceptable to stakeholders. Involve all interested parties early in the process and keep all parties (interested or otherwise) informed of the progress towards establishing criteria and protocols. Currently the success of reclamation focuses flora-based criteria of success, leaving fauna relatively, with the assumption that if the desired flora recovers so will the desired fauna. Incorporate key performance indicators (KPI) that will serve to determine success and assist in the development of best management practices.

MONITORING

Define the reasons for monitoring, types of data to collect, methods to collect the data, and how the data will be analyzed and interpreted. Establish the objectives of the monitoring program. Develop protocols to measure the criteria that integrate the objectives of the reclamation program without being exclusive to single aspects or land uses. Assemble a team of technical experts from multiple disciplines and include stakeholders. Determine the level of data quality needed. In the early stages of the program, involve outside agencies (Universities, researchers, government agencies, etc.) to determine opportunities for collaboration. Consider the necessity to monitor both flora and fauna in an integrated manner. Protocols developed to measure predominately flora may require modification or support through research (planned and conducted) to validate the assumption that criteria used to measure flora reflect decolonization of key fauna species.

Establish a reclamation monitoring program with methodologies to conduct and report reclamation monitoring. The reclamation monitoring program should assess vegetation, soils, and wildlife on sites associated with the project. Develop a reclamation monitoring program considerate of the following purposes:

- Assess the state of sites related to the project.
- Qualify and quantify the state of each site related to the project.

- Document the state of each site for inclusion in regulatory, internal, and external reporting.
- Evaluate the appropriateness and effectiveness of the reclamation program and monitoring criteria.
- Frequently and consistently measure against KPIs.
- Identify and propose corrective measures (e.g., for erosion, poor revegetation, subsidence, etc.).
- Report best management practices (e.g., revegetation, erosion control, wildlife reintegration, etc.) observed during the assessments.

Create data collection protocols (written procedures, such as formal standard operating procedures) to be used in sampling, monitoring, and managing the data. Create protocols that are concise and detailed to avoid interpretation by persons collecting and interpreting the data. Data that is accurate and consistent transcends changes in company staff and consultants resulting in data that is useful for the life of the project and applicable at time of application for reclamation certificate or closure.

DATA MANAGEMENT

The monitoring program is basically three steps: sampling, analysis, and interpretation. Sampling is time bound and therefore needs to be accurate and in a format that transcends technology changes (e.g., in a format that is continually supported, such as PDF or hard copy). Incorporate an information management system early (files and electronic). Common challenges include different and sometimes incompatible software between the consultant and company or that existing data is in obsolete format or in the case of spatial data, incorrectly georeferenced. These issues need to be identified and addressed ongoing.

Currently the dominant systems to store, process, and review data include computer aided drafting (CAD), geographical information system (GIS), tabular databases, and word processing related programs (Microsoft suite of programs such as Excel and Word). More progress is being made towards customized systems that integrate GIS and SharePoint systems to capture and manage documentation, including management decisions, and to store and retrieve documentation effectively and efficiently.

Data management systems should allow easy and transparent retrieval of information by or for third parties to review and audit for assurance and accountability (e.g., liability calculation, confirmation of treatments, etc.). Data management systems must incorporate a robust quality management system for both data entry and retrieval that supports the reclamation monitoring program.

Ensure that the data collected during the exploration phase (pre-disturbance) and reclamation is available and acceptable for subsequent environmental modeling and prediction studies, and that the data can be used for the analysis of the success of the reclamation program and stated objectives (Mclemore, Russell & Smith, 1999). Avoid the trap of collecting data simply for the sake of data collection.

Obtain available spatial datasets such as wet area mapping, land capability modeling, vegetation cover type, soil classification, surficial geology, wildfires, flooding, wet areas, wildlife habitat, etc. to analyze key site characteristics both pre- and post-disturbance. Use GIS to analyze data collected as part of the reclamation monitoring program to determine if a site or is on target to meet stated objectives and to make management interventions such as weed control, supplemental revegetation, etc.

REPORTING

Report the results of monitoring to show that the reclamation monitoring program is meeting objectives. Start reporting early by involving local stakeholders to determine the frequency, type, and style of reporting so that reporting is easily understood and transparent. Regulatory requirements for reporting may be already defined within existing legislation. Transparent and timely reporting will build trust with stakeholders, investors, and other companies within the industry. Incorporate existing systems (i.e., Google Earth) with existing spatial data to allow easy access by stakeholders.

SUMMARY

Accurate data (pre- and post-disturbance) that is managed to transcend technology/software changes, and which is suitable for use in data management and GIS systems of the time, will allow mine operators to monitor, analyze, improve, and report on reclamation programs and prove success and obtain regulatory sign-off.

CLOSING

This paper is prepared as a part of the presentation with the same name to be given at the annual British Columbia Technical and Research Committee on Reclamation to be held in Prince George, September 22-25, 2014.

REFERENCES

AENV (Alberta Environment). (n.d.). Reclaiming Alberta's Oil Sands. Retrieved from http://www.environment.alberta.ca/02012.html.

AESRD (Alberta Environment and Sustainable Resource Development). 2012. Upstream Oil and Gas Reclamation and Remediation Program. Retrieved June 11, 2013, from http://www.environment.alberta.ca/01108.html.

GOA (Government of Alberta). (2013). Environmental Protection and Enhancement Act – Conservation and Reclamation Regulation (130/2013). Retrieved from http://www.qp.alberta.ca/1266.cfm?page=1993_115.cfm&leg_type=Regs&isbncln=9780779731343.

GOA (Government of Alberta). (March 29, 2014). Environmental Protection and Enhancement Act Retrieved from http://www.qp.alberta.ca/documents/acts/e12.pdf.

Mclemore, V.T., Russell, C.C., & Smith, K.S. (1999). ADTI-MMS Sampling and Monitoring for the Mine-life Cycle. Retrieved from http://www.proceedings-stfandicard-2009.com/pdfer/Virginia_McLemore_B4_T3_ADTI-MMS-Sampling-and-Monitoring-for-the-Mine-life-Cycle.pdf.

MEM (Ministry of Energy and Mines). (n.d.a.). Reclamation Costing and Security - Reclamation Liability Cost Estimates. Retrieved from: http://www.empr.gov.bc.ca/MINING/PERMITTING-RECLAMATION/COSTING-SECURITY/Pages/default.aspx.

MEM (Ministry of Energy and Mines). (n.d.b.). Permitting and Reclamation. Retrieved from: http://www.empr.gov.bc.ca/Mining/Permitting-Reclamation/Pages/default.aspx.

RISK-BASED REMEDIAL PLANNING FOR THE ABANDONED EMERALD GLACIER MILL AND TAILING SITE

R.C. Dickin, M.Sc., P.Geo. CSAP, FGC.
J. Phibbs, M.Sc.,R.P. Bio.
M. Rankin, M.Sc., R.P. Bio.
M. Sanborn, M.Sc., R.P. Bio.

AECOM
13292 Production Way
Burnaby, B.C. V5A 4R4

ABSTRACT

The Emerald Glacier Mill and Tailings Site is an abandoned, base metal, ore processing site located on Crown Land approximately 80 km southwest of Houston BC that was prioritized for remedial action by the BC Crown Contaminated Sites Program (CCSP). The investigations, analysis and decisions required for developing an effective, risk based, remedial plan to return the site to a safe, healthy forest environment are described. Field investigations were conducted in a progressive manner to identify and quantify physical and chemical risks to human health and the environment and to develop remedial objectives. Detailed quantitative risk assessment was conducted to identify and prioritize the various exposure pathways that warranted mitigation.. Sustainability issues and conflicts with remedial action were identified, such as maintaining intact forest and riparian vegetation along the creek.

A risk based remedial plan was developed to address the following objectives:

i) Prevent physical erosion and off-site migration of the tailings as suspended sediments;
ii) Mitigate or eliminate contaminant exposure pathways that confer a high risk for the Site, including:
    a. Human Health - Remediate/manage site contamination to a state where future traditional or recreational human use will not cause unacceptable health risks, with specific consideration for ingestion of site water and locally harvested game; and
    b. Environmental Protection - Remediate/manage site contamination to a state where residual contaminated soil poses acceptable or tolerable risks to ecological receptors such as mammals, birds, soil invertebrates and plants.

The risk management that involved physical remediation included: a) containment and capping of the tailings; b) capping of the ore processing and mill area; and, c) decommissioning of a small water supply dam on the creek upstream of the tailings facility. Environmental contamination that was not subject to physical remediation, due to lesser and tolerable risk or sustainability conflicts, included metals in groundwater, stream sediments and soil located in high value forest areas.

KEY WORDS: Abandoned Mine, Risk Assessment, Remedial Planning, Crown Land, HHERA

PURPOSE

The purpose of this paper is to present a case history of how the environmental risks from metal contaminants at an abandoned mill and tailings site were investigated, quantified, and prioritized for physical remediation in order to return the land to productive forest and ecological habitat. In some instances trade-offs and risk tolerance were required for sensitive forest and riparian areas that had metal-contaminated soil above standards that could not be physically remediated without destroying important productive ecological habitat.

BACKGROUND

This project was completed under the BC Crown Contaminated Sites Program (CCSP). The CCSP is part of the LNG, Crown Land Opportunities and Restorations Branch (CLORB) within the Ministry of Forests, Lands and Natural Resource Operations (MFLNR). Within the MFLNR, the CCSP leads the management of contaminated provincial lands to reduce risks to human health and the environment. The CCSP works under a Cabinet approved policy that commits the MFLNR to identify and prioritize contaminated sites that are a provincial responsibility using a science-based and risk assessment approach that draws, in part, from the BC Contaminated Sites Regulation (CSR).

The Emerald Glacier mine and mill site is located in west-central BC on the Sweeny Lake/Morice-Tahtsa Forest Service Road, approximately 80 km southwest of Houston. Huckleberry Mine is located about 5 km further south.

Between 1915 and 1971, lead, zinc, copper, cadmium, silver and gold were intermittently mined from underground works located on an alpine ridge (~1500 m asl) approximately 2.5 km north of the mill site (~1000 m asl). Available information suggests that ore was transported off site for processing until 1966 when on site processing commenced at the Mill and Tailings Site (the Site). Mining and on-site milling ended in 1968. The Mill and Tailings site is located 2.5 km south at an elevation of 1000 m within 100 m of Tahtsa/Sweeny Lake Forest Services Road.. During operation, tailings were deposited in a low-lying area contiguous with and within the riparian zone of an unnamed creek that flows to Sweeney Lake. A small upstream dam and reservoir on the creek was used to supply water to the mill (Figure 1).

PHASED ENVIRONMENTAL FIELD INVESTIGATIONS

Environmental investigations are typically completed in a series of phases, particularly when the magnitude and extent of contamination is completely unknown at the outset. A phased approach is most likely to identify the extent and magnitude of the problem without undue effort and expense. In a phased approach, identified contamination is followed up with step-out samples in order to delineate the extent of contamination. Several phases are often required to fully investigate an abandoned site, particularly where little or no documentation of the historical activities that caused the contamination.

<u>Initial Investigations and Site Prioritization for Remediation</u>

CCSP has developed protocols for conducting preliminary site investigations of abandoned contaminated sites on Crown Lands, including historical mines and associated facilities. CCSP has also developed a standard methodology for prioritizing remedial efforts at the highest risk sites.

Figure 1 Emerald Glacier Mill and Tailing Site

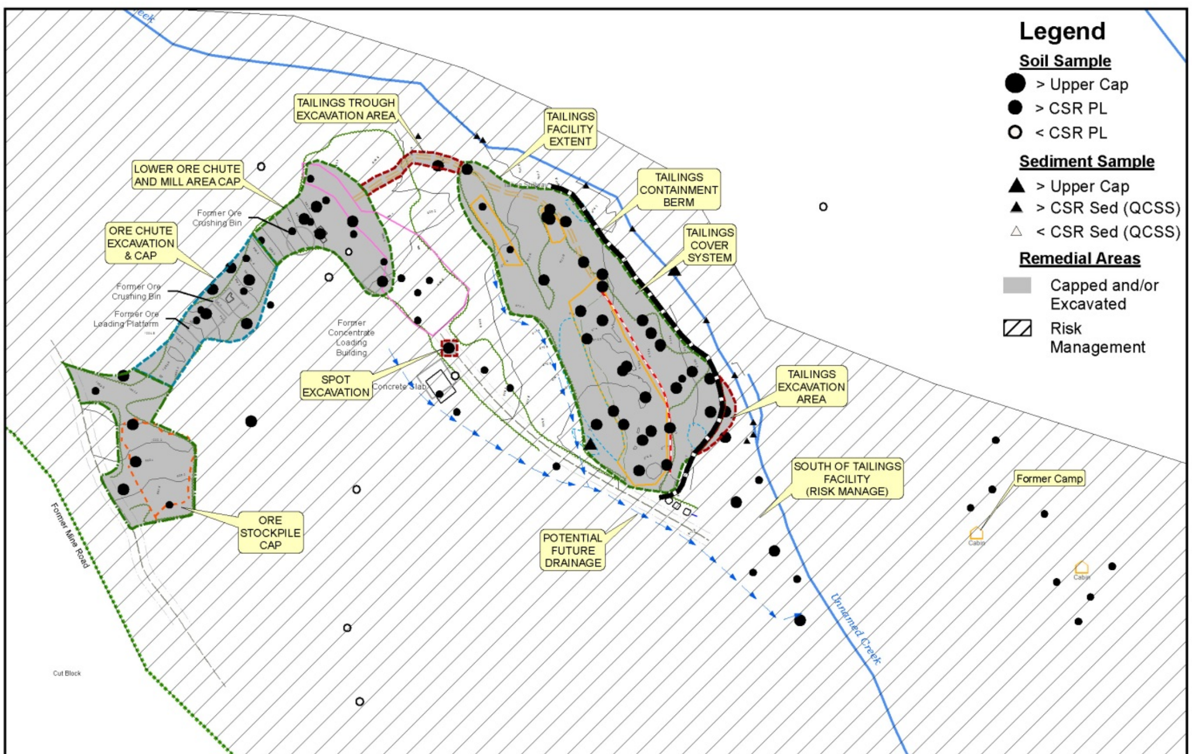


Figure 2 Soil/Sediment Metal Results and Remedial Areas

The initial environmental investigations at the Emerald Glacier Mine Site and Mill and Tailings Site were conducted between 2008 and 2010. This provided CCSP with sufficient data to assess relative risks between other sites on Crown Land in BC and to prioritize further assessment and remediation actions.

The Site is accessible to people due to proximity of the Forest Service Road. The two remaining mine cabins are located a short distance away along the Forest Service Road and are reportedly used by a local snowmobile club. CCSPs risk grading methodology prioritized the Mill and Tailings Site as a high risk site that warranted further investigation and remediation based on high metal concentrations (arsenic, copper, lead, and zinc) and accessibility to people. The Mine Site is located on a high mountain ridge that is much less accessible to people and therefore it was not considered a high priority for remediation.

Specific conditions at the Mill and Tailings Site that caused the high risk designation included the following factors:

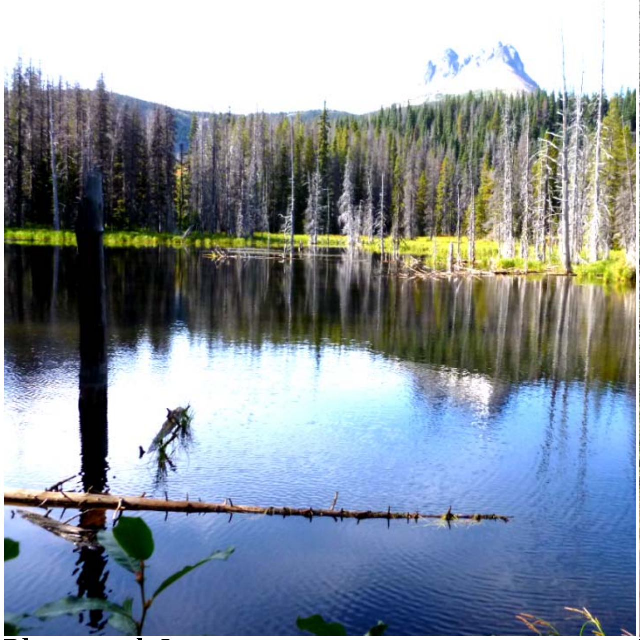
- Poor containment and lack of a cover system on the tailings, which are located within 10 m of a creek area (Photographs 1 and 2);
- Visual evidence of tailings erosion into the adjacent creek;
- Highly elevated metal concentrations (arsenic, lead, zinc, copper, cadmium) in mill soils and tailings;
- Elevated concentrations of metals in stream sediments;
- Potential for high creek flows to further erode the tailings;
- A poorly constructed, unmaintained dam for the mill water supply reservoir on the creek located 700 m upstream of the tailings facility that could fail and cause flooding and erosion of tailings. (Photograph 3)
- Dilapidated mill buildings posing a physical risk (Photograph 4);
- Close proximity to an active Forest Service Road and knowledge that the Site is accessed for recreational activities.

Table 1 provides a summary of the environmental investigation and remedial planning stages and how the findings of each investigation led to further investigations to resolve data gaps or to remedial planning decisions.

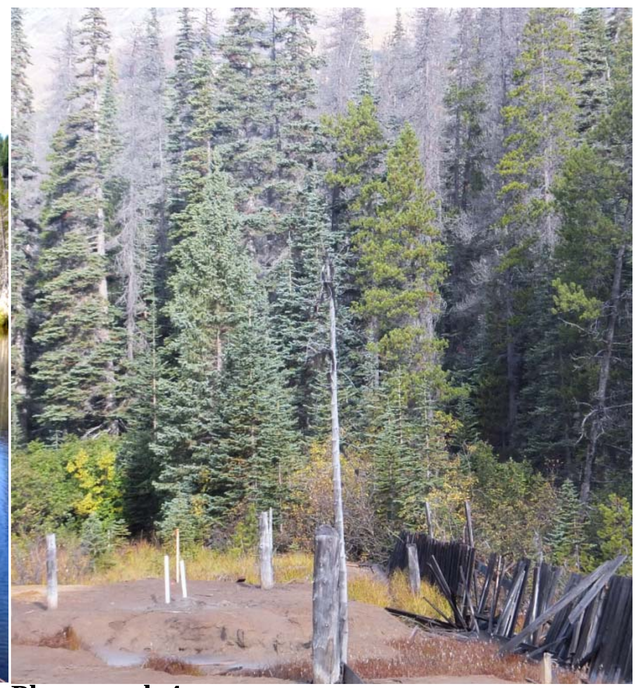
Figure 2 summarizes the soil and sediment sample metal concentrations relative to applicable Contaminated Sites Regulation (CSR) Standards and shows areas of concern that were eventually remediated by physical capping or were concluded to have acceptable/tolerable risk.

Photograph 1:
Tailings Disposal Area with Creek in Forest along Right Side of Photo

Photograph 2:
Mill Debris and Ore Bin at Base of Ore Chute



Photograph 3:
Mill Reservoir Photo taken from Dam



Photograph 4:
Tailing Area with Erosion Channel and Former Wooden Containment Structure. Creek is in Trees within 10 m to right side of Tailings area

Table 1 Phases of Environmental Investigation and Risk-Based Remedial Planning

Stage	Scope	Findings	Decision/Milestone
Modified Preliminary Site Investigation (PSI) 2008	Visual inspection, surface soil/rock, surface water, sediment sampling,	Elevated total and leachable metal concentrations (As, Cd, Co, Cu, Pb, Ag, Zn) in soil, sediment, tailings; PAG rock present but no surface water ARD effects noted	CCSP determines that Emerald Glacier Mill and Tailings Site is a high priority for a DSI.
Supplemental Site Investigation (Mine Site) and Limited Detailed Site Investigation (DSI) (Mill and Tailings Site), 2009-2010	Drilling, soil sampling with depth, XRD tailings mapping, groundwater and surface water sampling, delineation of tailings and soil chemistry	Mine Site – adequate neutralization potential and minimal surface water impact Mill Site – 2500 m^3 of tailings and metal contaminated soil delineated; erosion of tailings into creek; high metal concentrations in tailings and groundwater within tailings; but diluted in underlying gravel and no surface water impact	No further investigation of remote high elevation Mine Site required. Mill and Tailings Area beside Forest Service Road ranked as high priority for further assessment and remediation
Gap Analysis and Supplementary DSI Investigations, 2011	Topographic mapping; Further delineation of metals in tailings/soil; Sediment quality/benthic community/surface water quality toxicity effects assessment; Seasonal assessment of surface water and groundwater quality; Assessment of reservoir dam and creek hydrology	Tailings area delineated further;- As, Cu, Pb, Zn exceed high risk CSR Protocol 11 Upper Cap Standards; Sediment contamination not as widespread as expected but erosion of tailings is occurring; Dam in poor condition but low risk of catastrophic failure; Mill buildings and debris are physical hazard and prevent full sampling of area; Historic camp area metals in soil are minor risk	Need to remove buildings and debris prior to further sampling; Need to contain and cap tailings which are the highest priority; Adequate data for preliminary remedial option evaluation
Remedial Options Evaluation, 2012	Assessed feasibility, effectiveness and cost for four main options	1. Risk management alone 2. Partial on-site disposal 3. On-site disposal 4. Off Site disposal	Risk management alone inadequate; full off-site disposal neither feasible or affordable
Phase 1 Remediation (Building Removal) 2012	Removal of mill buildings and debris for further soil assessment	To remove physical hazard and facilitate complete contaminant delineation	BC Bid Tender for building removal

Stage	Scope	Findings	Decision/Milestone
Supplementary Investigations 2012	Topographic mapping; Borrow pit investigations; Geotechnical assessment of tailings berm and water supply reservoir dam; Step-out soil contamination delineation; Paired vegetation/ soil sampling; physiologically based extraction testing of soils; Riparian area habitat assessment; Tailings Acid-Base accounting and leachable metals testing, Groundwater quality	Identified potential borrow pit area and geotechnical requirements for tailing berm and dam decommissioning; Geochemical considerations for tailings capping; Potential need to cap ore stockpile area; Habitat assessments confirmed dense forest and riparian areas	Adequate information to proceed with HHRA and Risk-based Remedial Plan
Preliminary Detailed Human Health and Ecological Risk Assessment (HHERA) Winter 2013	Quantification of risks for the tailings facility, mill area, ore chute, ore stockpile areas, and adjacent forest and riparian areas. Test efficacy of preferred remedial option relative to site protection goals.	Preliminary results of the HHERA, based on the preliminary conceptual design of the preferred remedial option predicted unacceptable residual risks for both human and ecological receptors in the mill area, tailings area, ore chute and ore stockpile area.	Areas of highly contaminated soil in the mill area, tailings area, ore chute and ore storage area require further consideration in the proposed remedial option. No need or benefit to physical remediation of creek bed or riparian areas or adjacent heavily forested areas that are functioning adequately
Remedial Options Evaluation Update Winter 2013	Revised most favourable remedial options with updated environmental information and HHERA results. Cost estimate for budget purposes	1m cap and berm and drainage required for the tailings, mill area; 1m cap required for ore storage, ore chute and portion of mill area. New spillway required to lower reservoir water level by 1 m;	Decision to proceed with conceptual remedial design and issue RFP for detailed design build contract..

Stage	Scope	Findings	Decision/Milestone
Final Detailed Human Health and Ecological Risk Assessment (HHERA) Winter 2013	Quantify residual risks to human end ecological receptors following revision of the preferred remedial option based on preliminary HHERA results.	**<u>Human Health Risks</u>** • Negligible risks due to exposure to threshold contaminants through ingestion, inhalation, or dermal contact with residual contaminated soils, consumption of locally sourced drinking water and locally sourced game. • Potential increased incremental lifetime cancer risk (ILCR) as a result of ingestion of grouse tissue collected from the site. The exposure assessment carries significant uncertainties associated with a literature-sourced arsenic transfer factor and may overestimate the dose/risk for this exposure pathway. **<u>Ecological Receptors</u>** <u>Wildlife:</u> • Low risk of impacts to small wildlife receptors with small home ranges (e.g. rodents or birds). • Negligible risks to secondary wildlife consumers (e.g. red fox) in spite of conservative assumptions. • Negligible risks to wildlife populations and individual species under post remedial scenario. <u>Vegetation:</u> • Localized risks to plants in some areas of residual contamination. Risks are unlikely to result in population impacts for valued ecological receptors or major ecological impairment.	Proposed remedial activities are predicted to be: • Sufficiently protective of human health, when considering uncertainties and conservatism inherent in the assessment. • Protective of ecological receptors are a scale relevant to ecological protection at the population level.

Stage	Scope	Findings	Decision/Milestone
Conceptual Design of Preferred Remedial Option, Spring 2013	To provide sufficient information to facilitate proposal preparation and a fair bidding process	Provided areas, volumes, slopes and other information for proposed remedial works	Specifications for RFP; Applications for Water Act approval; BC Bid Design/ Build RFP.
Detailed Design; Construction, Summer- Fall 2013	Develop borrow pit, decommission dam; cap tailings, ore stockpile, ore chute and mill areas		Design approved; Permits acquired; Remedial construction completed
Post Remediation Monitoring 2014	Twice annual monitoring Visual inspection and photographic record; Groundwater and surface water sampling	Cap, berm, drainage and dam spillway integrity acceptable following spring run-off; groundwater and surface water quality similar or better than historical results; vegetation re-establishment started	Continued monitoring scope and frequency dependent on of physical conditions, water quality and re-vegetation establishment

INITIAL REMEDIAL PLAN

<u>Remedial Objectives</u>

In 2012 the CCSP initiated remediation of the Site with the following objectives:

i) Prevent physical erosion and off-site migration of the tailings as suspended sediments;
ii) Mitigate or eliminate contaminant exposure pathways that confer a high risk for the Site, including:
    a. Human Health - Remediate/manage site contamination to a state where future traditional or recreational human use will not cause unacceptable health risks, with specific consideration for ingestion of site water and locally harvested game; and
    b. Environmental Protection - Remediate/manage site contamination to a state where residual contaminated soil poses acceptable or tolerable risks to ecological receptors such as mammals, birds, soil invertebrates and plants

<u>Environmental Standards</u>

The BC Contaminated Sites Regulation (CSR) scheduled standards were applied for this project. Wildland land use was applied. Soil samples were compared to BC CSR Schedule 4 and 5 soil standards for (1) human contact, (2) toxicity to soil invertebrates and plants, and (3) livestock ingestion soil & fodder and 4) freshwater aquatic life. Sediment samples were compared to the Schedule 9 sensitive criteria. Surface water quality results were compared to the BC Approved and Working Ambient Water Quality Criteria. The applicable CSR Protocol 11 Upper Cap Concentrations were used to delineate higher risk areas within in the various sampling media.

Groundwater was compared to the Schedule 6 aquatic life standards. There are no water wells within many kilometers of the Site. CCSP added a notation to the Crown Lands management system that prevents any future use of groundwater or surface water for drinking water. Therefore the groundwater and surface water quality data were not compared to drinking water standards.

Preliminary Remedial Options Evaluation

Conceptual remedial options were evaluated and order of magnitude cost estimates were prepared for each Area of Environmental Concern (AEC). To simplify the cost estimates assessment, four overall project remediation scenarios were considered. These four scenarios cover the full range of potential costs and are considered appropriately representative of the four broad remedial approaches that could be taken: (1) risk management alone; (2) partial on-site capping (reduction of risk) with risk management; (3) extensive on-site capping (reduction of risk) with risk management; and (4) remediation to numerical standards with complete excavation and off-site disposal.

The preferred remediation scenario was Scenario 2 partial on-site disposal. This scenario actively remediates the highest risk contamination (erosion and direct exposure to tailings and soil with metal contamination above the CSR Upper Cap Standards) but uses a risk management approach to deal with lower risk aspects of the site.

It was determined that Scenario 1 Risk Management was not adequate to meet CCSP's objectives or regulatory requirements. Similarly, Scenario 4 Off-site Disposal was not a viable rehabilitation technique for an abandoned mine and the potential costs were not justifiable.

HUMAN HEALTH AND ECOLOGICAL RISK ASSESSMENT (HHERA)

A quantitative HHERA was conducted in accordance with Health Canada, Environment Canada, and BC Ministry of Environment Guidance documents based on inferred site conditions following the implementation of the preferred remedial option described above. The purpose of the HHERA was to predict residual risks to human health and ecological health, based on the remedial options put forward, and to refine the remedial options if the risk assessment identified unacceptable residual risks.

Based on the site investigations it was apparent that physical remediation was immediately required in some areas to reduce/eliminate certain risks posed by the Site, and that other areas were likely more suitable for managing the existing tolerable risks due to a variety of factors including:

1. potential conflicts between physical remediation and ecological impact,
2. evidence of limited environmental effects,
3. low probability of exposure to sensitive receptors or valued ecosystem components, and;
4. ecological risks in disagreement with the site protection goals.

Human receptors included in the quantitative risk assessment were assumed to be adults and toddlers engaged in traditional land use activities which may include hunting and gathering activities and summertime encampment and harvesting. People engaged in traditional land uses are expected to have the greatest potential magnitude of exposure based on duration of visit and activities they are involved in. By developing risk-based remedial strategies that protect a traditional lifestyle, other types of site use (e.g. hiking and non-hunting/camping short-term recreational visits) would be predicted to experience negligible risks.

Valued ecological components were identified based on a review of the federal and provincial conservation lists, species indigenous to the biogeoclimatic zone, site observations by AECOM biologists, habitat scale, and economic or cultural significance. For example, the wolverine (a provincially listed species) was not assessed based on their large habitat range relative to the very small footprint of terrestrial contamination; these conditions would result in a low exposure and ecological risk that may misrepresent more highly exposed species with smaller roaming ranges. Ecological receptors included in the HHERA were spruce grouse, red squirrel, red fox, western toad, and terrestrial vegetation.

For the purposes of the HHERA, the site was subdivided into two areas of environmental concern based on differences in expected site usage by human and terrestrial ecological receptors, and the physical characteristics of the site.

- AEC 1 - The lower site comprised of the mill pad, the tailings area, and the surrounding forest representing ~90% of the Site footprint contains contaminated soils, ground water and sediment associated with post milled materials and tailings.
- AEC 2 - The ore stockpile area is isolated from the remainder of the site by a steep forested slope, accounts for only 10% of footprint, is poorly vegetated and is dominated by large fragments of unprocessed rock and rock dust.

These areas were assessed as contiguous units, and total risks were adjusted for the relative proportion of the site comprised by each area.

Contaminant exposure point concentrations were based on the 95% upper confidence limit of the mean (95UCLM) where data allowed, otherwise the maximum concentration measured in a given medium was used. For dietary intake of plant or animal tissue, tissue concentrations were modelled using literature derived bioaccumulation factors. Exposure as a result of ingestion of contaminated soil was adjusted for the relative bioaccessibility based on Physiologically Based Extraction Testing (PBET) results. PBET approximates the amount of contaminant actually absorbed via the digestive tract.

Risks to human and ecological receptors were calculated based on toxicity reference values (TRV) sourced from either the US EPA or Health Canada (in accordance with BC MoE Technical Guidance 7). For human receptors, risks posed by non-carcinogenic contaminants were assessed by calculating a hazard quotient (HQ, also known as exposure ratio) using published tolerable daily intake (TDI) values. Risks posed by carcinogenic contaminants were estimated by calculating an incremental lifetime cancer risk (ILCR) for adult receptors only. Ecological receptors were assessed relative to non-threshold TRVs protective of effects to reproduction, growth and long term survival.

Risks to human receptors were deemed *negligible* if HQs were calculated to be below a value of 1 and ILCRs were below a value of 1.0×10^{-5}. Interpretation of the hazard quotients for ecological receptors was guided by the categorical descriptions provided below, which were defined to provide additional perspective for CCSP in relation to management of its site portfolio. It is important to note that the magnitude of the HQ exceedances is not linear, and after accounting for the ameliorating effects of conservative assumptions, HQs in the upper range of magnitude are essentially the same and indicate high potential for impacts.

- Negligible Risk of Impact(HQ < 1): Given the feasible exposure pathway, residual exposure is acceptable for the future, and may or may not warrant continued management and periodic monitoring;
- Low Risk of Impact (1 ≤ HQ < 10): Based on exceedances of no adverse effect levels and conservative assumptions, a low level of stressor effect(s) to long-term survival, growth, or reproduction of certain species may be present; if not mitigated, a decision to accept and manage of this sustained level of effect should be scrutinized and monitored;
- Moderate Risk of Impact (10 ≤ HQ < 20): Based on higher exceedances of no adverse effect levels, stressor effects towards the assessment endpoint are likely present and should either be resolved through a refined analysis, mitigated through risk management options, or the potential adverse consequences recognized and accepted;
- High Risk of Impact (HQ ≥ 20): High exceedances of no adverse effect levels underscore the need and priority to resolve the situation through a combination of refined analyses and/or mitigation of the causal risk factors.

RISK-BASED REMEDIAL PLAN

Remedial planning was conducted in conjunction with the HHERA in an iterative manner. Preliminary results of the HHERA, based on the initial remedial plan provided to the risk assessors, identified residual contamination posing appreciable risks to human and ecological health in the mill area, tailings area, ore chute and ore stockpile area. These preliminary results were used to modify and update the scope of physical remedial works to reduce risks to acceptable levels in these areas. After several iterations an acceptable Risk-Based Remedial Plan was developed.

The original preferred remedial option, Scenario 2 partial capping of contaminated areas, was modified based on the HHERA and discussions with the client. The preferred option was expanded to include capping of additional areas including the ore stockpile area, which was closer to Scenario 3, complete onsite disposal. A revised Risk-Based Remedial Plan was developed that:

- addressed the risks that were quantified in the HHERA;
- confirmed that the proposed remediation scenario was preferable to other options;
- updated the remediation areas, volumes and cost estimates;
- identified regulatory requirements for the remediation;
- included a post-remedial monitoring plan to evaluate the effectiveness of remediation;
- provided the basis for issuing a request for proposals for a design/build remediation contact.

The final Risk-based Remedial Plan is summarized below for the various areas of concern at the Site.

Tailings Area

Tailings represented the greatest potential exposure media for human and ecological receptors and appreciable risks were predicted. The tailings also were a physical risk to adjacent creek. The Risk-Based Remedial Plan recommended that the Tailings Area be subject to *in situ* capping similar to what was originally proposed but the cap extents were better defined due to the additional delineation, topographic mapping and borrow material investigations. The Risk-Based Remedial Plan recommended the following components.

- Construction of a containment berm around the NE and SE sides of the tailings deposits, capping of the tailings in place with locally available fine-grained till, construction of a containment berm and construction of appropriate drainage diversion ditches and sediment ponds.
- Minor excavation (0.3 m deep scrapes) of thin layers of tailings along the former tailings trough, eroded tailings between tailings facility and Unnamed Creek and eroded tailings along the Site access road.
- Risk management of tailings in high value forest or riparian areas that cannot be sustainably excavated.
- Revegetation of all excavated and capped areas.

Ore Stockpile/ Ore Chute Slope/ Mill Pad

Appreciable risks to human and ecological receptors predicted for ore stockpile, ore chute and a portion of the Mill pad areas. The Risk-based Remedial Plan recommended that remediation of this area include the following components.

- Off-site disposal of metal debris from Phase 1 demolition activities.
- The Ore Stockpile area and surrounding non-forested area will be capped with 1.0 m of fine-grained till.
- The steep Ore Chute slope will no longer be risk managed (due to high metal concentrations and potential for erosion). The upper slope of the Ore Chute is to be excavated to 0.5 m depth. Excavated material will be consolidated on top of the lower portion of the Ore Chute and NW portion of the Mill Pad.
- The NW portion of the Mill Pad and the entire ore chute will then be capped with 1.0 m of compacted fine-grained till.
- Drainage, erosion control and re-vegetation measures will be undertaken.

Historical Camp

The historic camp was predicted to pose negligible risk to human and ecological receptors of concern in the context of the overall site protection goals.

Adjacent Forested and Riparian Areas

Physical remediation involving excavation and 1 m till cap is unlikely to provide ecological benefit in a healthy forested environment. The probability of ecological effects outside the immediate area of contamination is considered negligible. Small areas of contaminated soil remaining in healthy and productive forested areas were therefore left in place.

Creek Sediments

Initial sediment testing and evidence of tailings erosion into the creek suggested the potential need for creek sediment remediation. A more comprehensive assessment based on the sediment quality triad approach (sediment chemistry, benthic community analysis, and laboratory toxicity testing) indicated minimal impacts from existing metal concentrations in creek sediment. Therefore, the Risk-Based Remedial Plan recommended no a physical remediation involving excavation or capping that would remove or damage mature riparian vegetation.

Surface Water

Surface water ingestion by humans from the creek was included for human and wildlife receptors in the HHERA. Risks to human receptors were determined to be negligible. Surface water ingestion for wildlife receptors represents less than 1% of total dose. Environmental effects assessment indicates effects are unlikely to the aquatic environment. The Risk-Based Remedial Plan recommended that surface water quality be risk managed by monitoring surface water quality in the post remediation period. A notation on the Crown Land management system was recommended to prohibit any future drinking water intakes on the creek near the Site.

Groundwater

Groundwater quality in the tailings has elevated metal concentrations. Metal concentrations are much less in groundwater from the sand and gravel below the tailing and the pH is within a neutral range. Groundwater from the tailings area discharges to the nearby creek but metal concentrations in the creek are near background levels. The Risk-Based Remedial Plan recommended that groundwater contamination be risk managed by monitoring groundwater quality in the post remediation period. A notation on the Crown Land management system was recommended to prohibit development of drinking water wells near the Site.

Water Supply Reservoir and Dam

The geotechnical inspection of the reservoir dam suggested that the failure risks presented by the dam condition required some degree of physical remedial action. Therefore, the Risk-based Remedial Plan recommended that the reservoir dam be decommissioned by lowering the water level 1 m by excavating and removing the decaying spillway and re-establishing the “natural” stream channel bypass.

Borrow Areas

A potential borrow area for clay or silt-rich glacial till was identified on the forestry cut block located less than 1 km east of the Site. CCSP was responsible for obtaining authorization. The contractor was required to confirm the volume of glacial till required, clear and grub the area and provide proper drainage and environmental controls, excavate the required volumes of fill material and rehabilitation the borrow area.

SUMMARY

The development of a Risk-based Remedial Plan for an abandoned mill and tailings site required an iterative investigation and remedial planning approach over a number of years that included:

- A sequence of phased environmental and engineering investigations to: identify physical and chemical risks; assess potential land use and habitat values; and delineate contamination in various areas and media
- Identification of remedial goals and applicable environmental quality standards;
- A human health and ecological risk assessment (HHERA) to quantify contaminant risks and need for remediation;
- An iterative remedial planning approach that includes an interdisciplinary review of the remedial plan's effectiveness in achieving project objectives.

REFERENCES

Worley-Parsons, 2009. Modified preliminary Site Investigation, Emerald Glacier Mine, Omineca Mining Divisions, BC, Prepared for MFLNRO, Crown Land Opportunities and Restoration Branch.

Keystone, 2011. Supplemental Site Investigation and Limited Detailed Site Investigation Emerald Glacier Mine and Mill Site. Prepared for MFLNRO, Crown Land Opportunities and Restoration Branch.

AECOM, 2012a. Emerald Glacier Mill Site - Supplemental Detailed Site Investigation. Prepared for MFLNRO, Crown Land Opportunities and Restoration Branch.

AECOM, 2012b Emerald Glacier Mill Site Remedial Options Evaluation. April 2012. Prepared for MFLNRO, Crown Land Opportunities and Restoration Branch.

AECOM, 2013a Emerald Glacier Mill Site – 2012 Field Report Prepared for MFLNRO, Crown Land Opportunities and Restoration Branch.

AECOM 2013b Emerald Glacier Mill Site – Emerald Glacier Mill Site: Detailed Quantitative Environmental Risk Assessment. March 2013. Prepared for MFLNRO, Crown Land Opportunities and Restoration Branch.

AECOM, 2014 *Emerald Glacier Mill Site Remedial Options Update. March* 2013. Prepared for MFLNRO, Crown Land Opportunities and Restoration Branch.

AECOM, 2012b *Emerald Glacier Mill Site Remediation Conceptual Design. May* 2013. Prepared for MFLNRO, Crown Land Opportunities and Restoration Branch.

REGULATORY CONSIDERATIONS FOR HISTORIC MINE REMEDIATION IN BC: ATLIN RUFFNER MILL AND TAILINGS CASE STUDY

Joanna Runnells, M.Env.Sc., P.Geo., EP
Gregg Stewart, P.Geo.

Crown Contaminated Sites Program, BC Ministry of Forests, Lands and Natural Resource Operations, Victoria, BC

ABSTRACT

A large number of historic mine sites across British Columbia are no longer operating and no responsible person exists or can be found. The clean-up of historic mine sites is undertaken as per provisions within the *Environmental Management Act* (EMA), and particularly the Contaminated Sites Regulation (CSR) and the Hazardous Waste Regulation (HWR).

The Crown Contaminated Sites Program (CCSP) of the BC Ministry of Forests, Lands and Natural Resource Operations leads the management of contaminated provincial lands where the province has accepted responsibility. The CCSP identifies and prioritizes sites using a science-based and risk assessment approach to reduce risks to human health and the environment.

Using the Atlin Ruffner Mill and Tailings Site as a case study, the application of the CSR and the HWR to the remediation of a historic mill and tailings will be discussed. The site is approximately 20 km from Atlin, BC and produced silver, lead, and zinc as well as gold, copper, cadmium, molybdenum, and tin. Prior to remediation, the mill and tailings site included a mill building with machinery and ore storage bins, two leveled areas (upper and lower mill pads), two trailers, a shack, an explosives shed, a tailings pond, two settling ponds, and an adit with flowing drainage.

Prior to remediation, the site was classified as High Risk under the CSR and metals in soils and mine wastes were classified as leachable hazardous waste under the HWR, requiring special permitting to be managed on site. The paper will focus on the application of the CSR and HWR to the mine site.

REGULATORY CONTEXT

The Crown Contaminated Sites Program (CCSP) is part of the LNG, Crown Land Opportunities and Restorations Branch (CLORB) within the British Columbia Ministry of Forests, Lands and Natural Resource Operations (FLNR). Within the FLNR, the CCSP leads the management of contaminated provincial lands to reduce risks to human health and the environment. The CCSP works under a Cabinet approved policy that commits the FLNR to identify and prioritize contaminated sites that are a provincial responsibility using a science-based and risk assessment approach.

The CCSP uses a number of criteria to identify and prioritize sites. One criterion is there are no one who may hold liability for the clean-up, i.e. there are no Responsible Persons, as defined in the *Environmental Management Act* (EMA). Metrics used for this criterion is the absence of any *Mines Act* permit which approves mineral production as well as the mine is located on Crown Land. If a *Mines Act* permit has existed, the permit holder and/or the BC Ministry of Energy and Mines may be Responsible Persons.

Most of the historic mine sites in British Columbia were active prior to development of the Health Safety and Reclamation Code for Mines in British Columbia and were never permitted. They do not meet the definition of an "abandoned mine" under the *Mines Act*, which “means a mine for which all permit obligations under this Act have been satisfied and in respect of which the mineral claims have reverted to the government.” They are better defined as a “historic mine site” under EMA, which “means an area (a) where mechanical disturbance of the ground or any excavation has been made to produce coal or mineral bearing substances, including a site used for processing, concentrating or waste disposal, and (b) for which a *Mines Ac*t permit does not exist and no identifiable owner or operator is taking responsibility for contamination at the site.”

Further, while owners and operators may be jointly and separately liable under EMA, a person is not responsible for remediation of a historic mine site if “the person acquired the mineral or coal rights at the site for the purpose of undertaking mineral or coal exploration activities and the exploration activities have not exacerbated any contamination that existed at the site at the time the person acquired those mineral or coal rights.” Historic mine sites that the CCSP address are therefore cleaned up under EMA provisions, including but not limited to the Contaminated Sites Regulation (CSR) and Hazardous Waste Regulation (HWR), not under reclamation provisions of the *Mines Act*.

SITE DESCRIPTION

The Atlin Ruffner Mill and Tailings site is located approximately 28 kilometres northeast of Atlin, BC on the northwest slope of Mount Vaughan. The site is location near the end of Ruffner Mine Road, 17 kilometres from the junction with the Atlin Highway. The mine covers a relatively large area with the mill situated at an elevation of 1180 metres above sea level (masl) and a number of adits and works ranging in elevation from 1190 to 1760 masl located at distances up to 4.8 km from the mill. Site drainage flows towards Vulcan Creek and other tributaries of Fourth of July Creek. The uninhabited McDonald Lake Indian Reserve is located approximately 2 km west and downgradient of the Site.

The Ruffner mining area was developed following discovery of mineralization in 1899. Lead, zinc, silver, copper, cadmium, and gold were mined and milled intermittently until 1981 from a series of mineralized veins, shears and lamprophyre dykes cutting granodioritic rocks of the Middle Jurassic Fourth of July Creek batholiths. The mineralization consists predominant of sphalerite, galena, aresenopyrite, pyrrhotitite, and chalocopyrite in a quartz-calcite gangue. Mining and exploration activities are currently inactive after a brief period of surface exploration in the late 1980’s. There is no *Mines Act* permit associated with the mine.

The mill and tailings area was developed by cutting into an esker formation on the side of Mt Vaughn. At least 3,535 tonnes of ore were milled at the site. Prior to remediation, the mill and tailings site included a mill building with machinery and ore storage bins, two leveled areas (upper and lower mill pads), two trailers, a shack, an explosives shed, a tailings pond, two settling ponds, and an adit with flowing drainage (Figure 1, Photos 1 through 3).

SITE CHARACTERIZATION

In 2008, the CCSP risk ranking process identified the mill and tailings area as a priority due to very high concentrations of arsenic, cadmium and lead in soils and concerns of direct exposure to both ecological receptors and recreational users. The mine workings area was ranked lower, human use is less frequent than at the mill site, and options for remediation of this area are limited; it was assessed through assessment of risks to the local environment. Remediation efforts focused on the higher risk mill and tailings area.

Phased environmental investigations between 2008 and 2011 identified that throughout the site, including in surface soils on the upper and lower mill pads, arsenic, lead, zinc, antimony, copper, silver, and cadmium concentrations were found to exceed CSR Wildlands Land Use standards. Concentrations in soils on the upper and lower mill pads, tailings, and settling ponds also exceeded the Upper Cap Concentration, classifying the site as a high risk site under the CSR. High risk sites have specific reporting requirements under the CSR, triggering BC Ministry of Environment (MOE) involvement into the independent remediation process.

In addition, a number of soil and tailings samples exceeded the HWR Leachate Quality Standards for various metals, classifying the material as Leachable Toxic Waste and triggering the HWR requirements. The mill building also contained hazardous waste including friable asbestos, painted debris with leachable lead concentrations exceeding 5 mg/L, and small quantities of residual fuel, grease blocks, batteries and other chemicals. Given the presence of Leachable Hazardous Waste, the age of the mill, and the lack of activity after 1981, the site is classified as a Historical Hazardous Waste Contaminated Site under the HWR. (See next section.)

Water draining from the adit at the mill site is believed to be from a single borehole groundwater source within the workings. Adit drainage was found to have relatively stable and low flow volume (<10L/s) and above the tailings, had concentrations of fluoride and arsenic only slightly higher than local streams. The adit drainage flows approximately 150 metres before entering the settling ponds. Downgradient of the lower settling pond, the drainage quickly infiltrated back into the highly porous ground without connecting to another surface water body. This water is believed to daylight 800 metres away in a wetland area before draining to Fourth of July Creek. There was no surface water pathway to a receiving environment or aquatic habitat.

Over three years and four sampling events, metal concentrations in groundwater at source areas at the site, with the exception of cadmium and zinc, met applicable CSR standards for freshwater aquatic life. Concentrations of cadmium and zinc attenuated to within 100 metres downgradient of the source areas (contaminated soils and tailings), and all groundwater receptors are located a significant distance away. Groundwater conditions suggested any metal leaching that was occurring was being attenuated quickly and metals were not being transported to receiving environments through leaching. Despite the groundwater conditions that suggested metal leaching wasn’t a major concern at the site, the exceedance of HWR leachate quality standards defined some site soils as Leachable Toxic Waste.

Wastes are classified as Leachable Toxic Wastes if they produce an extract with contaminant concentrations greater than those proscribed in Table 1 of Schedule 4, when subject to the US EPA Method 1311 Toxicity Characteristic Leaching Procedure (TCLP) test. This test seeks to replicate potential acidic conditions that waste material may be subject to under typical landfill conditions, and thus uses acetic acid/ sodium hydroxide as the reagent. This test does not do a good job of representing leachate under normal atmospheric weather conditions and may account for the difference between the test results and the observed site conditions. Leachable Toxic Wastes require permitting under the HWR.

The leachate testing completed (31 TCLP tests and 17 shake flask tests (3:1 solid to water; 24hr)) found the following.

- 6% exceed HWR leachable arsenic
- 22% exceed HWR leachable cadmium
- 33% exceed HWR leachable lead
- 4% exceed HWR leachable zinc

Sample selection was biased towards the highest total metal concentrations, so the testing may have overestimated the percentage of leachable material.

Based on the results, there was an estimated 4,500m^3 (i.e. one third of upper cap soil volume estimate) of Leachable Toxic Waste (soil and mill dust) present at the Site that needed to be accommodated in the remedial plan.

REMEDIAL PLAN AND PERMITTING

The objective of remediation was to provide a robust, long term solution that was cost-effective and achievable, reducing the long term risk related to chemical contaminants. The selected remedial plan focused on reducing exposure to contaminated soils by placement of a permeable, erosion resistant cap over all of the contaminated materials. The proposed remedial actions at the Site involved construction of landfill and cover facilities, creating a Secure Landfill. The Secure Landfill encompassed soil covers on the Settlement and Tailings Ponds, the Upper and Lower Pads and the constructed landfill facility at the Mill location. The remedial plan included:

- upgrading the access road;
- removal of Hazardous Wastes building materials to an approved off-site disposal site;
- diversion of adit drainage from the settling ponds to an infiltration area and drainage of the ponds;
- building demolition and compaction on the existing mill building footprint;
- consolidation of waste rock/ore concurrently with layers of compacted demolition debris on the mill building footprint;
- on-site isolation and capping of building debris, mill equipment and associated metals contaminated soils with a minimum of one metre of clean compacted sand and gravel;
- *in-situ* isolation and capping, with a minimum of one metre of clean compacted sand and gravel, of the tailings pond, settling ponds, and high risk contaminated soils on the upper and lower pads; and
- risk management of remaining contaminated soils.

Under the CSR, this risk based approach was acceptable remediation approach, but did not eliminate the need for a permit under the HWR.

The HWR primarily focuses on generation, transportation and disposal of hazardous wastes, but contains specific exclusions if only mine tailings or mine waste rock are managed at the facility or if the site is a Historical Hazardous Waste Contaminated Site. Historical Hazardous Waste Contaminated Sites are defined as “any land or groundwater which (a) was contaminated with hazardous waste on or before April

1, 1988, and (b) is no longer used for any activity which adds new hazardous waste contamination to the land at that location." The Atlin Ruffner Mill and Tailings area met the definition of a Historic Hazardous Waste Contaminated Site. While exclusions applied to the tailings, residual ore and waste rock, the metal contaminated soils on site, a result of mining operations, were not excluded from specific HWR requirements.

The remedial plan met the definition of an In Situ Management Facility which "means a facility used to (a) prevent or control the movement or release of hazardous waste contaminants, or (b) treat or destroy hazardous waste contaminants in soil or groundwater at an historical hazardous waste contaminated site in such a way that the physical location of the hazardous waste contaminants and the soil is not substantially altered." A director's prior approval is required for construction and operations of an In Situ Management Facility.

The remedial action, through construction of an on-site Secure Landfill, provided protection to human or ecological receptors equal to the protection offered by the HWR. The remedial actions were designed to account for site-specific levels of risk and to allow remediation to proceed under suitable cost and time frames. However, the remedial plan did not meet some of the HWR requirements for a Secure Landfill, even given the Historical Hazardous Waste Contaminated Site exclusions.

Given the proposed approach and the unique circumstances at the site, we submitted an Application for a Change in Requirements under Section 51 of the HWR related to the Secure Landfill. Section 51 allows the director to approve site specific requirements that vary from the regulation. The hazardous building materials and fuels, including the friable asbestos containing materials, the painted debris with leachable lead, and the small quantities of residual fuel, grease blocks, batteries and other chemicals were managed, transported and disposed of at off-site approved facilities in accordance with the HWR and therefore were not part of the Section 51 application.

The application requested changes to the following sections to reflect the site conditions and risk managed remedial approach: Part 4 Division 6 – Secure Landfills and Sections (26) Operational Requirements and (27) Performance Standards. The requested changes to the HWR requirements were primarily related to requirements for a Secure Landfill for the Leachable Toxic Waste, such as a change in the requirement for a liner as wastes were remaining *in situ*, and a change in the frequency in monitoring to reflect the level of risk, location, and climate of the site. The site is remote and only snow free and accessible from May to September, making the standard HWR monitoring frequency requirements overly onerous. A specified but reduced monitoring frequency was proposed due to the passive nature of the on-site Secure Landfill facility and the remote location. Another example was a change in the requirement to construct the Secure Landfill under a cover or roof, given the material had been exposed to precipitation for many decades already. The Section 51 application required each changed line in the regulation to be replaced by wording applicable to the site.

OUTCOME

An approval for creation of an In Situ Management Facility and a change in requirements under the HWR was granted in August 2012. This approval jointly met both CSR and HWR requirements and was the first of its kind issued by the BC Ministry of Environment. It was based on four key criteria of the remedial plan:

1. the hazardous wastes (contaminated soils and mine wastes) remained in place and were covered with a minimum "one metre of clean gravel that is naturally erosion resistant";
2. as a contingency measure, the cover design allowed for potential installation of a geomembrane system at a later date if evidence arises that infiltration of rainfall is resulting in groundwater contamination issues due to leaching from the facility;
3. surface water was diverted away from the facility; and
4. access to the facility for public recreational use was discouraged by the placement of earthwork features and rocks.

Upon receipt of the approval, the following remedial works were undertaken in August and September of 2012:

- Upgrading the access road including installation of a temporary bridge over Trident Creek and subsequent decommissioning following remediation (required a *Water Act* authorization);
- Removal of specific Hazardous Wastes from the buildings (residual fuel, chemicals and asbestos building materials) with approved off-site disposal (required tracking as per the HWR);
- Permanent diversion of the adit drainage from the settling ponds to an infiltration area constructed to the north and east of the settling ponds and drainage of the ponds;
- Building demolition and compaction on the existing mill building footprint;
- Consolidation of waste rock concurrently with layers of compacted demolition debris on the mill building footprint;
- On-site isolation and capping of building debris, mill equipment and associated metal-contaminated soils with a geotextile and a minimum of one metre of clean compacted sand and gravel from onsite borrow areas;
- In-situ isolation and capping, with a geotextile and minimum of one metre of clean compacted sand and gravel, of the tailings pond, settling ponds, and high risk contaminated soils on the upper and lower pads; and
- Decommissions of access roads through berms and ditching and signage restricting access.

In consultation with the mineral tenure holder, two small low-grade ore piles were consolidate near the edge of the capped area, capped, and surveyed. This allows the potential for the ore to be accessed in the future with minimal need for cap repairs.

Salvage was looked as an option for the mill equipment remaining on site but was not economic. The mill building was demolished, compacted, and capped in its original footprint. The remedial plan made use of the existing topography of the site. Removal of a waste rock retaining adjacent to the former mill and natural, uncontaminated hills used as borrow sources on either side of the mill allowed the area to be reformed into a continuous slope over the demolished mill building (Figure 2, Photos 4 and 5).

Given the remote location, clean sand and gravel from three undisturbed hills was used as a borrow source for the cover material. Given the source of the cover material, it was difficult to visually differentiate contaminated soils from the clean cover. A geotextile liner was used to cover all contaminated material prior to capping to ensure the two materials weren't mixed. It also proved to be invaluable to prove the cover thickness following a survey error, as it clearly showed the cap thickness in test pits.

Monitoring as per the requirements of the approval is ongoing. The cover must be inspected for overall integrity and performance at least once a year for the first five years and once every five years subsequently. Groundwater monitoring and sampling of compliance wells for dissolved metals is required on the same schedule, with reports provided to MOE annually. The groundwater monitoring is particularly important to ensure attenuation continues as expected. The monitoring frequency density has been higher than required under the authorization. Revegetation efforts will commence once it is confirmed that the addition of an impermeable cover is not required.

SUMMARY

Remediation of Historic Mine Sites must comply with the requirements of the British Columbia *Environmental Management Act*, the Contaminated Sites and Hazardous Waste Regulations and associated Protocols, Policies, Procedures, and Technical Guidance Documents issued by the Ministry of Environment. Remedial plans need to be developed taking site specific conditions into account. At Atlin Ruffner mill and tailings site, a remedial plan that successfully risk managed contamination under the CSR still required permitting under the HWR. Site specific conditions suggested a variance from the regulated conditions could provide equal protection to human or ecological receptors as offered by the HWR. Application for specific changes to the regulation was made and approval was granted.

The optional remedial solution for any historic contaminated site needs to be assessed on a site-specific basis with clear objectives and an understanding of the regulatory requirements. Investigating options for site specific regulatory changes may allow completion of remediation in a timelier and more cost effective manner, freeing up resources for the remediation of other sites.

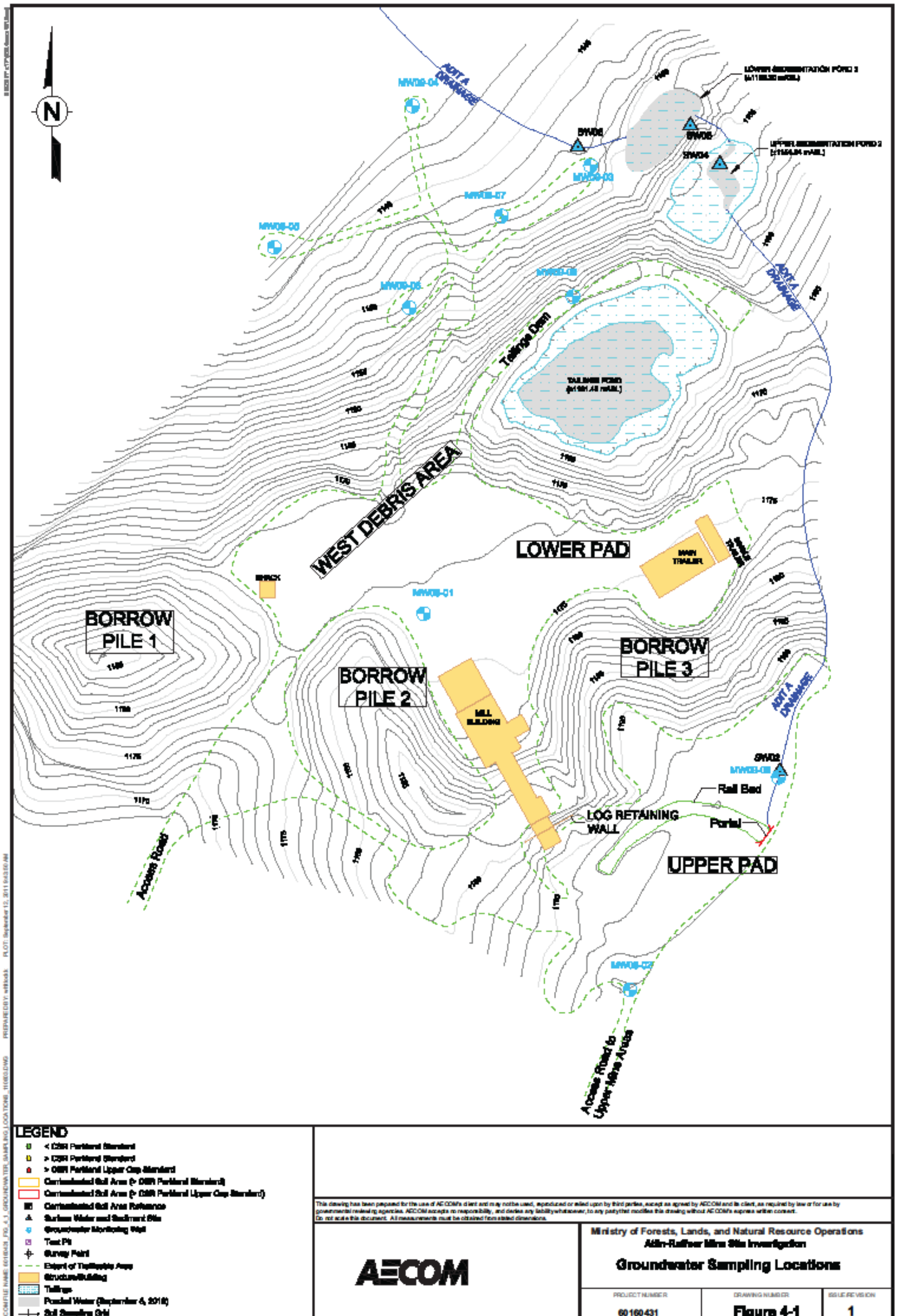


Figure 1: Site Plan Prior to Remediation of Atlin Ruffner Mill and Tailings

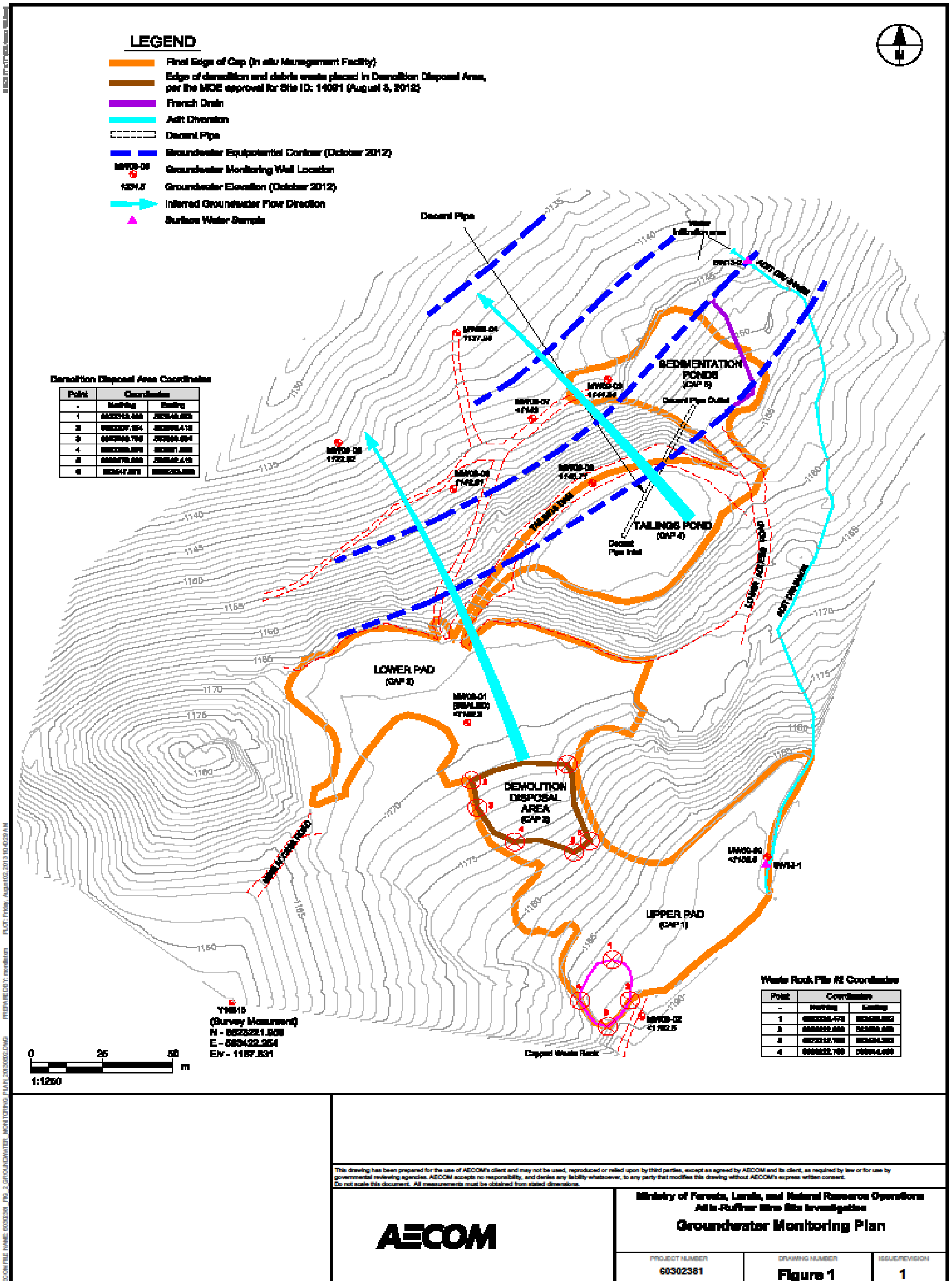


Figure 2: Site Plan Following Remediation of Atlin Ruffner Mill and Tailings

Photo 1: Adit, mill building, trailers, upper and lower mill pads, and portion of tailings pond prior to remediation (October 2008).

Photo 2: The settling ponds seen from the top of the tailings dam, prior to remediation (October 2008).

Photo 3: The lower mill pad, with the tailings and lower settling pond in distance prior to remediation (October 2008).

Photo 4: The site post-remediation, with the tailings area in the centre. (June 2013)

Photo 5: The site post-remediation, with the former mill area and access road in the foreground. (June 2013)

FARO MINE COMPLEX REVEGETATION ACTIVITIES

L. Carter, B.Sc., M.R.E.M.
P. Tobler, R.P.Bio., CPESC

EDI Environmental Dynamics Inc. (EDI)
2195 – 2ND Avenue,
Whitehorse, YT Y1A 3T8

ABSTRACT

In preparation for remediation and closure of the Faro Mine Complex (FMC) in central Yukon, significant work has been completed to develop and test revegetation methods at the mine site. Design and implementation of various revegetation field trials and strategies has been proceeding since 2007. This paper focuses on revegetation efforts and subsequent monitoring at two FMC sites: the Grum Overburden slope and Grum Sulphide Cell (GSC).

Grum Overburden slope revegetation trials cover an area of approximately two hectares with surficial soil characterization similar to material to be used as a reclamation cover for the entire FMC. Different grass seed mixes were applied with and without fertilizer, and woody species (alder, willow, and poplar) were planted in the seeded plots. Five years of monitoring has demonstrated that revegetation success primarily depended on fertilization, while site preparation method heavily influenced erosion protection.

Revegetation and site preparation at the 26 ha GSC was intended to provide ground cover for erosion protection and facilitate development of a long-term, self-sustaining ecological system integrated with the mine surroundings. Building on results from the Grum Overburden site, the 2012 revegetation prescriptions included hydroseeding, fertilization, planting of woody species, and testing of fertilization and hydration teabag-like packs. Results from the 2013 and 2014 monitoring programs provided preliminary insight on the success of the revegetation works at the GSC site.

Collectively, these sites signify many of the reclamation challenges faced by development projects across northern Canada, including a short growing season, harsh climate, and nutrient poor soils. The need to address multiple objectives across various time scales provides many research opportunities.

Key Words: Fertilizer; nurse and native seed; site preparation; hydroseeding; northern mining

INTRODUCTION

Yukon Government - Assessment and Abandoned Mines (YG-AAM) branch is responsible for planning for remediation and closure of the Faro Mine Complex (FMC), a large abandoned lead, zinc, silver and gold mine site in central Yukon. In preparation for closure, research has been conducted to test revegetation methods at the mine site. The short growing season, nutrient poor soils and the physical properties of the soils all combine to make establishment of vegetation at the FMC challenging. Therefore, establishing vegetation in FMC soils requires planning, care, and maintenance to ensure revegetation success. Revegetation must advance quickly to achieve the short-term goals of mitigating run-off (e.g. reducing run-off velocity) and reducing soil erosion but also must set the stage for natural succession trajectories.

Revegetation activities and trials have been conducted by EDI Environmental Dynamics Inc. (EDI) at two sites at the FMC: the Grum Overburden site and the Grum Sulphide Cell (GSC). Grum Overburden trials (~2 ha; 2009) were designed to identify the most effective combination of surface treatments and revegetation options to mitigate run-off and slope erosion. The GSC revegetation activities (~26 ha; 2012) were conducted using knowledge gained from outcomes at the Grum Overburden trials. Additional variation/experimentation was included to refine, evaluate, and provide further direction for future revegetation practices at the FMC.

This paper provides an overview of revegetation activities conducted and results and observations from the Grum Overburden trials and GSC revegetation treatments implemented in 2009 and 2012, respectively.

METHODOLOGY

Grum Overburden Trials

The Grum Overburden site was selected as a test site because surficial soil characterization and slope were considered to be representative of materials/topography intended for final cover during reclamation of the FMC. At an average altitude of 1,285 m, the site is approximately 260 m wide with a 70 m long slope and a gradient of about 3:1. The following objectives were addressed in the Grum Overburden trials:

- Determine vegetative cover for various grass seed mixes (with and without fertilizer);
- Evaluate erosion protection potential provided by prescribed application rates of grass mixes;
- Evaluate progress of inter-planting grass mixes with woody species using different methods/species;
- Evaluate planting and propagation methods and site suitability for large-scale implementation; and
- Evaluate efficacy of physical surface treatment options for erosion protection and vegetation establishment.

In a randomized block design, different grass seed mixes were applied with and without fertilizer, and woody species (alder, willow, and poplar) were inter-planted in the seeded plots. All test plots were established in the fall of 2009 and the following variables were included in the revegetation trials:

- Three grass seed mixes (agronomic, native, and nurse and native);
- Three woody plant treatments (horizontal and vertical willow and poplar stakes and alder seedlings), including some seeding of spruce and dwarf birch collected at site;
- Two fertilizer treatments: unfertilized and fertilized (8-38-15), 400 kg/ha and 200 kg/ha for initial year (2009) and first growing season (2010), respectively; and
- Three soil surface treatments (micro-rill, planar, and rough-and-loose).

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

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

During monitoring from 2010 to 2013, percent vegetation cover was estimated visually in each plot. In 2011 to 2013, percent cover was visually estimated using a rectangular quadrat (0.5 m²), six subsamples per plot, and calculating average percent cover.

Table 1 - Summary of seed mix and application rates

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

All live stems of poplar, willow, alder, and other woody plants were also counted in each plot. Live stems are defined as any shoot, or cluster of shoots, emerging independently of other shoots. Monitoring results continue to indicate that horizontal staking of willow/poplar produces more stems than vertical staking, over time. Horizontal staking is also less labour intensive than vertical staking for the number of stems produced. Alder planting of plugs (from seed collected onsite) appeared to be the most successful method of establishing woody stems.

<u>Grum Sulphide Cell Revegetation</u>

Key treatments in the 2012 GSC revegetation program were guided by outcomes from the Grum Overburden trials and included the following treatments:

- Nurse and native seed mix;
- Fertilizer included in hydroseeding;
- Surface treatment similar to rough and loose, ripping across the slope (more cost effective and achieved similar results);
- Nursery grown plugs from local plant materials; and,
- Horizontally staked willow and balsam poplar cuttings.

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

TerraStar Solutions Inc., a B.C. based company, applied the hydroseed from August 11 – 13, 2012. A native seed mix was used with annual rye grass as a nurse crop. The seed mix consisted of the following grasses (% by weight):

- 54.1% Slender Wheatgrass
- 17.1% Northern (Rocky Mountain) Fescue
- 0.5% Glaucous Bluegrass
- 7.1% Tufted Hairgrass
- 21.2% Annual Rye Grass

The seed was applied at a rate of 35 kg/ha over the 26 ha GSC. Additives to the hydroseed mix included fertilizer (18-18-18; 400 kg/ha) and mulch/tackifier mix (Hydrostraw® Guar Plus).

Treatments

Native woody vegetation plugs and cuttings were planted at the GSC not only for long-term erosion control and site stabilization, but also to initiate restoration of the site to a more natural state, along local successional trajectories. More than 14 ha of the 26 ha GSC were planted with woody species in either experimental plots or general planting plots. Specific treatments are as follows:

Plugs Only (P plots)

Woody vegetation plugs (willow, balsam poplar, birch, and alder) were planted without any amendments. The 'plugs only' treatment acts as a control for the treatments involving teabags. The treatment can also be compared to treatments that include horizontal staking.

Staking (S Plots)

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

Plugs and Staking (PS plots)

Planting stakes with plugs supplemented the plug numbers and potentially increases the number of resultant stems in the plot; thus, increasing density of woody vegetation. Stakes were planted alternately with plugs, both across and with the slope.

Fertilizer-packs (Pf plots)

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

Hydration-packs (Ph plots)

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

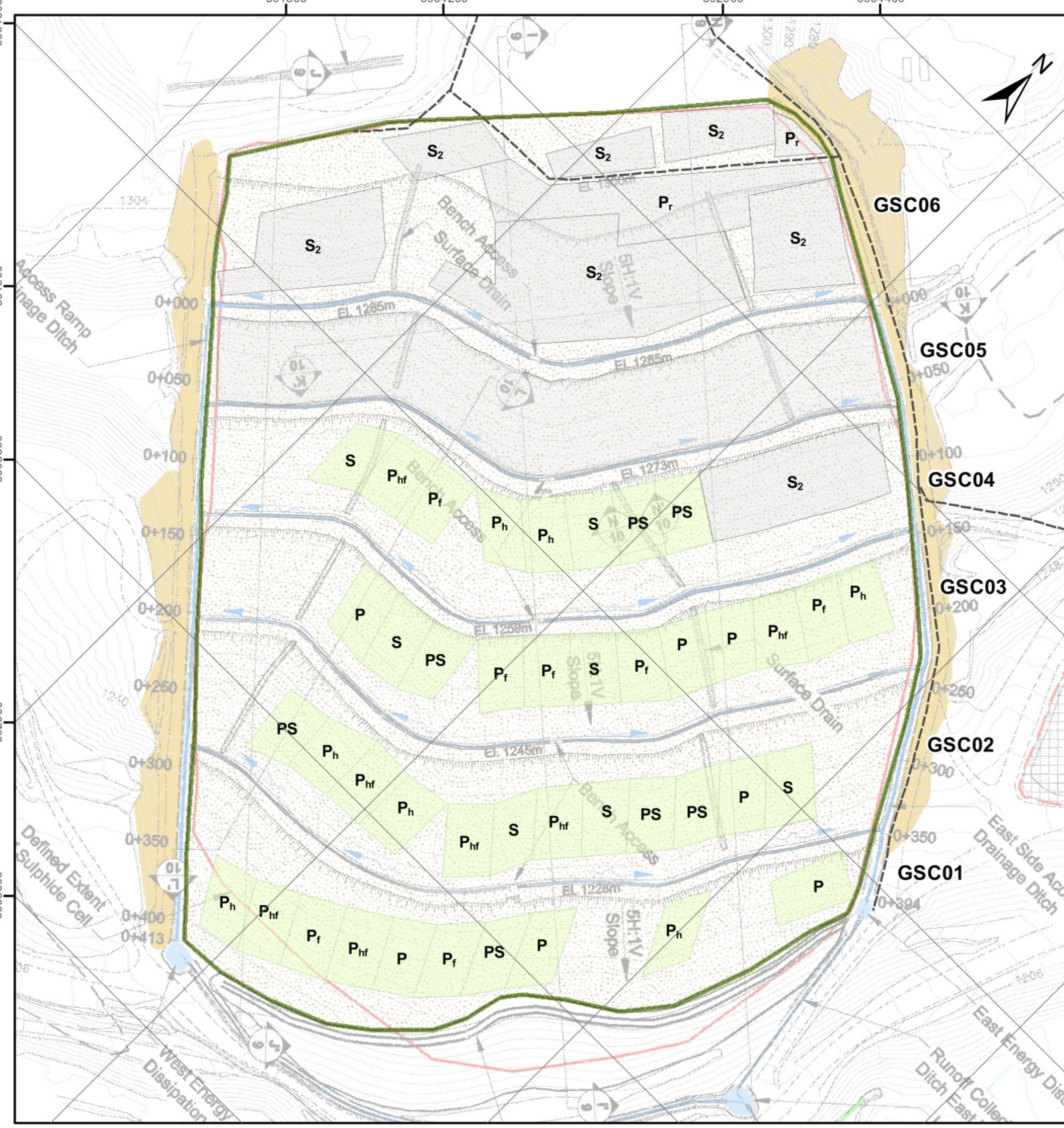
Genesis-pack (Phf plots)

One 40-gram Genesis-pack was placed in the planting hole and broken open prior to planting the plug. Genesis-packs release the following ingredients over twelve months: total nitrogen 8.00%; available phosphoric acid 5.00%; soluble potash 5.00%; sulfur 2.00%; boron 0.037%; humic acid (leonardite-derived) 7.5%; kelp meal (*Ascophyllum nodosum*) 5.0%; composted vegetative matter 30%.

General Shrub/Tree Planting Plots

In addition to the experimental plots, larger plots were set up in areas to add woody plant material treatments on a larger scale. The following treatments were implemented:

- S2 Plots: poplar and willow stakes were buried at a spacing of 2.5 m between centre points of stakes.
- Pr Plots: This area was planted with plugs that remained after all experimental plots were planted. Approximately 3,354 plugs were planted on the uppermost slope in a random arrangement. Some areas were planted at very high densities to simulate a more natural setting; other areas were planted with equidistance.



Legend

- Access Road
- Experimental Shrub / Tree Planting Plot
- General Shrub / Tree Planting Plot
- Treatment Area - Site Prep, Fertilizer & Hydroseeding

Label codes

P	Plugs
S	Stakes
PS	Plugs & Stakes
f	Fertilizer Paks
h	Hydration Paks
hf	Genesis Paks
r	Random Spacing
2	2 m Spacing

Grum Sulphide Cell Revegetation Works

Data Sources

Project data displayed is site specific. Access roads and plot data collected by EDI Environmental Dynamics Inc. (2012) and was obtained using Garmin GPS technology.

1:250,000 topographic spatial data provided by Geomatics - Yukon Government via online source (Corporate Spatial Warehouse) www.geomaticsyukon.ca.

National Road Network courtesy of Her Majesty the Queen in Right of Canada, Department of Natural Resources. All Rights Reserved.

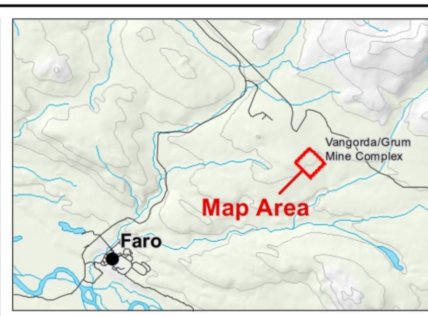
Background image provided by SRK Consulting.

Disclaimer
This document is not an official land survey and the spatial data presented is subject to

0 20 40 60 80 100
metres

Map Scale 1:4,000 (printed on 8.5 x 11)
Map Projection; North American Datum 1983 UTM Zone 8N

Drawn:	Checked:	Date: 28/08/2014	FIGURE 2
LG	MP / PT		



Yukon Energy, Mines and Resources
Énergie, Mines et Ressources

EDI

Path: J:\Yukon\Projects\2012\12_Y_0197_AAM_Faro Mine Complex Revegetation Activities 2012\Mapping\Fig2_GSC_RevegetationWorks_28Mar2013.mxd

RESULTS AND DISCUSSION

Grum Overburden Trials

Results of the 2013 Grum Overburden revegetation trial monitoring found that fertilizer application is necessary to establish herbaceous vegetation at the FMC. Unfertilized plots had minimal ground cover (less than 5%); vegetative cover on fertilized plots ranged from 5 to 24% in 2013. Results from 2013 indicated that the agronomic seed mix resulted in higher vegetative cover in two of three fertilized surface treatments and higher cover from the native seed mix in the third fertilized surface treatment. The 2013 results indicated that the native grasses could establish as successfully as other seed mixes, but needed more time to develop.

Regarding woody plant survival, it is unclear whether fertilizer application influenced success of willow/poplar stakes; however, alder establishment appeared to be negatively correlated with fertilizer application. For example, stem counts of alder in fertilized plots were an average of 50% lower than in unfertilized plots. This divergence may be attributed to competitive effects between alder and the grass cover. Results also showed consistently higher stem counts for horizontal staking in contrast to vertical staking.

Of all the surface treatments at the Grum overburden, the most effective at controlling erosion due to run-off was the rough-and-loose surface treatment, even in the areas with minimal vegetative cover. This treatment created micro-sites (depressions) providing areas for pooling water, ideal for germinating seeds; consequently, run-off was prevented (i.e. enhancing infiltration). The depression micro-sites appeared to be areas of competition between herbaceous vegetation and woody plants but most of the grass has now died off. Very little growth was observed in the rough-and-loose treatment on the upper parts of the hummocks.

The amount of vegetation cover established during these trials was not at a level that will meet the objectives of controlling erosion and stabilizing slopes alone. Nonetheless, the Grum Overburden trial results indicate that the rough-and-loose treatment with seeding, horizontal staking and fertilization can be key parts of a successful revegetation and erosion control prescriptions.

Grum Sulphide Cell

Monitoring at the GSC was conducted in mid-August 2013 and 2014. Data were gathered to determine revegetation success of herbaceous and woody species; site stability. The following activities took place as components of the GSC monitoring program.

- Soil sampling;
- Estimation of herbaceous vegetation cover;
- Assessment of woody species establishment (stems per hectare, health and vigour, height);
- Photo point monitoring;

Overall grass cover was not as high as expected, in 2013, likely due to a relatively dry spring. In 2014, overall cover from herbaceous species increased from 7.7% in 2013 to 20.7%. Cover was highest in the deeper cross-slope furrows of the slope where moisture appeared to be highest. Vegetative cover is still below a level to control erosion and stabilize slopes alone; however, combined with the site preparation, site erosion was minimal.

In 2013, all treatments of woody species planting resulted in higher densities than unplanted areas. Treatments with staking seemed to provide the best overall results with the highest average survival rate, better health and vigour ratings, and greater than average heights. Teabag amendments appeared to increase height and vigour of grass growing around woody species plug, potentially reducing survival of woody species due to competition.

In 2014, moss was commonly observed across the GSC and provided substantial cover in some areas to assist in soil/slope stabilization. Considerable natural regeneration of woody species was also observed and recorded throughout the site and appeared to be more prevalent in the areas of mossy growth. Woody stem planting appears to have been moderately successful; on-going data analysis will determine the success of individual planting treatments. Overall, the site preparation combined with the seeding/fertilizing prescription appeared to be effective in controlling erosion, while still allowing for the natural establishment of native mosses, forbs and woody stems.

GROWING OUR FUTURES: COMMUNITY-BASED TRAINING IN NATIVE PLANT HORTICULTURE FOR ABORIGINAL COMMUNITIES

Carla R. Mellott, MSc. AAg.[1]
Michael E. Keefer, MSc. PAg.[1]
Tim Brigham, MA[2]

[1] Keefer Ecological Services Ltd., 3816 Highland Road, Cranbrook, BC VIC 6X7
[2]Continuing Studies, Royal Roads University, 2005 Sooke Road, Victoria, BC V9B 5Y2

ABSTRACT

An increased demand for reclamation using native plants in British Columbia has created opportunities for stable, long-term economic opportunities in Aboriginal communities through employment in native plant horticulture. The ***Growing our Futures: Native Plant Horticulture*** training program was created collaboratively by staff at Royal Roads University, Keefer Ecological Services Ltd, West Moberly First Nations and Saulteau First Nations in order to provide training to Aboriginal communities in native plant seed collection, propagation and nursery management. The program was piloted in 2013 at **Twin Sisters Native Plant Nursery** located in West Moberly BC, and **Tipi Mountain Native Plants** located in Cranbrook, BC. Nine students from Twin Sisters nursery and seven students from Tipi Mountain nursery completed the *Growing our Futures* program. Following consultation with the leadership and other representatives of Saulteau and West Moberly First Nations, it was determined that Growing Our Futures should be developed for community-based delivery to support broad student participation, retention and success through enabling students to learn within their family, community and Elders support networks. In this paper, we provide an overview of the 2013 pilot year of the ***Growing our Futures: Native Plant Horticulture*** training program, discuss successes and lessons learned, and describe future directions for this program in 2014 and beyond.

KEY WORDS

Education, Indigenous, Native Plants, Propagation, Reclamation

BACKGROUND

Site reclamation necessitated by natural resource development activities is shifting away from the use of agronomic plants toward a biodiversity model aimed at re-establishing native plant cover. The use of native plants in reclamation has the potential to provide myriad environmental and social benefits and is supported by many Aboriginal communities interested in seeing the land restored with biologically and culturally appropriate plant communities. At the same time, there is a lack of skilled labour in rural British Columbia able to meet the growing demand for native plants. Trained personnel are needed across the reclamation cycle from developing reclamation plans, to producing the required native plant material, to the implementation of reclamation plans on disturbed sites. As one of the few rural populations in Canada currently experiencing population growth, Aboriginal communities have the potential to help address these and other labour shortages currently being faced in many northern and rural areas (Agriculture and Agri-Food Canada, 2011).

Many large-scale natural resource projects across Canada are located near Aboriginal communities (Government of Canada, n.d.), and the natural resource sector is the leading private sector employer of Aboriginal people (Agriculture and Agri-Food Canada, 2011). At the same time, rural Aboriginal people currently experience an unemployment rate nearly eight percentage points higher than non-Aboriginal people living in rural Canada (Statistics Canada, 2006). Programs and policies that provide training to Aboriginal people for local employment or entrepreneurial opportunities in the native plant horticulture and reclamation sector arguably make good economic, social and environmental sense by providing employment gateways to a growing Aboriginal labour force while increasing the supply of native plant material available for reclamation.

Growing our Futures is a training program in native plant horticulture developed in 2013 by Royal Roads University (RRU), Keefer Ecological Services (KES), Saulteau (SFN) and West Moberly First Nations (WMFN). The Program was designed to create capacity for native plant propagation within Aboriginal communities, facilitating greater participation by Aboriginal people in native plant reclamation and restoration programs occurring within their Territories. One of the key features emphasised by the leadership and other representatives of our First Nations partners involved in the *Growing Our Futures* program was in-community delivery. Aboriginally-centered, community-based training programs have been described as a means of increasing the participation, engagement and retention of Aboriginal students (Inuit Tapiriit Kanatami. 2007; Saskatchewan Ministry of Education, 2009).

In 2013, *Growing Our Futures* was piloted at two separate First Nations-owned native plant nurseries. Twin Sisters Native Plant Nursery, a joint venture of the Saulteau and West Moberly First Nations, was opened in 2013 at Moberly Lake in the Peace region of Northeastern BC. The students participating in the Twin Sisters program came from a variety of backgrounds but the great majority (over 90%) had no pre-existing nursery experience and none had any experience growing native plant species. Mountain Native Plant Nursery, owned by a member of the Ktunaxa First Nation, is a fully functional nursery that has been in operation under its current ownership since 2008. The students participating in the Tipi Mountain program were largely existing nursery workers wishing to broaden and deepen their skills and theoretical knowledge in native plant propagation. Many students from both groups possessed extensive traditional knowledge relating to the identification and typical uses of many local plant species. Tipi Nine students from Twin Sisters nursery and seven students from Tipi Mountain nursery completed the *Growing our Futures* program in December, 2013.

The response from students participating in the 2013 *Growing Our Futures* pilot program was extremely positive. As one student stated: “I never learned so much in my life – I love it.” Another student stated that the training has more than lived up to her expectations: “I am really happy and feeling very blessed to be a part of this. My mind has never felt so full in a good way!” All of the participating students made major measureable gains in their theoretical knowledge and in their development of practical nursery skills as demonstrated through the completion of a number of assignments in native plant identification, plant morphology, and plant propagation. A 2014 delivery of Growing Our Futures is planned for Southern Vancouver Island with the Tsawout First Nation.

PROGRAM ASESSMENT

Fundamental to the continued success of the *Growing Our Futures* program is a commitment to critical assessment and improvement of the program to ensure that it continues to serve the needs of participating Aboriginal students and communities. The remainder of this paper provides a qualitative assessment of our 2013 pilot programs at Tipi Mountain Native Plants and Twin Sisters Native Plant Nursery based on our experience as program planners and instructors. This assessment focuses on the extent to which the *Growing Our Futures* partnership delivered a program that reduced financial and logistical barriers for students; increased community collaboration; engaged community support for students, incorporated cultural knowledge and Elder involvement; made use of a flexible admission model; created bridges to employment opportunities for students; and implemented an innovative delivery approach informed by current Aboriginal models of teaching and learning.

Reduced Financial and Logistical Barriers to Accessing Post-Secondary Education

Although not exclusively targeted at rural Aboriginal communities, *Growing Our Futures* will be largely delivered in rural Aboriginal communities located in proximity to natural resource development. For this reason, *Growing Our Futures* was designed for community-based delivery, an approach that may reduce a number of the financial and logistical barriers experienced by rural Aboriginal students attending post-secondary programs in urban areas (Saskatchewan Ministry of Education, 2009). Delivering *Growing Our Futures* within host communities during our 2013 pilot deliveries avoided the need for students to relocate to access the training. Working closely with the First Nations communities involved in both pilot offerings, the *Growing Our Futures* partnership was also able to access substantial funding to support delivery costs, offer free tuition to students and provide financial support in the form of stipends and wage subsides for students, all of which may not have been possible outside a community-based training model. Delivering the program *in situ* also allowed students to make use of their existing support networks while attending the program. For example, students with children could make use of existing childcare arrangements that would not have been possible had they been required to move to a new location to access training.

Increase Opportunities for Collaboration with Aboriginal Communities

Community collaboration was positively enhanced through the use of the community-based delivery model. As discussed in the previous section, working with communities allowed us to access funding to both offer free tuition and to provide financial support to students. Our First Nations partners, Saulteau First Nations, West Moberly First Nations and Ktunaxa Nation Council also provided support and guidance with respect to intake, enrollment and reporting to the various funding agencies and logistical support with respect to room bookings and organizing program events. Other forms of collaboration with respect to curriculum development and modification were more challenging in both programs given the myriad responsibilities and limited time available to education coordinators and other staff in the participating communities. Community collaboration with respect to curriculum development and modification in future deliveries will be greatly enhanced if funding is provided to support the greater involvement of Aboriginal community administrators and contributing staff. In our 2014 delivery we have included budget provisions to pay for the contributions of both an education director/coordinator as well as a community life coach, both of whom will provide culturally appropriate student support and feedback throughout the program.

Effectively Incorporated Cultural Knowledge and Support for Elder Involvement

Elder involvement in training programs has been identified as a key element supporting Aboriginal student success (Inuit Tapiriit Kanatami, 2007; Saskatchewan Ministry of Eduacation, 2009; Assembly of First Nations, 2012). Elders can provide both culturally-appropriate student support as well as cultural enrichment, especially within a program that includes a focus on a number of plant species with cultural importance. During the TMNP training, for example our students spent three days with Wayne McCoy, a Ktunaxa Elder with substantial plant knowledge. In our 2014 delivery with the Tsawout First Nation, we have included a budget provision to support weekly workshops with community Elders.

We found it difficult to incorporate additional cultural knowledge outside of workshops and outings with Elders. An attempt to present both Western and Traditional forms of taxonomy, for example, with the intent to provide recognition and respect for the different yet equally valid ways of classifying and organizing organisms in the natural world, was not well-received when packaged in a very Western mode of teaching (a lecture) and presented by non-Aboriginal instructors. Our lesson from this experience is that cultural knowledge included in the program must arise from community members themselves. Future programming will be improved by creating space for student activities and discussions centred on the cultural knowledge held by students, Elders and guests. We believe spending as much time as possible in natural spaces and interacting directly with plants, will inspire deeper collaboration among students and instructors. Involving community education directors/coordinators as much as is possible (while, at the same time, respecting their time) will help identify locally appropriate approaches to incorporating cultural activities into the training program.

Effectively Utilize Flexible Admissions

Our experience is that skilled workers in native plant horticulture may come from myriad educational and experiential backgrounds and that the knowledge and skills that students can bring to a training program or place of employment may not be adequately captured by records of their prior educational history and/or other achievements. For this reason *Growing Our Futures* employs a flexible admissions model, wherein achievement of high school diploma or any other prior certification is not required for entry into the program. Admissions are instead focused on attitude, enthusiasm and life experience. Providing flexible admissions based on life experience has been recommended by various entities including the Saskatchewan Ministry of Education (2009).

Students in our 2013 pilot programs came from a variety of educational backgrounds and all brought a wealth of experience and knowledge to the program. A major lesson in the effective utilization of the flexible admissions model was the need to identify the essential skills for employment success in native plant horticulture and provide remediation to students where necessary. A lesson in calculating germination tests that involved ratios, for example, was not possible for some students in our 2013 offering with very limited math skills. In our 2014 delivery, we plan to address this by having instructors, the educational coordinator and life coach work with each student at the outset of the program to produce an individualized learning plan, which will include additional skills development where necessary. Time will be built into the program to support students in attaining the goals set out in their individualized learning plans.

Increased Opportunities for Work Experience and Networking

Both *Growing Our Futures* 2013 pilot programs were hosted by Aboriginally-owned native plant nurseries. Tipi Mountain students were largely existing employees of TMNP, and this training was offered to increase their skill, understanding, effectiveness and engagement with their work. The majority of these students continued with their employment at the nursery after they completed the program. Twin Sisters Native Plant nursery was newly opened in 2013 and five Twin Sisters students were employed by TSNP after completing the program. Periods of paid work-experience were interspersed among the six weeks of training in both projects, including nursery work at both locations and a seed collection contract at TSNP, allowed students to put their learning into practice.

In spite of these successes, we felt that we could make better use of the community-based delivery model in future deliveries to allow for increased networking as well as increase the employment skills training offered to students to better support them in finding and keeping meaningful employment. In 2014, workshops from the already proven and successful RRU-Continuing Studies Employment Readiness Access Program will provide over two weeks of employment skills training designed to help students transition from learning to employment by enhancing communication, leadership, management, problem solving, team-building, entrepreneurial and computer skills. Our 2014 offering will also involve field visits to local places of employment, which will serve the dual purpose of allowing students the opportunity to network while increasing potential employers' familiarity with the *Growing Our Futures* program.

Implemented Innovative Delivery Based on Current Aboriginal Education Models

Aboriginal students have demonstrated greater success in programs that are student-focused, cooperative and hands-on (Saskatchewan Ministry of Education, 2009). Not surprisingly, students in our 2013 pilot programs responded far better to active learning (activities and assignments) compared to passive learning (lectures and demonstrations). Students were also extremely engaged when working in groups toward a common goal of producing materials that demonstrated their knowledge. For example, we found that student-led group presentations were great opportunities for learners to review key concepts and to take real pride in what they had learned and what they had to share with other students. Asking the students to work on group presentations also helped with team building and, in many cases, will reflect the reality of their future workplace(s). In future offerings we aim to increase opportunities for student curiosity, imagination and initiative to guide their learning by having the students work in groups or individually to explore topical questions followed by the opportunity to share their learning with the rest of the class.

FUTURE DIRECTIONS: SEED TO SITE

After developing the *Growing Our Futures* program, our advisory board and industry partners identified a need for an increased number of trained personnel with the technical skills to participate across the entire native plant reclamation cycle: planning for reclamation, seed collection, growing out the required plant material, reclamation planting using native plants, and monitoring reclaimed sites. *Seed to Site: Native Plant Reclamation training* aims to link Aboriginal people with meaningful employment opportunities in natural resource reclamation in their home communities as well as support the development of Aboriginal-owned enterprises.

As with *Growing Our Futures,* the *Seed to Site* curriculum will be developed by the Growing Our Futures Partnership, which includes professionals from Royal Roads University, Keefer Ecological Services and representatives of our Aboriginal partners associated with Twin Sisters Native Plants Nursery. Additionally, we will also seek guidance from an advisory board composed of professionals in the native plant reclamation and natural resource development sectors, aboriginal representatives, and specialists in adult aboriginal education. Following the lessons we have learned with developing and delivering the *Growing Our Futures* program, *Seed to Site* will use an aboriginal education model that includes face-to-face, community-based instruction, flexible admissions, life coach and Elder support, culturally enriched content, hands-on learning, and the creation of personalized learning plans for essential ~~and~~ employment skills.

Once developed, *Seed to Site* can be delivered separately or concurrently with *Growing Our Futures* depending on the needs of the participating community. Both *Growing Our Futures* and *Seed to Site* can be delivered throughout British Columbia, and exported across Canada and internationally, while still employing British Columbians for program planning and instructional delivery. Our hope is to eventually build an instructional assistant team from previous *Growing Our Futures* and *Seed to Site* graduates in order to provide further employment opportunities to graduates and to provide direct graduate mentorship for future Aboriginal students.

CONCLUSION

Growing Our Futures works with Aboriginal students within their home communities to engage their knowledge, enthusiasm and passion for native plants while at the same time providing the necessary skills to successfully obtain and maintain employment in the native plant horticulture sector. Our objectives are to address the lack of rural capacity in native plant horticulture, enhance Aboriginal communities' ability to meaningfully participate in reclamation, and support positive and lasting relationships between First Nations and industry. Through the 2013 pilots we learned a number of important lessons about how adult Aboriginal students learn, the need for a strong cultural component in the program, the need to address gaps in employment skills, and the benefit of creating personalized training plans to assist students in achieving their educational and employment goals. Most importantly, we understand the need for continual program assessment and modification to ensure that *Growing Our Futures* remains responsive to the needs of both Aboriginal students and participating Aboriginal communities.

ACKNOWLEDGEMENTS

The development and delivery of the pilot program at Twin Sisters Native Plant nursery was supported by Agriculture and Agri-Food Canada through the Canadian Agricultural Adaptation Program (CAAP), delivered in British Columbia by the Investment Agriculture Foundation. Additional funding was provided by Walter Energy, the North East Native Advancing Society (NENAS), Aboriginal Affairs and Northern Development Canada (AANDC), Teck Resources, and cash and in-kind contributions from the development partners, West Moberly and Saulteau First Nations. Program delivery at Tipi Mountain nursery was supported through the Aboriginal Training for Employment Program (ATEP) and the British Columbia Aboriginal Mine Training Association (BC AMTA). Our 2014 delivery of Growing Our Futures is supported by the Aboriginal Community-Based Delivery Partnerships Program (ACBDPP). Special thanks to Alex Canning (Education Manager, West Moberly First Nation) and John Lewis (Director of Operations, West Moberly First Nations) for their thoughtful review and contributions to this paper and to Alex, John and Naomi Owens (Lands Manager, Saulteau First Nations) for support and collaboration throughout the development and delivery of Growing Our Futures.

LITERATURE CITED

Agriculture and Agir-Food Canada. 2011. Community Information Databased. Accessed July 02, 2014 from http://www.cid-bdc.ca/rural-facts?page=4

Assembly of First Nations. 2012. Supporting First Nations learners transitioning to post-secondary. Assembly of First Nations: Education, Jurisdiction, and Governance. Accessed July 05, 2014 from http//www/afn.ca/uploads/files/education2/postsecondarytransitions.pdf

Government of Canada. N.D. Canada's Economic Action Plan: Aboriginal people's participation in Canada's resource economy. Accessed July 02, 2014 from http://eap.gc.ca/en/backgrounder/r2d-dr2/aboriginal-peoples-participation-canadas-resource.

Inuit Tapiriit Kanatami. 2007. Post-secondary cases in Inuit education: Discussion paper no. 2 for the National Inuit Education Summit. Prepared by Silta Associates

Saskatchewan Ministry of Education. 2009. Post-Secondary Education: In Support of First Nations and Inuit Students. University of Saskatchewan, Aboriginal Education Research Centre, Saskatoon, SK & First Nations and Adult Higher Education Consortium, Calgary AB. Accessed July 02, 2009 from www.aerc.usask.ca

Statistics Canada. 2006. Census of Population, Statistics Canada catalogue no. 97-560-XCB2006031 (Canada, Code01). Accessed July 02, 2014 from http://www5.statcan.gc.ca.

TOWARDS AN ECOHYDROLOGIC CLASSIFICATION OF RECLAIMED WATERSHEDS: METHODS FOR ESTIMATING SOIL WATER REGIME ON RECLAIMED MINE WASTE MATERIALS; AND RELATIONSHIPS BETWEEN RECLAMATION AND SURFACE WATER BALANCES IN TECK'S RECLAIMED COAL-MINING WATERSHEDS

J. Straker[1]
M. O'Kane[2]
S. Carey[3]
T. Baker[1]
D. Charest[4]
R. Shurniak[2]

[1]Integral Ecology Group Ltd., Duncan, B.C. jstraker@iegconsulting.com
[2]O'Kane Consultants Inc., Calgary, AB
[3]McMaster University, Hamilton, ON
[4]Teck Resources Limited

ABSTRACT

The concept of a "soil moisture regime" or soil water regime is used worldwide to understand edaphic conditions, plant communities, and water balances. In particular, in western and northern Canada, Biogeoclimatic Ecosystem Classification ("BEC", B.C. and Yukon) and Ecological Land Classification ("ELC", AB and SK) use soil moisture regime as one of two primary variables in edaphic or edaptopic grids used to describe ranges of soil conditions and naturally occurring plant communities characteristics of these conditions. In most applications, soil moisture regime is a relative or unquantified parameter estimated from the presence of indicator plants, or from dichotomous keys using surficial material/soil properties observed in natural ecosystems. Applications of these estimation approaches to post-mining landforms and watersheds is challenging, because a) there are often few or no indicator plants and plant communities, and b) soils and surficial materials are reconstructed. Some quantitative approaches to estimation of properties that influence soil moisture regime (e.g., available water storage capacity) have been developed, but these are generally based on broad textural categories, and ignore the effects on water retention of all particles >2 mm. These approaches have limited utility to many post-mining materials, where the majority of particles may be >2 mm.

Methods for estimating soil moisture regime on reconstructed post-mining landscapes were developed using concepts from land-capability and biogeoclimatic-ecosystem classification systems combined with new analyses of effects of particle-size distribution on soil water retention. These methods are both quantified and objective, in that they provide consistent results when applied by different users. This paper discusses methods development, application, and testing of this estimation approach.

INTRODUCTION

One of the principal knowledge gaps in mine reclamation is that of the retention of water by and movement of water through surficial materials, and how this mediation of surface water balances influences both ecosystem development and watershed performance. This understanding is critical for better reclamation planning and projection of long-term characteristics of reclaimed ecosystems. It is also critical for improving understanding of hydrologic behaviour of reconstructed mine-affected watersheds and fate and transport of constituents of interest (CIs) in these watersheds. To date, most approaches to addressing this gap, where it has been addressed at all, have been borrowed from ecosystem classification systems, and are limited by some or all of the following factors:

1. they are qualitative or semi-quantitative, and there is limited attempt to evaluate and demonstrate their hydrologic validity;
2. they rely substantially on the presence of vegetation communities to provide information on edaphic conditions – in mine-reclamation settings where these communities are absent or introduced, there is insufficient information for application of these approaches; and/or
3. they have a narrow focus on a single aspect of the surface water balance, e.g., estimating water retention for revegetation planning. In these approaches, there is no attempt to provide a more comprehensive understanding of surface water balances, and how water retention and use by vegetation may influence deeper percolation and the water balance of the underlying mine-waste landform.

This paper presents some advances in this field that propose a quantitative basis for understanding reclamation surface water balances and their effects on both reclamation ecosystem development and the hydrologic behaviour of mine-affected watersheds.

BACKGROUND

Biogeoclimatic ecosystem classification

The foundation of ecological classification in British Columbia is "biogeoclimatic" ecosystem classification (BEC), in which biogeoclimatic units ("zones") represent broad geographic areas of similar macroclimate, and are recognized as influencing biological characteristics of resulting ecosystems (Meidinger and Pojar, 1991). Development of biogeoclimatic classification of the province began in 1949 with the immigration of Vladimir Krajina. From 1950-1975, ecosystem studies undertaken by Dr. Krajina and his students at the University of British Columbia resulted in the development of the BEC system for B.C., based primarily on climate, soils, and vegetation data. This work still forms the basis of the current BEC system and mapping of the province.

In the BEC system, biogeoclimatic zones are subdivided into subzones, which are in turn subdivided into variants, with each subdivision representing a reduction in climatic variability and geographic area (Lloyd et al., 1990). Within each subzone or variant, there are sequences of distinct ecosystems or "site series", with associated vegetation communities reflecting differences in topography and soil depth, texture, drainage, moisture regime, and nutrient regime. In this system, soil water availability is believed to have the greatest influence on ecosystem development. This availability is influenced by climate, but since climate is relatively uniform within a biogeoclimatic subzone or variant, variation in soil water availability at this level of classification results from influences of soil and topography on surface water balances (Lloyd et al., 1990). These influences are manifested in resulting plant associations, i.e., each site series has an assemblage of plants that are adapted to its edaphic conditions – a fundamental principle of the BEC system is that sites with similar physical properties have similar vegetation potential (Meidinger and Pojar, 1990). A subset of plants on a site – "indicator plants" – are diagnostic of edaphic conditions due to their adaptation to narrow ranges of conditions, e.g., soil water availability.

Biogeoclimatic classification as developed in B.C. is a unique system substantially informed by the Russian concept of phytoassociations across ecological gradients (e.g., Sukachev, 1928), and by the classification work of the phytosociologist Braun-Blanquet (e.g., 1932) in southern Europe. Its concepts have been adopted elsewhere in Canada, particularly in western and northern Canada (e.g., Alberta, Saskatchewan, Yukon), and in the western U.S. and Japan (e.g., Kojima, 1991). In North America, the BEC system shares similarities with the site-classification systems of Ontario and with Daubenmire's habitat-type system in the western United States (Pojar et al., 1987).

Soil water availability

In B.C., soil water availability is estimated using a concept called "soil moisture regime" (SMR), which conceptually reflects "the average amount of soil water annually available for evapotranspiration by vascular plants over an extended period of time (several years)" (Pojar et al., 1985). Krajina's classification work incorporated nine SMR classes ranging from driest (Class 0, or very xeric) to wettest (Class 8, or hydric) – this spectrum is sometimes referred to as a "hygrotope" (Pojar et al., 1985; Meidinger and Pojar, 1990; Klinka et al., 1984). The most common classifications of hygrotope – and those that are used in the BEC system – are classifications of *potential* hygrotope based on subjective inferences from site and/or vegetation features, and represent relative ranking of sites or factors in terms of potential soil water availability. A common example of this approach, although more complex and semi-quantified examples exist in the BEC system (e.g., Lloyd et al., 1990), is shown in Table 1.

Quantified and objective approaches to estimation of both potential and actual hygrotope or soil moisture regime are uncommon, and those that exist are limited in their application to specific places and/or ecosystems (e.g., Waring and Major, 1964). In B.C., Green et al. (1984, and summarized in Pojar et al., 1985) used a water-balance approach to develop an actual hygrotope, but it is based on intra-annual duration of water deficits and on presence of water tables, and although the authors provide defining features for their classes, methods for classifying sites according to this system are not provided.

Various land capability classification systems in Canada – beginning with agricultural land capability systems – have used available water storage capacity (AWSC, sometimes called available water holding capacity, or AWHC) as an index of soil water availability. Available water storage capacity is defined as the difference between the volumetric water content at field capacity (FC) and permanent wilting point (PWP), where:

- field capacity is the state at which rapid downward movement of water due to gravitational drainage has become negligible relative to removal through evaporation or evapotranspiration – typically occurring at tensions of 10-33 kPa, depending on soil texture; and
- permanent wilting point is the water content at which soil water is no longer available for plant uptake – although this content varies by plant species (and even within species), by convention PWP is defined as occurring at a tension of 1500 kPa.

AWSC can be expressed as a volumetric water content (%), as a depth of water (mm), or as a depth of water per unit depth of soil/surficial material (mm water/cm soil). A common practice has been to assign AWSC values based on soil texture: for example, B.C.'s *Land capability classification for agriculture in British Columbia* (B.C. Environment, 1983) provides AWSC values in mm water/cm of soil depth for soils of different textural classes. However, these systems, being initially focussed on agriculture, do not link AWSC to soil moisture regime, and to occurrence of typical natural ecosystems and/or larger hydrologic performance.

In northeast Alberta, the *Land capability classification system for forest ecosystems in the oil sands* (LCCS – Cumulative Environmental Management Association, 2006; first published in 1996) attempted to use earlier concepts (and values) of assigning AWSC to textural classes for application to mine-reclamation and forest-ecosystem settings. The LCCS equates a potential hygrotope very similar to that shown in Table 1 to numeric values calculated from texture-class-based AWSC, and some topography and surficial-material-depth modifiers. This approach represents an advancement in producing an objective and quantified potential hygrotope, but still has a number of limitations for broader application:

1. Consistent with conventional soil-science principles, calculation of AWSC in the LCCS is based solely on <2-mm particle-size fraction, with particles greater than 2 mm discounted on a volume basis (e.g., a material with 50% coarse fragments [>2 mm] and a fine-fraction [<2 mm] AWSC value of 1.0 mm/cm would have an aggregate AWSC of 50 mm). This has not been a substantial limitation in oil sands applications, because coarse-fragment contents are relatively insignificant, but it limits application of the LCCS approach to high-coarse-fragment-content settings like hard- and soft-rock mine wastes. Waring and Major (1964) state that evaluation of soil water in soils with coarse-fragment contents >10% must include some consideration of water stored in the coarse fraction, and report that in their study, disregarding water storage of the coarse fraction would have led to errors of up to 300% in determination of water storage.
2. Texture-based AWSC values in the LCCS apply uniformly across texture or material classes, and do not recognize or account for variation in particle-size distributions within these classes. For instance, the LCCS applies an AWSC value of 1.0 mm/cm to oil sands tailings, regardless of whether these tailings are complete, or are cyclone overflow or underflow products.
3. Although there has been substantial investigation and validation of the LCCS AWSC values (e.g., Barbour et al., 2010), and thus of their use as a potential hygrotope, there has been limited evaluation of the relationship between these values and actual soil water contents (the actual hygrotope), and of the relationship between these values and ecosystem development and landscape/watershed hydrologic performance.

The concept of soil moisture regime has been applied globally, based on duration or magnitude of growing-season water deficits, but typically involves relatively broad classes that can be mapped at a continental scale (e.g., Soil Survey Staff, 1999), versus local application to differentiate between ecosystems and hydrologic behaviours.

Table 1. Relative soil moisture regime classes and characteristics (after Meidinger and Pojar, 1991)

MOISTURE REGIME	DEFINING CHARACTERISTICS		FIELD RECOGNITION CHARACTERISTICS						SLOPE GRADIENT
				SOIL PROPERTIES					
	DESCRIPTION	PRIMARY WATER SOURCE	SLOPE POSITION	TEXTURE	DRAINAGE	DEPTH TO IMPERMEABLE LAYER	HUMUS FORM DEPTH	AVAILABLE WATER STOR. CAP.	
VERY XERIC 0	Water removed extremely rapidly in relation to supply; soil is moist for a negligible time after ppt	precipitation	ridge crests, shedding	very coarse (gravelly-S), abundant coarse fragments	very rapid	very shallow (<0.5m)	very shallow	extremely low	very steep
XERIC 1	Water removed very rapidly in relation to supply; soil is moist for brief periods following ppt	precipitation			rapid				
SUBXERIC 2	Water removed rapidly in relation to supply; soil is moist for short periods following ppt	precipitation						very low	steep
SUBMESIC 3	Water removed rapidly in relation to supply; water available for moderately short periods following ppt	precipitation	upper slopes, shedding	coarse to mod. coarse (LS-SL), mod. coarse frag.	rapid to well	shallow (<1m)	shallow	low	moderate
MESIC 4	Water removed somewhat slowly in relation to supply; soil may remain moist for a significant, but sometime short, period of the year. Available soil moisture reflects climatic inputs	precipitation in moderately to fine-textured soils & limited seepage in coarse -textured soils	mid-slope, normal, rolling to level	moderate to fine (L-SiL), few coarse fragments	well to moderately well	moderately deep (1-2m)	moderately deep	moderate	
SUBHYGRIC 5	Water removed slowly enough to keep the soil wet for a significant part of the growing season; some temporary seepage and possibly mottling below 20 cm	precipitation and seepage	lower slopes, receiving	variable, depending on seepage	moderately well to imperfect	deep (>2m)	deep	high	slight
HYGRIC 6	Water removed slowly enough to keep the soil wet for most of the growing season; permanent seepage and mottling present; possible weak gleying	seepage			imperfect to poor	variable, depending on seepage		variable, depending on seepage	
SUBHYDRIC 7	Water removed slowly enough to keep the water table at or near the surface for most of the year; gleyed mineral or organic soils; permanent seepage less than 30 cm below the surface	seepage or permanent water table	depressions, receiving	variable, depending on seepage	poor to very poor		very deep		
HYDRIC 8	Water removed so slowly that the water table is at or above the soil surface all year; gleyed mineral or organic soils	permanent water table			very poor	variable, depending on seepage		variable, depending on seepage	flat

METHODS DEVELOPMENT

Objectives of the proposed classification system

All of the classification systems described above – biogeoclimatic, hygrotope/SMR, and land-capability classifications – have substantially informed the classification system proposed here. However, the goals of this proposed system differ from these predecessors, in that it is intended to:

1. be broadly applicable to a range of climatic, physiographic, and surficial-material conditions (e.g., globally), yet have sufficient resolution to differentiate ecosystem characteristics and hydrologic performance at a local scale;
2. be capable of derivation solely from information on material particle-size distributions and topography, and not rely on observations of intact above-ground ecosystems for diagnosis;
3. be objective, quantitative, repeatable, and easily applied;
4. be capable of evaluation and validation or adjustment through analysis of related empirical observations, including relationships with non-mine ecosystems classified through standard BEC methods; and
5. provide useful interpretations for a range of mine-planning and reclamation-management considerations, including both cover placement/revegetation and understanding hydrologic behaviour at the mine landform-landscape-watershed scale.

Determination of AWSC

A standard particle-size distribution (PSD) ternary diagram for engineering interpretations was used as a framework for generating PSD-based AWSC values. This framework (based on the Unified System of Soil Classification) was used both to allow evaluation of the contribution of particles >2 mm (as opposed to conventional soil-science approaches), and to facilitate communication between mine planners/engineers and reclamation specialists. AWSC values were estimated from two databases of material characteristics[1] for all materials with measured particle-size distributions and water-retention curves. The materials were separated into 100 textural groups corresponding to subdivisions of the PSD ternary diagram, based on gravimetric proportions of coarse (>4.75-mm), sand (0.075-4.75 mm), and fine (<0.075-mm) particles. The average AWSC for each group was used to populate the ternary subdivision position. If a textural group had minimal or no AWSC data then an estimate was made from interpolation and/or extrapolation from surrounding positions. The resulting AWSC-populated ternary diagram is presented in Figure 1, where AWSC values are in mm water/cm material depth, and represent the centre point of each subdivision. Values in this table are preliminary, in that they provide a framework and enable testing of the proposed system, but it is recognized that they require further refinement prior to broad application. In order to allow consistent and repeatable use of this tool, software has been developed that will take input PSD information and consistently interpolate an AWSC value from the centre-point values, based on standard GIS interpolation algorithms. Input PSD information is based on all particles <100 mm. To facilitate more cost effective and reliable classification, low-technology field equipment has also been developed to allow rapid determination of the cobble-and-gravel separate (>4.75

[1] Databases included an internal database from O'Kane Consultants Inc. (OKC), based on properties of mine-waste and cover materials observed by OKC at different client mining sites around the world, and the other internal to SoilVision Systems Ltd.'s numerical modelling software (www.soilvision.com).

mm) based on a large volume of material, with subsequent determination of the sand and clay-and-silt separates (0.05-4.75 mm and <0.05 mm, respectively) based on laboratory analyses of smaller collected samples.

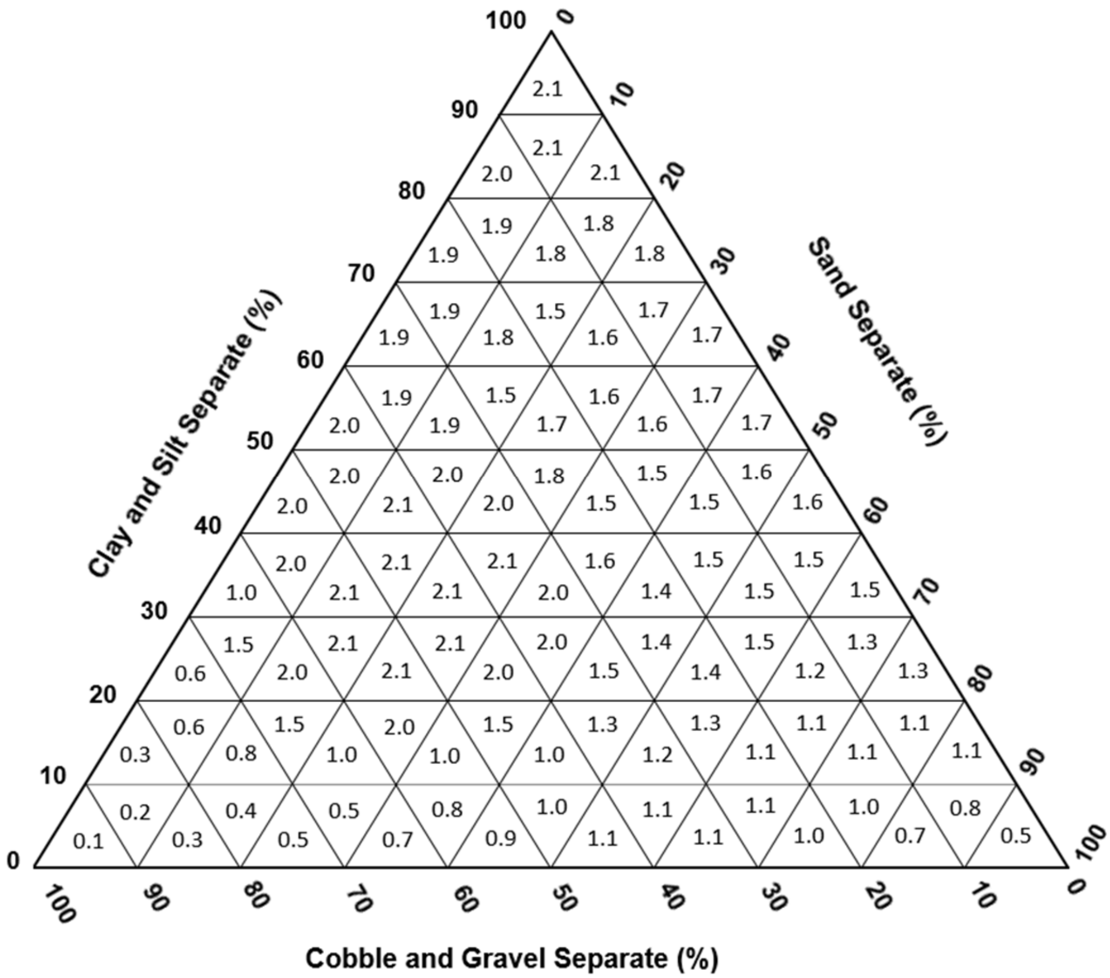


Figure 1. AWSC ternary diagram. AWSC values are in mm available water storage per cm of material depth, and represent the centre point of each subdivision of the diagram.

Values derived from Figure 1 are intended to represent a single material, and to be aggregated across a standard material profile or control section (typically 100 cm, but lesser sections could be used if stipulated). For instance, if a 50-cm soil cover were placed on mining waste rock or tailings, then one AWSC value would be calculated for the cover material, another would be calculated for the mine-waste material, and an aggregate AWSC would be generated by summing the values. If multiple layers were present within the soil cover (or mine waste), then an AWSC value would be calculated for each layer corresponding to depth and PSD data. For natural soils, calculation is based on horizon depths and characteristics. In the case of shallow soils over non-rooting-zone materials, the AWSC for the control section would be based only on the depth of the soil material, and thus would be reduced compared to a 1-m rooting zone.

Modification of AWSC values for energy regime

In B.C.'s BEC-based Terrestrial Ecosystem Mapping (TEM) system, the topographic effect on energy is recognized through "warm" and "cool" site modifiers. These modifiers are applied to slope angles >25% (14°; or >35% in coastal forest regions), with warm aspects being southerly or westerly (135°-285°), and cool aspects being northerly to easterly (285°-135°; Resources Inventory Committee, 1998). This TEM approach was modified for the current classification system as presented below in Table 2 to include a neutral energy regime on southeast and southwest slopes. This modification is based on two considerations:

1. the shift in energy regime on slopes as aspects change is more accurately a gradient of change rather than a categorical shift – the modified classification still uses categories, but incorporates a more gradual shift than an immediate shift from cool to warm as implied by the TEM system; and
2. evaluation of data on field-measured soil water-content profiles versus in comparison to AWSC indicates a better fit when southeast and southwest aspects are categorized as neutral than when they are classified as cool (southeast) or warm (southwest).

Table 2. AWSC energy modifiers

Energy class	Class definition	AWSC modifier
Neutral	Slopes <25% (<14°)	none
	Slope >25% (>14°); aspects 085-135° and 235-285°	
Warm	Slope >25% (>14°); aspect 135-235°	Calculated AWSC – 30 mm
Cool	Slope >25% (>14°); aspect 285°-085°	Calculated AWSC + 30 mm

Equation of modified AWSC values to soil moisture regime[2]

Adjusted AWSC values (PSD-based AWSC from Figure 1 plus any applicable energy modifiers from Table 2) were used to determine soil moisture regime, as outlined in Table 3. This table uses the classes of Krajina's potential hygrotope outlined in Table 1, but replaces the relative ranking of various criteria with ranges of adjusted AWSC. AWSC ranges for each SMR class are modified from the oil sands LCCS. The AWSC method for SMR determination applies only to upland (very xeric – mesic) SMRs, as wetter SMRs require input of seepage water or the presence of a water table within 100 cm of the soil surface, and are not dependent on soil storage. Thus determination of SMRs wetter than mesic in this system is based on observations of shallow groundwater seepage and/or the presence of a water table within the top 1 m of surficial materials.

[2] The term "soil moisture regime" is applied in this paper both to soils and to surficial materials in reclamation landscapes due to its history of use and understood meaning. However, in mine reclamation, many of the materials for which SMR can be estimated are not soils, but are mine wastes and/or salvaged parent materials. Thus SMR should more properly be understood as a soil or surficial-material moisture regime.

Table 3. Determination of SMR from adjusted AWSC.

SMR	Primary water source	Water-table depth (cm below ground surface)	Available water storage, surface 1 m (mm)
Very Xeric (0)	Precipitation and soil storage	>100	<60
Xeric (1)	Precipitation and soil storage	>100	60-89
Subxeric (2)	Precipitation and soil storage	>100	90-119
Submesic (3)	Precipitation and soil storage	>100	120-149
Mesic (4)	Precipitation and soil storage	>100	>150
Subhygric (5)	Precipitation and seepage	>100	>150, seepage contributes to supply
Hygric (6)	Seepage	30-100	n/a
Subhydric (7)	Seepage or permanent water table	0-30	n/a
Hydric (8)	Permanent water table	Water table permanently at or above soil surface	n/a

METHODS TESTING

The methods discussed above were developed and tested at reclamation-monitoring sites at seven mining operations in 2012-2014: Teck Resources Limited's (Teck) Elkview (EVO), Fording River (FRO), Greenhills (GHO), and Line Creek (LCO) metallurgical coal operations in southeastern B.C. and the Cardinal River (CRO) operations in west-central Alberta; at the Teck Highland Valley Copper Partnership's Highland Valley Copper mine in south-central B.C., and at Thompson Creek Metals' Endako mine in central B.C. Of particular relevance to testing are the five Teck coal mines, as in 2011, Teck commenced development of an integrated, multi-year and multi-disciplinary applied research & development program focused on managing water quality in mining-affected watersheds. In 2012-13, this program included installation of soil and meteorological instrumentation and soils and vegetation assessments at 12 reclamation sites at the above coal mines, to meet various research objectives including provision of updated data on reclamation conditions co-located and concurrent with information on meteorological and soil-moisture variables at each study site. This instrumented-site network and the data it provides supports increased understanding of how surface water balances and soil moisture regimes are affecting reclamation responses over time, and *vice versa*, as well as how reclamation approaches affect reconstructed landform water balances, and hence watershed hydrology.

The approach to estimation of soil moisture regimes discussed above was applied to these testing sites, and included adjustment of methods in response to interpretation of collected data.

AWSC estimates for different mine waste materials and reference sites

Figure 2 presents plotted PSD data from 65 mine-reclamation and non-mine reference sites on the AWSC ternary diagram. These data show both separation and similarities between different material types, as follows:

- Coal waste rock – sampled coal waste rocks generally have a cobble and gravel content from 50-80% of the entire sample, and a clay and silt content of ~5-20%. Their AWSC values range from approximately 0.4-1.5 mm/cm.
- Metal-mine waste rock – metal-mine waste rock in this study had a slightly lower cobble and gravel content than the coal waste-rock samples. These materials have a similar overall range of AWSC values, but slightly higher average AWSC. These differences between metal and coal waste-rock may be an artefact of differing physical-property sampling methods. Revised field sampling methods are under development to ensure consistent data collection across all study sites in future.
- Metals tailing – are composed entirely of fines, with approximately 5-20% clay and silt content.
- Cover materials – there is substantial variation in these materials, due to a range of source locations and material types, but they are generally finer than both metal and coal waste rocks. Their AWSC values range from 0.7-2.1 mm/cm, with a higher average AWSC than both waste rocks and tailings.
- Reference sites – these sites are similar to the mining cover materials (as most cover materials are sourced from sites like these), and generally have higher AWSC values than the mine wastes, for both ranges and averages.

Actual versus potential hygrotope

AWSC values calculated from the PSD ternary diagram provide quantification of the potential hygrotope, as they indicate the *capacity* for soil water storage (and eventual release as evapotranspiration, interflow, and/or net percolation), not actual storage. Actual storage is a product of the interaction between the potential hygrotope and local climate, which delivers precipitation for storage and energy for evaporation and transpiration. To evaluate the relationship between potential (calculated) and actual hygrotope, analyzed volumetric-water-content (VWC) and matric-potential (φ_m) data collected by O'Kane Consultants from the Teck instrumented study sites (metallurgical coal operations) were analyzed to derive mean growing-season *available* volumetric water contents (AWC) for each site. To do so, the VWC at permanent wilting point (PWP) was calculated for each material type (cover material, waste rock) from interpolated plots of VWC against φ_m for each sensor pairing. This gave each VWC sensor a VWC-at-PWP value, which was then subtracted from each of its VWC measurements to calculate AWC (water content above PWP) for all sensors. To calculate mean AWC from all sensors, each sensor's AWC was mathematically weighted according to rooting patterns observed at vegetated sites, with weight assigned for both root abundance and root size. Where rooting data did not exist, mean root patterns from all other sites have been applied. Reported AWC values are means of all daily measurements made during the 2013 growing season, which was defined by site-specific meteorological data using the criteria of five consecutive days of average daily temperatures over and under 5°C as the beginning and end[3] of the growing season (Alberta Agriculture and Rural Development, 2009).

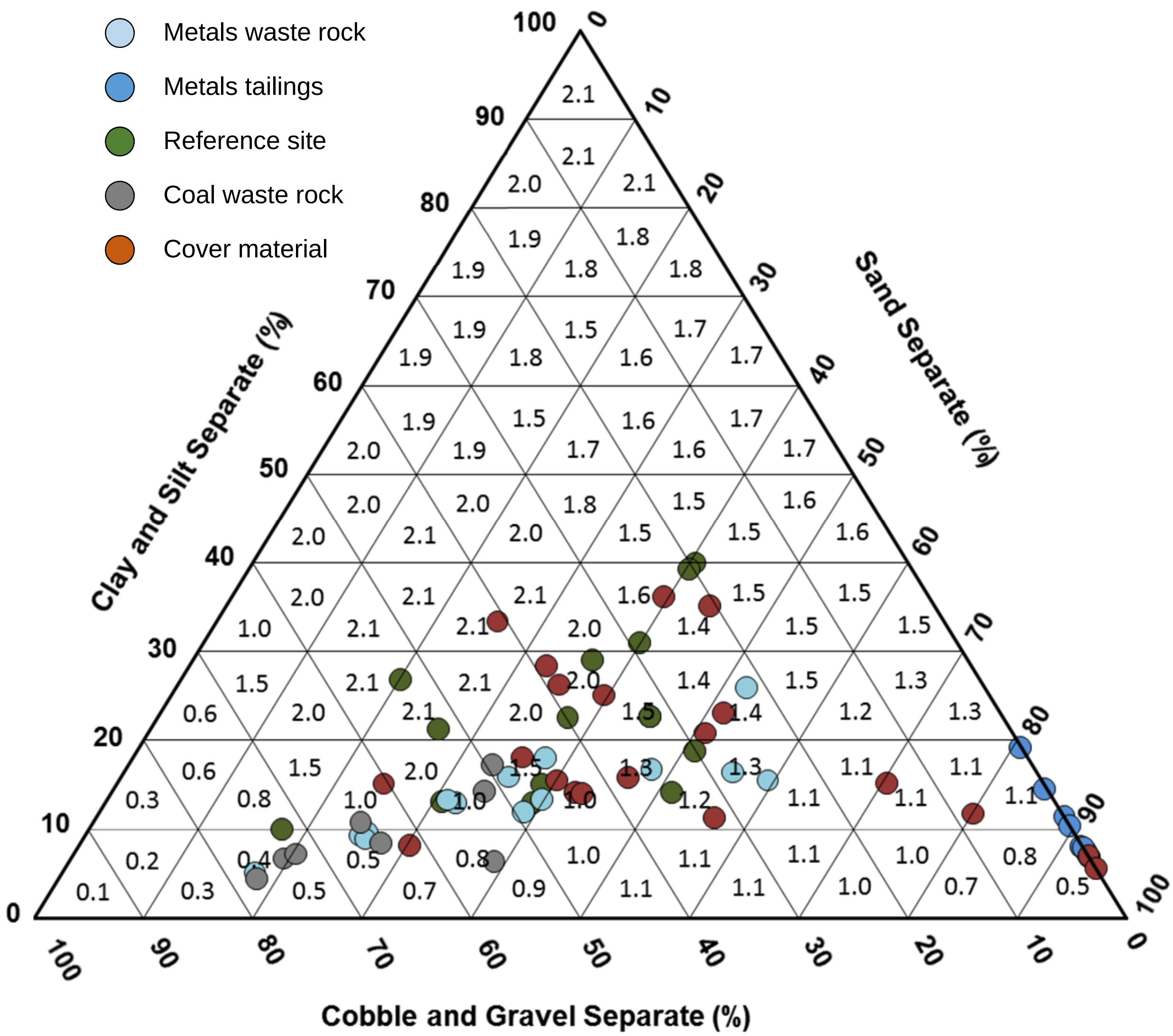


Figure 2. Plot of PSD-based AWSC values for mine-waste and cover-system materials, and non-mine reference sites.

<u>Data-based adjustment of AWSC modifiers</u>

The relationship between modified AWSC and mean AWC for the instrumented Teck coal study sites was usedto adjust energy modifiers (adjusted modifiers are presented in Table 2). Initial graphing of these data with the original (unadjusted TEM-based) energy modifiers yielded an R^2 value of 0.39, and showed one site in particular (the upper "x" in Figure 3) with a large fitting error. It was reasoned that the categorical nature of the original TEM-based energy correction – where sites transition from a 30-mm AWSC deduction on western aspects to a 30-mm AWSC addition on northwestern aspects – was too

[3] The end of the growing season cannot occur before August 1 regardless of temperature.

abrupt, and didn't recognize that, in fact, radiation is a continuous variable that is gradually declining over this range of aspects. Therefore, energy modifiers were adjusted to restrict the warm-aspect deduction to southeast to southwest aspects only, and to create a steep, southwest to west-northwest category with no modifier (neutral energy regime - Table 2). This adjustment produced the improved fit shown in Figure 3 (R^2 = 0.45).

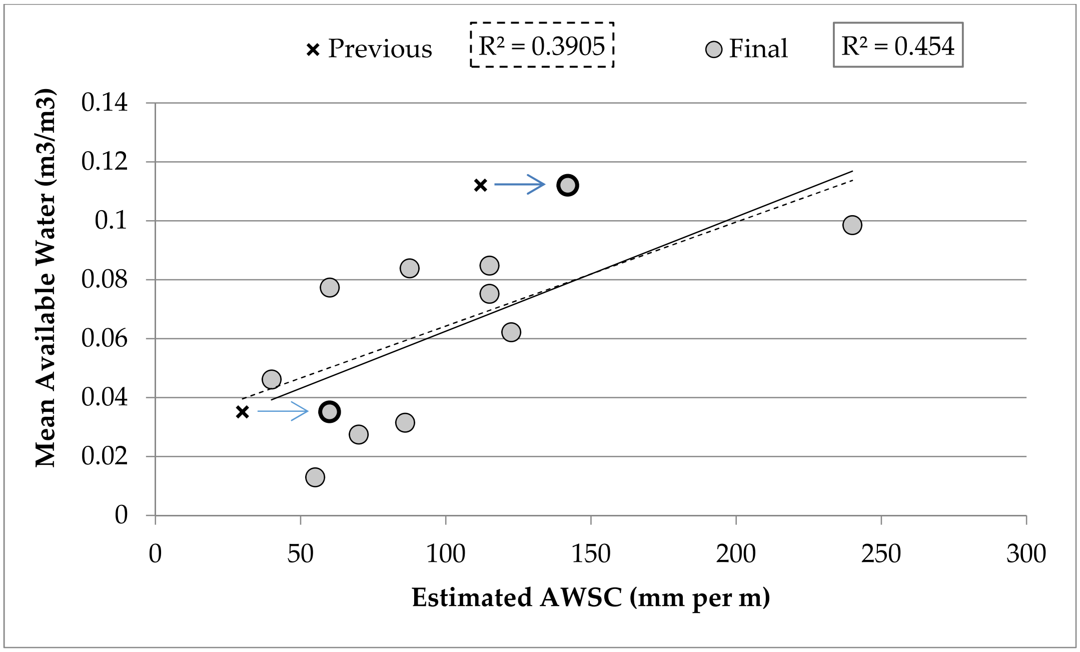


Figure 3. Mean available volumetric water content (AWC) by AWSC. Development of energy corrections was an iterative process based partially on agreement with soil moisture patterns. This figure depicts the alteration of energy corrections for two sites, where original AWSC values are plotted with crosses and have been shifted as indicated by blue arrows, which yields an improved correlation between modelled AWSC and actual AWC.

Analysis of fitting errors from the regression line in Figure 3 indicate that much of the observed deviation from the linear fit is attributable to differences in site vegetation: sites below the regression line show lower AWC than expected, primarily due to the presence of robust vegetation covers that are removing water through transpiration; whereas sites above the line are generally non-vegetated or sparsely vegetated, and thus have higher-than-expected AWC values, as water removal through evaporation alone cannot reduce VWC below field capacity through most of the profile.

<u>Predicted AWSC and SMR</u>

SMR was assigned for each of the 65 study sites using the PSD-based AWSC estimates with energy modifiers as described above. For the Teck coal research sites where AWC data is available, the AWSC-based SMR classification was evaluated using mean growing-season AWC (Figure 4). These data show general support for the current system, with mean growing-season AWC increasing for every SMR class, despite differences in vegetation development across these sites. On average, very xeric sites have less than 30% of the plant-available water that mesic sites have during the growing season, while xeric sites have approximately 50% of the plant-available water of mesic sites. Research sites at Endako and Teck

Highland Valley Copper lack continuous measurement of soil water contents, and so cannot be added to this database, but reference sites in these studies provide some ability to evaluate system fit, as predicted SMR using methods proposed in this paper can be related to potential hygrotope classification using standard subjective keys and the presence of indicator plants. All reference sites studied to date are zonal site series (in the SBSdw3 [Endako] and MSxk2 [Highland Valley Copper] biogeoclimatic variants) with mesic SMR – mean AWSC for these sites estimated with the proposed methods is 159 mm, which places them in the mesic SMR category according to the criteria presented in Table 3.

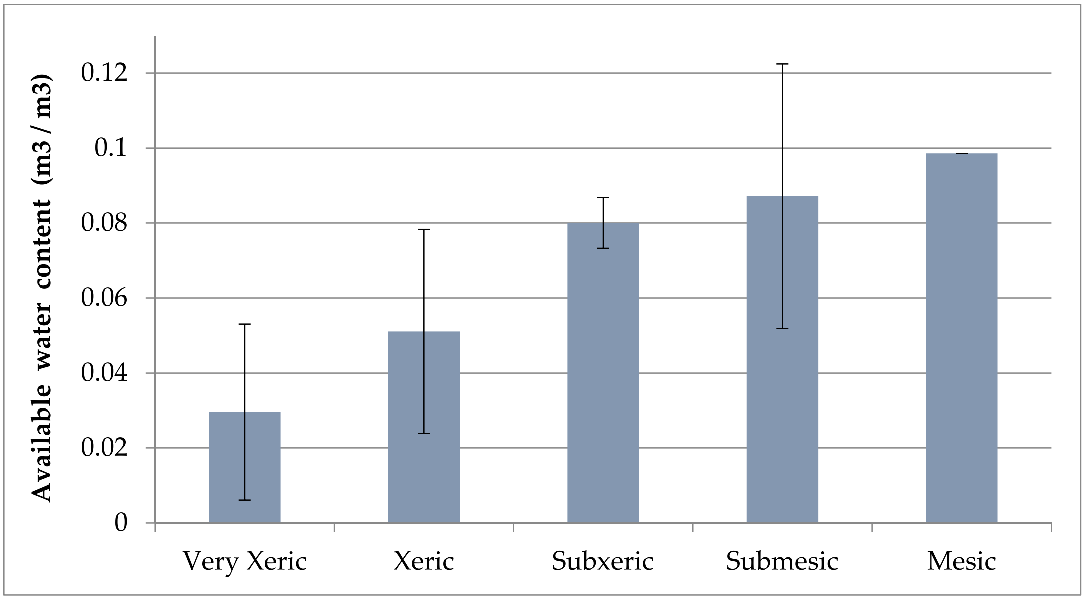


Figure 4. Mean AWC during the 2013 growing season at all sites classified by soil moisture regime. Error bars show one standard deviation of the mean.

APPLICATIONS

<u>Reclamation and revegetation planning</u>

The methods described above can be applied to both existing reclamation sites or planned future landscapes, using either measured or assumed PSD and topographic data. For instance, for an existing reclamation site, slope and aspect can be measured, and PSD data can be derived through field and lab measurements; for reclamation planning, projected post-mine topographies and assumed PSD characteristics can be used to estimate PSD-based AWSC, energy modifiers, and resulting SMR. BEC classification to the subzone-variant level using available mapping and elevation will then allow equation of calculated SMR to non-mine site series[4] – e.g., a site with a calculated AWSC value of 165 mm has a mesic SMR, which is hygrotopically equivalent to a zonal (frequently 01) non-mine site series. It is necessary to emphasize that numerical equivalency with respect to AWSC or surface-profile water supply does not equate to equivalency between reclaimed-mine site series and non-mine site series: although

[4] Equation of an estimated SMR using proposed methods to a site series also requires estimation of tropotophic position, or soil nutrient regime. Current data is insufficient to support determination of SNR through quantitative methods. Thus, the current proposed approach is to assume a very poor-poor nutrient regime (A-B) for reclaimed mine waste with no soil cover, and a medium nutrient regime (C) for sites with a soil cover.

sites with similar hygrotopic position have similar *capability* with respect to soil water supply, these sites have many other differences (e.g., soil biota, vegetation propagules), and their degree of actual similarity will depend on reclamation actions and stochastic events (e.g., climatic patterns during early reclamation establishment). To emphasize this distinction, the following nomenclature is proposed: non-mine sites are designated as per convention, e.g., "MSxk2 01" would indicate a zonal site series in the South Thompson Uplands Very Dry Cool Montane Spruce Variant, while a similar reclaimed site would be designated "MSxk2 RY_{01}", where "RY" is the TEM code for "reclaimed mine", and the RY subscript designates the reclamation site series[5].

Following designation of BEC subzone-variant and site series, knowledge of vegetation ecology can then be used for revegetation planning, with candidate revegetation species drawn from lists of species adapted to growth in the edaphic conditions of the site series, and tolerant of early successional conditions such as exposed mineral soil and full light. An example of this approach is shown in

Table *4*.

Assessment of equivalent capability

The approach discussed above can also be used as the basis of assessment of equivalent land capability, where land capability is defined as the potential for a site to achieve a specified land use(s) as a result of climatic, topographic, and soils/surficial-material-based limitations (B.C. Ministry of Energy, Mines and Petroleum Resources, 2008). Capability should not be confused with realized vegetation or ecosystem conditions, which, as discussed above, are based on capability but influenced by reclamation treatments and stochastic events. Given pre-development BEC mapping and the approach outlined in this paper, mapping and tabular data on relative areas can be produced, allowing comparison of pre-development and post-closure site series, which provides the basis for a comparison of pre- and post-mining capability with respect to hectares of target vegetation communities or habitat potential. An example of post-closure mapping allowing this capability-comparison approach is shown in Figure 5.

[5] In some cases it may be useful to group non-mine site series for creation of reclamation site series, where there is insufficient resolution to distinguish between two site series based solely on hygrotope. For instance, $RY_{02/03}$ could be used to designate a reclamation site series that spans the hygrotopic range of the 02 and 03 non-mine site series.

Table 4. Candidate revegetation species for the MSdk1 $RY_{102/103}$ reclamation site series. Species in bold font are those that are early-seral species characteristic of the analogous non-mine site series and commonly found in plots in these ecosystems.

Trees	**Douglas-fir**	Herbs	**Bluebunch wheatgrass**
	Lodgepole pine		**Fireweed**
	Hybrid white spruce		**Pearly everlasting**
	Trembling aspen		**Pinegrass**
	Western larch		**Ross's sedge**
Shrubs	**Birch-leaved spirea**		**Round-leaved alumroot**
	Common juniper		**Showy aster**
	Common snowberry		**Wild strawberry**
	Kinnikinnick		**Yarrow**
	Prickly rose		**Yellow penstemon**
	Saskatoon		Common horsetail
	Soopolallie		Fern-leaved desert-parsley
	Douglas maple		Junegrass
	Grouseberry		Lance-leaved stonecrop
	Low bilberry		Northern bedstraw
	Oregon grape		Purple-leaved willowherb
	Prairie rose		Sedges (terrestrial)
	Willows (terrestrial)		Sulphur buckwheat
			Yellow hedysarum
			Yellow penstemon

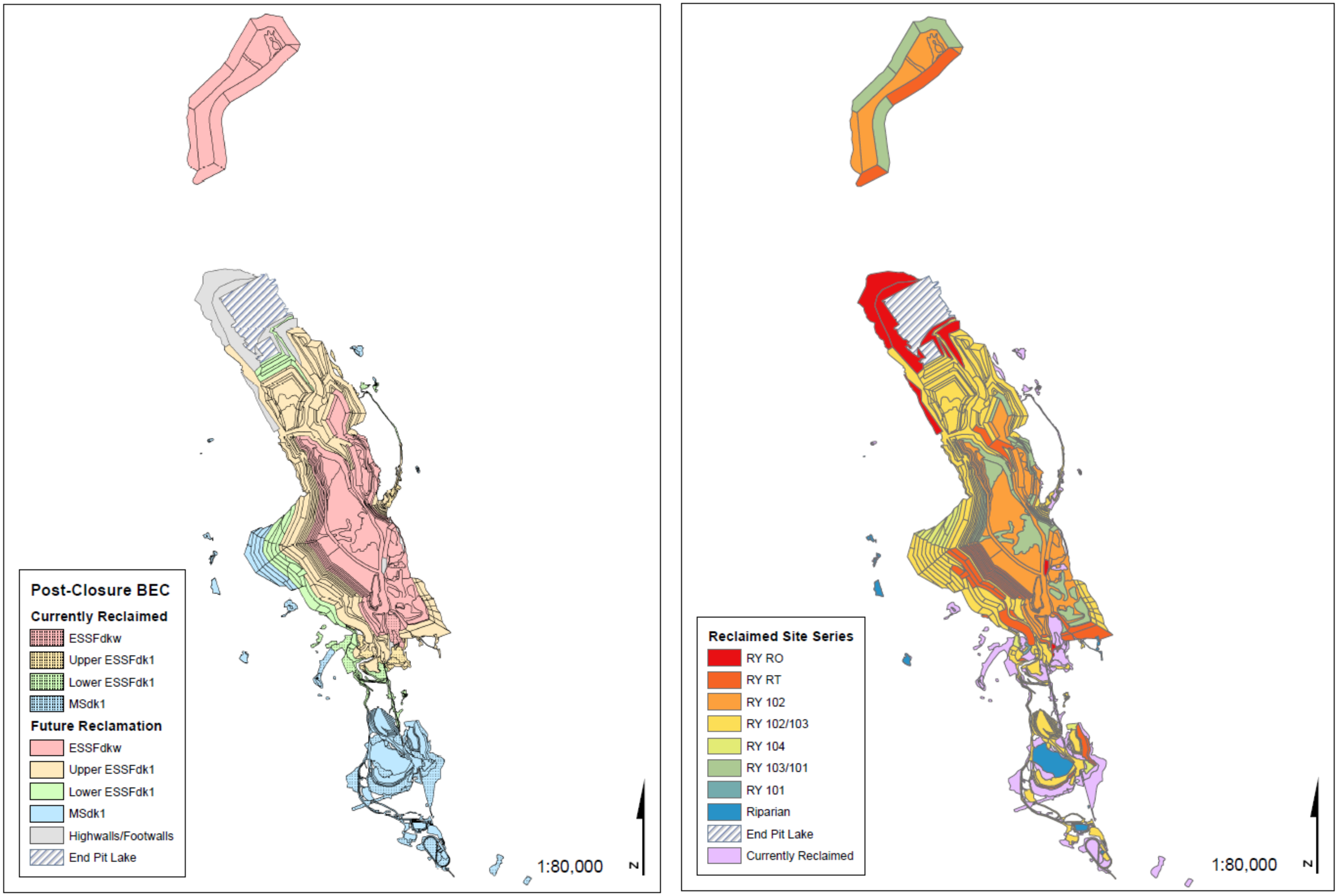


Figure 5. Example of post-closure BEC zone-subzone-variant and reclamation site series classification using methods proposed in this paper applied to a hypothetical post-closure topography.

Quantifying effects of mine-waste covers on surface water balances and ecosystem development

The final energy-corrected AWSC values for the Teck coal instrumented sites are plotted by cover type in Figure 6. This information shows that, while there are ranges in estimated AWSC by cover category, sites with salvaged soil or overburden covers have substantially higher AWSC than sites comprised of revegetated bare waste rock – the median SMR for sites with a salvaged soil or overburden cover is submesic, while the median SMR for vegetated waste rock is xeric. This information indicates that some form of salvaged-material cover is likely a necessary component of reclamation where reclamation goals include re-creation of submesic and mesic upland sites. This finding has implications for both mine planning/materials balances and assessment of equivalent capability, especially where capability and/or biodiversity goals indicate the need for replacement of wetter upland site series.

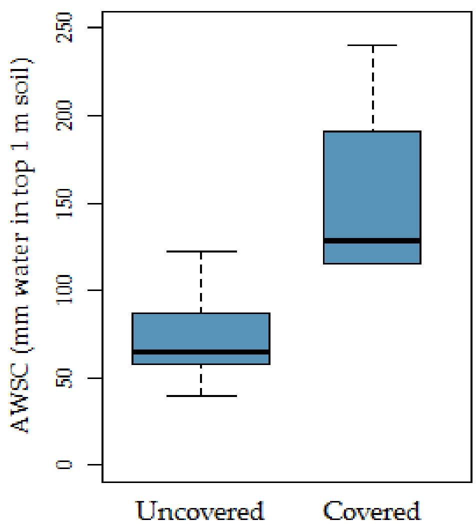


Figure 6. Modified AWSC by presence or absence of soil/overburden cover.

Net percolation and soil moisture regime

As discussed above, a primary goal of the proposed classification system is to not only provide information for reclamation planning, but also to contribute to the hydrologic understanding of how water is mediated by, and moves through, reclaimed landscapes and watersheds. In order to advance this objective, relationships between the proposed classification system and surface water-balance terms were evaluated. Preliminary water-balance estimates for a one-year period beginning October 1, 2012 were generated by O'Kane Consultants for the Teck coal instrumented sites, and are discussed in a companion paper by Birkham et al. (2014). Estimates of net percolation (NP) by estimated SMR class are presented in Figure 7. In general, the conceptual model is that net percolation decreases as SMR increases (i.e., from very xeric to mesic), as it is expected that the greater water storage associated with wetter moisture regimes would attenuate vertical movement of water (net percolation) and allow for more water removal through evapotranspiration. This is partially reflected in the fact that all moisture regimes other than very xeric have NP values approximately 30% lower than the very xeric class, but the conceptual model is not supported by the observed equivalence of NP data in the xeric to mesic classes. This is likely due to confounding influences of vegetation at these sites, as many sites with wetter SMR are also very recent soil-covered reclamation areas, and thus lack vegetation. Therefore, there is no transpiration removal of water from these sites, soil water contents remain at field capacity or higher throughout the majority of the profile, allowing near-continuous drainage, and as a result NP is high. Evidence from reclamation research in other jurisdictions confirms that as vegetation develops, evapotranspiration increases (e.g., Carey, 2011), with a corresponding decrease in net percolation. Continued collection of surface water-balance and vegetation community-evolution data will enable development of improved SMR-NP relationships for each stage of the reclamation trajectory. It is also recognized that the fit of the SMR-NP relationship may be limited by a current potential conflict in the classification system in which two primary factors lead to classification of sites as drier: 1) limited soil storage; and 2) higher insolation. The first of these factors tends to lead to increased NP, as precipitation moves quickly through the system and is removed as NP, while the second tends to lead to decreased NP, as winter precipitation (P_{snow}) is reduced through sublimation and evapotranspiration is increased. Work is underway to further develop the classification system to address this conflict.

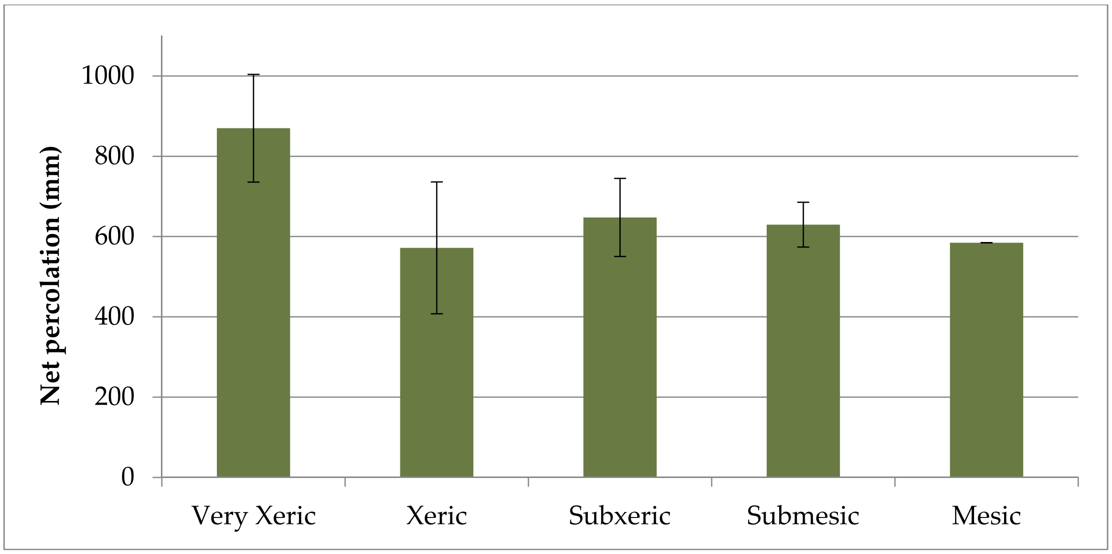


Figure 7. Net percolation values classified by SMR classes.

<u>Relationships between soil and vegetation variables and net percolation</u>

It was hypothesized that a large portion of the variation observed in net percolation can be explained by two key soil and vegetation variables: 1) AWSC (soil); and 2) leaf-area index (LAI – vegetation), with higher AWSC associated with higher LAI and lower NP, due to increasing ability to support vegetation (higher LAI), and increasing removal of water through transpiration (lower NP). Testing of this hypothesis through multiple-regression techniques indicated a significant negative relationship between LAI and NP, but no significant relationship between AWSC and NP. This result can be at least partially attributed to the low number of instrumented study sites, and to the fact that the high-AWSC sites are currently young and sparsely vegetated, and thus generally have high water contents and NP. In contrast, the two lowest-NP sites are low-AWSC mature vegetated coal waste rock with higher LAI values – thus, given current data, vegetation effects overwhelm any effects of AWSC. This effect is illustrated in Figure

8, which presents LAI and NP data, with a curvilinear (polynomial) regression line fit to the data. This relationship is statistically compromised by the fact that in these data, LAI and NP are not independent[6]. Thus R^2 or fitting-error information is not presented for this regression. Nevertheless, it is useful to present these data for conceptual reasons, as they illustrate the following:

1. On unvegetated or sparsely vegetated sites (LAI <1.5), approximately 70-85% of precipitation is transmitted through the upper 1 m of the reclamation cover and enters underlying waste as net percolation. Three unvegetated sites have NP over 80% of precipitation.
2. On moderately vegetated sites (LAI of 1.5-2.5), approximately 60-70% of precipitation is transmitted through the upper 1 m of the reclamation cover as net percolation;
3. On well-vegetated sites (LAI >3.5), approximately 35-55% of precipitation is transmitted through the upper 1 m of the reclamation cover as net percolation, a reduction in net percolation of 15-50% from the poorly or non-vegetated covers. The substantial differences in NP between these two high-LAI sites may reflect the influence of aspect (energy regime) on surface water balances – the lowest-NP value is from a south-facing site whose snowpack depth is approximately 30 cm towards the end of the growing season, and thus whose delivered winter

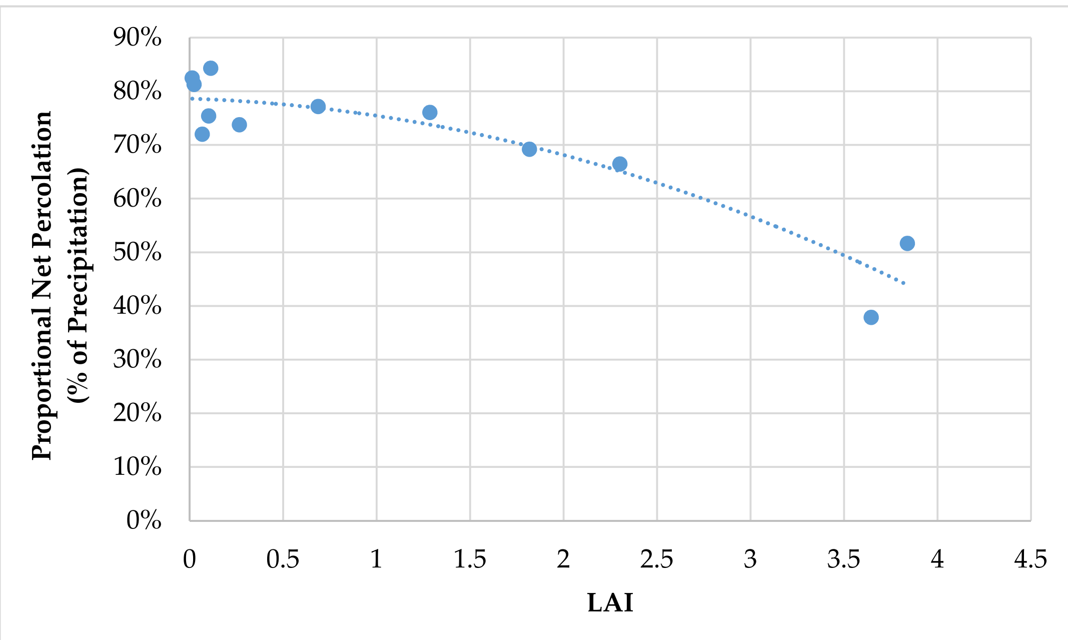


Figure 8. Net percolation as a proportion of precipitation by LAI.

precipitation (P_{snox}) is approximately 30 mm, resulting in reduced NP. In contrast, the higher-NP value of the pair is from an east-facing slope with less radiation and sublimation, giving it a winter/spring snowpack of approximately 60 cm, and thus higher delivered P_{snow} and higher NP, despite substantial removal of water through transpiration.

[6] A relationship between LAI and actual evapotranspiration (AET) was used to adjust NP values from O'Kane Consultants.

The soil moisture and climate data used to develop these relationships was collected in the period of October 2012-September 2013, which was a relatively wet year, and thus the NP values presented above represent the higher end of the expected range for these sites. Additional years of data will provide ranges of NP for sites under varying precipitation levels. These values also primarily reflect NP achieved (in a wet year) with varying degrees of vegetation established directly on waste rock. As the vegetation communities on soil covers continues to establish and mature, it is anticipated that these points will shift right and down (higher LAI and lower NP) on the diagram (Figure 8), and these sites should regularly achieve NP values <40% of precipitation, due to enhanced removal of water through transpiration.

SUMMARY

This paper proposes a quantified and objective hygrotopic classification system that is broadly applicable to a range of ecosystems, including mine-waste-based landforms and mine-affected watersheds. The classification system is informed by the B.C. biogeoclimatic ecosystem classification system, and by various land capability classification systems, including the system used for mine reclamation in Alberta's oil sands. Although the proposed classification system is initially based on potential hygrotope, the use of regional and local biogeoclimatic ecosystem classifications allows its translation into actual hygrotopes, based on regional and local climatic conditions. This translation from potential to actual hygrotope has been tested in two regions of western Canada on instrumented reclamation study sites. Initial results show promising relationships between predicted SMR using the proposed classification system and mean growing-season available water contents calculated from continuous measurement by *in situ* sensors, with increasing observed available water contents as SMRs predicted by the classification model progress from drier to wetter sites. In addition, the proposed classification system shows concordance with traditional ecosystem classification of non-mine reference sites where classification is based on indicator-plant presence and topographic/soil relationships.

The potential management applications of the classification system include reclamation and revegetation planning, assessment of equivalent capability, and quantification of the effects of surficial-materials management on landform surface water balances. A synthesis of data on reclamation variables (topography, surficial-material particle-size distribution, and vegetation cover) indicates strong relationships between these variables and surface water-balance terms (evapotranspiration and NP). These relationships demonstrate the importance of landform, surficial-material, and vegetation management for achieving reclamation objectives and simultaneously contributing to landform and landscape water management. Landform, cover and vegetation treatments designed to increase SMR and vegetation establishment and growth will lead to measurable reductions in net percolation in comparison to sites with less surface water storage and vegetation cover.

REFERENCES

Alberta Agriculture and Rural Development. 2009. *Growing season.* Agricultural Atlas of Alberta: Agricultural Climate Elements. Alberta Ministry of Agriculture and Rural Development: Edmonton, AB.

B.C. Environment. 1983. Land capability classification system for agriculture in BC. Province of BC, Victoria.

B.C. Ministry of Energy, Mines and Petroleum Resources. 2008. Health, safety and reclamation code for mines in BC. Victoria, B.C.

Barbour L., Chanasyk D., Hendry J., Leskiw L., Macyk T., Mendoza C., Naeth A., Nichol C., O'Kane M., Purdy B., Qualizza C., Quideau S., and Welham C. 2010. Soil capping research in the Athabasca Oil Sands Region, volume 1: technology synthesis.

Birkham, T., M. O'Kane, A. Goodbrand, L. Barbour, S. Carey, J. Straker, T. Baker, and R. Klein. 2014. Near-surface water balances of coal waste dumps. In preparation for the 2014 B.C. Mine Reclamation Symposium.

Braun-Blanquet, J. 1932. Plant sociology: the study of plant communities. Translated, revised, and edited by C.D. Fuller and H.S. Conrad. Hafner, London.

Carey, S. 2011. Multi-year assessment of water and energy exchange from an oil sands reclamation cover, Fort McMurray, Canada. Proceedings, Mine Closure 2011. Australian Centre for Geomechanics, Perth. 2:331-340.

Cumulative Environmental Management Association. 2006. Land capability classification system for forest ecosystems in the oil sands, 3rd ed. Alberta Environment, Edmonton.

Green, R., P. Courtin, K. Klinka, R. Slaco, and C. Ray. 1984. Site diagnosis, tree species selection, and slashburning guidelines for the Vancouver Forest Region. Province of British Columbia, Victoria.

Kojima, S. 1991. Classification and ecological characterization of coniferous forest phytogeocoenoses of Hokkaido, Japan. Vegetation, 96: 25-42

Lloyd, D., K. Angove, G. Hope, and C. Thompson. 1990. A guide to site identification and interpretation for the Kamloops Forest Region. Province of BC, Victoria.

Meidinger, D., and J. Pojar. 1991. Ecosystems of BC. Province of BC, Victoria.

Pojar, J., K. Klinka, and D. Meidinger. 1987. Biogeoclimatic ecosystem classification in BC. Forest Ecology and Management, 22: 119-154.

Pojar, J., K. Klinka, and D. Meidinger. 1985. Biogeoclimatic ecosystem classification in BC. Ministry of Forests, Victoria.

Resources Inventory Committee. 1998. Standard for Terrestrial Ecosystem Mapping in BC. Province of BC, Victoria.

Soil Survey Staff. 1999. Soil taxonomy. 2nd ed. Agriculture Handbook Number 436. US Department of Agriculture, Washington, DC.

Sukachev, V.N. 1928. Principles of classification of the spruce communities of European Russia. Journal of Ecology 16: 1-18

Waring, R., and J. Major. 1964. Some vegetation of California coastal redwood regions in relation to gradients of moisture, nutrients, light, and temperature. Ecological Monographs 34: 167-215.

NEAR-SURFACE WATER BALANCES OF WASTE ROCK DUMPS

T. Birkham, P.Eng.[1A], M. O'Kane, P.Eng.[1B], A. Goodbrand1*
S.L. Barbour[2]
S.K. Carey[3]
J. Straker[4], T. Baker[4]
R. Klein[5]

[1]O'Kane Consultants ([A]Cranbrook, BC [B]Calgary AB *previously)
[2]University of Saskatchewan, Saskatoon, SK
[3]McMaster University, Hamilton, ON
[4]Integral Ecology, Duncan, BC
[5]Teck Resources Ltd., Sparwood, BC

ABSTRACT

The near-surface water balance of mine impacted landscapes is a key control on re-vegetation performance, and on the hydrologic and water quality impact at the watershed scale. As part of Teck Resources Limited's applied research and development program focused on managing water quality in mine-affected watersheds, 12 sites representing a range of waste rock dump reclamation surface management options (i.e. soil cover, surficial mounding) were instrumented in 2012 to measure meteorological and soil water response and to quantify the near-surface water balances with a focal objective to improve estimates of ranges of net percolation into waste rock dumps under a range of scenarios.

Subsurface water and meteorological conditions varied substantially, as expected for the range of elevation, slope aspects, vegetation, soil covers, geographic location and surface preparation of the selected sites. Patterns in water balance trends emerged in the first year of analysis with net percolation (NP) into underlying waste rock typically decreasing for increased vegetation and soil cover, as well as for decreases in rainfall or snowmelt. Increased vegetation cover resulted in a greater volume of water removed from near-surface through evapotranspiration. The lowest NP (as % of water input) was estimated for a mature, reclaimed conifer forest site and a dense agronomic grass/alfalfa covered site. Net percolation estimated for a soil covered waste rock slope was approximately 15% (of water inputs) less than an adjacent bare waste rock slope. Decreased NP was partly attributed to greater water storage in the finer-textured soil cover. Net percolation through the soil cover is expected to further decrease with time as vegetation establishes relative to the bare waste rock slope. Net percolation for a mounded, bare waste rock slope was less than estimated for an adjacent smooth slope. Net percolation below a trough was similar to the smooth slope, but decreased at the crest and mounded mid-slope positions due to thinner snowpack (less snowmelt) from wind scouring. Additional monitoring and analysis of site-specific water balances will help define the shift in the relative proportions of water entering the deposits as vegetation matures.

KEY WORDS

Net percolation, soil cover, water balance, evapotranspiration, vegetation

INTRODUCTION

Mining activities undertaken by Teck Resources Limited (Teck) to access the metallurgical coal resource in the Elk Valley of southeastern British Columbia and in northwestern Alberta results in waste rock dumps with material susceptible to long term weathering and erosion processes. The waste rock is known to increase concentrations of some constituents of interest (CIs) in downstream surface water and groundwater systems. Post-depositional weathering of the waste rock is accelerated by oxygen ingress and weathering products can be transported with infiltrating water. These two processes are both controlled by the degree of saturation of the unsaturated waste rock.

The amount of water that infiltrates into waste rock landforms is a function of the near-surface water balance. This surface water balance is a key control on vegetation establishment and its long term performance and will affect the hydrology, hydrogeology and geochemistry of affected watersheds. For example, selenium (Se) concentrations in mine-affected rivers are reported to have increased over the last three decades (Lussier et al., 2003; Chapman et al., 2007) and the understanding of Se release and transport from waste rock dumps is still developing. Research has indicated that Se is more readily transported in the soluble form of selenate suggesting a strong link to water movement (Chapman et al., 2007).

Controls on the movement of water through and out of waste rock dumps include the volume, rate, and flow path of net percolation (NP) waters. The term 'net percolation' is defined as water that migrates into and downward in the material profile to depths not influenced by atmospheric processes (i.e. evapotranspiration). This water may eventually be released from waste rock to adjacent groundwater or surface water. Important aspects of NP include:

- Volume: NP volumes are a primary control on water volumes reporting as basal and toe seepage from waste rock dumps and have important implications for mine-affected watershed water balances and water management strategies. NP volumes may be an important control on solute loadings in basal and toe seepage;

- Rate: The flow velocity of NP through mine wastes, as well as the height of the waste deposit, will control the residence time of pore-water in the waste rock profile. Residence time may be an important consideration on the dissolution of CIs; and

- Flow Path: Water may move via preferential pathways (i.e. macro-pore flow) or via matrix flow, which is more distributed. Preferential flow through waste rock will decrease the surface area contacted by NP and residence time of NP in waste rock dumps, which are opposite to the trends for matrix dominated flow. Preferential flow paths will increase the time required to flush CIs from a waste rock dump compared to matrix dominated flow as NP following preferential flow paths bypasses and does not flush oxidation products from all of the waste rock.

As part of Teck's applied research and development program focused on managing water quality in mine-affected watersheds, 12 sites representing different waste rock dump reclamation surface management options were instrumented in 2012 to measure meteorological and soil water dynamics. Data from these sites was used to quantify the near-surface water balances with a focal objective to improve estimates of ranges of NP into waste rock dumps for a variety of conditions.

METHODS

The study sites are located at four coal mining operations in the Elk Valley, BC and one in northwestern Alberta: Line Creek (LC); Greenhills (GH); Elkview (EV); Fording River (FR); and Cardinal River (CR; Figure 1). The climate at all of the study sites is humid continental. Climate normals data (1981-2010, Environment Canada, Sparwood station at 1140 m asl) indicates the Elk Valley mines are located in a region with mean annual precipitation of 613 mm (411 mm as rainfall, 202 mm as snowmelt). Mean annual precipitation from climate normals data (1981 – 2010, Environment Canada, Jasper East Gate at 1003 m asl) for the Cardinal River area is 599 mm (448 as rainfall, 151 mm as snowmelt). The study sites monitoring instruments were commissioned in the summer and fall of 2012 and have been collecting data since. This paper reports on data for each site over one hydrologic year starting October 1st, 2012 through September 30th, 2013. A large rainfall event in the Elk Valley from June 18^{th} – 21^{st}, 2013 largely influenced the amount of precipitation observed in the first year of monitoring as this event resulted in 81% more rain in June than the monthly climate normal for June.

General Site Descriptions

The study sites were chosen to investigate the effect of elevation, surface soil placement, re-vegetation types, site orientation (slope/aspect) and microtopography on the near-surface water balance (Table 1). The study site locations include two instrumented sites at Greenhills Operation North Thompson mine area that are part of a paired cover trial with a uniform sloped soil cover (GH_NTC) and bare waste rock (GH_NTW) slope. A soil cover was also placed on the Turn Creek dump (FR_TCR) at Fording River Operation. Three sites (LC_BHE, LC_RHE and LC_RME) were installed on benched plateaus of the West Line Creek dump at the Line Creek Operation at different elevations and with varying degrees of vegetation cover. A reclaimed mature forest site (FR_CSP) at the Fording River Operation provided an optimal scenario to understand the effect of mature tree re-vegetation on near-surface water retention and net percolation. Data collected on a south-facing, vegetated slope of the Bodie Dump (EV_BRD) at the Elkview Operation allow quantification of the effect of a thick grass-legume vegetation cover and slope aspect. Similarly, instruments installed at the Cardinal River Operation north-facing (CR_B5D) and south-facing (CR_B4D) slopes will be used to observe differences due to aspect and soil placement. The sloped surface in the Greenhills Rosebowl mine area (GH_RMS) and Cardinal River Cheviot area (CR_CMS) have been prepared using a mounding technique creating crests, mid-slopes and troughs, which will help evaluate the effect of mounding on re-vegetation success and overall NP rates.

Performance Monitoring Instrumentation

All sites were instrumented with both soil profile monitoring (suction, temperature and water content) and meteorological stations with the exception of CR_B4D, which was installed with only a net radiation sensor. Components of a soil monitoring station included one primary *in situ* water content monitoring station consisting of eight thermal conductivity (pore-water suction) and eight time-domain reflectometry (TDR, water content) sensors, and four secondary *in situ* water content monitoring stations, each consisting of four TDR sensors. Meteorological stations measured rainfall (CS700 Tipping Bucket Rain Gauge), air temperature and relative humidity (HC2-S3 Probe), wind speed/direction (R.M. Young Model 05103AP-10 Wind Monitor), net radiation (Net Radiation Kipp & Zonen model NR-LITE2 Net

Radiometer), and snow depth (Sonic Ranger 50A). Eddy covariance stations were installed at LC_RHE, LC_BHE, FR_CSP, and GH_NTW, and included a gas analyzer (either LICOR Li-7200 closed-path for forest and grasses, or a Li-7700 open-path CO_2/H_2O analyzers for bare waste rock), an ultrasonic anemometer (Windmaster, Gill Instruments), an air temperature and relative humidity sensor (HMP45 sensor, Vaisala), and a net radiometer (CNR4, Kipp & Zonen). Nine Gee lysimeters were installed under the trough, crest, and slope areas of the GH_RMS study to directly measure net percolation rates. Lastly, 24 sap flow sensors were installed in 12 trees at FR_CSP to measure sap flow velocity and estimate water uptake rates by each tree.

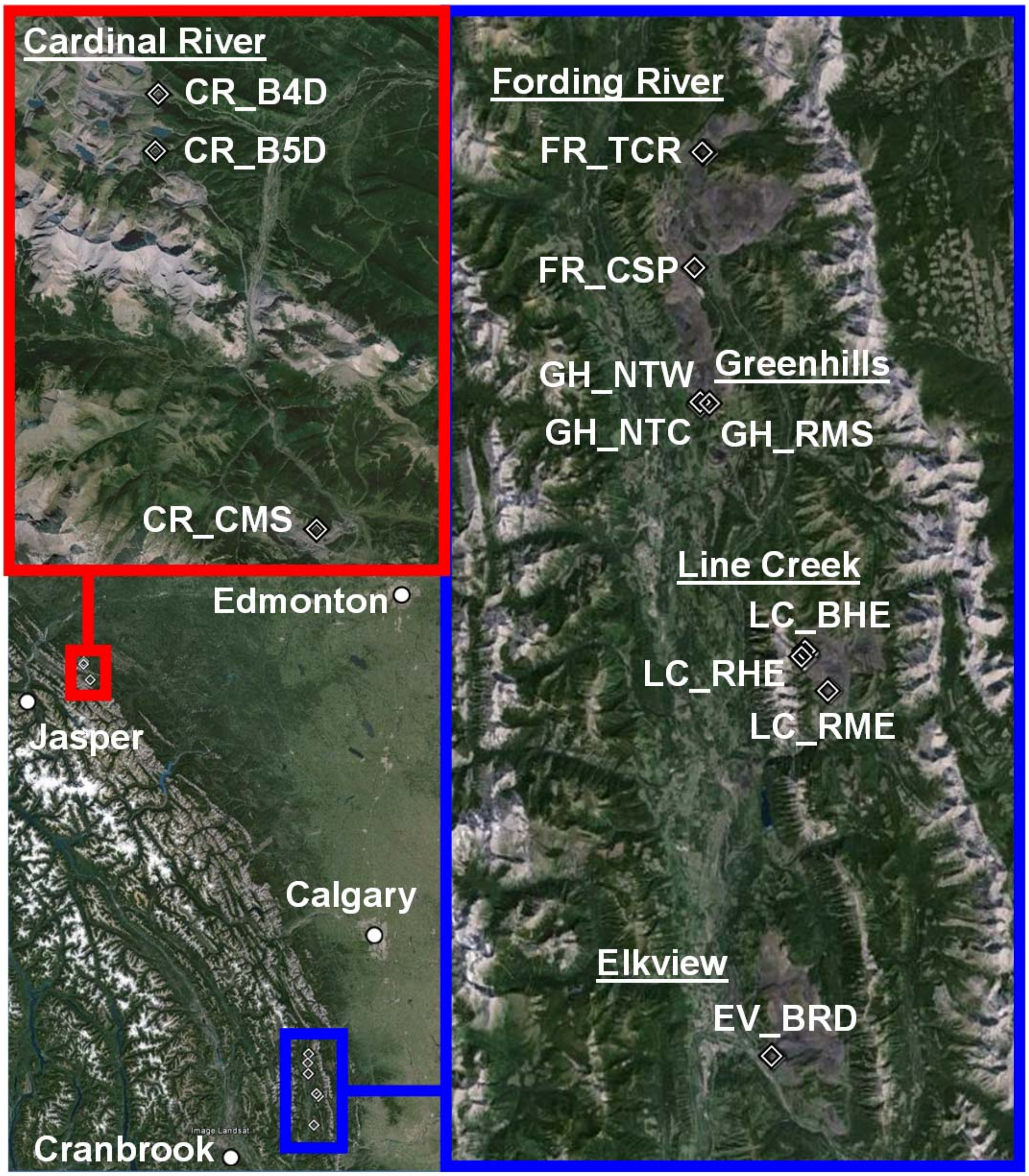


Figure 1 Aerial map showing locations of study sites.

Table 1
Summary of characteristics at study sites located in southeast BC and northwest AB

Site	Easting	Northing	Mine Area	Elevation (m asl)	Cover Soil	Surface	Slope (°)	Aspect	Topographic Shading	Vegetation
GH_NTW	5550564	651834.7	N Thompson Ck	1800	No	Smooth	26	W	No	Planted Seedlings
GH_NTC	5550651	651875	N Thompson Ck	1800	2-Lift Surface Soil	Smooth	26	W	No	Planted Seedlings/ Natural Regeneration
GH_RMS	5550429	652172.1	Rosebowl	1920	No	Mounds	26	SW	No	Planted Seedlings
LC_BHE	5535363	658257.9	West Line Ck	2150	No	Smooth/ Harrow	-	Level	No	Planted Seedlings
LC_RHE	5535020	658138.9	West Line Ck	2075	No	Smooth/ Harrow	-	Level	No	Agronomic Grasses/ Legumes
LC_RME	5533016	659655.7	West Line Ck	1790	No	Smooth/ Harrow	-	Level	No	Agronomic Grasses/ Legumes
FR_CSP	5559029	348453.1	C Spoil	1690	No	Smooth	26	E	No	~25 Year Old Regenerating Conifer Forest
FR_TCR	652279	5566138	Turn Creek	1800	Salvaged Soil/ Overburden	Smooth/ Harrow	-	Level	Yes	Planted Seedlings
EV_BRD	5510559	343964.9	Bodie Rock Drain	1470	No	Smooth	26	SW	No	Agronomic Grasses/ Legumes
CR_B5D	5879971	475328.5	Luscar B5	1630	Regolith	Smooth	26	NE	No	Agronomic Grasses/ Legumes
CR_B4D	5878010	474278.7	Luscar B4	1690	No	Smooth	26	S	Partial Winter	Agronomic Grasses/ Legumes
CR_CMS	5864953	479802.9	Cheviot 1930	1930	Salvaged Soil (Till)	Mounds	26	S	No	Planted Seedlings/ Natural Regeneration

*Substrate for all sites was waste rock.

Water Balance Analysis

Water balances were developed for each site over one hydrologic year starting October 1st, 2012 through September 30th, 2013. For these snow-dominated sites, water input was defined as the sum of rainfall minus rainfall interception, plus snowmelt (hereafter referred to as effective precipitation, P_{eff}). The following components of the water balance were measured or estimated to calculate daily changes in storage (ΔS) in the top ~1 m of material at the primary station profiles:

$$\Delta S = R + M - AE(T) - RO - NP \tag{1}$$

where: R is rainfall, M is snowmelt, $AE(T)$ is actual evaporation or evapotranspiration where appropriate, RO is surface runoff, and NP is net percolation (all units are mm d^{-1}). Runoff at all sites was assumed to be zero as a result of the high hydraulic conductivity of the waste rock and cover soils.

Precipitation

Rainfall (volume and intensity) was measured with a tipping bucket rain gauge (TBRG, Campbell Scientific 700). Precipitation was verified by comparing multiple TBRG records at sites with similar elevation, the precipitation record from the Sparwood, BC climate station (1140 masl, Environment Canada) and other total precipitation gauges operated by McMaster University and Teck. Precipitation was assumed to be rain if the mean daily air temperature was greater than 2°C.

Daily average snow depth was measured continuously using an automated Sonic Ranger 50A (SR50) sensor. Snow surveys completed at the end of March 2013 provided snow depth and density data to calculate a transect-average snow-water equivalent (SWE) across each site. A continuous record of snowpack SWE was estimated with snow density equations (Bartlett et al, 2006).

Melt

Point-based daily snow ablation amounts for each site were estimated using the energy balance equation and daily averages of meteorological data using the approach reported by Dingman (2002) and Burles and Boon (2011). The energy fluxes from ground heat flux and rain were assumed to be zero. Sensible heat was calculated as a function of measured wind speed and air temperature. Sublimation and evaporation were accounted for by the latent heat term which was a function of the vapour pressure gradient between the air and snow surface. The melt period was initialized with peak SWE and run to complete snowpack removal. The SR50 snow depth record in combination with field observations (including time-lapse photos) were used to determine the timing of snowpack disappearance.

Evapotranspiration

Potential evaporation was estimated using the Penman (1948) or Penman-Montieth combination method (Montieth, 1965) for sites with well-established vegetation. Canopy conductance (C_{can}) was estimated using leaf area index data collected in 2013 (Integral Ecology, 2013). A relationship between evapotranspiration (E(T)) and soil water deficits was used to relate potential evapotranspiration (PE(T)) to actual E(T) via the following equations:

$$AE(T) = PE(T) \cdot Available\ Water\ Ratio \tag{3a}$$

$$Available\ Water\ Ratio = (VWC - WP)/(FC - WP) \tag{3b}$$

where: VWC is the volumetric water content of the surficial soil profile (average of top 15 to 30 cm depending on vegetation and expected depth range accessible to evapotranspiration), WP is the wilting point and FC is the field capacity. WP and FC are defined as the volumetric water contents at 1500 and 10 kPa of suction, respectively. The available water ratio approached 1 at FC and declined as the availability of water decreased to the WP for plants (available water ratio = 0). Eddy covariance estimates of AE(T) were found to positively correlate with measured leaf area index (LAI) (Integral Ecology, 2013); therefore, AE(T) estimates calculated using Eq. 3a and 3b were constrained using the developed LAI-AE(T) relationship for conditions ranging from bare to vegetated ground.

Interception

Rainfall intercepted was assumed to be evaporated daily and estimated using a modified version of the forest rainfall interception model in Rutter and Morton (1977). Daily maximum rainfall interception was set equal to the canopy storage capacity for conifers (1 mm; Schmidt and Gluns, 1991). Intercepted snowfall in the trees at Fording River C-spoil was not considered during the first year of analysis.

Storage

Changes in water storage (ΔS) through the soil profile were calculated using the volumetric water content (CS616 Time Domain Reflectometry Sensors) measured at the primary monitoring station. The volume of water was weighted for sensor depth and summed over the entire profile.

Net Percolation

It was assumed that NP occurred when there was a downward vertical hydraulic gradient as calculated from the soil suction values between 75 and 100 cm depth and if there was a net loss in storage within the bottom half of the instrumented profile. For this case, NP was assumed to be equal to the net loss in storage from the bottom half of the profile, as it was assumed AE(T) fluxes would not affect water storage at that depth.

If the water storage capacity of the bottom soil profile exceeded the field capacity (FC; 10 kPa of suction) during snowmelt infiltration or rainfall events, then NP was estimated via the water balance in Eq. 1. This NP estimation method was used during snowmelt infiltration and rainfall events as there was often a surplus of water during melt and rainfall events if the gradient / storage method for calculating NP described above was used. It was assumed that seepage rates could be near steady-state during melt and rainfall events and that NP could be occurring even if there was not a measured decrease in storage.

Analytical Model Calibration

Water retention curves (WRC) were developed using the water content and suction monitoring data. These were then used to estimate values of FC and WP. Water retention curves, however, are difficult to develop for heterogeneous, coarse-textured materials. Estimates of AE(T) were most sensitive to FC and WP parameters; therefore, sensitivity of water balance estimates were assessed by varying the FC and WP values ± 2 to 10%. Measured water content and the match of measured and calculated water storage was used to inform FC and WP estimates and constrain AE(T) values. Assuming no runoff, water input that was not removed by AE(T) was then assumed to eventually report to NP.

RESULTS AND DISCUSSION

The soil water contents and meteorological conditions were variable, which was expected given the range of elevation, slope aspects, vegetation, soil covers, geographic location and surface preparation of the selected sites. Meteorological conditions at the various sites were similar but with key differences that can be attributed to elevation, geographic location, and slope aspect. Precipitation increased with increasing elevation. LC_BHE, the highest elevation site (2,150 m asl) had the greatest measured total annual precipitation (1,210 mm). For sites in the 1,500 to 1,800 m asl elevation range, total annual precipitation ranged from approximately 800 to 950 mm. Sites located at the Cardinal River Operation were greatly influenced by the dry, warm winds blowing down the eastern slopes of the Rocky mountains. Data records indicate snowpack disappearance occurred earlier on the south-facing CR_B4D slope compared to north-facing B5D likely due to higher solar energy input received on the south-facing slope. Earlier snowmelt and snowpack disappearance was also observed on other south-facing slopes including EV_BRD and CR_CMS.

Patterns in water balance trends emerged in the first year of analysis with NP into underlying waste rock typically decreasing for increased vegetation and soil cover, as well as for decreases in rainfall or snowmelt. The highest NP occurred at LC_BHE due to greater P_{eff}, low AE(T), and less water holding capacity in the coarse waste rock compared to other monitored sites (Table 2). Increased vegetation cover resulted in a greater volume of water removed from the near-surface through AE(T), that would have otherwise resulted in NP. For example, the lowest NP (as % P_{eff}) calculated was for the mature, reclaimed conifer forest site (FR_CSP) and the dense agronomic grass/alfalfa covered site (EV_BRD). Lower NP at EV_BRD compared to FR_CSP was attributed to a higher water holding capacity in the material underlying the thick grass cover combined with lower P_{eff} on a south-facing slope. As the individual site cover systems evolve, vegetation communities mature, and water balance trends become more apparent the mechanisms having the largest impact on controlling net percolation through waste rock dumps and achieving re-vegetation objectives can be identified, quantified and evaluated as potential watershed management tools.

Net percolation estimated for a soil covered, waste rock slope (GH_NTC) was approximately 15% (of P_{eff}) less than an adjacent bare waste rock slope (GH_NTW). Research has shown that cover systems have the potential to limit infiltrating water (i.e. NP) through a waste rock dump (MEND, 2012). Water storage in the shallow subsurface was greater in the finer-texture soil cover at GH_NTC compared to GH_NTW (Figure 2). NP through the soil cover is expected to further decrease in future years as vegetation establishes, thus more available soil water will be removed through AE(T) (Integral Ecology, 2012).

The water balances for the covered sites at FR_TCR and GH_NTC, and the mature forest site at FR_CSP, were analyzed over a shorter period than other sites as data was not available until later in the fall. NP estimates (as a % of P_{eff}) are likely slightly over-estimated for these sites as heavy rainfall in early October prior to the first day of the water balance estimations were not included in the P_{eff} values but may have been contributing to NP during the analysis period.

The weighted net percolation for a mounded, bare waste rock slope (GH_RMS), considering the overall effect of troughs and crests, was lower than that calculated for an adjacent smooth slope (GH_NTW) (Table 2). Net percolation estimates at the GH_RMS were corroborated by direct measurement of NP by lysimeters. Net percolation in a trough at GH_RMS was similar to the smooth slope at GH_NTW, but decreased at the crest and mounded mid-slope positions due to thinner snowpack (less snowmelt) from wind redistribution. Increased water availability in the troughs, as well as wind and sun shading, are expected to increase vegetation productivity over time in the troughs relative to the crests which will increase AE(T) and further decrease NP at GH_RMS.

Table 2

Cumulative annual water balance fluxes for all 12 study sites. All fluxes are presented in millimeters. Net percolation values are presented as a percentage of effective water inputs (rainfall plus snowmelt).

Site		Elev m asl	Description	Rain	Melt	Effective Precipitation (P_{eff})	Actual E(T)	Net Percolation (NP)	Meas ΔS	Calc ΔS	NP as % of P_{eff}	Sensitivity of NP (as % of P_{eff})
LC_BHE		2150	Bare, plateau	530	640	1170	**150**	970	60	50	**83%**	**79 – 85%**
LC_RHE		2075	Grass/legume, plateau	530	590	1120	**280**	780	65	65	**70%**	**68 – 71%**
LC_RME		1790	Grass/legume, plateau	630	270	900	**170**	640	90	85	**71%**	**68 – 73%**
GH_NTW		1800	Seedlings, bare, slope	510	430	940	**150**	750	30	35	**80%**	**77 – 83%**
GH_NTC[1]		1800	Seedlings, soil cover, slope	480	340	820	**180**	550	80	90	**67%**	**65 – 71%**
GH_RMS	**Trough**	1920	Seedlings, bare, mounded, slope	510	470	980	**150**	810[4]	10	15	**83%**	**79 – 85%**
	Crest	1920	Seedlings, bare, mounded, slope	510	130	640	**150**	450[5]	55	45	**70%**	**63 – 77%**
FR_CSP[2]		1690	Trees, bare, slope	620	350	970	**380**[6]	560	40	35	**58%**	**55 – 61%**
FR_TCR[3]		1800	Seedlings, soil cover, plateau	600	250	850	**150**	730	-30	-30	**86%**	**82 – 89%**
EV_BRD		1470	Grass/legume, slope	690	190	880	**320**	460	100	100	**52%**	**49 – 55%**
CR_B5D		1630	Grass/legume, slope	460	420	880	**330**	590	-45	-40	**67%**	**64 – 70%**
CR_B4D		1690	Grass, legume, slope	460	400	860	**250**	650	-45	-40	**76%**	**72 – 79%**
CR_CMS		1930	Seedlings, cover, mounded, slope	450	300	750	**200**	570	-20	-20	**76%**	**72 – 79%**

[*] Runoff assumed to be zero for all sites.

[1] Water balance period starting 18 Oct 2012

[2] Water balance period starting 15 Oct 2012

[3] Water balance period starting 25 Oct 2012

[4] Drainage measured using Gee lysimeters was 780 mm

[5] Drainage measured using Gee lysimeters was 65 mm

[6] Rainfall canopy interception was estimated to be 70 mm of AE(T)

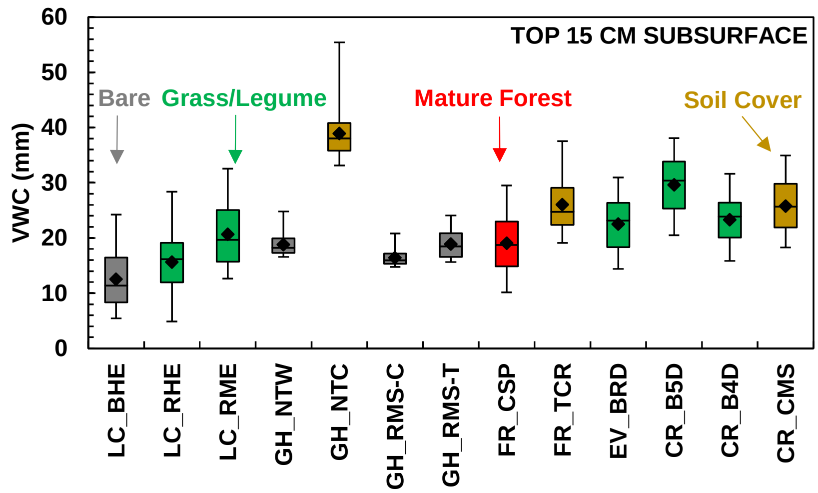


Figure 2 Box and whisker plot showing inter-site variability in mean daily VWC for the top 15 cm of material during the non-frozen period (May 1st to September 30th, 2013). Whiskers indicate the maximum and minimum values, the box represents the 25th and 75th percentiles, while the median values are represented by the mid-range lines. Mean values are presented as black diamonds to indicate the skew in the distribution of the data set.

CONCLUSION

Patterns in near-surface water balances have begun to emerge in the first year of monitoring of a range of waste rock dump reclamation surface management options. Net percolation as estimated from data collected from field moisture and climate instruments decreased with increasing vegetation cover and the addition of a surficial soil cover. Overall NP at a mounded waste rock site was estimated to be less than for an adjacent smooth waste rock slope. The effect of mounding on NP should be considered on a site-specific basis, however, as differences in precipitation, depth of troughs, slope aspect and prevailing wind direction will influence the results.

REFERENCES

Bartlet, P.A., MacKay, M.D., and Verseghy, D.L. 2006. Modified snow algorithms in the Canadian Land Surface Scheme: Model runs and sensitivity analysis at three Boreal forest stands. *Atmos. Ocean* 43: 207-222.

Burles, K. and Boon, S. 2011. Snowmelt energy balance in a burned forest plot, Crowsnest Pass, Alberta, Canada. *Hydrological Processes* 25: 3012-3029.

Chapman, P.M., Berdusco, R.J., and Jones, R. (2007). Update on the status of selenium investigations in the Elk River Valley, B.C.

Dingman, S.L. 2002. Physical Hydrology, 2nd Edn. Prentice-Hall Inc: Upper Saddle River, New Jersey; 600 pp.

Integral Ecology Group Ltd (2013). Teck Resources Limited Applied Research and Development Program: 2013 reclamation assessment – Draft.

Integral Ecology Group Ltd (2012). Teck Resources Limited Applied Research and Development Program: Evapotranspiration and vegetation contributions to soil water balances on reclaimed sites – an interpretive review. Project No. TCLWRD-11.

Lussier, C., Veiga, V., & Baldwin, S. (2003). The geochemistry of selenium associated with coal waste in the Elk River Valley, Canada. Environmental Geology 44 (8): 905-913.

MEND 1.61.5c. 2012. Cold Regions Cover System Design Technical Guidance Document. Canadian Mine Environment Neutral Drainage Program.

Monteith, J.L. 1965. Evaporation and environment. In New York, NY: Cambridge University Press, *Proceedings of the 19th Symposium of the Society for Experimental Biology*, 205-233.

Penman, H. L. (1948). Natural evaporation from open water, bare soil and grass. *Proceedings of the Royal Society of London. Series A. Mathematical and Physical Sciences* 193: 120-145.

Schmidt, R. A., and Gluns, D. R. 1991. Snowfall interception on branches of three conifer species. *Canadian Journal of Forestry Research* 21: 1262-1269.

GREENHOUSE TRIALS ON USE OF BIOCHAR VERSUS PEAT FOR LAND RECLAMATION PURPOSES

Elizaveta Petelina, B.Sc., MSEM[1]
Alexey Klyashtorin, Ph.D[2]
Tamara Yankovich, Ph.D[3]

[1]Remediation Specialist
[2] Senior Environmental Scientist
Saskatchewan Research Council, Saskatoon, Canada

[3]Radiation Protection Specialist
International Atomic Energy Agency, Vienna, Austria

ABSTRACT

Experimental testing has been conducted to determine the effect of amendment of soil with mineral fertilizer, biochar, and peat on emergence, survival, establishment, and productivity of northern boreal plant species grown on poor substrates in a controlled environment (in greenhouse). Four species (slender wheatgrass, rocky mountain fescue, American vetch, and common yarrow) were grown in pots containing poor sandy soil amended with mineral fertilizer and/or organic amendments (peat or biochar). The greenhouse temperature, lighting and pot watering regimes simulated the conditions of vegetation growing season in northern boreal forest. The trials showed that mineral fertilizer had a limited ability to promote plant growth compared to peat and biochar. Both types of organic amendments had similar positive effects on the establishment of two test species (American vetch and common yarrow) and on the growth of three test species (slender wheatgrass, rocky mountain fescue and American vetch). Peat had a stronger positive effect on emergence of slender wheatgrass and establishment of rocky mountain, while biochar promoted the common yarrow growth better than peat.

KEY WORDS

Revegetation, native boreal species, organic amendment, mineral fertilizer, soil

INTRODUCTION

Revegetation is a final stage of any land reclamation project aimed at provision of both wildlife habitat and long-term erosion control. In this respect, ideally, the vegetation cover will consist of native plant species, perform its ecosystem functions, and be self-sustaining, i.e. cover not require additional treatment after the plant community reaches its steady state; however, pursuing such an ambitious goal at disturbed sites in a boreal zone may face serious challenges, such as limited knowledge of the ability of plant species to recover quickly under harsh climatic conditions, and poor physical and chemical properties of growth media (Mining Watch Canada, 2001; Daniels and Zipper, 2010; Landline et al., 2011).

The purpose of this research was to run greenhouse trials using native boreal species and various soil amendments for application in disturbed areas at the Gunnar Mine Site, an abandoned uranium mine/mill site located on the northern shore of the Athabasca Lake and managed by the Saskatchewan Research Council (SRC). One of the project tasks is to establish self-sustaining vegetation on the engineered cover over tailings (SRC, 2014). The cover material is to be taken from the local airstrip and neighboring areas. The proposed borrow material is coarse sand with gravel inclusions and very low organic matter content (less than 0.1%), and has a limited capacity to support vegetation growth. It was recognized recently that biochar (a byproduct of pyrolysis) could promote soil fertility (Verheijen et al. 2009; Biederman and Harpole 2012; Adams et al. 2013), so it was decided to test the effectiveness of biochar, along with conventional soil conditioners (i.e. mineral fertilizer and peat), to promote establishment of vegetation.

METHODS

The greenhouse trials involved growing four plant species (i.e. Slender Wheatgrass (*Elymus trachycaulus*), Rocky Mountain Fescue (*Festuca saximontana*), American Vetch (*Vicia americana*), Common Yarrow (*Achillea millefolium*)) in pots containing combinations of borrow material collected in the vicinity of the Gunnar Mine Site with mineral fertilizer and two organic amendments (peat and biochar) (treatments are listed in Table 1). The experiment had a randomized design with five replicates of each soil mixture and plant species.

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

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

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

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

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

Before seeding, all pots were excessively watered to imitate spring snowmelt conditions, and fertilizer was applied to the corresponding pots after seeding. Saskatchewan Forage Council (SFC, 1998) recommendations on slender wheatgrass cultivation were used as a basis for fertilizer rates, which were 45 N kg/ha, 84 P2O5 kg /ha, and 112 K2O kg/ha for soils with poor nutrient content.

The trial time period was 12 weeks, which is close to the vegetation growing season at Gunnar. During the trial period, the pots were rotated weekly to avoid the edge effect, and were watered every third day at a rate imitating the northern Saskatchewan average monthly precipitation that varies from 38 mm (week 1-4) to 53 mm (week 5-12) (SRC, 2014). During the trials, the seedling number in every pot was measured weekly. In the third week of the trials, it was noticed that direct sunlight might overheat the soil mixtures with biochar because of its black color, impeding seed germination and growth. To avoid such undesirable effects, the greenhouse shades were closed. No other changes in temperature or the water regime were made. At the end of the experiment, the aboveground biomass from each pot was harvested, dried, and weighed.

The experimental data were further processed and analyzed to quantify the following indices: plant establishment rate, seedling emergence rate, seedling survival rate, and aboveground biomass dry weight. All data were tested for normality using the Shapiro-Wilk test. If data did not fit a normal distribution, the Kruskal-Wallis test, followed by the Conover-Iman test, were used to assess statistical differences in the response of the investigated indices to the soil treatments. If data were normally distributed, analysis of variance (ANOVA), followed by the Tukey's HSD (honestly significant difference) test, were run using the XLSTAT statistics program for all data groups. The significance level for all tests was 0.05.

RESULTS

Table 1 provides information on statistically significant effects of soil amendment application on the aboveground biomass dry weight (ABDW), seedling emergence rate (SER), seedling survival rate (SSR), and plant establishment rate (PER) for each plant species grown on the test soil mixtures.

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

SSR of the other three plant species (p varied from 0.144 to 0.977, depending on the index and plant species).

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

Biochar addition to the borrow material fostered the growth of all four plant species, increasing ABDW by a factor of 4 for slender wheatgrass (from 62 to 265 mg per pot; $p<0.001$), 8 for rocky mountain fescue (from 62 to 465 mg per pot; $p<0.001$), 20 for American vetch (from 8 to 161 mg per pot; $p<0.001$), and 121 for common yarrow (from 2 to 218 mg per pot; $p<0.001$). This treatment also promoted common yarrow seedling emergence and seedling survival, increasing SER and SSR by factors of 1.7 (from 39 to 66%; $p=0.037$) and 9 (from 11 to 96; $p<0.001$), respectively, which resulted in an increase of PER by a factor of 16 (from 4 to 63%; $p<0.001$). The favorable effect of biochar on mountain fescue and American vetch resulted in increases in SSR by factors of 1.3 (from 77 to 99%; $p<0.001$) and 3 (from 35 to 100%; $p<0.001$), respectively, for these species, which resulted in an increase in PER by a factor of 1.2 (from 59 to 73%; $p=0.013$) for rocky mountain fescue and 4 (from 13 to 49%; $p=0.001$) for American vetch. Biochar application, however, impeded slender wheatgrass SER, decreasing SER by a factor of 1.3 (from 87 to 69%; $p=0.006$), yet it had no pronounced effect on the SSR ($p=0.532$) or PER ($p=0.107$).

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

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

Table 1. Statistically significant effects of various soil treatments on plant establishment and biomass ($p<0.05$)

Compared soil mixtures	Ratio between the mean index values															
	Plant establishment rate, %				Seedling emergence rate, %				Seedling survival rate, %				Aboveground biomass dry weight, mg per pot			
	SW*	RMF	AV	CY	SW	RMF	AV	CY	SW	RMF	AV	CY	SW	RMF	AV	CY
BM-NPK : BM-control	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	59:35	n/a	100:62	n/a	n/a	n/a
BM-Peat : BM-control	n/a	94:59	49:13	75:4	n/a	94:78	n/a	87:39	n/a	98:77	100:35	86:11	209:62	373:62	111:8	131:2
BM-Biochar : BM-control	n/a	73:59	49:13	63:4	69:87	n/a	n/a	66:39	n/a	99:77	100:35	96:11	265:62	465:62	161:8	218:2
BM-Peat : BM-Biochar	90:69	94:73	n/a	n/a	90:69	94:74	n/a	n/a	n/a	n/a	n/a	86:96	n/a	n/a	n/a	131:218

* SW – slender wheatgrass; RMF – rocky mountain fescue; AV – American vetch; CY – common yarrow; BM – borrow material; NPK – mineral fertilizer

A summary of the effects of amendment of soil with peat and biochar on the tested plant species is as follows:

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

DISCUSSION AND CONCLUSION

The greenhouse trials demonstrated that mineral fertilizer had a limited ability to promote growth of boreal plant species, which is likely due to the low water holding capacity of the growing media (borrow material). Addition of mineral fertilizer only increased the amount of nutrients in the soil and did not improve water availability; therefore, plants did not improve, as a result.

Conversely, addition of organic amendments had a much more pronounced effect. Study results suggested that biochar could be a good substitute for peat as a soil amendment agent. Also, peat and biochar had different effects on the establishment and growth of different plant species. Establishment of rocky mountain fescue was promoted by peat application to the larger extent than by biochar, but growth of common yarrow was fostered by biochar application to the larger extent than by peat.

The establishment of slender wheatgrass was impeded by biochar application. This “negative” impact of biochar on seedling emergence can be explained by a darker coloration of the biochar-amended soil, which may have caused soil overheating when exposed to direct sunlight, compared to other test materials, which were lighter in color (borrow was yellowish and peat was brown). This could result in both increased seed embryo mortality, and increased evaporation from the borrow material and biochar mixture, which prevented seed germination and increased seedling mortality due to soil desiccation. It is likely that the biochar performance, in terms of plant establishment, could be improved by mulching the soil surface.

Although the trial results were positive, the final choice of a preferred organic amendment can only be made with consideration of the following aspects:

- Although greenhouse trials provide some general indication of the plant response to tested soil treatments, they are not sufficient to draw a definitive conclusion regarding the impact of soil amendment on plant growth and reproduction; to overcome this uncertainty, it is essential to conduct field trials under the Gunnar Mine Site conditions.

- The above-mentioned response of individual plant species to soil amendment will not necessarily be the same or similar to the response of a desired plant community; therefore, it is necessary to perform field trials using seed mixtures.
- Only one rate of mineral fertilizer was tested for each combination of soil and organic amendment. Additional trials should be run using different rates of mineral fertilizer to optimize the fertilizer rates and combinations.

In conclusion, although useful results have been achieved through greenhouse testing, complementary field trials are necessary to compare the soil treatment options under natural conditions, and a feasibility study should be undertaken to compare biochar versus peat as organic amendments.

ACKNOWLEDGMENTS

Funds required for this work were provided by SRC under CLEANS project. We wish to thank our colleagues Bob Godwin, Mary Moody, Miguel Providenti, and Jeff Thorpe for their advice and useful discussions during development of the project design and result interpretation. We are grateful to Dawn Bolin and Roxanne Suchan (SRC) for their help during the greenhouse trials. And we would like to single out David Telfer (TSM Telfer Seed Marketing) who kindly gifted seeds for our greenhouse trials.

REFERENCES

Adams M.M., T.J. Benjamin, N. C. Emery, S. J. Brouder and K. D. Gibson. 2013. The effect of biochar on native and invasive prairie plant species. Invasive Plant Science and Management 6(2): 197-207.

Biederman, L.A. and W.S. Harpole. 2012. Biochar and its effects on plant productivity and nutrient cycling: a meta-analysis. GCB Bioenergy, 5: 202-214.

Burton, C.M., and P.J. Burton. 2003. A Manual for Growing and Using Seed from Herbaceous Plants Native to the Interior of Northern British Columbia. Symbios Research & Restoration, Smithers, B.C. 168 p.

Daniels, W.L. and C. E. Zipper. 2010. Creation and management of productive minesoils. Virginia Cooperative Extension (VCE) publications. www.ext.vt.edu

Landine, P., D. Fenske, and R. Van Stone. 2011. Thirty years of revegetation experience at the Key Lake uranium mine in northern Saskatchewan. In: Bourie, A.B., M. Tibbett, and A. Beersing (eds.). Mine Closure 2011: Vol. 1 (pp. 47-56). Nedlands: Australian Centre for Geomechanics. 646 p.

MiningWatch Canada. 2001. Mining in remote areas: Issues and impacts -- A community primer. 36 p. http://www.mwatch.koumbit.org

SFC (Saskatchewan Forage Council). 1998. Seed production of Slender Wheatgrass. 12pp.

SRC (Saskatchewan Research Council). 2014. Gunnar site remediation project, environmental impact statement. SRC publication, Saskatoon, SK. 920 pp.

Verheijen, F., S. Jeffery. A.C. Bastos, M. van der Velde, and I. Diafas. 2009. Biochar Application to Soils - A Critical Scientific Review of Effects on Soil Properties, Processes and Functions. EUR 24099 EN, Office for the Official Publications of the European Communities, Luxembourg, 149pp.

FIELD TRIALS ON USE OF BIOCHAR VERSUS PEAT FOR LAND RECLAMATION PURPOSES

Elizaveta Petelina, B.Sc., MSEM[1]
Alexey Klyashtorin, Ph.D[2]
Tamara Yankovich, Ph.D[3]

[1]Remediation Specialist
[2] Senior Environmental Scientist
Saskatchewan Research Council, Saskatoon, Canada

[3]Radiation Protection Specialist
International Atomic Energy Agency, Vienna, Austria

ABSTRACT

Biochar application for revegetation purposes in northern Saskatchewan was studied to determine its effectiveness as a soil amendment in establishing sustainable vegetative soil covers. The abandoned Gunnar Mine Site, located on the northern shore of Lake Athabasca, served as a study area to test the effectiveness of biochar as a soil amendment. Field trials were carried out to compare the effect of biochar and peat application on the growth and establishment of native plant species. The field trials showed that peat promotes vegetation cover establishment better than biochar. Biochar also had a positive effect on vegetation recovery through both establishment of seeded plants and self-establishment of natural invaders (plant species not seeded during the experiment). Peat and biochar had different effects as soil amendments, depending on the plant species. It was shown that both peat and biochar can be used to promote plant establishment and growth, but biochar effectiveness may vary depending on its properties.

KEY WORDS

Revegetation, Native Boreal Species, Organic Amendment, Mineral Fertilizer, Soil

INTRODUCTION

The establishment of vegetation cover on disturbed mine sites is an important task of mine closure to protect the soil surface from wind and water erosion, to restore wildlife habitats, and to create opportunities for sustainable development of local Aboriginal communities. Properties of growth media is a significant factor of the revegetation success. In northern environments, fertile soil layer (topsoil) is often very thin, with low organic matter and nutrient contents. Topsoil also can be easily lost as a result of mining activities. Therefore, soil organic amendments and mineral fertilizers are often applied to improve soil properties and increase effectiveness of revegetation activities. Transportation of organic media to remote northern sites is very expensive because of their low bulk density, whereas local harvesting for organic materials (usually peat) destroys natural wetlands which are valuable natural habitat. As a result, there is an emerging need for alternative organic soil amendments to avoid costs associated with material transport, while minimizing detrimental impacts to wetland habitats.

Biochar is a solid material obtained from the carbonisation of biomass through pyrolysis that can be used as a soil amendment to improve its chemical and physical properties (Lehman and Joseph 2009; Verheijen et al. 2009). In addition, biochar application has been shown to create favorable conditions for soil microbiota, promote plant growth, and increase plant resistance to disease (Verheijen et al. 2009; Biederman and Harpole 2012; Elad et al. 2012). Therefore, this organic amendment can be beneficial for establishment of sustainable vegetative covers on soils. Potentially, biochar can be produced on-site or in nearby communities from local feedstock (e.g. organic wastes), which makes it a promising substitute for conventional organic amendments (Roberts et al. 2009). On the other hand, most biochar research is focused on its effect on cultivated crops and few studies consider its benefit to establishing native plant species (Elad et al. 2012; Adams et al. 2013; Sovu et al. 2013). Thus, there is still no unanimous opinion regarding the applicability of biochar to establish vegetative covers and which trades off can be associated with its application.

The purpose of this research was to test the effectiveness of biochar as a soil amendment for mine site remediation in northern Saskatchewan. The Gunnar Mine Site, located on the northern shore of Lake Athabasca, was selected as a study site for the research, since one of the project tasks is to establish self-sustaining vegetation on engineered covers that are to be installed over the Gunnar tailings deposits (SRC 2014). The cover (borrow) material, which will likely be taken from the local airstrip and neighboring areas, is coarse sand with gravel inclusions, that has a very low organic matter content (less than 0.1%), and a limited capacity to support plant establishment and growth. As a result, application of organic amendments and mineral fertilizer will be necessary to enhance its properties as a growth medium. In 2011, two organic amendments (peat and biochar) and mineral fertilizer were tested in a greenhouse to study the response of native plant species to various soil amendments and treatments. It was shown that both peat and biochar boost plant establishment and growth, but the response differs between plant species (Petelina et al., 2014). The greenhouse trails were carried out with isolated species; and obtained results had a limited validity in relation to effect of soil treatment on establishment of desired plant community under natural conditions. Therefore, it was decided to run field trails at the Gunnar Mine Site.

METHODS

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

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

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

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

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

- Rocky Mountain Fescue (*Festuca saximontana*) – 20%
- American Vetch (*Vicia americana*) – 20%
- Streambank Wheatgrass (*Elymus lanceolatus ssp. riparius*) – 10%
- Slender Wheatgrass (*Elymus trachycaulus*) – 10%
- Violet Wheatgrass (*Elymus violaceus*) – 10%
- Tufted Hairgrass (*Deschampsia caespitosa*) – 7%
- Rough Hair Grass (*Agrostis scabra*) – 7%
- Canada Buffaloberry (*Shepherdia canadensis*) – 6%
- Canadian Milkvetch (*Astragalus Canadensis*) – 4%
- Marsh Reed Grass (*Calamagrostis canadensis*) – 3%
- White Bluegrass (*Poa glauca*) – 1%
- Alpine Milkvetch (*Astragalus alpinus*) – 1%
- Prairie Crocus (*Anemone patens*) – 1%
- Fireweed (*Chamerion angustifolium*) – 0.1%

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

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

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

RESULTS

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

Figure 1 shows the total vegetation cover (TVC), seeded plant cover (SPC), rough cinquefoil cover (RCC) and strawberry blite cover (SBC) on the tested soil mixtures.

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

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

- peat at the low rate was combined with fertilizer at the high rate (TVC=33%, $p<0.001$; SPC=10%, $p<0.001$; RCC=15%, $p<0.001$)
- peat at the medium rate was combined with fertilizer at the high rate (TVC=39%, $p<0.001$; SPC=10%, $p<0.001$; RCC=15%, $p<0.001$)
- peat at the high rate was combined with fertilizer at the low rate (TVC=44%, $p<0.001$; SPC=8%, $p<0.001$; RCC=44%, $p<0.001$) and the high rate (TVC=28%, $p<0.001$; SPC=10%, $p<0.001$; RCC=25%, $p<0.001$).

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

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

Seeded Species			Invading species		
Scientific Name	English Name	Origin	Scientific Name	English Name	Origin
Agrostis scabra	rough hair grass	native	*Achillea millefolium*	common yarrow	native
*Astragalus alp*inus	alpine milkvetch	native	*Arabis hirsuta*	hirsute rock cress	native
Astragalus Canadensis	Canada milkvetch	native	*Arabis holboellii*	reflexed rock cress	native
Brassica napus	canola	exotic*	*Artemisia campestris*	sagewort wormwood	native
Calamagrostis canadensis	marsh reed grass	native	*Chenopodium album*	lamb's quarter	exotic
Chamerion angustifolium	fireweed	native	*Chenopodium capitatum*	strawberry blite	native
Deschampsia cespitosa	tufted hairgrass	native	*Crepis tectorum*	annual hawksbeard	exotic
Elymus trachycaulus	slender wheatgrass	native	*Geranium bicknellii*	bicknell's geranium	native
Elymus lanceolatus ssp. riparius	streambank wheatgrass	native	*Matricaria matricarioides*	pineapple weed	exotic
Elymus violaceus	violet wheatgrass	native	*Plantago major*	common plantain	exotic
Festuca saximontana	rocky mountain fescue	native	*Polygonum aviculare*	prostrate knotweed	native

Poa glauca	white bluegrass	native	*Potentilla bimundorum*	staghorn cinquefoil	native
Shepherdia canadensis	Canada buffaloberry	native	*Potentilla norvegica*	rough cinquefoil	native
Vicia americana	American vetch	native	*Rorippa palustris*	bog yellowcress	native
			Salix spp.	willow	native
			Taraxacum officinale	common dandelion	exotic

*- seeded by accident

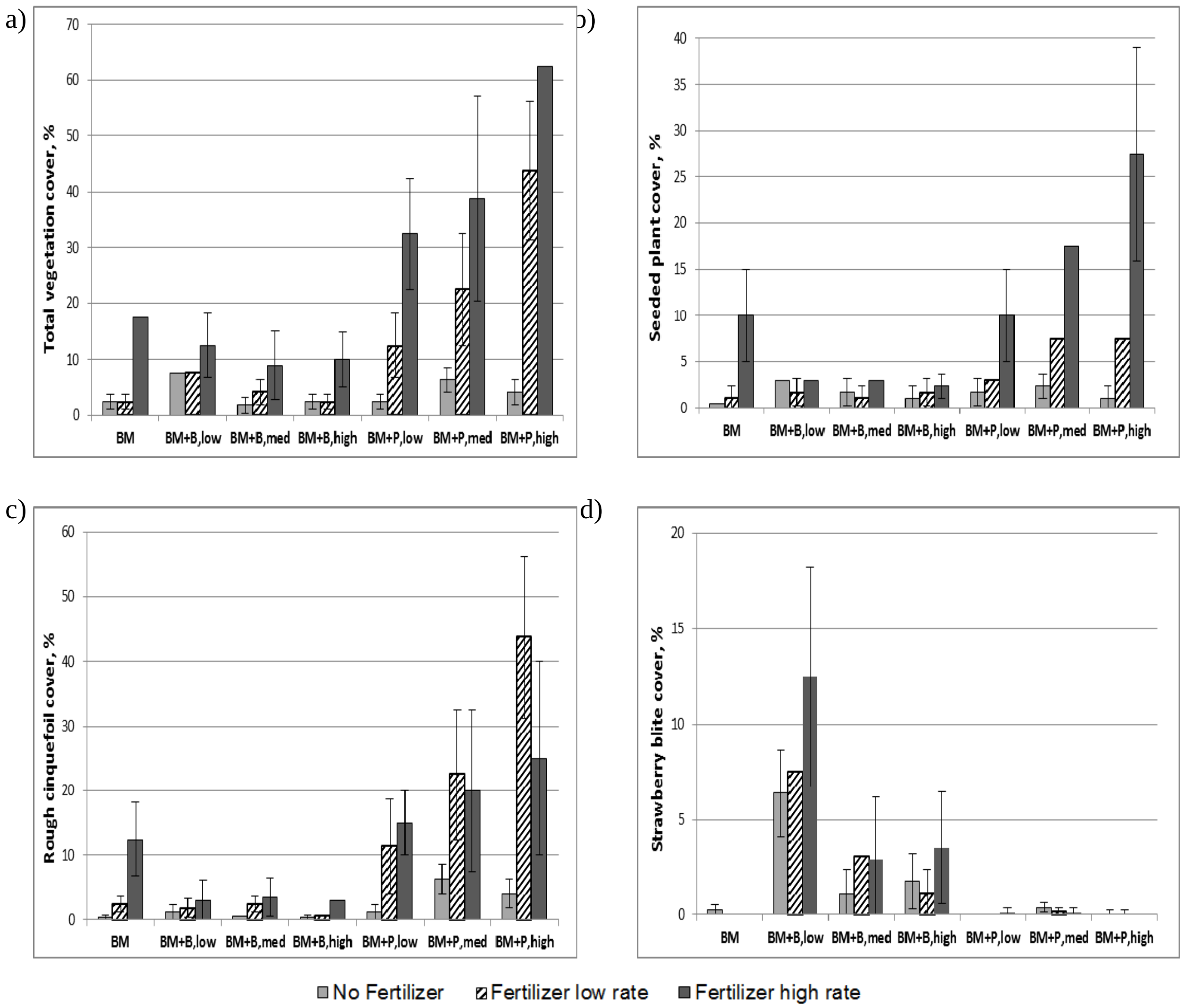


Figure 1. Effect of mineral fertilizer, biochar, and peat on the total vegetation cover (a), seeded plant cover (b), rough cinquefoil cover (c), and strawberry blite cover (d), for field trials. Error bars indicate standard deviation (absence of error bar means that the standard deviation is zero). BM – borrow material; B, low/med/high – biochar added at low/medium/high rate; P, low/med/high – peat added at low/medium/high rate.

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

Interestingly, when biochar was applied alone, the increase of its rate from low to high resulted in a significant decrease of TVC (from 6 to 2%, $p<0.001$), SPC (from 3 to 1%, $p=0.006$) and SCC (from 6 to

2%, $p=0.005$). When biochar was applied with fertilizer at the low rate, the same trend was observed, only for TVC. TVC decreased from 8 to 2% when the biochar rate was increased from low to high ($p<0.001$). There was no significant difference between the indexes when biochar at different rates was applied with fertilizer at the high rate (p varied from 0.092 to 1.000).

DISCUSSION AND CONCLUSION

Both biochar and peat promoted establishment of vegetation, but peat appeared to be more effective than biochar. The better performance of peat as an organic amendment is likely due to its high water holding and sorption capacity. Water holding capacity of the peat was eight times higher than that of the biochar; therefore, borrow material mixed with peat had a higher propensity to retain water and nutrients compared to the same rate with biochar. Peat also had a finer structure than chunky biochar, and borrow/peat mixtures were more homogenous than the soils amended with comparable rates of biochar.

Invader species abundance was different in plant communities grown without organic amendments versus with peat and biochar (e.g. rough cinquefoil dominated on the plots with peat and strawberry blite dominated on the plots with biochar). As the borrow material for the trials was taken from the same borrow area and boxes were placed and filled randomly, the observed difference in the species composition depended on treatment and species features with respect to their soil specialization. As peat had higher ability to hold and retain water and nutrients than biochar, therefore, its presence in the borrow material was more favorable for those plants normally growing in wetter areas, such as rough cinquefoil, tufted hairgrass, and other native grasses (e.g. marsh reed grass or streambank wheatgrass). Faster development of all these plant species under favorable conditions made them stronger competitors, i.e. they could develop faster than other species, impeding their development. Biochar addition to the borrow material also improved its properties, but to a lesser degree than peat (the water holding capacity of the biochar+borrow mixtures was approximately two-fold lower than that of the peat/borrow mixtures). Those plant species adapted for moist conditions were less competitive than other species on the soils with biochar. Thus, biochar addition to the borrow material created more favorable conditions for ruderal species such as strawberry blite, annual hawksbeard, reflexed rockcress, and sagewort wormwood, all known as pioneer species on disturbed areas with relatively low water and nutrient content.

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

Although both biochar and peat treatments showed a significant positive impact on plant establishment, peat is likely to be a more suitable organic amendment for the future revegetation at the Gunnar Mine Site.

The purpose of the Gunnar project is to establish vegetation cover which meets the following requirements:

- to ensure erosion protection of the engineering cover on the tailings areas;
- to impede the establishment of trees and shrubs, which could damage the cover due to penetration by roots and could bring contaminated groundwater to the surface via root transport;
- to be self-sustaining, i.e. able to persist for a long time without additional maintenance; and
- to minimize presence of exotic species to the lowest practicable level.

To meet the first and second requirements, the vegetation cover should cover at least 30-40% in the first year after seeding (Matheus and Omtzigt, 2012). Our results show that this requirement is achievable when the seed mixture is applied at a rate of 2000 PLS/m^2, and when seeded on soils amended with both peat and fertilizer, which in our case corresponded to the following treatments:

- peat at any rate (i.e. 80, 160, or 240 tonnes/ha) with fertilizer at the high rate (i.e. 45 N kg/ha, 84 P_2O_5 kg /ha, 112 K_2O kg/ha, and 20 S kg/ha), or
- peat at high rate (i.e. 240 tonnes/ha) with fertilizer at the low rate (i.e. 22 N kg/ha, 56 P_2O_5 kg /ha, 56 K_2O kg/ha, and 10 S kg/ha).

The above treatments also are beneficial to comply with the third and fourth requirements of the Gunnar project for the following reasons:

- The tallest grasses and the highest frequency of mature tufted hairgrass and wheatgrass were also observed at the above treatments. This suggests that plant communities for these treatments have the highest chances to be self-sustaining without additional investments.
- As discussed above, peat promotes the establishment of native plants, thereby increasing the plant community resilience to exotic species invasion.

In conclusion, it must be emphasised that the above results and recommendation are based on a single year of trials and should be used with caution for the following reasons:

- All the trial plant species are perennial, and one year is not adequate for most of the species to gain maturity. Until we have evidence that most seeded plants have reached maturity and started reproduction, we cannot make a decisive conclusion about self-sustainability of the newly established plant communities.
- Some species that emerged on the plots may not survive the harsh Gunnar winter; therefore, the plant winter hardiness should be also tested.
- The revegetation trials have been affected by the unusually wet conditions in July, 2012. Excessive watering of the trial plots after seeding could promote emergence, survival, and establishment of the seeded plants. It may also be particularly true for the plants associated with wetter conditions. Long-term observations of the plots will help to define which plant species were affected by the extreme weather conditions and which of them can persist under site-specific conditions at Gunnar.
- The species composition of the newly established plant community may change drastically with time, e.g. due to fertilizer depletion or exotic species invasion; therefore, longer-term observations are required to determine which revegetation technique is most successful for establishment of a self-sustaining and resilient ecosystem over the longer term.

Thus, monitoring of the trial plots should be continued for a few year to make conclusive recommendations for a larger scale revegetation effort at the Gunnar Mine Site.

ACKNOWLEDGMENTS

Funds required for this work were provided by SRC under the CLEANS project. We wish to thank our colleagues Bob Godwin, Mary Moody, Miguel Providenti, and Jeff Thorpe for their advice and useful discussions during the development of the project design and the interpretation of results. Our special thanks to Shasta Lysohirka and Michael Bendzsak (SRC), who assisted in the field during the experiment set up and data gathering.

REFERENCES

Adams M.M., T.J. Benjamin, N. C. Emery, S. J. Brouder and K. D. Gibson. 2013. The effect of biochar on native and invasive prairie plant species. Invasive Plant Science and Management 6(2): 197-207.

Bailey, A. W. and C. E. Poulton. 1968. Plant communities and environmental interrelationships in a portion of the Tillamook burn, northwestern Oregon. Ecology. 49:1–13.

Biederman, L.A. and W.S. Harpole. 2012. Biochar and its effects on plant productivity and nutrient cycling: a meta-analysis. GCB Bioenergy, 5: 202-214.

Elad, Y., E. Cytryn, Y. Meller Harel, B. Lew and E.R. Graber. 2012. The biochar effect: plant resistance to biotic stresses. Phytopathologia Mediterranea 50(3): 335-349.

Lehmann, J. and S. Joseph. 2009. Biochar for environmental management: an introduction. IN: Biochar for Environmental Management (Lehmann J. and S. Joseph eds). Earthscan, London. pp. 1–12.

Matheus, P. and Omtzigt T. 2012. Yukon Revegetation Manual. Available from http://yukoncollege.yk.ca/downloads/Yukon%20-Revegetation-Manual.pdf (accessed October 15, 2012)

Petelina, E., A. Klyashtorin, and T. Yankovich. 2014. Greenhouse Trials on Use of Biochar Versus Peat for Land Reclamation Purposes. 38th British Columbia Mine Reclamation Symposium.

Roberts, K., B. Gloy, S. Joseph, N. Scott and J. Lehmann. 2009. Life cycle assessment of biochar systems: Estimating the energetic, economic and climate change potential. Environment Science and Technology 44: 827-833.

Sovu, T.M., P. Savadago and P.C. Odén. 2012. Facilitation of forest landscape restoration on abandoned swidden fallows in Laos using mixed-species planting and biochar application. Silva Fennica 46(1): 39-51.

SFC (Saskatchewan Forage Council). 1998. Seed production of Slender Wheatgrass. 12pp.

SRC (Saskatchewan Research Council). 2014. Gunnar site remediation project, environmental impact statement. SRC publication, Saskatoon, SK. 920 pp.

Verheijen, F., S. Jeffery. A.C. Bastos, M. van der Velde, and I. Diafas. 2009. Biochar Application to Soils - A Critical Scientific Review of Effects on Soil Properties, Processes and Functions. EUR 24099 EN, Office for the Official Publications of the European Communities, Luxembourg, 149pp.

EFFECT OF SOIL AMENDMENTS (MATS AND HYDROSEEDING) ON ESTBALISHMENT SUCCESS OF FOUR NATIVE GRASSLAND SPECIES

M. Phillips, BNRSc Student[1]
W. Gardner, Ph.D., Associate Professor[2]
L. Fraser, Ph.D., Professor[2]
J. Karakatsoulis, Ph.D., Senior Lecturer[2]

[1] Undergraduate Honours Student, Department of Natural Resource Sciences,
Thompson Rivers University, Kamloops, BC, Canada

[2] Committee Members, Department of Natural Resource Sciences,
Thompson Rivers University, Kamloops, BC, Canada

ABSTRACT

Restoration of grassland ecosystems is challenging due to moisture limitation, competition from non-native invasive plants, and difficulty in establishing native grassland species. A manipulative experiment was designed to study the establishment for four native grassland species using combinations of two soil amendments in the lower grasslands of Kamloops, BC. The treatments included the use of a straw germination mat with seed placed either on top or underneath, hydroseeding, a combination of the straw matting with the hydro-seeding either below or on top, seeding without any soil amendment and a control with no seeding or amendment. Treatments were each replicated 6 times in a randomized complete block design. Approximately 150 seeds of each bluebunch wheatgrass (*Pseudoroegneria spicata*), rough fescue (*Festuca campestris*), brown-eyed Susan (*Gaillardia aristata*) and long-leaved daisy (*Erigeron filifolius*) were broadcast seeded per plot. Soil moisture and temperature, and cover of functional groups on each treatment were recorded and statistically analyzed. The highest cover by functional group was undesirables, none of which were seeded species, indicating the presence of plant propagules at this site. It was found that mat treatments, with seeding below, resulted in the greatest cover of seeded species and the most desirable ratio of grass, forbs and undesirable species.

KEY WORDS

Grassland restoration, straw mats, functional groups, invasive species

INTRODUCTION

The goal of ecological restoration is to restore damaged ecosystems to a functioning and sustainable state (Cook and Suski 2008). This can be difficult as there are still barriers to using native seeds (Oliveira *et al*. 2013, Oliveira *et al*. 2012). There is uncertainty that natives will not establish or compete well with agronomic or non-native species and are at higher risk of weed invasion (Oliveira *et al.* 2013; Thompson *et al.* 2006). Disturbance may have changed the nutrient availability, soil structure, and overall growing conditions making the location unsuitable for native vegetation (Bradshaw 1983). Native seeding is also associated with higher costs that may come from having to artificially recreate favorable conditions for native seedlings and increased seeding rates (Bradshaw 1983, Richards *et al*. 1998, Thompson *et al.* 2006). There are also reports of poor success initially, but this could be attributed to the fact we have little research on the performance and germination requirements for native seed (Oliveira *et al*. 2013, Oliveira *et al.*2012, Peterson *et al.* 2004). In many cases there is also a lack of accessibility to native seeds because of limited stockpiling, low productivity, lack of policy and poor political support (Richards *et al.* 1998).

Restoration of grassland ecosystems in western North America presents even larger challenges. Moisture availability is one of the most important challenges to overcome in native grassland restoration (Oliveira *et al.* 2013). Annual precipitation in these regions tends to have high temporal variability and a narrow window in which seedlings can take advantage of moisture, making the growing seasons unpredictable (Bakker *et al.* 2003). As well, non-native invasive species introduced from other continents are of great concern in grassland ecosystems across western North America (DiTomaso 2000). Invasive species flourish in disturbed sites which causes major problems in trying to establish native vegetation when these invasive species are already present (Young and Clements 2005). In general, interspecific competition for soil resources between native and invasive species results in different plant community outcomes which may deviate from the goals of restoration (Bakker *et al.* 2003).

There are different approaches for dealing with moisture limitations and invasive species in grassland restoration. It is common for treatments such as irrigation and fertilizers to be used to support seed germination and growth (Bakker *et al.* 2003, Peterson *et al.* 2004) but these methods also cause increases in undesirable species (Abraham and Corbin 2009). To deal with increased weed species, herbicides can be used. To collectively address these problems many amendments have been developed including mats and hydroseeding. Operationally, mats and hydroseeding have become increasingly popular due to claims of their ability to increase soil moisture and reduce weedy species, among other benefits (Steinfeld *et al.* 2007).

Although there is a move towards using native species in grassland restoration situations, there are challenges to successfully establish these species in arid to semi-arid environments. The current knowledge on methods to increase germination and establishment success of grassland native species is often limited to a few of the more dominant species such as the key grasses. There is also limited information available on the impact of using common operational techniques such as mats and hydroseeding in these dry systems. Although previous studies suggest that these types of soil amendments may increase soil moisture and germination success this has not been well documented with native species in a grassland environment. Therefore this study was designed to address some of these knowledge gaps.

This study utilized two common forms of soil amendments, a straw germination mat and hydroseeding, and four native grassland species. The overall objective of this study was to determine how different restoration treatments impact seeded native species establishment and growth of undesired agronomics and noxious weeds. The specific objectives were focused on how the seven treatments impacted:

(i) soil moisture and temperature; and
(ii) the canopy cover of different functional groups (grasses, forbs and undesirable species).

These results can be applied to many situations where grassland restoration may be necessary, including in mine reclamation, oil sands reclamation, habitat restoration and disturbances from other industries. The data will provide information on functional groups and their response to different site treatments.

METHODS

Study Area

The study area was located within a lower grassland region of Kamloops BC. McLean and Marchland (1968) describes this lower grassland region as a big sagebrush-blue bunch wheatgrass site. These types of sites are characterized as being between 335 m and 610 m in elevation, with 23-25 cm of rain a year with over half falling between April and October (McLean and Marchland 1968). The dominant species found in undisturbed areas are bluebunch wheatgrass (*Pseudoroegneria spicata*), Sandberg's bluegrass (*Poa secunda*), needle-and-thread grass (*Stipa comata*) and big sagebrush (*Artemesia tridentata*) (McLean and Marchland 1968).

The specific location of the experiment was 50° 40' 17'' N 120°22'18'' W at an elevation of 544 m. The site was a fenced compound at Thompson Rivers University. The vegetation surrounding the study site consisted of native and introduced lower grasslands species. Dominant species included bluebunch wheatgrass (*Pseudoroegneria spicata*), big sagebrush (*Artemesia tridentata*), Kentucky bluegrass (*Poa pratensis*), crested wheatgrass (*Agropyron cristatum*), and knapweed (*Centaurea* spp.). On the study site all above ground vegetation was removed using the bucket of a tractor on May 3rd, 2013. The site was then raked and large remnants of plant material were removed. The existing seed bed and some rootstock still remained and no herbicide was applied to the site.

Experimental Design

The experiment was set up as a randomized block design consisting of 7 treatments each replicated 6 times for a total of 42 plots (Table 1). The treatments were control (no seeding), seeding only, hydroseeding, straw matting with seed above the mat, straw matting with seed below the mat, hydroseeding under the straw mat and hydroseeding above the straw mat (Table 1). The design lay out had three plots along the width and 14 plots along the length. Each plot was a half meter square with 20 cm between plots width wise and 50 cm between plots length wise, creating a totally rectangle that was 1.9 m by 13.5 m. Blocking was applied due to a slope gradient and an increase in gravel content in the south plot which may affect soil moisture.

Table 1. The seven treamtents used in the study and the associated codes.

Definition	Code
Control, sand only	Cont
Seeded only	S
Hydroseeded	H-S
Straw matting with seed placed underneath the mat	M-SB
Straw matting with seed placed on top of the mat	M-ST
Hydroseeding underneath a straw mat	HM-SB
Hydroseeding on top of a straw mat	HM-ST

Four species were selected for seeding, 2 native grasses, *Festuca campestris* and *Pseudoroegneria spicata*, and 2 native forbs, *Gaillardia aristata*, and *Erigeron filifolius*. The grasses were selected because they are the dominant native species found in surrounding grasslands. The native forbs were selected based on unpublished data that showing that these two forbs had a relatively high germination success (K. Baethke, personal communications, 2013). *F. campestris, P. spicata* and *G. aristata* seeds were purchased from Purity Feeds, a local seed supplier. *E. filifolius* was hand-picked from a Lac du Bois Park at an elevation of 550 m. Seed packets consisting of approximately 150 seeds for each species, based on weight, were prepared for each seeded plot for a total of 600 seeds per half meter plot. Two tablespoons of sand was added to act as a carrier; the control seed packets contained sand, no seed Broadcast seeding occurred on May 3rd 2013, a windless day, on raked plots. Special care was taken to keep the package contents low to the ground to reduce seed drift.

The straw matting used, *Dewitt Straw Guard 200*, was designated as a seed germination blanket with some erosion control. The mat was cut into 0.5 m^2 squares for the specified plots. The edges were nailed into place to prevent the wind from blowing it away. To determine if placement of seed impacted establishment six replicate treatments had the seed placed on top of the matting and six replicate treatments had the seed placed underneath the matting.

The hydro-mulch used was *EcoFibre Premium Wood Fibre* produced by *Profile Products*. The hydroseeding slurry was mixed in one large batch and separated for each specified plot. The entire hydroseeding slurry contained 97.5 L of water, 3105.5 g of wood fibre, and three tablespoon of tackifier, a chemical binder which was provided with the wood fibre. Three litres of the slurry were taken at a time and the designated seed package was mixed in before it was placed on the plot. Special care was taken to make sure seeds did not settle to the bottom of this container before the slurry was poured onto the plot. The three litre mixture was then smoothed out over the plot using a plastic spatula.

On the day of seeding and for the 2 days directly after, the study site was watered for half an hour with an oscillating lawn sprinkler. After that plots received 2 litres of water twice a week until July 29th with the goal of assisting seed establishment. If soil moisture was above 10% water fraction by volume, the soil was at field holding capacity and no watering was done, which was the case through the month of May.

Precipitation and ambient temperature was recorded at a weather station located at 50°44'20.66"N, 120°25'58.14"W, 691 m in elevation, in another lower grassland region within Kamloops. This weather station was chosen because it was located at the most similar elevation compared to other weather station options in the area. The precipitation, maximum monthly temperature and minimum monthly temperature for 2013 were compared to the long term normal calculated from data from 1979 to 2013. Precipitation in April and May was above the 34 year average, but was then was drastically lower in June, July and August. The average monthly maximum air temperature throughout the growing season was consistently lower than the 34 year average and the average monthly minimum air temperature was higher than the 34 year average.

Data Collection

Soil moisture and temperature were taken twice a week during the time period of May 17 2013 to August 30 2013 using a *POGO Portable Soil Sensor* equipped with a *Stevens Hydra Probe Soil Moisture Sensor* with 6 cm probes used in conjunction with Windows *HydraMon 1.4* software program. The sensor probes were placed in the centre of each plot until the reading became steady, then the values were recorded. This was repeated for each plot starting from the north end of the plots to the south end of the plots. One plot, the sixth hydro-seeded replicate, was found to have too much gravel in the soil which prevented any soil moisture readings through the season.

At the end of the growing season, on September 9 and September 10 2013, canopy cover of all species present was recorded. Percent canopy cover by layer of individual species in the entire 0.5 m^2 plot was recorded. Plants that were rooted outside of the plot but hanging in were not included because their establishment would not have been from the treatment. To reduce bias data collection was done by a single observer.

Plant cover data was organized into three functional groups, grasses (seeded and other native grasses), forbs (seeded and other native), and undesirables (weedy species) (Table 2). Undesirable species consisted of any non-native grasses or forbs. This included noxious weeds and agronomic species.

Table 2. A list of the individual species that made up the three functional groups: grasses, forbs and undesirables. This table includes all species found in the lower grassland study site based on plant data sampling conducted September 9th and 10th, 2014.

Grass	Forb	Undesirable
Festuca campestris	*Erigeron filifolius*	*Kochia scoparia*
Sporobolus cryptandrus	*Gaillardia aristata*	*Centaurea spp.*
	Artemisia tridentata	*Agropyron cristatum*
	Grindelia squarrosa	*Sisymbrium spp.*
		Bromus tectorum
		Linaria vulgaris

Statistical Analysis

Soil moisture and temperature data was pooled by treatment to give an average temperature (degrees Celsius) and moisture content (percent water by fraction) for each date. Values for each treatment were averaged throughout the growing season and plotted to see if treatment impacted these variables. No statistical analysis was conducted on this data.

Statistics were analyzed using *IBM SPSS Statistics 20.* Data were tested against the assumptions of analysis of variance (ANOVA). Raw data did not fit the assumption of normality and was transformed by adding one and taking the log (10) value. Transformed data was deemed as normally distributed based on using a normal Q-Q plot and values for skewdness and kurtosis. The Levene's test was run on the transformed data and indicated that equality of variance was not met. However, the effect of inequality of variance is mitigated when the sample sizes are equal which was the case for our study. Therefore ANOVA was used for all data analysis.

To determine treatment impact on canopy cover a two-way ANOVA was run for each seeded species. If treatment effect was significant post hoc analysis was conducted using a Tukey's test. To determine treatment impact on functional groups a two-way ANOVA was run. If a significant effect was found then a Bonferroni adjustment was used to determine differences between the treatments. This adjustment is recognized as being a conservative post hoc test which allows a lot of confidence to be placed in significant outcomes.

RESULTS

Soil Temperature and Moisture

No differences in soil moisture or temperature were detected between different treatments in the top six centimeters of soil. Soil temperature was increasing and decreasing with changes in ambient temperature.

Canopy Cover of Functional Plant Groups

Blocking had no significant effect on canopy cover of the functional groups, but canopy cover was significantly impacted by both functional group ($p<0.001$) and treatment (Pp0.001). For functional groups the post hoc testing determined all comparisons were significantly different from each other (grass and forb $p = .01$, grass and undesirables $p < .001$, forb and undesirables $p < .001$) (Figure 1).

For treatment the post hoc testing detected a significantly higher total plant cover, of all species, in the mat seeded on top and mat seeded below treatments compared to hydroseeding on top of mat. There were no significant differences of total plant cover between the control and any treatment (Figure 2).

The highest mean cover of undesirable species was found in the control (μ=70.5%) and seeded only (μ=68.7%) treatments but decreased in the mat seeded below (μ=41.7%) and mat seeded on top treatments (μ=43.5%), and was lowest in the hydroseeding on top of mat (μ=26.8%) and hydroseeding below mat (μ=28.8%) treatments (Figure 7). The highest mean of grasses and forbs was found in the mat seeded on top (μ=32.6%, μ=14.1%) and mat seeded below (μ=49.5%, μ=12.8%) treatments. The control, seeded only and hydroseeding treatments had the highest undesirable cover. The hydroseeding on top of mat and hydroseeding below mat plots had the lowest cover of undesirables but also lowered the cover of grasses and forbs. The mat (both seeded on top and seeded below) treatments had the most desirable ratio of grass, forbs and undesirables (Figure 3).

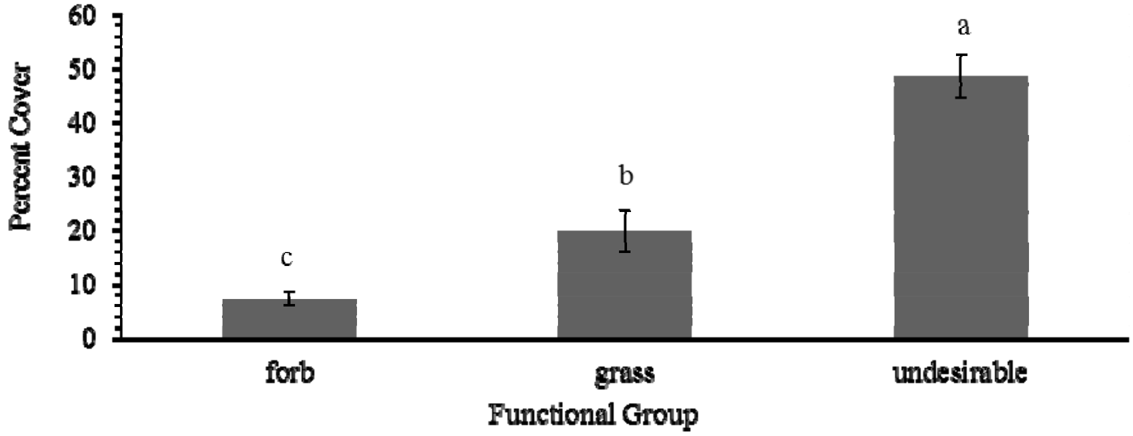


Figure 1. The average canopy covers of each functional group. The data represented as the bars and the error bars show the actual raw data cover data. Means with different letters represent significantly different results with a 0.05 *p* value. Post hoc groupings are based on transformed data.

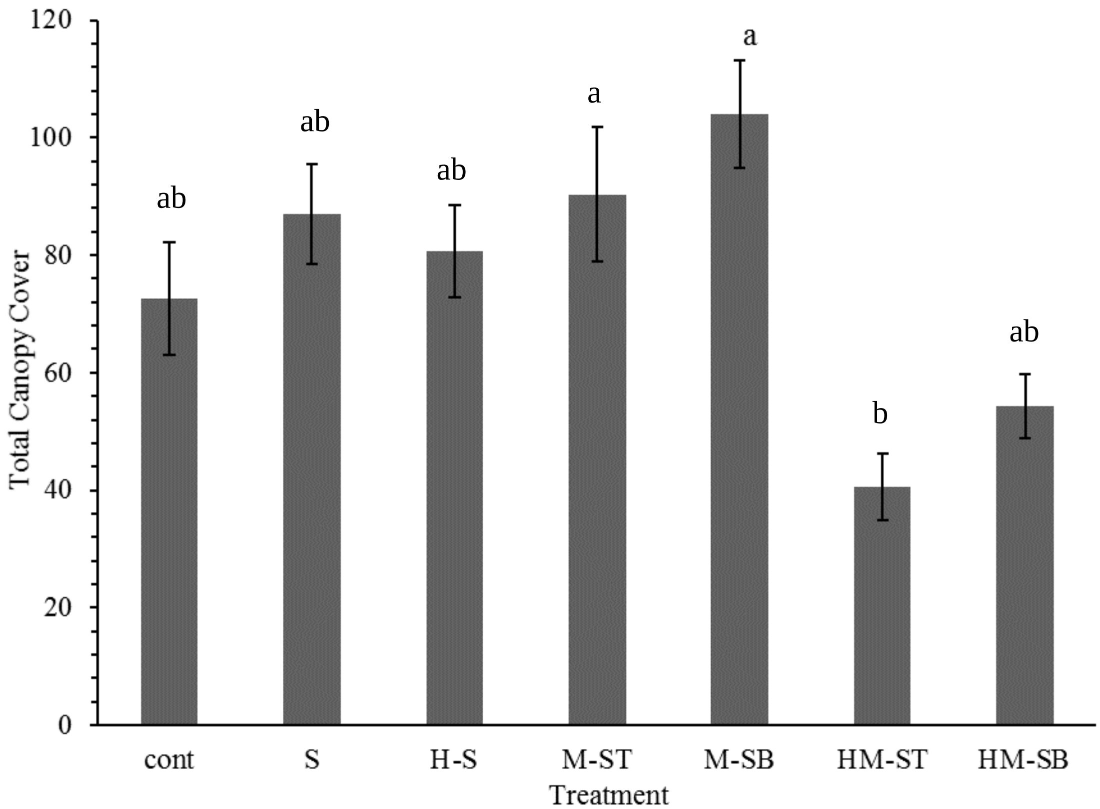


Figure 2. The average total combined cover of all species (all functional groups combined) by treatment type. The data represented as the bars and the error bars show the actual raw data cover data. Means with different letters represent significantly different results with a 0.05 *p* value. Post hoc groupings are based on transformed data.

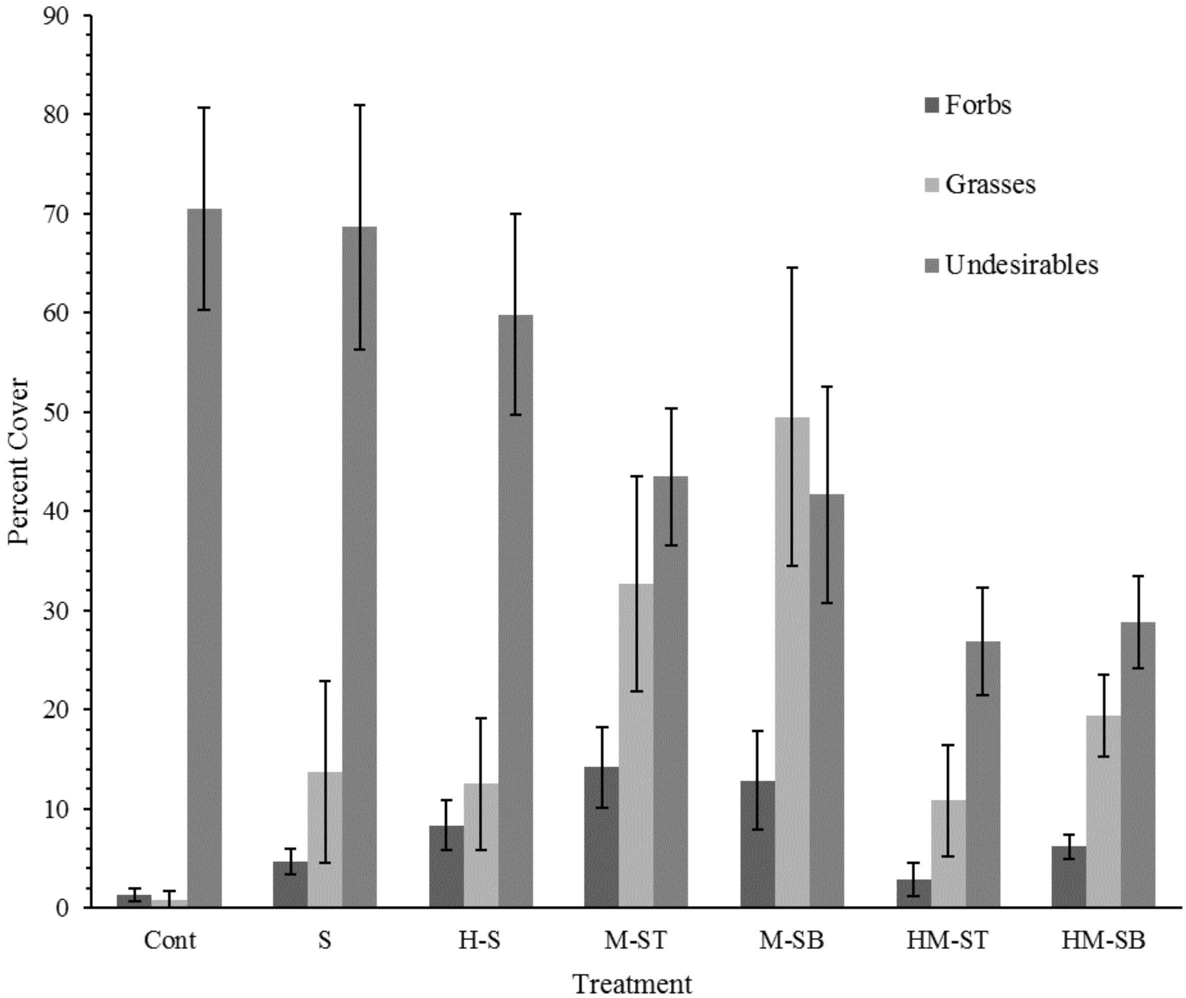


Figure 3. The average percent cover of each functional group, being grasses, forbs and undesirables, by treatment. The data represented as the bars and the error bars show the actual raw data cover data.

DISCUSSION

Soil Temperature and Moisture

The results show that treatments did not impact soil moisture and temperature. This outcome was most likely due to the methodology used to determine these variables. The *POGO Portable Soil Sensor* used to measure soil moisture and temperature had a 6 cm deep probe. The probe tracked soil moisture a few centimeters under the soil surface but was not able to capture differences on the surface of the soil. Other experiments have been able to detect differences in soil moisture using the gravimetric method and other types of soil probes for similar depths (Deutsch *et* al. 2010, Wicks *et al.* 1994). During the experiment, differences in moisture were observed, particularly if rain events had occurred within a few days prior. Matting amendments appeared to have more moisture present between the soil surface and the straw matting, while the top of the straw was dry. The hydroseeding slurry was also observed to be moist on some of these occasions as well. These indications of moisture were observed when the control and seeded only treatments were dry on the surface. The observations of the site are more consistent with the literature in terms of soil moisture than what was actually recorded (Deutsch *et al*. 2010, Wicks *et al.* 1994). Using different instrumentation may have captured these differences more accurately, for example the use of iButton data loggers on the soil surface as they can continuously log temperature and humidity.

<u>Canopy Cover of Functional Groups</u>

Our results show that undesirable species had a significantly higher cover than forbs and grasses. This may have been a result of the site location and the type of disturbance that was used to create this experiment. While the top few centimeters of soil were removed to create this experiment, the seed bank and roots in the soil were not eradicated and no herbicides were used to eradicate any live plant material present. The plant material and seed bank that would most likely be present would be similar to the vegetation surrounding the experiment. This mostly included *A. cristatum*, *Centaurea* spp., and *Sisymbrium* spp. which are all grouped in the undesirable functional group. Therefore, in this case, regardless of treatment these species were already going to be well established on the site and seeded species were going to have to compete. These conditions likely hindered the full ability of seeded species (Skousen and Venable 2008).

Regardless of species or functional group, treatments had statistically similar plant cover, ranging from 45% to 105% cover. The control had 73% cover and was not significantly different from any of the other treatments. This is because there was already a seed bank and live root mass on this recently disturbed site. This may not be the case on sites where there is a longer term disturbance and a limited seedbank or source of propagules are available. Hydroseeding on top of the mat resulted in significantly lower cover when compared to the mat. This indicates that the mat created the most favorable conditions, resulting in the highest mean cover, while the combination of hydroseeding on top of the mat created the least favorable conditions, likely smothering any vegetation that did start to grow beneath the amendment. Hydroseeding below the mat had the second lowest mean, but was not significantly different from the other treatments.

Even though total cover was similar across species the amount of forbs, grasses and undesirable species cover changed between treatments. The matted treatment resulted in the most desirable outcome having high levels of forb and grass cover (μ= 62%) while lowering undesirable cover to 40-50% versus the 70% found in the control and 68% in the seeded only. This is much higher for first year establishment seen in other seeding only studies (Skousen and Venable 2007). The straw mat may have reduced evaporation, increased water infiltration by preventing soil surface sealing due to rain drop splash, and reduced soil surface temperatures which over all created better growing conditions for seeded species (Cole 2007, Roberts and Bradshaw 1985). These components may have created the most favorable microsite for germination that was not captured in the temperature and moisture recordings (Peterson *et al* 2004, Simmons *et al.* 2011). The mat with seeding below had over three times the cover of grass and forbs compared to seeding alone and hydroseeding treatments. Mat treatments have previously been shown to significantly improve survivorship (Mallik and Karmin 2008). The decrease in the cover of undesirables in the mat treatments may have been due to increased competition from the seeded species. In many other cases straw mulch decreased weed species significantly, but this may be partially due to smothering from thick applications (Anzalone *et al.* 2010, Wicks *et al.*1994). In our study the straw matting created a relatively thin single layer with many openings to the soil surface. In a practical application of using mats some form of an herbicide would likely be used that would have reduced the cover of desirables further (Wicks *et al.* 1994).

The seeded only treatments had a very high cover of undesirables and lower cover of grasses and forbs. The undesirable species would have been quicker taking advantage of the conditions through earlier emergence (Oliveira *et al*. 2013). They would have also benefited from the springtime watering, but native species where still able to compete, reaching a mean cover of 26%. Undesirable species such as *A. cristatum* are quick to grow in the springtime to use available soil resources, restricting their use by desirable species (Oliveira *et al*. 2013). Other species, like *Centaura* spp., may have used their deep rooting to reach soil resources as a competitive advantage (Oliveira *et al*. 2013). The restricted growth of the grasses and forbs in the seeding and hydroseeding treatments is most likely due to competition with undesirables (Oliveira *et al*. 2013). There are many instances recorded in the literature that show that the weed species, grouped in this study as undesirables, benefit and compete better in disturbed locations and with watering (Banerjee *et al*. 2006).

The hydroseeding treatment had very similar results in its proportions of functional groups as seeding alone. Previous concerns that hydroseeding may inhibit seed establishment do not appear to be occurring in this case. However, without the watering in the spring there might not be the amount of cover of native species on this treatment (Andrés and Jorba 2000).When the hydroseeding and matting was combined the result was a low cover of all functional groups. The combination of the treatments likely acted as a barrier to all plants, regardless of functional group. The thickness and hardness of the combined treatment may have created an impermeable barrier.

The control had the highest mean of undesirables and the lowest amount of grasses and forbs, as would be expected. The reason we found an occurrence of the grass functional groups is because one of the control replicates had one occurrence of *Sporobolus cryptandrus* and one occurrence of *F. campestris*. The forbs that occurred on the control included one occurrence of each seeded forb and two occurrences of A*rtemisia tridentata.* The occurrence of the seeded species was likely due to drift off of plots from wind or water or human mishap during the time of seeding.

In general there was a decreased trend of cover of undesirables from the control to the mat. This trend reflects first the addition of the seeded species (seeded and hydroseeded) and then the increase in favorable conditions for seeded species (mat treatments) along with increased competition for the undesirables (Cole 2007, Mallik and Karmin 2008, Peterson *et al.* 2004, Roberts and Bradshaw 1985). Overall the use of mats provided the best ratio of desirable grasses and forbs to undesirable species while hydroseeding was similar to seeding alone.

<u>Study Limitations</u>

This study was short term, capturing only one growing season. Natives are often slow to emerge and other studies have found large increases in the second year (Oliveira *et al.* 2013). Seeding was also done in the spring and the following year may give seeded species an opportunity to take advantage of early spring moisture and snow melt resulting. Also the blocking effect with low replication may have made it more difficult to detect significant differences, specifically in the case of species with low cover. The use of herbicide before seeding may have helped to reduce competition and resulted in better establishment of the seeded species. Finally, the use of sensors that can capture temperature and moisture in smaller depth increments and more continuously over time would be beneficial to understanding the relationship between these different treatments and microclimate.

CONCLUSION

This study has added to the limited knowledge of native grasslands species and their ability to establish with the use of soil amendments. The results indicate that hydroseeding was similar to seeding alone, making this a viable option for use in grassland restoration where erosion control may be an issue, or over large areas at risk of wind and seed predation. However the use of straw matting resulted in the most favorable outcome, increasing the establishment of native seeded species and also decreasing the cover of undesirables. These results suggest that the use of matting is helping to address some of the site limitations. This is most likely because moisture is a limiting factor in these environments and mat treatments alter the moisture stress, although this study was not able to capture changes in soil moisture.

REFERENCES

Abraham J.K., Corbin J.D. 2009. California Native and Exoctic Perennial Grasses Differ in their Response to Soil Nitrogen, Exotic Annual Grass Density, and Order of Emergence. Plant Ecology. Vol. 201 (2). Pp. 445-456

Andrés P., Jorba M. 2000. Mitigation Strategies in Some Motorway Embankments (Catalonia, Spain). 2000. Resoration Ecology. Vol 8 (3). Pp. 268-275

Anzalone A., Cirujeda A., Aibar J., PardonG., Zaragoza C. 2010. Effect of Biodegradable Mulch Materials on Weed Control in Processing Tomatoes. Weed Technology. Vol 34(3). Pp. 369-377

Bakker J.D., Wilson S.D., Christian J.M, Li X., Ambrose L.G., Waddington J. 2003. Contingency of Grassland Restoration on Year, Site, Competition from Introduced Grasses. Ecological Applications. Vol. 13 (1). Pp. 137-153

Banerjee M.J., Gerhart V.J., Glenn E.P.2006. Native Plant Regeneration on Abandoned Desert Farmland: Effects of Irrigation, Soil Preparation, and Amendments on Seedling Establishment. Restoration Ecology. Vol 14(3). Pp. 339-348

Bradshaw A.D. 1983. The Reconstruction of Ecosystems: Presidential Address to the British Ecological Society. Journal of Applied Ecology. Vol 20 (1). Pp. 1-17

Cole D.N. 2007. Seedling Establishment and Survival on Restored Campsites in Subalpine Forest. Restoration Ecology. Vol.15 (3). Pp. 430-439

Cooke S.J. Suski C.D. 2008. Ecological Restoration and Physiology: An Overdue Integration. BioScience. Vol. 58(10). Pp. 957-968

Deutsch E.S., Bork E.W., Willms W.D. 2010. Separation of grassland litter and ecosite influences on seasonal soil moisture and plant growth dynamics. Plant Ecology. Vol. 209(1). Pp. 135-145

DiTomaso J.M. 2000. Invasive Weeds in Rangelands: Species, Impacts, and Management. Weed Science. Vol. 48(2). Pp. 255-265

Mallik A.U., Karmin M.N. 2008. Roadside Revegetation with Native Plants: Experimental Seeding and Transplanting of Stem Cuttings. Applied Vegetation Sciences. Vol. 11(4). Pp. 547-554

McLean A., Marchland L. 1968. Grassland Ranges in the Southern Interior of British Columbia. Canada Department of Agriculture. Kamloops BC. Pp. 3-7

Oliveira G., Clemete A., Nunes A., Correia O. 2013. Limitations to recruitment of Native Species in Hydroseeding Mixtures. Ecological Engineering. Vol. 57. Pp. 18-26

Oliveira G., Nunes A., Clemente A., Correia O. 2012. Testing germination of Species for Hydroseeding Degraded Mediterranean Areas. Restoration Ecology. Vol 20(5). Pp. 623-630

Peterson S.L., Roundy B.A., Bryant R.M. 2004. Revegetation Methods for High Elevation Roadsides at Bryce Canyon National Park, Utah. Restoration Ecology. Vol. 12(2). Pp. 248-257

Richards R.T., Chambers J.C. Ross C. 1998. Journal of Range Management. Vol 51(6). Pp. 625-632

Roberts R.D. and Bradshaw A.D. 1985. The Development of Hydraulic Seeding Technique for Unstable Sand Slopes. II. Field Evaluation. Journal of Applied Ecology. Vol 22 (3). Pp. 979-994

Simmons M.E., Wu X.B., Whisenant S.G. 2011. Plant and Soil Responses to Created Microtopography and Soil Treatments in Bottomland Hardwood Forest Restoration. Restoration Ecology. Vol 19(1). Pp. 139-146

Skousen J.G., Venable C.L. 2008. Establishing Native Plants on Newly-Constructed and Older-Reclaimed Sites along West Virginia Highways. Land Degradation & Development. Vol. 19. Pp. 388-396

Steinfeld D.E., Riley S.A., Wilkinson K.M., Landis T.D., Riley L. E. 2007. Roadside Revegetation: An Integrated Approach to Establishing Native Plants. Vancouver (WA): Rederal Highway Administration, Western Federal Lands Division. Pp. 301-326

Thompson T.W., Roundy B.A., McArthur E.D., Jessop B.D., Waldron B., Davis J.N. 2006. Fire Rehabilitation Using Native and Introduced Species: a Landscape Trial. Rangeland Ecology and Management. Vol 59 (3). Pp. 237-248

Wicks G.A., Crutchfield D.A., Burnside O.C. 1994. Influence of Wheat (*Triticum aestivum*) Straw Mulch and MEtolachlor on Corn (*Zea mays*). Weed Science. Vol 42(1). Pp. 141-147

Young J.A., Clements C.D. 2005. Exotic and Invasive Herbaceous Range Weeds. Rangelands. Vol 27(5). Pp.10-16