REVIEW OF PASSIVE SYSTEMS FOR TREATMENT OF ACID MINE DRAINAGE

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REVIEW OF PASSIVE SYSTEMS FOR TREATMENT OF ACID MINE DRAINAGE

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EXECUTIVE SUMMARY

One of the major issues facing the Canadian mining industry is the treatment of effluent during and after closure of a mining property. Effluent treatment may be complicated by the presence of acid mine drainage (AMD) which even under the best reclamation scenario may require long term collection and treatment. While chemical treatment or some other form of active effluent treatment has traditionally been conducted in Canada, greater consideration has been given recently to forms of passive treatment.

Passive treatment systems utilize the chemical, biological and physical removal processes that often occur naturally in the environment to modify the influent characteristics. Passive treatment systems were initially considered attractive to treat acid mine drainage due to their lower costs of construction, operation and maintenance, and their ability to operate at remote locations with limited operational requirements. The objective of this project was to review passive treatment systems, and make recommendations on their applicability to treat acid mine drainage in Canada.

Although passive systems have been proven at many locations around the world, the Canadian climate and aquatic environments present considerable challenges to large-scale use in Canada. Biologically driven systems have low activity in cold temperatures and drought, while storm and spring "freshet" events demand flexible, strong systems.

In this review, four major types of passive technologies for the treatment of acid mine drainage have been examined:

- anoxic limestone drains;
- constructed wetlands;
- microbial reactor systems; and,
- biosorption systems.

In accordance with the scope of work, this document provides:

- summary of known passive treatment technologies;
- maintenance and monitoring requirements;
- life expectancy and long term implications (>100 years);
- implications of treatment product disposal;
- estimate of costs for the technologies described based on generic cases;
- ability to meet Canadian Metal Mining Liquid Effluent Regulations, and to control toxicity;
- descriptions of case studies including: range of flow, temperature, water chemistry, and identification of limiting conditions where available in the literature (Appendix A); and,
- general assessment of the applicability of current passive treatment systems to Canadian mine sites.

ANOXIC LIMESTONE DRAINS (ALD'S)

The basic design of an ALD is a trench filled with high quality crushed limestone, sealed under plastic and geotechnical fabric, covered by soil, through which an unaerated, contaminated effluent stream flows by gravity. As it flows through the system, the acid mine drainage gradually dissolves the limestone, releasing calcium as bicarbonate, thus raising the pH.

Based primarily on studies conducted at coal mines in the United States, ALD's have been shown to be most effective for influent with dissolved oxygen, ferric iron (Fe^{3+}) and aluminum concentrations of less than 1 mg/L, and sulphate concentrations below 2,000 mg/L. At higher concentrations the limestone may become armoured with oxides or gypsum, reducing the rate of limestone dissolution or plugging the system. In either instance, the ability of the ALD to generate alkalinity may be significantly reduced, and failure of the system may occur.

As a result of these strict influent requirements, ALD's are expected to have only limited application to treatment of acid mine drainage at Canadian metal mines.

CONSTRUCTED WETLANDS

Constructed wetlands are ecological systems designed to optimize a variety of natural physical, chemical, microbial and plant-mediated processes. In a constructed wetland, influent AMD drains by gravity through the wetland, progressively undergoing metal removal and neutralization. Metals are removed by precipitation, chelation and exchange reactions, while neutralization is primarily achieved by the activity of sulphate reducing bacteria (SRB), or the increase in alkalinity from the chemical and microbial reactions including limestone dissolution.

Passive systems for AMD treatment have commonly used combinations of natural or constructed wetlands, *Sphagnum* peat and open ponds, supplemented by chemical amendments (mostly limestone) and organic substrate to increase alkalinity and reduce acidity. Sequential treatment of AMD to remove iron by oxidation, hydrolysis and settling in the aerobic stage, followed by SRB activity in an anaerobic stage to raise pH, is an effective combination.

For either aerobic or anaerobic cells, the design must maximize contact with the matrix, which can either be aerated water, or anaerobic substrate. It is essential that constructed wetlands are managed in terms of their individual components and their mutual interactions to gain a desired overall efficiency. Many of the metal removing mechanisms in a wetland are temporary and reversible, and can reach saturation; thereby reducing the wetland's efficiency and decreasing their cost effectiveness. In addition, para-reversibility represents a challenge for treatment product monitoring or disposal.

Constructed wetlands have the potential to address AMD treatment at some Canadian sites where sufficient surface area is available, and can form the preferred alternative in terms of costs, efficiency and environmental safety. The 'black box' design approach that has been used in the past and is still being suggested, is not recommended. The design should be based on an understanding of the interactions between the chemical, microbial and plant-mediated components of the system and the engineering, climate and hydrogeological realities of the treatment site.

A well designed, constructed wetland is an efficient accumulator of metals and reaction products. Key to its efficiency, is the continuous physical, chemical and biotic matrix in the wetland. This capacity will be limited during freezing or high flow conditions. As a result, constructed wetlands may be most applicable to Canadian mines having shorter and milder winters, and at sites where a constant rate of flow can be maintained. An alternative treatment method may be required during the winter and during spring runoff conditions, or else large retention ponds are required.

MICROBIAL REACTOR SYSTEMS

Microbial reactor systems or bioreactors may be in an open or closed configuration, referring to whether they are exposed to the atmosphere. In either instance, the microbial reactor shell contains a biodegradable substrate (usually agricultural products such as mushroom compost or straw) which supports the growth of micro-organisms, which in turn treat acid mine drainage.

The cellulose of the agricultural products is degraded by cellulolytic bacteria to generate free sugars and other metabolites, which are further metabolized to provide substrate for the fermentative anaerobes. Under anaerobic conditions these free sugars are fermented to short chain organic acids or short chain fatty acids, which are suitable substrates to support the growth of sulphate reducing bacteria (SRB). SRB reduce sulphate to hydrogen sulphide which precipitates metal ions as low solubility metal sulphides. Concurrently, the sulphate reducing bacteria consume hydrogen ions and produce carbon dioxide during their metabolism, causing an increase in the pH of the solution due to reduced concentration of free hydrogen ions and the buffering effect of the CO_2/bicarbonate buffer system.

Pilot plant data available suggests that bioreactors are a feasible technology for treatment of small AMD streams. Open bioreactors are expected to only be applicable at Canadian mines with mild or moderate winters. Closed bioreactors can operate anywhere that a fairly constant temperature can be maintained.

BIOSORPTION SYSTEMS

Micro-organisms, including bacteria, algae, fungi and yeasts, can efficiently accumulate heavy metals and radionuclides from their external environment. Biosorption systems in a wide variety of configurations rely on this ability to treat acidic drainage. Living cells can be used to treat effluent where metal concentrations are below toxic levels. The use of dead biomass in the form of commercial biosorbents eliminates the problems of metal toxicity, adverse climatic conditions, and the costs associated with nutrient supply and culture maintenance.

Only a limited number of studies have been conducted to date in regards to treating AMD with biosorption systems. While it does not appear that biosorption systems are an effective stand-alone treatment system for AMD, with further study they may become an alternative form of treatment of parts of an effluent stream, or as a final polishing step. The success of biosorption systems using living biomass during the winter is expected to be limited. Treatment efficiency will be lowered under poor growth conditions. Systems employing dead biomass are expected to have greater applicability, and may not be compromised by winter conditions as long as flow is maintained.

OVERVIEW

Passive treatment of acid mine drainage has a future in Canada, but is limited to applications where:

- flows are of relatively constant volume
- water temperature is greater than 7EC (eg. mine water or embankment seepage)
- water chemistry of low to medium strength acidity and metal concentration
- low concentrations of aluminum and iron
- low sensitivity of the receiving environment to upsets in the passive treatment system.

Further research and field experience is needed to more precisely specify passive treatments for AMD in Canada to ensure that metal mining liquid effluent regulations are met all the time. No "passive treatment" system is truly passive. All systems require monitoring and replacement of consumed alkalinity or organic-based nutrients for bacteria. Also, metal precipitates need to be removed and in some jurisdictions sludge falls under "hazardous waste" regulations and disposal may be a significant challenge.

SUMMARY OF PASSIVE TREATMENT TECHNOLOGIES FOR TREATMENT OF ACID MINE DRAINAGE

Technology	General Comments	Applicability / Limitations	Costs
Anoxic Limestone Drains	• low cost form of passive alkalinity addition • most research relates to AMD from US coal mines	• able to operate year round as long as flow continues • strict influent quality limits to ensure long-term alkalinity generation • expected to have limited application to Canada	• virtually no maintenance or other operating costs • capital costs in the range of $4,000 for a flow of 7.5 L/min and $22,500 for a flow of 125 L/min
Constructed Wetlands	• designed to optimize processes that occur in a natural wetland • internal cells focus on either aerobic or anaerobic processes • must be designed to ensure that the processes are optimized and do not counteract each other	• expected to have the greatest application to mines with moderate winters having secondary active treatment for the winter and spring runoff periods • ability to treat effluent during cold, harsh winters is unknown	• annual operating costs estimated at 10% to 20% of capital cost • generic wetland to treat 60 L/min is estimated at $85,000 • literature suggests capital costs of US $5 to US $32/m^2 are typical
Microbial Reactor Systems	• relies on microbial reactions, supported by a biodegradable carbon source • only short term studies have been completed to date, primarily under very low flow conditions (<1 L/min)	• limiting factor is rate of biodegradation of carbon source • unknown ability to treat moderate or high flows • may be applicable to small effluent streams and could complement other passive or active treatment systems	• operating are not significant; new substrate required semi-annually • capital costs for an open reactor (50 to 60 L/min) are estimated at near $33,500; a closed reactor system to treat 75 to 100 L/min, $56,000
Biosorption Systems	• metals are removed from solution by adsorption/absorption to living cells or non-living biomass • few field studies are available; most research is bench scale only	• may have application as a secondary form of treatment when integrated with another treatment system • success of systems employing living cells during the winter is doubtful	• insufficient information is available to assess at a full-scale level

SOMMAIRE

Un des principaux problèmes que l’industrie minière du Canada doit solutionner est le traitement des effluents durant et après la fermeture d’une propriété minière. Le traitement des effluents peut être compliqué par la présence et la collecte des eaux de drainage minier acide (DMA) qui, même dans le meilleur scénario de réhabilitation, peuvent nécessiter un traitement à long terme. Bien que le traitement classique des effluents au Canada ait été un traitement chimique ou une autre forme de traitement actif, on a récemment envisagé davantage des formes de traitement passif.

Les systèmes de traitement passif font intervenir des processus d’élimination chimique, biologique et physique qui existent souvent à l’état naturel dans l’environnement et modifient les propriétés de l’influent. Les systèmes de traitement passif ont d’abord été jugés attrayants pour le traitement des eaux de drainage minier acide à cause de leurs faibles coûts de construction, d’exploitation et d’entretien ainsi que de la possibilité de les mettre en oeuvre à des endroits éloignés avec des besoins opérationnels limités. L’objectif de ce projet était une revue des systèmes de traitement passif et la formulation de recommandations quant à leur applicabilité au traitement du DMA au Canada.

Bien que des systèmes passifs aient été éprouvés à bien des endroits dans le monde, le climat et l’environnement aquatique canadiens posent des défis considérables à leur utilisation à grande échelle au Canada. Les systèmes biologiques ont une faible activité par temps froid et en période de sécheresse, alors que les tempêtes et la crue printanière demandent des systèmes puissants et souples.

Dans cette revue, quatre principaux types de technologies passive de traitement des eaux de drainage minier acide ont été étudiées :

- les drains de calcaire anoxiques;
- les marécages aménagés;
- les réacteurs microbiens; et
- les biosorbeurs.

Conformément à la portée des travaux, ce document présente :

- un sommaire des technologies de traitement passif connues;
- les besoins d’entretien et de suivi;
- la durée de vie prévue et répercussions à long terme (> 100 ans);
- contraintes pour la déposition du produit de traitement;
- l’estimation des coûts des technologies décrites d’après des cas génériques;
- la capacité de respecter le Règlement canadien sur les effluents liquides des mines de métaux et d’en limiter la toxicité;
- la descriptions d’études de cas, entre autres : plage de débit, température, chimie de l’eau et identification des facteurs de contraintes décrits dans la littérature (appendice A); et
- l’évaluation générale de l’applicabilité des systèmes de traitement passif existants aux sites miniers du Canada.

DRAINS DE CALCAIRE ANOXIQUES (DCA)

Les DCA sont, fondamentalement, des tranchées remplies de calcaire concassé de grande qualité, isolées sous un plastique et un géotextile et recouvertes de terre, dans lesquelles un courant d'effluent contaminé non aéré s'écoule par gravité. Au fur et à mesure de leur écoulement dans le système, les eaux de drainage minier acide dissous le calcaire avec dégagement de calcium sous forme de bicarbonate, donc avec augmentation du pH.

D'après les résultats d'études réalisées dans des mines de charbon des États-Unis, les DCA se sont avérés plus efficaces dans le cas d'influent contenant des concentrations d'oxygène dissous, de fer ferrique (Fe^{3+}) et d'aluminium inférieures à 1 mg/L, et des concentrations de sulfates inférieures à 2 000 mg/L. À des concentrations plus élevées, le calcaire peut se revêtir d'oxydes ou de gypse, ce qui ralentit la dissolution du calcaire ou colmate le système. Dans les deux cas, la capacité d'alcalinisation du DCA peut être réduite de beaucoup et il peut y avoir défaillance du système.

Vu les spécifications strictes relative à l'influent, les DCA ne devraient avoir qu'une application limitée au traitement des eaux de drainage minier acide dans les mines de métal canadiennes.

MARÉCAGES AMÉNAGÉS

Les marécages aménagés sont des systèmes écologiques conçus pour optimiser divers processus naturels tant physiques, que chimiques, microbiens ou à médiation végétale. Dans le système construit, l'influent de DMA s'écoule par gravité dans le marécage et l'élimination du métal et la neutralisation sont progressives. Les métaux sont éliminés par précipitation, chélation et échanges, alors que la neutralisation est surtout le résultat de l'action des bactéries sulfatoréductrices (BSR), ou de l'augmentation de l'alcalinité due aux réactions chimiques et à l'action microbienne, y compris la dissolution du calcaire.

Les systèmes passifs de traitement des eaux de DMA sont habituellement une combinaison de marécages naturels ou construits, de mousse de sphaigne et d'étangs ouverts avec des amendements chimiques (surtout calcaires) et des substrats organiques pour augmenter l'alcalinité et réduire l'acidité. Une combinaison efficace est le traitement séquentiel des eaux de DMA pour enlever le fer par oxydation, hydrolyse et décantation, au stade aérobie, et l'action des BSR au stade anaérobie pour élever le pH.

La conception des cellules aérobies ou anaérobies doit maximiser le contact avec la matrice, qui peut être de l'eau aérée ou un substrat anaérobie. Il est essentiel que les marécages construits soient gérés en fonction de leurs composantes individuelles et de leurs interactions mutuelles pour qu'ils puissent avoir l'efficacité globale souhaité. Beaucoup des mécanismes d'élimination des métaux dans les marécages sont temporaires et réversibles; ils peuvent devenir saturés, ce qui réduit l'efficacité des marécages et diminue leur rentabilité. De plus, la réversibilité pose un défi pour le suivi et la déposition des produits de traitement.

Les marécages aménagés ayant une superficie suffisante ont le potentiel nécessaire au traitement des eaux de DMA de certains sites canadiens; ils peuvent alors constituer la solution de choix quant au coût, à l'efficacité et à la sécurité pour l'environnement. La conception selon l'approche de la ¨boîte noire¨, qui a été utilisée par le passé et qui est toujours proposée, n'est pas recommandée. La conception devrait être basée sur la compréhension des interactions entre les composantes du système - chimique, microbienne et à médiation végétale - ainsi que les condition d'ingénierie, de climat et d'hydrogéologie du site de traitement.

Un marécage aménagé bien conçu est un accumulateur de métaux et de produits de réaction efficace. La clé de son efficacité est la matrice physique, chimique et biotique continue de le marécage. Cette capacité sera réduite en période de gel et de crue. Les marécages aménagés peuvent donc être le plus applicable aux mines canadiennes où l'hiver est le plus court et le plus doux et aux sites où un débit constant peut être maintenu. Une autre méthode de traitement peut être requise durant l'hiver et la période de ruissellement printanier, sinon il faut de grands bassins de rétention.

RÉACTEURS MICROBIENS

La configuration des réacteurs microbiens ou bioréacteurs peut être ouverte ou fermée selon qu'ils sont ou non exposés à l'atmosphère. Dans les deux cas, la coque des réacteurs microbiens contient un substrat biodégradable (habituellement des produits agricoles comme le compost de champignon ou la paille) qui favorise la croissance des microorganismes effectuant le traitement des eaux de drainage minier acide.

La cellulose des produits agricoles est dégradée par les bactéries cellulolytiques, ce qui produit des sucres libres et d'autres métabolites, qui sont métabolisés plus à fond et donnent des substrats pour les agents fermentatifs anaérobies. En conditions anaérobie ces sucres libres sont transformés par fermentation en acides organiques à courte chaîne ou en acides gras, qui sont des substrats favorables à la croissance des bactéries sulfatoréductrices (BSR). Les BSR réduisent les sulfates en sulfure d'hydrogène, qui fait précipiter les ions métalliques en les transformant en sulfures peu solubles. En même temps les bactéries sulfatoréductrices consomment les ions hydrogène et produisent du dioxyde de carbone durant leur métabolisme, d'où une augmentation du pH de la solution par suite d'une réduction de la concentration en ions hydrogène et de l'effet tampon du système CO_2/bicarbonate.

Les données produites en usine pilote semblent indiquer que les bioréacteurs sont une technologie applicable au traitement des petits débits de DMA. Les bioréacteurs ouverts ne devraient être applicables qu'à certaines mines canadiennes où l'hiver est doux ou modéré. Les bioréacteurs fermés sont exploitables partout où une température assez constante peut être maintenue.

BIOSORBEURS

Les microorganismes - entre autres les bactéries, les algues, les champignons et les levures - concentrent efficacement les métaux lourds et les radionucléides présents dans leur environnement. Les biosorbeurs d'une grande variété de configurations font appel à cette capacité de traiter les eaux de drainage acide. Les cellules vivantes peuvent servir à traiter les effluents dont la teneur en métal est inférieure aux teneurs toxiques. L'utilisation de biomasse morte sous forme de biosorbants commerciaux élimine les problèmes de la toxicité des métaux, des conditions climatiques difficiles et des coûts de l'alimentation en nutriments et de l'entretien des cultures.

Seulement un petit nombre d'études ont été réalisées jusqu'ici sur le traitement des eaux de DMA avec des biosorbeurs. Bien qu'il ne semble pas que les biosorbeurs soient un système de traitement primaire efficace pour les eaux de DMA, il est possible que des études plus poussées en fassent une forme de traitement optionnel pour une partie d'un circuit d'effluent ou une étape de polissage. Le succès des biosorbeurs à biomasse vivante devrait être limité en hiver. L'efficacité du traitement sera moindre dans de mauvaises conditions de croissance. Les systèmes qui utilisent la biomasse morte devraient avoir une plus grande applicabilité qui pourrait ne pas être compromise par les conditions hivernales tant que le débit est maintenu.

APERÇU

Le traitement passif des eaux de drainage minier acide a de l'avenir au Canada, mais ses applications sont limitées aux cas où :

- les débits ont un volume relativement constant;
- la température de l’eau est supérieures à 7 EC (p. ex., eau de mine ou écoulement des digues);
- acidité de l’eau et concentration de métal dans l’eau faibles ou moyennes;
- faibles concentrations d’aluminium et de fer;
- faible sensibilité de l’environnement récepteur aux bouleversements dans le système de traitement passif.

Il faudra davantage de recherches et d’expériences sur le terrain pour définir plus précisément les systèmes de traitement passif des eaux de DMA au Canada afin que le Règlement sur les effluents liquides des mines de métaux soit respecté en totalité. Aucun système de "traitement passif" n’est vraiment passif. Tous les systèmes nécessitent un suivi et le remplacement de l’alcalinité consommée ou des nutriments organiques pour les bactéries . En outre, les précipités métalliques doivent être enlevés et, dans certaines juridictions les boues sont couvertes par le règlement sur les "déchets dangereux" et leur déposition peut constituer un défi important.

SOMMAIRE DES TECHNOLOGIES DE TRAITEMENT PASSIF APPLICABLES AUX EAUX DE DRAINAGE MINIER ACIDE

Technologie	**Commentaires généraux**	**Applicabilité / limitations**	**Coûts**
Drains de calcaire anoxiques	• façon passive et peu coûteuse d’augmenter l’alcalinité • la plupart des recherches porte sur les eaux de DMA de mines de charbon américaines	• capacité de fonctionner à longueur d’année à la condition que le débit soit continu • limites strictes sur la qualité de l’influent pour assurer la génération d’alcalinité à long terme • application limitée prévue au Canada	• coûts d’entretien et autres coûts d’exploitation à peu près nuls • investissements de l’ordre de 4 000 $ pour un débit de 7,5 L/min et de 22 500 $ pour un débit de 125 L/min
Marécages aménagés	• conçues pour optimiser les processus naturels des marécages aménagés • les cellules internes font appel à des processus aérobies ou anaérobies • la conception doit assurer que les processus sont optimisés et sans interaction négative	• devrait avoir sa plus grande application, là où l’hiver est modéré, dans les mines procédant à un traitement secondaire pendant l’hiver et les périodes de ruissellement printanier • la capacité de traiter l’effluent durant les hivers durs et froids est inconnue	• les coûts annuels d’exploitation sont estimés à de 10 % à 20 % des investissements • en général, le coût du marécage aménagé nécessaire au traitement de 60 L/min est estimé à 85 000 $ • la littérature semble indiquer des investissements typiques de 5 à 32 $(É.-U)/m^2
Réacteurs microbiens	• fait intervenir des réactions microbiennes, soutenues par une source de carbone biodégradable • seules des études à court terme ont été complétées jusqu’ici, principalement à un très faible débit (<1 L/min)	• le facteur limitatif est la vitesse de biodégradation de la source de carbone • capacité inconnue de traitement de débits moyens ou élevés • peut être applicable à de petits débits d’effluent et pourrait suppléer à d’autres systèmes de traitement passif ou actif	• exploitation peu importante; nouveau substrat requis deux fois par année • investissements pour un réacteur ouvert (de 50 à 60 L/min) estimés à environ 33 500 $; pour un réacteur fermé pouvant traiter de 75 à 100 L/min, 56 000 $
Biosorbeurs	• les métaux sont éliminés de la solution par adsorption/ absorption sur des cellules de biomasse vivante ou morte • il existe peu d’études sur le terrain; la plupart de la recherche a été réalisée à l’échelle du laboratoire	• peut avoir une application comme forme secondaire de traitement si intégrée à un autre système de traitement • durant l’hiver, le succès des systèmes à cellules vivantes est incertain	• il n’existe pas suffisamment d’informations pour une évaluation à grande échelle.

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This document is dedicated to the memory of Dr. Ron McCready. Ron was inquisitive, honest and a good friend of MEND.

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ABBREVIATIONS

ALD	anoxic limestone drain
AMD	acid mine drainage
BIO-FIX	Biomass Foam Immobilized Extractant
BOD	biological oxygen demand
CQVB	Centre Québecois de Valorisation de la Biomasse
DO	dissolved oxygen
MEND	Mine Environment Neutral Drainage Program
MMLER	Metal Mining Liquid Effluent Regulations and Guidelines
MRA	Metal Removal Agent
MWTP	Mine Waste Technology Program
NPDES	National Pollution Discharge Elimination System (United States)
ppm	parts per million
PVC	polyvinyl chloride
SAPS	successive alkalinity producing systems
SRB	sulphate-reducing bacteria
TSS	total suspended solids
TVA	Tennessee Valley Authority
USBM	United States Bureau of Mines

All abbreviations conform to SI units, or standard chemical terminology.

1.0 INTRODUCTION

1.1 BACKGROUND

One of the major issues facing the Canadian mining industry is the treatment of effluent during and after closure of a mining property. For some sites, effluent treatment is complicated by the presence of acid mine drainage (AMD), which may require long term treatment. Treatment may be in either an active or passive form. Passive treatment technology can be defined as any technology that takes advantage of chemical and biological processes that occur in nature, to ameliorate contaminated water. Generally, passive treatment systems are less dependent on an energy source supplied by man. Passive treatment systems were initially considered attractive to treat AMD due to their expected lower costs of construction, operation and maintenance, and their ability to operate at remote locations with limited operational requirements.

Research has been conducted primarily in the United States regarding passive treatment systems for AMD. This study provides an overview of the passive treatment systems and research that are available in the literature, and far enough advanced to allow an assessment of the system requirements, performance record, long term implications, and costs. Where insufficient data was available or other data gaps are present they are noted in the text. Some of the projects reviewed were conducted primarily for research purposes, and may not have been optimized for treatment capability. This should be considered when reviewing the information provided in the case studies. The intent of the study is to provide the user with a first step to assess the applicability of the various passive treatment systems currently available to conditions at their site.

In accordance with the scope of work, this document provides:

- summary of known passive treatment technologies;
- maintenance and monitoring requirements;
- life expectancy and long term implications (>100 years);
- implications of treatment product disposal;
- estimate of costs for the technologies described based on generic cases;
- ability to meet Canadian Metal Mining Liquid Effluent Regulations, and to control toxicity;
- descriptions of case studies including: range of flow, temperature, water chemistry, and identification of limiting conditions where available in the literature (Appendix A); and,
- general assessment of the applicability of current passive treatment systems to Canadian mine sites.

Detailed references specific to the individual passive treatment systems are provided. Summaries of key case studies are provided in Appendix A.

1.2 ACID MINE DRAINAGE IN CANADA

A study was recently completed for MEND outlining the characteristics of Canadian Acid Mine Drainage *(Study 3.22.1, Canada-wide Survey of Acid Mine Drainage Characteristics, December 1994)*. Data was collected for 72 different effluent streams representing all geographic regions, from 14 mining companies.

Gold, copper, lead, zinc and uranium mines provided information.

Considerable variability was found with respect to the level of contaminants present. The metals generally found in the highest concentrations in the effluent stream were zinc, iron, and aluminum. Table 1-1 provides a summary of data presented in this report.

A strong seasonality was shown in the data. Flow rates are significantly higher in the spring and autumn as compared to summer and winter. Copper and zinc concentrations are higher in the spring, while lead, iron and cobalt concentrations peak in winter. The lowest concentrations of zinc are measured in the summer and autumn, while copper and cobalt decrease in autumn. Sulphate and acidity concentrations are highest in the winter and lowest in the spring.

1.3 PASSIVE TREATMENT TECHNOLOGIES

Passive treatment systems for AMD use the chemical, biological and physical removal processes that occur naturally in the environment to modify the influent characteristics and ameliorate any associated environmental impacts. The major processes at work include:

- chemical: oxidation, reduction, coagulation, adsorption, absorption, hydrolysis, precipitation;
- physical: gravity, aeration, dilution; and,
- biological: biosorption, biomineralization, bioreduction, alkalinity generation.

Although stated as separate entities, in most treatment systems a combination of all of these processes are active at any time.

Four major types of passive treatment technologies have been selected for review in this document:

- anoxic limestone drains;
- constructed wetlands;
- microbial reactor systems; and,
- biosorption systems.

Each of these technologies are described briefly below, and discussed in detail within the sections that follow.

1.3.1 Anoxic Limestone Drains

Anoxic limestone drains (ALD) are a passive form of alkalinity addition for AMD having a net acidity. ALD's are essentially underground limestone beds through which an unaerated effluent stream, such as a waste rock seep, flows by gravity. As the effluent flows through the system, limestone is dissolved, and calcium and bicarbonate are introduced, adding alkalinity and increasing the pH of the stream. Key to the performance of the drain is the exclusion of oxygen. In the presence of oxygen, metal hydroxides are formed which may armour the surface of the limestone or plug spaces between limestone, making the drain ineffective and subject to failure. To reduce the potential for failure, experience suggests that ALD's only be used to treat effluent having dissolved oxygen, ferric iron and aluminum concentrations below 1 mg/L, and sulphate concentrations below 2,000 mg/L.

1.3.2. Constructed Wetlands

As the name implies, constructed wetlands are engineered ecological systems which use the biological processes commonly found in natural wetlands to modify effluent such as AMD. The effluent treatment reactions most commonly attributed to constructed wetlands include:

- exchange of metals by an organic-rich substrate;
- sulphate reduction and precipitation of iron and other sulphides;
- precipitation of ferric and manganese hydroxides;
- adsorption of metals by ferric hydroxides;
- metal uptake by living plants;
- filtering of suspended and colloidal material; and,
- adsorption or exchange of metals on algal material (after Wildeman 1993).

Wetlands can be designed and constructed to provide the aerobic and anaerobic conditions to support these microbial and abiotic reactions. A consideration of the hydraulics of flow is important to the effectiveness of the system. In the aerobic cells surface flow is promoted to maximize oxygen contact, while in the anaerobic components downward flow is maximized to minimize oxygen contact.

1.3.3. Microbial Reactor Systems

Microbial reactor systems or bioreactors are systems designed to promote microbial sulphate reduction which reduces sulphate to hydrogen sulphide under anaerobic conditions. The hydrogen sulphide reacts with metal ions in solution to form low solubility metal sulphide precipitates. At the same time, the sulphate reducing bacteria consume hydrogen ions and produce carbonate which increase the pH of the effluent being treated. Two types of microbial reactors are currently under study: open systems, which are open to the atmosphere; and closed systems, which are commonly established in tanks or other containers and can be established anywhere, including underground mine workings. Each of these systems require a biodegradable carbon source such as alfalfa or compost to support the bacterial growth.

1.3.4. Biosorption Systems

Biosorption systems are one of the most recent technologies developed for the passive treatment of AMD, but are frequently found in nature. The majority of research to date has been at the bench scale level. All biosorption systems rely on the absorption or adsorption of metal ions from solution to a biological material such as bacteria, algae, fungi and yeasts; by ion exchange, complex formation and precipitation in living or dead cells. Systems which utilize living cells include use of microbial mats and algal mixtures. Non-living systems use a variety of commercially available biosorbent materials based on bacteria, algae, peat moss and others. Apart from these process oriented descriptions, the biosorption systems developed to date have been quite divergent in design and function.

1.4 APPLICATION TO CANADA

The majority of published research and field applications of passive treatment systems have been designed to treat acidic coal drainage in the Appalachia region of the United States. Little information is available regarding the application of the passive treatment systems described above, under climatic conditions similar to Canadian winters or at Canadian mine sites. In general, biological based systems using living organisms will have limited effectiveness if exposed to winter climatic extremes unless considered in the initial design. The effectiveness of these systems may also be limited during spring runoff periods at Canadian mine sites, where there may be high flow conditions but only limited biological activity.

Passive systems have been shown to be most effective in treatment of AMD when designed for the specific characteristics of the influent and the location. This is particularly true of the biological components which resist 'black box' design methodologies. The first step in the design of a passive treatment system is the characterization of the effluent requiring treatment. This will include a detailed assessment of effluent quality and flow over an extended period of time, including sufficient data to determine any seasonal variations.

After a complete characterization, a preliminary passive treatment system can be designed. At most sites, treatment of AMD will require more than one form of passive treatment. At some sites it may be more appropriate to develop the systems in parallel in order to efficiently treat differing effluent qualities under varying flow conditions. This may be applicable at Canadian sites which have concentrated effluents under low flow conditions in the winter.

On completion of the design stage, a preliminary cost-benefit analysis should be conducted in order to better assess the appropriateness of passive treatment for a particular effluent. Passive treatment systems are not a magic solution to AMD treatment. At some sites, the effluent characteristics may be such that development of a passive treatment system may be an ineffective solution. If use of a passive system does not appear prohibitive, the design can be tested in laboratory studies, bench scale studies and pilot systems; each study collecting further information to increase the potential for success of the final application.

1.5 DOCUMENT ORGANIZATION

The document has been divided into individual sections, each detailing one of the passive treatment systems described briefly in Section 1.3.

- Section 2: Anoxic Limestone Drains;
- Section 3: Constructed Wetlands;
- Section 4: Microbial Reactor Systems; and,
- Section 5: Biosorption Systems.

Section 6 provides a conclusion for each of the technologies in terms of their applicability to Canadian conditions, costs, long-term implications and waste by-products; as well as requirements for future work and data gaps. Section 7 details all of the references cited within the main text of the document.

Appendix A which follows after Section 7 includes summaries of case studies reviewed in the preparation of the initial phase and Phase II of this project. The case study references are included in the template descriptions.

TABLE 1-1: SUMMARY OF ACID MINE DRAINAGE CHARACTERISTICS AT CANADIAN MINES (comparison of seasonal values, mg/L)

Parameter	MMLER	Base Metal		Gold		Uranium	
		Min.	Max.	Min.	Max.	Min.	Max.
pH	5	2.38	9.11	2.15	3.52	2.4	8.4
Acidity	-	10	23082	77	60000	3	3065
Sulphate	-	24	29579	360	150000	10	3089
Tot. Susp. Solids	50	0	943	23	1070	1	22
Tot. Diss. Solids	-	233	1050	45000	130000	35	6201
Copper	0.6	0.003	341.7	0.01	63	0.004	1.12
Nickel	1	0.003	36	0.1	12	0.01	1.6
Lead	0.4	0.001	23	0.019	1.9	0.01	0.32
Zinc	1	0.012	2099	2.05	27	0.007	2.1
Arsenic*	1	0	0.3	-	-	-	-
Iron	-	0.038	1394	7.65	20000	0.14	828
Cadmium*	-	0.001	1.92	0.009	0.12	0.002	0.004
Cobalt*	-	0.146	1.38	-	-	0.01	2.2
Aluminum*	-	0.0176	35.3	1100	4800	0.67	140
Manganese*	-	0.586	27	4.54	282	0.02	21

MMLER: Metal Mining Liquid Effluent Regulations (1977), maximum acceptable metal concentration and minimum acceptable pH value in a grab sample
Base Metal: Includes copper, lead, zinc and nickel operations
*limited data was available

(Compiled from MEND Study 3.22.1)

2.0 ANOXIC LIMESTONE DRAINS

2.1 INTRODUCTION

Anoxic limestone drains (ALD's) are a passive form of alkalinity addition for acid mine drainage (AMD) having a net acidity. In their simplest form, anoxic limestone drains are underground limestone beds through which a non-aerated effluent stream is directed for treatment. Anoxic limestone drains rely on passing the AMD to be treated through the drain at a rate that will allow the limestone to be gradually dissolved, releasing calcium and eventually bicarbonate. Burial of the drain excludes oxygen, helping to prevent armouring of the limestone with hydroxides. It allows the development of high CO_2 partial pressures which can produce high alkalinity concentrations beyond that possible under normal atmospheric conditions. ALD's are most commonly used to treat seeps, but may be used for other effluent streams if special measures are used to avoid introduction of oxygen, such as routing the effluent through a pipeline.

The majority of reported research conducted to date on anoxic limestone drains relates to the treatment of coal drainage in the United States. Three to four years of data available in the literature from operating ALD's, show that ALD's: increase calcium concentrations, increase alkalinity, reduce acidity, and as a result, increase pH. By increasing the pH to near neutral levels, the ALD conditions the influent stream so that metal removal by oxidation, hydrolysis and precipitation can occur further downstream.

Metal removal in an efficient ALD occurs under aerobic conditions downstream from the ALD rather than within the ALD. Reduction of metal concentrations between the inflow and discharge of the ALD indicates that armouring is occurring within the ALD. Significant reduction in metal concentrations within the effluent stream should occur, however, on its exposure to the atmosphere. Introduction of oxygen to the highly alkaline effluent will allow the formation and precipitation of metal hydroxides out of solution.

Summaries of case studies reviewed during preparation of this chapter are provided in Appendix A.

2.2 TECHNOLOGY DESCRIPTION

2.2.1 Physical Considerations

An anoxic limestone drain is essentially a trench filled with crushed limestone, sealed under plastic and geotechnical fabric, and covered by soil through which a contaminated effluent stream flows by gravity. Figure 2-1 provides a schematic representation of a typical ALD.

There is a general consensus that high calcium limestone is the most effective material for ALD construction. This is supported by Watzlaf and Hedin (1993) who conducted an experiment to assess the effectiveness of seven different qualities of limestones and dolomite. The limestone samples had a $CaCO_3$ content ranging from 82% to 99% and a $MgCO_3$ content of between 0.4% and 4.5%. The dolomite sample tested had a $CaCO_3$ content of 46% and $MgCO_3$ content of 38%. All of the limestone samples dissolved faster and produced a higher alkalinity than the dolomite sample; however, among the limestone samples there was no correlation between $CaCO_3$ content and alkalinity produced. Information regarding this experiment is provided in Appendix A.

In general, the size of crushed limestone chosen to construct the drain should be a compromise between allowing free flow and sufficient surface area for dissolution to occur. In most of the ALD's reviewed, limestone crushed to between 2 cm to 4 cm was used. Smaller sizes were anticipated to clog too readily, and hamper flow through the ALD. Hedin *et al.* (1994) reported that a number of ALD's constructed with limestone fines and small gravel failed by plugging. As a result, two recent articles suggest that larger particle sizes (8-25 cm, Faulkner and Skousen 1994; 'baseball sized', Hedin *et al.* 1994) should be used in ALD construction to increase hydraulic conductivity and reduce the potential for plugging.

The dimensions of anoxic limestone drains vary from site to site. ALD's are generally shallow in depth, and contain an effective limestone thickness of 1 m to 2 m which is covered by a minimum of 0.6 m of soil. Traditionally, ALD's have been narrow in width (0.6 m to 1.0 m), with sufficient length to provide the retention time required to reach chemical equilibrium based on the predicted flow regime. Drains of up to 20 m wide have also been shown to be effective, and produced alkalinity concentrations similar to more conventionally shaped systems (Hedin *et al.* 1994).

Dimensions for construction of an ALD at a specific location can be established once the volume of limestone required is determined. Hedin and Watzlaf (1994) have developed the following equation to determine the mass of limestone (M) required within an ALD to treat AMD:

$$M \;\; ' \;\; \frac{Q \;\, ?_b \;\, t_r}{V_V} \;\; \% \;\; \frac{Q \;\, C \;\, T}{x}$$

where: $\mathbf{Q}$ - flow (L/h)
$\mathbf{?_b}$ - bulk density of limestone (kg/L)
$\mathbf{t_r}$ - retention time (h)
$\mathbf{V_V}$ - bulk void volume (percent in decimal form)
$\mathbf{C}$ - required alkalinity (mg/L)
$\mathbf{T}$ - design life (h)
$\mathbf{x}$ - $CaCO_3$ content of the limestone (% in decimal form)

The most recent literature based on a variety of different case studies, suggests the following retention times (t_r) are generally appropriate:

Faulkner and Skousen (1994)	15 to 20 hours
Hedin and Watzlaf (1994)	14 to 23 hours, 15 hours recommended
Hedin *et al.* (1994)	14 hours

Experience has shown that beyond these retention times, the ALD will not achieve further alkalinity. These retention times were generally not used in the early ALD's represented in the literature. A design life (T) is chosen based on the expected period of treatment required.

The ALD should be designed to accommodate the maximum expected flow in order to prevent hydraulic failures and to ensure that the minimum retention time is maintained at all times. Maximum flow should be estimated from historical flow measurement data, or if this data is not available, by hydrological modelling. If the rate of flow through the drain is such that insufficient retention time is available, maximum alkalinity concentrations will not be achieved. Flow equalization may be necessary at some sites if the seasonal variations in flow are significant and maximum alkalinity concentrations are required. Average rates of flow treated vary considerably depending on the design of the individual ALD. Due to the relatively low cost for construction and maintenance of an ALD relative to other forms of alkalinity addition (passive and active), most ALD's have been over-designed to maximize longevity and ensure hydraulic stability.

Using this equation, to achieve 300 mg/L alkalinity over a preferred design life of 20 years (175,320 h) with a flow of 25 L/min (1,500 L/h) and limestone characteristics of: 1,600 kg/m^3 (1.6 kg/L) bulk density, 90% calcium carbonate and a bulk void volume of 50%; an effective mass of approximately 160 t of limestone would be required as shown below. This assumes that a retention time of 15 h is preferred (as per Hedin and Watzlaf 1994).

$$M\,(kg) = \frac{(1{,}500\ L/h)\ (1.6\ kg/L)\ (15\ h)}{0.50} + \frac{(1{,}500\ L/h)\ (300\ mg/L\ \left(kg/10^{6}\ mg\right)\ 175{,}320\ h)}{0.90}$$

$$= 159{,}660\ kg = 159.7\ t$$

Brodie *et al.* (1992) reported that a thickness in the order of 10 cm of the limestone at the base of the ALD may be ineffective if imbedded into low permeability soil at its base. The volume of limestone to be added (in this case 159.7 t) should be modified to address this loss of effective surface area.

Based on the volume of effective limestone required and an assumed depth, the length and width dimensions for the ALD for a particular site can be determined based on site constraints. The angle of the side slopes are not critical to the operation of the ALD, although near vertical slopes may facilitate construction (Brodie *et al.* 1992).

All of the ALD's reviewed included a cover of plastic and geotechnical fabric over the crushed limestone. Two layers of 10 mil plastic are preferred, although a single sheet of 20 mil plastic may also be used. The purpose of the plastic is to prevent oxygen penetration and maintain high CO_2 partial pressures (Brodie *et al.* 1991). Placement of geotechnical fabric on top of the plastic helps protect the plastic from rupture during construction, and resists root penetration.

The geotechnical fabric is covered with compacted soil (minimum of 0.6 m; Brodie *et al.* 1991), and revegetated with an aggressive species such as Sericea Lespedeza (*Lespedeza cuneata*) or Crown Vetch (*Coronilla varia L.*). These species discourage establishment of plants with deep roots (such as trees). Deep roots can provide an easy passageway for oxygen penetration into the ALD, and could potentially disrupt the physical structure of the drain. The surface of the soil is crowned in order to allow for eventual subsidence as the ALD gradually dissolves. The type of soil used was not consistent among the cases reviewed, although a compacted clay cap is preferred since it is a more effective oxygen barrier than other more permeable soils (Brodie *et al.* 1992).

It is critical that the ALD design considers the groundwater regime of the site, since the presence of saturated conditions (ie. complete internal flooding) is important to ALD operation. Under saturated conditions, more of the influent stream is in contact with the limestone, and hence, there is increased alkalinity generation (Faulkner and Skousen 1994). Saturated conditions can be obtained by constructing the drain at a shallow depth. If the ALD must be constructed to a greater depth to fulfil limestone requirements over a limited surface area, inundation of the limestone can be ensured by special design measures. Cay dyke(s) can be constructed within the ALD or at its toe to develop pond water, thereby raising the water level within the ALD; or alternatively, riser pipes can be used at the outflow (Hedin *et al.* 1994).

Table 2-1 summarized select applications of ALD's to treat acid mine drainage. Further detail regarding each of the references is provided in Appendix A.

2.2.2 Inflow Characteristics

For optimum effectiveness the influent to the ALD must meet certain quality guidelines. ALD's are most effective for influent with dissolved oxygen, ferric iron (Fe^{3+}) and aluminum concentrations of less than 1 mg/L, sulphate concentrations below 2,000 mg/L, and pH less than 6 (Kepler and McCleary 1994; Brodie *et al.* 1991). Higher concentrations of dissolved oxygen within the influent allow the oxidation of ferrous iron to ferric iron. Presence of ferric iron or aluminum with or without oxygen will cause metal hydroxides - $FeO(OH)$ and $Al(OH)_3$ to form.

Formation of $FeO(OH)$ and $Al(OH)_3$ may cause premature system failure due to limestone armouring and reduced limestone dissolution; or may plug the system and reduce flow. Hedin *et al.* (1994) reported that two ALD's recently constructed to treat AMD containing 20 mg/L aluminum became plugged within six to eight months of operation.

Recent evidence suggests that gypsum ($CaSO_4$) may form within ALD's if the influent contains high concentrations of sulphate (>2,000 mg/L) (Hedin and Watzlaf 1994). Gypsum formation may either decrease the rate of limestone dissolution if it forms on the limestone surface; or it may reduce flow if it forms within the open space between limestone particles. In either instance, the ability of the ALD to generate alkalinity may be significantly reduced, and failure of the system may occur.

If the influent contains greater than 50 mg/L iron (total), hydroxide precipitates will form on exposure to air. At these sites, an oxidation basin should be constructed downstream of the ALD to collect precipitates (Brodie *et al.* 1992). The basin should be designed for the desired retention time, allowing sufficient volume for precipitates to settle. Accommodation in the basin design for periodic dredging will increase effectiveness while reducing overall size requirements.

At sites where the influent contains concentrations above levels recommended for ALD treatment, construction of an anaerobic cell containing an organic substrate upstream of the ALD may be appropriate. Depending on the design of the cell, it may be possible to decrease dissolved oxygen concentrations, reduce ferric iron (Fe^{3+}) to ferrous iron (Fe^{2+}), or precipitate $Al(OH)_3$ prior to entering the ALD and therefore decrease the potential for armouring. However, adequate settling capacity is needed to settle precipitates. To decrease dissolved oxygen concentrations, an anoxic pond with a retention time of from 7 to 10 hours may be required based on a rate of 3.88 $mg/min/m^3$ (Wildeman *et al.* 1994). To decrease ferric iron concentrations or precipitate aluminum hydroxides, material containing sulphate reducing bacteria (SRB) could be added to the substrate. Anaerobic systems utilizing SRB's are discussed further in Section 4.0.

Use of a second ALD immediately after the first is not feasible, due to the elevated dissolved oxygen concentrations present in the effluent after exposure to the atmosphere (Hedin *et al.* 1994). With intervening treatment to reduce dissolved oxygen concentrations, a number of ALD's could operate effectively in series.

2.2.3. Maintenance and Monitoring Requirements

ALD's require little maintenance which is usually limited to: periodic inspection of the site, maintenance of the vegetation cover to ensure that deep-rooted plants do not become established on the soil cap, control of erosion, and periodic inspection of surface subsidence caused by limestone dissolution (if any). These inspections can be carried out annually or more frequently, as appropriate. After some length of time, the ALD will become ineffective when: chemical reactions exhaust the limestone; dissolution of limestone causes the surface to subside; or, channelized flow is initiated within the drain reducing the retention time and exposure of the flow to limestone surfaces.

Disposal may be required of precipitates which form downstream of the ALD on exposure of the effluent to oxygen. To reduce the potential for solubilization of precipitates, periodic dredging of oxidation basins, replacement of wetland substrate to remove metal precipitates, and burial of any sludges produced may be the most appropriate means for disposal. Dredging of precipitates will also help ensure that adequate retention times are maintained within the downstream system. Monitoring may be required to ensure the effectiveness of downstream systems or to fulfil research requirements.

Since ALD's are rarely used as a stand alone treatment process, monitoring for regulatory purposes is not generally required, however, monitoring may be necessary to ensure system effectiveness. This is particularly important during the initial period, when weekly monitoring may be appropriate. Periodic monitoring of the influent should be conducted to ensure that dissolved oxygen, ferric iron and aluminum concentrations remain below 1 mg/L and sulphate concentrations remain below 2,000 mg/L. If concentrations increase above these levels, it may be necessary to modify the system, so that potential armouring or plugging problems are corrected prior to system failure. Modification of the inlet to reduce oxygen inputs or pretreatment of the AMD for metal and sulphate removal may be required.

The monitoring requirement and its frequency during operation will depend on the amount of variation present in the influent quality and the necessity for consistent effluent quality at the site.

2.3 PERFORMANCE

Maximum alkalinity concentrations achievable by use of an ALD are reported to be approximately 300 mg/L $CaCO_3$ equivalence (Nairn *et al.* 1992), with a maximum sustained concentration of approximately 275 mg/L (Hedin *et al.* 1994). These maximum concentrations are a function of the CO_2 partial pressure within the ALD, and are not altered by increasing retention time. An open limestone system equilibrated to atmospheric concentrations of CO_2 would only be able to produce alkalinity in the range of 50 to 60 mg/L (Hedin *et al.* 1994).

ALD's can successfully neutralise AMD which has acidity of <150 mg/L, or approximately a pH of 4.5. If the influent contains levels of acidity >150 mg/L, an ALD may not be able to produce a net alkaline effluent, depending on the chemical characteristics of the influent. At these locations the ALD will still increase alkalinity, partially neutralize the water, and cause some oxidation and hydrolysis of metal to occur upon aerobic treatment, but a low pH effluent stream may still result. Further alkalinity addition may be required downstream by either passive (compost, anaerobic wetland) or active treatment.

Information was not available in any of the case studies reviewed regarding the toxicity of influent treated, or effluent released from ALD's. The chemical composition and toxicity of the ALD effluent is dependant on the quality of the influent. ALD's have been shown to increase alkalinity in most effluent streams to a net alkaline condition, which can improve the effectiveness of downstream treatment, which may reduce the toxicity of the effluent. Toxic levels of alkalinity are not generated by ALD's.

Although long term data is not available, the research conducted to date suggests that ALD's can be expected to be effective for 20 to 80 years (Brodie *et al.* 1992) and perhaps even longer (>100 years), if influent quality is within the required criteria and the system is properly designed and constructed. The life of an individual ALD is dependent on the initial design of the ALD, including: the rate of flow into the system, and the volume of limestone available for treatment. Eventually the ALD will become ineffective when the volume of limestone available is exhausted by chemical reactions. Prior to this occurring, the surface can subside re-introducing oxygen and rendering the system ineffective; or, channelized flow may be initiated within the drain reducing the retention time and exposure to limestone surfaces.

Information was not available in the literature reviewed regarding the potential for armouring of the limestone or plugging of the system by retention of metals and development of hydroxides under low dissolved oxygen conditions (<1 mg/L) over a long period of time (>100 years). In order to better assess the effectiveness of ALD's for treatment of AMD, long-term data is required along with an investigation of other potential problems not immediately identifiable from the limited amount of data currently available.

Performance data for nineteen operating ALD's is provided in Faulkner and Skousen (1994). This data, as well as performance data from other operating ALD's are summarized in Appendix A. Since the ALD is an underground system open to inflow from the sides as well as at the point of input, some of the performance data available in the literature may not accurately represent actual treatment effectiveness if the ALD captures non-target water. If uncontaminated water flows into the ALD, the ALD will appear to be more effective than expected due to dilution. Conversely, if contaminated groundwater flows into the system, the effluent discharged could be more contaminated than the original influent.

2.4 APPLICABILITY

Anoxic limestone drains are an effective, inexpensive form of passive alkalinity addition for AMD having dissolved oxygen, ferric iron and aluminum concentration below 1 mg/L, and sulphate concentrations below 2,000 mg/L (based on the results of treatment of acid mine drainage at coal mines in the United States). Presence of elevated concentrations of theses components in the seep or influent stream requiring treatment may cause premature system failure by armouring of limestone, or plugging the system and reducing flow. Most acid mine drainage in Canada does not meet these influent requirements (Table 1.1). ALD's may, however, be appropriate at Canadian sites which require treatment of an effluent stream meeting these criteria.

As a result of these strict influent requirements, anoxic limestone drains may have only a limited long-term application for Canadian mines. At sites which have elevated concentrations of metals and acidity, ALD's may still be able to provide a less expensive shorter term solution than chemical treatment; however, a cost-benefit study may be prudent to compare ALD's to conventional chemical treatment and other forms of passive treatment (such as a compost wetland). At some sites where surface area is available, cost benefit analyses may show that periodic reconstruction of ALD's as they fail may continue to be a cost effective solution for alkalinity addition.

Insufficient information was available to assess the result the presence of other metals more typically found at Canadian mine sites (such as copper, nickel, lead and zinc) may have on the effectiveness of the treatment system.

None of the case studies available for review described ALD's constructed in Canada, or locations with climates similar to the mining areas of Canada. The majority of the information available is based on treatment of coal drainage in Pennsylvania, West Virginia and other parts of the Appalachia. No references were found to indicate whether or not temperature had an impact on alkalinity generation or ALD effectiveness (apart from impacts from reduced flow due to freezing effects). Very limited information was available regarding the operation of ALD's during the winter (under either complete freeze or partial thaw conditions).

The information reviewed does not suggest that ALD's will be less effective in a more extreme northern climate. Exothermic reactions associated with AMD, assisted by the insulating capacity of the snow and soil cover, may allow the ALD to remain open and operable throughout the winter. As long as flow continues and is kept in contact with the limestone surface, it is expected that the ALD will continue to operate and alkalinity will be generated. It is unknown if frost heave would have a significant impact on ALD operation in the long term. Further information is required regarding the application of the technology to northern climates.

Anoxic limestone drains are not a stand-alone treatment method for AMD. The function of ALD's is to add alkalinity to the effluent stream, which may in turn improve the effectiveness of downstream treatment systems, and cause precipitation of metal hydroxides. In this manner, ALD's may improve the quality of effluent released from a site; however, the chemical composition and toxicity of the ALD effluent itself is highly dependant on the quality of the influent. As a result, the ability of ALD effluent to meet Canadian water quality criteria, such as the Canadian Metal Mining Liquid Effluent Regulations, is highly dependant on the quality of the influent.

2.5 COST

Anoxic limestone drains are a relatively inexpensive form of alkalinity addition. The total construction costs of the ALD's in the case studies reviewed were less than $30,000 (Cdn.), including: labour and supervision, equipment rental, and materials (high calcium limestone, plastic liner, geotechnical fabric, seed and fertilizer).

A breakdown is provided below in Canadian dollars, for the construction of ALD's similar to existing ALD's in Pennsylvania and Alabama which were designed in the past to treat flows in the range of 7.5 L/min and 125 L/min respectively for drainage associated with coal mining operations (Nairn *et al.* 1992; Brodie *et al.* 1992). These construction costs assume that the site is located at a non-remote location, and does not require special construction methods. Neither of these cases have the ideal retention time of 14 to 23 hours currently preferred.

Case 1: Anoxic limestone drain 50 m long, 0.6 m wide and 2.5 m deep (average), flow rate of 7.5 L/min, 24 h retention time:

Excavation		$ 650
	(75 m^3 @ $8.50/$m^3$)	
Crushed Limestone - High $CaCO_3$		1,750
	(70 t @ $25.00/t)	
Geotextile - Nonwoven, 270 R		50
	(40 m^2 @ $1.25/$m^2$)	
Geomembrane - Two layers, 10 mil PVC liner		400
	(80 m^2 @ $5.00/$m^2$)	
Native Backfill (local material)		450
	(45 m^3 @ $10.00/$m^3$)	
Revegetation - seed, mulch, fertilizer		100
	(40 m^2 @ $2.50/$m^2$)	
Management	(lump sum)	600
ESTIMATE (Cdn$)		**$ 4,000**

Previous chemical treatment costs for the site that this case was based on were reported as over $10,000 (Cdn.) annually for plant operation and maintenance, sludge handling and disposal and labour (Nairn *et al.* 1992). Over $2,000 of the total amount was allocated for chemical purchases. At this site, the total construction cost for the ALD is less than the annual costs for chemical treatment.

Case 2: Anoxic limestone drain 80 m long, 4.0 m wide and 1.5 m deep (average), flow rate of 125 L/min, 62 h retention time:

Excavation		$ 4,100
	(480 m^3 @ $8.50/$m^3$)	
Crushed Limestone - High $CaCO_3$		10,000
	(400 t @ $25.00/t)	
Geotextile - Nonwoven, 270 R		400
	(320 m^2 @ $1.25/$m^2$)	
Geomembrane - Two layers, 10 mil PVC liner		3,200
	(640 m^2 @ $5.00/$m^2$)	
Native Backfill		2,000

	(200 m^3 @ \$10.00/m^3)	
Revegetation - seed, mulch, fertilizer		800
	(320 m^2 @ \$2.50/m^2)	
Management	(lump sum)	2,000
ESTIMATE (Cdn$)		**$ 22,500**

Previous conventional treatment costs for the operating ALD that Case 2 was based on were in the range of approximately $28,500 (U.S.) annually for NaOH and $17,000 (U.S.) for labour; resulting in a total annual cost of approximately $45,500 (Brodie *et al.* 1991). NaOH is a very costly form of alkalinity, suitable only for small remote flows.

A ‘typical’ ALD constructed at most locations in Canada is expected to cost in the range of $4,000 to $25,000 depending on chosen dimensions and design flow. This estimation would not apply to more remote sites, or sites where establishment of an ALD would require extensive excavation or blasting. This cost does not include construction of related downstream systems such as oxidation basins or constructed wetlands. Prior to construction of an ALD, the expected influent to the system should be characterized to ensure that an ALD is the best choice for alkalinity addition. A cost-benefit analysis should also be conducted once the anticipated life of the system is known.

Maintenance costs for ALD’s are not expected to be significant. Apart from monitoring costs which might be required to ensure the effectiveness of downstream systems, costs should be limited to periodic inspection of the site and maintenance of the vegetation cover. Inspection of the ALD may be required weekly immediately after construction, eventually being reduced to monthly as the cover becomes established. Once the vegetation cover is well established and erosion problems are not anticipated, inspection at most sites can then be conducted seasonally. Dredging and disposal of precipitates may also be required, and may be considered either within the maintenance cost of the ALD, or as part of the cost for additional downstream treatment.

2.6 OUTSTANDING ISSUES

As compared to other biologically based passive treatment systems, the mechanisms of treatment within an ALD are well understood. Future work should focus on the following areas:

- applicability to Canadian climatic conditions;
- effectiveness and life expectancy for influent outside of the recommended concentrations (as is likely at Canadian sites); and,
- characteristics, volume, and disposal options of the precipitates generated downstream of the ALD.

ALD’s have only been in operation for a short number of years, and as a result, long-term (>100 years) information is not available regarding hydraulic stability and effectiveness. Studies by various groups have indicated that ALD’s can be expected to be effective for 20 to 80 years if properly designed and constructed (Brodie *et al.* 1992). Information is not available at this time regarding the potential for armouring of the limestone or plugging of the system by retention of metals and development of hydroxides under low dissolved oxygen conditions over a long period of time.

Experience has shown that the presence of organic matter and microorganisms may in fact reduce ALD effectiveness. At a number of sites where hay was added hoping to increase ALD effectiveness, the rate of limestone dissolution was in fact slowed since the hay encouraged microbial growth which reduced the available pore space. In addition, the hay was found to clog openings within the system and cause hydraulic failure (Faulkner and Skousen 1994).

Recent research has however, demonstrated the possible effectiveness of combining ALD technology with sulphate reduction mechanisms. In a Successive Alkalinity Producing System (SAPS), influent is directed down through a rich organic wetland substrate containing sulphate reducing bacteria, into a limestone bed, and out through effluent pipes. SAPS are expected to have the potential to generate alkalinity beyond the 300 mg/L obtainable in an ALD and may reduce the wetland treatment area required. In addition, they can be combined in series with other systems (Kepler and McCleary 1994). SAPS may become an effective passive treatment technology for AMD; however, insufficient information is available at this time to fully assess its potential applicability for Canadian mines.

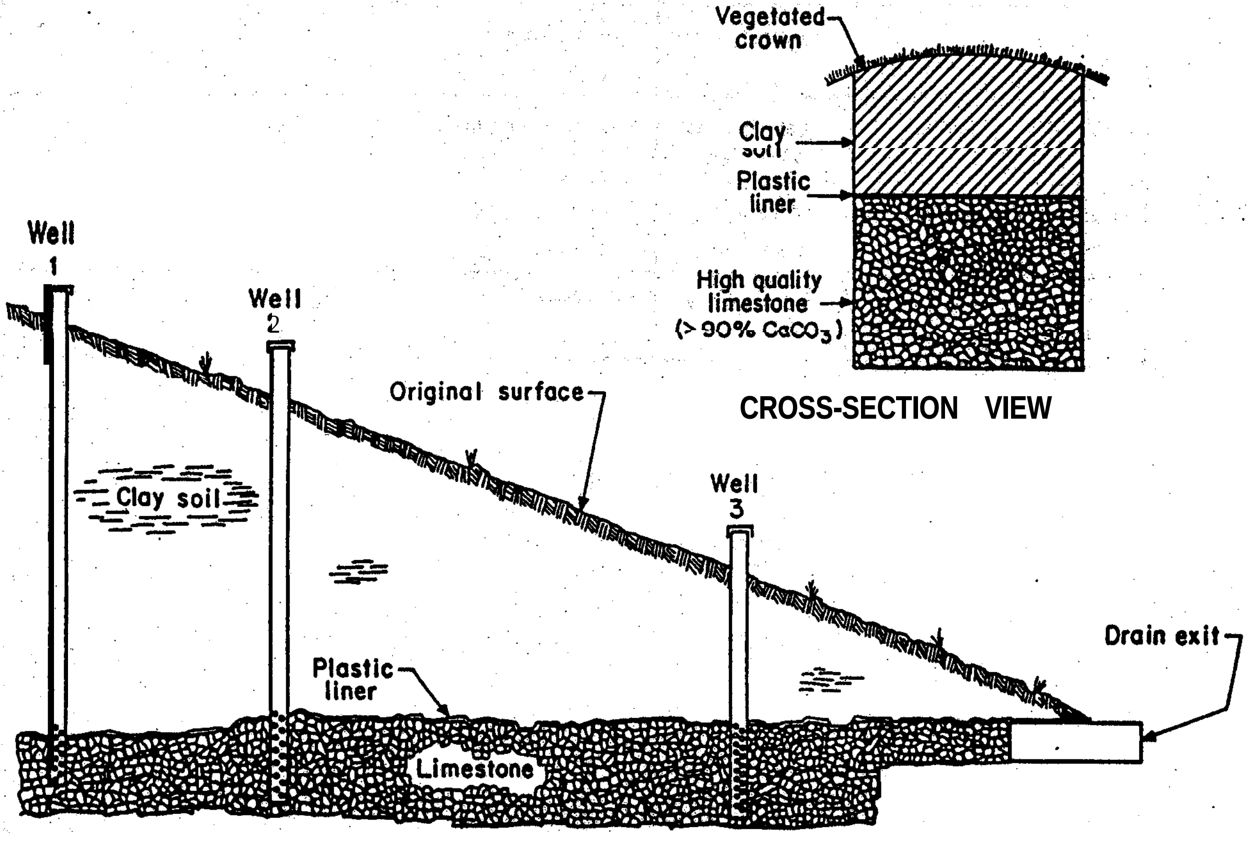

FIGURE 2-1: CROSS SECTION THROUGH AN ANOXIC LIMESTONE DRAIN
NOTE: WELLS ARE FOR MONITORING PURPOSES (From: Hedin et al. 1994)

TABLE 2-1: DESIGN CHARACTERISTICS OF SELECT ALD APPLICATIONS

Reference	Location	Flow Rate	Dimensions (Width-Length-Depth) or Limestone Volume
Brodie *et al.* (1992)	Alabama	155 L/min	4.5 m x 80 m x 1.5 m
Brodie *et al.* (1992)	Tennessee	1,855 L/min	1.5 m x 425 m x 3 m
Nairn *et al.* (1992)	West Virginia (19 ALD's)	4 L/min - 87 L/min	8 t - 250 t
Faulkner and Skousen (1994)	Appalachia (21 ALD's)	0.5 L/min - 265 L/min	35 t - 945 t
Hedin and Watzlaf (1994)	Pennsylvania	7.5 L/min	70 t

3.0 CONSTRUCTED WETLANDS

3.1 INTRODUCTION

Wetlands are composite ecological systems where a variety of physical, chemical, microbial and plant-mediated processes occur which can effect significant changes in the water chemistry. These include oxidation, reduction, precipitation, chelation, adsorption, complexation, sedimentation, filtration, active plant uptake and microbial mechanisms. Plant growth and decay in the wetland provide a constant supply of degradable organic material with readily available organic carbon. The organic matter provides ion-exchange and adsorption sites while stimulating a consortium of bacterial activity, and setting into motion a number of biotic and abiotic processes. Wetland plant surfaces also provide large attachment surfaces for bacterial growth, and act as hydraulic baffles which increase the contact between the nutrients in the water and the bacterial flora.

Passive systems for acid mine drainage (AMD) treatment have commonly used combinations of natural or constructed wetlands, *Sphagnum* peat and open ponds, supplemented by chemical amendments (mostly limestone) and organic substrate to increase alkalinity and reduce acidity. There are two primary objectives:

- raise pH, remove acidity
- remove metals, such as:
    - sulphides (CuS, PbS, ZnS, CdS, NiS, FeS_2)
    - hydroxides ($Fe(OH)_3$, $Al(OH)_3$, $Mn(OH)_2$)
    - carbonates ($FeCO_3$, $MnCO_3$, $ZnCO_3$)

Sequential treatment of AMD to remove iron by oxidation, hydrolysis and settling in the aerobic stage, followed by sulphate reducing bacteria (SRB) activity in an anaerobic stage to raise pH, is an effective combination.

In constructed wetlands, influent AMD drains by gravity through the wetland, progressively undergoing metal removal and neutralization. Metals are removed in the aerobic zones by precipitation, chelation and exchange reactions. Neutralization is primarily achieved in anaerobic zones by the activity of SRB and the increase in alkalinity associated with specific chemical and microbial reactions, such as the progressive dissolution of limestone beds. There are two general types of constructed wetlands: aerobic wetlands and anaerobic wetlands. Usually, a passive treatment system will incorporate both aerobic and anaerobic environments appropriate to carry out microbial and abiotic reactions.

Aerobic wetland systems are designed to maximize the effectiveness of oxidation reactions. Oxidation reactions cause the precipitation of metals primarily as oxides and hydroxides, which become incorporated into the wetland sediment. Aerobic processes have been demonstrated to remove considerable levels of metals, however, the oxidation process causes a net increase in acidity by the release of H^+ ions, and/or the consumption of carbonate. By contrast, anaerobic wetlands rely on the lack of oxygen and the promotion of chemical and microbial processes to generate alkalinity. Reactions occur within the water saturated and oxygen depleted substrate and not within the shallow water column. The substrate usually consists of high organic content material such as sawdust, manure, or compost which act as a nutrient source for reducing bacteria, overlying a carbonate source, usually limestone (Hedin *et al.* 1994). Figures 3-1 and 3-2 provide schematic representations of possible wetland designs for AMD treatment.

Wetlands are by nature heterogeneous, characterized by horizontal and vertical inhomogeneity in terms of nutrient distribution, organic and inorganic substrate, redox, water chemistry and microbial productivity. It is possible to alter the degree of this inhomogeneity and improve the ability of the system to treat AMD by introducing design modifications in an existing or a constructed wetland. Hydraulics of flow play an important role in ensuring that the microbial populations and the substrate get maximum contact with the AMD. Percolation of the water through the vertical and horizontal layers of the wetland is crucial for developing a highly efficient system. Without proper engineering, wetlands do not normally meet the high demand for alkalinity, buffering capacity and readily degradable organic carbon required for treating high loadings of AMD.

Once well established, the wetland can generate large deposits of organic detritus which decompose and provide anaerobic zones. Wetland plants have the unique ability to transfer oxygen from their photosynthetic activity to the roots and rhizomes via the *aerenchyma* tissues. Commonly used emergent plant species in wetlands can transfer up to 45 g $O_2/m^2/d$ to the rhizomes creating an aerobic environment for oxidation and precipitation reactions. The radial loss of oxygen from wetland plant roots can range from 100 to 400 mg $O_2/m^2/h$ (Watson *et al.* 1988).

An efficient wetland is a capable accumulator with the collected metals and reaction products stored in the substrate and biomass. This is particularly true of the systems designed to remove heavy metals. Unlike the wetlands designed to remove organic pollutants such as in the treatment of sewage where they can function well for more than 20 to 30 years before requiring any major construction works, AMD-treating wetlands will reach toxic levels much sooner, rendering their long-term applicability (>100 years) doubtful. Organic pollutants are degraded to simpler molecules, water and carbon dioxide and, therefore, are not conserved in the wetland.

Appendix A contains summaries of constructed wetland case studies reviewed during this project.

3.2 TECHNOLOGY DESCRIPTION

3.2.1. Chemical and Biological Considerations

As described above, constructed wetlands are built to optimize the processes that occur in a natural wetland. Organic substrates commonly employed in constructed wetlands include: spent mushroom compost, *Sphagnum* peat, hay bales, manure, sawdust, wood shavings and barley mash. Cattail and sedges have been used to develop wetlands, and in some cases other aquatic vegetation species have been allowed to emerge to provide the plant cover. Cattail density in most constructed wetlands has been limited to less than 10 plants/m^2, and plant coverage has been only partial or less than optimal. Most wetland operations have not used fertilization to increase the plant growth and productivity, which may be a factor in the observed inefficiency of most operations.

Cattail roots and peat are known to have appreciable cation exchange capacity to remove metal ions.

Many of the metal removing mechanisms in a wetland are temporary and reversible, and can reach saturation thereby reducing the wetland’s efficiency. As a composite system the wetland develops a natural microbiological environment which, to a large degree, can be modified to achieve site-specific design objectives. Processes such as accumulation and decay of organic detritus over a period of time can produce an oxygen demand far in excess of what the wetland plants can replenish, resulting in contiguous or widely distributed anaerobic zones. Accordingly, the prevalence of reducing conditions and anaerobic microbial degradation processes produce short chain organic acids or short chain fatty acids which support the growth of SRB. In essence, the wetland offers a wide spectrum of microbial and chemical environments. A highly optimized wetland environment without the plants would resemble an open microbial reactor system.

Algal growth in wetlands treating AMD can provide a significant pathway for metal removal and can effectively function as a source of readily degradable organic carbon. Decaying mats of algae create anaerobic zones and enhance metal removal by precipitation and stimulate SRB activity. Physiologically engineered algae have been shown to accumulate metals up to 10% of their weight, with a corresponding concentration factor of up to 5,700 (Lakshman 1989).

Enhancement of microbial activity and plant productivity can produce long term metal removal and pH neutralization if certain critical design parameters are introduced. It is essential to recognize that opposing mechanisms such as oxidation and reduction, neutralization and acidification, and solubilization and precipitation, will occur throughout the wetland unless they are streamlined by specific design constraints or opportunities.

3.2.2. Physical Considerations

Sizing influences the design and cost of the wetland operation. Published data shows that efficiency is not a function of sizing. Wetlands differ greatly in terms of substrate chemistry, plant coverage and density, open areas, flow hydraulics, retention time, and other design characteristics as shown in Table 3-1. Wetland design should consider the impact of precipitation and the spring runoff on the flow velocities in the wetland. It is necessary to control the turbulence and prevent the establishment of preferred channels to minimize or eliminate erosion.

Since metal removal and neutralization rates depend significantly on these design parameters, it is not appropriate to define or recommend a sizing for wetland construction unless the system components are standardized. A number of sizing recommendations have been proposed based on criteria such as hydraulic loading and influent pH, Fe and Mn values (Wieder *et al.* 1989). Hellier *et al.* (1994) recommended a composite sizing index based on the combined requirements of influent acidity, Fe and Mn loadings. These recommendations, however, refer to coal mine drainage where iron and manganese are often the only metals observed above regulatory limits. Table 3-2 shows a representative selection of sizing and loading recommendations; Table 3-3 summarizes the crucial system requirements and the importance given to them in current operations; and, Table 3-4 summarizes methods available to increase system efficiency.

3.2.3. Maintenance and Monitoring Requirements

The most important components of a constructed wetland are the plant cover and organic amendments.

Once the wetland species are established it is essential to maintain a healthy growth and maximize their productivity and cover. The low nutrient status of AMD and other types of mining wastewater necessitates periodic fertilization of the wetland operation to supply plant nutrients. The composition, rate and frequency of the applied fertilizer depends on the water quality of the influent.

In general, there is a paucity of diagnostic data from wetlands that can be used to monitor the deterioration in the wetland treatment capability over time (and with which to implement corrective measures). The deeper ponds of the wetland system designed to provide anaerobic and settling zones should be periodically monitored to prevent silting. Organic amendments need to be excavated and suitably disposed once saturation levels are reached. Piezometers should be installed to various depths to monitor subsurface flows and their quality to determine the longevity of the substrate.

In addition to routine water quality input and output data to fulfil regulatory requirements, analysis of plant tissue, soil and substrate samples should be undertaken on a less frequent cycle. Analyses should include total nitrogen, phosphorus, iron, manganese, calcium and cation exchange capacity, and should be completed at least once a month during the active operating season. Subsurface flows should be sampled once every two weeks to determine the percolation characteristics. Wetland components should be monitored closely to implement modifications or detect changes in the flow hydraulics that diminish the treatment efficiency.

Metals accumulate and are deposited as complexes in various layers in the wetland. This is an important consideration at Canadian operations where the metal loadings in acid mine drainage generally exceed those studied in the United States. At some time the treatment components in a wetland will become saturated with metals and reaction products, and will need to be regenerated, and the metal precipitates removed. Quantification of treatment product generation and establishment of plans for their disposal must be an integral part of the wetland design strategy.

The economics of treatment include the cost of waste disposal. A good wetland design will invariably optimize the system for maximum treatment and minimum waste generation. The collection, excavation, storage and disposal of treatment waste is necessarily a site-specific question. The volume, biodegradability and toxicity of treatment products will dictate the available options. Controlled incineration, zero-discharge (leachate-controlled) landfill, and waste exchange are some of the available options. Reference to any waste characterization in the published literature is limited. The wetland studies reviewed did not appear to address the problem of generation and disposal of treatment products as a part of their design strategy.

Disposal of treatment wastes from engineered wetlands has not been adequately addressed in current studies. Streamlined procedures for safe disposal of spent substrate and metal-contaminated plant material appropriate to the Canadian conditions and environmental standards need to be developed.

3.3 PERFORMANCE

It is essential that constructed wetlands are managed in terms of their individual components and their mutual interactions to gain a desired overall efficiency since they are by nature heterogeneous and composite systems. Wetland operations for AMD treatment developed in the past have not been targeted for maximum metal removal by active plant uptake and microbial activity. Disappointing performance of many wetland systems can be attributed to the design intention to view the wetland as a black box, and develop a system based on the average throughput and empirical data. Wetland efficiency depends on the interaction of the chemical constituents of the influent with the physical, chemical and biotic matrix in all parts of the wetland. It is therefore crucial that the influent has predictable access to the active matrix throughout the period of operation.

Constructed wetlands have been in operation from a few months to several years. Compared to other forms of passive treatment, considerably more information is available for wetlands. Among the Tennessee Valley Authority (TVA) wetlands the oldest operating wetland was constructed in 1976. The Fabius Impoundment was constructed in 1985 and data have been collected from July 1985 to October 1991. The oldest U.S. Bureau of Mines (USBM) wetlands were constructed in 1987. Several mid-course modifications have been implemented however, which preclude a reliable interpretation of the data for determining their long term response. No data have been reported on the toxicity of discharges after wetland treatment.

One of the most in-depth studies of constructed wetlands, their processes and efficiency at treating acid mine drainage, is detailed in Wildeman *et al.* (1990; 1992; 1993). The investigators followed the Big Five Tunnel wetlands over a number of years, through a variety of design modifications.

The major thrust of wetland design and operation has so far been to use wetlands as a renewable source of organic carbon, and as zones for SRB and metal precipitation activities. A lack of a design strategy to optimize the hydrologic, physical, chemical and biotic aspects of a wetland has resulted in diverse types of treatment operations producing contradictory and often misleading data. For example, rushes and reeds in natural and constructed wetlands have been reported to accumulate significant quantities of heavy metals (Stephenson *et al.* 1980); the roots and rhizomes of these plants are known to develop encrustations of oxidized iron deposits. Seidel (1976) found that cattail accumulated 1.1 g Fe/kg and 779 mg Mn/kg on a dry weight basis in a constructed wetland, while bulrush accumulated 0.78 g Fe/kg and 1,200 mg Mn/kg.

From unpublished data from the TVA, Hammer (1990) indicated cattail leaves and stems in wetland cells heavily loaded with AMD contained 5.0 g Fe/kg and 4.1 g Mn/kg. Stark *et al.* (1992) reported that cattail roots in the wetland accumulated five times more iron than the roots in a control wetland. Other reports (Sencindiver 1988) indicate that metal removal by wetland plants is negligible. The reported low metal removals by wetland plants is due to a lack of adequate contact between the plant roots and the AMD, and the very low plant coverage and plant density.

Flow and water quality data for 16 wetlands showed that the surface areas recommended by TVA were 2 to 74 times the areas recommended by the USBM for the same operations. One wetland which was effective in lowering the iron concentrations was only 10% larger than the USBM recommended size. Three other wetlands with sizes closer to the USBM sizing did not produce substantial improvement in the discharged water (Wieder *et al.* 1989). The large variability present in sizing recommendations is essentially due to the black box approach to the wetland design which has neglected a number of crucial system parameters. The parts of the wetland accessed by the influent during one flow regime may not be available during a higher or lower flow regime, resulting in a highly variable treatment efficiency.

Data from Stark *et al.* (1992) showed that the iron reduction corresponded to greater wetland area, high cattail densities and increased plant cover. In an earlier paper Stark *et al.* (1990) reported a seasonal variability in the treatment efficiency in the following decreasing order: efficiency in summer > fall > winter > spring. Hedin *et al.* (1990) found that for a 548 m^2 wetland there was a strong relationship between iron removal and the area-adjusted iron loading, but no such correlation existed for a smaller (330 m^2) and a larger (2,797 m^2) wetland. Stillings *et al.* (1988) and others reported an inverse relationship between iron removal and the influent flow volumes.

Seasonal variability was noticed in the Simcoe wetland operation with the iron removal being the highest in summer and lowest in spring. Over the treatment period from November 1985 to October 1989, the average reductions in total Fe, acidity, alkalinity and sulphate were 62%, 74%, 70% and 7% respectively, while the total Mn and total suspended solids (TSS) increased by 5% and 25%; pH values remained almost the same at 6.5. The mean iron removal rates for 1986, 1987, 1988 and 1989 were 49.25%, 65.5%, 74.5% and 66%, respectively (Stark *et al.* 1990). A few studies required chemical treatment of the effluent with NaOH periodically to meet the discharge limits. Further case study results are provided in Appendix A.

Stark *et al.* (1992) reported that iron removal rates increased with the increased plant density and coverage during the period of operation. Their data indicated that cattail roots and rhizomes directly absorbed or precipitated iron. Stillings *et al.* (1988) studied the removal and retention of iron and manganese in a cattail dominated wetland. Their data indicated a good correlation between the basin length traversed and iron removal although this correlation for manganese removal was not as significant.

Most of the studies reviewed reported significant iron removal rates far higher than manganese removal rates. Few reported results for any of the other metals common to Canadian metal mine effluent streams (such as copper, lead, nickel, zinc). In almost all cases, discharges consistently met the U.S. National Pollution Discharge Elimination System (NPDES) limits for TSS. In general, the following trend appears in the studies for removal rates, in the decreasing order of effectiveness: TSS > Fe > pH > Mn. Hedin *et al.* (1994) reported that all of the 18 wetlands monitored produced significant iron removal, with an average rate of 23 grams per day per square metre ($g/d/m^2$) for sites receiving alkaline influent; the sites receiving highly acidic influent has removal rates ranging from 3.1 to 15.4 $g/d/m^2$. Iron removal at the acidic sites was approximately half of the values obtained from the alkaline sites. At the acidic sites the SO_4 removal ranged from 1.5 to 8.9 $g/d/m^2$. Under laboratory conditions, using 2.5 L reactor columns, Fyson (1995) reported substantial removal of nickel and arsenic by muskeg sediments.

Twelve of the wetlands managed by the TVA consistently met the NPDES monthly average discharge limits without chemical treatment. Average discharge values were: pH of 6 to 9, Fe < 3.0 mg/L, Mn < 2.0 mg/L, and TSS < 35 mg/L (Brodie 1993). At the Fabius Impoundment 1, total Fe was reduced from 69 mg/L to 0.9 mg/L, total Mn was reduced from 9.3 mg/L to 1.6 mg/L, and the pH was increased from 3.5 to 6.8. Total suspended solids discharge requirements were readily met (Brodie 1993). For comparison purposes, the Canadian Metal Mining Liquid Effluent Regulations and Guidelines (MMLER) for these parameters (acceptable concentration in a grab sample) are: pH, minimum of 5.0; TSS, maximum of 50 mg/L. Most of the other parameters regulated under the MMLER were not monitored at the full-scale constructed wetlands reviewed (see Appendix A). Eger *et al.* (1991) did show reductions in concentrations of nickel and zinc in the influent to meet the MMLER of 1 mg/L for each parameter using wetland cells over a short monitoring period and at a pilot scale; prior to treatment, concentrations were up to 7.98 mg/L for nickel and 1.96 mg/L for zinc.

Many systems have tried to maximize contact of flow with substrate by reducing short-circuiting and high velocities using hay bales and serpentine paths, however, the results have been inconsistent. Most wetland systems treating AMD are surface-flow operations where the opportunities for contacting the root-zone of the wetland plants are quite limited. In fact, in a subsurface flow arrangement, a 0.6 ha constructed wetland reduced dissolved iron from more than 80 mg/L to less than 1 mg/L in 110 L/min average flow; immobilizing 4,636 kg/a of iron and other metal oxides and hydroxides (Watson *et al.* 1988; Brodie *et al.* 1988). McIntire (1990) found a dramatic increase in pH when AMD was allowed to infuse into the organic substrate through drain pipes. Little improvement in pH and alkalinity was noted when AMD flowed across the surface; however, clogging due to deposition of iron oxyhydroxide flocs was experienced. Wetlands should maximize the drainage of AMD through the anoxic layers and the root zone of the planted areas. Clogging of the substrate has been prevented to some degree of success using perforated underground drain pipes.

Clear data illustrating the response of the systems to variations in influent hydraulic and chemical conditions has not yet been produced. Brodie (1993) reported that data from 12 TVA wetlands indicated no correlation between iron removal, and influent alkalinity and hydraulic loading; or between manganese removal, and wetland size and hydraulic loading. Manganese removal correlated well with influent alkalinity and acidity concentrations. Significant metal removal occurred to meet the NPDES limits. Factors influencing the ability of the wetlands to ameliorate the AMD were identified as: hydrology, iron and alkalinity concentrations; and such wetland characteristics as, depth, area, hydraulics, vegetative and microbial species, extent, and substrate.

3.4 APPLICABILITY

Wetland technology holds promise as a technically viable and economically attractive technology for treating AMD. Its application as a recommended treatment method in Canada will require the development of data under conditions that are unique to Canadian sites.

AMD in Canada has low flow rates and high metal concentrations in winter, and high flow and low metal concentrations in spring, essentially due to dilution from spring runoff. The following table shows the general characterization of AMD in Canada (McCready and Salley 1988).

Parameter	Base Metal Tailings Effluents	Uranium Tailings Drainage Water
pH	2.3 - 3.9	2.3 - 3.9
Acidity	<1 - 33,728 mg/L	2,850 - 6,500 mg/L
SO_4	11 - 17,000 ppm	5,000 - 7,500 ppm
Fe	<1 - 9,860 ppm	1,000 - 2,000 ppm
Mn	<1 - 187 ppm	<1 - 40 ppm

The response of constructed wetlands to the seasonal variations in the loadings is unknown. Changes to the wetland design will be necessary to ensure that the wetland components maintain a maximum exposure to the substrate and wetland plants under highly variable loadings.

Canadian mine sites are characterized by long, cold winters and short, warm summers. Winter operation of wetlands depends for its success on the temperature regime and the designed strategies for minimizing the impact of low temperatures on the hydraulics and microbiology of operation. The design of subsurface flow should consider the thermodynamics of heat loss in the hydraulics of wetland flow. For example, the thermal conductivity of organic detritus in the wetland is 0.05 W/(m.°C), about 44 times lower than the thermal conductivity for ice at 0°C. Thus, layers of organic detritus can be used to develop an effective insulation for reducing the heat loss during the flow in the wetland.

Wildeman *et al.* (1993) provides the following recommendations to improve the operation of constructed wetlands in cold winter climates:

- ensure effluent delivery systems are insulated, so that the thermal energy of the effluent is maintained;
- where possible, wetlands should be situated to optimize winter sun exposure;
- insulate the top of the cell, or design wetland cells with excess surface flow; and
- implement subsurface flow systems to take advantage of the thermal energy within the substrate.

Wetland efficiency is a function of the operating temperature. As a result, adequate redundancy should be built into the system to maintain target discharge water quality throughout the operating period. Data from constructed wetlands designed to operate under extreme conditions in Canada for treating sewage are helpful in delineating design parameters for AMD treatment (Lakshman, 1979, 1994). No information is available, however, on the effect of freeze-thaw cycles on the rate of decomposition of organic substrate, production of humic compounds, and the availability of biodegradable organic carbon. However, microbial-based systems, such as constructed wetlands, effectively shut down at temperatures below 7.5°C. In Canada effluents are usually 0-5°C in winter.

3.5 COST

If optimized, constructed wetlands may be more cost-effective than active treatment methods in terms of construction, and annual operation and maintenance costs. The major cost components are equipment, labour, land acquisition (if necessary), substrate and design. In addition, the annual operation and maintenance costs are expected to be between 10% to 20% of initial capital costs. Brodie (1993) has given the general cost distribution provided in Table 3-5 based on TVA experience (primarily coal drainage).

Construction costs will vary significantly depending on site-specific parameters. Some sites may have advantages such as: bedrock depth, availability of construction materials, substrate and liners. Apart from construction costs, the cost of monitoring, routine supervision, nutrient addition and costs associated with treatment product disposal should also be considered in any cost-benefit analysis. Environmental factors such as groundwater contamination and periodic disposal of spent substrate should be included in cost assessments.

Table 3-6 provides an overview of published costs derived from operations in the United States. These should be applied with caution in Canada, since variations in the local costs, labour, material, and land costs may be significant.

The cost estimate that follows has been developed for a generic wetland treating acid mine drainage at a Canadian mine based on:

Flow rate: 60 L/min or 86.4 m^3/d
Hydraulic Retention Time = 5 d
Depth of Operation = 1.0 m
Construction Depth = 1.5 m
Surface Area = 432 m^2

Cost Estimate

Excavation	648 m^3 @ \$8.50/m^3	\$ 5,500
Gravel	130 m^3 @ \$20.00/m^3	2,600
Substrate	Peat - 200 m^3 @ \$60/m^3	12,000
	Hay, Manure - 100 m^3 @ \$40/m^3	4,000
Planting	500 h @ \$12.50/h, casual, unskilled labour	6,250
Supervision	Professional time 100 h @ \$100/h	10,000
Hardware	Delivery pipe, under drains, discharge culvert (lump sum)	5,000
Miscellaneous	(20%)	9,000
Design	(lump sum)	30,000
ESTIMATE (Cdn\$)		**\$ 84,350**

Annual Operation and Maintenance

Inspection	Trained field crew. 200 h/a, @ \$35/h	\$ 7,000
Fertilization	3 applications/a (lump sum)	1,500
Maintenance	(lump sum)	1,500
ESTIMATE (Cdn\$)		**\$ 10,000**

3.6 OUTSTANDING ISSUES

Disappointing performance of many wetland systems can be attributed to the design intention to view the wetland as a black box, and develop a system based on the average throughput and empirical data. Wetland efficiency depends on the interaction of the chemical constituents of the influent with the physical, chemical and biotic matrix in all parts of the wetland. It is therefore crucial that the influent has predictable access to the active matrix throughout the period of operation.

There is no evidence to show that current wetland applications have attempted to size this access in terms of chemical and hydraulic loadings, residence time, flow velocity, availability of chemical sites and microbial populations. The data obtained from large scale operations to date have related to the overall performance, and as such, it is impossible to determine from the available data the contribution and efficiency of wetland plants in the amelioration of AMD. It is difficult to predict the performance and formulate reliable guidelines for hydraulic and chemical loadings. The major impact on efficiency of constructed wetlands has been due to the chemical and organic amendments.

Constructed wetlands have the potential to address the AMD treatment problem under Canadian conditions and can form an alternative in terms of costs, efficiency and environmental safety at some sites. A number of issues have to be addressed through pilot scale and controlled studies to develop viable treatment scenarios and to fill the following existing information gaps:

- effect of short summers, plant density, plant cover, periodic fertilization and enhanced root development on the rate of metal uptake by cattail;
- effect of large volumes of low concentration flows during spring runoff and the consequent reduced hydraulic retention time on the wetland response;
- effect of increased organic detritus production by cattail cover and its decomposition on the metal removal capacity of the wetland;
- variation in the vertical permeability of the wetland and the organic substrate as a function of the length of operation;
- contributions of enhanced algal growth and decay in the wetland on the uptake and precipitation of metals, maintenance of anaerobic zones and recycling of nutrients;
- provision for lack of plant and biological activity in winter;
- designs for ensuring efficient distribution of AMD in wetlands;
- contribution of abiotic processes to the metal removal efficiency from high-concentration and low-flow regimes during winter;
- designs for increasing the vertical percolation of AMD through the substrate;
- development and optimization of sequential sedimentation, oxidation, plant uptake and anaerobic stages for metal removal and neutralization;
- development of mechanistic approaches to wetland designs for sites characterized by short growing season, variable flow-concentration regimes and very low winter temperatures; and,
- development of mechanistic optimization procedures for designing wetlands for AMD treatment in terms of algal and plant uptake, substrate chemistry and microbial mechanisms.

Examples of constructed wetlands are shown in Figures 3.1 and 3.2.

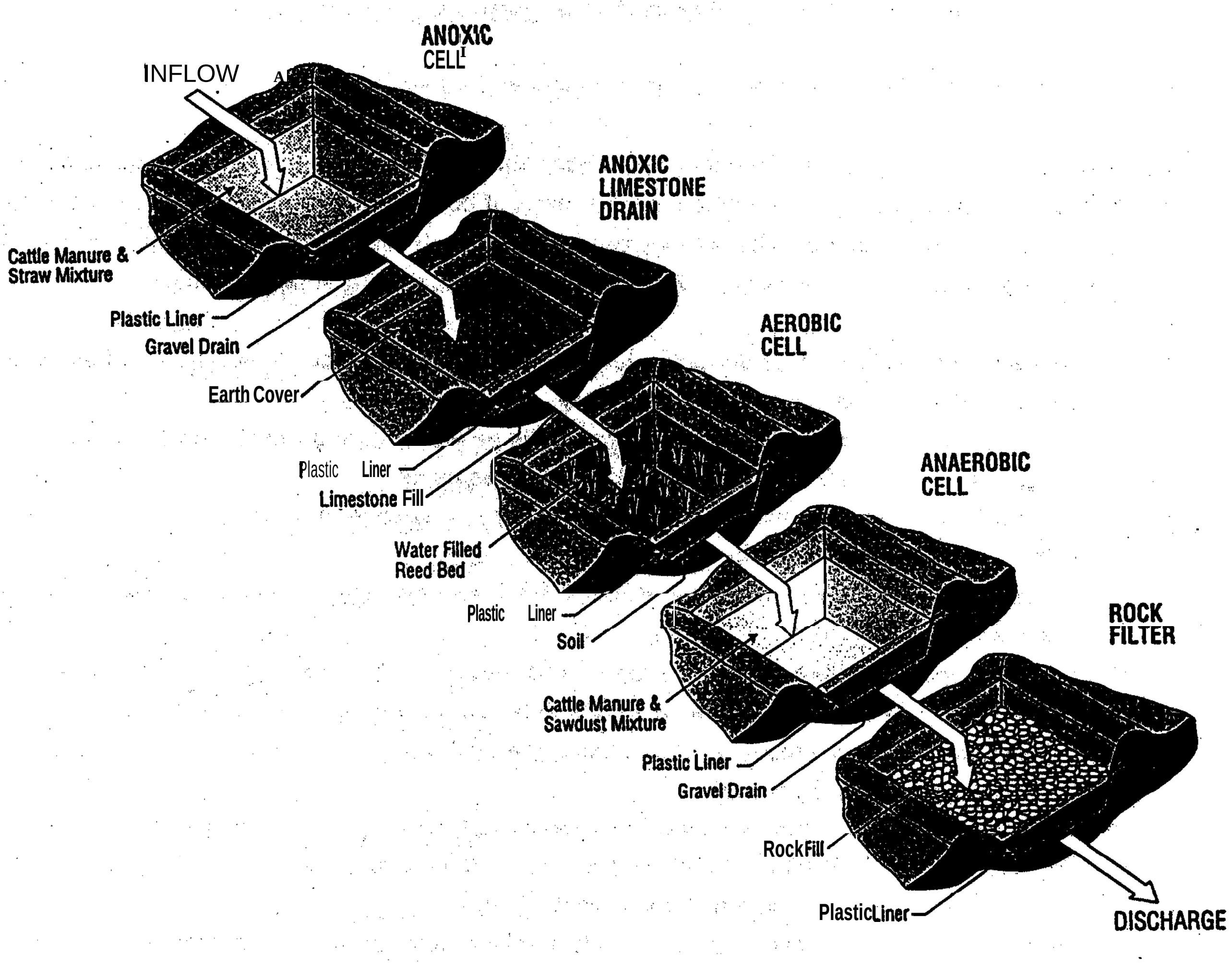

FIGURE 3-1: PASSIVE TREATMENT SYSTEM SHOWING AEROBIC AND ANAEROBIC CELLS IN A CONSTRUCTED WETLAND (Modified from: Cambridge 1995)

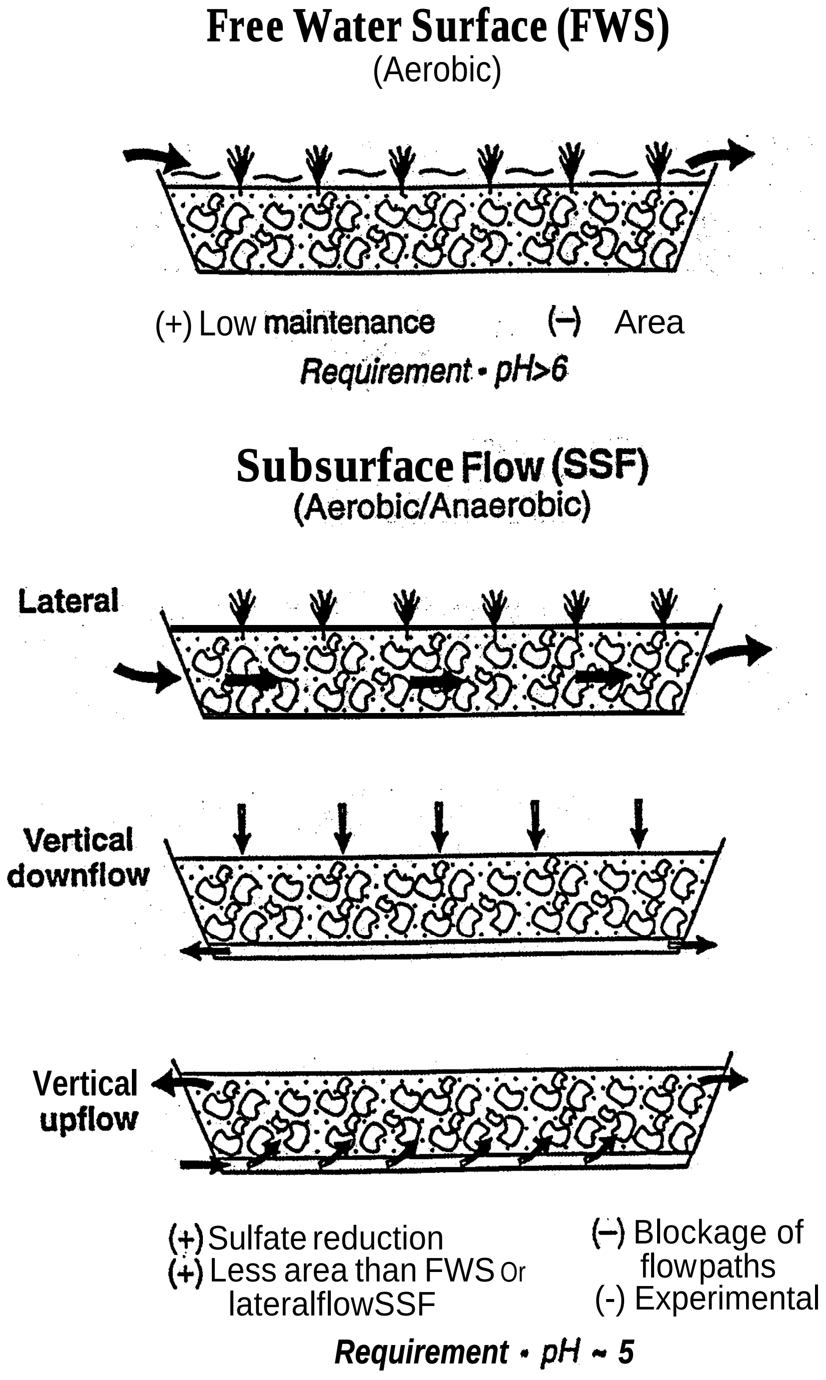

FIGURE 3-2: CONSTRUCTED WETLAND CELLS SHOWING POTENTIAL FLOW PATHS
(From: Jones et *al.* **1995)**

TABLE 3-1: OPERATIONAL DATA FOR SELECT WETLANDS

System	Effluent	Size (m^2)	Flow (L/min)	Retention time (d)	Age (mo)	Substrate
IMP 1	coal	5700	73	nd	72	nd
Tracy - Large	coal	420	30-57	3.8-7.5	12	nd
Tracy - Small	coal	108	23-30	1.88-2.47	12	Limestone/peat
Simcoe #4	coal	2623	328	4.2	48	Limestone/ mushroom compost
Kentucky	coal	180	5.9	5.1	25.6	Peat
Kentucky	coal	180	5.9	5.1	25.6	Mushroom compost
Big Five Tunnel	AMD	18.3	3.6	3.4	36	Mushroom compost
Big Five Tunnel	AMD	18.3	3.6	3.4	24	Peat/manure/ saw dust

nd = no data

(Kalin 1993)

TABLE 3-2: GUIDELINES AND CASE STUDIES DATA FOR WETLAND AREAS

Case Study	Sizing Value	Applicability
Kleinmann *et al.* (1991)	2 m^2/mg Fe/min	Inflow pH< 5.5; outflow Fe=3 mg/L
	0.75 m^2/mg Fe/min	Inflow pH > 5.5
	7 m^2/mg Mn/min	Infl. pH <5.5; outflow Mn=2 mg/L
	2 m^2/mg Mn/min	If inflow pH >5.5
Watson *et al.* (1989)	928 m^2/L/s	Range from 61.1 to 10,700 m^2/L/s
Kleinmann *et al.* (1991)	4.9 m^2/L/min (19-38 L/min)	pH>4.0; Fe<50 mg/L; Mn<20 mg/L
Kleinmann *et al.* (1991)	0.15 m^2/mg Fe/min 0.75 m^2/mg Mn/min 200 to 500 m^2/kg acidity/d	nd
Girts *et al.* (1987)	15 m^2/L/min	nd
Brodie *et al.* (1989)	0.7 L/m^2/min; 1.1 m^2/mg Fe/min; 2.8 m^2/mg Mn/min	nd
Hedin *et al.* (1994)	250 m^2 for 5 kg Fe/d 1,300 m^2 for 5 kg Fe/d	Alkaline waters Acidic waters

TABLE 3-3: SYSTEM PARAMETERS FOR CONSTRUCTED WETLANDS

Category	**Factors to be Considered**	**Currently considered in operating wetlands?**
Input to the Wetland	Hydraulic Loading Chemical Loading Chemical Species in the Influent Chemical Water Quality	Yes
Wetland System	Wetland Plants: Plant species, density and coverage Nutrient supply to plants	No
	Wetland Construction: Shape and configuration (serpentine channels, lagoons, bottom slope, depth)	Some cases
	Substrate: Vertical and horizontal permeability	No
	Non-Wetland Areas: Open ponds, ditches	Impact on the efficiency not considered
Hydraulics of Flow	Flow velocities, flow depth, flow paths through the substrate, short-circuiting of flows, clogging of vertical percolation, development of preferred channels of flow in the substrate, erosion, variation in the residence times	No
Operational Mode	Variable hydraulic and chemical loadings to synchronize with the seasonal variation in the wetland capacity, dilution due to precipitation, temperature effects	No

TABLE 3-4: CONSTRUCTION METHODS FOR INCORPORATING SYSTEM PARAMETERS

System Parameter	Construction Parameters
Increase Contact Time, Hydraulic Retention Time	Add hydraulic baffles, undulating and serpentine paths, reduce short-circuiting, flow-velocities and erosion
Creation of Aerobic Zones	Increase shallow areas, plant density and turbulence, avoid deep flows and accumulation of organic detritus
Creation of Anaerobic Zones	Increase flow depth, deposition of organic detritus, use amendments to supply readily decomposable organic matter
Increase Plant Uptake	High plant density and maximize areal plant coverage, maximize subsurface flow to increase root zone contact, provide adequate nutrient supply by periodic fertilization
Increase Treatment Efficiency	Increase flows during high plant activity periods, summer and fall; reduce flows during low activity periods, winter and spring

TABLE 3-5: TYPICAL COST ESTIMATES FOR CONSTRUCTED WETLANDS

Cost Factor	**Percent of Total Cost**	**Typical Total Cost (US$)**
Design	12%	$6,000
Permitting	2%	$1,000
Land Acquisition	12%	$6,000
Construction:		
- Equipment and Labour	24%	$12,000
- Materials	12%	$6,000
- Supervision	8%	$4,000
Operation and Maintenance	18%	$9,000
Miscellaneous	12%	$6,000
TOTAL	**100%**	**$50,000**

(Brodie *et al.* 1993)

TABLE 3-6: SELECTED PUBLISHED COST ESTIMATES FOR CONSTRUCTED WETLANDS

Source	Cost Estimates (US$)
Brodie *et al.* (1988)	\$ 12.18/m^2
Hiel and Kerins (1988)	Large wetland (418 m^2): \$ 66.53/m^2 Small wetland (111 m^2): \$ 139.89/m^2
Brodie (1989)	Colbert wetland: 0.9 ha, equip. 32.4%, labour 30.4%, O/H 37.2%; (\$ 1.19/m^2 Windows wetland: 2.9 ha; (\$ 6.98/m^2) Kingston wetland: 0.9 ha, equip. 28.6%, Labour 43.3%, O/H 8.1%; (\$14.21/m^2)
Watson *et al.* (1989)	Mean value: \$ 10,000 Wetland sizes: 93 m^2 to 6,070 m^2; mean: 1,550 m^2
Wieder *et al.* (1989)	USBM survey of wetland costs (prior to 1986): less than \$ 2.96/m^2 TVA survey of wetland costs (prior to 1986): \$ 3.58/m^2 - 32.03/m^2
Brodie (1993)	Range: \$ 5.40/m^2 - \$ 32.40 /m^2; Mean: \$ 10.80/m^2
Brodie (1993)	Total cost for 14 wetlands: \$45,000 (1985 \$). Annual costs from 1985 to 1990: \$ 13,000; 1993 costs: less than \$ 1,000 annually
Hellier *et al.* (1994)	Cost is proportional to the wetland area. Median cost \$ 32.29/m^2 (1992 \$); Monthly operation and maintenance cost \$ 100

4.0 MICROBIAL REACTOR SYSTEMS

4.1 INTRODUCTION

Microbial reactor systems or bioreactors are treatment systems that contain a biodegradable substrate to support the growth of organisms which metabolize the substrate and generate short chain organic acids. The short chain organic acids promote microbial sulphate reduction. These systems are strictly dependent on the bacterial consortium in contrast to other biological passive systems which are based on a combination of microbial activity, plant and/or macrophyte adsorption, precipitation or uptake of metals. Microbial activity within the bioreactor may or may not be supplemented with inorganic chemical reactions, such as the adjustment of pH by dissolution of limestone.

Bioreactors rely on several microbial reactions to treat acid mine drainage. The cellulose of the agricultural products must be degraded by cellulolytic bacteria to generate free sugars and other metabolites. Aerobic and facultative heterotrophs will further metabolize these products to provide substrate for the fermentative anaerobes. Under anaerobic conditions these free sugars are fermented to short chain organic acids or short chain fatty acids which are suitable substrate to support the growth of the sulphate reducing bacteria (SRB). The SRB's then reduce sulphate to hydrogen sulphide which raises the pH, and will react with the metal ions in solution and precipitate them as low solubility metal sulphides.

Concurrently, the SRB's consume hydrogen ions and produce carbon dioxide during their metabolism, causing an increase in the pH of the solution due to a reduced concentration of free hydrogen ions and the buffering effect of the CO_2/bicarbonate buffer system. Other microbial activities related to ammonia producers, methanogens and metal reducing species also contribute to the increased alkalinity observed during treatment of acid mine drainage.

Case studies reviewed during preparation of this chapter are summarized in Appendix A. The pilot plant data presented in the literature indicates that bioreactors are a feasible technology for the treatment of small AMD streams and could be complementary to other passive or active AMD treatment systems. Microbial reactor systems are not truly "passive" treatment systems since considerable maintenance and monitoring is required.

4.2 TECHNOLOGY DESCRIPTION

4.2.1 Physical Considerations

Microbial reactor systems are either an open aerobic configuration where the surface of the reactor is exposed to the atmosphere (Béchard *et al.* 1991), or a closed anaerobic configuration where the surface is not exposed to the atmosphere (Kalin 1993). A third type of bioreactor under development at the University of Waterloo, is a bioreactive wall which is placed at the leading edge of an underground acidic plume (Blowes *et al.* 1995). An example of an open configuration microbial reactor system is shown in Figure 4-1.

Two types of closed microbial reactors are shown in Figure 4-2.
All microbial reactor systems require a biodegradable carbon source to support bacterial growth. Most researchers have attempted to use agricultural products (flax, timothy hay, wheat straw, alfalfa) and/or spent mushroom compost as a carbon source, primarily for economic reasons and availability. There is no consensus in the literature regarding the required substrate for optimal microbial growth, although the characteristics of the substrate mixture may limit optimal microbial growth and activity. Rabenhorst *et al.* (1992) reported that a 1:2 mixture of straw and manure was the best substrate. Béchard (1994) indicated that alfalfa was a better substrate for AMD mitigation than either straw or timothy hay. Using alfalfa as substrate (C:N = 13:1) in small microcosms, Béchard was able to produce effluent waters which met environmental guidelines.

To be effective in the treatment of AMD the total microbial consortium must be active, and the microbial production of hydrogen sulphide must be stochiometrically greater than the concentration of precipitable metal ions present in the AMD. The treatment system must have an anaerobic zone to support the sulphate reducing bacteria and have sufficient volume to allow a 10 to 14 day retention time of the AMD within the system. The bioreactor should have an iron oxidation and removal system at the influent to reduce the amount of iron sulphides produced within the reactor.

Although SRB's have been isolated from extreme environments in regards to salinity and temperature (Barghoorn and Nichols 1962; Satake 1977; Gyure *et al.* 1990), they tend to grow best at near neutral conditions and at moderate temperatures in the range of 20 to 30EC. As bioreactors are dependent on microbial and fungal growth, and metabolism, they are very sensitive to temperature and are expected to become ineffective below 5°C, and above 37 to 40°C. To prevent winter freeze-up, open bioreactors should have a minimum depth of 1.5 to 2.0 m. This depth combined with the protective effect of the overlaying substrate should allow the system to operate throughout the winter.

The cellulolytic and heterotrophic organisms required to generate the carbon substrate for the SRB's grow best at moderate temperatures in a neutral pH environment. SRB's will not grow below a pH of 5.5. As a result, the initial cell of the treatment system must produce sufficient buffering capacity in the long term to raise the pH to greater than 5.5 in the rest of the system to sustain SRB growth.

To date, there have only been short term pilot studies on various types of bioreactors, some of which are described briefly in Table 4-1 and summarized in Appendix A. Some of these pilot studies were reviewed by Kalin (1993). As can be noted from Table 4-1, most of the biological reactor studies to date have been conducted at very low flow rates (< 1 L/min). The CANMET Biotrench study which had a flow rate of 23 L/min, increasing to as high as 60 L/min during heavy maritime rain storms is the only exception. Unfortunately, this bioreactor was not successful.

4.2.2. Maintenance and Monitoring Requirements

Both open and closed bioreactors for treatment of AMD will require periodic maintenance. Open reactor systems will also require maintenance to prevent precipitate plugging of the sub-surface pipes or weirs, and erosion of the lined channels or system embankments. Closed reactor systems will likely require maintenance of transfer pumps and periodic cleaning of the pipes to remove metal precipitates or silt which may accumulate within the system. Closed systems must be constructed using fibreglass tanks, or a sulphide and acid resistant metal.

The cellulosic substrate used in the microbial reactor is expected to require supplementation with fresh substrate every six to twelve months. Eventually the degraded cellulosic substrate will become saturated with iron and other precipitated metals, and will have to be removed and replaced with fresh substrate material. This is estimated to be required every three to seven years, depending on the flow rate and the metal ion content of the AMD being treated. As the substrate removed from the bioreactors will be saturated with metal sulphides, the waste should be buried under anaerobic conditions to ensure that the metals do not have an opportunity to resolubilize. The toxicity of this waste material will be dependent on the metallic ion content and stability of the precipitates formed in the AMD. If the drainage contains high levels of such metals as cadmium, arsenic and selenium, the waste material will be quite toxic.

In all the studies to date researchers have monitored chemical parameters such as: alkalinity, acidity, pH, BOD, COD, suspended solids, NH_4-N, NO_3-N, SO_4 , phosphorus, aluminum, copper, iron, manganese, zinc, cadmium and nickel, concentrations in the influent and effluent. In addition to chemical parameters, the Eh at various locations within the reactor should be monitored, since SRB's can only survive at Eh values of less than -150 mV (Postgate 1959) and methanogens require an Eh of less than -330 mV.

The biological parameters of the system have not been thoroughly monitored and identified. This is despite the fact that biological reactor systems are dependent on a consortium of microorganisms in order to neutralize acidic drainage. Sulphate reducing bacteria, anaerobic cellulolytic bacteria, aerobic cellulolytic bacteria, cellulolytic fungi, methanogens and heterotrophic fermentative organisms were found to be present in laboratory reactors, and their role in AMD mitigation was investigated in small laboratory microcosms (Béchard *et al.* 1993).

4.3 PERFORMANCE

The results of microbial reactor systems in treating AMD have been variable to date. All of the studies reviewed showed an increase in pH of the effluent relative to the influent in each system in the range 1.1 to greater than 3.0 pH units. Acidity reduction in the drainage ranged from 53% to 100%. Dvorak *et al.* (1991) have shown greater than 98% removal of Mn, Zn, Cd and Ni in the Palmerton bioreactor. At that site, zinc levels were reduced from 302 mg/L, well above the Canadian Metal Mining Liquid Effluent Regulations and Guidelines (MMLER) maximum monthly acceptable average concentration for zinc of 0.5 mg/L to 0.42 mg/L. Nickel concentrations were reduced from 0.85 mg/L to 0.03 mg/L (the MMLER maximum acceptable average concentration for nickel is 0.5 mg/L). Dvorak encountered excellent results and was able to sustain his system through the winter by the addition of sucrose as a carbon source. The results with other bioreactor systems have been less promising, as demonstrated in Table 4-2.

In addition to these result, Staub and Cohen (1992) reported metal removal from the Eagle Mine effluent of 28 up to 100%. The lower values were for manganese and zinc; while almost complete removal of copper, cadmium, lead and iron were achieved with a final effluent pH of 6.5-6.9. These pH values would meet the MMLER minimum acceptable grab sample pH of 5. The initial pH of the AMD was 2.6 to 3.0.

In the CANMET Biotrench study (Béchard *et al.* 1991), encouraging results were obtained over a 13 week period. Sulphate reduction was noted, aluminum concentration in the effluent water fell to undetectable values, and significant quantities of iron were removed. Weekly additions of 4 to 5 kg of sucrose and small amounts of urea promoted bacterial growth throughout the reactor. The reactor stopped functioning in early summer after new material was added. Oxygen became incorporated into the system, resulting in a loss of anaerobic conditions. The researchers were unable to re-establish the effectiveness of the biotrench the following spring.

Béchard *et al.* (1993) found that low solubility iron sulphides such as greigite and pyrite were produced by reactions within the system; elemental sulphur also accumulated in the laboratory reactors. Berner (1969), Farrand (1970), Sunagawa *et al.* (1971) and Sweeney and Kaplan (1973) have shown that mackinawite and greigite are precursors to pyrite framboids in the laboratory and in organic sediments. Sweeney and Kaplan (1973) have shown that elemental sulphur reacts with these immature crystals to bring about crystal maturation in the form of framboidal pyrite.

Nakamura's study (1988) ran for 113 days and was able to meet Japanese standards in the effluents at a flow rate of 720 L/d. The author stated however, that the cost of nutrients in his study were too high to be economically viable in large scale industrial use.

If the system is properly maintained and is effectively removing greater than 95% of each of the soluble metals from the AMD with a concurrent increase in solution pH to 6.5 or greater, the effluent from the microbial reactor system might be expected to be non-toxic from the perspective of metal concentrations. The presence of high BOD or H_2S could, however, influence effluent toxicity. With proper maintenance and replacement of the cellulosic substrate as it is metabolized, the system should be stable for an extended period of time. Although long term data is not available, it is anticipated that with proper maintenance, both the open and closed bioreactor systems could operate indefinitely; barring any radical change in the climatic conditions affecting the open bioreactor system.

4.4 APPLICABILITY

The pilot plant data presented in the literature indicates that bioreactors are a feasible technology for the treatment of small AMD streams and could be complementary to other passive or active AMD treatment systems. The overall applicability of either type of reactor in Canada will depend on the ability of the systems to treat higher flow regimes typical of Canadian mine sites.

Open bioreactor systems could be used at Canadian mines (most suited to coastal areas or areas with moderate winters) provided they are designed to prevent winter freeze-up, and have been oversized to allow the system to tolerate shock-loadings of the AMD during spring runoff. Closed bioreactor systems could be installed at any Canadian site where a fairly constant temperature can be maintained, either by installation underground (such as in underground mine workings), or in a heated building.

The only open bioreactor system in Canada that was reviewed was the channel system constructed at the Halifax Airport to treat acid rock drainage. However, this system shut down completely during a drought and did not effectively operate when the contaminated flow was reestablished. This system was partially successful, and indicated the need for additional research to make the technology industrially viable. Closed reactors have the advantage of being a contained system that can be insulated or placed within a building to ensure effective operation during the winter months. Both systems are a supplementary treatment system applicable to low flow AMD streams and are not suitable for high volume effluents from large tailings areas.

4.5 COST

For the purposes of presenting a cost estimate two idealized reactor systems have been described and costed. As demonstrated in Appendix A, alternative designs have been utilized, and would have different costs.

Case 1: Open Reactor System

The generic open reactor used for cost estimating purposes consists of three cells 20 m in length, 2 to 3 m in width, and 2 m in depth, separated by porous dams or clay berms with transfer pipes passing through the berm to facilitate migration of AMD from one cell to another. The AMD treatment cells should be preceded by an iron oxidation lagoon or a channel, 50 m long, filled with riprap to create turbulence in the AMD stream. This size and depth of cell should overcome the loss of anaerobic conditions experienced by Béchard. The cells should be lined with a 15 to 25 cm compacted clay lining, or an acid-resistant fabric liner to prevent seepage into the groundwater; filled with a mixture of alfalfa and manure (2:1), and inoculated with anaerobic mud containing SRB. This system should be capable of treating 50 to 60 L/min, with a 10 to 14 day retention time.

Cost Estimate

Excavation	(750 m^3 @ \$8.50/m^3)	\$ 6,400
Bentonite Mat	(1,500 m^3 @ \$10.00/m^3)	15,000
Riprap	(100 m^3 @ \$30.00/m^3)	3,000
Substrate	(100 m^3 (150 t) @ \$40.00/m^2)	6,000
Management	(Lump Sum)	3,100
ESTIMATE	**(Cdn \$)**	**\$ 33,500**

Case 2: Closed Reactor System

A closed system bioreactor could consist of a 38,000 litre closed, fibreglass tank. A metering pump and a solution distribution system will be required to introduce the AMD into the top of the reactor; effluent could be released by gravity flow through a discharge pipe. The reactor would be filled with a cellulosic substrate similar to that described for an open reactor system above, and inoculated in the same manner with SRB's. This size of tank should be able to accommodate a 75 to 100 L/minute flow rate with a 7 to 14 day retention time. An AMD storage/iron oxidation lagoon should precede the closed reactor system in order to provide iron removal and allow a constant flow rate to the reactor.

Cost Estimate

Fibreglass Tank		\$ 30,000
Metering Pump		5,000
Pipe	(100 m @ \$75.00/m)	7,500
Substrate	(140 m^3 (210 t) @ \$40.00/m^2)	8,400

Management	(Lump Sum)	5,100
ESTIMATE	**(Cdn $)**	**$ 56,000**

Significant cost savings can be obtained for the closed reactor system if it is constructed with clean, used equipment.

The cost of constructing an open bioreactor system is 40% less than the cost of an equivalent capacity closed bioreactor system. The cost of an open system will vary, depending on the soil type and location. If blasting is required to achieve the required depth of the open bioreactor the cost will escalate substantially. Closed bioreactor costs can be accurately determined, although the cost will increase if a building or insulation of the bioreactor is required. Maintenance costs for both reactor systems are not expected to be significant. Fresh cellulosic substrate additions will be required every 4 to 6 months and removal of precipitated metals from the sediments of the reactors will be required every 3 to 7 years depending on the metal loading present in the AMD. Monitoring costs will be identical for both systems.

4.6 OUTSTANDING ISSUES

Microbial reactor systems appear to have a significant potential for application to the Canadian mining industry, although at this time most of the biological reactor studies to date have been conducted at very low flow rates (< 1L/min). Further study will be required in order to be able to demonstrate whether there may be a broader application. If larger reactor designs with much higher flow rates are not developed, this technology is expected to have little or no application in the mining industry.

To date researchers have concentrated their monitoring efforts on the chemical and physical data and only Béchard (1993) has attempted to monitor the biological parameters of the system. Additional studies are required to understand the microbial interactions that occur to allow for the biological mitigation of AMD. The substrate for these organisms include short chain organic acids, fatty acids and alcohols, but to date it has not been confirmed that fermentative and cellulolytic organisms can generate these essential nutrients at a rate sufficient to sustain the anaerobes.

Future studies should place a greater emphasis on the engineering design of microbial reactor systems, and the optimizing of microbial activity involved in alkalinity generation. Such studies would not only bring about an optimization of the bioreactor systems, but would also provide insight into the development of an effective wetland system.

None of the researchers have tested these systems for sudden, large changes in AMD loadings. The bioreactors must be able to survive changes in flow rate, changes in the ionic composition of the AMD and changes in the AMD solution temperature.

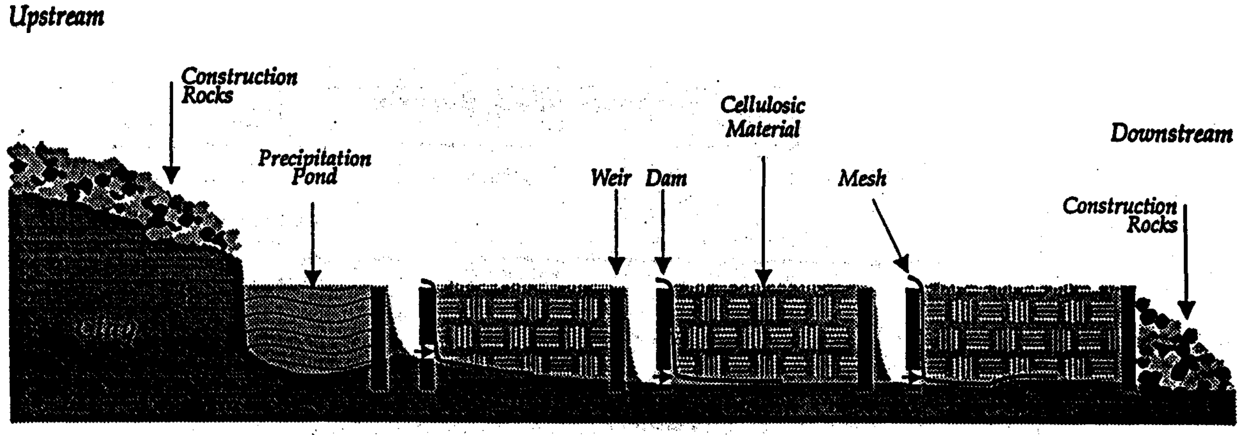

FIGURE 4-1: EXAMPLE OF TREATMENT OF ACID MINE DRAINAGE IN AN OPEN MICROBIAL REACTOR SYSTEM (From Béchard 1995)

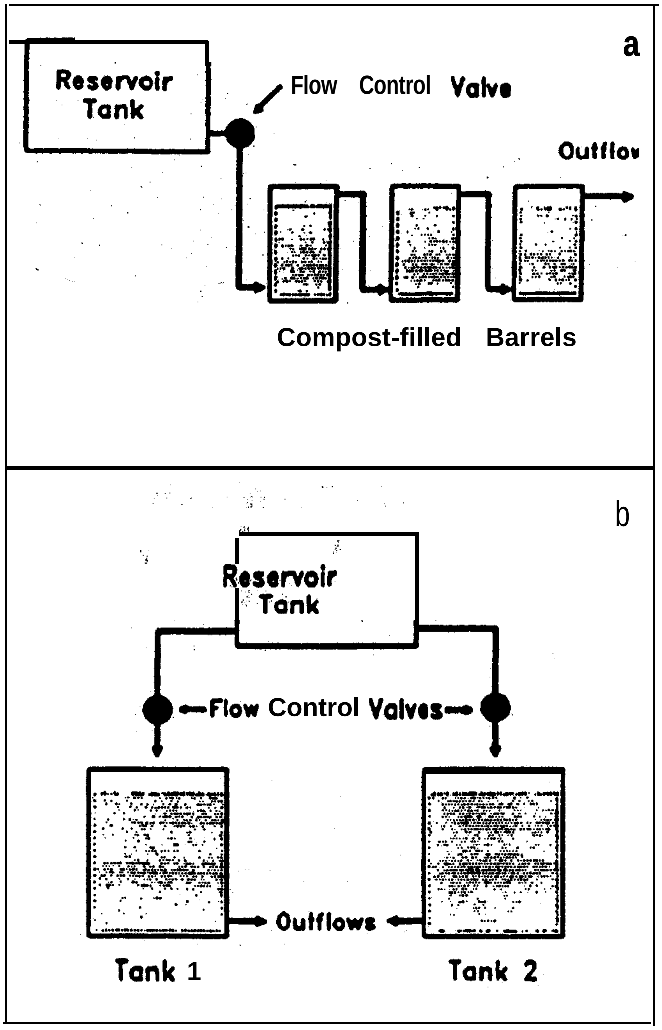

FIGURE 4-2: BLOCK DIAGRAMS OF THE (a) PITTSBURGH AND (b) PALMERTON CLOSED MICROBIAL REACTOR SYSTEMS
(From Dvorak et *al.* 1992)

TABLE 4-1: SELECT BIOREACTOR APPLICATIONS FROM LITERATURE

Reference	Location	Flow Rate	Reactor Size	Medium
Nakamura (1988)	Japan	0.5 L/min	0.2 m^3	Desulfovibrio Medium
Béchard *et al.* (1991; 1995)	Halifax	23 L/min	48 m^3	Straw/Wood
Dvorak *et al.* (1991)	Pittsburgh	0.055 L/min	0.6 m^3	Mushroom Compost
Staub *et al.* (1992)	Denver	0.4 L/min	1.5 m^3	Manure/Hay
Dvorak *et al.* (1992)	Palmerton	0.07 L/min	4.5 m^3	Mushroom Compost
Kalin (1993)	Makela	1 L/min	0.2 m^3	Flax Straw
Kuyucak and St-Germain (1994)	Mattabi	nd	0.002 m^3	$CaCO_3$/Pyruvate
Blowes *et al.* (1995)	Sudbury	nd	1.44 m^3	Leaf compost, bark, lime, sediment
Brickett (1995)	W. Pennsylvania	nd	nd	Mushroom Compost

TABLE 4-2: SUMMARY OF MICROBIAL REACTOR SYSTEM PERFORMANCE

Reference	Location	Influent		Effluent		Change (%)	
Dvorak *et al.* (1991)	Pittsburgh	Alk	0	Alk	349	Alk	+inf
		Acid	201	Acid	15	Acid	-93
		pH	3.2	pH	6.4	- -	- -
		Fe	53	Fe	8	Fe	-85
Dvorak *et al.* (1992)	Palmerton	Alk	17	Alk	1583	Alk	+9311
		Acid	520	Acid	1	Acid	-100
		pH	6.2	pH	7.1	- -	- -
		Mn	26	Mn	0.5	Mn	-98
		Zn	317	Zn	0.3	Zn	-99.9
		Cd	0.3	Cd	0.00	Cd	-99
		Ni	0.9	Ni	0.02	Ni	-98
Kalin (1993)	Makela (ARUMators)	Alk	146	Alk	3035	Alk	+2078
		Acid	1197	Acid	301	Acid	-75
		pH	5.37	pH	7	- -	- -
		SO_4	3663	SO_4	567	SO_4	-85
		Al	1.37	Al	3.47	Al	+253
		Cu	0.017	Cu	0.18	Cu	+1050
		Fe	281	Fe	1.15	Fe	-99
		Mn	11.1	Mn	15.8	Mn	+42
		Ni	60.6	Ni	0.6	Ni	-99
	Makela (Test Cells)	Acid	330	Acid	138	Acid	-58
		pH	3	pH	4.4	- -	- -
		SO_4	1905	SO_4	1620	SO_4	-15
		Al	21.6	Al	0.85	Al	-96
		Cu	4.9	Cu	0.57	Cu	-88
		Fe	13	Fe	20.7	Fe	+59
		Mn	4.4	Mn	7.5	Mn	+70
		Ni	40.9	Ni	6.9	Ni	-83
Kuyucak and St-Germain (1994)	Mattabi	pH	3.34	pH	6.23	- -	- -
		SO_4	4000	SO_4	2275	SO_4	-43
		Al	131	Al	117	Al	-11
		Cu	0.2	Cu	0.2	Cu	0
		Fe	133	Fe	91	Fe	-30
		Cd	2.4	Cd	0.06	Cd	-97

(from Kalin 1993; Table 2e)

5.0 BIOSORPTION SYSTEMS

5.1 INTRODUCTION

Biosorption systems are a treatment alternative that relies on the removal of metal ions from solution by adsorption/absorption to biological material. Micro-organisms, including bacteria, algae, fungi and yeasts, can efficiently accumulate heavy metals and radionuclides from their external environment. Metal ions are captured through processes involving ion exchange, complex formation and precipitation with living or dead cells. In addition to being a promising concept for bioremediation of acid mine drainage, biosorption is currently being successfully used to concentrate metals in place of more conventional methods such as ion-exchange resins (uranium - Cuif *et al.* 1989, Tsezos *et al.* 1989; gold - Torma 1986; cobalt - Kuyucak and Volesky 1986).

Most of the wetlands initially designed to detoxify AMD were constructed to mimic natural *Sphagnum* moss (peat) wetlands, because peat demonstrates significant heavy metal adsorption capacity (biosorption). Eger and Lapakko (1988) reported that a natural peatland in Minnesota removed 80% of the nickel and nearly 100% of the copper from tailings drainage. Similar experiences did not lead to such conclusive results in Canada. Kalin *et al.* (1990) reported that AMD with high heavy metal concentrations and extreme pH caused considerable damage to peat bog vegetation in Nova Scotia. *Sphagnum* moss is not always readily available, has proved difficult to transplant, and has tended to accumulate metals to toxic levels after several months of exposure to mine drainage (Hedin *et al.* (1994). As a result, ponds or drainage systems based only on the action of micro-organisms (living or dead) were developed. Some organisms used as biosorbents to treat elevated heavy metal concentrations in mine drainage are listed in Table 5-1.

Biosorption systems rely on either living cells or non-living biomass. Intimate contact of the influent with the biosorbents must be maximized to promote adsorption/ion exchange reactions. Living cells can be used to treat effluent where metal concentrations are below toxic levels. Although some heavy metals are essential for micro-organisms metabolism at low concentrations (Cu, Zn, Mn), others have no known essential biological function (Au, Ag, Cd, Pb, Hg, Al) and can be highly toxic. The use of dead biomass eliminates the problems of metal toxicity, adverse climatic conditions, and the costs associated with nutrient supply and culture maintenance. The metabolism-independent binding of metal species occurs at the cell walls for non-living cells. The cell walls of bacteria, algae and fungi are efficient metal biosorbents and initial binding may be followed by inorganic deposition of increased amounts of metal, even up to 50% of the dry weight (Gadd 1990). Figure 5-1 provides a schematic representation of a typical biosorption system.

Most of the potential techniques for biological treatment of metal contamination have been developed at the bench-scale level. Few field studies on treatment of AMD by biosorption are currently available in the literature. The studies reviewed are summarized in Table 5-2 and in Appendix A.

5.2 TECHNOLOGY DESCRIPTION

5.2.1. Living Biosorption Systems

Living micro-organisms have been used in a number of studies to treat AMD. In laboratory experiments, microbial mats immobilized on glass wool were used to remove zinc and manganese from contaminated waters (Vatcharapijarn *et al.* 1994). Duggan *et al.* (1992) utilized an algal mixture, primarily composed of *Cladophora* (green algae) collected from a local pond, plus limestone, to remove manganese from solution and raise the pH of mine drainage waters. Smith and Kalin (1989) utilized an "underwater meadow" of *Nitella flexilis* (an attached macrophytic algae), in a lake receiving water leaving a uranium mine, to act as a filter which through its continuous growth and decay, relegates the contaminants Ra^{226} and to a lesser extent U, from the water to sediments. This natural treatment system is referred to as the Chara process. Tsezos (1987, in Barnes *et al.* 1991) utilized algae ponds containing *Spirogyra* sp., *Chara* sp., *Oscillatoria* sp. and other green algae to reduce the concentrations of uranium, molybdenum and selenium in mine water in a field test.

Microbial mats composed of heterotrophic and autotrophic micro-organisms (cyanobacteria - blue-green algae - green filamentous algae) have been used in constructed ponds (40 - 46 m^2) to reduce the dissolved manganese and iron concentrations from mine drainage (Bender and Phillips 1994; Phillips *et al.* 1994; Wildeman *et al.* 1993). The micro-organisms are held together by slimy secretions, that fix the ecosystem to a variety of substrates. Since these mats are both nitrogen fixing and photosynthetic, they are self-sufficient solar-driven ecosystems with few growth requirements. In the pilot pond studies using microbial mats only low flow rates of 2 to 5 L/min were treated.

Bender and Phillips (1994), Phillips *et al.* (1994), and Wildeman *et al.* (1993) reported the use of microbial mats (mainly the blue-green algae *Oscillatoria* sp., green filamentous algae and *Chromatium* sp.) in small ponds (40 - 46 m^2) to reduce the dissolved concentrations of manganese and iron in mine drainage. Cyanobacteria - algal mats were harvested on-site, developed into silage-microbe mats in the laboratory, and applied to constructed ponds. The ponds were excavated to 30 cm depth and lined with PVC film. Limestone (2 to 3 cm in diameter) was added to develop alternating high and low areas, ranging from 2 to 30 cm in depth, spaced 1 m apart. The pond was fed effluent by gravity; its rate controlled by valves to between 2 and 5 L/min.

A microbial mat slurry (blended in water) was broadcast over the pond in three applications of 1 to 1.5 L during a four week period. Silage prepared from grass clippings added organic acids and several species of bacteria to the water. Within two months, a thick green mat covered the entire pond surface and the limestone below. Thus, the metal-contaminated water essentially flowed between a double layer of mat. Parameters measured in the pond included flow rate, dissolved O_2 concentrations, pH, Eh, conductivity, and Fe and Mn concentrations in both the influent and effluent waters.

5.2.2. Non-living Biosorption Systems

The utilization of non-living biomass to lower metal concentrations in mine drainage is also reported in the literature. Commercially available biosorbent materials are based on inactivated biomass (Table 5-3). When commercial biosorbent materials are used, it is crucial that flow be controlled to permit a tight exchange between the dry biomass and the AMD.

One of the biosorbent materials used to treat AMD is BIO-FIX (biomass - foam immobilized extractant). BIO-FIX beads are fabricated from dried non-living biomass blended into a high-density polysulfone dissolved in an organic solvent. BIO-FIX beads are prepared from commercially available raw materials such as: marine algae (*Ulva* sp.), blue-green algae (*Spirulina* sp.), yeast (*Saccharomyces cerevisiae*), common duckweed (*Lemna* sp.), and finely ground peat (*Sphagnum* peat moss). The beads can be enclosed in meshed polypropylene bags and placed either in a trough or directly into a waste stream, or in conventional equipment (stirred tanks, fixed-bed columns, fluidized-bed columns). Contaminants removed from effluent included: As, Cd, Cu, Ag, Pb, Mn and Zn. The beads may be especially useful for dilute effluent containing metal concentrations of less than 15 mg/L. Bennett *et al.* (1991) and Jeffers *et al.* (1989) suggested the use of BIO-FIX to extract heavy metals from dilute waste streams.

Bennett *et al.* (1991) conducted a field test with a seventeen compartment trough, 2.1 m long by 30.5 cm wide by 40.6 cm deep, containing bags of BIO-FIX beads. Weirs were positioned such that wastewater from the inactive gold mine alternatively flowed down through a bag under a weir, then up through the adjacent bag, and over the next weir. The bags at the head of the trough (which were most fully loaded with metals), were periodically removed for regeneration to ensure continued effectiveness for metal extractions. The bags immediately below the first bags were moved up into the front compartments. This procedure continued until all bags were repositioned and freshly eluted bags were placed in the lower sections of the trough.

Médiaflex$_{MC}$ is a biofilter composed of *Sphagnum* moss and of Biofil, a carbon material having adsorption and chemical precipitation properties. Médiaflex has been tested in the field in Canada to treat different types of wastewater, including AMD effluents, in order to remove heavy metals such as Fe, Zn, Cu, Ni, Cr and Cd (CQVB 1993, Bélanger *et al.* 1995). Bélanger *et al.* (1995) reported the use of a large Médiaflex$_{MC}$ biofilter (15 x 36 m - 540 m^2) in a basin having a capacity of 6,000 m^3 receiving effluent from a now closed copper mine. The flow was set to approximately 250 m^3/d.

Darnall *et al.* (1989) suggested the use of a preparation of immobilized algae (AlgaSORB II), prepared from *Chlorella vulgaris*, to recover such metals as copper, mercury or cadmium from contaminated waters. Levels of these metal ions can be reduced in concentration from low ppm levels to low ppb levels.

Other biosorbents have also been proposed, including: calcium-alginate beads (using algin extracted from species of brown seaweed) to remove Cd, Ba, U and Zn from uranium mine tailing waste water (Barnes *et al.* 1991); chitosan (a natural biopolymer from chitin: McKay *et al.* 1986); and immobilized algal biomass (*Spirulina* sp.) in preformed polyurethane foam (Fry and Mehlhorn 1994).

5.2.3. Maintenance Requirements

No data was found in the literature reviewed regarding long term maintenance for biosorption systems designed to treat AMD. The scarcity of case studies and their short duration do not permit an accurate identification of maintenance and monitoring requirements for biosorption systems. Several living ecosystems used are reported to be "self-sustaining and maintenance-free" (Smith and Kalin 1989). Nutrients are sometimes provided to these treatment systems, but they are typically designed to be inexpensive and to require little maintenance. Constructed microbial mats are reported to be durable, highly resilient, self-sustaining ecosystems with few requirements for good growth. Once established, the mat persists for long periods without nutrient supplements (Bender and Phillips 1994). Algae are being tested to treat other types of wastewater, so knowledge acquired there can be used in AMD decontamination (de la Noüe *et al.* 1992).

Despite this, it is reasonable to suppose that biological ponds will require periodic maintenance. In the long term, systems will become clogged with metal precipitates, or the conditions that facilitate contaminant removal may be compromised.

BIO-FIX beads require frequent eluting and recharging to ensure continued effectiveness for metal extractions. Removal of metals from BIO-FIX beads is accomplished using dilute mineral acids in order to ensure efficient removal, and the ability to use the beads for repeated extraction-elution cycles (Jeffers *et al.* 1989). The higher the concentration of heavy metals in the influent, the more often the biomass must be eluted to remove the metals. These eluted metals have to be disposed of, most frequently as hydroxides which requires the use of lime or sodium hydroxide.

New peat substrate can be added to constructed wetlands to enhance their performance to remove heavy metals and prolong their efficient life by providing new adsorption sites. Similarly, in biological ponds, substrate (organisms) will eventually have to be removed and replaced by fresh materials. Enhancing the treatment cells with new substrate provides a lower metal content and therefore higher metal uptake potential than the old substrate. This material can be replaced when the metal adsorption/absorption capacity is exhausted, to increase the life of the system (Eger and Melchert 1992).

There is little reported information on disposal of the contaminated biomass after saturation. Used microbial mats in constructed ponds are expected to require periodic removal and disposal; however, no information was given in the literature concerning this point. The short duration of the studies conducted to date do not permit prediction of how often this used biomass will require removal and disposal.

When algae eventually die after prolonged metal exposure, they fall to the bottom of the treatment cells and become gradually buried, thereby potentially immobilizing the metals for long periods. A feature of internally accumulated metals (living biomass) is that their extraction or recovery is difficult. Potentially expensive physical or chemical treatments, including acid extraction or incineration of the biomass, may be required (Gadd 1986). With dead biomass, the surface bound metals are easily removed by relatively simple procedures which do not adversely affect the biomass, which can then be re-used. After elution of the BIO-FIX beads, eluates can be recycled at each successive elution and become progressively enriched with metals. The acid solution product eluates can be further processed using conventional hydrometallurgical techniques such as ion exchange and solvent extraction if the products have any economic value (Jeffers *et al.* 1989); otherwise, the eluate must be appropriately disposed. Contaminated biomass, and eluates from the acid washing of dead biomass, are the final waste products of biosorption systems. The quantities, metal concentrations, toxicity and frequency of waste disposal are not known due to the lack of any long-term study.

5.3 PERFORMANCE

No data was found in the literature reviewed regarding long term performance for biosorption systems designed to treat AMD. The relatively few field studies reported do not permit generalizations about water quality results or the potential toxicity of effluent released from these systems. The total amount of metal removal occurring in the treatment cells is a function of the input water quality, residence time, influent temperature, the distribution of flow within the cell, and the capacity of the organisms to remove metals from the solution.

Concentrations of a large number of metals have been reported to be reduced using biosorbents (As, Ba, Cd, Cu, Fe, Hg, Pb, Mn, Mo, Ra, Se, Zn, U). The few field studies available regarding biosorption and mine drainage bioremediation focused primarily on manganese removal, and on metals released from uranium mine tailings (Mo, Ra, Se, U). Manganese is a contaminant common to most metal and coal mine drainage. Mn^{2+} is difficult to remove from solution without chemical treatment due to the high pH required to form insoluble manganese oxides, carbonates or sulphides. Soluble manganese forms encrustations (black precipitates) in the algal mat.

In the studies reported in Table 5-2, algal mixtures were effective in reducing dissolved manganese levels and raising the pH of the influent. Algal biomass in most passive treatment systems such as constructed wetlands is, however, very limited, so its contribution to metal removal will rarely reach significant levels. A very productive algal system accumulating 50,000 mg/kg manganese would only remove 4 mg/L of manganese from solution, assuming currently-sized wetland systems (Hedin 1989, in Kleinmann 1989).

Metal precipitation is believed to be a least partially due to the increase in pH and production of O_2 during photosynthesis, associated with algal growth, as the precipitation of Mn oxides usually requires a pH between 8 and 9. Primary Mn removal processes are thought to be: adsorption to algal cell walls, auto-oxidation due to photosynthesis, and adsorption to Mn oxides that have previously formed. The formation of manganese oxides, specifically MnO_2, is a desirable Mn removal method due to the extreme insolubility and large sorption capacities of manganese oxides (Duggan *et al.* 1992). An increase in pH due to algal growth in tailing ponds has been observed at some sites.

BIO-FIX beads are also well adapted for secondary treatment or polishing of effluent, since they readily absorb metals from the near-neutral solutions released from precipitation processes (Jeffers *et al.* 1989). Immobilized dead biomass biosorbents tolerate low pH and high temperatures better than conventional ion exchange materials, and offer a higher loading capacity at low metal contaminant levels (Smith *et al.* 1994).

The removal of toxic heavy metals and radionuclides from liquid wastes (as in AMD) by biosorption can result in effluent detoxification and therefore safe environmental discharge in compliance with government standards for environmental protection (Gadd 1990).

5.4 APPLICABILITY

Passive treatment systems using biosorbents to lower metal concentrations in AMD are of value as a form of secondary treatment, when integrated with other types of treatment systems. An algal mixture containing *Cladophora* may be used as a second stage process to remove manganese and raise the pH of water that has passed through a constructed wetland treatment (Duggan *et al.* 1992). In Bender and Phillips (1994), the biosorption system polishes the effluent released from a system including an anoxic limestone drain, oxidation pond, and trickle filters to remove oxidized iron. If appropriate, limestone may be added in the microbial/algal mat passive treatment system to help increase pH.

Biosorption technology using living organisms (such as algae, bacteria) may be useful in some circumstances as a secondary form of treatment, depending on the concentration and toxicity of metal(s) in a given effluent. They may have use in processes where large volumes of contaminated effluents with low concentrations of metals must be treated in holding ponds.

Due to the severity of the winters in Canada, the success of biosorption systems using living organisms year round is doubtful, and will likely require treatment support from other more conventional methods. Living organisms are not expected to maintain good growth during the cold season, and hence treatment efficiency as a biosorbent will be lowered. Biosorption systems may be applicable at certain locations where constructed ponds are deep, algal growth is sustained during the winter, and the ponds do not freeze to an extended depth; or at sites where an alternative form of treatment is used during cold weather. Duggan *et al.* (1992) reported visible effects of winter temperatures on the health of the algal biomass; most algal cells sank below the water surface. Philips *et al.* (1994) reported that a viable algal mat was maintained throughout the winter, but the temperatures experienced during that study were mild (4-6°C) when compared to the severity of typical winters in Canada. Nordic strains of cyanobacteria isolated from arctic climates can possibly be developed having potential as biofilters and for alkalinity generation, even under winter conditions.

Dead biomass also appears to offer more potential as a secondary form of treatment in Canada. Biosorption technology employing immobilized biomass in a column system analogous to ion exchange may be applicable for effluent polishing, or for recovery of low level metal values from process streams. In regions of Canada which endure long, harsh winters, it may be feasible eventually to treat drainage biologically without wetlands or living biomass ponds, as in a pipeline-type reaction vessel filled with dead biomass (Kleinmann 1989). Duggan *et al.* (1992) reported that non-living algae retained a high affinity for metal adsorption during the winter, as long as flows are maintained. A test conducted at 4°C with BIO-FIX beads to determine the effect of temperature on metal loading showed that a reduction in solution temperature did not significantly decrease metal sorption. Bennett *et al.* (1991) suggested the potential of using BIO-FIX beads for treating isolated waste streams and seeps during the winter months.

5.5 COST

The field studies consulted did not provide any information about cost of the preliminary treatment systems, either for construction or operation and maintenance. In addition, the short time duration of the experiments and the limited scale does not provide sufficient information to develop detailed cost estimates for a full-scale treatment system. Economy could probably be realized over a more conventional chemical treatment system, only if the maximized operated system is self-sustainable for long periods without need for additional treatment (addition of nutrients, removal/disposal of biomass).

5.6 OUTSTANDING ISSUES

There are many data gaps in the literature regarding the use of biosorption systems to treat AMD. There are few field studies, and those available are of short time duration and low flow rates. Little is known about costs, life expectancy, performance records, maintenance requirements or the potential toxicity of effluent released from these systems.

Much of the research to date has documented changes in water quality without a detailed study of the processes. As a result, it is difficult to scale up, predict effective long-term performance (>100 years), or optimize wastewater treatment (Kleinmann 1989). Depending on the quality of the influent, a biosorption system may or may not produce an effluent which meets the Canadian Metal Mining Liquid Effluent Regulations and Guidelines. It does not appear that biosorption systems are an effective stand-alone treatment system for AMD.

At this time, the greatest requirement is the acquisition of additional information, preferably at a field scale, under Canadian or similar climatic conditions. The lack of data currently available ensures that virtually any data collected will be useful. This might include:

- effect of climate on the living biomass;
- potential for treatment of effluent with high metal concentrations;
- design criteria for pilot scale or full scale applications;
- efficiencies of pilot or full scale systems;
- applicability of biosorption components to other passive treatment systems, such as constructed wetlands;
- design criteria to use immobilized biomass in a column system (akin to ion exchange);
- operation and maintenance requirements; and,
- system life and waste disposal requirements.

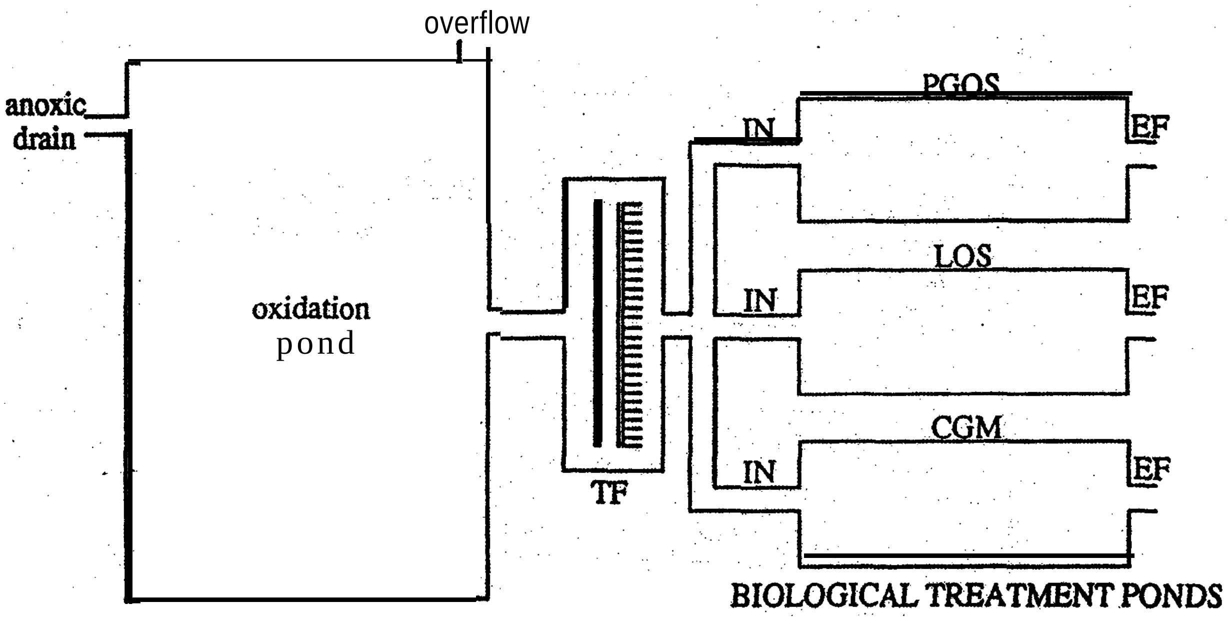

FIGURE 5-1: SCHEMATIC REPRESENTATION OF A BIOSORPTION SYSTEM (TF -trickling filter; IN - influent water; CGM - Cyanobacteria-algae mat pond; LOS - limestone-Oscillatoria pond; PGOS - pea gravel-Oscillatoria; EF - effluent water. Modified from Phillips et al. 1994).

TABLE 5-1: ORGANISMS USED AS BIOSORBENTS TO TREAT MINE DRAINAGE CONTAMINATED WITH HEAVY METALS

Living Organisms	Reference
Microbial mat	Bender and Phillips 1994 Phillips *et al.* 1994 Wildeman *et al.* 1993 Vatcharapijarn *et al.* 1994
Cladophora sp.	Duggan *et al.* 1992
Nitella flexilis	Smith and Kalin 1989
Spirogyra sp. *Chara* sp. *Oscillatoria* sp.	Tsezos 1987 (cited in Barnes *et al.* 1991)

Dead Biomass	Reference
BIO-FIX (*Sphagnum* moss, others)	Bennett *et al.* 1991 Jeffers *et al.* 1989
Médiaflex$_{MC}$	CQVB 1993
Chitosan	McKay *et al.* 1986
Alga SORB II *Chlorella vulgaris*	Darnall *et al.* 1989
Spirulina sp. in polyurethane foam	Fry and Mehlhorn 1994
Calcium-alginate	Barnes *et al.* 1991

TABLE 5-2: SUMMARY OF BIOSORPTION FILED EXPERIMENTS FOR ACID MINE DRAINAGE

Organisms	Metal	Experiments	Results
Algal mixture of mainly filamentous green algae + limestone Duggan *et al.* (1992)	Mn	Bench scale study: 2, 97 L reservoirs Static System Influent: pH 5.8, Mn 32 mg/L Flow System influent: pH 6.3, Mn 30-100 mg/L, flow of 2-4 mL/min	Effective in reducing Mn and substantially raised the pH. Static effluent: pH > 8.0, Mn < 0.3 mg/L Flow effluent: Mn < 5 mg/L, removal rate: 0.19 g/d/m^2
Cyanobacteria-algal mat pond + limestone Wildeman *et al.* (1993)	Mn (Fe)	Pilot pond study (46 m^2) Influent: pH > 6.0, Fe 1-4 mg/L, Mn 3-7 mg/L	Reduced Mn concentrations; removal complete at 3.3 L/min Effluent: Mn removal at 2.5 g/d/m^2, pH > 7.0
Algal mat consortium (mainly blue-green algae) + limestone Phillips *et al.* (1994)	Mn (Fe)	Pilot scale field test Pond: 40 m^2 Flow: 4.2 L/min	Mn and Fe efficiently reduced, even in winter (4 to 6EC), Mn removal rate of 2.59 g/d/m^2
Nitella flexilis "Chara" process Smith and Kalin (1989)	Ra^{226} U	Lake downstream of a U mine pit. Biomass 100 to 1,000 g/m^2. Water depth 0.6 to 1.9 m.	Estimated to reduce annual loading of Ra^{226} by 41 to 100% and of U by 1.5 to 3.2%. Relegate them from water to sediments.
BIO-FIX beads enclosed in meshed polypropylene bags Bennett *et al.* (1991)	Mn Fe	Field test of a multi-compartment trough with 16 bead bags Influent: pH 5.5, Fe 40 mg/L, Mn 3.0 mg/L	Bag rotation scheme utilizing 6 bags (36 L of beads) every 96 h. Effluent: Fe < 10 mg/L, Mn < 0.05 mg/L
Médiaflex$_{MC}$ Bélanger *et al.* (1995)	Cu Fe Ni Pb Zn	Pilot scale field test 6,000 m^3 of influent treated (~250 m^3/d) Cu 14 ppm, Fe 88 ppm, Ni 0.8 ppm, Pb 0.9 ppm Zn 108 ppm, pH 3.7-4.4	Effluent: pH > 8.8, reduced Cu and Fe concentrations > 99%, 93% for Zn, 82% for Ni, 70% for Pb.

TABLE 5-3: CHARACTERISTICS OF COMMERCIAL BIOSORBENTS

Product	Developer	Biosorbent Type	Support Material	Metals Treated
MRA	AMT-BIOCLAIM	Caustic treated killed bacteria	Polymer	Cationic and anionic form
Alga SORB	Bio-Recovery	Algae	Silica gel	Cationic metals
BIO-FIX	US Bureau of Mines	Peat moss, *Spirulina* and others	Polysulfone polymer	Cationic metals
Médiaflex$_{MC}$	Serrener Consultation Inc.	*Sphagnum* peat moss	Biofil$_{MC}$	Cationic metals

AMT = Advanced Mineral Technology
BIO-FIX = Biomass Foam Immobilized Extractant
MRA = Metal Removal Agent

(Adapted from Smith *et al.* 1994)

6.0 DISCUSSION

6.1 APPLICABILITY OF PASSIVE TREATMENT SYSTEMS TO CANADIAN MINES

Based on the information currently available relating primarily to treatment of coal drainage, anoxic limestone drains are an effective, inexpensive form of passive alkalinity addition for acid mine drainage having dissolved oxygen, ferric iron and aluminum concentrations below 1 mg/L, and sulphate concentrations less than 2,000 mg/L. As a result of the strict influent guidelines, ALD's may have only a limited application for Canadian mine sites. At sites which have elevated concentrations of these parameters, ALD's may still provide an inexpensive method of alkalinity addition over the short term. At some sites, cost benefit analysis may show that despite periodic reconstruction after failure, ALD's may continue to be a cost effective solution in the long term.

Wetland technology holds promise as a technically viable and economically attractive technology for treating AMD. Its application as a recommended treatment method in Canada will require the development of data under conditions that are unique to Canadian sites. It is accepted that constructed wetlands are less expensive to construct and operate than active treatment systems; however, this is true only if the wetland systems operate efficiently, and effluent does not require additional chemical treatment. The existing design approach is not recommended. Systems for Canadian operations need to be developed based on an understanding of the interactions between the chemical, microbial and plant-mediated components in the system, rather than the traditional black-box approach.

Microbial reactor systems are expected to be most effective for the treatment of low flow AMD streams such as seeps from waste rock storage areas. Under optimal conditions microbial reactor systems could produce an effluent that would meet environmental guidelines. The applicability and efficiency of open bioreactors may be constricted in part by climatic considerations. If designed accordingly, closed systems are unaffected by climate, and are able to operate year round.

Biosorption technologies are not far enough developed or proven to fully assess their applicability to Canadian conditions and Canadian mine sites. More data from pilot scale and controlled studies are required. At this time biosorption systems using non-living material are expected to have the greatest applicability, since living organisms are unlikely to maintain good growth and treatment efficiency under severe winter conditions. It is not expected that biosorption systems will develop into a stand-alone treatment system, although with continued research they may develop into a final or intermediary polishing system for acid mine drainage having low metal concentrations.

Emerging technologies including: *in situ* bacterial treatment (Brickett 1995; Christensen *et al.* pers. comm.); porous reactive walls (Blowes *et al.* 1995); biological sulphate reduction (Mine Waste Technology Program; MWTP nd), remote mine site technologies (MWTP nd); and successive alkalinity-producing systems (Kepler and McCleary 1994) may be applicable, however insufficient information is available at this time to assess their ability to treat Canadian acid mine drainage.

6.2 ABILITY TO MEET CANADIAN EFFLUENT REGULATIONS

The primary federal legislation relating to acid mine drainage is the Metal Mining Liquid Effluent Regulations (MMLER). Effluent quality standards are listed for seven parameters: arsenic, copper, lead, nickel, zinc, total suspended solids and radium 226, along with a lower limit for pH. Provincial and other regulatory agencies use these standards, or else more stringent criteria. The ability of passive treatment systems to consistently meet the MMLER is a function of the quality of the influent and the effectiveness of the system. Very little information was available in the case studies reviewed regarding the toxicity of effluent. However, it will also be a function of influent quality and system effectiveness. Nevertheless, since toxicity is frequently related to ammonia, organics and cyanide the production of a non-toxic effluent may not be assured by the use of passive treatment methods.

ALD's are not a stand-alone treatment method for AMD. Their function is to add alkalinity to the effluent stream, which may in turn improve the effectiveness of downstream treatment systems, and cause precipitation of metal hydroxides. In this manner, ALD's may improve the quality of effluent released from a site; however, the chemical composition and toxicity of the ALD effluent itself is highly dependant on the quality of the influent. Little information is available regarding the metals of most concern at Canadian mine sites, since the majority of data available to date is related to U.S. coal mines. As a result, the ability of ALD effluent to meet Canadian water quality criteria is uncertain.

Twelve of the wetlands managed by the Tennessee Valley Authority consistently met the U.S. National Pollution Discharge Elimination System's (NPDES) monthly average discharge limits without chemical treatment. Unfortunately, most of the parameters regulated under the MMLER were not monitored at the full-scale constructed wetlands reviewed (see Appendix A). Average discharge values had a pH of 6 to 9 and total suspended solids (TSS) <35 mg/L. At the Fabius Impoundment 1 the pH was increased from 3.5 to 6.8 (Brodie 1993). The MMLER for these parameters (acceptable concentration in a grab sample) are: pH, minimum of 5.0; and TSS, maximum of 50 mg/L. Eger *et al.* (1991) did show reductions in concentrations of nickel and zinc in the influent to meet the MMLER of 1 mg/L for each parameter (prior to treatment concentrations were up to 7.98 mg/L for nickel and 1.96 mg/L for zinc), using wetland cells over a short monitoring period and at a pilot scale.

The results of microbial reactor systems in treating AMD have been variable to date, although a number of the case studies reviewed show promise of meeting the MMLER. All of the studies reviewed showed an increase in pH of the effluent relative to the influent in each system in the range of 1.1 to greater than 3.0 pH units. Acidity reduction in the drainage ranged from 53% to 100%. Dvorak et at. (1991) have shown greater than 98% removal of Mn, Zn, Cd and Ni in the Palmerton bioreactor. At that site, zinc levels were reduced from 302 mg/L (well above the MMLER maximum monthly acceptable average concentration for zinc of 0.5 mg/L) to 0.42 mg/L. Nickel concentrations were reduced from 0.85 mg/L to 0.03 mg/L (MMLER maximum acceptable average concentration for nickel is 0.5 mg/L). Staub and Cohen (1992) reported metal removal from the Eagle Mine effluent of 28 to 100%; the lower values were for manganese and zinc, while almost complete removal of copper, cadmium, lead and iron were achieved with a final effluent pH of 6.5-6.9.

No data was found in the literature reviewed regarding long term performance for biosorption systems designed to treat AMD. The relatively few field studies reported do not permit generalizations about water quality results or the potential toxicity of effluent released from these systems, although concentrations of a large number of metals have been reported to be reduced using biosorbents (As, Ba, Cd, Cu, Fe, Hg, Pb, Mn, Mo Ra, Se, Zn, U).

6.3 MAINTENANCE AND MONITORING REQUIREMENTS

Maintenance and monitoring requirements for passive treatment systems are generally less than comparable conventional chemical treatment plants. ALD’s require the least maintenance and monitoring since they do not have biological components. Biological-based passive treatment systems may require periodic replacement of components once metal saturation levels are reached, or metal precipitates clog the system.

Details regarding the maintenance and monitoring requirements for each of the systems described in this document are provided in the individual chapters and outlined in Table 6-1.

6.4 LIFE EXPECTANCY AND LONG TERM IMPLICATIONS

Insufficient information is available for passive treatment systems to fully assess their potential life expectancy. Limited information is available regarding handling and disposal of the potentially toxic waste products created by concentrating metals in effluent in: aerobic ponds downstream of ALD’s, wetland plants and substrate, cellulosic substrate of microbial reactors, or biosorption materials (or elute for non-living materials). Some generalities regarding long term performance can however, be made at this time.

The research conducted to date suggests that ALD’s can be expected to be effective for over 20 years (Brodie *et al.* 1992) and perhaps even longer, up to 100 years. Eventually the volume of limestone available will be exhausted, the surface may subside, or channelized flow may be initiated, rendering the system ineffective. The potential for retention of metals and development of hydroxides under low dissolved oxygen conditions (<1 mg/L) over a long period of time (>100 years) has not been assessed, and thus, the extrapolation of limited data over such a long period of time may be presumptuous.

With proper maintenance and replacement of the cellulosic substrate as it is metabolized, microbial reactor systems should be stable for an extended period of time. No long term data is available, although it is anticipated that with proper maintenance, both open and closed bioreactor systems could operate indefinitely; barring any radical change in the climatic conditions affecting the open bioreactor system.

Constructed wetlands are designed to be efficient accumulators of metals and reaction products, which are stored in the substrate and biomass. Unlike the wetlands designed to remove organic pollutants such as in the treatment of sewage where they can function well for more than 20 to 30 years before requiring any major construction works, AMD wetlands will reach toxic levels much sooner, rendering their long-term applicability (>100 years) doubtful.

6.5 ALTERNATIVE TECHNOLOGIES

While this document has addressed the potential for passive treatment of acid mine drainage, it is not the only technology available. Considerable research has been conducted for, or by, MEND and other agencies with respect to management of AMD. The end goal is to reduce the environmental liability associated with mining in Canada.

Two primary technological approaches to acid mine drainage are possible: prevention and control, and treatment. Each of these are considered briefly below.

6.5.1. Prevention and Control

The consensus of operators and reported in the literature is that prevention of AMD is the preferred approach. Prevention incorporates any measures designed to eliminate or reduce the impact of acid generation, prior to its initiation. This may take place during any stage of mine development, but may be most practical during the feasibility and design stages of the project.

Prevention can occur by eliminating one of the elements essential for acid generation to occur: exclusion of water; exclusion of oxygen; removal or isolation of sulphides; maintaining frozen conditions; pH modification; and control of bacterial action (SRK 1989).

Examples of technologies investigated by MEND are reported in:

- subaqueous disposal (lake disposal, man-made impoundments, submarine disposal) *(Study 2.11.1d, A Critical Review of MEND Studies Conducted to 1991 on Subaqueous Disposal of Tailings, July 1992);*
- wet barriers (flooding of existing tailings sites, in-pit disposal, flooded mine working) *(Study 2.15.1, Study on Water Covers for Decommissioning of Mine Waste Disposal Sites, Active Project);*
- dry covers (soil, synthetic membranes, organic material) *(Study 2.20.1, Evaluation of Alternate Dry Covers for the Inhibition of Acid Mine Drainage from Tailings, March 1994);*
- thickened tailings disposal *(Study 2.23.2ab, Hydrologic and Hydrogeologic Evaluation of the Thickened Tailings Disposal System at Kidd Creek, October 1993);*
- porous envelope surround *(Study 2.23.3, Investigation of the Porous Envelope Effect at the Fault Lake Tailings Site, May 1995);*
- microbial plugging *(Study 2.44.1, Microbial Plugging of Uranium Mine Tailings to Prevent Acid Mine Drainage, December 1992);*
- separation of sulphides *(Study 2.45.1a, Separation of Sulphides from Mill Tailings, June 1994);* and,
- storage in permafrost *(Study 6.1, Preventing AMD by Disposing of Reactive Tailings in Permafrost, December 1993).*

6.5.2. Treatment

Collection and treatment of AMD may be the preferred approach available for some sites, particularly operating or abandoned mines. Treatment may be either an active or passive form; the primary difference is whether energy is supplied directly to the treatment process. Passive treatment systems generally rely on energy supplied by the environment from sunlight, and/or gravity to operate, and require limited intervention by man. Passive treatment systems including: anoxic limestone drains, constructed wetlands, microbial reactor systems and biosorption systems are considered in detail within this document.

Considerable research has been completed in MEND-related studies regarding passive treatment for AMD. The majority of this information was not in a case study form amenable for use in preparation of this document. Some of the key MEND studies related to passive treatment that provided early technical support to some of the case studies described in this report, are:

- *Study 3.12.1, Assessment of Existing Natural Wetlands Affected by low pH, Metal Contaminated Seepages, May 1990;*
- *Study 3.11.1, Treatment of Acidic Seepages using Wetland Ecology and Microbiology, July 1993;* and,
- *Study 3.12.2, Panel Wetlands - A Case History of Partially Submerged Pyritic Uranium Tailings Under Water, February 1993.*

Active treatment most commonly relies on the addition of alkaline agents to ameliorate the effects of AMD. Once neutralized, heavy metal ions hydrolyze and precipitate out of solution. Lime (slaked) and ground limestone are the materials most commonly used to neutralize acid mine drainage. Other materials that can be used include: caustic soda, soda ash, ammonia, and sodium sulphide. One of the studies completed by MEND, reviewed the current status of chemical treatment of AMD *(Study 3.32.1: Acid Mine Drainage - Status of Chemical Treatment and Sludge Management Practices, June 1994)*.

The primary issue relating to both active and passive treatment systems is the disposal of sludge which develops as metals precipitate out of solution. The sludge may contain gypsum, heavy metal hydroxides, heavy metal arsenates, calcium arsenate and, heavy metal sulphides. As such, the stability of the material and toxicity in the long term is an important issue.

One aspect of treatment technology towards which future research could be directed is the combining of passive and active technologies to treat a single effluent stream. Each type of treatment technology has unique advantages, which if used together could provide an optimal treatment system. This type of combined active and passive treatment system would likely be necessary at any Canadian mine site to ensure that water quality regulations could be met at all times of the year. An active system could be used during high flow periods, such as during spring runoff, when the passive system may become overloaded and less effective. The passive system could be used during the summer to reduce operating costs and labour requirements.

At the Wheal Jane Mine in the United Kingdom, both active and passive treatment systems have been used in an attempt to effect a permanent closure strategy (Cambridge 1995). An active treatment system comprising lime dosing, flocculation and precipitation is used as a short term measure, in hopes that the pilot plant passive treatment system can be proven to be effective. Figure 3-1 provides a schematic representation of part of the pilot plant system being tested at Wheal Jane. An active system was not considered in the long term due to the lack of capacity of existing facility to store sludge and anticipated high operating costs.

6.6 COST ASSESSMENT OF TREATMENT TECHNOLOGIES

As outlined in earlier chapters, an assessment of the potential costs for passive treatment in Canada is difficult, since they vary according to a number of site-specific factors. Traditionally, passive treatment systems have been considered attractive due to their expected lower costs of construction, operation and maintenance, and their ability to operate at remote locations with limited operational requirements.

Two studies have been completed recently which compare on a generic and/or site-specific basis the costs for passive and active treatment of AMD: Lawrence and Poulin, 1995; and Gusek, 1995.
Tables 6-2 and 6-3 provide a summary of the results of their investigations. Passive treatment appears to have a significant advantage over conventional lime treatment, primarily due to reduced operating costs. Their applicability to Canadian conditions, however, remains uncertain as outlined earlier in this document.

TABLE 6-1: SUMMARY OF MAINTENANCE AND MONITORING REQUIREMENTS FOR PASSIVE TREATMENT SYSTEMS

Passive System	Maintenance Requirements	Monitoring Requirements
Anoxic Limestone Drains	Once established, little maintenance is required.	Primarily for regulatory purposes; need to monitor influent to ensure DO, Fe^{3+}, Al are <1 mg/L, pH <6 and SO_4 <2,000 mg/L.
Constructed Wetlands	Periodic fertilization may be required, along with general maintenance.	Water quality input and output data should be monitored for regulatory purposes including: nitrogen, phosphorous, iron, manganese, calcium and CEC monthly, subsurface flows should be sampled bi-weekly to assess percolation characteristics. Periodic analysis of plant tissue, soil and substrate also important.
Microbial Reactors - Open	Prevent plugging of sub-surface pipes or weirs and erosion of channels or embankments. Addition of substrate every 6 to 12 months; complete replacement every 3 to 7 years. Nutrient addition may be required.	Monitoring required to ensure proper environment for SRB's to thrive. Alkalinity, acidity, pH, Eh, BOD, COD, TSS, ammonia, nitrate/nitrite, sulphate, Al, Cu, Fe, Mn, Zn, Cd, Ni. Biological parameters need also to be monitored.
Microbial Reactors - Closed	Transfer pumps and periodic cleaning of pipes to remove precipitates and silt. Addition of substrate every 6 to 12 months; complete replacement every 3 to 7 years. Nutrient addition may be required.	Monitoring required to ensure proper environment for SRB's to thrive. Alkalinity, acidity, pH, Eh, BOD, COD, TSS, ammonia, nitrate/nitrite, sulphate, Al, Cu, Fe, Mn, Zn, Cd, Ni. Biological parameters need also to be monitored.
Biosorption Systems - Biological Ponds	Limited information available. Nutrient addition may be required. Metal precipitates will require removal from system components; eventually dead algae at bottom of pond may require removal and disposal.	No information was available. Expected to include: DO. pH, Eh, conductivity, and metals.
Biosorption Systems - Non-living	Frequent eluting and recharging. Addition of new material as metal adsorption/adsorption capability is exhausted.	No information was available. Primarily for regulatory purposes. Expected to include: pH and metals.

TABLE 6-2: SUMMARY OF METAL REMOVAL COST DATA - ACTIVE AND PASSIVE TREATMENT TECHNOLOGY

Site Characteristics

Location	Flow (L/s)	pH	Iron (mg/L)	Copper (mg/L)	Zinc (mg/L)	Lead (mg/L)	Total Metals (mg/L)
Site 1	6.3	4	250	0	0	0	270
Site 2	31.5	4	250	0	0	0	270
Site 3	63	4	250	0	0	0	270
Gallen	30.5	3.5	500	10	40	2.6	553
Sullivan	300	4.5	250	0.15	22	5	277
Equity	91.7	2.3	800	120	80	0	1000

Economic Comparison

Location	Passive Treatment			Conventional Lime Treatment		
	RCC (‘000$)	AAOC (‘000$)	NPV ($/kg)	RCC (‘000$)	AAOC (‘000$)	NPV ($/kg)
Site 1	310	40	0.46	286	68.5	0.7
Site 2	1460	120	0.33	1510	342.5	0.71
Site 3	2800	220	0.32	2300	685	0.66
Gallen	5208	140	0.4	1465	1030	0.89
Sullivan	25670	560	0.38	30102	3342	0.88
Equity	26626	610	0.37	3373	5280	0.8

RCC: raw capital cost (1995 US$)
AAOC: amortized annual operating cost (1995 US$)
NPV: net present value (1995 US$)

From: Gusek, 1995.

TABLE 6-3: COMPARISON OF METAL REMOVAL COST DATA - ACTIVE AND PASSIVE TREATMENT TECHNOLOGY

Treatment Technology	**Capacity (m^3/d)**	**Influent (mg/L)**		**Capital Cost (million US$)**	**Operating Costs**	
		Metals	**Acidity/ Sulphate**		**US$/$m^3$**	**US$/kg SO_4 or acidity**
Lime [1]	4650	-	500/-	0.9	0.19	0.38
Lime [1]	4650	-	5,000/-	1.1	0.88	0.18
Lime [1]	7900	1000	-/8,500	3.4	0.6	0.07
Lime [2]	4650	-	5,000/-	1.4	0.2	0.39
Lime [2]	4650	-	5,000/-	1.7	0.85	0.17
Wetlands	5500	270	-/-	2.8	0.06	-
Wetlands	7900	1000	-/8,500	26.6	0.1	0.01

[1] Conventional lime treatment

[2] High density lime treatment

From: Lawrence and Poulin, 1995.

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APPENDIX A

CASE STUDIES OF PASSIVE SYSTEMS FOR THE TREATMENT OF ACID MINE DRAINAGE

ANOXIC LIMESTONE DRAINS
PHASE I CASE STUDIES

TECHNOLOGY	Anoxic Limestone Drains ALD 1
PROJECT SCALE	Full scale and laboratory tests
REFERENCE	"Use of Passive Anoxic Limestone Drains to Enhance Performance of Acid Drainage Treatment Wetlands". Brodie, G.A., Britt, C.R., Tomaszewski, T.M. and H.N. Taylor. 1991. Proceedings ASSMR, Durango Colorado. pp. 211-228.
LOCATIONS	Two locations described: TVA Fabius Coal and Impoundment 4 ALD.
DESCRIPTION	17 ALDs installed in Alabama. Information provided of several different drains. Basic design includes: an open, unlined trench or excavation backfilled with gravel-sized (3/4 to 1 1/2 in) crushed high-$CaCO_3$ limestone (>90% available $CaCO_3$; depth of limestone dependant on maximum probable flow and desired longevity (TVA and TWPC have used 2 to 5 ft); covered with plastic (20 mil or 2 layers of 10 mil preferred) to preclude oxygen infiltration and CO_2 exsolution and geotechnical fabric to protect plastic from damage; overlain by clay soil (min of 2 ft) to again reduce oxygen infiltration (crowning of soil recommended to allow for eventual ALD subsidence); revegetated by an aggressive species to discourage tree establishment
DESIGN CRITERIA	Effective lifespan for Impoundment 4 ALD is 50 years (modified to include loss of hydraulic conductivity over time - calculated lifespan of 102 years)
ADDITIVES	not applicable
COSTS	TVA Fabius Coal total cost of $18,950: Design and Engineering $3500 Labour and Supervision $3000 Equipment Rental (dozer, excavator, loader) $8000 Limestone (400 T @ $7/T) $2800 Plastic $ 600 Geofabric $ 800 Seed, mulch, fertilizer $ 250 Previous cost for conventional treatment was approx. $20,000 for NaOH and $10,000 for operation and maintenance for seven months. TWPC costs for two ALDs at mine sites were $8000 and $6000 respectively for limestone, labour and heavy equipment operation (no rental).
TIME OF OPERATION / INFORMATION	n.d.
PERFORMANCE DATA	TVA Impoundment 4 Influent / Effluent pH 5.5 6.3 Ca^{2+} 14 mg/l 56 mg/L Total Fe 85 mg/L 65 mg/L Total Mn 17 mg/L 15 mg/L
OVERALL EFFECTIVENESS	Produces increased alkalinity which may act to buffer downstream wetland systems.
TOXICITY	n.d.
MAINTENANCE REQUIREMENTS	n.d.
MONITORING REQUIREMENTS	Not required for operation effectiveness.
TREATMENT PRODUCT DISPOSAL	Not applicable
ENVIRONMENTAL CONDITIONS	n.d.
COMMENTS	

<table>
<tr><td>TECHNOLOGY</td><td>Anoxic Limestone Drains ALD 2</td></tr>
<tr><td>PROJECT SCALE</td><td>Full scale</td></tr>
<tr><td>REFERENCE</td><td>"Anoxic Limestone Drains to enhance Performance of Aerobic Acid Drainage Treatment Wetlands - The Tennesee Valley Authority". Brodie, G.A., Britt, C.R., Tomaszewski, T.M. and H.N. Taylor. 1992. West Virginia Surface Mine Drainage Task Force Symposium, Morgantown West Virginia.</td></tr>
<tr><td>LOCATIONS</td><td>Two case studies: 1) Alabama; 2) Tennessee</td></tr>
<tr><td>DESCRIPTION</td><td>1) Upstream of the IMP4 constructed wetland a trench 10-15 ft wide, 5 ft deep and 260 ft long excavated and backfilled with 400 tons of 3/4 to 1 1/2 inch crushed limestone; then covered by two layers of 10 mil plastic; a layer of geofabric followed by 2 feet of local soil (sandy-silty loam) formed the cover. The surface was slightly crowned and seeded with a mixture including sericea lespedeza, mulched and fertilized. Sericea will discourage establishment of trees whose roots could penetrate the cap and render the ALD ineffective.
2) Upstream of the KIF 006 constructed wetland an ALD with the following dimensions was constructed: 5 ft wide, 5-10 ft deep, 1400 ft long. The ALD runs parallel to a reclaimed ash disposal embankment; entrance to the ALD occurs along its length. Trench was lined with filter fabric, and filled with 3,600 tons of 3/4 inch to 1 1/2 inch crushed limestone. The stone was covered with geotechnical fabric and 2 to 6 feet of compacted local clay loam. The ALD cover was established at a 3:1 slope blending it in with the existing ash pile, and revegetated with a mixture including sericea lespedeza.</td></tr>
<tr><td>DESIGN CRITERIA</td><td>1) Flow rates averaged 34 gal/min through the entire limestone drain / constructed wetland system.
2) Flow rates averaged 408 gal/min into the system.</td></tr>
<tr><td>ADDITIVES</td><td>Not applicable</td></tr>
<tr><td>COSTS</td><td>1) Total cost approximately $19,000; compares to annual costs chemical treatment between October 1985 and May 1990 of approximately $20,000 for NaOH and $10,000 for operation and maintenance. 2) n.d.</td></tr>
<tr><td>TIME OF OPERATION / INFORMATION</td><td>1) April 1990 to ?; no indication of timing of effluent quality data.
2) September 1991 constructed.</td></tr>
<tr><td>PERFORMANCE DATA</td><td>
<table>
<tr><td>1)</td><td>Inflow to <u>Entire System</u></td><td><u>ALD Outflow</u></td><td>Wetland Outflow Pre-ALD constr. <u>(typical)</u></td><td>Wetland Outflow Post-ALD constr. <u>(typical)</u></td></tr>
<tr><td>pH</td><td>5.5</td><td>6.3</td><td>3.1</td><td>6.3</td></tr>
<tr><td>Tot Fe</td><td>65</td><td>85</td><td>6.0*</td><td>1.0*</td></tr>
<tr><td>Tot Mn</td><td>17</td><td>17</td><td>1.6</td><td>0.2</td></tr>
<tr><td>Acidity</td><td>>200</td><td>nd</td><td>350</td><td>40</td></tr>
<tr><td>Alkal.</td><td><30</td><td>nd</td><td>0</td><td>100</td></tr>
</table>
<table>
<tr><td>2)</td><td><u>Inflow to System</u></td><td>No effluent data available</td></tr>
<tr><td>pH</td><td>5.5</td><td></td></tr>
<tr><td>Tot Fe</td><td>170</td><td></td></tr>
<tr><td>Tot Mn</td><td>4.4</td><td></td></tr>
<tr><td>Acidity</td><td>>400</td><td></td></tr>
<tr><td>Alkal.</td><td><40</td><td></td></tr>
</table>
</td></tr>
<tr><td>OVERALL EFFECTIVENESS</td><td>1) Some armouring of limestone with iron in the upper zone of the ALD may have resulted from ground water level fluctuations and periodic oxygen influxes. 2) n.d.</td></tr>
<tr><td>TOXICITY</td><td>n.d.</td></tr>
<tr><td>MAINTENANCE REQUIREMENTS</td><td>1) To stop armouring of upper limestone zone, a berm was constructed at the ALD discharge point to raise water levels within the drain approximately 3-4 feet. 2) n.d.</td></tr>
<tr><td>MONITORING REQUIREMENTS</td><td>Monitoring not required for operation.</td></tr>
<tr><td>TREATMENT PRODUCT DISPOSAL</td><td>Not applicable</td></tr>
<tr><td>ENVIRONMENTAL CONDITIONS</td><td>n.d.</td></tr>
<tr><td>COMMENTS</td><td>1) *Data published in text of document does not appear to correspond to attached graph which indicated Total Iron decreased from 6.9 mg/L to 0.2 mg/L.
2) Data on effectiveness / outflow not included.</td></tr>
</table>

TECHNOLOGY	Anoxic Limestone Drains ALD 3
PROJECT SCALE	Full scale
REFERENCE	"Treatment of Acid Mine Drainage by Passive Treatment Systems". Faulkner, B.B and J.G. Skousen. 1994. Proceedings of Third International Conference on Abatement of Acidic Drainage. pp. 250-257.
LOCATIONS	West Virginia, U.S.
DESCRIPTION	Constructed of 10 to 20 mil plastic; overlain by #57 (gravel-sized) limestone surrounded by filter fabric inside of plastic; topped by hay bales in most instances (see below).

DESIGN CRITERIA

	Limestone (mt)	Materials	Flow (L/min)
Greendale:			
11	51	hay, #57	5
21	66	hay, #57	6
41	18	#57	6
42	21	hay, #57	19
43	43	hay, #57	68
44	36	#57	19
51	8	hay, #57	4
52	14	hay, #57	7
61	22	15-25 cm, #57	19
101	9	#57	11
102	14	15-25 cm, #57	19
103	61	hay, #57	15
108	34	hay, #57	23
121	19	15-25 cm	7
131	15	hay, #57	4
Kodiak:	128	8-12 cm, #4	15
Lillybrook:	7	#57	60
Preston:	90	#57	13
Lobo Cap:	250	hay, #57	87

ADDITIVES	Hay added - found that it encouraged growth of organisms on limestone, reducing pore space and dissolution of limestone.
COSTS	n.d.
TIME OF OPERATION / INFORMATION	Approx. 2 years data summarized below.

TECHNOLOGY

Anoxic Limestone Drains

PERFORMANCE DATA

	pH	pH	Acidity (mg/L)		Alkalinity (mg/L)	
	In	Out	In	Out	In	Out
Greendale:						
11	3.1	6.0	195	90	0	30
21	2.9	5.7	700	0	0	160
41	2.6	6.1	280	140	0	30
42	3.1	6.4	290	100	0	100
43	2.9	6.2	600	70	0	75
44	3.1	5.6	280	220	0	25
51	2.8	3.4	1020	600	0	0
52	2.8	4.7	700	350	0	1
61	2.8	4.3	2000	1000	0	0
101	2.6	4.8	1200	600	0	6
102	2.8	3.7	600	310	0	0
103	2.8	6.1	266	145	0	80
108	2.9	6.0	370	80	0	50
121	3.0	6.8	800	0	0	340
131	3.1	6.1	750	30	0	80
Kodiak:	2.8	4.3	2210	30	0	0
Lillybrook:	5.7	6.4	170	0	23	259
Preston:	3.5	6.1	775	413	0	308
Lobo Cap:	2.9	6.2	470	9	0	131

	Fe (mg/L)		Mn (mg/L)		Al (mg/L)		SO_4 (mg/L)	
	In	Out	In	Out	In	Out	In	Out
Greendale:								
11	130	110	24	19	3	0	300	500
21	190	160	160	80	28	0	1900	1400
41	160	140	40	50	4	0	1000	1000
42	22	70	95	90	24	0	1000	1400
43	120	110	90	50	80	0	1800	900
44	210	150	95	50	24	0	1100	1100
51	140	130	200	160	100	70	2400	2000
52	130	100	170	130	55	30	1600	1600
61	300	360	140	140	140	100	3000	2000
101	150	300	140	120	70	10	2300	2000
102	300	100	120	90	94	17	2060	1500
103	300	210	120	90	94	0	2100	1500
108	80	80	105	96	9	1	1300	1300
121	30	30	45	12	50	0	1100	600
131	41	10	160	140	88	2	2660	2000
Kodiak:	124	1	45	2	287	3	2900	140
L'brook:	60	43	21	21	0	0	101	45
Preston:	570	350	12	12	50	1	1330	1330
L' Cap:	96	58	3	4	18	0	780	575

OVERALL EFFECTIVENESS

Significant changes to pH, acidity and alkalinity were noted as above. Metals were also altered at some sites where armouring of the limestone was occurring.

TOXICITY

n.d.

MAINTENANCE REQUIREMENTS

n.d.

MONITORING REQUIREMENTS

Extensive monitoring conducted to determine effectiveness - not required for operation.

TREATMENT PRODUCT DISPOSAL

Not applicable

ENVIRONMENTAL CONDITIONS

n.d.

COMMENTS

Organic matter such as hay, should not be added as it encourages microorganism growth reducing pore space and dissolution of limestone. Use of larger limestone particle sizes (8 to 25 cm) increases hydraulic conductivity and may reduce plugging by metal hydroxides. 15 to 20 hours of residence time required; water saturated drains are more effective. Armouring was observed where inadequate measures were taken to prevent oxygen intrusion. The use of piping with ALDs is being reduced because of plugging problems.

TECHNOLOGY

Anoxic Limestone Drains ALD 4

PROJECT SCALE

Full scale

REFERENCE

"The Effects of Anoxic Limestone Drains on Mine Water Chemistry". Hedin, R.S. and G.R. Watzlaf. 1994. International Land Reclamation and Mine Drainage Conference, Volume 1. pp. 185-194.

LOCATIONS

21 ALDs located across Appalachia: Rid-2L, Hathaway, AROAD, Rid-2R, Rid-1, TVA-2, Morrison, Schnepp, Willi, Jennings, Howe-2, Howe-1, Shade, TVA-4, REM-R, Fawn, Ohiopyle (no locations given).

DESCRIPTION

n.d.

DESIGN CRITERIA

	Rid-2L	Hathaway	AROAD	Rid-2R	Rid-1	Morrison
Limestone (mt)	114	945	nd	162	108	64
Flow (L/min)	0.8	nd	265	0.5	5	7
Mass/flow (mt/L/min)	142.5	nd	nd	324	24	9.4
Theoretical Retention Time (hr)	698	nd	nd	1588	118	46

	Schnepp	Willi	Jennings	Howe-2	Howe-1
Limestone	132	182	364	132	455
Flow	57	5	92	53	92
Mass/flow	2.3	36.4	4.0	2.5	4.9
Theor.R.T.	11	178	19	12	24

	REM-L	Shade	TVA-4	REM-R	Ohiopyle
Limestone	125	35	364	124	225
Flow	82	15	131	115	218
Mass/flow	1.5	2.3	2.8	1.1	1.0
Theor.R.T.	7	11	14	5	5

ADDITIVES

Not applicable

COSTS

n.d.

TIME OF OPERATION / INFORMATION

n.d.

PERFORMANCE DATA

Changes in concentrations (mg/L)
(see text for calculations)

	Alk In	Alk Out	pH In	pH Out	Tot. Fe	Fe^{3+}	Fe^{2+}	Al	Mn
Rid-2L	0	469	2.3	6.2	-1214	-680	-534	-486	-12
Hath'y	0	385	3.3	6.5	-141	nd	nd	-23	-30
AROAD	0	350	2.8	6.6	+23	-10	+33	-19	-2
Rid-2R	0	306	3.7	6.3	-53	-1	-52	-31	0
Rid-1	3	290	4.7	6.5	+22	-1	+23	-2	+4
TVA-2	0	280	3.5	6.7	-16	nd	nd	0	-6
Morr'n	23	271	5.3	6.3	-65	0	-65	-1	-9
Schnepp	0	191	3.3	6.3	-35	nd	nd	-7	+11
Willi	0	187	2.7	6.3	-48	-48	0	-62	-11
Jenn'gs	0	177	3.3	6.3	-11	-10	-1	-21	0
Howe-2	24	174	5.9	6.5	-5	0	-5	0	0
Howe-1	33	161	5.6	6.2	-2	0	-2	0	-1
Shade	0	123	3.5	6.6	-2	0	-2	-22	+4
TVA-4	nd	120	4.9	6.7	-135	nd	nd	0	-21
REM-R	0	69	4.3	5.5	-82	-32	-50	-2	-4
Fawn	0	0	3.5	3.7	+28	0	+28	+49	+5
Ohio'e	0	0	3.3	3.6	-7	-6	-1	-25	-16

OVERALL EFFECTIVENESS

Excluding two sites, all ALD showed significant alkalinity and pH changes. Retention time was an important consideration. In addition several sites (Rid-1 and Fawn) unintentionally collected non-target acid mine drainage and, therefore appear less effective. Concentrations of alkalinity reached maximum levels after 14-23 hours of contact; retention times of over 23 hours did not markedly increase concentrations.

TOXICITY

n.d.

MAINTENANCE REQUIREMENTS

not applicable

MONITORING REQUIREMENTS

Not required for operation.

TREATMENT PRODUCT DISPOSAL

not applicable

TECHNOLOGY	Anoxic Limestone Drains ALD 4
ENVIRONMENTAL CONDITIONS	n.d.
COMMENTS	Mass of limestone required for treatment can be calculated by: $M = \frac{(\text{vol. flow})(\text{ls. bulk density})(\text{ret. time})}{\text{bulk void volume}} + \frac{(\text{vol. flow})(\text{pred. alk. conc.})(\text{design life})}{CaCO_3 \text{ content in ls.}}$ ls: limestone Concentration of alkalinity reached maximum levels after 14-23 hours of contact.

<table>
<tr><td>TECHNOLOGY</td><td>Anoxic Limestone Drains (also addressed under constructed wetlands) ALD 5</td></tr>
<tr><td>PROJECT SCALE</td><td>Full scale</td></tr>
<tr><td>REFERENCE</td><td>"Generation of Alkalinity in an Anoxic Limestone Drain". Nairn, R.W., Hedin, R.S. and G.R. Watzalf. 1992. ASSMR Proceedings Duluth, Minnesota. pp. 206-219.</td></tr>
<tr><td>LOCATIONS</td><td>Paint Township, Clarion County, Pennsylvania</td></tr>
<tr><td>DESCRIPTION</td><td>An anoxic limestone drain was constructed in order to provide the initial treatment for a seep from the toe of a spoil pile as part of a constructed wetland system. A trough 50 m long, 0.6 m wide was filled with: 1 m of 2.5 - 15 cm limestone; covered by two layers of 5 mil plastic and 0.3 to 3 m of on-site clay (to restore original topography). High quality local limestone used, composed of approx. 93% $CaCO_3$, 1.2% $MgCO^3$, 6% other.</td></tr>
<tr><td>DESIGN CRITERIA</td><td>Flow rates at drain exit ranged from 1 to 12 L/min; mean and median flow rates of 7.5 L/min and 6.9 L/min. Retention time allowed saturation to be attained. At a mean alkalinity generation of 117 g/m^3, maximum longevity is 65 years; structural failure due to dissolution of limestone and subsequent subsidence and oxygen infiltration expected to occur earlier. Estimated longevity of 30 years.</td></tr>
<tr><td>ADDITIVES</td><td>Not applicable</td></tr>
<tr><td>COSTS</td><td>Total cost estimated at $1300 included:
<table>
<tr><td>hypothetical equipment rental 10 hours @ $35/hr</td><td>$350</td></tr>
<tr><td>70 tons limestone @ $10/ton</td><td>$700</td></tr>
<tr><td>plastic liner 600 ft^2 @ $0.05/$ft^2$</td><td>$ 30</td></tr>
<tr><td>labour (approx)</td><td>$180</td></tr>
<tr><td>seed and fertilizer (approx)</td><td>$ 40</td></tr>
</table>
Previous chemical treatment of AMD cost approximately $1640/yr, in chemical costs alone; total annual costs of over $7000/yr (including operation and maintenance, sludge handling and disposal, and labour).
Long-term cost for ALD estimated at $0.12/day as compared to $20.00/day for chemical treatment.</td></tr>
<tr><td>TIME OF OPERATION / INFORMATION</td><td>Data represents October 1990 to October 1991.</td></tr>
<tr><td>PERFORMANCE DATA</td><td>
<table>
<tr><th></th><th colspan="2">Before drain constr.</th><th colspan="2">After drain constr.</th></tr>
<tr><td>pH</td><td>4.7</td><td>(3.0-5.3)</td><td>6.5</td><td>(6-6.5)</td></tr>
<tr><td>Acidity (mg/L)</td><td>438</td><td>(298-611)</td><td>83</td><td>(0-143)</td></tr>
<tr><td>Alkalinity(mg/L)</td><td>15</td><td>(0-24)</td><td>255</td><td>(204-335)</td></tr>
<tr><td>Total Fe (mg/L)</td><td>208</td><td>(102-411)</td><td>168</td><td>(97-213)</td></tr>
<tr><td>Fe^{2+} (mg/L)</td><td>208</td><td>(124-261)</td><td>168</td><td>(97-206)</td></tr>
<tr><td>Mn (mg/L)</td><td>44</td><td>(5-61)</td><td>46</td><td>(33-54)</td></tr>
<tr><td>Al (mg/L)</td><td>0.6</td><td>(<0.2-1)</td><td>0.2</td><td>(<0.2-1)</td></tr>
<tr><td>Ca (mg/L)</td><td>117</td><td>(74-128)</td><td>229</td><td>(205-245)</td></tr>
<tr><td>Mg (mg/L)</td><td>116</td><td>(66-139)</td><td>109</td><td>(84-133)</td></tr>
<tr><td>Na (mg/L)</td><td>46</td><td>(25-59)</td><td>37</td><td>(28-45)</td></tr>
<tr><td>SO_4 (mg/L)</td><td>1332</td><td>(830-1757) 1168</td><td>(875-1350)</td><td></td></tr>
</table></td></tr>
<tr><td>OVERALL EFFECTIVENESS</td><td>Dramatic changes to water chemistry (alkalinity, acidity, pH, Ca) reflected in results above, and further downstream. After exiting the drain iron oxidation and hydrolysis caused iron to precipitate in the adjacent aerobic channel and settling pond.</td></tr>
<tr><td>TOXICITY</td><td>n.d.</td></tr>
<tr><td>MAINTENANCE REQUIREMENTS</td><td>n.d.</td></tr>
<tr><td>MONITORING REQUIREMENTS</td><td>Extensive monitoring conducted to determine effectiveness - not required for operation.</td></tr>
<tr><td>TREATMENT PRODUCT DISPOSAL</td><td>Not applicable</td></tr>
<tr><td>ENVIRONMENTAL CONDITIONS</td><td>n.d.</td></tr>
<tr><td>COMMENTS</td><td>ALD are an inexpensive partial treatment system that generates alkalinity and facilitates subsequent metal removal. Maximum alkalinity concentrations achievable appear to be about 300 mg/L as $CaCO_3$ equivalent.</td></tr>
</table>

TECHNOLOGY Anoxic Limestone Drains ALD 6

PROJECT SCALE Laboratory scale

REFERENCE "A Method for predicting the Alkalinity Generated by Anoxic Limestone Drains". Watzlaf, G.R. and R.S. Hedin. 1993. West Virginia Surface Mine Drainage Task Force Symposium, Morgantown, West Virginia.

LOCATIONS Water quality assessed from two sites: Howe Bridge, Jefferson County, Pennsylvania; and Morrison Site, Clarion County, Pennsylvania.

DESCRIPTION Experiment involved use of sealed cubitainers filled approximately 2/3 full with 4.0 kg of clean 1/2 inch to 1 3/8 inch limestone or dolomite.

Three stages of the experiment:
1) Three cubitainers established adjacent to the Howe Bridge ALD; filled with the same ALD influent; alkalinity concentrations measured 11 times during the 72 hour experiment in cubitainer and ALD.
2) Howe Bridge mine water; ten cubitainers with five different types/grades of limestone; alkalinity monitored for 11 days.
3) Morrison; fourteen cubitainers with seven different limestones and dolomite were filled with untreated AMD seepage; alkalinity monitored for 11 days.

DESIGN CRITERIA not applicable - single time addition of AMD to fill container

ADDITIVES not applicable

COSTS n.d.

TIME OF OPERATION / INFORMATION 72 hours or 11 days; See above

PERFORMANCE DATA

Influent (mg/L)	pH	Acid	Tot Fe	Mn	Ca	Mg	SO_4
Howe Bridge	5.73	516	267	39	153	102	1290
Morrison	5.42	460	239	58	132	143	1490

Alkalinity (mg/L)	Influent	Effluent
Howe Bridge	39	190
Morrison	29	294

Water Alkalinity (mg/L) after two days in cubitainer (various stone compositions):

Stone	$MgCO_3$ %	$CaCO_3$ %	Howe Bridge In	Howe Bridge Out	Morrison In	Morrison Out
1	1.4	91	39	187	29	320
2	1.1	92	39	nd	29	315
3	nd	94	39	172	29	283
4	1.4	94	39	183	29	300
5	0.4	99	39	167	29	285
6	4.5	82	39	157	29	275
7	38	46	39	nd	29	149

OVERALL EFFECTIVENESS Data between the cubitainer and field ALD data for experiment #1 corresponded satisfactorily.

For experiments #2 and #3, alkalinity increased substantially with all rock types; high quality limestone was the most effective. Overall mine water chemistry also had an impact; limestone was more effective at adding alkalinity to Morrison site AMD which started with a lower alkalinity. The most significant difference between the two sites was the Morrison water had a CO_2 partial pressure 65% higher than at Howe Bridge.

TOXICITY n.d.

MAINTENANCE REQUIREMENTS not applicable

MONITORING REQUIREMENTS not applicable

TREATMENT PRODUCT DISPOSAL not applicable

ENVIRONMENTAL CONDITIONS n.d.

TECHNOLOGY	Anoxic Limestone Drains	ALD 6
COMMENTS	#1: Changes in alkalinity were greatest during the first 24 hours, but continued to increase unless experiment stopped (48 hours). #2 and #3: alkalinity in the cubitainers increased rapidly over the first two days, and then increased more slowly for the remaining 9 days. There was little difference in the amount of alkalinity generated by the different qualities of limestone; however, the dolomite samples were considerable less effective. The limestone with the lowest calcium content did; however, produce the effluent with the lowest alkalinity. The use of cubitainers allows a determination of amount of alkalinity that will be generated by an ALD prior to its construction.	

TECHNOLOGY	Anoxic Limestone Drains ALD 7
PROJECT SCALE	Pilot Scale
REFERENCE	"Treatment of Waste Rock Drainage with Limestone Beds". Lapakko, K. and D. Antonson. 1990. In: Acid Mine Drainage Designing for Closure. pp. 273-283.
LOCATIONS	n.d.
DESCRIPTION	A polyethylene tank (132 cm in diameter; 152 cm in height) filled with 2020 kg of high-calcium limestone chips to a thickness of 104 cm, was placed below ground level in a steel tank to prevent damage from freezing soils. The limestone is covered by 5 cm of water. Receives drainage through a PVC pipe onto a splash plate; effluent released through an underdrain.
DESIGN CRITERIA	Flow through the system ranged from 0.032 L/s to 1.514 L/s, averaging 0.41 L/s. Retention times ranged from 15 minutes to 12 hours, with an average time of 57 minutes.
ADDITIVES	not applicable
COSTS	n.d.
TIME OF OPERATION / INFORMATION	188 days; April 26 to October 31 1989.

PERFORMANCE DATA

	Influent	Effluent
pH	5.0	6.85
Net alkalinity	-51	24
Copper (mg/L)	0.97	0.50
Nickel (mg/L)	15.6	14.2
Cobalt (mg/L)	1.2	1.1
Zinc (mg/L)	2.8	2.5

Net alkalinity = alkalinity - acidity
520 kg of limestone dissolved by end of test

OVERALL EFFECTIVENESS	Over the flow range above, 70.2 mg of alkalinity was released for every litre of flow. Both pH and net alkalinity were fairly consistent, indicating that they were not highly dependent on the volume of flow treated or retention time.
TOXICITY	n.d.
MAINTENANCE REQUIREMENTS	n.d.
MONITORING REQUIREMENTS	Flow regulation by weir; other monitoring for assessment of success only.
TREATMENT PRODUCT DISPOSAL	n.d.
ENVIRONMENTAL CONDITIONS	n.d.
COMMENTS	Flow impedance through the system was a problem which could be avoided by use of a larger size of limestone. An upstream settling pond would assist in removing suspended solids and reducing clogging of the limestone bed.

CONSTRUCTED WETLANDS
PHASE I CASE STUDIES

TECHNOLOGY Constructed Wetlands WTLD 1

PROJECT SCALE Pilot to full scale

REFERENCE "Achieving Compliance with Staged, Aerobic, Constructed Wetlands to Treat Acid Drainage". Brodie, G.A. 1991. Proceedings Reclamation 2000: Technologies for Success, Volume 1, Durango Colorado. pp. 151 - 174.

LOCATIONS Tennessee Valley Authority; Alabama and Tennessee.

DESCRIPTION Twelve staged, aerobic, constructed wetlands to treat acid drainage at reclaimed coal mines, a coal preparation plant, and at coal fired power plants.

DESIGN CRITERIA Wetlands were sized based on hydraulic loading.

Wetland	Area (m^2)	Cells	Loading (GDM) Fe	Loading (GDM) Mn	Ave. Flow (LDM)	Operating Date
950	3400	3	.42	.28	.04	1-76
IMP1	5700	4	1.27	.17	.02	7-85
IMP4	2000	3	6.13	1.58	.09	11-85
WCF5	6600	4	--	--	.21	7-90
WCF6	4800	3	13.0	.59	.09	6-86
WCF19	25000	3	.5	.2	.03	6-86
IMP2	11000	5	5.32	1.72	.13	6-86
IMP3	1200	3	1.1	.34	.07	10-86
RT2	7300	3	2.47	.73	.05	9-87
950NE	2500	4	2.44	2.0	.22	9-87
KIF6	9300	3	41.43	1.07	.24	10-87
COF	9200	5	.03	.24	.05	10-87
DLL	7550	4	.73	.4	.07	5-90

Paper also suggests sizing wetlands based on chemical loading.

ADDITIVES n.d.

COSTS

IMP1	Total cost was $43,000 (1985 U.S. dollars). Annual costs have been about $13,000 due to repairs on the prototype and design and extensive monitoring.
IMP3	Built in 1986 at a cost of $40,000.
IMP4	Total cost was $28,000 (1985 dollars) with $32,000 average annual costs through May 1990, primarily for chemical treatment. The anoxic drain costs $19,000 and paid for itself in about 8 months.

TIME OF OPERATION / INFORMATION

	Date Constructed
IMP1	June, 1985
IMP3	1986
IMP4	Fall, 1985

PERFORMANCE DATA Systems currently removing Fe at rates between 0.4 and 21.3 (GDM) of wetlands and Mn at rates between 0.15 and 1.87 GDM. Loading rates range between 0.03 - 41.5 GDM for Fe and 0.17 - 2 GDM for Mn. All of the wetlands meet compliance for total suspended solids.

Wetland	Influent Water (mg/L) pH	Fe	Mn	TSS	Effluent Water (mg/L) pH	Fe	Mn	NFR	Effluent Water Flow Ave.	Max.
950	5.7	12.0	8.0	20.0	6.5	1.1	1.6	5.4	83	341
IMP1	6.1	69.0	9.3	9.5	6.7	0.9	1.8	3.0	73	693
IMP4	6.3	65.0	16.8	21.0	6.3	0.4	0.6	6.0	131	693
WCF5	--	--	--	--	8.4	2.2	0.7	--	973	2057
WCF6	5.6	150.0	6.8	--	3.9	6.4	6.2	--	289	1495
WCF19	5.6	17.9	6.9	--	4.3	3.3	5.9	--	492	6360
IMP2	3.5	40.0	13.0	9.0	3.1	3.4	13.0	0.8	1016*	1540*
IMP3	6.3	15.8	4.9	21.4	7.0	0.5	0.7	9.0	58	250
RT2	5.7	45.2	13.4	--	6.8	0.6	1.8	3.2	277	1155
950NE	6.0	11.0	9.0	19.0	6.9	0.6	0.8	5.0	385	1386
KIF6	5.5	170.0	4.4	40.0	2.9	82.5	4.6	--	1574	2271
COF	5.7	0.7	5.3	--	6.7	0.7	3.5	--	288	408
DLL	6.2	10.0	5.5	23.0	6.4	2.1	2.2	10.0	385	7700

* - Also receives pumpage from slurry lake up to 4800 L/min.
NFR = non filterable residue

OVERALL EFFECTIVENESS Nine systems produced consistent compliance quality discharges without chemical treatment. Four systems not achieving compliance have high Fe and zero alkalinity in the inflow that results in low pH due to Fe hydrolysis. These systems were modified with anoxic limestone drains.

TOXICITY n.d.

TECHNOLOGY	Constructed Wetlands WTLD 1
MAINTENANCE REQUIREMENTS	Fertilization, maintenance of dikes and spillways, and addition of new ponds to further treat wetlands discharge.
MONITORING REQUIREMENTS	Water quality monitoring; substrate and vegetation sampling.
TREATMENT PRODUCT DISPOSAL	n.d.
ENVIRONMENTAL CONDITIONS	n.d.
COMMENTS	Significant water quality improvement has occurred at all of the wetlands

TECHNOLOGY

Constructed Wetlands WTLD 2

PROJECT SCALE

Full scale

REFERENCE

"Aerobic Constructed Wetlands and Anoxic Limestone Drains to Treat Acid Drainage: An Overview of the Tennessee Valley Authority Program". Brodie, G.A. 1993. Constructed Wetlands Workshop for Electrical Power Utilities, Chattanooga, TN.

LOCATIONS

IMP 1 constructed wetland was constructed in May, 1985 to treat acid drainage emanating from an earthen dam impounding fine coal refuse at TVA's Fabius Coal Preparation Plant in Alabama.

DESCRIPTION

15 staged, aerobic constructed wetlands. The wetlands generally consist of pretreatment stage (anoxic limestone drain and/or oxidation basin) followed by several cells of shallow to deep (0.1 to 2.0 m) cattail (*Typha spp.)*marsh-ponds. Most of the systems have been constructed in streams receiving groundwater created by the acid drainage; a few sites required diversions to route the drainage to the wetlands system. Some systems are followed by a final polishing pond. Wetland systems built before 1988 were sized hydraulically and then increased if the site allowed. Cell areas were arbitrarily increased in size if very poor quality water was to be treated. Spillways were designed for handling the maximum probable flow and protected against erosion with riprap or non-biodegradable erosion-control fabric planted with wool grass (*Scirpus cyperinus*) or sedge (*Cares* sp.). Shapes were dictated by existing topography, geology, and land availability. The number of cells was determined by site topography, hydrology, and water quality. Average water depth ranged from 15 - 30 cm.

DESIGN CRITERIA

Fifteen staged, aerobic constructed wetlands to treat acid drainage at reclaimed coal mines and preparation plants, and at fossil power plants.

Wetland	Area (m^2)	Cells	Loading (GDM)		Ave. Flow (LDM)	Operating Date
			Fe	Mn		
950	3400	3	.42	.28	.04	1-76
IMP1	5700	4	1.27	.17	.02	7-85
IMP4	2000	3	6.13	1.58	.09	11-85
WCF5	6600	4	--	--	.21	7-90
WCF6	4800	3	13.0	.59	.09	6-86
WCF19	25000	3	.5	.2	.03	6-86
IMP2	11000	5	5.32	1.72	.13	6-86
IMP3	1200	3	1.1	.34	.07	10-86
RT2	7300	3	2.47	73	.05	9-87
950NE	2500	4	2.44	2.0	.22	9-87
KIF6	9300	3	41.43	1.07	.24	10-87
COF	9200	5	.03	.24	.05	10-87
DLL	7550	4	.73	.4	.07	5-90
HROOO	40000	5	--	--	--	6-91

ADDITIVES

Fertilized generally once with a phosphorous-potassium fertilizer such as 0-12-12 at 400 kg/ha.

COSTS

Total cost for the IMP1 wetlands was $43,000 (1985 dollars). Annual costs from 1985 - 1990 were about $13,000 due to repairs on the prototype design and extensive monitoring. Operations and maintenance costs in 1993 are less than $1,000 annually. Chemical treatment would have been approximately $250,000 from 1985 to 1991.

TIME OF OPERATION / INFORMATION

1985 and on going.

TECHNOLOGY

Constructed Wetlands WTLD 2

PERFORMANCE DATA

Wetland	Influent Water (mg/L)				Effluent Water (mg/L)				Effluent Water Flow	
	pH	Fe	Mn	TSS	pH	Fe	Mn	NFR	Ave.	Max.
950	5.7	12.0	8.0	20.0	6.5	1.8	2.6	2.0	148	341
IMP1	3.1	58.6	8.0	78.0	6.8	0.9	1.5	3.0	64	681
IMP4	4.4	101.2	23.3	20.0	7.7	0.6	1.1	3.0	132	681
WCF5	--	--	--	--	8.4	2.2	0.7	--	973	2057
WCF6	5.6	372.0	7.2	62.0	3.9	6.4	6.2	--	289	1495
WCF19	5.6	17.9	6.9	--	4.3	3.3	5.9	--	492	6360
IMP2	3.5	40.0	13.0	9.0	3.1	3.4	13.0	0.8	1016*	1540*
IMP3	6.8	14.2	4.3	22.0	6.9	0.5	0.7	8.0	53	246
RT2	5.7	45.2	13.4	--	6.8	0.7	2.1	4.0	329	1155
950NE	6.0	11.0	9.0	19.0	6.9	0.6	0.8	5.0	385	1386
KIF6	5.5	170.0	4.4	40.0	2.9	82.5	4.6	--	1574	2271
COF	5.7	0.7	5.3	--	6.7	0.7	3.5	--	288	408
DLL	6.4	7.6	3.3	23.0	6.5	1.2	0.9	5.0	291	7700
HROOO	4.5	40.0	17.0	--	6.3	1.5	1.5	10.0	4000	--

* - Also receives pumpage from slurry lake up to 4800 L/min.
NFR = non filterable residue

Constructed wetlands are currently removing Fe at rates between 0.4 and 21.3 GDM of wetlands and Mn at rates between 0.15 and 1.87 GDM. Loading rates are 0.03 - 41.5 GDM for Fe and 0.17 - 2.0 GDM for Mn. All of the wetlands meet compliance for total suspended solids.

On average hydraulic loadings were between 0.02 - 0.24 LDM of wetlands. Maximum hydraulic loading ranges were 0.06 - 1.47 LDM and averaged 0.42 LDM.
Fe loading in the systems ranges from 0.03 GDM to 41.4 GDM. Fe removal rates ranges from 0.0 GDM to 21.3 GDM, corresponding to 0 to 99% removal.
Mn loading ranges from 0.17 to 2.00 GDM. Mn removal ranges from 0.15 to 1.87 GDM, corresponding to 0 to 96% Mn removal. The low removal rates are all associated with low pH (2.9 - 3.9)

OVERALL EFFECTIVENESS

The discharge of IMP1 has consistently been in compliance with total Fe reduced from 69 mg/L to 0.9 mg/L and total Mn from 9.3 mg/L to 1/6 mg/L. The pH has increased from 3.1 to 6.8 due to a limestone roadbed located under the dam. Aquatic flora and fauna in the constructed wetlands and receiving stream have shown rapid growth, expansion, and diversification.

Significant water quality improvement has occurred at all of the wetlands. Twelve systems have produced discharges that consistently meet NPDES monthly average discharge limitations (pH of 6 - 9; Fe < 3.0 mg/L; Mn < 2.0 mg/L; TSS < 35.0 mg/L) with no chemical treatment.

TOXICITY

n.d.

MAINTENANCE REQUIREMENTS

n.d.

MONITORING REQUIREMENTS

pH, total Fe and Mn, and total suspended solids. Effluent samples from were obtained during daylight hours generally within the second and fourth weeks of the month. Sampling was always initiated within two weeks of system start up.

TREATMENT PRODUCT DISPOSAL

n.d.

ENVIRONMENTAL CONDITIONS

n.d.

COMMENTS

Drainage with high Fe (> 50 mg/L) and acidity has not been amenable to treatment with aerobic wetlands alone, primarily due to Fe hydrolysis and resultant low pH.

There is no correlation between Fe removal and influent alkalinity, Fe removal and wetlands size, or Fe removal and hydraulic loading. There is a good correlation between Mn removal and influent alkalinity and acidity concentrations.

Data suggests that Mn co-precipitation on Fe-oxides as circum-neutral pH is a likely mechanism of Mn removal.

A preferred design may consist of: an initial anaerobic limestone trench at the source of the seepage to passively add alkalinity; a large deep settling basin to accumulate oxidized and precipitated Fe sludges; and, a two or three cell constructed wetland for Mn and further Fe removal.

TECHNOLOGY	Constructed Wetlands WTLD 3
PROJECT SCALE	Pilot scale
REFERENCE	"An Evaluation of S...ted Wetlands Acid Drainage Treat Systems", Brodie, G.A., Hammer, D.A. and D.A. Tomljanovich. 1988. U.S. Bureau of Mines Information Circular 9183. pp. 389 - 398.
LOCATIONS	Tennessee Valley Authority: Acid Drainage Wetlands Research Facility , Jackson County, Alabama.
DESCRIPTION	20 cells each consisting of a buried, half round, fibreglass culvert 6.3 m x 1.1 m with various substrates. An unlined pond in spoil material was planted with marsh/wet meadow vegetation. Pond and pea gravel cells served as controls. Substrates were clay from the B horizon at an undisturbed site, mine spoil, pea gravel, topsoil from the A horizon of an agricultural field, substrate from a natural wetland below an acid seep, and substrate from a natural wetland without acid drainage. Three cells were planted with bulrush (*Scirpus cyperinus*), the remainder were planted with cattail (*Typha latifolia*)
DESIGN CRITERIA	Designed to test the performance of various substrates under two flow regimes of 1 L/min and 0.5 L/min.
ADDITIVES	6 - 12 - 12 fertilizer.
COSTS	n.d.
TIME OF OPERATION / INFORMATION	January 14 to September 9, 1987 under a flow of 0.5 L/min.
PERFORMANCE DATA	Seep water had a pH of 5.9, 37 mg/L Fe, and 16 mg/L Mn. Influent water had a pH of 5.9, and a concentration of 32.3 mg/L dissolved Fe, 14.8 mg/L dissolved Mn, and 32.1 mg/L TSS. Effluent concentrations of dissolved Fe ranged from 4.8 mg/L to 7.2 mg/L (topsoil/cattail). TSS effluents ranged from 13.9 mg/L (acid wetland/cattail) to 14.5 mg/L (natural wetland/cattail). pH values ranged from 6.2 (natural acid wetland/cattail) to 6.4 (topsoil/cattail). Dissolved Mn ranged from 13.9 mg/L (acid wetland/cattail) to 14.5 mg/L (natural wetland/cattail).
OVERALL EFFECTIVENESS	All substrate types significantly reduced dissolved Fe and TSS, and increased pH. The removal efficiency improvement pattern common to all experimental cells suggests a major biological component in the removal process.
TOXICITY	n.d.
MAINTENANCE REQUIREMENTS	n.d.
MONITORING REQUIREMENTS	Individual cell flow rates, inflow and outflow water samples, and substrate sampling.
TREATMENT PRODUCT DISPOSAL	n.d.
ENVIRONMENTAL CONDITIONS	n.d.
COMMENTS	Results suggest that substrate type is relatively unimportant to removal treatment efficiency since the desired plant-substrate-microbe complex became established in each type; microbial inocula were unnecessary, and; vegetation may substantially improve treatment efficiency.

TECHNOLOGY Constructed Wetlands WTLD 4

PROJECT SCALE Pilot to full scale

REFERENCE "Constructed Wetlands for Acid Drainage Control in the Tennessee Valley". Brodie, G.A., Hammer, D.A. and D.A. Tomljanovich. 1988. U.S. Bureau of Mines Information Circular 9183. pp. 325 - 331.

LOCATIONS Alabama and Tennessee

DESCRIPTION

Eleven wetlands ranging in size from 3.5 m^2 to 113.0 m^2 per average flowing litre per minute, and 2.0 m^2 to 41.0 m^2 per maximum flowing litre per minute. Wetland shapes varied because of existing topography, geology, or land availability. Average water depth ranged from 15 - 20 cm with shallower and deeper areas to provide for species diversification and to provide recharge zones and aquatic fauna refuge in drought events. *Typha latifolia* was set into the substrate at about nine plants per square metre, and stems broken above the water level to prevent windfall and to stimulate new growth. Wetlands were fertilized with a phosphorous-potassium fertilizer such as 0-12-12 at 400 kg/ha. Mosquito fish (*Gambusio affinis*) were stocked for insect pest control.

DESIGN CRITERIA

All effluent discharge limitations were to be met:

Total Fe:	<3.0 mg/L
Total Mn:	<2.0 mg/L
pH:	6.0 - 9.0
Nonfilterable residue:	<35 mg/L

Wetland	Area (m^2)	Cells	Treatment Area m^2/mg/min	
			Fe	Mn
WC018	4800	3	0.2	4.2
King006	9300	3	0.2	5.0
IMP4	2000	3	0.4	2.0
950NE	2500	2	0.7	0.8
RT-2	7300	3	0.7	2.3
IMP2	11000	5	0.7	2.1
IMP3	1200	3	1.1	2.8
WC019	25000	3	2.8	7.4
950-1&2	3400	3	3.4	5.1
COL013	5700	4	3.6	11.8
KIF6	9200	5	45.6	6.0

ADDITIVES n.d.

COSTS

Wetland	% Equip.	% Labour	% Overh	Total ($000's)	$/$m^2$
RT-2	32.6	53.4	14.0	26.4	3.58
IMP2	70.2	12.3	17.5	57.0	5.25
WC018&019	--	--	--	209.0	6.98
IMP1	19.0	49.6	31.4	42.9	7.46
950NE	38.1	46.0	15.9	35.2	13.8
IMP4	28.2	26.8	45.0	28.0	14.12
King006	28.6	43.3	28.1	131.7	14.21
IMP3	34.8	43.5	21.7	40.2	32.03

Costs from design to operation averaged $12.18/$m^2$.

TIME OF OPERATION / INFORMATION June, 1985 to August, 1987

PERFORMANCE DATA

Wetland	Influent Water (mg/L)				Effluent Water (mg/L)				Flow	
	pH	Fe	Mn	NFR	pH	Fe	Mn	NFR	Ave.	Max.
WC018	5.6	150.0	9.8	---	3.9	6.4	6.2	---	70	1495
King006	4.2	153.0	4.9	40.0	---	---	---	---	379	2271
IMP4	4.9	135.0	24.0	42.0	4.6	3.0	4.0	6.0	42	49
950NE	6.0	11.0	9.0	19.0	6.6	0.5	0.2	49.1[1]	348	1673
RT-2	5.7	45.2	13.4	---	6.7	0.8	0.2	2.0	238	681
IMP2	3.1	40.0	13.0	9.0[2]	3.1	3.4	14.0	0.8[2]	400	2200
IMP3	6.3	13.0	5.0	28.0	6.8	0.8	1.9	4.7	87	379
WC019	5.6	17.9	6.9	---	4.3	3.3	5.9	---	492	6360
950-1&2	5.7	12.0	8.0	20.0	6.5	1.1	1.6	5.4	83	341
IMP1E	6.3	30.0	9.1	57.3[3]	6.5	0.9	2.1	2.8	53	227
COL013	5.7	0.7	5.3	---	6.7	0.7	13.5	---	288	408

NFR = non filterable residue

[1] - one effluent sample to date

[2] - one sample, July 1987

[3] - from preconstruction instream sample

TECHNOLOGY	Constructed Wetlands WTLD 4
OVERALL EFFECTIVENESS	Efficiencies ranged from 9% to 99% removal for total Fe, and 9% to 98% removal for total Mn.
TOXICITY	n.d.
MAINTENANCE REQUIREMENTS	Post-construction activities included fertilization and maintenance of dikes, spillways or other control structures and pest control.
MONITORING REQUIREMENTS	Flow, total Fe and Mn, and nonfilterable residue.
TREATMENT PRODUCT DISPOSAL	n.d.
ENVIRONMENTAL CONDITIONS	n.d.
COMMENTS	Offers preliminary design guidelines based on results.

TECHNOLOGY Constructed Wetlands WTLD 5

PROJECT SCALE Full scale

REFERENCE

"Constructed Wetlands for Treatment of Ash Pond Seepage". Brodie, G.A., Hammer, D.A. and D.A. Tomljanovich. 1989. Constructed Wetlands. pp. 211 - 219.

LOCATIONS

Widows Creek Steam Plant, Guntersville Reservoir in Jackson County, Alabama;
Kingston Steam Plant, Watts Bar Reservoir in Roane County, Tennessee;
Colbert Steam Plant, Pickwick Reservoir in Colbert County, Alabama.

DESCRIPTION

Widows: Dikes and spillways were built in April 1986 to create a 0.5 ha wetland consisting of three vegetated cells. Cattail and rush was planted in cell 1, and *Iris versicolor* and *Eleocharis quadrangulata* in cells 2 and 3. In May 1986, three earthen dikes and spillways were built to expand the area of existing wetlands to 2.5 ha.

Kingston: A small natural stand of cattail and bulrush stand was enlarged by grading and dike construction in August and September 1986 to create a 0.9 ha treatment area consisting of 3 vegetated cells.

Colbert: Construction of four earthen dikes to increase the size of a natural wetland and enlarge the treatment area to 1 ha in July 1987.

DESIGN CRITERIA

Widows: Average flow rate is 170 L/min with 186 mg/L Fe, yielding a treatment area of 0.2 m^2/mg Fe/min.

Kingston: Average flow was 1370 L/min with 170 mg/L Fe for a loading rate of 0.06 m^2/mg Fe/min.

Colbert: Average influent flow was 496 L/min with 7.0 mg/L Mn yielding a treatment area of 1.3 m^2/mg Mn/min.

ADDITIVES

Widows: Fertilizer applied after planting at 6.75 MT/ha. In January 1987 a drip feed NaOH system was installed at the inlet to cell 3, and discharges have met compliance standards to date.

Kingston: Lime and fertilizer, mushroom compost. Final effluent will be pumped to the ash pond for treatment before discharge until the wetlands system performance improves.

Colbert: NaOH treatment was continued in the final two cells of the system to reduce Mn to permit limits.

COSTS

System	Area (ha)	% Equip.	% Labour	% Overh.	Total ($000's)	$/$m^2$
Colbert	0.9	32.4	30.4	37.2	7.9	1.19
Widows	2.9	--	--	--	209.0	6.98
Kingston	0.9	28.6	43.3	28.1	131.7	14.21

TIME OF OPERATION / INFORMATION

Widows: 1984 - 1988
Kingston: Late summer 1986 - 1988
Colbert: July 1987 - 1988

PERFORMANCE DATA

Widows: Influent
Average total Fe 186 mg/L
Average total Mn 7.1 mg/L
Average pH 6.0
Average flow rate 170 L/min
Treatment area 0.2 m^2/mg/Fe/min.
System removed over 97% of the Fe, but only 9% of the Mn, and average discharge pH decreased 2.1 during the first eight months of operation.

Kingston: Influent
Total Fe 40 - 45 mg/L
Total Mn 4 - 5 mg/L
pH 5.5
Average flow rate 1370 L/min
System removed 85% total Fe and little Mn; pH consistently decreased 3

Colbert: Average influent flow was 496 L/min with 7.0 mg/L Mn yielding a treatment area of 1.3 m^2/mg Mn/min. Fe was less than 1.0 mg/L.
Effluent flow had a pH of 6.6, total Fe < 1 mg/L, higher levels of total Mn (22.5 mg/L).

OVERALL EFFECTIVENESS

Fe removal efficiency was higher than Mn for all three systems.

TOXICITY n.d.

MAINTENANCE REQUIREMENTS

Widows: Treated for a heavy infestation of armyworms (*Simyra henrici*) in August and October 1986, and again in July and September 1987.

Kingston: Bare areas were re-planted, fertilized in 1987 and 1988; compost and crushed limestone applied in 1988.

TECHNOLOGY	Constructed Wetlands WTLD 5
MONITORING REQUIREMENTS	Flow, metals, pH.
TREATMENT PRODUCT DISPOSAL	n.d.
ENVIRONMENTAL CONDITIONS	n.d.
COMMENTS	Kingston and Widows lacked sufficient treatment for the loading rate. High length to width ratios at Kingston may have reduced treatment efficiencies. Colbert discharge Mn has been high, this may be due to construction disturbance. Potential remedies include the possible lack of Fe-Mn coprecipitation at Colbert, alkalinity buffering and sulfate reduction mechanisms for Widows and Kingston, and flow reductions or increased treatment areas at Widows and Kingston.

TECHNOLOGY	Constructed Wetlands WTLD 6
PROJECT SCALE	Full scale
REFERENCE	"Staged, Aerobic Constructed Wetlands to Treat Acid Drainage - Case History of Fabius Impoundment 1 and Overview of the Tennessee Valley Authority's Program". Brodie, Gregory A. 1993. Wetland Design for Mining Operations. pp. 1 - 14.
LOCATIONS	Impoundment 1 (IMP1) is located at Fabius Coal Preparation Plant, Alabama.
DESCRIPTION	Fourteen staged aerobic wetlands to treat acid drainage; twelve are operated by the Tennessee Valley Authority. The aerobic wetlands generally consist of a pretreatment stage (anoxic limestone drain and/or oxidation basin) followed by several cells of shallow to deep (0.1 - 2.0 m) cattail marsh-ponds. Some systems are followed by a final polishing pond which improve long-term capacity and minimize storm event flushing of Fe and Mn precipitates from the wetlands. Wetland shapes varied and were dictated by existing topography, geology, and land availability. The number of cells were determined by site topography, hydrology, and water quality. Average water depth ranged from 15 - 30 cm with some deeper and shallower areas to provide for species diversification. Isolated deep pockets of up to 2.0 m were included in many cells to provide for aquatic fauna refuge in drought events. Vegetation was hand dug and planted on the same day as digging. Cattail (*Typha sp.*) was set into the substrate as 0.3 m canters in early systems and 1.0 m canters in later ones. Post-construction activities includes the addition of new ponds to further treat the wetlands discharge.
DESIGN CRITERIA	Same as in "Achieving compliance with staged, aerobic constructed wetlands to treat acid drainage", Brodie, 1991.
ADDITIVES	Wetlands were generally fertilized only once with a phosphorous-potassium fertilizer such as 0-12-12 at 400 kg/ha.
COSTS	Total cost of the IMP1 wetlands was $43,000 (1985 dollars). Annual costs from 1985 - 1990 were about $13,000 due to repairs and to extensive monitoring. Current operations and maintenance costs are less than $1,000 annually. Costs to chemically treat the acid drainage entering the wetland would have been approximately $250,000 from 1985 - 1990.
TIME OF OPERATION / INFORMATION	IMP1 was constructed in May 1985. Data collection period - July 1985 to October 1991 and on going.
PERFORMANCE DATA	Same as in "Achieving compliance with staged, aerobic constructed wetlands to treat acid drainage", Brodie, 1991. IMP1 reduced total Fe from 69 mg/L to 0.9 mg/L and total Mn from 9.3 mg/L to 1.6 mg/L. pH has been increased from 3.5 to 6.8. Wetlands were on average hydraulically loaded between 0.02 - 0.24 L/day/m^2 (LDM). Maximum hydraulic loading ranges were 0.06 - 1.47 LDM and averaged 0.42 LDM. Fe: Ten of the 12 wetlands produced discharges in compliance with Fe limitations. Loading on the systems ranged from 0.03 GDM to 41.4 GDM. Fe removal rates ranged from 0.0 GDM to 21.3 GDM or 0 to 99% Fe removal. Mn: Nine wetlands produced discharges in compliance with total Mn limitations. Mn loading ranges from 0.17 GDM to 2.00 GDM. Mn removal ranges from 0.15 GDM to 1.87 GDM, corresponding to 0 to 96% removal. Systems with zero alkalinity have removed 0 to 16.5 %, while systems with alkalinity greater than 62 mg/L and with excess acidity as high as 248 mg/L have removed 85 - 97% of the Mn load. pH: Nine of the systems increase or maintain inflow pH to produce discharges in compliance for pH. Three systems cause pH reductions due to Fe oxidation and hydrolysis, and are being modified with anoxic limestone drains. TSS: All systems produced discharges in compliance with TSS limitations.
OVERALL EFFECTIVENESS	Significant water quality improvement has occurred at all of the 12 operating wetlands. Nine systems have produced discharges that consistently meet NPDES monthly average discharge limitations (pH = 6 - 9; Fe < 3.0 mg/L; Mn < 2.0 mg/L; TSS < 35 mg/L) with no chemical treatment. Where regulatory limits were not entirely achieved, cost savings were realized as a reduction in chemicals needed for further metals precipitation via pH adjustment. pH has increased primarily due to a limestone roadbed located under the earth dam.
TOXICITY	n.d.
MAINTENANCE REQUIREMENTS	n.d.
MONITORING REQUIREMENTS	Monitoring requirements included pH, total Fe and Mn, and total suspended solids (TSS). Effluent samples from the wetlands were taken during daylight hours generally within the second and fourth weeks of the month. Sampling was always initiated within two weeks of system startup.
TREATMENT PRODUCT DISPOSAL	n.d.
ENVIRONMENTAL CONDITIONS	n.d.

TECHNOLOGY Constructed Wetlands WTLD 6

COMMENTS

Based on the results, aerobic wetlands systems should be designed for 4.0 - 110.0 GDM of Fe removal depending on pH, alkalinity, and Fe concentrations of inflow. One wetland cell should be constructed for each 50 mg/L Fe inflow due to the need for re-aeration of oxidation of this amount of Fe.

Fe: Removal in the wetlands is very efficient for loadings up to 13 GDM, and less efficient where loading exceeds 41 GDM. There is no correlation between Fe removal and influent alkalinity, Fe removal and hydraulic loading.

Mn: The low removal rates were all associated with low pH (2.9 - 3.9) systems. Removal was very efficient for loadings as high as 2.0 GDM. There is no correlation between Mn removal and wetland size, and hydraulic loading. There is a good correlation between removal and influent alkalinity and acidity concentrations. Low Mn removal is associated with zero alkalinity. Data suggests than Mn co-precipitation on Fe-oxides at circum-neutral pH is a likely mechanism of Mn removal.

A preferred design for a wetland system may consist of: an initial anaerobic limestone trench at the source of the seepage to passively add alkalinity; a large, deep settling basin to accumulate oxidized and precipitated Fe sludges; and a two or three cell constructed wetland for Mn and further Fe removal.

Factors affecting the ability of wetlands to ameliorate acid drainage include hydrology, Fe and alkalinity concentrations, and wetland characteristics such as depth, area, hydraulics, vegetative and microbial species and extent, and substrate.

TECHNOLOGY	Constructed Wetlands WTLD 7
PROJECT SCALE	Full scale
REFERENCE	"Treatment of Acid Drainage with a Constructed Wetland at the Tennessee Valley Authority 950 Coal Mine". Brodie, G.A., Hammer, D.A. and D.A Tomljanovich. 1989. Constructed Wetlands. pp. 201 - 209.
LOCATIONS	Near Flat Rock, Jackson County, Alabama
DESCRIPTION	Impoundment 3 was a sediment basin receiving acid mine drainage from the 950 coal mine converted to a three celled wetland. Cell 1 was filled and regraded to a uniform flat bottom using borrow from a sandstone spoil band. Cells 2 and 3 were graded uniformly, and a dike constructed using compacted material from cells 2 and 3 which consisted of alluvially derived silt loams. Two spillways consisted of channels lined with erosion fabric planted with aquatic vegetation; the final spillway was riprapped and fitted with a discharge pipe for monitoring flow. Freeboard was 0.5 m, pond depths averaged between 0.3 m and 0.5 m. The final size was 0.13 ha. Hydraulic loading was 0.7 L/m^2/min, and a chemical loading of 1.1 m^2/mg/min for Fe and 2.8 m^2/mg/min for Mn.
DESIGN CRITERIA	Designed to meet effluent criteria. The hydraulic loading was approximately 0.7 L/m^2/min, and a chemical loading of 1.1 m^2/mg/min for Fe and 2.8 m^2/mg/min for Mn.
ADDITIVES	Disturbed areas were seeded, mulched, and fertilized with a mixture of grasses. Ponds were planted with 1600 cattails (*Typha latifolia*) and woolgrass (*Scripus cyperinus*), fertilized with 12-12-12 at 450 kg/ha, and limed at 9000 kg/ha.

COSTS

Item	Cost ($)
Construction	5200
Construction equipment	4700
Materials and supplies	2800
Labour	18200
Supervision and administration	5300
Design and site evaluation	5000
Total wetland capital cost	41200
Annual operation	3700
Total pre-wetlands annual cost	28500

TIME OF OPERATION / INFORMATION	Construction initiated in the late summer of 1986. Data collection from November 20, 1986 to June 23, 1988.
PERFORMANCE DATA	Generally, the wetland has increased pH from 6.1 to 6.9, reduced total Fe from 14.3 mg/L to 0.8 mg/L, reduced total Mn from 4.8 mg/L to 1.1 mg/L, and total suspended solids from 24 mg/L to 7 mg/L.
OVERALL EFFECTIVENESS	Six months after wetland construction 17 additional aquatic species were found in the receiving waters compared to a previous total of 2. Twenty vegetative species were present in the impoundment 13 months after construction, with dominant species: *Typha*, *Scirpus*, and *Juncus acuminatus*.
TOXICITY	n.d.
MAINTENANCE REQUIREMENTS	n.d.
MONITORING REQUIREMENTS	pH, total Fe, total Mn.
TREATMENT PRODUCT DISPOSAL	n.d.
ENVIRONMENTAL CONDITIONS	n.d.
COMMENTS	Hydraulic loading rates were below average when compared to other wetland treatment installations by TVA. Results from Impoundment 3 suggests that wetlands may be a long-term, self maintaining treatment system capable of producing high-quality water from moderately polluted inflows.

TECHNOLOGY	Constructed Wetlands WTLD 8
PROJECT SCALE	Laboratory / pilot scale
REFERENCE	"Application of Constructed Cattail Wetlands for the Removal of Iron from Acid Mine Drainage". Calabrese, J.P. *et al.* 1990. Second International Conference on the Abatement of Acidic Drainage. Volume 3. pp. 559 - 575.
LOCATIONS	West Virginia
DESCRIPTION	Four simulated wetlands constructed in a greenhouse environment. Dimensions: 480 cm x 60 cm x 60 cm. Planting Medium: 40 cm mixture of commercially purchased Sphagnum peat moss (90% peat moss, 10% soil), underlain with 15 cm of limestone gravel The wetlands were inoculated with a mixed anaerobic sediment slurry to introduce sulfate-reducing bacteria prior to planting from a volunteer *Typha* wetland successfully treating AMD seeps from a reclaimed surface mine permit. *Typha latifolia* collected from two field sites were mixed and planted in a regular pattern to a density of 32 plants per wetland. Root systems were allowed to establish and stabilize for 3 to 6 months before AMD addition. Average detention time was four days. Two cells were established as controls.
DESIGN CRITERIA	Naturally occurring AMD was added at a rate of 60 L/day. Control cells received tap water at an equivalent rate.
ADDITIVES	n.d.
COSTS	n.d.
TIME OF OPERATION / INFORMATION	10 to 12 months.
PERFORMANCE DATA	AMD composition (influent) averaged: Fe 438 mg/L; Mn 4 mg/L; sulphate 2900 mg/L; total acidity 1835 mg/L, and a pH of 2.75. Collected pore water showed a decrease in Eh from -30 to -300 mV, and an increase in pH, from 2.0 to 6.5, with depth and distance respectively. Sulfate concentrations decreased with depth (ca. 200 mg/kg), due to the establishment of active sulfate-reducing bacteria. Reduced iron sulfides accumulated as 80% acid volatile sulfides/20% pyrites. Dissolved Ca increased with depth indicative of continuing limestone dissolution due to the permanent anoxic conditions.
OVERALL EFFECTIVENESS	Overall, the wetlands retained nearly 66% of the total applied Fe over a one year period.
TOXICITY	n.d.
MAINTENANCE REQUIREMENTS	n.d.
MONITORING REQUIREMENTS	Flow, load, Fe concentration. Wetland sediments collected and analyzed.
TREATMENT PRODUCT DISPOSAL	n.d.
ENVIRONMENTAL CONDITIONS	n.d.
COMMENTS	Iron sulfide formation could account for between 20 to 28% of the total iron removed from AMD during passage through the wetlands. An advantage of mixed aerobic/anaerobic wetlands which are underlain with limestone is the production of alkalinity both by sulfate-reduction to sulfides and from the continuous dissolution of calcium carbonates under anoxic sediment conditions. Production of dense, relatively stable iron sulfides represents a renewable metal retention mechanism which is unavailable in aerobic systems.

TECHNOLOGY	Constructed Wetlands WTLD 9
PROJECT SCALE	Pilot scale
REFERENCE	"Evaluation of Acidic Mine Drainage Treatment in Constructed Wetland Systems". Dietz, J.M., Watts, R.G. and D.M. Stidinger. 1994. International Land Reclamation and Drainage Conference and Third International Conference on the Abatement of Acidic Drainage. pp. 70 - 79.

TECHNOLOGY

Constructed Wetlands

WTLD 9

LOCATIONS

Six wetland treatment systems at three sites: Canoe Creek, Jennings Environmental Education Center, and Cucumber Run, Pennsylvania.

DESCRIPTION

Surface flow wetland treatment system.

Multi-cell design with each unit comprised of 0.3 to 0.5 m deep of spent mushroom compost, a water depth of 0 to 0.1 m, and locally transplanted *Typha latifolia* and other plant species. Two sites (K2 and CUC1) included an initial open water basin. K2 incorporated limestone below the compost substrate. Size was based on available area for construction. K2 contained a pretreatment unit and four compost units for a total treatment area of 952 m^2. K3 contained three compost units for a total treatment area of 224 m^2. JEEC contained four treatment units at a total treatment area of 1,161 m^2. CUC1 contained one treatment unit with a total area of 1,340 m^2, but contained an open water area that reduced the compost area to 1,140 m^2. CUC2 contained two compost units for an area of 92 m^2. CUC3 contained four compost units for a total treatment area of 460 m^2.

DESIGN CRITERIA

Designed to reduce AMD impacts on receiving streams and to achieve the treatment goals for Fe (3.0 mg/L), Al (5.0 mg/L), and Mn (7.0 mg/L).

ADDITIVES

n.d.

COSTS

n.d.

TIME OF OPERATION / INFORMATION

Wetland
K2: August 1988 to May 1991
K3: July 1988 to May 1991
JEEC: April 1989 to February 1991
CUC1: June 1990 to December 1991
CUC2: June 1990 to December 1991
CUC3: June 1990 to December 1991

PERFORMANCE DATA

Influent

Wetland	Total Fe (mg/L)	Total Al (mg/L)	Total Mn (mg/L)	Lab pH	Acidity (mg/L)	Sulphate (mg/L)
K2[1]	98.3	0.8	12.2	4.64	126.5	594
K3	41.0	2.4	30.7	5.20	114.3	1388
JEEC	22.4	18.5	6.2	2.90	258.4	516
CUC1	85.6	15.0	3.9	2.60	392.0	303
CUC2[1]	36.3	1.2	4.2	3.27	108.3	718
CUC3	10.6	4.2	4.8	3.12	106.5	332

[1] - Influent values are based on flow adjusted averages of multiple influent data.

Effluent

Wetland	Total Fe (mg/L)	Total Al (mg/L)	Total Mn (mg/L)	Lab pH	Acidity (mg/L)	Sulphate (mg/L)
K2	6.7	0.42	8.5	4.90	6.0	462
K3	2.7	3.4	28.0	4.41	40.6	1202
JEEC	14.5	14.3	6.0	3.33	153.6	519
CUC1	61.9	12.0	4.1	2.70	272.0	312
CUC2	12.9	1.0	4.8	3.51	56.2	754
CUC3	5.5	3.0	4.3	4.01	26.8	313

On average significantly lowered metal concentrations for Fe by 35% to 95%, Al by 0% to 50%, and Mn by 0% to 30%, and acidity concentrations by 40% to 90%.

OVERALL EFFECTIVENESS

Performance did not substantially diminish for the two year duration. Observed fluctuations in effluent quality were due to variable influent quality. Acidity removal was not affected by changes in influent AMD chemistry.

TOXICITY

n.d.

MAINTENANCE REQUIREMENTS

n.d.

MONITORING REQUIREMENTS

Influent and effluent sampling and sampling from intermediate points between individual treatment units done biweekly and monthly. Water samples analyzed for pH, alkalinity, acidity, sulfate, total iron, ferrous iron, total manganese, total aluminium, and hardness. Water flow was measured to evaluate loadings and removal rates.

TREATMENT PRODUCT DISPOSAL

n.d.

ENVIRONMENTAL CONDITIONS

n.d.

COMMENTS

A surface flow wetland treatment system design criteria of 6 $g/d/m^2$ for acidity removal is recommended to predict sizing requirements for future wetland treatment system construction.

TECHNOLOGY	Constructed Wetlands WTLD 10
PROJECT SCALE	Pilot scale
REFERENCE	"Anoxic Cattail Wetland for Treatment of Water Associated with Coal Mining Activities". Duddleston, K.N., Fritz, E., Hendricks, A.C. and K. Roddenbery. 1992. Achieving Land Use Potential Through Reclamation, Duluth Minnesota. pp. 249 - 259.
LOCATIONS	Norton, Virginia
DESCRIPTION	An anoxic subsurface flow wetland constructed to treat AMD through the enhancement of bacterial sulfate reduction. The wetland was 0.7 acres (200 feet in length x 150 feet in width), with a one foot deep limestone substrate beneath one foot of weathered pine bark mulch. Designed to prevent channelling and increase retention time and to force water through the anaerobic zone. Six inch perforated pipes were placed in the limestone bed to serve as an underground drain. Cattails planted in selected areas only.
DESIGN CRITERIA	Average flow was 20 gpm.
ADDITIVES	n.d.
COSTS	Annual costs for chemical treatment were $7,200 (U.S.) before wetland construction. Annual costs for chemical treatment averaged $1,000 (U.S.) after wetland construction. Cost of converting one of the three sediment ponds into a wetland: $25,800 (U.S.).
TIME OF OPERATION / INFORMATION	Influent and effluent water samples were collected from January 1991 to February, 1992; substrate samples were taken once monthly from February 1991 to August 1991.
PERFORMANCE DATA	Influent and effluent pH was 7.0 Fe concentration: decrease from 5 mg/L to 2 mg/L (average) Mn concentration: decrease from 4 mg/L to 1.5 mg/L (average) Sulfate concentration: decrease from 400 mg/L to 200 mg/L (average) Population of sulfate reducing bacteria: increase of 1.2 x 10^4 to 3.7 x 10^5 microorganisms per gram dry substrate. No significant difference in the population size of sulfate reducers between areas with and without cattails.
OVERALL EFFECTIVENESS	pH, Fe, and Mn met instream compliance standards, and no additional chemical treatment was necessary. The state of Virginia requires a pH of 6.0 - 9.0, an in stream iron concentration of 2 mg/L, and in stream manganese of 1 mg/L. The reduction of sulfate was consistent over the sampling period, dropping an average of 360 mg/L as the AMD passed through the wetland.
TOXICITY	n.d.
MAINTENANCE REQUIREMENTS	n.d.
MONITORING REQUIREMENTS	Flow rate, pH, Fe and Mn concentrations, bacteria population. Comparison of sulfate reducing bacteria population size between areas with and without cattails. Inlet and outlet water sampled and analyzed weekly. Substrate samples were taken monthly to enumerate sulphate reducing bacteria. Samples (50 g) were taken from random areas within the wetland using a pitchfork.
TREATMENT PRODUCT DISPOSAL	n.d.
ENVIRONMENTAL CONDITIONS	n.d.
COMMENTS	Weather problems and difficulties with the construction of the boxes intended to compare treatment capability between different substrates. Wetland went through a 2 month establishment period in which successful treatment of iron and manganese was not evident. A determination of optimum bacterial densities for the removal of metal, how these densities can be achieved, and information about the environmental requirements that regulate their activities is needed. The wetland must be designed to take advantage of the sulfate reducing capacity available.

TECHNOLOGY Constructed Wetlands WTLD 11

PROJECT SCALE

Pilot scale

REFERENCE

"The Design of a Wetland Treatment System to Remove Trace Metals from Mine Drainage". Eger, P. and G. Melchert. 1992. American Society for Surface Mining and Reclamation Meeting, Duluth Minnesota.

LOCATIONS

Duluth Complex stockpiles, Dunka Mine, northeastern Minnesota.

DESCRIPTION

The Duluth Complex material contains Cu, Ni, and Fe sulfides. Discrete seepages appear at the base of the stockpiles and generally flow continuously from early April to Late November.

Four cells were constructed in a natural wetland. Each cell was 6 m x 60.5 m long and was surrounded by a compacted peat berm. To hydrologically isolate the cells, a sand-bentonite cut-off ditch was installed in the centre of the berms surrounding each cell. A design residence time of 40 - 48 hours was selected for the cells.

Cell 1: Unmodified natural wetland; vegetation primarily sedges (*Cares sp.*) and grasses (*Calamograstis sp.*); water depth was 5 cm.

Cell 2: Modified wetland; shallow trenches spaced about 4.5 m apart and were about 60 cm deep, dug perpendicular to the flow path; sedges and grasses from the surrounding area were translated into the cell, water depth was 5 cm.

Cell 3: Modified wetland; hay bales placed to create serpentine flow, 5 cm of straw placed on the bottom of the cell to encourage sulfate reduction; cell was planted with cattails (*Typha latifolia*) at a spacing of 1 per metre; water depth was 15 cm.

Cell 4: Modified wetland; peat berms constructed across the cell, perpendicular to flow; cattails planted at 1 m spacing; water depth was 15 cm. In 1991 15.2 cm of a mixture of 1 part well decomposed reed sedge peat from an unimpacted wetland to 2 parts peat screenings from a shpagnum peat processing facility. Water depth varies from 0 to 5 cm.

DESIGN CRITERIA

Average flows from the various seepages range from 0.5 L/sec to 14 L/sec. Flows exceeding 100 L/sec were observed after periods of heavy precipitation.

The goal was to collect data for the design of full-scale treatment systems for the stockpile drainages. Cells were designed so that a variety of water levels, vegetation, and flow regimes could be tested.

ADDITIVES

n.d.

COSTS

n.d.

TIME OF OPERATION / INFORMATION

Data collection began in August 1989.

PERFORMANCE DATA

Annual median influent concentrations of Ni is on the order of 3 - 30 mg/L. Cu, Co and Zn were generally less than 5% of the Ni values. Median pH ranges from 5.0 to 7.5, but most of the stockpile drainages have pH greater than 6.5.

The input stockpile drainage can be characterized as a high hardness neutral drainage whose primary contaminant is Ni. Ni concentrations into the cells have ranged from 0.11 to 3.8 mg/L. In 1991, after the collection point for the stockpile was moved, average nickel concentrations of the input water increased from 0.66 mg/L in 1989 and 1990 to 2.0 mg/L in 1991. Average hardness is around 2300 mg/L as $CaCO_3$ with a pH range of 6.5 - 7.9. Cu and Zn generally met water quality criteria, Co and Ni routinely exceeded the criteria, sometimes by more than an order of magnitude.

Nickel removal for 1989 - 1990 (average Ni input concentration was 0.56 mg/L, input concentration ranged from 0.11 to 2.1 mg/L)

Cell	Average Outflow Nickel	% Red. in Conc.	Overall Mass Removal (%)	Water Level (cm)
1	0.10	85	83	5
2	0.09	87	86	5
3	0.23	66	68	15
4	0.46	32	40	15

TECHNOLOGY Constructed Wetlands WTLD 11

PERFORMANCE DATA CONT..

Outflow water quality as a function of residence time for Cell 4, 1991:

Time Period	Flow Rates (gpm)		Residence Time (hrs)	Ni Conc. mg/L		% Red. in Ni Conc.
	In	Out		In	Out	
7/10 - 7/30	1.3	1.3	28	1.9	0.11	94
7/31 - 8/21	2.0	1.8	20	2.0	0.18	91
8/22 - 10/9	1.3	1.2	31	2.1	0.15	93
10/10 - 10/24	2.2	2.1	17	2.0	0.9	55
10/25 - 11/22	3.1	3.1	12	1.7	1.1	38

Note: All values are averages. For period 7/10 - 7/30 outflow = inflow because of 9.5 cm of rain during this period.

Outflow water quality as a function of residence time for Cell 1, 1991:

Time Period	Outflow Rates (gpm)	Residence Time (hrs)	Ni Conc. mg/L		% Red. in Ni Conc.
			In	Out	
6/13 - 8/1	1.0	40	2.241	.0147	93
8/2- 9/6	1.4	29	2.058	0.318	85
9/7 - 9/27	1.9	21	1.950	1.150	41
9/28 - 10/9	0.9	44	1.820	1.670[1]	8
10/10 - 11/13	0.6	67	1.878	1.251	33

[1] only one data point.

OVERALL EFFECTIVENESS

In 1989 and 1990, the largest reduction in Ni concentration occurred in the shallow water cells (Cells 1 and 2). Cell 4 consistently has the lowest Ni removal. When the peat mixture was added to Cell 4 in 1991, Ni removal increased dramatically. In general, for the shallow waters, residence times of 48 hours provided good removal. For Cell 4, Ni removal decreased as residence time decreased. Reduction of Ni concentration also appears to depend on the time of year, with removal rates decreasing in the fall.

TOXICITY

n.d.

MAINTENANCE REQUIREMENTS

n.d.

MONITORING REQUIREMENTS

Flow, Ni concentration, residence time.

TREATMENT PRODUCT DISPOSAL

n.d.

ENVIRONMENTAL CONDITIONS

n.d.

COMMENTS

Key factors in any wetland design should include: drainage characteristics such as average and peak flow, water quality, mass loading, and anticipated changes over time; wetland characteristics such as size, type of system (overland or subsurface flow) and type of substrate. Design factors should include effluent requirements (both average and maximum allowable concentration), residence time, performance data which should include average and minimum metal removal and seasonal effects, and finally the expected lifetime of the wetland.

To have significant sulfate reduction, the following must occur:

- an aerobic zone must be established;
- a source of readily decomposable organic matter must be present;
- elevated sulfate concentrations must occur in the drainage, and;
- transport of the mine drainage to the anaerobic zone must occur.

TECHNOLOGY

Constructed Wetlands WTLD 12

PROJECT SCALE

Pilot scale

REFERENCE

"The Use of Wetlands to Remove Trace Metals from Mine Drainage". Eger, P., Melchert, G., Antonson D. and J. Wagner. 1991. International Symposium, Constructed Wetlands for Water Quality Improvement.

LOCATIONS

Duluth Complex stockpiles, Dunka Mine, northeastern Minnesota.

DESCRIPTION

Four cells constructed in a natural wetland. Each cell was 6 m x 30.5 m and was surrounded by a compacted peat berm.

Cell 1: Unmodified natural wetland (wet meadow, type 2); water dispersed across natural wetland; vegetation was primarily sedges (*Carex* spp.) and grasses (*Calamograstis* spp.); water depth was 5 cm.

Cell 2: Modified wetland; shallow trenches were constructed with a backhoe spaced approximately 4.5 m apart and approximately 60 cm deep, dug perpendicular to the flow path; sedges and grasses from the surrounding area were transplanted into the cell; water depth was 5 cm.

Cell 3: Modified wetland; hay bales placed to create serpentine flow, 5 cm of straw placed on the bottom of the entire cell to encourage sulfate reduction; cell planted with cattails (*Typha latifolia*) at a spacing of 1 per metre; water depth was 15 cm.

Cell 4: Modified wetland; peat berms constructed across the cell, perpendicular to flow; *Typha latifolia* planted at a spacing of 1 per metre, water depth was 15 cm.

DESIGN CRITERIA

The cells were designed so that a variety of water levels, vegetation, and flow regimes could be tested. A design residence time of 40 - 48 hours was selected. Average flows from the various seepages range from 0.5 L/sec to 14 L/sec, maximum flows exceeding 100 L/sec were observed after periods of heavy precipitation. The input flow rates into Cells 3 and 4 were decreased from 11.4 to 5.3 L/min in order to increase the residence time to around 48 hours.

ADDITIVES

n.d.

COSTS

n.d.

TIME OF OPERATION / INFORMATION

August 1989 to October 1990.

PERFORMANCE DATA

Summary of wetland treatment results for Cell 1.

Source and Date	Input Range (mg/L)	Output Range (mg/L)	Mass Removal	Residence Time (hrs)
W3D 8/1/90 - 9/16/90	pH 6.5-7.9	6.5-7.7	NA	18-48*
	Cu 0.002-.002	0.001-0.010	NS	
	Ni 0.11-2.1	0.020-0.39	87	
	Co 0.003-0.047	0.002-0.043	NS	
	Zn 0.02-0.07	0.006-0.08	33	
Seep 1 + W3D 9/17/90 - 10/5/90	pH 6.6-7.4	6.6-7.45	NA	34
	Cu 0.11-0.18	0.006-0.023	90	
	Ni 4.27-6.12	0.04-0.63	94	
	Co 0.27-0.5	0.006-0.050	90	
	Zn 0.65-0.94	0.019-0.037	96	
Aquifer X 10/11/90 - 10/19/90	pH 6.65-7.15	6.45-7.00	NA	34
	Cu 0.08-0.24	0.005-0.007	95	
	Ni 1.75-7.98	0.56-0.64	59	
	Co 0.18-0.26	0.003-0.008	97	
	Zn 1.42-1.76	0.024-0.058	96	
Aquifer X 10/20/90 - 10/26/90	pH 6.73-7.30	6.73-7.35	NA	22
	Cu 0.21-0.35	0.004-0.022	73	
	Ni 1.91-1.96	0.95-1.20	41	
	Co 0.21-0.24	0.029-0.040	81	
	Zn 1.87-1.96	0.055-0.320	90	

* - 30% of values exceeded 48 hours.
Overall Mass Removal in %.

OVERALL EFFECTIVENESS

In general for both years there was little difference between the input and output concentrations for the major cations and anions (Ca, Mg, Na, K, and sulfate), and for the trace metals Cu and Co.

Output concentrations for iron, alkalinity and acidity were slightly higher than the input values, while significant decreases in output concentrations were observed for pH, Ni, and Zn. pH decreased by about 0.2 - 0.3 pH units. Average Zn concentrations decreased from 0.03 mg/L in the input water to 0.01 - 0.02 in the outflows of all cells.

Every cell was successful in removing Ni from solution. In general the cells which contained 5 cm of water were more effective in reducing Ni concentrations. Lower water levels and longer residence times probably account for the increased removal. Shallow cells removed less Ni, but were more efficient than the deeper cells.

TECHNOLOGY	Constructed Wetlands WTLD 12
TOXICITY	n.d.
MAINTENANCE REQUIREMENTS	n.d.
MONITORING REQUIREMENTS	Flow; influent and effluent water quality; residence time.
TREATMENT PRODUCT DISPOSAL	n.d.
ENVIRONMENTAL CONDITIONS	n.d.
COMMENTS	Measures were taken to reduce seepage loss to a minimum. All cells functioned according to design except Cell 4. In Cell 3, the original serpentine path was disrupted when the hay bales subsided. Most of the flow in the treatment cells occurs across the surface, therefore, the deeper the water, the less contact there will be with the metal removal sites on the peat.

TECHNOLOGY Constructed Wetlands WTLD 13

PROJECT SCALE Pilot or full scale

REFERENCE "Treatment of Acid Mine Drainage by Passive Treatment Systems". Faulkner, B.B. and J.G. Skousen. 1994. International Land Reclamation and Mine Drainage Conference and the Third International Conference on the Abatement of Acidic Drainage, Pittsburgh, PA.

LOCATIONS West Virginia

DESCRIPTION Five wetland systems (Keister 1, Keister 3/2, S. Kelly, Pierce, and Z & F) were constructed. Each wetland was designed specifically for the site. In general, 0.6 to 1 m of organic material composed of peat and hay overlie about 15 to 30 cm of limestone. Hay bales were usually used as barriers to slow and direct water through the wetland. All wetlands except the Keister wetlands has about 23 cm of limestone at the base.

S. Kelly: The design attempted to encourage subsurface flow under and through the organic strata. The flow was directed downward into the substrata by the use of a geotextile fabric placed under and on the upstream side of the hay bale barriers.

Pierce: The design employed the classic approach to wetland construction and uses surface flow over a limestone enriched, organic substrate.

Z & F: Construction employed organic substrates but encouraged subsurface flow by means of 6-inch plastic pipes under earthen barriers.

DESIGN CRITERIA Designed to treat flows ranging from 4 to 98 L/min and acidity concentration from 170 to 2400 mg/L.

Site	Limestone(mt)	Flow(L/m)	Area(m^2)	Materials
Keister 1	0	17	408	compost
Keister 3/2	0	19	929	compost
S. Kelly	780	95	1417	LS, compost, no plts
Pierce	450	98	813	LS, compost
Z & F	476	30	863	LS, compost

ADDITIVES n.d.

COSTS Keister 3/2: The capital cost of installing the Keister 3/2 wetland,f $225,000, suggests that chemical treatment might have been cheaper with caustic soda costing $8,500 per year or ammonia at $9,700 per year.

Z & F: Construction cost was $110,000. Average removal of the acid load was 67% for a total of 25.5 mt (28 st) of acid per year. Chemical treatment for this amount of acid would be $22,000 per year with caustic soda. This cost would likely double with additional costs for attendant labour and sludge handling.

TIME OF OPERATION / INFORMATION Wetlands were constructed in 1990. Data covers approximately 3 years of operation.

PERFORMANCE DATA

Site	pH		Acid mg/L		Fe mg/L		Mn mg/L		Al mg/L		SO_4 mg/L	
	In	Out	In	Out	In	Out	In	Out	In	Out	In	Out
Keister 1	3.1	5.4	252	59	23	9	23	20	27	13	673	466
K'str 3/2	4.0	4.8	106	56	11	3	11	11	10	8	356	347
S. Kelly	2.9	3.1	1383	1366	105	33	51	50	157	163	2558	2660
Pierce	3.3	4.4	118	57	10	2	8	7	9	7	217	227
Z & F	2.5	3.5	2388	801	376	86	51	40	206	76	2821	1662

Reduction of Acidity

Site	Load in[1] g/day	Load out g/day	% Removal	Load/area gdm²
Keister 1	6015	1413	76	14.7
Keister 3/2	2885	1524	47	3.1
S. Kelly	189187	186629	3	133.5
Pierce	16378	7912	52	20.1
Z & F	105287	35316	67	122.0

[1] Loads were calculated by L/m x mg/L x 1.44 = g/day

Removal of Iron

Site	Load in g/day	Load out g/day	% Removal	Load/Area gdm²
Keister 1	549	215	62	1.3
Keister 3/2	299	82	73	0.3
S. Kelly	14345	4509	68	0.1
Pierce	1388	278	80	1.7
Z & F	16578	3792	77	19.2

TECHNOLOGY	Constructed Wetlands WTLD 13
OVERALL EFFECTIVENESS	Results show that the Keister wetlands without limestone are similar in acid and iron removal as wetlands with limestone. The Z & F wetland has reduced metals consistently since its construction. Only a mild pH enhancement has been seen despite removal of 67% of the acidity. Much of the acidity reduction is associated with the removal of iron (77%) and perhaps aluminium (63%), while manganese has sometimes been decreased and sometimes increased through the system.
TOXICITY	n.d.
MAINTENANCE REQUIREMENTS	n.d.
MONITORING REQUIREMENTS	Flow, acidity, metals concentration.
TREATMENT PRODUCT DISPOSAL	n.d.
ENVIRONMENTAL CONDITIONS	All wetland systems exhibit seasonal variability with respect to acidity and metal removal efficiency. Generally, removal efficiencies increased during the summer/fall and decreased during winter/spring.
COMMENTS	Keister 3/2: The site is very remote with poor access and the intermittent flows make chemical treatment inefficient. The receiving stream has not had to bear wide fluctuations in pH that are common to chemical treatment systems with intermittent flows. S. Kelly: The attempt to direct the flow was unsuccessful and most of the flow appeared to pass through and over the hay bale dikes, rather than under them. Pierce: The system has averaged about 42% removal of acidity and about 80% of the iron with substantial seasonal variation. It has consistently removed a small percentage of aluminum (25%) and manganese (11%) except during large flow events. In all cases the limestone was coated and covered by metal precipitates, and it appeared that acid water was not flowing through the limestone because of the metal covering. Limestone substrates in the wetlands did not appear to improve the wetlands metal removal efficiency.

TECHNOLOGY	Constructed Wetlands WTLD 14
PROJECT SCALE	Pilot scale
REFERENCE	"Bell Mine: Experimental Wetlands Project". Gormely Process Engineering, R.U. Kistritz Consultants Ltd. and Microbial Technologies. 1991. British Columbia Acid Mine Drainage Task Force Project.
LOCATIONS	Newman Peninsula, Babine Lake, approximately 64 km northeast of Smithers, B.C.
DESCRIPTION	Ponds: small, large, and batch mixing ponds Batch Mixing Ponds: Dimensions at soil surface: 5 m x 60 m (large) 2.5 m x 20 m (small) Depth: Gravel and Filter 300 mm Soil 300 mm Water 200 mm Freeboard 500 mm Total: 1300 mm Side Slopes: 1.5 H : 1 V Liner: 30 mil HDPE Inflow: 8 L/min (large) 2 L/min (small)
DESIGN CRITERIA	- To determine the performance and capacity of the experimental wetland system to remove metals under different seasonal conditions and varying flows and metal concentrations from the waste stream in a northern climate and mine environment. - To relate the performance and capacity measurements to underlying mechanisms of metal fixation in the sediments, with particular reference to sulphate reducing bacteria. - To survey changes in plant species composition and growth in the experimental ponds. - To determine metal tissue concentrations in the transplanted vegetation. - To develop improved design criteria and demonstration data that will permit a technical and economic assessment of the process as a means of acid mine drainage treatment. Lake water is added to seepage entering the batch (mixing) pond in order to obtain a Cu concentration between 1 to 10 mg/L. Water level can be varied within the two ponds.
ADDITIVES	n.d.
COSTS	n.d.
TIME OF OPERATION / INFORMATION	End of July 1991 and ongoing.
PERFORMANCE DATA	1991 results: copper is being removed to levels below 0.01 mg/L consistently. Input copper concentrations of 0.3 mg/L to 1.0 mg/L yield a discharge from the wetland of 0.005 to 0.012 mg/L. Copper levels for all plant tissue samples ranged from 2.3 to 240 ppm. Below ground tissue metal levels were higher than above ground levels. Highest tissue metal levels were found in cattail roots.
OVERALL EFFECTIVENESS	n.d.
TOXICITY	n.d.
MAINTENANCE REQUIREMENTS	n.d.
MONITORING REQUIREMENTS	Plant tissue samples were analyzed for 29 parameters including phosphorus. Sediment samples were analyzed for 29 parameters and sulphur reducing bacteria counted. Water sampling was done daily by grab samples and composites.
TREATMENT PRODUCT DISPOSAL	n.d.
ENVIRONMENTAL CONDITIONS	n.d.
COMMENTS	

TECHNOLOGY	Constructed Wetlands WTLD 15
PROJECT SCALE	Pilot scale, evaluation of sixteen reactor cells.
REFERENCE	"Palmerton Zinc Superfund Site Constructed Wetlands". Haffner, W.M. 1992. Achieving Land Use Potential Through Reclamation. Duluth Minnesota. pp. 260 - 267.
LOCATIONS	Zinc Corporation of America Superfund Site in Carbon County (Palmerton, PA) about 75 miles northwest of Philadelphia, PA.
DESCRIPTION	Sixteen reactor cells (14 x 17 x 4 feet deep) were arranged in four rows (eight served as nitrification cells, and the remaining eight were denitrification cells), filled with varied ratios of mixed mushroom compost, sphagnum peat, bog peat and two lime sources. Four additional cells (4 x 4 x 4 feet) were filled with high-calcium iron rich material (IRM) to serve as either pre or post treatments for the wetlands. Tanks were planted with *Typha latifolia* sprigs and equilibrated with approximately 1 gal/min influent rates.
DESIGN CRITERIA	Removal of metals and nitrogen from industrial influent. Design criteria focused on maximizing influent contact with the media to promote adsorption/ion exchange reactions and minimizing short circuiting or overland flow. Average inflow was 0.45 gal/min.
ADDITIVES	Inorganic salts ($ZnSO_4$, $MnSO_4$, and NH_4NO_3) were injected to give influent concentrations of 300 mg/L Zn, 30 mg/L Mn, and 20 mg/L NO_3^- three days per every other week.
COSTS	n.d.
TIME OF OPERATION / INFORMATION	Evaluated over a six month period: August 1991 to February 1992.
PERFORMANCE DATA	Nitrification cells removed at least 80% of ammonia, total N, or nitrate. Denitrification cells further reduced to 83% of the concentrations of these same three parameters. IRM pre-treatment removed 63% and 83% of the Mn and Zn respectively, from the 30.1 mg/L Mn and 309 mg/L Zn in the influent. The nitrification and denitrification cells increased total Mn removal to 84% (4.9 mg/L) and 93% (2.1 mg/L). By the time water has passed the final wet meadow cells, 97.6% (0.7 mg/L) of Mn has been removed. Zinc showed a 95% removal (2.8 mg/L) following denitrification. Post-IRM and meadow cell removals for Zn declined to 96% and 97% respectively. This may be due to Zn release from IRM.
OVERALL EFFECTIVENESS	Excellent removal efficiencies were experienced for nitrogen, Mn, and Zn.
TOXICITY	n.d.
MAINTENANCE REQUIREMENTS	n.d.
MONITORING REQUIREMENTS	Flow meters monitored both inflow (0.5 gal/min) and outflow. Water was monitored weekly at fifteen *in-situ* points throughout the wetland for pH, redox, temperature (air and water), dissolved oxygen, and chemical oxygen demand (COD). Other monthly sampling parameters included N, C, alkalinity, electrical conductivity, nutrients and metals.
TREATMENT PRODUCT DISPOSAL	n.d.
ENVIRONMENTAL CONDITIONS	n.d.
COMMENTS	The greatest contributions of vegetation appear to be: to anchor the substrate, fix atmospheric CO_2 for eventual CH_2O release in the rhizosphere, and to provide an oxygenated environment for chemical precipitation to occur in the substrate. Preliminary results suggest that IRM is beneficial for metals removal before contact with wetlands media, however, a metal sorptive or IRM longevity point may be approached. Removal efficiencies of the different substrates was not examined.

TECHNOLOGY

Constructed Wetlands WTLD 16

PROJECT SCALE

Full scale experimental

REFERENCE

"Sizing and Performance of Constructed Wetlands: Case Studies". Hedin, R.S. and R.W. Nairn. 1990. Proceedings of the 1990 Mining and Reclamation Conference and Exhibition, Volume II, Charleston, West Virginia. pp. 385 - 392.

LOCATIONS

Somerset Wetland, Somerset County, PA.
Latrobe Wetland, Westmoreland County, PA.
Friendship Hill, Friendship Hill National Historic Site, Fayette County, PA.

DESCRIPTION

Three Pennsylvania wetlands. All were constructed with a mushroom compost substrate and planted with *Typha* spp.

Somerset treated water draining from 12 year old surface mine spoils. Wetland consists of two cells (277 m^2 and 268 m^2) connected in series. Each cell was constructed with 30 cm of crushed limestone and 45 cm of mushroom compost. A dense growth of cattails covered both cells.

Latrobe treats water draining from both reclaimed surface spoils and an abandoned drift mine. The site consists of 3 rectangular cells in series (694 m^2, 802 m^2, and 1301 m^2). Substrate consists of 10 cm of crushed limestone covered with 30 - 45 cm of mushroom compost. Cattail cover about 80% of the wetland, most of the open water was due to muskrat activity during the summer of 1988.

Friendship Hill Wetland treated water from a small first order stream that drains an abandoned drift mine about 1 km upstream from the site. Site consists of six wetland cells connected in parallel. All cells contain 15 cm of gravel covered with 45 cm of spend mushroom compost. Cells were planted with cattails in October 1988, and their growth resulted in approximately 75% coverage by July 1989.

DESIGN CRITERIA

Constructed to treat acid mine drainage. Designed to develop sizing criteria and to develop better sizing standards that incorporates contaminant concentrations as well as flow.

ADDITIVES

n.d.

COSTS

n.d.

TIME OF OPERATION / INFORMATION

Somerset built in 1984, first sampled in March, 1987; monthly sampling of water and flow rates was initiated in November, 1988.

Latrobe was constructed in June, 1987. Sample collection from the influent, effluent and between each cell began in July 1988. Flows measured at influent and effluent stations began in October 1988.

Friendship Hill was constructed in the summer of 1988. Data was collected in the summer of 1989. Water samples were collected from the common influent pool and from the effluent of each cell.

PERFORMANCE DATA

Performance was evaluated by calculating area-adjusted iron loadings and removals as Fe g/day/m^2 (gdm). Area adjusted iron removal = the difference between influent loading and effluent loading.

Somerset:

Date	Flow		Inflow Fe		
	L/min	pH	mg/L	g/day	GDM
03-20-87	19	3.2	193	5280	9.7
11-22-88	24	4.0	24	829	1.5
03-22-89	49	3.9	51	3599	6.6
04-11-89	29	3.7	122	5095	9.3
05-18-89	44	3.4	148	9377	17.2
06-16-89	25	3.8	337	12132	22.3
07-11-89	21	3.3	284	8588	15.7
08-16-89	10	3.3	310	4464	8.2
09-14-89	8	5.1	243	2799	5.1
10-24-89	11	5.1	169	2977	4.9
12-14-89	6	4.6	231	2096	3.9

Average influent pH = 4.0, range from 3.2 to 5.1

Iron removal was independent at loadings above 15 GDM (average removal = 10.6 GDM) and averaging 54% of loading at loadings less than 15 GDM.

Iron concentrations and loading varied considerably, and effluent iron concentrations varied considerably ranging from 3 mg/L to 159 mg/L.

Average area-adjusted iron removal was 6.3 GDM.

TECHNOLOGY Constructed Wetlands WTLD 16

PERFORMANCE DATA (cont'd)

Latrobe:

Date	Flow		Inflow Fe		
	L/min	pH	mg/L	g/day	GDM
10-14-88	38	3.1	215	11765	7.9
11-11-88	38	3.1	226	12366	8.3
11-28-88	56	3.0	194	15644	10.5
12-15-88	30	3.3	262	11318	7.6
01-10-89	53	3.0	210	16027	10.7
02-01-89	71	2.9	214	21879	14.6
02-23-89	273	3.1	104	31056	20.8
03-22-89	330	2.9	79	37541	25.1
04-19-89	173	3.0	85	21175	14.1
05-16-89	189	2.9	45	12247	8.2
05-23-89	114	3.0	49	8044	5.4
06-07-89	140	3.1	62	12499	8.4
After system modifications:					
08-89	37	4.5	119	9340	4.2
09-89	33	4.6	121	5750	3.8
10-89	34	4.6	136	6659	4.5

Average influent pH = 3.0 before modifications, after modifications, pH averaged 4.6.

Iron removal averaged 2.7 GDM at flows > 100 L/min, and 4.3 GDM at flows < 100 L/min. The overall average removal was 3.6 GDM.

Iron concentrations ranged from less than 50 mg/L during high flow periods to more than 200 mg/L during low flow periods. Iron removal ranged from 2.4 to 6.7 GDM.

Average area-adjusted iron removal rate was 3.0 GDM before modification. When post modification data are included, the overall removal is 3.6 GDM.

Friendship Hill:

Date	Flow		Inflow Fe		
	L/min	pH	mg/L	g/day	GDM
05-16-89	7.6	2.8	108	1182	10.8
05-23-89	7.6	2.6	119	1302	11.8
06-06-89	7.6	2.8	182	1992	18.1
06-21-89	7.6	2.7	85	930	8.2
07-25-89	7.6	2.8	82	897	8.2
08-08-89	3.8	2.8	139	758	6.9
08-23-89	3.8	2.6	144	785	7.1
09-06-89	3.8	2.6	195	1063	9.7
09-19-89	3.8	2.6	176	959	8.7
10-17-89	3.8	2.7	82	447	4.1
11-01-89	3.8	2.7	130	709	6.5
11-14-89	3.8	2.7	133	725	6.6
11-21-89	1.9	2.8	130	354	3.2
11-28-89	1.9	2.8	135	367	3.3
12-05-89	3.8	2.8	176	959	8.7
12-13-89	3.8	2.8	183	997	6.1
12-27-89	3.8	2.7	183	997	9.1
01-03-90	3.8	2.8	111	605	5.5

Average influent pH = 2.7

Overall iron removal averaged 3.3 GDM.

Iron concentrations of the influent ranged from 82 to 195 mg/L, area adjusted loadings into the cells varied from 3 to 19 GDM.

Average area adjusted removal rate for the period of May 16 to September 19 were 5.0 GDM for cell A2, 4.1 GDM for cell B2, and 4.3 GDM for cell C2. The overall average removal for this period was 4.5 GDM.

Over the entire observation period, the average removal of cell A2 was 3.5 GDM; cell B2, 2.8 GDM; and cell C2, 3.7 GDM.

OVERALL EFFECTIVENESS

Somerset: Strong relationship between area-adjusted iron loading and removal.

Latrobe: Highly variable effect on water chemistry from a concentration perspective. From an iron removal perspective, the performance was less variable. No relationship existed between area-adjusted iron loading and removal.

Friendship Hill: Cells displayed similar patterns of iron removal. Little relationship between iron loading and removal.

Complete iron removal for all sites did not occur, even at very low loadings.

TOXICITY n.d.

MAINTENANCE REQUIREMENTS n.d.

MONITORING REQUIREMENTS Flow, iron loading and concentration.

TREATMENT PRODUCT DISPOSAL n.d.

TECHNOLOGY	Constructed Wetlands WTLD 16
ENVIRONMENTAL CONDITIONS	Friendship Hill: With the onset of cooler temperatures, iron removal by all cells became more variable.
COMMENTS	Overall these sites were used to develop preliminary wetland sizing criteria based upon iron loadings. In situations where mine drainage has flow > 50 mg/L, loading-based criteria resulted in significantly larger wetlands than conventional flow-based criteria. Somerset: Influent iron loading, which was primarily a function of concentration, was more variable than iron removal. The relationship between the two appeared asymptotic. Area adjusted iron removal displayed less variability than the area-adjusted loading. The lowest iron removal rates were associated with very low influent loading, not with any apparent failure of the wetland during periods of high loading. The highest removal rates occurred when loading rates were also highest. Loading and removal rates were calculated for individual cells, resulting in higher estimates of area-adjusted loading and removal that were obtained for the whole wetland system. Latrobe: Periodic problems with leakage of water through a berm in the third cell prevented use of all the chemical and flow data collected from the final effluent station. Analytical efforts therefore focused on the first two cells. Loading calculations were based on influent flow rates. Extensive modifications were made to the system in the summer of 1989, most of the effects of the modification on influent water chemistry appeared to stabilize in August, 1989. Variation in Fe loading was primarily a consequence of flow variation. A significant relationship between loading and removal was not found. Loading and removal rates were calculated for individual cells, resulting in higher estimates of area-adjusted loading and removal that were obtained for the whole wetland system. Friendship Hill: The narrow range of loadings at this site prevented detailed analysis of loading removal relationships. Flow rates were changed three times during the study period and ranged from 1.9 to 7.6 L/min. On several days, effluent samples contained more iron than influent samples. During this period water levels were lowered by 7 - 10 cm, which may have influenced iron removal results.

TECHNOLOGY	Constructed Wetlands WTLD 17
PROJECT SCALE	Full scale experimental
REFERENCE	"The Use of an Artificial Wetland to Treat Acid Mine Drainage". Hendricks, A.C. 1991. Second International Conference on the Abatement of Acidic Drainage. pp. 594 - 558.
LOCATIONS	Southwestern Virginia.
DESCRIPTION	0.61 ha, 8 cells of approximately equal size, substrate composed of approximately 30.5 cm of limestone (5 to 7.5 cm in diameter) below 30.5 cm of bark mulch. Drains were placed in the limestone layer in order that water could be routed through it (water discharged on the surface went through the mulch layer, through the limestone layer, and discharged on the surface of the next cell). The cells were in series and separated by a 3 m berm of packed clay. *Typha latifolia* and *Scirpus robustus* were planted every 45 cm. At the peak of the 1990 growing season, the plant density was approximately 10 per m^2. The total water surface over the 8 cells was 0.405 ha. Flow ranged from 30 to 100 gpd.
DESIGN CRITERIA	Designed for the treatment of water from an abandoned pyrite mine. The inlet weir was designed to allow a maximum flow of 100 gdm. Flow ranged from 30 to 100 gdm.
ADDITIVES	n.d.
COSTS	n.d.
TIME OF OPERATION / INFORMATION	October 15, November 7, December 12, 1990?
PERFORMANCE DATA	<u>Influent</u> <u>Effluent</u> pH: 3.5 pH: 6.1 Total Fe: 326 mg/L Total Fe: 23 mg/L
OVERALL EFFECTIVENESS	The pH of the AMD was increased and iron removal was achieved as stated in the objective. Total iron levels decreased over 90% as the AMD travelled through the wetland.
TOXICITY	n.d.
MAINTENANCE REQUIREMENTS	n.d.
MONITORING REQUIREMENTS	Flow, dissolved oxygen, pH, redox potential, total iron.
TREATMENT PRODUCT DISPOSAL	n.d.
ENVIRONMENTAL CONDITIONS	n.d.
COMMENTS	When anaerobic water was released on the surface of the subsequent cell, iron hydroxide precipitated on the surface of that cell, thereby removing a portion of the iron. No iron hydroxide was observed on the stone and no red coatings were observed inside the drain pipes. Anaerobic and reducing conditions can be achieved in a wetland that will allow the limestone in the wetland to neutralize AMD and prevent the limestone from becoming coated with iron hydroxide. It is not clear how much of a role the plants and bacteria play in the reduction of iron levels and the enhancement of pH. Reasons for this are: all of the data was collected after the plants have become senescent; and the temperatures in the wetland were decreasing during the study period which would impair the activities of the sulfate reducing bacteria. It has also been shown that iron will stay in solution longer at colder temperatures, thereby reducing the efficiency of the wetland to remove iron.

TECHNOLOGY

Constructed Wetlands WTLD 18

PROJECT SCALE

Pilot scale to experimental full size

REFERENCE

"The Tracy Wetlands: A Case Study of Two Passive Mine Drainage Treatment Systems in Montana". Hiel, M.T. and F.J. Kerins. 1988. U.S. Bureau of Mines Information Circular 9183. pp. 352 - 358.

LOCATIONS

Great Falls, Montana.

DESCRIPTION

Two constructed wetland treatment systems: Large and Small Tracy Wetlands. Both utilized a peat substrate planted predominantly with *Typha latifolia* for metal removal, and limestone gravel and aeration structures for pH buffering. Large Tracy measured 30 m x 14 m, and had a surface area of approximately 418 m^2. Small Tracy wetland had two parallel impoundments each measuring 18 m x 3 m. The total surface area of Small was approximately 111 m^2.

DESIGN CRITERIA

The primary criteria used for the design of the wetlands included: sizing the wetland to allow for treatment of the quality of water expected during all seasons and following precipitation events; a surface area of 294 m^2/L/s was considered a minimum size. Minimizing water velocities and maximizing retention time; avoiding "short-circuiting" by the formation of flow channels; providing water depths varying from 5 cm to 46 cm. Providing optimum wetlands soil for emergent hydrophytes (such as *Typha*) composed of decomposed organic matter (peat) with some mineral soil content. The minimum depth of the peat was 0.3 m. Providing for: placing cattail sod mats over approximately 40% of the surface area of the wetlands; a crushed limestone-filled channel downstream of the wetland for moderation of pH; and, aeration structures along the limestone channel.

Large Tracy treated flows of 0.95 L/second, the Small Tracy treated flows of 0.15 L/second.

ADDITIVES

n.d.

COSTS

	Large ($)	Small ($)
Site Development/Inflow Control Facilities	5510	4125
Wetland Construction	5565	2335
Materials		
Water	780	375
Bentonite	975	750
Limestone Gravel	4260	1470
Peat	6760	3380
Vegetation Sod Mats	440	220
Wetland Flow Control/Aeration Structures	3430	2940
Total	27810	15595
Surface Area (m^2)	418	111
Cost/Area ($/$m^2$)	66.53	139.89

TIME OF OPERATION / INFORMATION

Both constructed during July and August, 1986.
Water quality data: July 10, 1986 - September 30, 1987.
Typha latifolia data: December 23, 1986 and September 21, 1987.

PERFORMANCE DATA

Large Tracy

Influent		Effluent	
Flow:	0.5 - 0.95 L/second		
pH:	2.7	pH:	2.58
Total Fe:	284 mg/L	Total Fe:	271 mg/L
Total Al:	178 mg/L	Total Al:	180 mg/L
Total Mn:	1.51 mg/L	Total Mn:	1.67 mg/L
Sulfate:	2618 mg/L	Sulfate:	2683 mg/L
Specific Cond.:	3349	Specific Cond.:	3340

Small Tracy

Influent		Effluent	
Flow:	0.38 - 0.5 L/second		
pH:	3.1	pH:	2.8
Total Fe:	148.5 mg/L	Total Fe:	94.1 mg/L
Total Al:	46.7 mg/L	Total Al:	45.7 mg/L
Total Mn:	1.2 mg/L	Total Mn:	1.3 mg/L
Sulfate:	1560 mg/L	Sulfate:	1551 mg/L
Specific Cond.:	2414	Specific Cond:	2559

Specific Conductivity measured in u mho/cm. All values are averages, total number of samples is 21 taken between July 10, 1986 and September 30, 1987.

OVERALL EFFECTIVENESS

Both wetlands were relatively ineffective in improving the water quality of the acid mine drainage. Low system retention times and minimal contact between the peat and acid mine drainage are primary reasons for the ineffectiveness of the systems.

TOXICITY

n.d.

TECHNOLOGY	Constructed Wetlands WTLD 18
MAINTENANCE REQUIREMENTS	n.d.
MONITORING REQUIREMENTS	Inflow, outflow water sampling, soil and vegetation sampling.
TREATMENT PRODUCT DISPOSAL	n.d.
ENVIRONMENTAL CONDITIONS	n.d.
COMMENTS	No plant stress has been observed and plant growth has been vigorous. Potential uptake of metals by *Typha* is not significant when compared with the actual loading of the system. The vegetation would provide only a minor contribution of the total metal retention of a functioning system. Because the removal of metals in both systems was unsuccessful, the limestone gravel in the channels downstream of each wetland became armoured in less than two weeks. Retention times were shorter than desired. Retention times were approximately seven hours for the Large and three hours for the Small. Low times were due primarily to the relative impermeableness of the peat once it is saturated. Two primary reasons for the ineffectiveness of the system was: both systems were undersized which resulted in very low retention times and limited the contact between the AMD and the peat moss which is the primary heavy metal removal medium. The systems designs did not force the AMD to flow through the peat at any location. The designs maximized the length of the flow paths through each system, but most of the flow was above the AMD-peat interface.

TECHNOLOGY	Constructed Wetlands WTLD 19
PROJECT SCALE	Full scale experimental
REFERENCE	"Metal Speciation and Retention Patterns in a High Metal Load Acid Mine Constructed Wetland of Southeastern Kentucky". Karathanasis, A.D. and Y.L. Thompson. 1990. Second International Conference on the Abatement of Acidic Drainage, Volume 2. pp. 485 - 498.
LOCATIONS	Jones Branch, McCreary County, KY
DESCRIPTION	11,000 square feet of ponded surface area for treatment. Wetland constructed by placing a layer of crushed limestone (KY #9's, 3/8 inch) on top of a graded, compacted floor treated with bentonitic clays to minimize seepage. Limestone layer is 9 inches thick below an 18 inch layer of spent mushroom compost in which cattails were planted. Watered initially with clean water to allow plants to recover from the stress of being transplanted. AMD was released into the wetland at a rate that would allow the plants to gradually become tolerant of the low pH water.
DESIGN CRITERIA	Ponds and cells sized according to expected high and low flows from an abandoned coal deep mine. Additional design considerations concluded site conditions, climate, hydrology, water chemistry, access and expected maintenance needs. Flow path length and residence time are critical. The project provides 480 square feet of surface area per gallon at 23 gallons per minute and 150 square feet of surface area per gallon at 75 gallons per minute of flow, the projected normal range in flow conditions based on observations.
ADDITIVES	n.d.
COSTS	n.d.
TIME OF OPERATION / INFORMATION	Constructed in the spring of 1989. Sampled twice (February and May, 1990), and monthly for the first six months of operation (June to December, 1989).

PERFORMANCE DATA

	Inlet			Outlet		
	June - Dec '89	Feb '90	May '90	June - Dec '89	Feb '90	May '90
pH	2.75	3.75	3.75	7.25	2.75	3.0
Al	0.475	0.10	0.35	0.05	0.05	0.40
Fe	15.0	9.0	17.5	2.0	2.0	14.5
Zn	0.007	0.0015	0.002	0.001	0.0	0.001
Mn	0.35	0.15	0.225	0.1	0.125	0.20
SO_4	42	4	13	10	4	12

Values in mM except for pH. All values read from graphs presented in paper.

OVERALL EFFECTIVENESS	After 8 months of operation and during periods of high flow rates (> 10 gallons/min) the efficiency of the wetlands was drastically reduced, apparently due to insufficient size and metal overloading. No major metal differences were observed during high flow rate periods between input-output concentrations, although input concentrations varied due to dilution effects. pH and SO_4^{2-} have been found to be the major components affecting metal speciation in AMD. The residual metal forms (sulphates, sulfides, carbonates, and oxyhydroxides) were dominant for every metal throughout the wetland except for two cells in which the exchangeable Mn and Zn species prevailed.
TOXICITY	n.d.
MAINTENANCE REQUIREMENTS	n.d.
MONITORING REQUIREMENTS	Substrate samples were collected from the upper 15 cm of selected cells.
TREATMENT PRODUCT DISPOSAL	n.d.
ENVIRONMENTAL CONDITIONS	n.d.
COMMENTS	Findings of the study suggest that metal distribution and speciation patterns in surface effluents and substrates of field-constructed wetlands and the mechanisms controlling metal retention can be sufficiently predicted by laboratory-simulated wetland models, but significant variations can result in wetland efficiency due to metal loading, flow rate variability and construction design.

TECHNOLOGY

Constructed Wetlands WTLD 20

PROJECT SCALE

Pilot scale

REFERENCE

"Wetland Sizing, Design, and Treatment Effectiveness for Coal Mine Drainage". Kepler, D.A. 1990. Proceedings of the 1990 Mining ad Reclamation Conference and Exhibition, Volume II, Charleston, West Virginia. pp. 403 - 408.

LOCATIONS

Site 1 wetland is located in Venango County, PA.; Site 2 wetland is located in Clarion County, PA.

DESCRIPTION

Site 1: The wetland is divided into two parallel series of four ponds each, with a common initial collection pond. The arrangement of the ponds was such that the order of planting was reversed in three opposite channels. This design allowed for comparisons to be made as to the effectiveness of both vegetative make-up and area/volume considerations in Fe removal.

Drainage is associated with the Lower Kittanning Coal Seam and has been flowing unabated from an abandoned drift mine opening since at least 1940.

Site 2: Characterized by an open water pond covering approximately 125 m^2 and a depth of 1.5 m, followed by a series of four *Typha* dominated cells, each measuring approximately 6 m by 30 m (180 m^2 each). The open water pond was designed to act as a sedimentation basin and as a retention pond for iron removal, because the relatively low flows and high pH values found lent well to this design.

Mine discharge to the wetlands is a reflection of the overburden associated with the Upper Clarion Coal Seam which was surface mined in the early 1970's.

The predominant planting (excluding the algal ponds) for the two sites was *Typha latifolia*. Both wetlands were excavated and sealed with on-site clay and incorporated 30 cm of composted manures and hay as substrate in the areas dominated by the *Typha* plantings. The *Typha* were planted as core samples on roughly 0.6 m centres, but spread to essentially 100% surface coverage by the end of the first complete growing season. Core samples refer to hand dug, intact rhizome/soil units.

DESIGN CRITERIA

Mean total flow into Site 1 during the sampling period was 106 L/min, with a range of 77 - 143 L/min. Mean total flow into Site 2 during the sampling period was approximately 12 L/min, with a range of 2 - 50 L/min.

ADDITIVES

n.d.

COSTS

n.d.

TIME OF OPERATION / INFORMATION

Site 1 was constructed in October, 1987. Sample period was November 1987 through March 1989; Site 2 was constructed in August, 1988. Sample period was August 1988 through September 1989.

PERFORMANCE DATA

Mean source quality of AMD at Sites 1 and 2:

	pH	Alkalinity CaCO$_3$	Acidity CaCO$_3$	Sulfate mg/L	Fe^{2+} mg/L	Fe (tot.) mg/L	Mn mg/L	Al mg/L
Site 1	5.5	50	250	2100	35	90	50	1
Site 2	5.7	28	145	970	135	180	20	<1

Total iron removal rates per pond in gdm for Site 1:

Configuration	Mean	Range
TTAA		
T1	25.3	8 - 53
T2	26.1	7 - 68
A1	11.1	5 - 28
AATT		
A1	16.5	1 - 35
A2	14.7	5 - 35
T1	13.3	4 - 28
T2	14.2	1 - 31

TTAA - *Typha* dominated ponds; T -*Typha* ponds
AATT - Algal dominated ponds; A - Algal ponds

Typical water quality at Site 2:

	pH	Alkalinity CaCO$_3$	Acidity CaCO$_3$	Sulfate mg/L	Fe^{2+} mg/L	Fe (tot.) mg/L	Mn mg/L	Al mg/L
Source	5.7	28	145	970	135	180	20	<1
Pond Outlet	3.0	0	125	825	32	34	19	<1
Cell 1 Outlet	6.7	58	0	650	12	15	18	<1
Cell 2 Outlet	6.9	145	0	580	6	8	18	<1

OVERALL EFFECTIVENESS

Site 1: The *Typha* dominated ponds at this site removed more iron from the flows on a per area basis than did their algal ponds counterpart, and appear to be more effective at removing iron at higher loads than at lower loading rates.

Site 2: Site 2 data further suggest that iron is more effectively precipitated in large versus small quantities.

TOXICITY

n.d.

TECHNOLOGY	Constructed Wetlands	WTLD 20
MAINTENANCE REQUIREMENTS	n.d.	
MONITORING REQUIREMENTS	Both wetlands have a minimum of 14 months of weekly water analyses collected from various points within the systems. Monitoring points were established at AMD source and final wetland discharge points and at the distinct inflow/outflow points of the individual sections of the entire wetlands systems. Typical analyses included pH, alkalinity, acidity, conductivity, total and ferrous iron, total and dissolved manganese, aluminum and sulfate.	
TREATMENT PRODUCT DISPOSAL	n.d.	
ENVIRONMENTAL CONDITIONS	Iron removal rates at Site 2 were most effective during May, June, and July (the three highest flow periods) and lowest during February, which was the lowest flow period. These months also represented the respective extremes in actual iron loadings. Seasonal findings are limited because both wetlands were relegated to one season's worth of data. The iron removal rate at Site 2 in December was slightly greater than that in August, but December's source loading was also greater than in August.	
COMMENTS	At both sites removal rates correlated positively with loadings at the source. Data are discussed mainly in terms of total mineral loadings, which allows comparisons of iron removal between the parallel channels, as opposed to concentrations. An attempt was made to provide equal flows to both series over the course of the study, but variations occurred. Flows were inadvertently not monitored at Site 2 during the initial months of collection, but were determined for the latter of the existing analyses. Data from May and June, 1988 were discussed separately because of anomalies in the data resulting from physical alterations to the wetland during this time period.	

TECHNOLOGY	Constructed Wetlands	WTLD 21
PROJECT SCALE	Full scale experimental	
REFERENCE	"Windsor Coal Company Wetland: An Overview". Kolbash, R.L. and T.L. Romanoske. 1989. Constructed Wetlands. pp. 788 - 792.	
LOCATIONS	Windsor Coal Company's Beech Bottom Mine, West Virginia Schoolhouse Hallow Refuse Area	
DESCRIPTION	Wetland was constructed on the Schoolhouse Hollow Refuse Area (30 acres) which received refuse from the early 1900s until 1980. Contaminated seepage is collected and channelled to a small pond at the refuse pile.	
DESIGN CRITERIA	0.5 cm limestone, sterile mushroom compost, and cattail (*Typha*) plants. The wetland was installed on the reclaimed disposal area 60 m below the hill crest on a 4:1 slope; a 36-mil Hypalon liner was chosen to protect against slope saturation. The wetland was excavated to 2:1 side slopes, with cut material used as fill for the down-slope side. Total excavation was 3.4 m by 26 m, to a total surface area of 117 m^2. A rock pipe drain was installed below the liner to carry water from a 4 L/min. seep at bottom grade of the excavation. A one-piece Hypalon liner was placed and keyed into the side berms of the wetland. Cattails were installed on 46 cm centres, and seep water was directed into the wetland. Even water distribution and flow was ensured by minor weir adjustments.	

TECHNOLOGY Constructed Wetlands WTLD 21

ADDITIVES

Collected seepage is treated with caustic soda, although maintaining a set pH limit has been difficult and fluctuating pH levels have caused occasional elevations of iron.

COSTS

Item	Cost ($)
Equipment Use	
40 hr backhoe at $60/hour	2400
40 hr dozer at $60/hour	2400 (incl. mobilization, operator and fuel)
Material	
Mushroom compost (30 yd at $21/yd)	630
Limestone substrate (17.8 tons at $7.25/ton)	129.05
Underdrain stone 12.23 tons at $15.30/ton	178.12
Type 316 S.S. weir	912.37
36-mil Hypalon liner	2896.20
Misc. pipe and fitting	680.86
Cattail plants	2275
Total	12510.60

TIME OF OPERATION / INFORMATION

Wetland construction began on August 10, 1987, and was completed on August 14, 1987. Data collection was from September, 1987 to May, 1988.

PERFORMANCE DATA

	Fe (mg/L)		Mn (mg/L)		pH	
	Inlet	Outlet	Inlet	Outlet	Inlet	Outlet
Maximum	88	51	3.7	3.5	4.0	7.25
Minimum	39	5	1.7	1.5	3.0	3.0

Approximate values read from graphs provided in paper.

OVERALL EFFECTIVENESS

Maintaining a set pH has been difficult and fluctuating pH levels have caused occasional elevations of iron concentration.

The wetland removed 50% of the incoming Fe. Within wetland water sampling indicated a gradual reduction of dissolved Fe without sharp transition zones. Manganese removal was not as efficient; inlet and outlet values were essentially unchanged. Seep manganese concentrations ranged from 1.5 to 3.8 mg/L, and inlet manganese increased since February 1988. Except for three incidents, the pH did not change from inlet to outlet., and remained between 3.0 and 4.0. The highest outlet pH from one of the three instances was 7.25.

TOXICITY

n.d.

MAINTENANCE REQUIREMENTS

n.d.

MONITORING REQUIREMENTS

Flow, pH, Mn and Fe concentrations.

TREATMENT PRODUCT DISPOSAL

n.d.

ENVIRONMENTAL CONDITIONS

n.d.

COMMENTS

The wetland minimizes a refuse pile seep as well as enhances the area wildlife and environmental quality. Pond location and size contribute to the difficulty in meeting iron limits on a continuous basis because the pond cannot be enlarged and requires frequent, expensive cleanout. The Hypalon liner seems to be enhancing algae growth. The effect of preventing interaction between wetland plants and the substrate is not known. Continued monitoring will measure sulfate, inlet flow, and will evaluate Hypalon liner long-term effects.

TECHNOLOGY

Constructed Wetlands WTLD 22

PROJECT SCALE

Full scale experimental

REFERENCE

"Ecological Benefits of Passive Wetland Treatment Systems Designed for Acid Mine Drainage: With Emphasis on Watershed Restoration". McCleary, E.C. and D.A. Kepler. 1994. International Land Reclamation and Mine Drainage Conference and the Third International Conference on the Abatement of Acidic Drainage. pp. 111 - 119.

LOCATIONS

R.E.M. Orcutt/Smaill site, Union County, Jefferson County, PA.

DESCRIPTION

The initial design consisted of an aerobic wetland that was sized to the available area of approximately 0.12 ha.

DESIGN CRITERIA

Constructed to deal primarily with the high iron loadings (> 20,000 g/d). Flow into the wetland was measured at 102.9 L/min.

ADDITIVES

n.d.

COSTS

Annual treatment costs for discharge is $125 per year (based on an alkalinity value of 125 mg/L). The installation costs for the anoxic limestone drain (ALD) and successive alkalinity producing systems (SAPS) were roughly $10,000 for materials and labour. To neutralize 149 mg/L of acidity (acidity at L5 subtracted from acidity of L1) chemically, roughly 33 kg of NaOH would be required per day. Sodium hydroxide currently costs $ 0.40/lb, on average, which translates to approximately $ 29/day and $ 10,600/year for chemical treatment; additional funds would be required for staff to operate the treatment equipment.

TIME OF OPERATION / INFORMATION

Constructed in 1987.

PERFORMANCE DATA

Monitoring Location	pH	Fe (mg/L)	Acid (mg/L)	Acidity Loading (g/d)	Fe Loading (g/d)
L1 (ALD)	6.18	155.0	213	31,561	22,967
L2	4.00	46.3	161	23,856	6,860
L3	3.75	29.8	152	22,523	4,416
L4	3.33	25.8	146	21,634	3,823
L5 (Effluent)	3.98	9.0	64	9,483	1,333

Flow measured at 102.9 L/min.
Source: U.S. Bureau of Mines laboratory data, January 18, 1993.

Data for 11 constructed wetlands is also presented in the paper and include the following:

Site	Year const.	Influent Effluent	Sample Size, n	pH	Fe (mg/L)	Mn (mg/L)	Acidity (mg/L)	Alk. (mg/L)	Flow (L/mi)
Somerset	1984	I	43	4.4	162	50	373	0	47
		E	40	5.5	18	33	69	n.d.	--
Donegal	1987	I	29	7.1	5	8	n.d.	202	501
		E	28	7.4	<1	2	n.d.	n.d.	--
Emlenton	1987	I	40	4.7	89	77	320	15	55
		E	40	3.2	15	73	271	0	--
Latrobe	1987	I	43	3.5	125	32	617	0	86
		E	43	3.7	56	29	343	n.d.	--
Piney	1987	I	39	5.8	1	15	n.d.	60	468
		E	39	5.8	1	15	n.d.	n.d.	--
FH	1988	I	73	2.6	123	9	929	0	15
		E	73	2.9	137	10	674	0	--
Blair	1989	I	12	6.2	52	30	n.d.	166	11
		E	8	7.0	<1	5	n.d.	n.d.	--
Cedar	1989	I	26	6.3	92	2	n.d.	336	156
		E	27	6.4	41	2	n.d.	n.d.	--
Keystone	1989	I	28	6.3	37	<1	n.d.	142	8606
		E	28	6.4	32	1	n.d.	n.d.	--
Shade	1989	I	20	6.0	<2	23	n.d.	31	10
		E	20	6.8	<1	10	n.d.	n.d.	--
Morrison	1990	I	34	6.3	150	42	n.d.	271	7
		E	24	6.6	<1	11	n.d.	n.d.	--

Source: Hedin et al, 1994, in press.
FH Friendship Hill National Historic Site

TECHNOLOGY Constructed Wetlands WTLD 22

PERFORMANCE DATA (CONT.)

Site Location[1]	Sample Location	Flow (L/min)	pH	Alk. (mg/L)	Acidity (mg/L)	Iron (mg/L)
Damariscotta:[2] Schnepp Road	Influent	54.5	3.2	0	300	45
	Effluent	--	7.0	56	0	2
USBM:[3] Howe Bridge[4]	Influent DRI	137.0	5.87	32.0	507	307.12
	Influent PO	84.5	6.10	40.0	344	227.61
	Effluent	--	6.49	75.0	<10	47.68
USBM:[3] Orcutt/Smaill[4]	Influent L1	135.6	5.84	107.0	275	199
	Influent R1	105.7	5.19	48.0	895	508
	Effluent	--	5.48	37.0	109	37.3

[1] All sites in Union Township, Jefferson, PA.

[2] Data from Damariscotta, 23 August 1993.

[3] U.S. Bureau of Mines laboratory data (4 May 1993, Orcutt/Smaill; 27 July 1993 Howe Bridge).

[4] Two source discharges are present at both Howe Bridge, DR1 and PO; and Rocutt/Smaill, L1 and R1. These discharges combine for one effluent discharge at both sites.

OVERALL EFFECTIVENESS

Removal rates for the R.E.M. Orcutt/Smaill site did not meet expectations, and in the spring of 1992 revisions were made to the system that incorporated an ALD and a SAPS. The passive system additions significantly improved water quality. Fe removal increased from 70 - 75% to 95%. Acidity values were also reduced from > 200 mg/L to 50 - 60 mg/L (associated mostly with Mn). Laboratory data was chosen as representative of a date on which passive treatment systems are considered to be at their lowest effective treatment levels.

TOXICITY

n.d.

MAINTENANCE REQUIREMENTS

n.d.

MONITORING REQUIREMENTS

pH, acid loading, Fe concentration and loading, flow.

TREATMENT PRODUCT DISPOSAL

n.d.

ENVIRONMENTAL CONDITIONS

n.d.

COMMENTS

Recent activities utilizing passive treatment systems as an approach to watershed restoration have met with great success.

<table>
<tr><td>TECHNOLOGY</td><td>Constructed Wetlands WTLD 23</td></tr>
<tr><td>PROJECT SCALE</td><td>Full scale experimental</td></tr>
<tr><td>REFERENCE</td><td>"Incorporation of Bacterial Sulfate Reduction into Constructed Wetlands for the Treatment of Acid and Metal Mine Drainage". McIntire, P.E. et al. 1990. National Symposium on Mining, University of Kentucky. pp. 207 - 213.</td></tr>
<tr><td>LOCATIONS</td><td>Friendship Hill National Historic Site, Fayette County, PA.</td></tr>
<tr><td>DESCRIPTION</td><td>Inflow to the wetland comes from a stream severely polluted with water draining from an abandoned drift mine 1 km upstream. The wetland is divided into three cells. The first cell is a holding pond that does not treat the AMD. The remaining two cells are each divided into three different treatment lanes separated by Fiberglas sheeting. In cell two the AMD flows across the surface of the organic substrate, but in cell three, subsurface infusion pipes can be used to carry AMD down to the bottom of the organic substrate layer. Individual ball valve controls are used to adjust AMD water flow into each lane. The six treatment lanes were each constructed with a 15 cm layer of gravel on the bottom, covered by a 46 cm layer of pent mushroom compost. The basal gravel layers in the cells consisted of crushed limestone and noncalcareous river gravel. Whole cattail plants were transplanted to all cells and the compost was flooded with 8 to 20 cm of AMD water.</td></tr>
<tr><td>DESIGN CRITERIA</td><td>Designed to use sulfate reduction in the treatment of AMD by inducing water flow through anaerobic organic substrate. Flow rate was 7.1 L/min.</td></tr>
<tr><td>ADDITIVES</td><td>n.d.</td></tr>
<tr><td>COSTS</td><td>n.d.</td></tr>
<tr><td>TIME OF OPERATION / INFORMATION</td><td>November, 1988 to August 1989.</td></tr>
<tr><td>PERFORMANCE DATA</td><td>Inflow:
pH of about 2.5.
Total Fe concentration of 50 to 250 ppm.
Sulfate concentration of 1000 to 25000 ppm.

Outflow (When AMD forced through substrate)
pH increased as high as 6.5
Alkalinity increased up to 500 ppm.
Total Fe concentration decreased from 200 to 20 ppm.

Bacterial sulfate reduction rates in the organic substrate ranged from 2 to 600 nmol/cm^3/day.</td></tr>
<tr><td>OVERALL EFFECTIVENESS</td><td>Bacterial sulfate reduction rates in the wetland were comparable to those found in coastal marine systems, and high enough to significantly affect the water quality of AMD. Iron concentrations decreased, pH increased, and alkalinity increased when the mine drainage was first forced through the anaerobic zone. This was due to both the inherent chemical characteristics of the organic substrate and the bacterial activity. No improvement in water quality was observed after the AMD had been forced thorough the substrate for several weeks; this was apparently due to the exhaustion of the alkalinity present in the substrate by high AMD flow rates.

When subsurface infusion pipes were turned on and the AMD forced into the anaerobic substrate, an immediate increase in pH and alkalinity and a net removal of total iron were observed. Little improvement in water pH or alkalinity was observed when the AMD flowed predominantly across the surface of the organic substrate. When subsurface infusion pipes were turned off, the observed improvements in water quality disappeared.

Measurements of pH in the porewater of the organic substrate showed that the pH generally remained between 6.0 and 7.0, even though the surface water usually had a pH of 2.5 to 3.5. This high porewater pH most likely reflects the influence of alkaline substrates present in the compost and those produced by the natural bacterial population. When the Plexiglas dam was used, the increased flow of AMD through the relatively small volume of compost located on either side of the dam exhausted the neutralization of the compost.

The redox potentials measured below 15 cm deep in the organic substrate decreased steadily, indicating gradual exhaustion of reduced species.</td></tr>
<tr><td>TOXICITY</td><td>Laboratory experiments were done to evaluate potential metal toxicity to sulfate-reducing bacteria. The experiments demonstrated that sulfate-reducing bacteria were active in solutions containing high concentrations of metals.

<table>
<tr><th></th><th colspan="3">Metal Concentration (ppm)</th><th></th></tr>
<tr><th>Metal</th><th>Toxicity</th><th>After Inoculation[1]</th><th>After Growth[2]</th><th>Percent Removal</th></tr>
<tr><td>Cd</td><td>104</td><td>35</td><td>0.1</td><td>>99</td></tr>
<tr><td>Cu</td><td>> 14</td><td>14</td><td>< 0.1</td><td>>99</td></tr>
<tr><td>Mn</td><td>>223</td><td>223</td><td>60</td><td>73</td></tr>
<tr><td>Ni</td><td>>211</td><td>211</td><td>0.4</td><td>>99</td></tr>
<tr><td>Pb</td><td>> 5</td><td>5</td><td>0.5</td><td>89</td></tr>
<tr><td>Zn</td><td>310</td><td>194</td><td>2</td><td>99</td></tr>
</table>

[1] Metal concentration immediately after inoculation.
[2] Metal concentration at the end of experiment.</td></tr>
<tr><td>MAINTENANCE REQUIREMENTS</td><td>Clogging problems had to be addressed.</td></tr>
</table>

TECHNOLOGY	Constructed Wetlands	WTLD 23
MONITORING REQUIREMENTS	Surface water samples collected from nine locations biweekly for eight months, pH measures, sediment samples collected and interstitial porewater collected.	
TREATMENT PRODUCT DISPOSAL	n.d.	
ENVIRONMENTAL CONDITIONS	Sulfate reduction rates remained consistently low through the winter but increased as the summer progressed, consistent with the predicted overall increase in biological activity with seasonally high temperatures.	
COMMENTS	The subsurface infusion pipes worked well for two weeks each of the first two times they were used; after that time, they failed when organic debris and iron oxyhydroxide floc clogged the pipes. The continuous supply of a limited number of simple organic compounds is necessary to ensure ongoing bacterial sulfate reduction activity. It is probable the much of the initially observed improvement in water quality was due to the chemical characteristics of the spent mushroom compost. Flow patterns are likely to occur that use up the neutralization potential of the compost along major flow paths and result in less effective neutralization. Bacterial sulfate reduction can be expected to contribute neutralization potential to the compost with time. The data implies that there is no significant neutralization advantage in using limestone gravel in the construction of wetlands employing anaerobic systems.	

TECHNOLOGY	Constructed Wetlands WTLD 24
PROJECT SCALE	Full Scale Experimental
REFERENCE	"The use of Bacterial Sulfate Reduction in the Treatment of Drainage from Coal Mines". McIntire, P.E. and H.M. Edenborn. 1990. Proceedings of the 1990 Mining and Reclamation Conference and Exhibitions, Charleston WV, Volume II. pp. 409 - 415.
LOCATIONS	Friendship Hill National Historic Site, Fayette County, PA.
DESCRIPTION	Inflow to the wetland comes from a stream severely polluted with water draining from an abandoned drift mine 1 km upstream. Divided into three cells. The first cell is a holding pond that does not treat the AMD. The remaining two cells are each divided into three different treatment lanes separated by Fiberglas sheeting. In cell two, the AMD flows across the surface of the organic substrate, but in cell three, subsurface infusion pipes can be used to carry AMD down to the bottom of the organic substrate layer. Individual ball valve controls are used to adjust AMD water flow into each lane. The six treatment lanes were each constructed with a 15 cm layer of gravel on the bottom, covered by a 46 cm layer of spent mushroom compost. The basal gravel layers in the cells consisted of crushed limestone and noncalcareous river gravel. Whole cattail plants were transplanted to all cells and the compost was flooded with 8 to 20 cm of AMD water.
DESIGN CRITERIA	Designed to maximize contact between mine drainage and the anaerobic zone of the organic substrate where sulfate reduction takes place. Approximately 3.3 x 10^5 L of AMD of 1000 mg/L acidity passed through the compost during 31 days of operation. Flow rate was 7.5 L/min.
ADDITIVES	n.d.
COSTS	n.d.
TIME OF OPERATION / INFORMATION	Built in the summer of 1988. Data collected from November 1988 to August 1989.
PERFORMANCE DATA	Inflow water has a pH of about 2.5 Total Fe concentration of 50 to 250 ppm Sulfate concentration of 1000 to 2500 ppm When mine drainage was forced through the anaerobic zone: Fe concentrations decreased from 237 to 27 ppm pH increased from 2.9 to 6.5 Alkalinity increased from 0 to 1077 ppm Bacterial sulfate reduction rates in the organic substrate ranged from 2 to 600 nmol/cm^3/day. Measurements of pH in the porewater of the organic substrate showed that the pH generally remained between 6.0 and 7.0, even though the surface water usually had a pH of 2.5 to 3.5. Redox potentials measured below 15 cm deep in the organic substrate decreased steadily, which is an indication of gradual exhaustion of oxidized chemical species and the accumulation of reduced species.
OVERALL EFFECTIVENESS	When AMD was forced into the anaerobic substrate, an immediate increase in pH and alkalinity and a net removal of total iron were observed. Little improvement in water pH or alkalinity was observed when the AMD flowed predominantly across the surface of the organic substrate, and the observed improvements in water quality were not present. The concentration of total reduced sulphur compounds also increased in the wetland substrate with time, confirming that by-products of bacterial sulfate reduction do accumulated in the wetland substrate and can probably be used to affect the quality of AMD under properly engineered conditions.
TOXICITY	n.d.
MAINTENANCE REQUIREMENTS	Gravel became armoured with iron oxyhydroxides and other deposits. An alternative method of introducing AMD to the anaerobic regions without the use of pipes was devised which involved the installation of a sheet of Plexiglas in a test lane of the wetland, placed so that water in the lane was dammed behind the sheet and was forced to flow downward beneath the dam, somewhat analogous to hay bale dams.
MONITORING REQUIREMENTS	Surface water samples were collected from nine locations in the wetland biweekly for eight months. pH was measured in the field. Alkalinity, Acidity and total iron concentration, sulfate concentration were monitored. Organic substrate cores were taken, and interstitial pore water samples were collected.
TREATMENT PRODUCT DISPOSAL	n.d.
ENVIRONMENTAL CONDITIONS	Sulfate reduction rates remained consistently low throughout the winter but increased as the summer progressed, consistent with the predicted overall increase in biological activity with seasonally higher temperatures.

TECHNOLOGY	Constructed Wetlands	WTLD 24
COMMENTS	Sulfate reduction rates are high enough to significantly affect the water quality of acid mine drainage. The sulfate reducing bacteria effectively precipitate many heavy metals as insoluble sulfides and may also be useful in treatment processes designed to improve the water quality of metal mine drainage. Increasing contact between AMD and the anaerobic zone of the organic substrate improved the quality of the water. Soon after the subsurface infusion pipes were turned on they became clogged with iron oxyhydroxide floc and organic debris. Sulfate reduction occurs at relatively high rates, but these rates are highly variable in the heterogeneous environment of the wetland organic substrate, and they vary seasonally. Data suggests that there is no significant acid neutralization advantage in using limestone gravel in the construction of wetlands employing anaerobic wetlands. After the Plexiglas dam was installed, initial water quality improvements were observed; however, after a few weeks of operation, the water quality was no longer affected presumably due to the exhaustion of alkalinity in the substrate and the disruption of the anaerobic microbial community. High porewater pH likely reflects the influence of alkaline substances present in the compost as well as those produced by the natural bacteria population. When the dam was used, the increased flow of AMD exhausted the neutralization potential of the compost. Porewater pH values deep in the compost above the dam after its operation were all low (3.2 to 4.5). The percentage of AMD that is neutralized can increase significantly by enhancing the sulfate reduction rate or by lowering the flow rate of AMD through the system.	

TECHNOLOGY	Constructed Wetlands WTLD 25
PROJECT SCALE	Laboratory scale
REFERENCE	"Adding a Carbon Supplement to Simulated Treatment Wetlands Improves Mine Water Quality". Stark, L.R., et al. 1990. Second International Conference on the Abatement of Acidic Drainage, Volume 2. pp. 465 - 483.
LOCATIONS	Pennsylvania State University
DESCRIPTION	Non-circulating continuous flow system in which a concentrated stock solution of mine water is prepared and stored in a 1600 L fibreglass reservoir in a greenhouse. The water is pumped to a series of chambers where the solution is diluted, mixed, and adjusted to the experimental pH, metal concentration, and flow rate, and distributed to individual simulated wetlands lanes. The stock solution was prepared from ferrous sulfate, sulphuric acid, and tap water. Each simulated wetland consisted of a wooden rectangular structure coated with fibreglass resin and lined with plastic. The dimensions of each land (160 cm in length x 25 cm in width x 15 cm in depth) were suitable to allow the development of a habitat with an oxidizing and reducing zone simulating nature. The lanes were level and filled with 6.6 kg (dry weight) treated spent mushroom compost.
DESIGN CRITERIA	Inlet water quality was identical for each simulated wetlands. Two cells received liquid whey supplements, three cells (control, and two replicates) consisted of simulated wetlands not receiving whey supplements. None of the cells contained plants. A moderate flow rate was used in order to preserve the substrate reduction-oxidation profile found in wetland environments. ph and flow rate were maintained at approximately 4.0 and 30 mL/min, respectively. The wetlands received water containing 50 mg/L Fe for 10 weeks (Phase I), followed by water containing 150 mg/L for 9 weeks (Phase II). These treatments equate to a loading of 5.4 g Fe/m^2/day and 16.2 g Fe/m^2/day, respectively.
ADDITIVES	600 mL of whey was injected into the substrate adjacent to the inlet of two cells on a weekly basis.
COSTS	n.d.
TIME OF OPERATION / INFORMATION	Wetlands were assembled in early 1990.
PERFORMANCE DATA	Inlet pH was fairly stable near 4.0, outlet pH for both treatments exceeded 6.0 for the duration of the experiment. In Phase I outlet iron concentrations declined to near zero.

Phase I (Low)			Phase II (High)	
pH	Max.	Min.	Max.	Min.
Inlet	5.5	3.9	4.5	3.2
non-whey	7.9	6.9	7.2	6.1
whey	8.1	6.5	6.9	6.1
Fe conc. (mg/L)				
Inlet	60	22	180	120
non-whey	21	3	77	4
whey	18	1	42	2
Sulfate conc. (mg/L)				
Inlet	250	225	550	450
non-whey	275	240	500	380
whey	250	190	350	75

Data read from graphs presented in the paper.

OVERALL EFFECTIVENESS	The weekly injection of sweet cheese whey into the substrate near the lane inlets retained more iron and has significantly lower outlet sulfate levels than those lanes not receiving the supplement. Iron and sulfate removal continued to improve throughout the course of the experiment in the whey treated lanes; no such trends were seen in the untreated lanes. No significant differences between the supplemented and unsupplemented wetlands were found for the parameters outlet pH, H$^+$, redox potential, Ca^{+2}, alkalinity, and acidity. Dissolved organic carbon and dissolved oxygen also did not differ significantly.
TOXICITY	n.d.
MAINTENANCE REQUIREMENTS	n.d.
MONITORING REQUIREMENTS	Inlet and outlet concentrations of pH, Fe, Ca, and alkalinity.
TREATMENT PRODUCT DISPOSAL	n.d.
ENVIRONMENTAL CONDITIONS	n.d.

TECHNOLOGY	Constructed Wetlands WTLD 25
COMMENTS	It is likely that the whey supplement, rich in readily metabolized carbon, vitamins, and minerals, stimulated sulfate reduction beyond sulfate reduction levels present in unsupplemented lanes. The process of sulfate reduction is likely to be critical to the long term effectiveness of constructed wetlands. In wetlands receiving mine waters, carbon is likely to be limiting to sulfate reduction. At high sulfate concentrations (-> 450 mg/L) some other nutrients must be limiting bacterial metabolism. Supplementation with a carbon and other nutrient source should improve water treatment. The high carbon to nitrogen ratio (approximately 18:1) indicates that carbon is not in short supply, readily available carbon to bacterial metabolism may be limiting. Whey stimulates the production of laccase, which in turn hastens the solubilization of the cellulose fraction of the straw, thus releasing readily available carbon.

TECHNOLOGY Constructed Wetlands WTLD 26

PROJECT SCALE Full scale experimental

REFERENCE "Iron Loading, Efficiency and Sizing in a Constructed Wetland Receiving Mine Drainage". Stark, L.R., et al. 1990. Proceedings of the 1990 Mining and Reclamation Conference and Exhibition, Volume II, Charleston, West Virginia. pp. 393 - 401.

LOCATIONS Near Coshocoton, Ohio.

DESCRIPTION The Simco wetland has been in operation and continuously monitored since 1985. Wetland consists of three wetland cells in sequence separated by small ponds. The substrate consisted of a layer of crushed limestone (15 cm) covered with a layer of spent mushroom compost (45 cm). The total area of the system was 3,196 m^3 (including small ponds between cells and connecting ditches). Bed slope was 1%. *Typha latifolia* rhizomes were transplanted from a donor site to an initial density of 3 to $4/m^2$. In 1989, a fourth wetland cell was added, bringing the system area to 4,138 m^2.

DESIGN CRITERIA Wetland installed primarily to treat a seep that contained high levels of total irons and was initially planted with *Typha latifolia*. Flow rate into the wetland ranged from 114 to 1,116 L/min, and closely mirrored iron loading.

ADDITIVES n.d.

COSTS n.d.

TIME OF OPERATION / INFORMATION Winter 1985 to Fall 1989.

PERFORMANCE DATA Mean (±one standard deviation) water quality at Simco #4 inlet and outlet. Data collection period was from November 1985 to October 3, 1989.

	pH (mg/L)	Total Fe (mg/L)	Total Mn (mg/L)	Acidity (mg/L)	Alkalinity (mg/L)	Sulfate (mg/L)	TSS (mg/L)
Inlet	6.5±0.2	111±26	2.0±0.6	123±82	87±33	1146±225	40±24
Outlet	6.4±0.3	42±23	2.1.6	32±25	26±18	1067±230	50±27
Efficiency (%)	-2	-62	+5	-74	-70	-7	+25
# samples	(74)	(86)	(86)	(86)	(86)	(70)	(73)

Outflow (L/min) 328±186, N=105
Temp. (°C) 12.8±1.6, N=1.6

When outlet iron concentrations were <25 mg/L, the iron loading average 47 kg Fe/day. When outlet concentrations exceeded 25 mg/L, loadings averaged about 57 kg Fe/day. When Fe loadings were <47 kg Fe/day, the mean outlet iron concentration was 25.1 ± 11.0 mg/L (mean ± S.D.)

OVERALL EFFECTIVENESS Efficiency in removing total iron averaged 62% from wetland installation to October 3, 1989 (about four years). The range is from a low of 11% in December 1985, shortly after construction, to a high of 94% from July to September 1988. Generally, the system has improved in efficiency with age. As loading to the wetland system increased, the efficiency decreased with respect to total iron.

TOXICITY n.d.

MAINTENANCE REQUIREMENTS n.d.

MONITORING REQUIREMENTS Water samples and flow rate samples were taken every second or third week.

TREATMENT PRODUCT DISPOSAL n.d.

ENVIRONMENTAL CONDITIONS Seasonal efficiencies (%: Inlet Fe - Outlet Fe/Inlet Fe) for total iron:

	1985	1986	1987	1988	1989
Spring (March - May)	--	37	55	63	61
Summer (June - Aug.)	--	56	62	91	73
Fall (Sept. - Nov.)	--	53	73	82	64
Winter (Dec. - Feb.)	28	51	72	62	--

Higher efficiencies occurred in the summer and fall, lower efficiencies usually occurred during the winter and spring. Winter efficiency improved with each successive winter except during the winter of 1988.

Highest loadings occurred during the highest flows (winter and spring), the lowest loadings occurred during the lowest flows (summer and fall). The depressed efficiencies of winter and spring may simply be a result of higher loadings, as opposed to temperature-dependent biological reactions.

TECHNOLOGY	Constructed Wetlands	WTLD 26
COMMENTS	The calculated loading range is an estimate because water samples and flow rates were sampled every second or third week. Highest loads to the system occurred in the winter and spring, relatively low loadings occurred during the summer and fall. Iron loadings closely correspond to flow rates. The recommended iron loading should be under 47 kg Fe/d. In order to ensure an effluent # 25 mg/L, preferred loadings would be in < 28 kg Fe/d. The efficiency of the wetland is directly related to Fe loading; 47% of the observed variation in efficiency can be explained by the iron loading rate.	

TECHNOLOGY

Constructed Wetlands WTLD 27

PROJECT SCALE

Full scale experimental

REFERENCE

"Iron Retention and Vegetative Cover at the Simco Constructed Wetland: An Appraisal Through Year Eight of Operation". Stark, L.R., et al. 1994. International land Reclamation and Mine Drainage Conference and the Third International Conference on the Abatement of Acidic Drainage. pp. 89 - 98.

LOCATIONS

Simco wetland, near Coshocton, OH.

DESCRIPTION

The Simco wetland has been in operation and continuously monitored since 1985. Wetland consists of three wetland cells in sequence separated by small ponds. The substrate consisted of a layer of crushed limestone (15 cm) covered with a layer of spent mushroom compost (45 cm). The total area of the system was 3,196 m^3 (including small ponds between cells and connecting ditches). Bed slope was 1%. *Typha latifolia* rhizomes were transplanted from a donor site to an initial density of 3 to 4/m^2. In 1989, a fourth wetland cell was added, bringing the surface area to 4,138 m^2.

DESIGN CRITERIA

Wetland installed primarily to treat a seep that contained high levels of total irons and was initially planted with *Typha latifolia*. Flow rate into the wetland was 451.29 ±272.7 L/min.

ADDITIVES

n.d.

COSTS

n.d.

TIME OF OPERATION / INFORMATION

1985 to 1994 and ongoing.

PERFORMANCE DATA

Mean (± 1 standard deviation) values for inlet and outlet water parameters at the Simco wetland, 1985 - 1993.

Parameter	Inlet	Outlet	N
pH	6.56±0.16	6.58±0.75	190
Total Fe (mg/L)	89.37±0.16	22.63±20.09	190
Total Mn (mg/L)	1.72±25.56	1.96±0.65	191
Acidity (mg/L)	133.80±0.39	28.04±22.94	191
Alkalinity (mg/L)	87.64±26.82	44.02±26.93	191
Conductivity (µomhos/cm)	1,790.10±360.71	1,701.97±336.17	189
TSS (mg/L)	37.60±16.87	29.71±22.05	190
Sulfate (mg/L)	984.07±233.79	912.63±219.27	174
Temperature (°C)	12.85±1.26	12.00±7.38	128
Flow rate[1] (L/min)	451.29±272.70	427.50±270.90	95

[1] Paired data from 1987 - 1993; previous to this, only outflow data is available.

Flow (L/min)	< 100 to < 1300
Area adjusted retention (g/m^2/d)	> 1 to < 27.5

OVERALL EFFECTIVENESS

Inlet and outlet water chemistry: Until early 1989, total Mn entering the wetland exceeded Federal standards of 2.0 mg/L. Most other mine water parameters decreased in concentration as a result of passage through the wetland.

Inlet and outlet Fe concentration: Inlet Fe has declined over the last 10 years from 200 mg/L to 75 mg/L, however, chemical treatment of NaOH was required periodically.

Total Fe treatment efficiency: Fe retained by the wetland increased over time. Efficiency has been higher during the winter warmer months and lower during cooler months. The treatment efficiency improved each succeeding winter with one exception.

Area adjusted Fe retention: As flow rate increased, retention per unit area also increased.

The general trend over time is towards an increase in treatment efficiency.

TOXICITY

n.d.

MAINTENANCE REQUIREMENTS

n.d.

MONITORING REQUIREMENTS

Water was sampled every 2 to 3 weeks, water samples were taken from the system inlet and outlet, as well as from individual wetland cell outlets and the site permit outlet. pH, flow rates and temperature were taken in the field. Water samples were analyzed for total and dissolved Fe, total Mn, acidity, alkalinity, conductivity, sulphates, and total suspended solids.

TREATMENT PRODUCT DISPOSAL

n.d.

ENVIRONMENTAL CONDITIONS

n.d.

TECHNOLOGY	Constructed Wetlands	WTLD 27
COMMENTS	Wetland treatment is likely dependent on critical design parameters namely size of the wetland as compared to flow rate and influent metal concentrations, the net acidity of the influent water, and the metal load to the wetland system (Hedin and Narin, 1992). Treatment efficiency for iron has improved over time to its current level above 90%. The addition of a fourth cell in 1989 is probably at least partly responsible for the increase in treatment efficiency. After 8 years of operation, the wetland has shown no signs of declining Fe retention or pH mitigation. This may be due to a variety of factors such as the presence of moderate mine water, wetland design and modifications, the role of cattails, and vegetation patterns.	

TECHNOLOGY

Constructed Wetlands WTLD 28

PROJECT SCALE

Full scale experimental

REFERENCE

"The Simco #4 Wetland: Biological Patterns and Performance of a Wetland Receiving Mine Drainage". Stark, L.R., et al. 1992. Achieving Land Use Potential Through Reclamation, Duluth Minnesota. pp. 550 - 562.

LOCATIONS

Peabody Coal-American Electric Power Simco #4 deep coal mine site near Coshocton, OH.

DESCRIPTION

Three wetland cells covering an area of approximately 3,000 m^2 separated by mixing pools were constructed in a strip pit west of the backfill area. The area was graded and a 6 inch layer of crushed limestone was applied and covered with an organic-rich, deep-rooting medium (18 inches) in which lime was incorporated. *Typha* rhizomes were planted with a density of 3 -4/m^2. Diversion ditches and hay bale dikes were installed in the fall of 1987. Prominent plants besides cattail are *Leersia oryoides* (cutgrass).

DESIGN CRITERIA

System was constructed in cells to provide proper gradients to restrict flow velocities and to promote uniform water dispersal through the wetland. Segmenting the wetland provided a gradual introduction of the discharge water into the wetland, allowing the vegetation to establish prior to the introduction of the entire discharge.

Flow rate measured at the outflow of the wetland ranged from 100 - 160 gpm from September 1986 to August 1987, but declined to 50 - 65 gpm in September 1987.

ADDITIVES

Periodically, the wetland has been limed and/or fertilized with phosphate (di-ammonium phosphate, 0-45-0) in an attempt to increase iron removal and promote the growth of algae.

Date	Lime (lbs)	Fertilizer (lbs)
Dec 1985	--	200
Feb 1986	--	200
Mar 1986	--	200
Jul 1986	8,000	--
Nov 1986	950	200
Dec 1986	1,000	200
Jan 1987	--	150
Apr 1987	--	150
Jul 1987	--	150
Sep 1987	--	150

COSTS

n.d.

TIME OF OPERATION / INFORMATION

1985 - 1987: Construction started on October 14, 1985 and completed on November 20, 1985.

PERFORMANCE DATA

Average pH values by season

	Location				
	Seep	Cell 1	Cell 2	Cell 3	Ditch End
Pre-wetland (83-85)	6.16	--	--	--	5.88
Winter (85-86)	6.54	6.07	5.92	6.03	6.07
Spring (86)	6.40	6.52	6.46	6.40	6.50
Summer (86)	6.40	6.40	6.22	6.22	6.24
Fall (86)	6.46	6.16	6.16	6.47	6.16
Winter (86-87)	6.34	6.48	6.48	6.36	6.50
Spring (87)	6.25	6.45	6.40	6.27	6.43
Summer (87)	6.36	6.33	6.31	6.19	5.99
Fall (87)[*]	6.16	6.50	6.43	6.26	6.18

[*] After site modifications of August 31, 1987.

Total iron (mg/L) by season

	Location				
	Seep	Cell 1	Cell 2	Cell 3	Ditch End
Pre-wetland (83-85)	180	--	--	--	122
Winter (85-86)	116	99	93	84	75
Spring (86)	106	93	81	65	55
Summer (86)	127	98	77	56	32
Fall (86)	144	123	105	74	58
Winter (86-87)	113	100	86	60	52
Spring (87)	100	78	63	42	38
Summer (87)	87	68	59	34	26
Fall (87)[*]	100	62	59	26	19

[*] After site modifications of August 31, 1987.

PERFORMANCE DATA (CONT.)

Average percent reductions in total iron concentration by season, cumulative (cum.) for the wetland proper (wtld), and end ditch; N=4-7 for each season.

	Location Cell 1	Cell 2	Cell 3	Ditch End	Cum. Wtlnd	Cum. end ditch
Winter (85-86)	14	6	12	10	28	35
Spring (86)	12	12	18	16	37	47
Summer (86)	22	20	27	42	56	75
Fall (86)	17	14	29	22	53	59
Winter (86-87)	17	15	30	8	51	53
Spring (87)	21	17	33	13	55	62
Summer (87)	22	17	42	22	62	70
Fall (87)*	38	41	56	26	73	81

* After site modifications of August 31, 1987.

Average total acidity (mg/L) by season

	Location Seep	Cell 1	Cell 2	Cell 3	Ditch End	% Red.
Pre-wetland (83-85)	207	--	--	--	187	10
Winter (85-86)	95	82	76	70	65	32
Spring (86)	52	38	36	26	22	58
Summer (86)	92	57	32	15	1	99
Fall (86)	88	64	59	41	20	77
Winter (86-87)	54	38	27	20	20	63
Spring (87)	57	25	22	14	15	74
Summer (87)	76	24	15	3	6	87
Fall (87)*	61	3	17	-4	-3	100

* After site modifications of August 31, 1987.

Winter = Dec - Feb, Spring = Mar - May, Summer = Jun - Aug, Fall = Sept. - Nov.
N=5 for pre-wetland, N=4-7 for post-wetland.

OVERALL EFFECTIVENESS

Despite the liming, pH was unaffected, and iron removal efficiency did not change significantly. However, the iron removal efficiency of the ditch below the wetland was distinctly elevated in the summer of 1986, coinciding with the largest application of lime.

The seep pH of 6.0 is unaltered by the wetland. Cannot discriminate between the pH effect on iron solubility and the pH effect on biological surfaces and cation exchange. Iron removal two years after wetland construction is up 80 - 90% over pre-wetland iron removal.

Biological activity in the form of microbial, algal, and cattail-cutgrass populations is much greater following wetland construction and vegetation cover increased considerably from the first to the second summer. Any effect of substrate apart from biological activity cannot account for all of the iron removal. Cattail roots and rhizomes are directly absorbing or precipitating some of the iron. Wetland Cell 3 was the most effective portion of the constructed wetland in improving the water quality.

Modifications done in the fall of 1987 resulted in an immediate increase in iron and manganese removal, pH and alkalinity decreases.

TOXICITY

n.d.

MAINTENANCE REQUIREMENTS

n.d.

MONITORING REQUIREMENTS

Water quality, vegetation, and aerobic microbial populations have been monitored since June, 1986. Sediments sampled at 20 locations along the length of the wetland, and analyzed for microbial activity. Microbial counts included fungi, actinomycetes, and bacteria.

TREATMENT PRODUCT DISPOSAL

n.d.

ENVIRONMENTAL CONDITIONS

Only a slight reduction in iron removal efficiency between summer and winter (56% to 51%) was observed. Winter efficiency was 28% (overall efficiency) approximating pre-wetland efficiency. During the second winter (1986-87) efficiency increased to 51%. This is probably attributable to biological activity.

COMMENTS

Fe reduction corresponds to greater wetland area, higher cattail density and increased plant coverage. Cattail and cutgrass coverage increased from 1986 to 1987. Roots of cattail accumulated up to 5 times the iron content of roots from a control site.

Cell 3 contained the greatest amount of biological mass, and the largest cell. Microbial populations were no greater in this cell than in the remaining two. Cell 3 received water with the lowest level of iron, therefore its higher efficiency may reflect the water pretreatment effect of previous cells or that iron is more easily removed by a wetland at lower incoming levels.

The cattail rhizosphere may be a conducive environment for aerobic microbes such as species of *Thiobacillus* that oxidize iron. It is likely that hay, much like sawdust, encourages anaerobic sulphate-reducing bacteria that precipitate FeS - FeS_2 (Tuttle et al. 1969). The role of algae is not yet understood. Anaerobic bacteria has not been counted in the wetland, but the presence of sulfate reduction by bacteria were obvious. The oxidation of Fe was evident in the wetland as indicated by the deposition of $Fe(OH)_3$ and the formation of yellow-boy. Iron oxidizing bacterial of the genus *Thiobacillus* was found throughout the wetland.

<table>
<tr><td>TECHNOLOGY</td><td>Constructed Wetlands WTLD 29</td></tr>
<tr><td>PROJECT SCALE</td><td>Full scale experimental</td></tr>
<tr><td>REFERENCE</td><td>"Iron and Manganese Removal in a <i>Typha</i> - Dominated Wetland During Ten Months Following its Construction". Stillings, L.L., et al. 1988. Mine Drainage and Surface Mine Reclamation Conference. pp. 317 - 325.</td></tr>
<tr><td>LOCATIONS</td><td>n.d.</td></tr>
<tr><td>DESCRIPTION</td><td>A 110 x 20 foot basin was levelled and a 6 inch layer of agricultural-grade limestone was placed and covered with a 10 inch humic layer of an equally proportioned mixture of peat, compost, and sandy soils. The limestone was placed as a distinct layer and not mixed with the humic soil to avoid coating the limestone with a ferric hydroxide precipitate. <i>Typha</i> were collected from surrounding areas and planted at 1 foot centres, and hay bails were placed within the basin. Outflow from the wetland underwent chemical treatment and flowed through a polishing pond before being discharged.</td></tr>
<tr><td>DESIGN CRITERIA</td><td>Constructed to remove iron and manganese from a surface seep flow. Design was based on a flow estimate of 10 gpd following the recommended size of 200 ft^2 of wetland/1 gpm of flow (Kleimnann et al, 1986). Iron concentrations can not be greater than 7 ppm daily, or an average of 3.5 monthly. Manganese limits are 4 ppm daily and 2 ppm for a monthly average (USEPA, 1993).</td></tr>
<tr><td>ADDITIVES</td><td>Chemical treatment facility following outflow location and final polishing pond. Chemical treat of the wetland effluent was discontinued in February.</td></tr>
<tr><td>COSTS</td><td>n.d.</td></tr>
<tr><td>TIME OF OPERATION / INFORMATION</td><td>Constructed in September 1986.
Samples collected on a bi-weekly basis from October 1986 to July 1987.</td></tr>
<tr><td>PERFORMANCE DATA</td><td>
<table>
<tr><td>Average flow</td><td>10 gpm</td></tr>
<tr><td>Average pH</td><td>5.5</td></tr>
</table>
Average Mass Concentrations at Inlet (ppm)
<table>
<tr><td>Fe</td><td>13</td></tr>
<tr><td>Mn</td><td>30</td></tr>
<tr><td>SO_4^{-2}</td><td>990</td></tr>
</table>
The following values have been read from graphs presented in the article:
<table>
<tr><td><u>Mass Budget (kg)</u></td><td></td><td></td><td><u>Concentration (ppm)</u></td><td></td><td></td></tr>
<tr><td></td><td><u>Inlet</u></td><td><u>Outlet</u></td><td></td><td><u>Inlet</u></td><td><u>Outlet</u></td></tr>
<tr><td>Iron</td><td></td><td></td><td>Iron</td><td></td><td></td></tr>
<tr><td>Maximum</td><td>34</td><td>15</td><td>Maximum</td><td>23</td><td>10</td></tr>
<tr><td>Minimum</td><td>2.5</td><td>0.5</td><td>Minimum</td><td>5</td><td>0</td></tr>
<tr><td>Manganese</td><td></td><td></td><td>Manganese</td><td></td><td></td></tr>
<tr><td>Maximum</td><td>56</td><td>42.5</td><td>Maximum</td><td>55</td><td>42</td></tr>
<tr><td>Minimum</td><td>17</td><td>16.5</td><td>Minimum</td><td>18</td><td>18</td></tr>
</table>
74% of the 186.4 kg of Fe introduced to the system was removed. The smallest percentage of Fe removal was 43%, the largest percentage was 94%. Mn removal was only 8.3% of the 368.5 kg introduced into the wetland. Largest Mn removal was 40%.</td></tr>
<tr><td>OVERALL EFFECTIVENESS</td><td>Overall iron retention of the wetland was 74% of the total input. Manganese retention was not as successful with only 8%.
Flow appeared to influence metal removal in the wetland system.
Fe and SO_4^{-2} were removed from solution while the water flowed through a ditch leading from the seep to the wetland.</td></tr>
<tr><td>TOXICITY</td><td>n.d.</td></tr>
<tr><td>MAINTENANCE REQUIREMENTS</td><td>n.d.</td></tr>
<tr><td>MONITORING REQUIREMENTS</td><td>Surface waterflow volume was recorded with inlet and outlet weirs, and rainfall was measured by a continuous recording rain gauge. Water samples were collected at the inlet and outlet of the wetland, and at 33 surface locations. Substrate interstitial water was sampled from six, two level piezometer nests. All samples were analyzed for Ca, Fe, K, Mg, Na, Mn, P, and SO_4^{-2}, on a bi-weekly basis.</td></tr>
<tr><td>TREATMENT PRODUCT DISPOSAL</td><td>n.d.</td></tr>
<tr><td>ENVIRONMENTAL CONDITIONS</td><td>n.d.</td></tr>
</table>

TECHNOLOGY	Constructed Wetlands	WTLD 29
COMMENTS	Periods of increased flows corresponded with the smallest percent reduction of influent mass. The ability to remove Fe and Mn varied with time and location. Inlet flow volume, basin length, and water flow through the substrate were important factors affecting the retention of influent Fe and Mn. Cores taken one month after construction showed that the substrate layers had compacted to approximately 5.5 inches of limestone, and 7 inches of humic material. The wetland responded to increased flow volumes by decreasing the percentage of Fe removal; Fe concentrations at the outlet increased with increasing outflow volume. Effects of length and width on the removal of Fe and Mn from surface water were examined with an analysis of variance. Time effects were not examined. Width was not found to be significant in the removal of either metal, length was significant in both cases, with iron being more affected. Statistics suggest that the rate of iron removal slows during the time it takes water to flow from inlet to outlet. This is a likely possibility since reaction rates can slow as the reactions become more dilute. A wetland designed with a more permeable substrate might provide greater contact between soil and water, provided water remains shallow.	

TECHNOLOGY Constructed Wetlands WTLD 30

PROJECT SCALE Pilot scale

REFERENCE "The Accumulation of Iron Sulfides in Wetlands Constructed for Acid Coal Mine Drainage (AMD) Treatment". Taddeo, F.J. and R.K. Wieder. 1991. MEND Conference Proceedings. pp. 529 - 547.

LOCATIONS Earle C. Clements Job Corps Satellite Facility, Greenville, Kentucky.

DESCRIPTION Five constructed wetlands with different organic substrates.

DESIGN CRITERIA Each wetland was approximately 183 m^2 in area (6.1 m x 30 m). 4 - 6 mm thick liner, 30 cm thick substrate. Wetlands had a substrate of *Sphagnum* peat to which 75 kg of pelleted limestone and 70 kg of 5-10-5 or 10-10-10 fertilizer has been surface applied on a quarterly basis (Peat L&F); *Sphagnum* peat (Peat); sawdust; straw and manure; spent mushroom compost). *Typha latifolia* planted in 1 m x 1 m centres. Wetlands built adjacent to an AMD impacted stream draining 202 ha. Average flow was 9950 L/d with a mean pH of 2.29 and mean concentrations of dissolved Fe and SO_2^- of 121 and 2984 mg/L respectively.

ADDITIVES n.d.

COSTS n.d.

TIME OF OPERATION / INFORMATION July, 1989 to November, 1990
(16 months)

PERFORMANCE DATA

		Changes rel. to initial Iron sulfide-Fe conc. (mg Fe/g dry substr.)			Iron sulfide-Fe
Substrate	Surface	Deep		accum. kg Fe/Wetland	
Peat L&F	5	6		15	
Peat		0.5	0.5		4.5
Sawdust		6	6		75
Straw/manure		14	14		150
Mush'm Compost		20	10		200

		Changes in tot. non-water-soluble iron (mg Fe/g dry substr.)			Tot. non-water-soluble Fe
Substrate	Surface	Deep		accum. kg Fe/Wetland	
Peat L&F	125	30		200	
Peat		100	10		245
Sawdust		100	25		690
Straw/manure		65	35		575
Mush'm Compost		65	25		614

All data read from graphs presented in the paper. Surface is 0-5 cm, deep is at a depth of 25 - 30 cm.

OVERALL EFFECTIVENESS Increases in iron sulfide-Fe concentrations were statistically similar in the surface and the deep substrates in both the *Sphagnum* peat wetland to which lime and fertilizer had been surface applied as well as in the *Sphagnum* peat wetland except for month 1. There were gradual, continual increases in the concentrations of iron sulfide-Fe for the other three wetlands. These increases were more pronounced in the deep substrate for the sawdust wetland. Increases in the iron sulfide-Fe concentrations were more pronounced in the surface substrate of the mushroom compost wetland. Increases in the Fe sulfide-Fe concentration in the straw/manure wetlands were statistically similar in the surface and deep substrates. All wetlands accumulated some Fe as Fe sulfides. The order of accumulation of iron sulfide was mushroom compost > straw/manure > sawdust > *Sphagnum* peat with lime and fertilizer > *Sphagnum* peat.

TOXICITY n.d.

MAINTENANCE REQUIREMENTS n.d.

MONITORING REQUIREMENTS Substrate samples were collected from each wetland one month and every three months after AMD exposure. Twelve surface (0 - 5 cm) and twelve deep (25 - 30 cm) samples were collected from an 3 x 4 grid from each wetland. Flow parameters measured were flow rate, pH, and concentrations of dissolved Fe and SO_4^{-2}.

TREATMENT PRODUCT DISPOSAL n.d.

ENVIRONMENTAL CONDITIONS n.d.

COMMENTS Sulfate reduction occurs in substrates of wetlands receiving AMD drainage. Total Fe accumulation was considerably greater in the sawdust, straw/manure, and mushroom compost wetlands than in the two *Sphagnum* wetlands, and Fe sulfides have the greatest contribution of total Fe accumulation in the straw//manure and mushroom compost wetlands. Data does not support the suggestion that shallow substrate (20 cm) depths area not conducive to the development of permanent anoxic conditions necessary of sulfate reduction.

TECHNOLOGY

Constructed Wetlands WTLD 31

PROJECT SCALE

Full to pilot scale

REFERENCE

"Preliminary Considerations Regarding Constructed Wetlands for Wastewater Treatment". Wieder, K.R., et al. 1989. Constructed Wetlands. pp. 297 - 305.

LOCATIONS

Sixteen wetlands - no locations

DESCRIPTION

n.d.

DESIGN CRITERIA

Flow	Wetland Area (m^2)			
(L/min)	Actual	Recommended by:		
		BOM	TVA (Fe)	TVA (Mn)
50	270	245	2990	3080
38	204	186	3040	3800
43	420	211	24424	451
38	108	186	11248	319
455	3000	2230	61425	4095
80	2597	392	28960	20160
170	4800	833	19125	2312
42	2000	206	11340	7056
348	2500	1705	2871	6264
238	7300	1166	8068	6378
400	11000	1960	32000	26400
87	1200	426	848	870
492	25000	2411	6605	6790
83	3400	104	747	1328
53	5700	260	1192	965
288	9200	1411	151	3053

ADDITIVES

n.d.

COSTS

According to the BOM survey, construction costs prior to 1986 averaged less than \$ 2.96/m^2. The TVA's wetland construction costs prior to 1986 ranged from \$ 3.58/m^2 to \$ 32.03/m^2. Both studies indicate that equipment and labour costs are major components. Subsequent maintenance costs are minimal.

TIME OF OPERATION / INFORMATION

n.d.

PERFORMANCE DATA

Flow (L/min)	pH	Fe (mg/L)	Mn (mg/L)	Is the Wetland Effective (Fe)*
50	5.5	29.9	8.8	No
38	5.5	40	50	Yes
43	2.7	284	1.5	No
38	3.1	148	1.2	No
455	6.0	180	4.5	Yes
80	2.9	181	36	Yes
170	5.6	150	6.8	Yes
42	4.9	135	24.0	Yes
348	6.0	11.0	9.0	Yes
238	5.7	45.2	13.4	Yes
400	3.1	40.0	13.0	Yes
87	6.3	13.0	5.0	Yes
492	5.6	17.9	6.9	Yes
83	5.7	12.0	8.0	Yes
53	6.3	30.0	9.1	Yes
288	5.7	0.7	5.3	Yes

*Wetland effectiveness refers to Fe only. If the constructed wetland did not result in a substantial decrease in Fe concentration in outflow water as compared to Fe concentration in inflow water, the wetland was considered to be ineffective.

OVERALL EFFECTIVENESS

Wetlands areas recommended using the TVA approach are between 2 and 74 times greater than the areas recommended using the BOM approach. In three sites where the constructed wetland failed to substantially improve the mine drainage water, the wetlands were sized more closely with the BOM recommendations than the TVA recommendations. One wetland, however, was only 10% larger than the BOM recommended area and was effective in lowering Fe concentrations.

TOXICITY

n.d.

MAINTENANCE REQUIREMENTS

A wetland at its metal retention capacity may require sludge removal and disposal incurring additional costs.

MONITORING REQUIREMENTS

n.d.

TREATMENT PRODUCT DISPOSAL

n.d.

TECHNOLOGY	Constructed Wetlands WTLD 31
ENVIRONMENTAL CONDITIONS	n.d.
COMMENTS	A frequently used approach to sizing a constructed wetland for mine treatment, according to the U.S. Bureau of Mines, follows a rule of thumb that 200 ft^2 of wetland is needed for each gal/min of flow (or approximately 4.9 m^2 of wetland area for each L/min of flow). This guideline was intended for flows of 5 - 10 gal/min (19 - 38 L/min) with pH values of 4.0 or above and Fe and Mn concentrations less than 50 mg/L and 20 mg/L respectively. The Tennessee Valley Authority provided guidelines based on chemistry and flow of water treated at their sites from wetlands constructed for acid mine drainage with in situ substrate and with typically relatively low organic matter content. The data recommends the following guidelines: to achieve Fe concentrations of 3 mg/L at the wetland discharge, if the inflow drainage has a pH of <5.5, 2 m^2 of wetland is needed for each mg/min, if the inflow drainage has a pH of >5.5, 0.75 m^2 of wetland is needed for each mg/min. Two achieve an outflow Mn concentration of 2 mg/L, if the inflow drainage has a pH of <5.5, 7 m^2 of wetland is needed for each mg/min of Mn entering the wetland, whereas if influent pH is >5.5, 2 m^2 of wetland is needed for each mg/min of Mn entering the wetland.

TECHNOLOGY

Constructed Wetlands WTLD 32

PROJECT SCALE

Pilot, experimental and full scale

REFERENCE

"Performance Expectations for Loading Rates for Constructed Wetlands". Watson, J.T., et al. 1989. Constructed Wetlands. pp. 319 - 351.

LOCATIONS

Various locations in Europe and North America.

DESCRIPTION

Constructed wetland systems for small municipal systems, and constructed wetland systems for acid mine drainage.

Two types of systems were investigated:
- conventional surface flow systems with an exposed free water surface, and;
- subsurface flow systems with water levels below the surface of a permeable substrate.

A constructed wetland designed for surface flow consists of basins or channels, soil or another suitable medium to support emergent vegetation, and relatively shallow water flowing through the unit. If seepage needs to be prevented, a liner is incorporated into the design. Channels are typically long and narrow, ensuring approximate plug-flow conditions.

A subsurface flow system consists of a trench or bed containing a medium that supports growth of emergent vegetation. Media used have included crushed stone, gravel, and different soils, either alone or in combination. Most beds are underlain by impermeable material to prevent seepage and assure water level control. Wastewater flows laterally and is purified using contact with media surfaces and the vegetation root zones. The subsurface zone is saturated and generally anaerobic, but excess oxygen conveyed through the plant root system supports aerobic microsites adjacent to roots and rhizomes.

DESIGN CRITERIA

Site dependent. According to the Tennessee Valley Authority (TVA), for acid mine drainage, loading rates of 2.0 and 7.0 m^2/mg/min for iron an manganese, respectively, are recommended for waters with pH of 5.5. If pH is greater than 5.5, the loading rate is increased to 0.75 m^2/mg/min for iron and 2.0 m^2/mg/min for manganese.

Hydraulic loading rates for acid mine drainage systems range from 61 to 11,000 m^2/L/s (8640 to 0.81 cm/day). A rule of thumb value of 294 m^2/L/s (29.4 cm/day) has been suggested by one source. Preliminary guidelines developed by the TVA are 0.75 - 7.0 m^2/mg/min depending on pH and type of metal.

Paper includes an extensive discussion of guidelines for design based on reaction kinetics for both surface and subsurface flow wetlands.

ADDITIVES

n.d.

COSTS

see performance data

TIME OF OPERATION / INFORMATION

Data published in 1987.

PERFORMANCE DATA

Twenty sites	Mean	Range
Size (m^2)	1550	93 - 6070
# of basins (ponds)	3	1 -7
Size of basin (m^2)	795	19 - 6070
Flow rate (L/s)	1.3	0.06 -12.6
Water Depth (m)	0.3	0 -2
Water chemistry (n=11)		

	Inflow		Outflow	
	Mean	Range	Mean	Range
pH	4.9	3.1 - 6.3	6.0	3.5 - 7.7
Acidity (mg/L)	170	ND - 600	40	ND - 140
Fe (mg/L) 33	0.4 - 220	1.2	0.05 - 7.	
Mn (mg/L) 26	8.7 - 54		15	0.3 - 52
SO_4 (mg/L)	950	270 - 1600	740	160 - 150

Construction Cost: Mean, $10,000; Range, $1,500 - $65,000
Note: ND = below detection limits. Data published in 1987.

OVERALL EFFECTIVENESS

Systems with humic substrates are potentially effective at metal removal because of their ion exchange capacity. However, ion exchange capacity of humic materials or mosses will have limited functional longevity. A *Sphagnum* bed with an area of 1600 m^2 and a depth of 30 cm treating a mine drainage flow of 40 l/min and an iron concentration of 200 mg/L may be saturated in only 2.2 years.

Constructed wetlands (marshes) have high removal efficiencies for iron, and lower efficiencies for manganese. Iron and manganese removal efficiencies ranged from 0 to 99% (median of 96%) and -8% to 96% (median of 83%), respectively, in one study of 10 constructed wetlands.

pH changes in *Sphagnum* wetlands range from a decrease to no change in newly constructed systems to an increase in an existing natural system. Cattail and mixed-species systems tend to increase pH from inlet to outlet. Mean influent pH of 20 wetlands was 6.0. Median effluent pH for 10 wetland systems was 6.5.

Sphagnum systems tend to decrease pH, while cattail and mixed-species systems tend to increase pH.

TECHNOLOGY	Constructed Wetlands WTLD 32
TOXICITY	n.d.
MAINTENANCE REQUIREMENTS	n.d.
MONITORING REQUIREMENTS	Flow, hydraulic loading, metal concentration, pH.
TREATMENT PRODUCT DISPOSAL	n.d.
ENVIRONMENTAL CONDITIONS	n.d.
COMMENTS	Only limited information is available on metals removal. Wetlands soil (natural and acid) were initially more efficient substrates than spoil, normal soil, and others, but differences were minimal at the end of the first growing season and thereafter. Algal death, precipitation, and burial immobilizes metal for long periods. Most systems treating acid mine drainage are surface flow systems. Subsurface flow systems can provide more contact surface for absorption, but they are likely to clog from substantial deposits of insoluble metal compounds. One important acidity generation mechanism is production of humic substances, much of which consists of organic acids. In wetlands receiving acid mine drainage, oxidation of ferrous iron to ferric iron and subsequent hyrolyzation to ferric hydroxide may be an important source of acidity. Formation of carbon dioxide and its soluble forms, carbonate and bicarbonate, will increase acidity. Reduction of nitrate and sulfate results in a net production of alkalinity. Removal of carbon dioxide increases pH and alters forms of alkalinity but not amounts. This is common in algal-dominated systems such as nutrient-rich ponds, but a reversal occurs at night.

TECHNOLOGY Constructed Wetlands WTLD 33

PROJECT SCALE Pilot scale

REFERENCE "Constructed Wetlands that Emphasize Sulfate Reduction: A Staged Design Process and Operation in Cold Climates". Wildeman, T.R., et al. 1992. Proceedings - 24th Annual Meeting Canadian Mineral Processors.

LOCATIONS Big Five Pilot Wetlands in Idaho Springs, CO.

DESCRIPTION Original design and construction was completed in the summer and fall of 1987, and consisted of 3 cells 18.6 m^2 in size. Cell A contained a mushroom compost substrate, Cells B and C contained a substrate of equal parts of peat, aged manure, decomposed wood. In August 1989, Cell B was divided onto two identical cells that can be operated in either up flow or downflow configurations. Cell E was constructed when Cell B was remodelled. The cell was constructed using the original substrate from Cell B, and designed to operate as a downflow, subsurface wetland. Cells B and E were designed as subsurface flow cells.

DESIGN CRITERIA

Initial Operation - December 11, 1987

	Flow Rate (gpm)
Cell A	1.0
Cell B	1.0
Cell C	1.0

August 19, 1988

	Flow Rate (gpm)
Cell A	0.51
Cell B	0.24
Cell C	0.34

Cell A Modification - February 21, 1989

	Flow Rate (gpm)
Cell A	0.28
Cell B	0.31
Cell C	0.32

May 6, 1989

	Flow Rate (gpm)
Cell A	0.92
Cell B	0.83
Cell C	0.81

August 1, 1989

	Flow Rate (gpm)
Cell A	0.30
Cell B	0.48
Cell C	0.43

Cell B Modification - October 3, 1989

	Flow Rate (gpm)
Cell B-up	0.29
Cell B-down	0.16
Cell E	0.062

November 5, 1989

	Flow Rate (gpm)
Cell B-up	0.22
Cell B-down	0.24
Cell E	0.11

August 19, 1988

	Flow Rate (gpm)
Cell B-up	0.21
Cell B-down	0.20
Cell E	0.10

The area of Cells A, B, and C is 200 ft^2; the area of Cells B-up, B-down, and E is 100 ft^2.

ADDITIVES n.d.

COSTS n.d.

TIME OF OPERATION / INFORMATION December 11, 1989 to August 1, 1989 (Cell A, B, and C).
October 3, 1989 to January 13, 1990 (Cell B up and downflow, and Cell E).

TECHNOLOGY Constructed Wetlands WTLD 33

PERFORMANCE DATA

Water Sample	Mn (mg/L)	% red.	Fe (mg/L)	% red.	Zn (mg/L)	% Red.	Cu (mg/L)	% Red.	pH
Initial Operation - December 11, 1987.									
Mine Drainage	34	--	32	--	10.6	--	1.02	--	2.8
Cell A	27	21	18	45	7.8	27	0.44	57	4.6
Cell B	33	1	24	26	9.8	12	0.89	12	3.1
Cell C	34	0	22	32	9.6	9	0.91	10	3.3
August 19, 1988									
Mine Drainage	26	--	37	--	8.1	--	0.91	--	2.9
Cell A	25	4	20	56	<0.1	100	0.17	81	5.5
Cell B	26	0	15	59	6.1	24	0.55	40	3.2
Cell C	25	4	11	70	5.8	28	0.38	58	3.5
Cell A modification - February 21, 1989									
Mine Drainage	27	--	32	--	9.3	--	0.56	--	3.0
Cell A	22	16	12	63	4.5	52	<0.05	100	5.1
Cell B	27	0	28	13	6.1	34	0.82	0	3.4
Cell C	25	7	31	3	7.2	23	0.26	53	3.5
May 6, 1989									
Mine Drainage	30	--	42	--	10.4	--	0.76	--	3.0
Cell A	33	-10	28	33	7.8	25	<0.05	100	3.5
Cell B	27	10	10	76	6.2	40	0.36	53	3.0
Cell C	29	3	9	79	6.4	38	0.46	39	3.2
August 1, 1989									
Mine Drainage	32	--	43	--	9.4	--	0.75	--	2.9
Cell A	31	3	39	9	5.2	45	<0.05	100	4.1
Cell B	26	19	24	44	6.6	30	0.46	39	3.1
Cell C	32	0	18	58	4.8	48	0.09	88	3.7
Cell B Modification - October 3, 1989									
Mine Drainage	35	--	46	--	9.9	--	0.66	--	3.2
Cell B-up	34	3	39	15	9.3	6	0.59	11	3.5
Cell B-dn	23	34	8	83	0.8	92	<0.05	100	6.5
Cell E	24	31	<1	100	<0.01	100	<0.05	100	6.3
November 5, 1989									
Mine Drainage	32	--	38	--	8.7	--	0.61	--	2.9
Cell B-up	31	3	17	55	8.4	3	0.48	21	3.6
Cell B-dn	20	38	<1	100	6.0	31	<0.05	100	5.9
Cell E	20	38	<1	100	<0.1	100	<0.05	100	6.5
January 13, 1990									
Mine Drainage	31	--	33	--	9.0	--	0.59	--	2.9
Cell B-up	30	3	33	0	8.9	1	0.49	17	3.2
Cell B-dn	27	13	12	64	8.1	10	0.44	25	3.2
Cell E	28	10	10	70	<0.1	100	<0.05	100	6.0

Cell B-dn = Cell B-downstream

OVERALL EFFECTIVENESS

Cu and Zn were reduced to below detection limits of 50 mg/L, Fe was reduced by at leat 90% to below 3 mg/L, pH was raised from below 3 to above 6. From the initial flow on September 1, 1989, removal of Cu, Zn, and Fe has been 100%, and Mn removal averages 25%.

TOXICITY

n.d.

MAINTENANCE REQUIREMENTS

n.d.

MONITORING REQUIREMENTS

n.d.

TREATMENT PRODUCT DISPOSAL

n.d.

ENVIRONMENTAL CONDITIONS

A key factor in sulfate reduction is to insure that the optimum micro-environment for sulfate reducers be maintained. The most important environmental conditions are reducing conditions and a pH of around 7.

Reasons for winter success of Cell A:
It is continuously in the sun.
More of the water flows through the substrate rather than across the surface.
The inlet is small and insulated so the energy within the water is not lost.
Cell B: Operated over the winter primarily because the surface of the cells were insulated with hay and plastic.
Cell E: Is shallow but is a subsurface flow cell. Much of the water that enters the cell flows across the surface and over the spillway. This excess water insulates the substrate.

Guidelines for winter operation:

- Use thermal energy within the mine drainage water to the best advantage. Insure that delivery systems are insulated. Keep inlet structures small and insulated.
- Place wetland cells so they receive winter sun. If this cannot be completely achieved, at least insure that outlets are in winter sun.
- Insulate the top of the cell with excess mine water or with hay and plastic.
- Insulate the outlet and allow a method for the effluent to flow away from places where it could cause freezing problems.
- Design subsurface flow systems. The thermal energy within the substrate will aid operation, whereas in a surface flow system the waters are exposed to the elements.

COMMENTS

From the beginning Cell A performed better than Cells B and C. It was realized that treatment was primarily through microbial process within the substrate and that plants were not needed to insure success. Sulfate reduction followed by sulfide precipitation is the primary heavy metal removal process. A downflow, trickling filter style of configuration achieves the best contact of water with the substrate.

Removing efficiencies depends largely on loading factors. At high flows through the substrate, sulfide will be the limiting reagent, the microbial environment will be under stress to produce more sulfide, the pH of the micro-environment will drop, and removal will be inconsistent. At low flows, iron will be the limiting reagent, the excess sulfide will insure a reducing environment and a pH near 7, microbial populations will remain healthy, and removal of the metal contaminants will be consistent and complete.

TECHNOLOGY	Constructed Wetlands WTLD 34
PROJECT SCALE	Pilot scale
REFERENCE	"Metal Removal Efficiencies from Acid Mine Drainage in the Big Five Constructed Wetland". Wildeman, T.R., et al. 1990. Proceedings of the 1990 Mining and Reclamation Conference and Exhibition, Volume II, Charleston, West Virginia. pp. 417 - 424.
LOCATIONS	Idaho Springs-Central City mining district of Colorado.
DESCRIPTION	A number of different substrate compositions in the wetland have been used. The system has a length of 18.3 m, a width of 3.05 m and a depth of 1.22 m, and is divided into three 6.1 m long cells with an area of 18.6 m^2. Flow into each cell is controlled by valves, and rock boxes were constructed at the upstream end of each cell forcing water to flow downward into the substrate. Cells contain a 1 m organic substrate. Cell A was filled with fresh mushroom compost, which consists of approximately 50% animal manure and 50% barley mash waste. Substrate of Cell B consisted of a mixture of equal parts of peat, aged steer manure, and decomposed wood shavings and sawdust. Cell C was lined to a depth of 10 - 15 cm with 5 - 8 cm limestone rock, then filled to a 1 m depth with the same organic mixture as Cell B. Cattail sedge and rush were transplanted from similar locations to each of the cells.
DESIGN CRITERIA	A retention time of 70 hours is estimated for a flow rate of 4 L/min.
ADDITIVES	n.d.
COSTS	n.d.
TIME OF OPERATION / INFORMATION	December 11, 1987; February 13, May 31, August 19, December 18, 1988; February 21, May 6, August 1, December 3, 1989. Focus of the paper is on Cell A.

PERFORMANCE DATA

	Dec. 11/87		Feb 13/88		May 31/88		Aug 19/88	
	Mine Drainage	Output A	Mine Drainage	Output A	Mine Drainage	Output A	Mine Drainage	Output A
Mn	34	27	28	27	25	25	26	25
% Dec.	--	21	--	4	--	0	--	4
Fe	32	18	28	18	44	27	37	20
% Dec.	--	45	--	36	--	39	--	46
Zn	10.6	7.8	8.2	8.9	8.1	5.4	8.1	<0.1
% Dec.	--	27	--	28	--	33	--	100
Cu	1.02	0.44	0.89	0.14	0.75	0.03	0.91	0.17
% Dec.	--	57	--	84	--	96	--	81
SO_4^{-2}	1750	1560	1750	1690	1500	1330	1460	650
pH	2.8	4.6	3.3	4.7	3.0	4.3	2.9	5.5
Flow (L/min)	--	4	--	4	--	4	--	1.9

	Dec. 18/88		Feb 21/89		May 6/89		Aug 1/89	
	Mine Drainage	Output A	Mine Drainage	Output A	Mine Drainage	Output A	Mine Drainage	Output A
Mn	29	28	27	22	30	33	32	31
% Dec.	--	3	--	19	--	- 10	--	3
Fe	38	31	32	12	42	28	43	39
% Dec.	--	18	--	63	--	33	--	9
Zn	9.2	8.6	9.3	4.5	10.4	7.8	9.4	5.2
% Dec	--	7	--	52	--	25	--	45
Cu	0.80	0.62	0.56	<0.01	0.76	<0.05	0.75	<0.05
% Dec	--	23	--	100	--	100	--	100
SO_4^{-2}	1710	1710	1860	1690	1760	1690	1740	1700
pH	3.0	3.4	3.0	5.1	3.0	3.5	3.0	4.1
Flow (L/min)	--	4.6	--	1.1	--	3.3	--	1.1

	Dec 3/89	
	Mine Drainage	Output A
Mn	26	30
% Dec.	--	-3
Fe	43	21
% Dec.	--	49
Zn	9.0	1.3
% Dec	--	14
Cu	0.58	<0.05
% Dec	--	100
SO_4^{-2}	1700	1690
pH	3.0	6.0
Flow (L/min)	--	1.1

Note: All metal concentrations in mg/L.

TECHNOLOGY	Constructed Wetlands WTLD 34
OVERALL EFFECTIVENESS	Metal concentration decreases range from basically none for Mn to essentially complete removal for Cu. Decreases in concentrations for Fe of up to 63% and for Zn of up to 100% are found. Decreases in sulfate concentrations of about 10% are typical in Cell A. Fe and Zn generally show a greater removal as the flow rate is decreased but Cu and Mn are not easily related. Below a flow of about 1.5 L/min, Cu is uniformly removed and below this flow appreciable Zn is always removed. It appears that below 1.8 L/min, Mn is not removed from solution and may be redissolved. If metal removal occurs through sulfide formation, Cu removal may also occur at higher flow rates because copper sulfides are very insoluble. The Cu removal process is not consistent. pH and sulphide removals vary with flow, with better removal as flow is decreased. Eh generally decreases with flow rate. Data indicate that processes in the substrate affecting pH, Eh, and sulfide concentration operate at much greater efficiencies below a certain flow rate. The three parameters are related to microbiological processes in the wetland.
TOXICITY	n.d.
MAINTENANCE REQUIREMENTS	n.d.
MONITORING REQUIREMENTS	Mine drainage water and cell effluents were sampled about twice monthly with more frequent samplings in the summer seasons. At least two duplicate water samples were taken monthly. Data for the duplicates have been averaged.
TREATMENT PRODUCT DISPOSAL	n.d.
ENVIRONMENTAL CONDITIONS	Mass removal data indicate that there are some tendencies for greater metal removal rates and pH increases during the summer season than for winter; however, Mn showed no trend. Some of the largest fluctuations in mass removal data occur in spring when large fluctuations in temperature occur. Small changes in temperature of 3 - 4 °C can cause the interdependent community of microbes operating in a reducing system to go dormant for several days until the microbes are able to readjust to the different temperature. Such a variable could greatly affect seasonal trend, and it may produce the greatest instability in the fall and spring when temperature fluctuations are the largest.
COMMENTS	The decrease in metal concentrations and the increase in pH is related to the rate of drainage flow through the wetland. For Fe and Zn, decreases in concentrations generally vary inversely with flow rate as do increases in pH. This information can be used in determining the maximum loading rate for any given mental removal efficiency. Mass removal data (mg/d/m^2) indicate that there are some tendencies for greater metal removal rates and pH increases during the summer season for the winter. No obvious relationships are found between mass removal data and flow rate for Fe, Zn, Cu, and Mn. In 1988, the clogging of the mine drainage flow lines inadvertently allowed observations of much larger increase in pH and larger decreases in metal concentrations with reduced flow. Sulfate reducing bacteria were found to be dominant throughout the wetland substrates whereas metal-oxidizing bacteria were found only very near the surface. Rock boxes did not work because the accumulation of metal hydroxides appeared to clog the downward flow of water into the substrate. Eh of the effluent and the decrease in sulfate concentration is very responsive to changes in flow rate. Processes in the substrate affecting Eh, pH and sulfate concentration operate at much greater efficiencies below flow rates of about 1.5 L/min. Consistently good results for Cu, Fe, and Zn removal are obtained using a substrate of fresh mushroom compost with flow rates less than 1.5 L/min in a system with an area of 18.6 m^2. Mass removal data does not appear to be as good an indicator of how well a wetland is removing pollutants.

<table>
<tr><td>TECHNOLOGY</td><td>Constructed Wetlands WTLD 35</td></tr>
<tr><td>PROJECT SCALE</td><td>Pilot scale</td></tr>
<tr><td>REFERENCE</td><td>"Passive Bioremediation of Metal from Water using Reactors or Constructed Wetlands". Wildeman, T.R., et al. 1993. Proceedings of the 2nd International Symposium on In Situ and On-Site Bioreclamation.</td></tr>
<tr><td>LOCATIONS</td><td>Big Five Tunnel in Idaho Springs, Colorado.</td></tr>
<tr><td>DESCRIPTION</td><td>Anaerobic wetland containing a number of cells for acid mine drainage. Cell E was designed using the guidelines below.</td></tr>
<tr><td>DESIGN CRITERIA</td><td>The following guidelines were applied when designing Cell E:

Wetland substrate was formulated so that organic material necessary for metabolism is in high abundance and the soil can provide acid buffering capacity at a pH above 7.
Microbial processes that transform strong acids such as H_2SO_4 in weak acids such as H_2S were promoted.
The products of these reactions were used to precipitate metal contaminants as sulfides (CuS, ZnS, PbS, CdS), hydroxides ($Al(OH)_3$, $Ce(OH)_3$), and carbonated ($MnCO_3$).
To remain effective, the reactions that consume H^+ have to predominate over the reactions that produce H^+.

Flow data not given.</td></tr>
<tr><td>ADDITIVES</td><td>n.d.</td></tr>
<tr><td>COSTS</td><td>n.d.</td></tr>
<tr><td>TIME OF OPERATION / INFORMATION</td><td>The pilot scale facility has been operating and performance monitored for more than 2 years. A number of cells, including Cell C was constructed between 1987 and 1991. Removal efficiencies for Cell E have been collected over a 27 month period beginning in September 1989.</td></tr>
<tr><td>PERFORMANCE DATA</td><td>Mine drainage influent: (mg/L)
<table>
<tr><td>pH</td><td>SO_4^{-2}</td><td>Cu</td><td>Fe</td><td>Mn</td><td>Zn</td></tr>
<tr><td>3.0</td><td>1720</td><td>0.57</td><td>39</td><td>31</td><td>8.6</td></tr>
</table>
Cell E Removal Trends (ratios)
<table>
<tr><td></td><td></td><td colspan="2">Outflow Conc./Inflow Conc.</td><td></td></tr>
<tr><td><u>Parameter</u></td><td><u>Maximum</u></td><td><u>Minimum</u></td><td></td></tr>
<tr><td>Mn</td><td></td><td>11.5</td><td>0.60</td></tr>
<tr><td>Fe</td><td></td><td>1.50</td><td>0.80</td></tr>
<tr><td>Cu</td><td></td><td>0.0</td><td>0.0</td></tr>
<tr><td>Zn</td><td></td><td>0.20</td><td>0.0</td></tr>
<tr><td>SO_4</td><td></td><td>1.0</td><td>0.0</td></tr>
</table>
Data read from the graph presented in the paper. Removal is determined by dividing the wetland effluent by the mine drainage influent concentrations. If removal was complete, the ratio will be close to zero.</td></tr>
<tr><td>OVERALL EFFECTIVENESS</td><td>There was consistent and complete removal of Cu and Zn. Removal of Fe changes with the seasons; it is good in the summer and poor in the winter.</td></tr>
<tr><td>TOXICITY</td><td>n.d.</td></tr>
<tr><td>MAINTENANCE REQUIREMENTS</td><td>n.d.</td></tr>
<tr><td>MONITORING REQUIREMENTS</td><td>n.d.</td></tr>
<tr><td>TREATMENT PRODUCT DISPOSAL</td><td>n.d.</td></tr>
<tr><td>ENVIRONMENTAL CONDITIONS</td><td>Removal of Fe changes with the seasons being relatively high in the summer and low in the winter. Because FeS is more soluble than CuS or ZnS, the winter increase is probably due to the reduction in activity of sulfate-reducing bacteria with decrease in temperature. Reduction of bacterial activity appears to be confirmed by the fact that there is a smaller decrease in sulfate concentration in the winter. Because Mn is the most soluble sulfide, its removal is inconsistent.</td></tr>
<tr><td>COMMENTS</td><td>The following guidelines should be followed for wetland design: the pH of the effluent water should be above 5.5 and dissolved bicarbonate should be present. Any processes that will raise the pH and add alkalinity, such as anoxic limestone drains, should be used. Precipitation of iron and manganese oxyhydroxides should be promoted. Precipitation is a primary removal process and other metal contaminants are removed by adsorption onto these precipitates or by precipitating as carbonates. Plants should be incorporated into the design because photosynthesis is a primary process for raising pH, adding oxygen to the water, and supplying organic nutrients.</td></tr>
</table>

TECHNOLOGY

Constructed Wetlands WTLD 36

PROJECT SCALE

Pilot system testing wetland effectiveness on treatment of effluent from a base or precious metal mining operation containing abundant pyrite.

REFERENCE

"Use of Wetlands for Treatment of Environmental Problems in Mining: Non-coal Mining Applications". Wildeman, T.R. and L.S. Laudon. 1989. Constructed Wetlands. pp. 221 - 231.

LOCATIONS

Description of sites in Ontario, Minnesota, and Montana, and a more detailed description of the Big Five Tunnel in Idaho Springs.

DESCRIPTION

Ontario: Red Lake pilot system - Contaminated subsurface water was intercepted and treated in a polishing ditch and pond that provided additional surface area for precipitation of iron hydroxide. In a pond receiving seepage from tailings, cattail growth was promoted. Maintaining reducing conditions for the seepage prevents further release of precipitate by algae which is the polishing agent. Algal growth in a tailings pond was stimulated by adding material that increased surface area growth.

Minnesota: A naturally occurring white cedar wetland at Danka Mine - LTV Steel Mining Company's open pit taconite mine in northeastern Minnesota receives contaminated drainage from mine dewatering and stockpile runoff. The major source of metals is stockpile drainage.

Montana: Constructed wetland near Sand Coulee.

Colorado: The Big Five Tunnel Site was constructed as a pilot treatment system with controlled inputs and outputs. Drainage flow and chemistry is nearly constant throughout the year. Three cells of 19 m^2 each were constructed. Cell A was filled with mushroom compost; Cell B was filled with a mixture of one-third peat, one-third aged manure, and one-third decomposed wood products; Cell C was filled with 10 - 15 cm of limestone rock (5 cm pieces) and covered with the same organic mixture as Cell B.

DESIGN CRITERIA

Stockpile drainage from the Danka Mine in Minnesota averages 18 mg/L Ni, and 0.62 mg/L Cu; and has flow rates ranging from zero to 16 L/sec.

ADDITIVES

Ontario: Algal growth in a tailings pond was stimulated by adding material that increased surface area growth.

COSTS

n.d.

TIME OF OPERATION / INFORMATION

Samples at the Big Five Tunnel Site were taken on 11 December 1987, 13 February 1988, 4 May 1988 and 31 May, 1988.

PERFORMANCE DATA

Concentrations in mg/L of constituents and percent metal reduction of waters at the Big Five Tunnel Pilot Project:

	Mine Drainage	Output A	Output B	Output C
December 11, 1987				
Al	5.8	2.7	5.0	5.0
%Red.	--	53	14	14
Cu	1.02	0.44	0.89	0.91
%Red.	--	57	12	10
Fe	32	18	24	22
%Red.	--	45	26	32
Mn	34	27	33	34
%Red.	--	21	1	0
SO_4^{2-}	1750	1560	1430	1520
pH	2.8	4.6	3.1	3.3
February 13, 1988				
Al	5.9	2.7	5.6	5.7
%Red.	--	54	5	3
Cu	0.89	0.14	0.92	0.92
%Red.	--	84	0	0
Fe	28	18	28	28
%Red.	--	36	0	0
Mn	28	27	31	29
%Red.	--	4	0	0
SO_4^{2-}	1750	1690	1780	1700
pH	3.3	4.7	3.4	3.4

TECHNOLOGY

Constructed Wetlands WTLD 36

PERFORMANCE DATA

May 4, 1988

Al	5.7	4.1	5.1	5.9
%Red.	--	28	10	0
Cu	093	0.14	0.71	0.80
%Red.	--	85	23	14
Fe	32	21	19	24
%Red.	--	34	40	20
Mn	30	26	26	28
%Red.	--	13	13	7
SO_4^{2-}	1760	1720	1550	1600
pH	3.0	3.7	3.1	3.1

May 31, 1988

Al	4.8	3.0	4.7	4.8
%Red.	--	38	3	0
Cu	0.75	0.03	0.64	0.68
%Red.	--	96	15	9
Fe	44	27	17	21
%Red.	--	39	61	52
Mn	25	25	25	25
%Red.	--	0	0	0
SO_4^{2-}	1500	1330	1570	1220
pH	3.0	4.3	3.0	3.0

OVERALL EFFECTIVENESS

Ontario: Results from a tailings site in Sudbury indicate precipitation of oxides is the main metal removal mechanism and metal uptake by algae is secondary. Experiments with algae conducted in tailings ponds at several sites indicate that algae concentrate uranium, zinc, copper, nickel, and radium-223 under alkaline conditions.

Minnesota: During a year long study at the Dunka Mine peatland, overall Ni and Cu removal in the peatland was 84% and 92% respectively, with peat uptake accounting for most of the metal removal.

Montana: Laboratory experiments on substrate samples showed little remediation potential in the constructed wetland when compared to a natural wetland that had been receiving acid mine drainage from an abandoned lead-zinc mine. In situ study of the natural wetland found removal efficiencies from metals to be 70% for Fe, 14% for Cu, 5.8% for Zn, 0.7% for Mn, and 0.3% for Cd. The constructed wetland substrates did not effectively remove iron or acidity, due to reduced microbial populations.

Big Five Tunnel, Colorado: Based on four sets of samples, the section with mushroom compost (Cell A) had the highest metal removal efficiency and effluent pH. Metal reduction ranges from almost none for Mn to complete removal for Pb and Cu in Cell A.

TOXICITY

n.d.

MAINTENANCE REQUIREMENTS

n.d.

MONITORING REQUIREMENTS

Flow and metal concentrations and loads, and pH.

TREATMENT PRODUCT DISPOSAL

n.d.

ENVIRONMENTAL CONDITIONS

Big Five Tunnel Site, Colorado: Although sulfate in the water is not significantly reduced, sulfate-reducing bacteria are present even during cold weather. Algae were present through winter and spring, with the largest concentration in Cell A. September transplanting did not allow plants to become established, however, all species were vigorously growing in the spring.

COMMENTS

Big Five Tunnel Site, Colorado: Iron hydroxide precipitate was present in all sections of the wetland; iron oxidizing bacteria not present in the original materials are present in the substrates.

MICROBIAL REACTOR SYSTEMS
PHASE I CASE STUDIES

TECHNOLOGY	Microbial Reactor Systems MRS 1
PROJECT SCALE	Pilot scale field test (concurrent laboratory test)
REFERENCE	"Microbial Process for the Treatment of Acidic Drainage at the Halifax Airport" , Bechard, G., Goudey, P, Rajan, S. and R.G.L. McCready. 1991. MEND Conference Proceedings. pp 171-183.
LOCATIONS	Halifax Airport, Nova Scotia
DESCRIPTION	AMD was treated in cells containing straw and wood shavings
DESIGN CRITERIA	AMD waters diverted from ditch to 3 clay-lined cells 16.8 m x 1 m (l x w) in series
ADDITIVES	Sulphate reducing bacteria from laboratory test; sugar and urea added fall 1989, late spring 1990, fall 1990.
COSTS	nd
TIME OF OPERATION / INFORMATION	August 1989 - December 1990
PERFORMANCE DATA	From thaw (April 1990) to July 1990, graphed data showed: Influent Effluent pH 3 - 5.8 6 - 6.8 Alum 8 - 50 mg/L < 4 mg/L Iron 10 - 70 mg/L 0 - 60 mg/L Iron removal decreased after June 1990; after July 1990 all data became inconsistent.
OVERALL EFFECTIVENESS	Iron removal efficiency declined in June; author suggests due to either oxidizing conditions in first cell or saturation of woodwaste and straw with iron hydroxides. Straw was added to the system after three months to increase biomass content and re-establish reducing conditions in the first cell; at that time the system became ineffective in removing aluminum and iron because of the reintroduction of oxygen during addition of straw; anaerobic conditions did not become reestablished during the period of study described. Successful mitigation of AMD at pH 3.5 was recorded for 18 months of a similar laboratory scale test; iron for 12 months; aluminum for 5 months.
TOXICITY	n.d.
MAINTENANCE REQUIREMENTS	Sugar and urea added in late spring and fall; straw and swamp mud mixed in July 1990; sulphate reducing bacteria and straw added on top September 1990.
MONITORING REQUIREMENTS	Flow rate, water temperature, pH monitored in field; samples collected for weekly analysis of pH, acidity and various other parameters; microbial analysis at two depths monthly
TREATMENT PRODUCT DISPOSAL	nd
ENVIRONMENTAL CONDITIONS	Mitigation of AMD occurred at temperatures as low as 0EC, system was not designed to regulate flow sufficiently enough to account for precipitation changes.
COMMENTS	

TECHNOLOGY	Microbial Reactor Systems MRS 2
PROJECT SCALE	Bench scale
REFERENCE	"The Use of Bench Scale Permeameters for Preliminary Analysis of Metal Removal from Acid Mine Drainage by Wetlands". Bolis, J.L., Wildeman, T.R. and R.R Cohen. 1991. National Meeting of the American Society for Surface Mining and Reclamation.
LOCATIONS	National and Quartz Hill Tunnel drainages, Central City-Idaho Springs area, Idaho, USA.
DESCRIPTION	AMD from abandoned mines passed through one of three bench scale permeameters containing various high-alkalinity substrates.
DESIGN CRITERIA	Permeameters consisted of 32-gallon plastic pails fitted with PVC piping and without valve controls. Three cells designed: Cell A contained dry non-inoculated substrate, Cell B contained non-inoculated substrate soaked for one week with city water prior to operation, and Cell C contained soaked substrate inoculated with sulfate reducing bacteria.
ADDITIVES	Substrate consisted of cow manure and planter soil in a 3:1 ratio, with a gravel bed.
COSTS	n.d.
TIME OF OPERATION / INFORMATION	Experiment operated for 19 weeks from June - October 1990. Flowrates for the National Tunnel cells were increased from 10 to 30 mL/min after week 6, with Cell A flow configuration changed from downflow to upflow during weeks 9-13. Flowrates for Quartz Hill Tunnel were ramped from 1 to 3 mL/min, and Cell A flow configuration changed from downflow to upflow after week 13.
PERFORMANCE DATA	

		Influent (mg/L)	Effluent (%decrease, or mg/L) A	 B	 C
National Tunnel					
	pH	5.2 - 5.9	6.6 - 7.7	6.2 - 7.7	6.4 - 7.7
	Cu	0.08 - 0.45	20 - 98.3%	33.3 - 97.5%	50 - 98.3%
	Fe	34.1 - 54.8	82.3 - 98%	54 - 99.9%	72.4 - 99.8%
	Mn	19 - 24.8	7.5 - 98.7%	21.1 - 96.7%	21.9 - 94.0%
	Zn	7.9 - 9.5	94.1 - 99.4%	93.2 - 99.4%	95.5 -99.9%
	SO_4	772 - 1027	664 - 1317	47 - 3260	86 - 1200
Quartz Hill					
	pH	2.3 - 2.9	7.1 - 8.6	6.4 - 8.0	6.4 - 8.2
	Cu	47.5 - 69.8	99.3 - 99.9%	98.9 - 99.9%	98.6 - 99.9%
	Fe	626 - 900.5	98.7 - 99.9%	82.2 - 99.9%	93.9 - 99.9%
	Mn	70.5 - 95.9	98.1 - 99.2%	38.6 - 99.4%	47.4 - 99.0%
	Zn	101.5 - 190	95.4 - 99.9%	94.3 - 99.9%	96.8 - 99.9%
	SO_4	3090 - 4380	280 - 3430	188 - 3574	109 - 3340

OVERALL EFFECTIVENESS	In general, removal of metals was achieved in excess of 95% and a pH increase of 5.6 to 7.7 for National Tunnel; > 99% metals removal and increase in pH from 2.5 to 7.4 for Quartz Hill Tunnel.
TOXICITY	n.d.
MAINTENANCE REQUIREMENTS	Regular checks were performed on permeameter flowrates and periodic cleanings performed to rid system of iron hydroxide clogging.
MONITORING REQUIREMENTS	n.d.
TREATMENT PRODUCT DISPOSAL	n.d.
ENVIRONMENTAL CONDITIONS	n.d.
COMMENTS	Experiments indicate that bench scale permeameters can be used to evaluate metal removal from AMD and to attain design criteria for larger scale wetlands.

TECHNOLOGY	Microbial Reactor Systems MRS 3
PROJECT SCALE	Pilot scale
REFERENCE	"Treatment of Metal-Contaminated Water Using Bacterial Sulfate Reduction: Results from Pilot-Scale Reactors". Dvorak, D.H., et al. 1991. MEND Conference Proceedings.
LOCATIONS	i) Experimental Mine Site, U.S. Bureau of Mines Pittsburgh Research Centre, Pittsburgh, PA ii) Smelting Residues Dump, former New Jersey Zinc Co. plant, Palmerton, PA.
DESCRIPTION	AMD in experimental mine site treated through anaerobic compost-filled capped barrels. Metal-contaminated drainage from smelter site treated through anaerobic compost-filled covered tanks.
DESIGN CRITERIA	Pittsburgh barrel system consisted of 3 x 200 L barrels in series, receiving water from a 1,140 L tank. Palmerton system consisted of 2 independent 4,500 L tanks receiving water from a 3,500 L supply tank. Both systems contained a loosely packed compost mixture.
ADDITIVES	Compost mixture consisted of spent mushroom, manure, hay, straw, corn cobs, and wood chips, conditioned with gypsum and limestone. Contained 50-60% organic matter and 10-15% pulverized limestone.
COSTS	n.d.
TIME OF OPERATION / INFORMATION	Pittsburgh barrel system was operated underground at a constant temperature of 10 EC. Allowed 3.5 weeks of leaching (3,940 L of flow) and results collected for 14 weeks after that period (6,110 L of flow). Mean flow rate was 55 mL/min for duration of experiment. Palmerton system allowed to leach for 6 weeks (4,140 L of flow) and measured for 16 weeks (11,000 and 19,300 L of flow, respectively) thereafter. Mean flow rate was 70 mL/min increased to 131 mL/min after 4 weeks.

PERFORMANCE DATA

	Pittsburgh		Palmerton	
	Influent	Effluent	Influent	Effluent
pH	3.7	6.9	6.3	6.9
Alkalinity (as $CaCO_3$)	17	632	17	1102
Fe (total)	67	< 0.2	--	--
Al (dissolved)	7	< 0.2	--	--
Ca	294	518	275	720
Zn	--	--	302	0.42
Mn	--	--	24	0.50
Ni	--	--	0.85	0.03
Cd	--	--	0.29	< 0.005
Sulfate	973	712	2989	2352
Dissolved Sulfide	nd	34	nd	109

OVERALL EFFECTIVENESS	Both systems lowered contaminant metal concentrations by > 95%, sulfate concentrations by > 30%, and both produced effluent with high alkalinity due to limestone dissolution.
TOXICITY	n.d.
MAINTENANCE REQUIREMENTS	n.d.
MONITORING REQUIREMENTS	Influent, effluent, and porewater samples collected at 1-2 week intervals throughout course of experiments.
TREATMENT PRODUCT DISPOSAL	n.d.
ENVIRONMENTAL CONDITIONS	Palmerton reactor system operated at temperatures between 18 and 24 EC. Pittsburgh system operated at a constant 10 EC.
COMMENTS	Experiments suggest that anaerobic reactors with organic matter can remove metals from contaminated water. Precipitation of metals as insoluble sulfides by bacterially generated H_2S was identified as an important process. Enhancement of metal retention would be achieved by increased bacterial sulfate reduction by nutrient augmentation or raising reactor temperature.

TECHNOLOGY	Microbial Reactor Systems MRS 4
PROJECT SCALE	Pilot scale
REFERENCE	"Microbially-Mediated Metal Removal from Acid Mine Drainage". Fyson, A., Kalin, M. and M. Smith. Boojum Research Ltd.
LOCATIONS	Seepage station at INCO Copper Cliff Tailings Area near Sudbury, Ontario.
DESCRIPTION	AMD passed through four test cells.
DESIGN CRITERIA	Test cells 100 m by 20 m wide. Four cells in total, two for iron precipitation in oxidizing conditions, and two for acidity and metal removal in reducing conditions using ARUM. Flow rate of 1 L/min. Retention time was 168 days for Cells 1 & 2, 131 days for Cells 3 & 4.
ADDITIVES	Flax straw added to Cells 3 & 4 in 1989. Floating cattail rafts added to Cells 3 & 4 in 1991 to decrease water circulation and oxygenation. Compressed alfalfa pellets added to 3 & 4 in 1992 as an organic carbon source.
COSTS	n.d.
TIME OF OPERATION / INFORMATION	Operated from 1989 through 1992. Paper discusses results over a 5 month period in 1992. Samples analyzed for July, August, and October 1992.
PERFORMANCE DATA	Mean Influent, May-October; Effluent, October 1992 (Flow = 0.1L/min) pH: 5.7; 6.14 Eh: 281 mV; 234 Conductivity: 2,753 mS/cm; -- Acidity: 625 mg/L; 23.4 g/day (162.5 mg/L) Al: < 1 mg/L; 0.14 g/day (0.97 mg/L) Cu: < 1 mg/L; 0.001 g/day (0.007 mg/L) Fe: 256 mg/L; 3.18 g/day (22.08 mg/L) Ni: 29.8 mg/L; 0.46 g/day (3.18 mg/L) S: 844 mg/L; 60.1 g/day (417.4 mg/L)
OVERALL EFFECTIVENESS	Test cells removed > 88% of inflowing Fe, 77% of Ni, 39% of S and reduced acidity by 72%.
TOXICITY	n.d.
MAINTENANCE REQUIREMENTS	Flax straw added to test cells in 1989, alfalfa pellets in 1992 as organic amendments. Macrophyte cover (cattails) added in 1992.
MONITORING REQUIREMENTS	n.d.
TREATMENT PRODUCT DISPOSAL	n.d.
ENVIRONMENTAL CONDITIONS	n.d.
COMMENTS	

<table>
<tr><td>TECHNOLOGY</td><td>Microbial Reactor Systems MRS 5</td></tr>
<tr><td>PROJECT SCALE</td><td>Lab scale</td></tr>
<tr><td>REFERENCE</td><td>"In Situ Treatment of Acid Mine Drainage by Sulphate Reducing Bacteria in Open Pits: Scale-Up Experiences". Kuyucak, N. and P. St-Germain. 1994. International Land Reclamation and Mine Drainage Conference.</td></tr>
<tr><td>LOCATIONS</td><td>N/A</td></tr>
<tr><td>DESCRIPTION</td><td>Investigation of feasibility of treatment of AMD using SRB processes in various batch and continuous mode reactors. Evaluation of different nutrient sources, substrates, temperature, and inoculum concentrations was also performed.</td></tr>
<tr><td>DESIGN CRITERIA</td><td>Batch reactors were 2 L in volume, and experiments performed for 35-day periods. Larger reactors (5-L, 280-L drum, and a 5-m high, 160-L column) were operated in continuous mode. Reactors consisted of a limestone sediment inoculated with a 1% v/v sulphate reducing bacteria (SRB), with nutrient layered at the bottom of the reactor.</td></tr>
<tr><td>ADDITIVES</td><td>Two main nutrient sources tested: cellulosic (wood pulp, sawdust, bark, maple leaves, oat straw, fuel peat and horticultural peat); and organic wastes (cow manure, distillers dried grains, and brewer's dried grains). Control tests used chemical nutrients, including lactic acid, $(NH_4)_2SO_4$, and K_2HPO_4. Moderate strength (F-group) AMD was used.</td></tr>
<tr><td>COSTS</td><td>Based on the sulfate reduction rate found, and assuming that AMD contains 20 mol/d of metals, that a ton of substrate costs $40 and generation rate of H_2S of 0.15 mol/m^3/d; the amount of substrate required would be 40,000 tonnes at a cost of $1.6 million (Cdn). For a pit 5,000 m^2 in area, the substrate layer would be 8 m deep. Theoretical life would be 120 years, but likely less than this reduced as the efficiency would be uncertain due to its thickness and the mass transfer limitations due to lower temperatures.</td></tr>
<tr><td>TIME OF OPERATION / INFORMATION</td><td>Small scale reactors operated for 35 days. Large scale reactors operated for 14 months.</td></tr>
<tr><td>PERFORMANCE DATA</td><td>
<table>
<tr><th></th><th>Influent (Average)</th><th>Range for Effluent (over 14 months for drum reactor)</th></tr>
<tr><td>pH</td><td>2.5</td><td>4.8 - 6.8</td></tr>
<tr><td>Al</td><td>173 (mg/L)</td><td>nd - 1.8</td></tr>
<tr><td>As</td><td>1.0</td><td>nd - 0.09</td></tr>
<tr><td>Cd</td><td>1.5</td><td>nd - 0.04</td></tr>
<tr><td>Cu</td><td>47</td><td>nd - 0.17</td></tr>
<tr><td>Fe</td><td>160</td><td>0.13 - 8.99</td></tr>
<tr><td>SO_4</td><td>4,000</td><td>1682 - 3720</td></tr>
<tr><td>Mn</td><td>38</td><td>1.11 - 26.0</td></tr>
<tr><td>Zn</td><td>350</td><td>nd - 11.4</td></tr>
</table>
</td></tr>
<tr><td>OVERALL EFFECTIVENESS</td><td>As, Cd, and Cu reduced to near undetectable levels under all circumstances. Al, Fe, and Zn fluctuated between nd and 10-20 mg/L. Reduction of Mn poor throughout experiment.</td></tr>
<tr><td>TOXICITY</td><td>n.d.</td></tr>
<tr><td>MAINTENANCE REQUIREMENTS</td><td>n.d.</td></tr>
<tr><td>MONITORING REQUIREMENTS</td><td>n.d.</td></tr>
<tr><td>TREATMENT PRODUCT DISPOSAL</td><td>n.d.</td></tr>
<tr><td>ENVIRONMENTAL CONDITIONS</td><td>43 % decrease in SRB activity observed in large scale reactors when temperature was reduced from 20 to 10EC. Improved activity was observed when contact between nutrients and AMD water was performed by means of resuspension of the nutrient bed. Method of influent addition was found to effect fluctuations in pH, implying better contact between nutrients, influent and SRB results in more constant results.</td></tr>
<tr><td>COMMENTS</td><td>Loadings associated with open pits are too high for passive biological systems but the concept is viable for low load situations. Further research into inexpensive nutrients and overcoming effects of low temperatures needs to be done.</td></tr>
</table>

TECHNOLOGY	Microbial Reactor Systems MRS 6
PROJECT SCALE	Pilot scale
REFERENCE	"The Use of Bacterial Sulfate Reduction in the Treatment of Drainage from Coal Mines". McIntire, P.E. and H.M. Edeborn. 1990. Proceedings from the 1990 Mining and Reclamation Conference and Exhibition.
LOCATIONS	Friendship Hill National Historic Site, Fayette County, Pennsylvania.
DESCRIPTION	Pilot scale experimental wetlands system designed to treat stream discharge from an abandoned drift mine.
DESIGN CRITERIA	Wetland consisted of 3 cells. Cell 1 was a holding pond with peat and hay bales as substrate. Cells 2 and 3 contained 3 divided treatment lanes, separated by Fibreglass sheeting. AMD flows across the surface of the organic substrate of Cell 2. Subsurface infusion pipes were implemented in Cell 3 to allow the option of carrying AMD to the bottom of the substrate. The 6 treatment lanes were each constructed of 15 cm of gravel at the base (crushed limestone in 4 of the 6, noncalcareous river gravel in the remaining 2), followed by 46 cm of mushroom compost. Cattail plants transplanted to all cells.
ADDITIVES	n.d.
COSTS	n.d.
TIME OF OPERATION / INFORMATION	Constructed in 1988 and operated continuous through to August 1989.
PERFORMANCE DATA	Influent water has a pH of approximately 2.5, total Fe concentration of 50 - 250 ppm, and sulfate concentration of 1000 - 2500 ppm. During course of experiment, outflow water pH increased to as high as 6.5, alkalinity increased up to 500 ppm, and total Fe dropped to as low as 20 ppm (observed during use of subsurface infusion pipes in Cell 3).
OVERALL EFFECTIVENESS	Study showed that increasing contact between substrate and AMD (i.e. use of infusion pipes) improved water quality. Little improvement in water pH or alkalinity was observed when AMD was allowed to flow predominantly across the surface of the substrate. No significant acid neutralization advantage was found in using limestone gravel.
TOXICITY	n.d.
MAINTENANCE REQUIREMENTS	Subsurface infusion pipes were found to become clogged with iron oxyhydroxide (FeOOH) floc and organic debris. Armouring of gravel with FeOOH was found to occur. An alternative method of introducing AMD to the anaerobic regions of the wetland was devised through the use of plexiglass dam sheeting.
MONITORING REQUIREMENTS	Surface water samples collected from 9 locations biweekly for eight months.
TREATMENT PRODUCT DISPOSAL	n.d.
ENVIRONMENTAL CONDITIONS	Sulfate reduction rates remained consistently low during the winter months of the experiment.
COMMENTS	Preliminary results indicate that bacterial sulfate reduction rates were high enough to significantly affect water quality of AMD. Observed improvement in water quality was due to both the inherent chemical characteristics of the substrate used (i.e., alkalinity of compost), and the bacterial activity.

TECHNOLOGY	Microbial Reactor Systems MRS 7
PROJECT SCALE	Pilot Scale
REFERENCE	"Biological Metal Removal from Mine Drainage", K. Nakamura. 1988. U.S. Bureau of Mines Circular 9183.
LOCATIONS	Yanahara Mine, Okayama Prefecture, Japan.
DESCRIPTION	Treatment of AMD through a fluidized bed-type anaerobic reactor.
DESIGN CRITERIA	Working volume of reactor was 180 L. Raw AMD drainage fed through a 3 m^3 powdered limestone neutralizing tank to bring to pH 5 prior to entry to the reactor vessel. Bacterial culture used was a strain of Desulfovibrio vulgaris. Reactor also fortified with nutrient solution and coagulant.
ADDITIVES	Nutrient solution consisted of yeast extract, sodium lactate and inorganics such as K_2HPO_4, NH_4Cl, Na_2SO4, and $MgSO_4$.
COSTS	Conclusion reached that nutrients used in process were too expensive for full scale system.
TIME OF OPERATION / INFORMATION	Pilot plant was run for 113 days continuously. Feed volume of effluent was increased in steps from 180, 360, and finally to 720 L/day.
PERFORMANCE DATA	Influent (mg/L) pH 2.5 - 2.65 Fe^{2+} 171-303 Total Fe 693-815 Zn 41.5 - 52.1 Acidity 1991 - 2442 Cu 9.4 - 16.8 Cd 0.18 Mn 7.4 As 0.21 Effluent (mg/L) Fe^{2+} < 1.0 Zn < 0.2
OVERALL EFFECTIVENESS	AMD treated successfully (i.e. to below Japanese standards) at flow of 720 L/day. COD level of reactor effluent found to be high, 800 - 1000 mg/L
TOXICITY	n.d.
MAINTENANCE REQUIREMENTS	n.d.
MONITORING REQUIREMENTS	n.d.
TREATMENT PRODUCT DISPOSAL	n.d.
ENVIRONMENTAL CONDITIONS	Reactor temperature, operated at between 9.0 and 30.2 EC, dependant on weather.
COMMENTS	

TECHNOLOGY	Microbial Reactor Systems MRS 8
PROJECT SCALE	Lab scale
REFERENCE	"Determination of the Rate of Sulfide Production in a Constructed Wetland Receiving Acid Mine Drainage". Reynolds, J.S., Machemer, S.D., Wildeman, T.R., Updegraff, D.M. and R.R Cohen. 1991. Proceedings from 1991 National Meeting of the American Society for Surface Mining Reclamation.
LOCATIONS	Water samples collected from the Big Five Pilot Wetlands in Idaho Springs, Colorado.
DESCRIPTION	Lab scale system was designed to determine an approximate in situ rate of sulfide production by sulfide-reducing bacteria by measuring the change in acid volatile sulfides in substrate over time.
DESIGN CRITERIA	A 2-litre sample of substrate was collected from the wetland containing mushroom compost and 12 litres of mine drainage from the Big Five Tunnel adit.
ADDITIVES	Serum nutrient amendments consisted of sodium lactate or 30 mL of an extract of hay. Control serum samples also run, poisoned with sodium azide.
COSTS	n.d.
TIME OF OPERATION / INFORMATION	n.d.
PERFORMANCE DATA	n.d.
OVERALL EFFECTIVENESS	Summer in situ rate of sulfide production from the Big Five Wetland was estimated to be 1.2 µumol/g/day.
TOXICITY	n.d.
MAINTENANCE REQUIREMENTS	n.d.
MONITORING REQUIREMENTS	n.d.
TREATMENT PRODUCT DISPOSAL	n.d.
ENVIRONMENTAL CONDITIONS	Serum bottles operated at 70 FE to simulate summer conditions.
COMMENTS	Minimal sulfide production in poisoned control samples indicates that process is predominantly microbiological. Results indicate that addition of lactate or hay significantly increases sulfate production. Trends observed were similar to those in output waters of the wetland, indicating that serum bottle systems could be used in initial treatability studies.

TECHNOLOGY	Microbial Reactor Systems MRS 9
PROJECT SCALE	Pilot scale
REFERENCE	"A Passive Treatment System as a Bioreactor: Treatment Efficiency, pH Increase, and Sulfate Reduction in Two Parallel Reactors". Staub, M.W. and R.R.H. Cohen. 1992. In: Achieving Land Use Potential Through Reclamation, Duluth Minnesota.
LOCATIONS	Eagle Mine Superfund Site, 100 miles west of Denver, Colorado.
DESCRIPTION	AMD was treated through two pilot scale bioreactor tank systems using cow manure and hay as substrates.
DESIGN CRITERIA	AMD waters collected at source and diverted through two parallel bioreactor treatment systems - i) a single stage (SS) reactor consisting of a 500 gallon high-density polyethylene cylindrical tank with hay and cow manure/hay substrate layers, and a landscape fabric/pea gravel bed; and ii) a double stage (DS) reactor consisting of 2x200 galllon HDP tanks, with the first stage containing substrate and the second stage a fabric/gravel bed only. Flows through systems initialized at 50 mL/min based on a loading rate of 0.125 gal/min/100 ft^2, doubled to 100 mL/min after five weeks operation, 200 mL/min after 11 weeks, and 400 mL/min by 16 weeks.
ADDITIVES	Each reactor system was inoculated with 15 pounds of substrate from the Big Five constructed wetlands in Idaho Springs, Colorado. Manure/hay substrate layer was prepared in a 4:1 volumetric ratio.
COSTS	n.d.
TIME OF OPERATION / INFORMATION	July 25/91 - November 26/91. Stopped due to freezing.

PERFORMANCE DATA

	Influent	Effluent - SS	Effluent - DS (2nd tank)
pH	2.6 - 3.0	5.5 - 7.9 (final)	6.5 - 6.9 (final)
Eh	650-750 mV	Slightly +ve at 200mL/min	Slightly +ve at 200mL/min
Cond. (mmhos)	2,400 - 5,070	Increased over time	Increased over time
Cu	15 ppm	94 - 100% removal	92 - 100% removal
Cd	0.91 ppm	71- 100 %	83 - 100 %
Pb	0.7 ppm	47 - 100 %	53 - 100 %
Fe	270.0 ppm	92 - 100 %	81 - 99 %
Mn	200.0 ppm	32 - 100 %	55 - 99 %
Zn	200.0 ppm	45 - 100 %	79 - 100 %
Sulfate	4,400 ppm	0 - 43 %	0 - 28 %

OVERALL EFFECTIVENESS	Treatment successful at all flow rates. Increase in treatment continued gradually through substrate level, achieving increased pH, decreased Eh over time, and 90 - 100 % removal of nearly all metals. Conductivity found to increase for both reactors, explained by leaching of organic ion groups from substrate. Sulfate concentration not reduced (actually increased) at 50 mL/min flow rate in effluent off SS and DS.
TOXICITY	n.d.
MAINTENANCE REQUIREMENTS	n.d.
MONITORING REQUIREMENTS	Samples collected weekly from each system via three wells built in reactor tanks located at various depths of substrate (10", 20" and 30"). Flow rate, temperature, pH, Eh, and conductivity measured in the field. Samples analyzed, filtered and analyzed for Cu, Cd, Pb, Fe, Mn, Zn, and sulfate.
TREATMENT PRODUCT DISPOSAL	n.d.
ENVIRONMENTAL CONDITIONS	Mitigation of AMD successfully occurred at temperatures from 2 to 9 °C. Ceased after freeze-up.
COMMENTS	Experiments suggest that effective treatment can be obtained with higher flow rates per given area, indicating that large scale plants may be built as towers rather than shallow broad wetlands.

TECHNOLOGY	Microbial Reactor Systems MRS 10
PROJECT SCALE	Pilot scale
REFERENCE	"Constructed Wetlands that Emphasize Sulfate Reduction: A Staged Design Process and Operation in Cold Climates". Wildeman, T.R., et al. 1992. Proceedings from the 24th Annual Meeting of Canadian Mineral Processors.
LOCATIONS	Big Five Pilot Wetlands in Idaho Springs, Colorado.
DESCRIPTION	AMD passed through anaerobic wetland cells. Wetland in operation since 1987.
DESIGN CRITERIA	Initial three cells (A, B, and C) 18.6 m^2 in area. Cell B remodelled in 1989 to allow for upflow or downflow configurations based on a trickling flow filter process.
ADDITIVES	Substrates for Cell A is mushroom compost; B, C, D, and E consist of equal parts peat, aged manure, and decomposed wood.
COSTS	n.d.
TIME OF OPERATION / INFORMATION	Cells operated year round from 1988-1992.
PERFORMANCE DATA	Big Five Drainage (Concentrations in mg/L, %R= % reduction). Flow in mL/min.

Initial Operation - December '87

	Mn	%R	Fe	%R	Zn	%R	Cu	%R	pH	Flow Rate	
Drainage	34		32		10.6		1.02			2.8	
Cell A	27	21	18	45	7.8	27	0.44		57	4.6	1.0
Cell B	33	1	24	26	9.8	12	0.89		12	3.1	1.0
Cell C	34	0	22	32	9.6	9	0.91		10	3.3	1.0

August '88

	Mn	%R	Fe	%R	Zn	%R	Cu	%R	pH	Flow Rate	
Drainage	26		37		8.1		0.91			2.9	
Cell A	25	4	20	56	<0.1	100	0.17	81	5.5	0.51	
Cell B	26	0	15	59	6.1	24	0.55		40	3.2	0.24
Cell C	25	4	11	70	5.8	28	0.38		58	3.5	0.34

CELL A MODIFIED February '89

	Mn	%R	Fe	%R	Zn	%R	Cu	%R	pH	Flow Rate	
Drainage	27		32		9.3		0.56			3.0	
Cell A	22	19	12	63	4.5	52	<0.05	100	5.1	0.28	
Cell B	27	0	28	13	6.1	34	0.82		0	3.4	0.31
Cell C	25	7	31	3	7.2	23	0.26		53	3.5	0.32

May '89

	Mn	%R	Fe	%R	Zn	%R	Cu	%R	pH	Flow Rate	
Drainage	30		42		10.4		0.76			3.0	
Cell A	33	-10	28	33	7.8	25	<0.05	100	3.5	0.92	
Cell B	27	10	10	76	6.2	40	0.36		53	3.0	0.83
Cell C	29	3	9	79	6.4	38	0.46		39	3.2	0.81

August '89

	Mn	%R	Fe	%R	Zn	%R	Cu	%R	pH	Flow Rate	
Drainage	32		43		9.4		0.75				2.9
Cell A	31	3	39	9	5.2	45	<0.05	100	4.1	0.30	
Cell B	26	19	24	44	6.6	30	0.46		39	3.1	0.48
Cell C	32	0	18	58	4.8	48	0.09		88	3.7	0.43

TECHNOLOGY

Microbial Reactor Systems MRS 10

PERFORMANCE DATA (CONT)

CELL B MODIFIED October '89

	Mn	%R	Fe	%R	Zn	%R	Cu	%R	pH	Flow Rate
Drainage	35		46		9.9		0.66		3.2	
Cell B (Up)	34	3	39	15	9.3	6	0.59	11	3.5	0.29
Cell B (Down)	23	34	8	83	0.8	92	<0.05	100	6.5	0.16
Cell E	24	31	<1	100	<0.1	100	<0.05	100	6.3	0.062

November '89

	Mn	%R	Fe	%R	Zn	%R	Cu	%R	pH	Flow Rate
Drainage	32		38		8.7		0.61		2.9	
Cell B-U	31	3	17	55	8.4	3	0.48	21	3.6	0.22
Cell B-D	20	38	<1	100	0.8	92	<0.05	100	6.5	0.16
Cell E	24	31	<1	100	<0.1	100	<0.05	100	6.3	0.062

January '90

	Mn	%R	Fe	%R	Zn	%R	Cu	%R	pH	Flow Rate
Drainage	31		33		9.0		0.59		2.9	
Cell B-U	30	3	33	0	8.9	1	0.49	17	3.2	0.21
Cell B-D	27	13	12	64	8.1	10	0.44	25	3.2	0.20
Cell E	24	31	<1	100	<0.1	100	<0.05	100	6.0	0.10

OVERALL EFFECTIVENESS

Total heavy metals concentration reduced by 99%, pH raised from 2.5 to average of above 7. Since inception of Cell E removal of Cu, Zn, and Fe has been 100%, with Mn removal averaging 25%

TOXICITY

n.d.

MAINTENANCE REQUIREMENTS

n.d.

MONITORING REQUIREMENTS

n.d.

TREATMENT PRODUCT DISPOSAL

n.d.

ENVIRONMENTAL CONDITIONS

Cells operated during winter months were found to operate effectively provided that cells are adequately insulated, pathways are designed to allow flow away from freezing areas, and if possible, subsurface flow systems are incorporated to minimize exposure to the elements.

COMMENTS

Sulfate reduction and subsequent precipitation of metal sulfides by anaerobic processes appears to be primary mechanism for AMD reduction in wetlands and that plants are not necessary for subsurface processes. The "Limiting Reagent Concept" suggests that loading factors should be set to insure that heavy metal concentrations are the limiting factor to ensue healthy microbial populations. Permeability of the substrate was also determined to be a critical design factor.

BIOSORPTION SYSTEMS
PHASE I CASE STUDIES

TECHNOLOGY	Biosorption Systems BS 1
PROJECT SCALE	Pilot scale
REFERENCE	"Passive Treatment Methods For Manganese: Preliminary Results from Two Pilot Sites". Wildemann, T.R., et al. 1993. National Meeting of the American Society for Surface Mining and Reclamation.
LOCATIONS	Fabius Coal Mines/Hard Rock Constructed Wetlands, Jackson, Alabama; and Boston Mine, Durango, Colorado.
DESCRIPTION	1) Fabius: Treatment of effluent from an oxidation and settling pond (the Hard Rock Constructed Wetlands); with focus on manganese removal; using a cyanobacteria-algal mat (CGM). 2) Boston: anaerobic reactor/wetland for treatment of AMD.
DESIGN CRITERIA	1) CGM system: Pilot pond area 46 m^2 in size with initial influent flow rate of 1 L/min. Influent water from the wetlands system enters a trickling filter to remove suspended $Fe(OH)_3$ and then leaves from one pipe to three ponds. All ponds constructed with 4-foot wide rock baffles separated by one-foot wide troughs. Average depth of water in troughs is 30 cm. Two ponds contained 2.5 cm of high-calcium limestone and the third contained 1 cm pea gravel. Ponds built on pad 1 foot depth lined with 10 mil polyethylene. Flow controlled by valves. 2) Anaerobic system: Area 45m by 10m. Reactor 1.1m deep. Substrate consisted of mushroom compost with white wheat straw, cotton seed meal, poultry waste and lime (soil pH of 7.5) Limestone fines mixed with compost in ratio of 2:8. 10% of compost was fresh to ensure SRB activity. Liner for reactor made of compacted clay. Collection trenches incorporated into seeps due to slope stability concerns. trenches connected to an inflow gallery made of perforated plastic pipe and extending length of system. 10 cm thick gravel bed installed beneath substrate to promote drainage. Landscape fabric (TYPAR) used to keep fines from clogging system. A 20 cm layer of hay was placed on top of the reactor for insulation. Gravel-lined spillway trench designed downstream of reactor to polish water and promote algal/bacterial oxidation of Mn and Fe.
ADDITIVES	1) CGM system: microbial strains (Oscillatoria spp., green filamentous algae, and Chromatium spp.) were harvested from the Hard Rock wetlands and developed in the lab into silage-microbe mats. 2) Anaerobic reactor: n.d.
COSTS	n.d.
TIME OF OPERATION / INFORMATION	Flow through CGM system initiated July 1992 through March 1993. Flow started at 1 L/min and increased to 3.3 L/min in October. Anaerobic system at Durango started September 1992 and ran through the winter. Initial flow of 40 L/min through system. Measurements shown for samples collected September-November.

PERFORMANCE DATA		CGM system (flow of 3.3 L/min) (October 1992)		Anaerobic Reactor	
		Influent	Effluent	Influent	Effluent
	Diss. O_2	5.7	16.0	-	-
	pH	6.6	7.7	2.6-2.9	3.2-3.8
	Eh (mV)	430	440	720-730	540-600
	Cond.(uS/cm)	703	775	8500-10830	9240-12200
	Mn (mg/L)	9.7	0.1	18-25	N.A.
	Fe	11.9	0.4	130-480	N.A.
	No field information available on Boston Mine wetland.				

OVERALL EFFECTIVENESS	CGM mat found effective in reducing Mn concentrations. At high flow rate, removal capacity to achieve below a Mn concentration of 2 mg/L is 2.5 gram/m^2/day.
TOXICITY	n.d.
MAINTENANCE REQUIREMENTS	Some question of whether CGM system could be sustaining during winter months. CGM system may also need clean-out of precipitate and organic debris. Anaerobic reactor was found to clog, thus reducing water flow. No indication of whether anaerobic system would avoid freezing during winter months due to sever winter weather - influent water has considerable thermal activity, however.
MONITORING REQUIREMENTS	n.d.
TREATMENT PRODUCT DISPOSAL	n.d.
ENVIRONMENTAL CONDITIONS	n.d.
COMMENTS	Preliminary results for both systems. See follow-up study (MRS 11). Cyanobacteria algal mat found effective in Mn removal.

TECHNOLOGY	Biosorption Systems BS 2
PROJECT SCALE	Pilot scale

TECHNOLOGY	Biosorption Systems BS 2
REFERENCE	"Manganese and Iron Removal from Coal Mine Drainage by Use of a Green Algae-Microbial Mat Consortium". Phillips, P., et al. 1994. International Land Reclamation and Mine Drainage Conference on the Abatement of Acidic Drainage. pp. 99 - 108.
LOCATIONS	Fabius Coal Mines/Hard Rock Constructed Wetlands, Jackson, Alabama.
DESCRIPTION	See BS 1. Continuation of 1992 study using cyanobacteria algal mat to remove Mn. Project designed to examine feasibility of applying mats to remove Mn and Fe from AMD. Study also to evaluate performance of mats under various environmental conditions.
DESIGN CRITERIA	See BS 1. Approximately 6 weeks required to achieve a full mat cover.
ADDITIVES	See BS 1.
COSTS	n.d.
TIME OF OPERATION / INFORMATION	See MRS 10. Flow adjusted on a daily basis due to iron hydroxide clogging. Average flows were 2.8, 3.8, and 4.2 L/min in the three ponds (Pea gravel substrate (control) pond, Limestone substrate (control) pond, and Cyanobacteria algae (active) pond, respectively).
PERFORMANCE DATA	For a 4.2 L/min flow, in winter months, Mn removal rates for the CGM pond ranged from 2.59 g/day/m^2 at 2m downstream, 3.87 g/d/m^2 at 5m, and 0.74 g/d/m^2 at 8m. Removal rats by the two control ponds were 1.5 to 8 times lower than those attained at similar downstream points. Dried algal mat sample Mn concentrations decreased from 2.67 mg/g at 1 m from influent point to 0.45 mg/g at near effluent point.
OVERALL EFFECTIVENESS	CGM pond found to remove Mn and Fe by 83% and 98%, respectively (based on mat sample concentrations).
TOXICITY	n.d.
MAINTENANCE REQUIREMENTS	n.d.
MONITORING REQUIREMENTS	n.d..
TREATMENT PRODUCT DISPOSAL	n.d.
ENVIRONMENTAL CONDITIONS	Winter water temperature was 5 EC. Through the 1993 winter the algal mat remained viable. The mat was significantly damaged after a large snowfall thaw in March 1993. In general, winter-summer and day-night metal removal rates for the CGM pond were found to be essentially the same.
COMMENTS	Mn and Fe removal shown to be effectively removed by use of a cyanobacteria algal mat.

TECHNOLOGY	Biosorption Systems BS 3
PROJECT SCALE	Bench scale
REFERENCE	"The Aerobic Removal of Manganese from Mine Drainage by an Algal Mixture Containing Cladophora". Duggan L.A., et al. 1992. National Meeting of the American Society for Surface Mining and Reclamation.
LOCATIONS	N/A
DESCRIPTION	Bench scale study of determining Mn removal rate using Cladaphora algae and test its resistance to high Mn concentrations and severe weather conditions.
DESIGN CRITERIA	Two reservoirs designed from small plastic swimming pools, 1.1 m in diameter. Each pool contained 97 L of effluent from the Big Five Wetland and 5 L of pond scum comprised of Cladophora. One reservoir contained 12 kg of limestone (designated as pond LS, the other pond as NoLS). Pools allowed to stand outside for duration of experiment.
ADDITIVES	Water occasionally added to account for evaporation.
COSTS	n.d.
TIME OF OPERATION / INFORMATION	Experiment ran for 4 months from August to December. Reservoirs static for first two months of experiment. A flow system was installed the last two months of the experiment.
PERFORMANCE DATA	Big Five effluent contained 32 mg/L Mn and had a pH of 5.8. Under static conditions, LS pond contained 5.4 mg/L and had a pH of 8.8 by day 6 of experiment. Similarly, the NoLS pond had 14 mg/L of Mn and a pH of 8.6 by day 6. On Day 20 a dose of 40 L of 100mg/L Mn, pH 4.9, was added to the ponds. By Day 25, the NoLS pond had 51 mg/L Mn and a pH of 7.6; the LS pond had a pH of 8.1 and 13.1 mg/L Mn. For the pump flow conditions, the mine drainage effluent used had a pH of 6.3 and Mn concentration of 23 mg/L. Flow rates ranged from 2.1-4.5 mL/min for the NoLS pond and 1.5- 4.5 mL/min for the LS pond over the 56 days. Mn was added to both systems at a concentration of 100 mg/L for 5 days during this period and averaged 28-65 mg/L for the duration. Both ponds showed consistent removal to below 5 mg/L, with 85% of samples collected below 2 mg/L.
OVERALL EFFECTIVENESS	The LS pond generally showed higher removal efficiencies than the NoLS pond (0.19 g/d/m^2 vs. 0.15 g/d/m^2) for both static and flow conditions. Both reservoirs showed tolerance to high Mn concentrations and were not drastically affected by severe winter weather conditions.
TOXICITY	n.d.
MAINTENANCE REQUIREMENTS	n.d.
MONITORING REQUIREMENTS	n.d..
TREATMENT PRODUCT DISPOSAL	n.d.
ENVIRONMENTAL CONDITIONS	Temperatures during experiment dropped to as low as freezing. Severe weather conditions had a visible effect on the biomass of algae but did not affect removal efficiency.
COMMENTS	Photosynthesis by algae appears to play an important role in contributing to Mn removal due to its increasing pH; thus making Mn precipitation more amenable. Cladophora displayed a high tolerance to Mn concentration and to severe weather conditions. Concluded that an aerobic wetland utilizing Cladophora and limestone would have an estimated Mn removal rate of 0.17gdm under adverse conditions and possibly higher under warmer sunnier conditions.

MICROBIAL REACTOR SYSTEMS
PHASE II CASE STUDIES

TECHNOLOGY

Microbial Reactor Systems MRS 11

PROJECT SCALE

Pilot scale

REFERENCE

"Microbial Treatment of Acid Mine Drainage at Halifax International Airport". Béchard, G., Koren, D.W., Rajan, S. and R.G.L. McCready. 1995. CANMET, Mineral Science Laboratories. Division Report MSL 95-3 (OP&J).

LOCATIONS

Halifax International Airport, Nova Scotia, Canada.

DESCRIPTION

The test site was on Halifax International Airport property near a ditch which receives drainage from runways 24 and 33. Three clay lined cells were built in sequence parallel to the ditch.

DESIGN CRITERIA

The cells were designed to be narrow, shallow and long to prevent channelling. Each cell was 16.8 m long by 1 m wide, by almost 1 m deep. The cells were separated from each other by a clay wall. Water flowed from one cell to the other through three plastic pipes of 150 mm diameter and was returned to the ditch downstream. The water capacity of the cells was expected to be 23 L/min.; excess water was designed to flow into the spillway channel.

ADDITIVES

All cells were filled with fresh planer shavings and partially decomposed straw in a proportion of 1:3; a total of 54 kg of sucrose (Lantic sugar), 6 kg of urea fertilizer (CIL 46-0-0) and 1 kg of phosphate rock were distributed among the cells. The phosphate rock contained fluorapatite, $Ca_5(PO_4)_3F$, $Na_3Ca_2(SO_4)_3F$ and quartz, SiO_2. Used straw and wood shavings from the laboratory reactor, which contained a variety of microorganisms including sulphate reducing bacteria (SRB), were incorporated into the cells as a bacterial inoculum.

To promote initial growth of microorganisms throughout the cells during the fall of 1989, 40 kg of Lantic sugar and 4 kg of urea fertilizer were added weekly from Day 6 to Day 113 to the acidic water as it entered the first cell. In efforts to promote microbial activity after treatment activities had ceased, the following additions were made: sugar and urea as nutrient additives to acidic water entering the first cell (weekly, Day 296-326 and 388-498; 687; 744; 749); straw as nutrients and swamp mud as a source of microorganisms into the cells (Day 339); used straw from the laboratory reactor as a source of microorganisms to the cells (Day 348); fresh straw as nutrients to the top of the three cells (Day 395); and ten bales of alfalfa hay as nutrients into the three cells (Day 764).

COSTS

n.d.

TIME OF OPERATION / INFORMATION

881 days beginning on August 8, 1989.

PERFORMANCE DATA

All values taken from graphs presented in the paper for the first 500 days of monitoring.

Influent					Effluent				
Parameter	Max.	Day	Min.	Day	Parameter	Max.	Day	Min.	Day
pH	5	100	2.5	480	pH	6.75	150	3	450
Al (mM)	1.8	7	0	365	Al (mM)	2.25	375	0	275
Fe (mM)	1.5	25	0.12	500	Fe (mM)	2.25	380	0	80
Ammonium*	1.5	375	0	90	Ammonium*	2.75	140	0	110
nitrate + nitrite (mg/L)	9	280	0	150	nitrate + nitrite (mg/L)	3	15	0	150

Parameter	Max.	Day	Min.	Day
Flow Rate (L/min)	0	320	24	450
Temperature (°C)	24	360	0	110

Parameter	Sampling Point			
	1	2	3	4
pH	4	5	5.75	6.25
Acidity (mg/L)	225	190	140	50
aluminum (mM)	1	0.5	0.25	0.1
Iron (mM)	0.6	0.75	0.6	0.25
Sulphate (mM)	3.75	3.25	2.5	1.75

* = mM

OVERALL EFFECTIVENESS

The treatment process effectively treated acidic drainage at temperatures as low as 0°C, indicating that the cold Canadian climate does not preclude the use of microorganisms to treat AMD in the field.

A minimum depth of submerged substrate was shown to be required: anaerobiosis was never established within substrate only 3.2 cm deep, even with nutrient additives. The cellulosic material which was used to fill the cells was partially decomposed and clumped promoting channelling flow. Channelling could be minimized, at least initially, if the cellulosic material was blown into the cells using equipment which could break down the clumps and allow for a more regular layering of the material.

The cells did generate ammonium, which should be monitored in environments where this parameter is regulated.

Iron was consistently removed, however not always sufficiently to meet effluent objectives, possibly because iron incompletely reacted with biogenic sulfide. An additional step where reduction of ferric iron to ferrous iron is required for all iron to be removed as a metal sulfide.

TOXICITY

n.d.

TECHNOLOGY	Microbial Reactor Systems MRS 11
MAINTENANCE REQUIREMENTS	After the first winter, local collapsing of the clay liner was observed and some reconstruction work was required. Flow pipes were cleaned of ferric iron precipitates to prevent plugging (Day 348, 617, 687, 708 and 798). Ferric iron precipitates which accumulated in the pond upstream of the first cell were dredged on Day 703. Screens on the pipes at the inlet of the first cell were replaced on Day 708.
MONITORING REQUIREMENTS	Monitored weekly for an 881-day period, except during winter months when it was covered by ice and snow (Days 113-155; 163-252; 498-617). Field monitoring included water temperature, flow rate, pH. Water samples were taken directly upstream of cell #1, at the downstream end of cell #1 and cell #2 and downstream a v-notch located after the final cell, before the effluent reenters the original bed of the stream. Samples were laboratory tested for pH, Al, Fe, total acidity, sulphate, ammonium, oxidized nitrogen (nitrate + nitrite), phosphate, total dissolved solids and total organic carbon. For microbial analyses, biomass samples were taken once a month at locations **a** and **b** down the length of each cell. The following counts were done: total fungal, total aerobic heterotrophic bacteria, and SRB.
TREATMENT PRODUCT DISPOSAL	No secondary pollution was generated by the field reactor.
ENVIRONMENTAL CONDITIONS	n.d.
COMMENTS	It is expected that in the field in a system operating at a steady-state, ammonium, the preferred nitrogen source for many microorganisms, would be consumed as soon as it is produced by the microbial communities. The need to clean pipes on a regular basis is inconvenient. The reactor is designed to provide both aerobic and anaerobic zones for microorganisms, so that the formation of ferric iron precipitates cannot be prevented.

TECHNOLOGY	Microbial Reactor Systems MRS 12
PROJECT SCALE	Laboratory Batch and Column Experiments, and Small Scale Field Study
REFERENCE	"*In-Situ* Treatment of Mine Drainage Water Using Porous Reactive Walls". Blowes, D.W., Ptacek, K.R., Waybrant, K.R., and J.G. Bain. 1995. In: Biotechnology and the Mining Environment, BIOMINET, 11th Annual Meeting Proceedings. Lortie, L., Gould, W.D. and S. Rajan (eds), pp. 103-115.
LOCATIONS	Nickel Rime mine site, Sudbury, Ontario.
DESCRIPTION	The study was designed to assess the long-term effectiveness of a range of different organic materials as determined from batch and column experiments, as well as the use of porous, permeable, geochemically-reactive walls which incorporate bacterially mediated sulphate reduction for acid mine drainage treatment. These geochemically engineered walls were installed in the pathway of flowing tailings-derived groundwater resulting in *in-situ* treatment.
DESIGN CRITERIA	Batch Experiments Batch experiments were conducted using poplar sawdust, leaf compost, and composed sheep manure as sources of organic carbon, and anaerobic creek sediment as a source of sulphate-reducing bacteria. Agricultural limestone was also added to ensure optimum pH conditions for the bacteria. The organic carbon sources, creek sediment, limestone, and simulated mine drainage water were placed in well-sealed, glass reaction flasks. Anaerobic conditions were maintained during the experiments. Column Experiments Three 40 cm x 5 cm columns were packed with reactive materials including wood chips, sawdust, composted municipal sewage sludge, leaf compost, and composted sheep manure. A continuous flow of simulated mine drainage water was pumped through the columns. Small-Scale Field Test A test cell measuring 1.2 m x 1 m x 1.2 m; (1,w,d) was installed 75 m from the base of the tailings dam, near the front of the Fe (II)$^{-}$ and SO_4^{-} rich plume. The reactive mixture added to the test cell contained composted leaf mulch, pine bark and wood chips as sources of organic carbon, agricultural limestone for pH buffering, and creek sediment as a source of sulphate-reducing bacteria. The reactive mixture was combined with course sand to maintain sufficient permeability.
ADDITIVES	Batch Experiments After a period of 1865 hours, the batch reactor was spiked with an input solution containing 3530 mg/L SO_4, 660 mg/L Fe and 0.9 mg/L Zn. Nitrogen gas was injected into the reaction flasks to prevent entrance of O_2 while aliquot samples were collected. Column Experiments In August 1994, the source water was changed to a solution containing approximately 3000 mg/L SO_4 and 800 mg/L Fe. The solution also contained low concentrations of Mn, Ni, Pb and Zn.
COSTS	n.d.
TIME OF OPERATION / INFORMATION	Batch Experiments Approximately 4200 hours. Column Experiments The preliminary column experiment was initiated in August 1993, additional column experiments were initiated in March, 1994. Small-Scale Field Trail First cell was installed in October, 1993. In September, 1994 two additional cells were installed 125 m down gradient of the tailings dam. Water in the initial test cell was sampled 7 months after the cell was installed.
PERFORMANCE DATA	Batch Experiments Sulphate reducing conditions developed rapidly (<500 hrs) after the initial acclimation period as indicated by a 50% decline in SO_4 concentrations from >1500 mg/L to <700 mg/L. The removal of metals (Fe and Zn) from solution was also rapid. Throughout the experiment the pH remained near neutral (6.5 - 7.0). After the addition of the spike, removal of SO_4 and dissolved metals was rapid, indicating the potential for sustained sulphate-reduction reactions.

TECHNOLOGY	Microbial Reactor Systems MRS 12
PERFORMANCE DATA (CONT.)	Column Experiments The columns were saturated with SO_4-rich water containing 500-1000 mg/L SO_4. Within 30-40 days a population of sulphate-reducing bacterial became acclimated in the column and reducing conditions were established. The sulphate-reduction reaction resulted in removal of SO_4 from an input concentration of 500-1000 mg/L to <20 mg/L. When the composition of the source water was changed, preliminary results indicated a partial removal of SO_4 with an increase in removal over time, and complete removal of Fe. Small-Scale Field Trail Water influent to the test cell had concentrations of Fe(II) of 500-900 mg/L; SO_4 of 2300-3600 mg/L; pH near 5; Eh near 450 mV; and a mean annual temperature of ~10°C. Alkalinity of the water, attributed to carbonate species, was less then 30 mg/L. The odour of H_2S was detected randomly throughout the aquifer, suggesting that indigenous sulphate-reducing bacteria are active. Analysis of groundwater samples collected from the cell indicated that metal attenuation was occurring. Sulphate reduction and dissolution of limestone added to the cell acted in increasing the pH to near neutral (6.3 to 7.1) and alkalinity to 1000 mg/L $CaCO_3^-$. Iron (II) concentrations decreased from 1000 mg/L to <50 mg/L after travelling <0.5 m through the cell. The lowest iron concentrations were 10 mg/L. Sulphate concentrations also decreased with travel distance in the cell, dropping from 3500 mg/L to <1000 mg/L over 0.5 m. Lowest SO_4 concentrations in the cell are ~5 mg/L. Dissolved metals such as Al, Ni, and Zn decreased to below drinking water quality guidelines. Hydrogen sulfide was present at 0 to >50 mg/L.
OVERALL EFFECTIVENESS	Batch Experiments All experiments indicated a potential for sulphate reduction and metal removal, provided sulphate was not limited. Column Experiments Sulphate reducing conditions were continued at experimental low-rates of 3-10 m/yr, and a dimension less velocity of 8-24 pour volumes/yr. When source water was changed, preliminary results indicated partial removal of SO_4 and complete removal of Fe. The removal of sulphate increased as the bacterial population apparently acclimated to the increased SO_4 concentrations. Small-Scale Field Trial Metal attenuation within the cell occurred, pH was near normal, and metals were reduced to below drinking water guidelines. The extent of metal atttenuation increased along the flowpath through the cell. The presence of H_2S, in addition to the low Eh in the cell, suggests that sulphate reduction is the dominant process leading to the removal of dissolved metals.
TOXICITY	n.d.
MAINTENANCE REQUIREMENTS	n.d.
MONITORING REQUIREMENTS	Batch Experiments Aliquot samples were analysed for SO_4, Fe, Mn, Zn, Ni, Ca, K, Na, Mg, COD, pH and Eh. Column Experiments Monitoring of influent and effluent Small-Scale Field Trial Water in the test cell was sampled 7 months after the cell was installed. Water influent and effluent samples were collected and analysed.
TREATMENT PRODUCT DISPOSAL	n.d.
ENVIRONMENTAL CONDITIONS	n.d.
COMMENTS	Results suggest that the use of porous reactive walls is a potential alternative for the prevention and remediation of acid mine drainage.

TECHNOLOGY Microbial Reactor Systems MRS 13

PROJECT SCALE Pilot Scale

REFERENCE "The Feasibility of Using Anaerobic Water Treatment at the Hardin Run Clay Mine". Dvorak, D.H. 1996. Report to Crescent Brick Company, New Cumberland, West Virginia.

LOCATIONS Hardin Run Clay Mine, Hancock County, West Virginia.

DESCRIPTION A 34 m^3 water treatment reactor was installed inside of the air course portal of the Hardin Run clay mine in order to determine the feasibility of using anaerobic neutralization to treat the acidic drainage from the mine.

DESIGN CRITERIA U-shaped containment created by building dams and walls inside the portal tunnel, filled with a mixture of spent mushroom compost and ¼-½ inch (No. 1) limestone gravel. The volumetric ration of compost-to-limestone in the substrate bed was decreased steadily from 5:1 at the influent terminus to 32:1 at the effluent terminus. This decrease corresponded to an expected pattern of more rapid limestone dissolution occurring towards the influent terminus. On both sides of the reactor, the substrate bed terminated in a 48° angle of repose. During operation, the substrate bed was kept saturated with water so as to impede oxygen penetration. Free pools of water occupied the remaining space between the walls and sloping ends of the substrate bed. Water flowed through a 6-inch diameter pipe to the opposite end of the containment, where control plumbing directed it either into the containment or out to a soda ash treatment system already in place. A 172 L operating volume (3 ft. deep x 2 ft diameter) dissolver tank was used for adding sugar to the reactor influent. The elevation difference between water entering and exiting the containment induced the flow of water to follow the U-shape of the containment. The U-shape design was selected to: 1) Increase the length-to-cross-sectional-area ratio of the substrate be in an effort to enhance the extent of water-substrate contact, 2) Place all of the control plumbing at the readily-accessible portal mouth, 3) Provide easy access for both influent and effluent sampling, and 4) Allow drainage from the mine to completely bypass the reactor when desired. Because the reactor required physical containment and protection from freezing temperatures, it was decided to construct it inside one of the portal tunnels.

Composition and Physical Characteristics of the Substrate Bed

Total Volume	26.7 m^3
Void Volume	14.9 m^3
Average Depth	1.04 m
Maximum Depth	1.18 m
Substrate Dry Weight	11,550 kg
Compost Moist Weight	17,950 kg
Limestone Gravel Weight	3470 kg
Total $CaCO_3$ Content	3930 kg

Dimensions of the Water Treatment Reactor

Overall Length	11.0 m
Width of Influent Corridor	1.22 m
Width of Effluent Corridor	1.52 m
Portal Tunnel Height	2.8 m
Height of Mine Pool Dam	2.0 m
Interior Area	31 m^2
Operating Volume	34 m^3

The reactor was modified to induce two types of flow: interstitial flow and channelized flow. Flow was also characterized based on five intervals: Interval I began when periods of surface flow across the top of the substrate bed were finally eliminated, and reactor flow became entirely interstitial (percolating through the substrate bed). Interval II began with the first addition of sugar to the reactor as an alternative organic carbon source. Interval III represents a period when sugar was not added. Interval IV began with resumption of sugar addition and ended with the onset of channelized flow. Interval V covers the period during which the reactor was operated with channelized flow.

Intervals of Reactor Operation

Interval	Duration (Weeks)	Flow Pattern	Carbon Source	Throughput (m^3 water)
Initial	27.6	Surficial	Compost	2279
I	2.7	Interstitial	Compost	139
II	14.3	Interstitial	Sugar	326
III	7.4	Interstitial	Compost	141
IV	16.6	Interstitial	Sugar	206
V	22.4	Channelized	Sugar	213

ADDITIVES For most of the time after June, 1995, cane sugar (sucrose) was added to the reactor as a bacterial carbon source. Sugar was usually added to the reactor every 3-4 days. During February and March, and after May 19, 1995, it was added at weekly intervals. The sugar dosage ranged between 4.6 and 18 kg/wk. During one 2-week period and one 4½-week period, ammonium chloride was added together with the sugar, in an effort to examine if sulphate reduction was being limited by the availability of nitrogen. During Interval II, sugar was added at a dosage of 4.7 kg/wk during the first five weeks, dosage doubled during the sixth week, and almost doubled again to 17.4 kg/wk for the remaining 8 weeks of the interval. During Interval IV sugar dosage was 18 kg/wk.

COSTS Costs (US dollars) for the reactor installation included: $310 for substrate material, $960 for containment materials, $620 for plumbing, $540 for trucking, $1300 for miscellaneous material and equipment, for a total of $3730, plus 600 person-hours of labour. Substantially lower costs would have been possible if the goal had not been scientific evaluation.

TIME OF OPERATION / INFORMATION August 1993 to August 1995.

PERFORMANCE DATA

Influent water characteristics based on 88 samples collected between August, 1994 and August, 1995 were as follows:

Parameter	Median (mg/L)	Range (mg/L)
Total Iron	347	150 - 718
Manganese	11	5 - 20
Aluminum	32	14 - 53
Calcium	185	123 - 275
Sodium	406	203 - 792
Sulphate	2530	1410 - 4490
pH	3.0	2.8 - 3.5

Interval I

Effluent pH higher than 5.7 and neutralized about 60% of the influent acidity, but removed only about 25% of the influent iron.

Interval II

Neutralization of 900-1500 mg $CaCO_3$/L influent acidity was achieved at the highest sugar dosage. Iron removal was 54-65%, however, the reactor effluent still contained 200-300 mg/L iron, and required aeration and clarification prior to final discharge.

Interval III

Iron removal and acidity neutralization declined to approximately the same levels that were observed in Interval I prior to sugar addition.

Interval IV

Net alkaline effluent was discharged within the first 16 days. 39 days before iron concentrations were lowered by more than 50%. After 18 days the reactor operated at considerably longer hydraulic retention times and lower (wintertime) temperatures. During interstitial flow, substrate bed clogging with precipitates, biomass, and/or gas pockets was encountered at different times during interstitial flow. Clogging also occurred deeper within the substrate bed over time.

Interval V

Effluent pH fell below 5.5 for 18 weeks. Total alkalinity generation was often less than 50% of that observed during Intervals I and III, and less than 20% of that observed during II and IV. Generally, retention times longer than 10 days seemed necessary to achieve significant neutralization with channelized flow. During the last 3½ weeks when the retention time was longer than 21 days, the effluent pH rose above 6.3, a net-alkaline effluent was being discharged, and iron concentrations were lowered by greater than 95%. The lowest effluent iron concentration was 0.4 mg/L.

During the entire period between installation and September 1995, approximately 1010 kg of $CaCO_3$ (26% of the total amount originally present) was dissolved and removed from the reactor.

OVERALL EFFECTIVENESS

The results indicated that although anaerobic neutralization may be a cost-competitive and convenient treatment method, the reactor design needs to be developed further before this method can be implemented for full-scale treatment at the mine. Most important variables influencing reactor performances were found to be sugar addition (presence of absence) and the flow pattern (surface, interstitial, or channelized). Decline in performance during Interval III confirmed the reliance of the reactor performance on sugar addition.

During all five operation intervals, the reactor typically lowered aluminum concentrations by greater than 99%, except for those times during channelized flow when the effluent pH was lower than 5. The reactors usually had little effect on manganese concentrations. The most significant removal of manganese occurred near the end of Interval V, when manganese decreased by 53% to a concentration of 4.3 mg/L, effluent alkalinity was 525 mg/L, and iron concentrations was 1.5 mg/L. During periods of active sulphate reduction, the reactor lowered influent concentrations of zinc, nickel, cobalt and probably copper by greater than 90% from influent concentrations in the range of 0.5 - 2 mg/L.

The reactor reliably neutralized the mine water when interstitial flow was used in combination with sugar addition, and retention times were longer than 6 days. However, iron removal was much less consistent, so subsequent aeration and clarification was often necessary. Only partial neutralization could be achieved in the absence of sugar addition, and channelized flow generally required retention times longer than 10 days and resulted in less stable performance.

TOXICITY

Suspended solids and dissolved ammonium, phosphate, and H_2S can have a negative impact when discharged into surface waters. Effluent concentrations of ammonium-nitrogen, and phosphorus may not remain under acceptable levels if nitrogen and phosphorus supplements were added to the reactor continuously. Effluent chemical oxygen demand could also pose a pollution problem and might compete with iron oxidation in the soda ash treatment system.

Although the effluent concentrations of total suspended solids, total organic carbon and H_2S did not seem to have constituted a significant pollution problem, the frequent black colour and disagreeable smell of the effluent might pose a nuisance problem. the oxygen demand of the effluent and the evolution of carbon dioxide gas can deplete oxygen in the reactor headspace, and thereby pose a considerable hazard for anyone who enters the reactor to perform maintenance.

TECHNOLOGY	Microbial Reactor Systems MRS 13
MAINTENANCE REQUIREMENTS	To alleviate clogging during intervals with interstitial flow, the substrate surfaces in the infusion crib pools were scarified using a combination of boot-scraping and perforation with a stick or pipe. The reactor was serviced every 3-4 days. Scarification of the substrate bed was necessary once every 2-4 weeks during interstitial flow. Cleaning of the control plumbing was rarely necessary. Black iron sulfide deposits accumulated in the effluent control plumbing very slowly, and may never have required removal. Orange ferric hydroxide deposits accumulated more rapidly in the effluent control plumbing, and required removal once every few months. The most significant clogging problem encountered with the control plumbing was the development of flow-obstructing, gelatinous biomass blooms downstream from the dissolver tank during combination of sugar addition and lower reactor flow rates. The total annual effort required to operate and maintain the reactor would be about 62 person-hours, with sugar addition, process monitoring, scarification, and miscellaneous maintenance each requiring about 26, 17, 13, and 6 person-hours, respectively.
MONITORING REQUIREMENTS	Influent and effluent net acidity, total iron concentration, *in-situ* pH, and total alkalinity generation (influent net acidity minus effluent net acidity) for periods of interstitial and channelized flow were determined.
TREATMENT PRODUCT DISPOSAL	The reactor substrate bed accumulated a mixed sludge consisting of iron sulfide, aluminum hydroxides, and elemental sulfur. The total dry weight of accumulated sludge was about 347 kg, which corresponds to a wet volume of about 580 L. Sludge disposal was not discussed.
ENVIRONMENTAL CONDITIONS	n.d.
COMMENTS	Discussion of the neutralization process and efficiency are also provided in the paper.

TECHNOLOGY	Microbial Reactor Systems	MRS 14
PROJECT SCALE	Pilot and field scale.	
REFERENCE	"Sulfate-Reducing Bacteria Demonstration Project". Mine Waste Technology Program. Environmental Protection Agency, Department of Energy. Montana Tech., Implemented by MSE Inc..	
LOCATIONS	Lilly/Orphan Boy mine, an abandoned hard-rock mine site near Elliston, Montana. Pilot scale tests were conducted at the Western Environmental Technology Office in Butte, Montana.	
DESCRIPTION	Demonstration project focuses on source control technology, specifically, biological sulphate reduction for the reversal or slowing of the process of acid generation and to improve water quality at sites affected by acid generation. Pilot scale tests were done to determine the effectiveness of the technology at the temperature of 8°C.	
DESIGN CRITERIA	Design involved using the subsurface mine working of the Lilly/Orphan Boy mine and the contained acidic mine water as an *in-situ* biological reactor. Two platforms were suspended by cables in both sides of a shaft below the static water level and were secured at the surface. An organic substrate was added to the system since sulphate reducing bacteria need a source of carbon in the form of simple organic nutrients. Substrate composed of cow manure, decomposed wood chips, and alfalfa were placed in the shaft and was supported by the platform. Injection wells were drilled into the tunnel for the placement of substrate. The acid mine drainage flows upward through the substrate in the shaft and horizontally through the substrate in the tunnel. The treated water subsequently flows out of the mine through the portal. Pilot scale testing was conducted in packed-bed reactors (4-foot plexiglass columns) operated in an up-flow configuration to more closely model the actual conditions of the hydrogeologic flow of water entering the shaft. A total of 7,000 gallons of mine water from the Lilly/Orphan Boy mine were treated during pilot scale testing.	
ADDITIVES	n.d.	
COSTS	n.d.	
TIME OF OPERATION / INFORMATION	Field demonstration began in late August, 1994, final reporting to be conducted in the fall of 1997.	
PERFORMANCE DATA	n.d.	
OVERALL EFFECTIVENESS	n.d.	
TOXICITY	n.d.	
MAINTENANCE REQUIREMENTS	n.d.	
MONITORING REQUIREMENTS	n.d.	
TREATMENT PRODUCT DISPOSAL	n.d.	
ENVIRONMENTAL CONDITIONS	Sulphate reducing bacteria require reducing conditions in the solution and will not tolerate aerobic conditions for extended periods. Sulphate reducing bacteria need a source of carbon in the form of a simple organic nutrient.	
COMMENTS	The technology is also believed to act as a source control by slowing or reversing the process of acid generation. Because biological sulphate reduction is an anaerobic process, it reduces the quantity of dissolved oxygen in the mine water and increases pH, thereby slowing or stopping the production of acid.	

BIOSORPTION SYSTEMS
PHASE II CASE STUDIES

TECHNOLOGY	Biosorption Systems BS 4
PROJECT SCALE	Laboratory Scale
REFERENCE	"Nickel and Arsenic Removal From Mine Wastewater by Muskeg Sediments". Fyson, A., Kalin, M. and M.P. Smith. 1995. In: Biotechnology and the Mining Environment, BIOMINET, 11th Annual Meeting Proceedings. Lortie, L., Gould, W.D. and S. Rajan (eds), pp. 103-115.
LOCATIONS	Not applicable
DESCRIPTION	Experiments were carried out in 2.5 L column reactors to assess the capacity of muskeg sediment to treat an arsenic (97 mg.L^{-1}) and nickel (72 mg.L^{-1}) rich seepage from a waste rock pile of a Saskatchewan mining operation. The study was designed to further define the rates of contaminant removal and the removal capacity of sediments to help establish design criteria for scale-up in the field.
DESIGN CRITERIA	2.5 L reactors of acrylic tubing with an internal diameter of 100 mm were set up. Muskeg sediment (500 mL) was placed in the reactors followed by the addition of 900 mL of waste rock pile seepage water. After a 24 hour settling period, 5 g of potato waste was sprinkled on the water surface to minimize sediment disturbance. 2 L wide-mouthed glass jars, containing seepage water and potato waste only, provided sediment-free controls to facilitate interpretation of the role of the sediment materials in contaminant removal processes. For some reactors and jars, 500 mL of water from the water column was removed by syphoning, and the reactor or jar was 'recharged' with 'fresh' seepage water to assist in determination of arsenic and nickel removal capacity of a particular sediment material.
ADDITIVES	500 mL of Muskeg sediment; 900 mL of waste rock pile seepage water (pH of 3.9; 72 mg.L^{-1} nickel; 97 mg.L^{-1} arsenic; 21 mg.L^{-1} nitrate-N; 21 mg.L^{-1} ammonium -N; and 29 mg.L^{-1} phosphorus); 5 g of potato waste.
COSTS	n.d.
TIME OF OPERATION / INFORMATION	129 days.
PERFORMANCE DATA	Seepage water used contained the following characteristics: Parameter Concentration (mg/L) Nickel 72 Arsenic 97 Sulphate 21 Nitrate-N 21 Ammonium-N 21 Phosphorus 29 pH 3.9 Data from the uncharged jars shows an early decline in Eh and the establishment of reducing conditions associated with the decomposition of potato waste and the release of volatile fatty acids, these acids contributed to the observed increase in acidity. The reducing conditions supported the reduction in concentrations of nitrate and iron. The removal of phosphate and ammonium may be due in part to uptake by microorganisms. Minor arsenic and nickel removal are attributable to a lack of iron and sulfide ions, respectively, and in the absence of sediments there are fewer sites for adsorption. In the reactors there was a steady reduction in conductivity, indicating a net removal or ions from solution. The decrease in Eh and rise in pH and acidity follows a similar pattern to that of the jars. The removal pattern in the reactors for nitrate, ammonium and phosphate was also similar to the jars. However, nickel and arsenic were effectively removed indicating that the presence of sediments appear to be essential for substantial nickel and arsenic removal. When degradable organic material (potato waste) is added, effective removal (>90%) of nickel and arsenic was achieved within 60 days and was associated with reducing conditions and an increase in pH from 4 to >7. Frequent replacement of seepage water in the column above the sediment has shown that sediments can remove at least 3.8 g.L^{-1} sediment for both nickel and arsenic. Analysis of sediments established that most of the arsenic and nickel is in the surface layer of the sediment. Removal rates of 50 µg.m^{-2}.min^{-1} for nickel and 80 µg.m^{2}.min^{-1} for arsenic were estimated for the reactors receiving one dose of seepage waters. For the reactors recharged every 1 to 2 weeks, rates of 170 µg.m^{-2}.min^{-1} for nickel and 250 mg.m^{-2}.min^{-1} were obtained.
OVERALL EFFECTIVENESS	For the control jars, despite the presence of reducing conditions in the jars, conditions were not favourable for arsenic and nickel removal when sediment was absent. In the 2.5 L reactors, the sediments appear to be essential for substantial nickel and arsenic removal. In laboratory conditions and in the presence of organic amendment, muskeg sediments have a considerable capacity for removal of both nickel and arsenic from wasterock pile seepage water.
TOXICITY	n.d.
MAINTENANCE REQUIREMENTS	n.d.
MONITORING REQUIREMENTS	Redox potential, electrical conductivity and pH, acidity, alkalinity, nickel, arsenic, iron, nitrate-N concentrations in water and sediment samples.
TREATMENT PRODUCT DISPOSAL	n.d.

TECHNOLOGY	Biosorption Systems BS 4
ENVIRONMENTAL CONDITIONS	n.d.
COMMENTS	

TECHNOLOGY	Biosorption Systems BS 5
PROJECT SCALE	Full scale
REFERENCE	"Biological Polishing of Mine Waste Water: Bioaccumulation by the Characeae". Smith, M.P. and M. Kalin. 1989. Biohydrometallurgy. pp. 659-665.
LOCATIONS	Northern Saskatchewan, Canada.
DESCRIPTION	Paper reports on the biomass distribution and growth rates of the Characean underwater meadow within a uranium mining waste management area, as well as on the concentrations of Ra^{226} and uranium in the biomass. These parameters are used to determine the expected removal capacity of the Chara process for application in a drainage basin in northern Saskatchewan, Canada. The total area of the drainage basin is 10 km^2, and includes a mined-out open pit (formerly Lake 1), Lake 2 and Lake 3. Lake 2 is the first water body downstream of the pit. It has a surface area of about 150,000 m^2 and an average depth of 1.5 m. A dam and overflow structure at the outlet of Lake 2 control the water level. From Lake 2, water flows through the Bog and a series of beaver dams before discharging into Lake 3, a lake very similar to Lake 2. The sources of Ra^{226} and uranium are waste rock piles located at the head of the drainage basin and Lake 1 sediments, which had reached the western portion of Lake 2 at the time the pit was developed. A dense population of *Nitella flexilis* (a Characean species) was identified in the main flow channels of the Bog between the outlet of Lake 2 and the inlet of Lake 3. *Chara vulgaris* and *Nitella flexilis* were found to be the major macrophytes in the lower part of the drainage basin in Lake 3. Lake 2 was devoid of any macrophytic growth.
DESIGN CRITERIA	Not stated in paper if the Bog naturally developed or was constructed. The annual Ra^{226} loading to Lake 2 from the waste rock piles and the sediments is about 4.1×10^8 Bq. The annual uranium loading from the same sources is 6.9×10^8 mg/yr^{-1}.
ADDITIVES	not applicable
COSTS	n.d.
TIME OF OPERATION / INFORMATION	Growing season of 1988.
PERFORMANCE DATA	Lake 2 Outlet Lake 3 Inlet Ra^{226} 0.43 $Bq.l^{-1}$ 0.08 $Bq.l^{-1}$ U 0.60 $mg.l^{-1}$ 0.18 $mg.l^{-1}$ The observed decrease occurred in a relatively small area of the Bog. The underwater meadow has a density distribution of 100 $g.m^{-2}$ to 1000 $g.m^{-2}$, in a water depth ranging from 0.6 to 1.9 m. The entire underwater meadow undergoes a complete growth turnover at least once per season. Concentrations of Ra^{226} and uranium in the biomass range from 10 to 30 $Bq.g^{-1}$ and 0.66 $mg.g^{-1}$ to 1.29 $mg.g^{-1}$, respectively.
OVERALL EFFECTIVENESS	The Chara system was estimated to be capable of reducing the annual loading of Ra^{226} by 41 to 100% and the uranium loading by 1.5 to 3.2%
TOXICITY	n.d.
MAINTENANCE REQUIREMENTS	n.d.
MONITORING REQUIREMENTS	n.d.
TREATMENT PRODUCT DISPOSAL	n.d.
ENVIRONMENTAL CONDITIONS	Throughout the growing season, *N. flexilis* was never found below 1.9 m, the depth at which light was attenuated to 5% of the incident radiation. As the quantity of light diminishes with depth, the spectral quality of light which reaches the vegetation canopy also changes, affecting both the photosynthetic rates and the photomorphogenic switch mechanisms. Thus these factors determine the depth distribution and the standing biomass.
COMMENTS	Chara refers to an attached, macrophytic algae.