# THE USE OF ENGINEERED WETLANDS TO TREAT MINE DRAINAGE

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## ABSTRACT

Minewater and leachates from waste accumulations at coal and metal mine sites can be major sources of surface- and ground-water contamination. These mine drainage (MD) streams can be difficult to handle because of their natures (turbid, variable flow rates, sometimes very high flow rates, low or high pHs, high levels of sulphates, and/or high levels of dissolved metals). Effective, reliable treatment of these wastewaters is important. Active treatment of MD in conventional wastewater treatment facilities may be suitable while mines & their mills are in operation, but can be expensive to maintain after mine closure. Semi-passive systems such as limestone drains are much more economic but still require some long term maintenance including periodic reagent replacement. Passive treatment systems such as constructed wetlands (CWs) are still more economic to build and operate and can continue to operate with relatively little attention long after mine decommissioning.

Modern constructed wetlands technology developed in the late 1970s and early 1980s. Many early CWs failed to achieve their designers′ goals as morphology was primitive and proper engineering design principles were rarely followed. CW design evolved through several stages to rectify such limitations through the *kinds of wetland basin (cell) types* used (e.g., from ponds and artificial bogs to free water surface and sub-surface flow cells), in *morphology* (e.g., from small facilities with one or few, long irregularly shaped cells to the current multiple train, multiple rectilinear cell, low aspect ratio systems), in the *volumes of water* that they could handle (e.g., from relatively low flow rates to many thousands of cubic metres per day), in *sizing methods* used (i.e., from early *loading method* relationships based on rule-of-thumb hydraulic and/or contaminant loadings to the modern, rational or *mass balance method* based on reaction kinetics), and in *engineering design* (from *ad hoc* designs to the use of formal civil and chemical design engineering techniques). The technology of CWs for municipal and agricultural wastewater treatment is now mature and there are tens of thousands of them in operation around the world.

Testing of the use of CWs for the treatment of industrial wastewaters such as MD began almost as soon as they began to be used for municipal wastewater treatment. Early mining industry designers first attempted to build open water wetlands vegetated with *Sphagnum* moss, but these were unsuccessful and were quickly supplanted by so called *aerobic wetlands* vegetated with cattails (*Typha* sp.). These proved useful for treating low flows of moderately-contaminated, net alkaline coal mine drainage, but were of limited use for other services.

Aerobic wetlands were quickly supplemented by *anaerobic (compost) wetlands.* These are ones that contain thick layers of organic substrate. They can treat small flows of acid rock drainage (ARD) with higher iron and dissolved oxygen contents than can aerobic wetlands. Later designs such as Successive Alkalinity Producing Systems (SAPS) added layers of alkalinity-generating limestone to the organic ones.

Engineered wetlands (EWs) are special, advanced kinds of constructed wetlands in which operating conditions are more actively monitored, manipulated and controlled than is the norm with ordinary CWs. In them, some of the multiple reactions/operating processes that are carried out in single cells in CWs are optimized in separate cells in series. This KISS principle make them much easier to design, build, operate and maintain than combination systems such as SAPS.

EWs may be categorized as semi-passive systems. Many kinds of wetland cells, limestone drains, and advanced MD treatment systems such as aerobic and anaerobic bioreactors can be expressed as engineered wetland cells. EWs are much more versatile than ordinary CWs and can operate under more extreme conditions of contamination, flow rate and environmental stress than can ordinary constructed wetlands. Engineered wetlands can be used to bridge the gap between active treatment and eventual, fully passive treatment in the ordinary constructed wetlands that they can evolve into over time.

This paper describes the evolution of CW technology, its historical and current application in the mining industry, some preliminary results from an operating, demonstration scale EW system treating a smelter-related leachate containing high levels of dissolved zinc, cadmium and arsenic, and the potential wider use of EWs to greatly increase the scope for wetlands by the industry.

## CONSTRUCTED WETLANDS FOR WASTEWATER TREATMENT

Ecological engineering, which is that sector of the environmental industry that involves the design, manipulation, control, and/or management of natural or bioengineered ecosystems that provide benefits to human society and the environment. It involves sustainable and ultimately self-designing systems (Higgins et al., 1999). Ecological engineering does not seek to restore ecosystems to earlier, pristine states, but tries instead to effectively manage them in order to achieve some set environmental or social purpose involving anthropogenic wastes or damage (e.g., the clean up of wastewater in a constructed wetland).

Bioengineered ecosystems such as CWs are not operated, nor do they behave, like natural ecosystems, and are usually highly stressed due to the introduction into them of high levels of contaminants in wastewaters. Stresses may be positive (the introduction of high levels of available carbons sources and nutrients that greatly stimulate wetland plant growth), or negative (such high levels of biochemical oxygen demand - BOD, ammonia, dissolved metals, etc. so as to be phytotoxic).

Modern CWs usually consist of a number of individual rectangular and/or irregularly-shaped basins (cells) connected in series and surrounded by berms of earth, clay, rock, concrete or other materials. They are often associated with a variety of ancillaries (e.g., ditching, pumps, control structures, bays) to form a *constructed wetlands treatment system*. Other kinds of wastewater treatment facilities (e.g., lagoons, cascades, land treatment fields, conventional wastewater treatment units) may also form part of a CW system, and water in them usually flows in two or more parallel trains. While biodiverse, native plants can be used in CWs to mitigate wastewater pollutants, monocultures of aggressive, macrophyte plants such as cattails and reeds often thrive in their artificial, highly stressed environments and quickly displace other plants.

Three types of cells (basins) may be used in a CW system: pond, free water surface (FWS), and sub-surface flow (SSF) cells. **Pond wetland** cells, as the name suggests, are shallow pools vegetated around the peripheries (10 – 30% coverage) and having major portions of their surfaces consisting of open water in which floating or submergent vegetation is found. They are most commonly used in conjunction with other types of constructed wetland cells.

**Free Water Surface** wetland cells are artificial marsh ecosystems and water in them flows on the surface in a basin through emergent wetland vegetation. In them, the submerged portions of wetland plants as well as soil and detritus act as attachment surfaces for biofilms of micro-organisms. These and physical filtration are responsible for much of the pollution removal. CWs involving mostly FWS cells are the most common type.

With a **Sub-Surface Flow** CW cell, water flows just under the surface of a porous material (substrate). Pollutant removal is via the substrate (and often in vegetation root systems growing in it). Although wetland vegetation can be present growing on it, its surface is largely dry. Generally, a SSF wetland cell consists of one or more beds of rock, gravel, aggregate, or sand. SSF wetland cells are usually smaller in area than FWS ones for the same levels of pollutant removal, and can tolerate higher loadings. The SSF wetland mode is often used where the wastewater being treated is noxious or odorous; where a higher degree of freeze protection is desired, where the attraction of wildlife (especially waterfowl) may be undesirable (e.g., at airports); and/or where ample, economic supplies of substrate material are readily available.

SSF wetlands are much more complex to design, engineer and build correctly so that proper hydraulic control is maintained and desired performance achieved. With poorly designed SSF wetlands (and many of the ones built earlier were), the substrates cannot handle the flows involved and/or plug up, resulting in the surfacing of wastewater flows. (While this leads to reduced performance, it is not a complete disaster, as the failure mode of a SSF wetland cell is a FWS wetland cell.)

There are three kinds of SSF wetland cells: those which are horizontally-fed (the most common) and in which the wastewater enters and flows horizontally through the substrate (HSSF cells), and those that wastewater moves vertically in the substrate (VSSF cells). VSSF wetland cells may be fed either upflow from wastewater distributors in or below the base of the substrates, or downflow. Some types of downflow VSSF cells also can be fed by sprayers or perforated pipes lying of the surfaces of the cells.

Constructed wetland ecotechnology is now well developed and can provide good removals of BOD, suspended solids, and pathogens from wastewaters (Kadlec & Knight, 1996). Ecological engineers are now able to design municipal, agricultural and many industrial wastewater treatment CWs with as much confidence as to their operability and pollutant removal levels as with comparable conventional wastewater treatment technology.

## THE DESIGN AND SIZING OF CONSTRUCTED WETLANDS

The design of modern CWs has evolved continuously from the time that the first pilot, demonstration and full scale systems were built in the late 1970s and early 1980s. Although many early CWs for treating municipal wastewaters such as those at Listowell in Canada from 1984 to 1986 (Herslowitz, 1986) were judged to be failures at the time because did not fulfil their designers′ expectations, they served as valuable lessons in the evolution of modern constructed wetland ecotechnology.

The designers of early CWs focused on the demonstration of the concept, ecology, vegetation types, biology, and R&D matters. The first kinds considered were simple ponds. These were soon followed by systems that attempted to mimic natural wetland systems such as bogs (dominated by *Sphagnum* mosses) or marshes (dominated by emergent macrophytes such as cattails). Combinations of ponds and artificial marshes were then tried, often in association with other types of wastewater treatment systems such as land treatment. Marsh wetland cells quickly became the dominant types used. Meadow/marsh/pond systems began to be studied in the early 1970s at Brookhaven, NY, USA (Kadlec & Knight, 1996), and from this evolved the early marsh/pond/marsh design (i.e., a FWS wetland cell followed by a pond, then another FWS wetland cell).

Although some early designs involved rectilinear cells in parallel flow trains, most had only one train consisting of one, or a best a very few, cells of variable geometry. The concept of multiple train, multiple cell treatment wetland *systems* took some time to become the norm. Similarly, early CWs usually involved the treatment of relatively small volumes of water and it is only recently that FWS and SSF wetland systems capable of routinely treating very large flow rates of wastewater have been demonstrated successfully (e.g., a 2.7 ha, six train, 12 cell SSF wetland at Edmonton International Airport in Western Canada is designed to treat up to 20 L/s of glycol-contaminated stormwater resulting from cold weather aircraft de-icing with glycol, Higgins et al., 2001).

The critical importance of chemical engineering and civil engineering for successful CW design, construction and operation was little appreciated early on, and many builders of constructed wetlands had little or no engineering training. Many designs involved cells with very high aspect ratios (length/width), with many of these having high sinuosity to get as much wetland into as little space as possible. Both design concepts have since been discredited. Gradually, however, the work of wetland engineering pioneers such as Reed (Reed et al., 1995) and Kadlec (Kadlec & Knight, 1996) led to the definition of modern engineering principles for CW design.

The usual first step in designing a constructed wetland system is to size it (make a first approximation of its surface area, **A**). Early on, sizings were based on empirical data based on influent loadings of water and/or pollutants. While this rule-of-thumb *loading method* has been used to size many wetlands, it is of limited utility as it does not take into account the impacts of water depth, temperature, or residence time. Although the old loading-based sizing method is still used to size small septic and sewage treatment wetlands, for larger modern constructed wetlands and ones treating wastewaters other than sewage, it has been superceded by the *rational* or *mass balance method* that considers wetlands to be horizontal biofilters simulated by chemical reactors in which first order reaction kinetics are involved. Early versions of this model assumed that constructed wetlands could be simulated by plug flow reactors (Reed et al., 1995), but recent data from actual operating wetlands has shown that they are better simulated by a series of continuously stirred reactors (the tanks-in-series, TIS, version of the model).

Under the TIS mass balance method, a constructed wetland can be sized using the following equation:

$$\mathbf{C/C_i = 1/(1 + k_v}\tau/\mathrm{N})^{\mathbf{N}} \quad (1)$$

where **N** is the number of tanks in series, **C** is the concentration of any one of several contaminants in the wastewater, **i** represents the inlet, $\tau$ the residence time of a volume of wastewater in the wetland, and $\mathbf{k_v}$ the first order volumetric rate constant for its removal in the CW.

The hydraulic residence time, $\tau$, can be given by:

$$\boldsymbol{\tau = \varepsilon Ah/Q} \qquad (2)$$

In equation 2, $\varepsilon$ is the porosity in the cells (that of the substrate in SSF wetlands or that of the vegetation/detritus/sediment system in FWS ones); **h** is the average water depth in the wetland, and **Q** is the flow rate of wastewater through the wetland.

If $k_v$ is known, solutions to Equations 1 and 2 for the size (area A) of a constructed wetland needed to obtain a desired pollutant removal level can be determined using known wastewater flow rate(s); the inlet concentrations of each pollutant of importance; and the target (desired) outlet concentrations of these same contaminants. Wetland sizings may be carried out for desired levels of removals at a particular temperature of pollutants such as BOD, suspended solids, ammonia nitrogen, nitrate nitrogen, phosphorus, pathogens, dissolved metals of various kinds, and other contaminants.

The values of $k_v$ for particular contaminants can be obtained from published values of it or its areal equivalent, k (where $k = k_v h$, ) in the literature or by carrying out a treatability test. In ordinary CWs, the porosity is relatively constant (~95% for FWS wetlands and ~40% for the gravel of many SSF ones), as is water depth (h ~ 0.2 - 0.4 m for FWS wetlands and ~ 0.5 - 0.7 m for SSF ones), so areal rate constants are usually adequate for design. However, with pond wetlands and the engineered versions of other kinds of constructed wetlands (see below), these values may vary by larger amounts and volumetric rate constants are more appropriate. Whichever is used, care must be exercised when using literature data for their values, as most of these are for the wetland treatment of municipal and agricultural wastewaters under limited conditions and probably will not apply for the treatment of other types of wastewaters. (The authors of this paper operate pilot-scale CW test units in Canada in Ottawa, at a campus of the University of Guelph in Ontario, and a major lead-zinc smelter facility in British Columbia, and these are used to determine the rate constants and other scale up parameters needed to successfully design modern constructed wetlands.)

As mentioned, CWs can be sized by solving the above equations for each of the contaminants of concern in a wastewater. Whichever of the calculations yields the largest area, the result shows that the reduction of that pollutant in the system is deemed to be controlling and it will be the one on which more detailed design can later be based. The values of the rate constants are different for pond, FWS and SSF wetlands, with the highest values being found for the latter, confirming that SSF wetlands provide better treatment for a given surface area. The reason is not hard to see. Constructed wetlands can be viewed as slow rate, attached growth systems. The area available for microbial attachment in SSF ones varies with the volume of the wetted substrate (a three dimensional matrix), while that in pond or FWS wetlands varies with the surface of the sediment, plants, and detritus (roughly a two dimensional area, albeit a highly folded one).

A SSF wetland is generally more costly to construct than a FWS wetland of equivalent size. In many cases, this may be compensated for by better SSF treatment efficiency (higher rate constants) and better operating capability. Once initial estimates of surface area, A, are found, the detailed design of modern constructed wetlands requires the application of standard chemical and civil engineering methods including flow calculations based on the use of the Manning Equation for FWS wetlands and Darcy's Law for SSF ones (Reed et al., 1995, Kadlec & Knight, 1996).

## ENGINEERED WETLANDS

Ordinary CWs are very good at removing BOD and suspended solids (75 - 95% + removals in many cases), pathogenic bacteria & viruses (2 to 5 log removal), non-dissolved metals, oil & grease and lighter organics from wastewaters flowing through them, especially where large proportions of the contaminants are associated with particulate matter (as is often the case with municipal wastewaters). Particulate materials settle in the relatively quiescent areas of wetland cells and are filtered out by plants, sediments, detritus, substrates and biofilms. However, ordinary constructed wetlands are not as effective in removing some dissolved contaminants such as nutrients, salts and many metals in solution. This is compounded if the wastewater being treated is *recalcitrant* (difficult to handle) because of its nature (e.g., saline, very highly contaminated, alkaline or acidic, large volume, intermittent in flow, and/or cold).

Nitrogen compounds removal in a CW can be used as an example. Their removal from wastewater in a constructed wetland cell is largely bacterially-controlled: organic nitrogen compounds are mineralized to ammonia by facultative bacteria, the resultant ammonia is nitrified to nitrates by aerobic bacteria, and the nitrates then may be denitrified to nitrogen gas by anaerobic bacteria. Ordinary CW cells contain oxic and anoxic zones where the respective reactions can occur, but conditions are not ideal for any of them and, on average, only 20 - 40% overall total nitrogen removal is achieved. Moreover, the bacterial nitrogen removal reactions are highly temperature sensitive and proceed much slower in cold water (Reed et al., 1995, Kadlec & Knight, 1996). As a result, the removal of nitrogen often will control the sizing of a CW, which could be much smaller if ways to remove nitrogen contaminants more efficiently could be devised.

What was needed was a new configuration/operating mode for CWs that could allow them to remove contaminants such as ammonia and high levels of dissolved metals as well, that would operate better in cold weather, and that would be able to handle especially recalcitrant wastewaters such as many kinds of mine drainage.

The answer may be found with *engineered wetland*s[1]. These are an advanced type of CW in which process conditions and/or operations are designed, modified, manipulated and/or controlled in such a manner as to allow contaminant removals to be optimized. At the same time, cold weather operability is improved, as is the ability to deal with otherwise adverse conditions. (Higgins et al., 1999, Higgins, 2000 a, b, c).

Constructed wetland systems may be "engineered" in many ways. For example, influent streams may be varied in flow rate (or turned off) to periodically enhance redox conditions; effluents from various points in the wetland system may be recycled to other points; ordinary substrates in some SSF cells may be replaced with special ones having specific qualities (e.g., the ability to permanently chemically adsorb or precipitate certain pollutants from wastewaters passing through them); things may be added at certain points (e.g., heat, chemicals, air); wetland vegetation may be selected for its phytoremediating properties; and/or a variety of other options may be considered (Higgins, 2000a, b, c).

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[1] 1 *Although the names "engineered wetland" and "constructed wetland" are sometimes used interchangeably, it is appropriate to define the former more narrowly by limiting the engineered designation to apply only to the more advanced, semi-passive systems. All engineered wetlands are constructed wetlands, but not all constructed wetlands are engineered ones.*

As mentioned, modern CWs generally consist of a number of cells arranged in series in several parallel trains (Higgins & MacLean, 1999, Higgins et al., 2001). This too can be exploited in engineered wetlands by designing them such that specific reactions (e.g., aerobic nitrification) are carried out in certain oxic cells while potentially antagonistic reactions (e.g., anaerobic denitrification) are carried out in different, anoxic cells in a series, allowing very much higher conversions under both regimes.

For example, aeration air may be introduced under the gravel substrate of an engineered SSF wetland cell, increasing the nitrification rate of ammonia-contaminated wastewater passing through it from the average 25 - 35 % conversion (oxidation) to nitrates normal for CWs (Reed et al., 1995) to almost 100% (Higgins, 2000a). Under these conditions, wetland cell substrate thicknesses can be increased from the usual 0.5 - 0.9 m to 1 - 3 metres or more, vastly decreasing the surface areas required, while at the same time improving efficiencies and winterability, and reducing needed residence times. Similarly, anaerobic denitrification reactions can then take place in other, separate cells of the engineered wetland train, ones in which conditions are manipulated to ensure maximum reduction of the nitrate to nitrogen gas. In addition, when anaerobic reactions are confined to a few cells of a constructed wetland train, vegetation on these cells may be dispensed with, and deep layers of water placed over their gravel substrates, ensuring anoxic conditions and permitting operation even under thick ice in winter.

Another example of a way to "engineer" a SSF wetland is to replace the gravel substrate in one or more of its cells with an alkaline aggregate material such as steel slag. Under these conditions, phosphorus removal can be increased from the normal 20 - 40% level to virtually 100%, even for a case where the influent stream was a highly recalcitrant, very acidic gypsum stack leachate containing almost 500 mg/kg of phosphate (Higgins, 2000a). Kinetic rate constants for the engineered wetland cells in the example were 24 to 32 times higher than those normally found for the removal of phosphorus from wastewater in ordinary CWs. Arsenic and fluoride can also be removed from wastewaters containing them using this method.

Engineered wetland cells may be of the pond, FWS or SSF kinds, but SSF ones are the most common. Because of the need to operate them in more active ways (e.g., by adding air in some cases or using materials such as limestone that are used up in them over time in other cases), EWs are not fully passive systems and should be regarded as *semi-passive*. They do, however, have the potential to evolve into passive, ordinary CWs over time if influent contaminant concentrations decline naturally over time, ending the need for the more aggressive treatment methods.

The operating costs for an EW will usually be higher than those for an ordinary CW, and one will only be economic where its special qualities (e.g., smaller sizes, lower capital costs, the ability to handle recalcitrant wastewaters better, winter operability) more than compensates for this.

The term "wetland" conjures up heterogeneous, open water areas vegetated by biodiverse growths of aquatic plants. It must be emphasized that modern CW, and especially EW, *systems* need not involve open water (e.g., SSF cells); if planted, may not involve wetland plants (e.g., the use of un-vegetated cells or vegetating SSF cells with phytoremediating terrestrial plants growing hydroponically;) are usually highly regular in morphology (e.g., trains of rectilinear cells in series); can be associated with many kinds of conventional mechanical equipment (e.g., pumps, aeration systems); and can involve as part of their systems other kinds of wastewater treatment technology (e.g., cascades, overland flow fields).

## THE TREATMENT OF MINE DRAINAGE

ARD is formed when pyrite (and other mineral sulphides) are exposed to air (oxygen) and water. The bacterially and/or chemically-catalyzed oxidation of this pyrite releases ferrous iron, sulphate and hydrogen ions (protons, $H^+$). The ferrous iron may then further oxidize to ferric iron, which then may hydrolyze resulting in the precipitation of ferric oxyhydride, producing more hydrogen ions.

The main sources of acid rock drainage are the contact of oxygen-containing water with the workings of base metal, precious metal and coal mines; the passage of water through pyrite-containing mine tailings accumulations; and contact of water with pyrite-containing, removed cover material, base rocks, coal beds and/or mine rock refuse areas. Depending on the pH and nature of the rock involved. ARD may mobilize a wide variety of metal ions into solution. Generally, Fe, Mn, Al, Cu, and Zn are found in the highest proportions.

ARD may be managed by **prevention** (stop it from forming in the first place), **control** (limit or control the migration of water and/or materials/contaminants that might cause ARD), **treatment** (deal with it after it has been formed), or **enhancement** (cause it to be formed faster than normal to "burn" acid-causing materials out of waste-generating accumulations over relatively short times).

ARD treatment may be carried out by **active**, **semi-passive** or **passive** methods and there are a variety of biotic and abiotic processes available for each. Often, active treatment processes are used at operating facilities, while passive and semi-passive systems are preferred for facilities that are no longer in operation yet still continue to generate mine drainage (Skoussen, 1998, Brown et al., 2002). Passive systems are generally more economic to build and operate than semi-passive systems, which in turn are cheaper than active ones.

The treatment of ARD containing a wide suite of dissolved metals can involve creating either aerobic conditions (minewater oxidation) and anaerobic conditions (sulphate reduction) or both. It aims to raise pH, remove suspended solids, allow metals that easily form (oxy)hydroxides to form and precipitate, and, in some cases, to remove sulphates as metal sulphides. ARD may be neutralized by contact with limestone or similar alkaline materials, but the resulting neutral mine drainage (NMD, with pHs from 6 to 7) may still contain elevated levels of certain dissolved metals and sulphates.

Typically, mine drainage contains >500 mg/L of sulphates and this salinity can be removed from it by reducing the sulphates to sulphides, by biological uptake, and/or by the formation of organic esters on plant decomposition in a vegetated wetland cell (biosorption). The former (sulphate reduction) depends on anaerobic heterotrophic microbes called **sulphate-reducing bacteria (SRB)**. SRB occur in the sediments and other anoxic zones of wetlands and can use organic carbon and sulphates in their metabolisms. The basic equations for SRB-mediated sulphate reduction can be represented by:

$$\mathbf{2CH_2O + SO_4^{=} + H^+ \quad Y \quad H_2S + 2\,HCO_3^-} \qquad (3)$$

where $CH_2O$ represents a carbon source and Me is a typical dissolved divalent metal cation in MD. The hydrogen sulphide quickly reacts with any dissolved divalent cationic metals in the water (e.g., Zn, Cd, Ni) to precipitate them as relatively insoluble metal sulfides.

$$Me^{2+} + H_2S \rightarrow MeS\downarrow + 2\,H^+ \qquad (4)$$

It is accordingly an excellent method of removing certain metals (Hard et al.,1997) It is noted that two moles of alkalinity and one mole of acidity are the net result of reactions 3 and 4, so the action of the SRB also is to raise alkalinity and buffer the solution. There are many kinds of SRB. Generally, SRB do not flourish at pHs below 5.5 and prefer higher levels of alkalinity, with 6.6 being optimal (Govind et al., 1999).

SRB can be exploited in *anaerobic bioreactors* which remove contaminants such as sulphates and dissolved metals from NMD by passing it through a suitable packed bed under anaerobic conditions. Anaerobic bioreactors can be designed as active treatment facilities or as semi-passive ones. Mine drainage can be fed into them using surface distributors or sprayers. In colder climates, buried inlet and outlet distributors can be used to direct flow through the reactors. In active modes, the precipitated metal sulfides can be separated out, while in passive modes they accumulate in the packed bed and/or in downstream filters (e.g., FWS wetland cells). Where the MD being treated contains a dissolved metal with economic value (e.g., zinc), the packed beds and their filters can later be "harvested" to recover the metal for re-use.

As may be seen from Equation 3, an adequate supply of sulphate is necessary in the NMD for the reactions to work. The carbon source can be any type of carbonaceous material (e.g., sawdust, biosolids, manure) submerged in a wetland cell's water, the decaying roots/detritus of the wetland plants growing in a wetland cell, an added soluble carbon-based liquid material (e.g., methanol) (Hard et al., 1997). Solid organic materials in an anaerobic bioreactor not only serve as carbon sources but also physically retain metal sulphides.

Like all microbes, SRB are sensitive to temperature, and reaction rates are lower where colder water is involved. However, unlike open water algae, SRB are prokaryotes and are less affected by cold than the eukaryotes. They consist of varying populations of bacteria, some of which are cold-adapted and can thrive down to temperatures as low as 4 ºC, with increased numbers of bacteria compensating for the lower reaction rates (Fortin et al., 2000). Anaerobic bioreactors can be used ("expressed") as cells in an engineered wetland system.

The most common active method of treating ARD is by liming. By this method, ARD is mixed with slaked lime or limestone in a vessel or pond, and the metals are removed as part of a precipitated sludge. There are serious limitations with this active approach in terms of application and effectiveness. The operating costs for an active treatment plant for the very long periods that ARD may be generated may be prohibitive. Also, sludges that the process generates are unstable mixtures of metal hydroxides that have in turn to be managed and disposed of (Govind et al., 1999). In an effort to develop more passive methods, **open limestone channels** (OLCs) were developed. These involve putting down layers of limestone in ditches and running the ARD over them. This adds alkalinity and raises pH, as follows:

$$CaCO_3 + H^+ \rightarrow Ca^{2+} + HCO_3^- \qquad (5)$$

OLCs are semi-passive as eventually the limestone will be used up and will require replacement. Many OLCs are successful only for limited periods before dissolved and atmospheric oxygen produces precipitates that partially "armour" the limestone particles, forming hard surface layers that inhibit more reactions.

It was found that some of the armouring limitations of limestone channels sometimes could be overcome by using another kind of semi-passive system, an **Anoxic Limestone Drain** (ALD). With ALDs, ARD is passed through a buried underground channel or cell containing crushed limestone and capped with impermeable clay layers and plastic liners. ALDs generate bicarbonate alkalinity in the ARD flowing through them. The ARD will produce hydroxide and other sludges after it passes out of the ALD, and accommodation must be provided downstream (in pits or wetlands) to allow them to settle out. ALDs are being superceded by more versatile **Oxic Limestone Drains (OLDs)** which are designed in much the same way but are designed to treat ARD that contains elevated levels of dissolved oxygen, ferric iron, and/or dissolved aluminum that would plug an ALD. In OLDs, materials that might armour it (e.g., ferric hydroxide) and/or plug up its limestone beds (e.g., aluminum hydroxide) are controlled by operation methods, and swept out of the drain for removal later. ALDs and OLDs can be expressed as cells in engineered wetland systems.

## CONSTRUCTED WETLANDS FOR TREATING MINE DRAINAGE

Studies in the late 1970s and early 1980s in the USA determined that MD passing through natural *Spaghnum* bogs did not adversely affect them (Huntsman et al., 1978, Girts & Kleinmann, 1986). This led to abortive attempts at the pilot and full scale level to use artificial *Sphagnum* bogs for the treatment of acidic coal mine drainage (Kleinmann & Girts, 1987, Brown et al., 2002.) Beginning in the 1980s, R&D on CWs for treating ARD then focused on the potential of using open water, cattail-vegetated wetlands to reduce ferrous iron and raise pH (Kadlec & Knight, 1986). These are called *aerobic wetlands* and are designed with very shallow water depths (often only a few cm). They are FWS wetlands, and their concept is to provide residence time and aeration for metal oxidation and hydrolysis so that the metal hydroxides of iron, aluminum and manganese can be precipitated out and retained (Skousen, 1998). Often, they are preceded by cascades and other methods of oxygenating ARD.

Aerobic wetlands are said to be most appropriate where the MD being treated contains net alkalinity to neutralize metal acidity (Skousen, 1998). Until very recently, sizings were based on the old loading method, with a hydraulic loading rate of 200 $m^3$/ha.d and a seven day retention time being recommended for some (Wile et al., 1985). Others suggested sizing them for contaminant loadings of 10 g/$m^2$.d for Fe removal and up to 2 g/$m^2$.d for Mn removal if sufficient alkalinity is present (Hedlin et al., 1994). Sizings are considered valid only for relatively low MD flow rates, and were applicable for the treatment of certain kinds of coal mine drainage.

The aerobic wetland concept was quickly followed *anaerobic or compost wetlands*. These are very shallow (2 - 8 cm deep) cattail planted or un-vegetated open water wetlands containing an organic substrate (e.g., compost) 30 - 70 cm thick. Flow through the substrate of these types of FWS wetlands is encouraged (i.e., operation in an almost semi-SSF mode). Anaerobic wetlands are recommended for use where the mine drainage has net acidity. They are suggested for the treatment of small flows of poorer quality ARD with higher Fe and dissolved oxygen contents than can be treated in aerobic wetlands.

Sizings for anaerobic wetlands also have been based on empirical data (e.g., 20 $m^2$ of wetland per $m^3$ of flow/day determined by averaging results for 48 constructed wetlands for treating coal mine drainage in Pennsylvania, USA) (Hellier, 1989). In another use of the loading method, the Tennessee Valley Authority in the United States, a major investigator of the use of CWs for treating coal mine drainage, recommended that they be sized using hydraulic loadings of between 15 x $10^{-3}$ m/d and 42 x $10^{-3}$ m/d (Reed et al., 1995).

Later loading method sizing methods were based on area-adjusted criteria presented in terms of grams of Fe removed per $m^2$ of wetland per day. Still later designs proposed a sizing criterion based on net acidity. Each of these was found to be unsatisfactory for the more complex suites of contaminants found in metal mine drainage and even those for coal mine MD were applicable only for limited ranges of contaminant chemistries at low flow rates. It was found that where flows were greater than 50 L/min, loading method approaches resulted in sizings considerably larger than did other approaches (Brown et al., 2002). Indeed, for the high mine drainage flow rates typically of many metal and coal mining facilities (tens of L/s), the loading method is of little value.

Only recently has consideration been given when sizing CWs for treating mine drainage of using the modern mass balance method used in other areas. The oxidation of ferrous iron, the precipitation and sedimentation of ferric oxyhydroxides and iron sulphide oxidation each has been shown to follow first order kinetics (Brown et al., 2002). It is reasonable to expect that the removals of other MD contaminants in a CW do so as well. In addition, the use of reaction kinetics for wetland sizing can accommodate both situations of high flow/low contaminant concentration and those of low flow/high concentration, something that the loading method cannot do.

The use of the modern method demands determining first order kinetic rate constants for each of the contaminants of interest (see Equation 1, above), and the highly variable chemistries and operating situations of mine drainage flows make every sizing situation unique. A study carried out on published results from 35 wetlands treating coal mine ARD has confirmed that *"... pollutant removal is better described by first order kinetics"* (Tarutis et al., 1999) but indicated a wide range of values for the rate constants for Fe and Mn removals, so published data for other wetlands are no guideline for design. The only practical ways to determine them is by carrying out a pilot scale treatability test with the actual (or simulated) MD under consideration.

By 1989, there were at least 300 CWs in operation in the US at mining facilities (Kadlec & Knight, 1986) but few of these were designed using ecological engineering principles. Reed states that in many cases *"....the sizing and configuration of [many of] these systems was not rationally based"* (Reed et al., 1995). Cell geometries, water depths, residence times, hydraulics, and other aspects for them were decidedly sub-standard in relation to modern wetlands engineering methods. Because the performances of many such badly designed aerobic and anaerobic wetlands was poor and metals removals were, according to one influential author, *"...variable and unpredictable"* (Weider, 1989), negative perceptions of the use of CWs to treat MD were engendered in some circles. This had a somewhat similar inhibitory effect on the use of constructed wetlands for mine drainage treatment as did the likewise erroneous perceptions that the early Listowell experience had for CWs treating municipal wastewaters.

For CWs used by the mining industry, little attention has been given to optimal cell geometries, water depth, hydraulics, temperature effects, or winter operabilities. In addition to the poor design problems endemic with aerobic wetlands, anaerobic wetlands and anaerobic bioreactors have had to contend with those of substrate blinding and plugging due lack of understanding by many of their designers of basic chemical engineering methods for the flow of fluids in porous media.

There was, and still is in some circles, little appreciation of the lessons learned for modern constructed wetland design in areas outside the mining industry when ridiculous statements such as "*Designs which imitate natural wetlands, such as varied shape and slope are more aesthetically pleasing ...*" and " *... a more serpentine shape outperformed a rectangular shape....*" are made in documents advising on the design of CWs for the mining industry (Smith, 1997). Too often, wetland sizings continue to be based on the old loading method, and, while better attention has recently been paid to mine drainage geochemistry, designs based on modern wetlands engineering principles are still rare. Indeed, even the concept that CWs need to be designed as multi cell, multi-train *systems* has been slow to be appreciated.

Nevertheless, throughout the 1990s, constructed wetlands technology for the treatment of mine drainage continued to develop and systems combining CWs with other types of MD treatment such as ALDs became more common. Before the end of the twentieth century, over 1000 CWs had been built to treat mine drainage (Skoussen, 1998).

**ENGINEERED WETLANDS FOR TREATING MINE DRAINAGE**

Successive Alkalinity Producing Systems are another type of wetland system which has been developed recently to treat ARD by adding alkalinity and raising pH (Watzlaf et al., 2000, Brown et al., 2002). In a SAPS cell, ARD flows down through a bed of compost which is intended to reduce ferric iron to its ferrous state, thereby limiting armouring in a limestone bed which lies below the compost. Drainage is out of bottom distributors. Indeed, a SAPS can be regarded as a combination of an ALD and compost wetland. While SAPS are improvements on ALDs, because of their multi-layer substrate designs, they are hard to build, difficult to operate, expensive and messy to clean out when the inevitable substrate replacement time occurs, and generally have been built with as little attention to modern wetlands designs principles than were the ones described above.

If protection from limestone armouring is a major consideration, it is far more sensible to precede an ALD cell in an engineered wetland treatment train by a VSSF wetland cell in series containing an organic substrate mixed with gravel (a far more sensible arrangement than trying to mix organics with dissolving limestone). This accomplishes the same thing as a SAPS but with cells that are far easier to operate, are far less prone to channelization and plugging, and which are much easier to clean out when substrate replacement is required.

An even better EW design for ARD treatment is to oxidize ferrous iron in an ARD to ferric ions in a *Thiobacillus*-inoculated, aerated VSSF cell containing gravel, then to pass it through an OLD into which sufficient recycle NMD is added to ensure turbulent conditions in the OLD's limestone substrate. Precipitates formed in the OLD are swept out of it (before they can cause armouring and/or plugging) into downstream pond and/or FWS cells where they settle/are filtered out. Part of the clean effluent from these cells is recycled to be mixed with the ARD into the OLD cell. Pilot scale testing of this engineered wetland process configuration is now underway at Ottawa using a high aluminum, simulated ARD from a large open cast coal mine in New Zealand (Fe/Al/$SO_4$ of 50/100/800 mg/kg, respectively).

In a semi-passive engineered wetland such as that described above, the pump around recycle mode might be used when ARD flows and contaminant flows were high. As ARD flows and pollutant concentrations declined naturally with time, the pump around mode could be terminated and a dosing syphon placed on the OLD's outlet to allow potentially armour-producing precipitates in it to be periodically flushed out under a more passive operating mode (Vinci & Schmidt, 2002). As ARD quality continued to improve over time, eventually the EW cells could be converted into ordinary FWS constructed wetland cells operating in fully passive modes. Over very long periods of time, the CW could be allowed to evolve into a natural wetland.

In a modern EW system, VSSF/OLD/FWS cells such as those described above can be used to remove from ARD those metals that can be precipitated as in limestone-based systems (i.e., metals such as iron, manganese and aluminum precipitating at pHs below the maximum 6.5 – 7.0 range achievable by limestone neutralization). The remaining metals, those not precipitatable under oxic conditions below 6.5 to 7.0 pH (Cu, Zn, Ni, Cd), and those which do not form insoluble hydroxides, can then be removed in a following anaerobic bioreactor cells.

## ENGINEERED WETLANDS DEMONSTRATION UNIT

A demonstration scale, five cell engineered wetland system treating for treating an NMD has already been built and has been operating since 1997 at the Teck Cominco lead-zinc smelter in Western Canada. It treats cadmium- and zinc-contaminated landfill leachate, and seepage from the site of an old arsenic scrubbing pond. The system involves two upflow, anaerobic bioreactor VSSF wetland cells ($700m^3$/18 m by 30m, $600m^3$/18 m by 25m) with biosolids-based substrates (60% pulp mill biosolids, 35% sand and 5% cow manure underlain by a layer of dolomitic limestone), followed by three gravel HSSF cells (5m by 10m, 5m by 10m & 10m by 30m, all 0.7 m thick) vegetated with *Typha* and other plants, and a final pond wetland cell (20m by 25m by 0.8m deep) (Mattes, 2002, Mattes et al., 2002, Mattes et al., 2003).

The following table presents average metal removal results for this engineered wetlands system from May, 2000 to late January, 2001 during which 2.84 MM L of NMD was treated, and from May, 2001 to late January, 2002 during which 3.68 MM L of NMD was treated (Mattes et al., 2003).

**Table 1**
**METALS REMOVAL IN ENGINEERED WETLAND SYSTEM**

| | Average Influent Concentration (mg/L) | Effluent Concentration (mg/L) | Removal (%) |
|---|---|---|---|
| **Arsenic** | 70 | 1.3 | 98.1 |
| **Cadmium** | 5 | 0.03 | 99.4 |
| **Zinc** | 355 | 15 | 95.7 |

This demonstration engineered wetland system is running well and it has already shown excellent results, summer and winter. Work continues with it to optimize performance and it is felt that much lower zinc effluent levels (averaging < 1 mg Zn/kg) will soon be achievable on a regular basis.

The EW demonstration unit's main purpose now is not to show that anaerobic bioreactors and HSSF cells can be used as part of engineered wetland systems for removing dissolved metals from NMD (it has already done that!), but to work out the practical aspects needed as designs progress to larger, full scale systems (e.g., pH control, hydraulics, winterability, compaction, precipitates, best distributor types). For example, several years of operations seem to indicate that precipitated sulphides will not plug up substrate beds as the sulphides are very fine and colloidal in nature.

A two train, anaerobic bioreactor/HSSF cell EW demonstration system for treating only the arsenic-contaminated stream has been built alongside the system described above and it will be operated in parallel in future to try to zero in on the exact mechanisms of arsenic removal in engineered wetland cells.

A full scale, 2.4 ha engineered wetland system to treat almost 1700 $m^3$/d of NMD containing approximately 15 mg/L of zinc is being planned.

## CONCLUSION

Modern CWs treating other kinds of industrial wastewaters have shown that such systems operating at very high MD flow rates (> 15 L/s for SSF and > 25 L/s for FWS) can treat stormwater, process waters and cooling water. Ones treating even larger volumes of water are on the horizon. Contaminant removal kinetic rate constants for them can be determined using treatability tests, so the highly limited loading method need no longer be used in wetland sizing.

Engineered wetlands show excellent potential as the next generation ecotechnology for the treatment of mine drainage. Their multiple cell designs allow the various reactions needed to treat ARD or NMD to be carried out in highly efficient manners in the successive cells of their treatment trains. The engineered wetland concept for treating mine drainage streams has already been proven at pilot and demonstration scale facilities. Full scale facilities are being planned.

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