ABSTRACT

Aspects of research work undertaken by the Wetland Ecology Research Group at University College Dublin are summarised here. Wastes from mining activities generally contain high concentrations of heavy metals and other toxic substances. Reclamation methods to treat these wastes include the use of wetlands, for revegetation of mine tailings under flooded conditions and for the treatment of tailings water. Both natural and constructed wetlands are frequently employed for the treatment of mine wastes. Through a complex array of plant, soil and microbial interactions contaminants, such as heavy metals and sulphates, can be successfully removed from wastewater. Suitable vegetation can stabilise the tailings sediment, thereby preventing it from being dust-blown or leached into the surrounding environment. Our research suggests that these two techniques for treatment of mine wastes are successful and economically viable.

INTRODUCTION

Wetlands are defined as having (1) a water table above or at the soil surface for a significant proportion of the year, which is a determining factor in their make-up of the ecosystem, (2) an emergent vegetation characteristic of wet biotopes (often containing a large proportion of helophytes), and (3) a soil characteristic of wet biotopes (anoxic, chemically reduced) (Mitch and Gosselink 1986).

Wetlands are attractive as an endpoint in the rehabilitation of mine wastes, such as tailings and tailings water, for two reasons. First, pollutants originating from mining activities, such as metals and sulphur, are relatively immobile when present under waterlogged conditions (Gambrell 1994). Second, pollutants are retained by the wetlands from water passing through the wetlands (Hammer 1989; Dunbabin and Bownmer 1992). Both characteristics are largely due to the same processes. Permanently waterlogged wetland soils are generally anaerobic, because of the relatively low diffusion rate of oxygen through water compared to air. In addition, micro-organisms present in such soils respire using terminal electron acceptors other than oxygen. Such organisms can, for example, reduce ferric iron to its ferrous form, or reduce sulphate to sulphide. The formation of highly insoluble sulphide from soluble sulphate in particular is important. Not only does that process lead to the precipitation of sulphur, but also co-precipitation of metals, including iron, zinc, lead and cadmium. Once metal sulphides have precipitated, they are stable and insoluble providing the soil remains anaerobic (McIntire and Edenborn 1990; Dvorak et al. 1992). Wetlands can therefore be used in several aspects of rehabilitation of mine wastes. First, mine tailings can be revegetated under wetland conditions, using wetland plants, and second, the quality of water originating from mining operations can be improved by passing it through wetlands, whether they are naturally-occurring or constructed specifically for that purpose (Hammer 1989).

Wetland soil processes, interactions with the vegetation, and the application of wetlands for the rehabilitation of metal mine tailings are among the subjects addressed by our research group. This paper presents an overview of some of our recent research projects.

Wetland plants for vegetation of mine tailings

Outside Ireland, wetland vegetation has been successfully established on mine tailings (Nawrot 1994; Beckett et al. 1997). Within Ireland, this approach to revegetation of mine tailings has not yet been applied, but has been proposed for the Lisheen Mines, Co. Tipperary. Ireland has a net precipitation level greater than the evapotranspiration level, therefore the supply of water for the establishment of wetlands should not pose a problem. However, characteristically, mine tailings have a low nutrient content and high concentrations of potentially toxic metals and sulphur compounds, both of which can be problematic for the successful establishment of plants. Nutrient supply to the plants can be improved by adding fertiliser. Alternatively, plants that have low nutrient re-
requirements can be used. The latter solution is more attractive as it reduces the cost of the reclamation process. In addition, plants that are used for revegetation purposes can survive higher metal concentrations than plants that are not accustomed to such conditions.

Beining and Otte (1996) observed that the amphibious floating sweetgrass (Glyceria fluitans) was growing very well on tailings in a pond near the abandoned lead–zinc mine at Glendalough, Co. Wicklow. This was the first time that this species was reported to grow under such conditions and a study was initiated to investigate whether the species was suitable for revegetation purposes (McCabe 1998). Some results have been published already (McCabe and Otte 1997), while other data are intended for publication (McCabe et al., in press), and a brief outline of these will be given here.

In a short-term (five weeks) greenhouse experiment and a longer term (fourteen months) field experiment, G. fluitans (R. Br.) from a metal–contaminated site (Glendalough tailings pond) and a non-metal contaminated site (Lough Dan) grew equally well on metal mine tailings from Glendalough under flooded conditions (McCabe and Otte 1997). However, growth of both populations was significantly reduced under non-flooded conditions compared to flooded conditions.

Glyceria fluitans from Lough Dan also grew successfully on lead–zinc tailings from the active tailings pond at Outokumpu Zinc–Tara Mines in an outdoor microcosm experiment. Findings of the short-term (five weeks) experiment (McCabe and Otte 1997) suggested that treatment of G. fluitans with NPK fertiliser was of no measurable benefit to the plants. However, the long-term (thirteen months) experiment indicated that treatment of tailings with NPK fertiliser (700kg ha\(^{-1}\)) significantly improved growth and biomass production of G. fluitans. It appears that growth of G. fluitans responds slowly (after a period of three–four months) to treatment with fertiliser.

The minimal fertiliser requirements of G. fluitans and the ability to grow vigorously on mine tailings of elevated metal concentrations (220–360μmol g\(^{-1}\) Zn; 10–120μmol g\(^{-1}\) Pb; 310–410μmol g\(^{-1}\) Fe) favour the use of this wetland plant for revegetation purposes. It is essential, however, that a cover of standing water is maintained over the tailings in order to ensure that G. fluitans is continually grown under flooded conditions.

Filtering of metals from contaminated water passing through a ‘volunteer’ wetland

Wetlands can also be used for quality improvement of contaminated water (Brix and Schierup 1989; Hammer 1989). Biogeochemical and physical processes, as well as uptake by plants, lead to reduced concentrations of contaminants, including nitrogen, phosphorus and metals, as the water passes through the wetlands. Naturally occurring, so-called ‘volunteer’ wetlands, as well as constructed wetlands, can be used for the treatment of polluted water. Many studies have shown the effectiveness of such systems in reducing concentrations of contaminants in water, but the question still remains as to how the system itself and its longevity are affected by the accumulation of toxic substances (Walski 1993). If treatment wetlands deteriorate within a relatively short period of time (10–20 years) then this approach would not be attractive for municipal and industrial purposes. Most constructed treatment wetlands are younger than fifteen years and, therefore, have not been active long enough for an accurate assessment of the impact of accumulation of pollutants on their longevity. Natural, volunteer wetlands may have been receiving pollutants for a much longer period of time. Such situations are rather rare, but one example exists at Glendalough, Co. Wicklow. A marsh adjacent to an abandoned lead–zinc mine, which had its peak activity during the 1880s and finally closed in the 1950s, still receives metal-contaminated water. This water, having passed through the marsh, enters the Upper Lake. We investigated the retention of metals by the marsh (Beining and Otte 1996) and from the results we calculated the remaining capacity of the marsh to accumulate metals as well as its expected longevity (Beining and Otte 1997).

Samples of soil and porewater at depths of about 10cm were taken along the length of the marsh (Beining and Otte 1996). In order to calculate the retention of metals by the marsh, its maximum capacity for accumulation of metals, the capacity used to date, and its expected longevity, the following assumptions were made (Beining and Otte 1997):

1. the part of the marsh being studied was hydrologically separated from the Glenealo river running adjacent to the marsh;
2. the abandoned mine site acts as the source of metals entering the marsh;
3. if retention of metals from water occurred, then concentrations of metals in porewater should decrease from the source (the abandoned mine site) towards the lake;
4. due to metal retention from the porewater, a gradient of decreasing concentrations of metals in the soil from the source towards the lake would be established; and
5. the soil was homogeneous throughout the marsh.
The first assumption seems reasonable given the structure of the marsh. A riverbank clearly separates the river from the marsh, and water can be seen flowing from the mine site into the marsh and out towards the lake in a direction parallel to the river. Decreasing concentration gradients in porewater were found for zinc and arsenic (cadmium and lead were present in concentrations below the detection limit of the atomic absorption spectrophotometer used for the analysis). In soil, decreasing concentration gradients were found for arsenic, cadmium, lead and zinc.

Concentrations of iron were not found in porewater or soil patterns, suggesting that the iron in the marsh does not originate from the mine site, nor is it retained by the marsh. The patterns for porewater concentrations suggested a 95% retention of zinc and a 65% retention of arsenic. The patterns for soil concentrations suggested that 16% of the marsh’s capacity to accumulate cadmium had been used up to date, while the values for lead, zinc and arsenic were 25%, 26% and 30%, respectively. Therefore, this marsh seems to have been effectively retaining metals for at least a century, and based on its present capacity to bind metals, can be expected to continue to do so for at least another two centuries.

**METHOD**

Two identical experimental systems were constructed on the grounds of the Randalstown tailings facility of Outukumpu Zinc-Tara Mines Ltd, Co. Meath, during the summer of 1997. Each system comprises of three compartments in series—an ‘inflow’ pond receiving untreated tailings water overflowing into a wetland compartment, which in turn overflows into an ‘outflow’ pond receiving the now treated water (Fig. 1). Waterproof baffles in each wetland compartment serve to increase the flow path of the water, thereby increasing the potential for sulphate retention. On site a computer (ACS Pentium PC) connected to the pumps regulates the flow of tailings water through the systems. Also connected to the computer are four permanent industrial-grade electrodes (Rosemount Solu Cube® Analyser Model 2700), one situated in each of the four ponds. These facilitate continuous and simultaneous monitoring of conductivity and temperature. Data are logged into a database, every half-hour for the initial two months, thereafter every three hours, 24 hours a day, and can be accessed remotely via a portable modem. This makes it possible to monitor the performance of the systems from our laboratory at the Belfield campus of University College Dublin using pcANYWHERE 32 software.

The wetland compartment of each system is filled with approximately 50cm depth of a **Sulphate retention by a constructed wetland**

Retention of substances in wetlands, including metals and sulphur, from water passing through them is accommodated by the physico-chemical characteristics of the wetland components (Dvorak et al. 1992). In theory, the prevailing anaerobic, chemically-reduced conditions in wetland soils lead to the reduction of sulphate to sulphide, which may precipitate with metals to form insoluble metal sulphides, or may evolve as hydrogen sulphide. Therefore, levels of sulphate and metals tend to be reduced in water after passage through wetlands. These processes are mediated by interactions between micro-organisms, soil and plants (Hammack and Hedin 1991; Ledin and Pederson 1996; Roane et al. 1996). The aim of the project described here is to design a wetland for treatment of the sulphate-rich tailings water originating from the tailings facility of Outukumpu Zinc-Tara Mines Ltd, Co. Meath. Typically this water contains 300–2200μg SO$_4^{2-}$ L$^{-1}$, compared to background levels of 70μg SO$_4^{2-}$ L$^{-1}$ in the nearby Yellow River (Knight Piesold 1996). If successful, such a system is expected to be a more economical and more sustainable approach to the chemical treatment processes currently employed.
mixture of Spent Mushroom Compost (25%) and fine grit (75%). This mixture was chosen because pilot experiments had shown that it combined good permeability with optimal growth of plants. Mushroom compost is also regarded as one of the best organic substrates for sulphate-reducing micro-organisms (Ledin and Pederson 1996). At the bottom of the inflow and outflow ponds in each system, a layer of about 25cm of a 1:6 mixture of Spent Mushroom Compost and fine grit was deposited to provide a substrate for the invertebrate species that spontaneously inhabit the systems.

The planting density chosen was based on similar research on constructed wetlands (Szczepanska and Szczepanska 1982; Kadlec and Alvord 1989). During August 1997 the wetland compartments of each system were planted with Typha latifolia (four plants per m²) and Phragmites australis (nine plants per m²). Glyceria fluitans (seven plants per m²) was added in August 1998.

Flow rates were set at 300–500mL min⁻¹. These rates were adapted to fit the size of the systems based on the values given for other operational systems as described by Crites (1994). The theoretical residence time for the systems is 52 days, but the applicable value has yet to be confirmed.

Here we report a select number of parameters measured, namely, volunteer species (invaders), pH, redox potential, conductivity and sulphate concentrations in water. The site was visited on a monthly or bi-weekly basis and some parameters were monitored continuously. The pH was measured using a glass combination electrode connected to a pH meter (WTW pH90). Redox potential was measured using a platinum electrode connected to a mV meter (WTW pH90). Conductivity was measured using the industrial electrodes mentioned above. For the analysis of sulphate a Dionex ion chromatograph was used. Invading flora were identified using the standard key of the British Freshwater Biological Association.

## RESULTS

Because the systems have only been in operation for about one year, and several operational problems occurred, the data set collected is somewhat scattered and incomplete. Despite this, a number of useful observations have been made. The plant species introduced to the systems—Phragmites australis, Typha latifolia and Glyceria fluitans—were rapidly established. Some P. australis flowered, but T. latifolia was particularly successful, growing to 164cm tall in some cases and with at least ten inflorescences in each of the wetland compartments.

Table 1 lists the volunteer invertebrate and plant species observed during early 1998. Seven invertebrate species and two plant species had invaded the systems at this stage. Some of these, such as the Chironomids and Coleopterans, had already been observed a few weeks after the systems were filled with tailings water.

The pH of the water passing through the systems remained in the neutral to alkaline range, with values varying from 6.8 to 9.1 (Table 2). Observations of redox potential indicated that the wetland soil environment is chemically reduced, with values generally lower than −100mV (Table 2).

The average conductivity (of two to six measurements taken around the same time as the water samples for sulphate analysis) was positively corre-

### Table 1—Inventory of invertebrate and plant species recorded from January to April 1998 in the experimental treatment systems at the Randalstown tailings facility of Outukumpu Zinc-Tara Mines Ltd, Co. Meath.

<table>
<thead>
<tr>
<th>Group</th>
<th>Species</th>
<th>Occurrence</th>
</tr>
</thead>
<tbody>
<tr>
<td>Invertebrates</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Chironomidae</td>
<td>Chironomis spp.</td>
<td>Both systems, all compartments</td>
</tr>
<tr>
<td></td>
<td>Psectrotanypus varius</td>
<td>Both systems, all compartments</td>
</tr>
<tr>
<td>Coleoptera</td>
<td>Dytiscus marginalis</td>
<td>Both systems, all compartments</td>
</tr>
<tr>
<td>Copepoda</td>
<td>Cyclops spp.</td>
<td>Wetland compartment of system 1 only</td>
</tr>
<tr>
<td>Ephemeropera</td>
<td>Cleon dipterum</td>
<td>Both systems, all compartments</td>
</tr>
<tr>
<td>Hemiptera</td>
<td>Sigara lateralis</td>
<td>Both systems, all compartments</td>
</tr>
<tr>
<td>Hirudinea</td>
<td>Helobdella stagnalis</td>
<td>Wetland compartment of system 2 only</td>
</tr>
<tr>
<td>Plants</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Amblystegiaceae</td>
<td>Calliergon cuspidatum</td>
<td>Both systems, wetland compartments only</td>
</tr>
<tr>
<td>Ranunculaceae</td>
<td>Ranunculus tripartitus</td>
<td>Wetland compartment of system 2 only</td>
</tr>
</tbody>
</table>
Fig. 2—Correlation between conductivity (μS cm⁻¹) and sulphate concentrations (mg L⁻¹) in tailings water running through the experimental systems. The overall correlation coefficient for log-transformed values is $r^2 = 0.641$.

lated with the sulphate concentrations of the water in the inflow and outflow ponds (Fig. 2). The correlation coefficient, $r^2 = 0.641$, indicates that 64% of the variation is explained by the relationship between conductivity and sulphate. Differences in sulphate concentrations between compartments are not evident from Fig. 2. These are better illustrated when the average sulphate concentrations for each of the compartments are considered (Fig. 3). As experimental system 1 was the only system to receive interceptor ditch water consistently over the period of operation discussed here, data from that system only will be regarded (tailings water for system 2 originated from two possible sources, run-off from a vegetated tailings pond nearby, or from interceptor ditch water, and has not yet been analysed systematically). Average concentrations varied between 1300 mg L⁻¹ in the interceptor ditch in September and 250 mg L⁻¹ in the outflow in November. Statistical analysis (analysis of variance) showed that differences between compartments were significant ($P < 0.001$). The data illustrate that between the interceptor ditch and the outflow, sulphate concentrations were reduced by 64% in September, 14% in October, and 62% in November of 1998.

### DISCUSSION

As the treatment systems have been in operation for approximately one year only, the data must be regarded with caution. However, the performance of the systems is promising. It is apparent that both planted and volunteer species are growing healthily, and several invertebrate species have invaded the system, indicating that the tailings water is not toxic to these organisms. This is consistent with observations that *G. fluitans* plants could easily be grown on tailings from Tara Mines (McCabe and Otte 1997; McCabe 1998). Most of the species that have invaded the systems so far are typical colonisers (Brusca and Brusca 1990). Species like *Dytiscus marginalis* may dwell in the systems for a short period of time only, because they can fly into or out of the systems at will (Friday 1988).

<table>
<thead>
<tr>
<th>Compartment</th>
<th>Systems</th>
<th>Date</th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>September 98</td>
<td>October 98</td>
<td>November 98</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Eh (mV)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Wetland</td>
<td>1</td>
<td>$384 \pm 130$</td>
<td>$-122 \pm 53$</td>
<td>$-120 \pm 21$</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>2</td>
<td>$266 \pm 19$</td>
<td>$-190 \pm 10$</td>
<td>$-75 \pm 24$</td>
<td></td>
<td></td>
</tr>
<tr>
<td>pH</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Interceptor ditch</td>
<td>1</td>
<td>n/a</td>
<td>n/a</td>
<td>n/a</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Inflow pond</td>
<td>1</td>
<td>$8.4 \pm 0.3$</td>
<td>$8.3 \pm 0.1$</td>
<td>$8.1 \pm 0.1$</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Wetland</td>
<td>1</td>
<td>$7.1 \pm 0.2$</td>
<td>$7.5 \pm 0.7$</td>
<td>$6.8 \pm 0.1$</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>2</td>
<td>$8.7 \pm 0.6$</td>
<td>$8.1 \pm 0.3$</td>
<td>$7.1 \pm 0.0$</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Outflow</td>
<td>1</td>
<td>$8.5 \pm 0.0$</td>
<td>$8.4 \pm 0.2$</td>
<td>$8.4 \pm 0.0$</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>2</td>
<td>$9.1 \pm 0.1$</td>
<td>$7.9 \pm 0.1$</td>
<td>$8.2 \pm 0.0$</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
The conductivity of the water appears to correlate with the sulphate concentrations. Conductivity is an overall measure of the ionic strength of the water to which all dissolved ions contribute. Our data indicate that sulphate is one of the more important ions determining conductivity in this particular type of water, as would have been expected from the high concentrations of sulphate present. Thus conductivity may be used to monitor the retention of sulphates in the systems and, possibly, in future large-scale applications. However, as one ion is removed from solution another may be displaced, in which case net conductivity may not change.

The sulphate data suggest that concentrations are reduced considerably by the systems. However, these data indicate that the largest reduction occurs between the interceptor ditch and the inflow of the actual experimental systems, suggesting that sulphates are removed from the water in the header tanks, possibly through precipitation. These and other (sub-) compartments of the system, such as the wetland soils where sulphide is expected to precipitate, are being incorporated into the regular monitoring program at present.

The redox potentials of the soils in the wetland compartments observed so far are relatively low and typical of sulphate-reducing environments. This is in accordance with the apparent odour of sulphides emerging from the disturbed soil. Furthermore, a recent study into the bacterial communities of the systems clearly revealed the presence of sulphate-reducing bacteria (Wharton 1999).

The data so far indicate that improvement of tailings water quality using treatment wetlands is feasible. Questions that remain unanswered and will be addressed in future research are:

(i) Where does the sulphate that is removed from the water go?
(ii) Does the removal of sulphate from the tailings water lead to displacement of other potentially harmful substances?
(iii) Are metals retained from the tailings water by the wetlands?
(iv) What is the expected life span of the treatment wetlands?

CONCLUSION

The Wetland Ecology Research Group at UCD focuses on, among other things, the application of wetlands for rehabilitation of mine wastes under Irish conditions. Current findings are in accordance with those from outside Ireland, that is that wetlands can be successfully used for both revegetation of tailings and treatment of tailings water. Wetlands have a strong capacity for the retention of pollutants, including those originating from mining activities. The establishment of a wetland cover over tailings provides a promising alternative to the more traditional dryland option.

Critics of both applications frequently doubt the longevity of these systems. Where the use of wetlands for treatment of polluted water is concerned the answer is simple—size matters. If a wetland is built sufficiently large to manage the input of pollutants, then it should be functional for many decades. This was illustrated by our research on the 'volunteer' wetland at Glendalough (Beining and Otte 1996; 1997). Restriction in the use of wetlands for treatment of wastewater is therefore determined by the available space for construction of such a system. Revegetation of tailings with wetlands should be sustainable for indefinite periods of time. The vegetation component provides the source of organic matter needed to drive the chemical reduction of sulphides and the subsequent precipitation of metal sulphides. Through these processes the metals and sulphates are returned to the form they were derived from originally in the mining process, as many metal ores are sulphide in nature. Therefore, wetlands can be used to complete the recycling of mine wastes from sulphides back to sulphides.

ACKNOWLEDGEMENTS

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REFERENCES


