Field Scale Trials Treating Acidity in Coal Pit Lakes Using Sewage and Green Waste

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Abstract

Acidic mine pit lakes represent a potentially valuable resource to both the environment and the community if the water can be remediated to an appropriate standard. Beneficial end uses include: aquaculture, water for irrigation, recreation, and for nature conservation. Additions of organic material to support sulfate reducing bacteria (SRB) may convert sulfate back to sulfides, removing acidity and metals in the process.
As a before-after-control impact (BACI) design, a field-scale manipulative experiment monitored post-dosing water quality in one treatment and three control pit lakes over 19 months. From July 2006 to January 2007, a 70 ML lake was filled with dried sewage sludge (60 t), liquid sewage sludge (3,190 t) and municipal green waste (980 t). Monitoring of this new treatment lake and the remaining control lake and other control lakes then continued for another 19 months at monthly intervals.
Control lake water chemistry was generally stable, or at worst predictable and able to be explained by groundwater influx and heavy cyclonic rainfall events. Following organic additions, treatment lake water chemistry displayed large pH increases. Water chemistry of this treatment lake was best explained by internal sulfate reduction processes. Nevertheless, these pH increases suddenly declined after 12 months of increase. This decline may be due to surface water acidity inputs and mixing during heavy rainfall events leading to re-oxidation of iron and iron compounds or to exhaustion of the dosed organic carbon.
These field-scale experimental observations suggest that addition of low-grade organic materials for remediation of acid mine waters at field scale shows promise. Further monitoring and analysis is required to access the degree of treatment that can be achieved and how long this treatment will continue.

Key words: pit lakes, AMD, bioremediation, sulphate reduction

Introduction

Large-scale open-cut mining activity has left a legacy of many thousands of mine pit lakes worldwide (Klapper and Geller 2002). Pit lakes form in these voids when dewatering ceases in a mine pit that extends below the watertable. As groundwater, rainfall and surface runoff slowly fills the pit, this water may react with oxidised seams and pit walls resulting in dissolution and oxidation of exposed minerals (Castro and Moore 1997). Pit lakes of low water quality may threaten the health and well-being of both local communities and the natural environment (Doupé and Lymbery 2005; McCullough, Hunt et al. in review).
Acid Mine Drainage is arguably the greatest environmental problem facing water management in the international mining industry (Gray 1997; Harries 1998). However, the large quantities of water in pit lakes represents a potentially valuable resource to mining companies, the environment and community; if appropriate water quality can be achieved (McCullough and Lund 2006). Mining lease pit lakes therefore can represent significant short-term resources to adjacent operations, leading to reduced pressure on regional natural water resources over a longer period.
In situ microbial sulfate reduction through addition of labile carbon as organic substrates has potential to be an efficient and effective remediation approach for the treatment of AMD contaminated waters. (Kleeberg 1998). Microcosm experiments have indicated that sewage and greenwaste may be suitable organic materials for the treatment of AMD waters (McCullough, Lund et al. 2006; McCullough, Lund et al. 2008). This paper reports on a field-scale experiment testing the suitability of greenwaste and sewage sludge as an organic matter source for *in situ* AMD pit lake remediation.

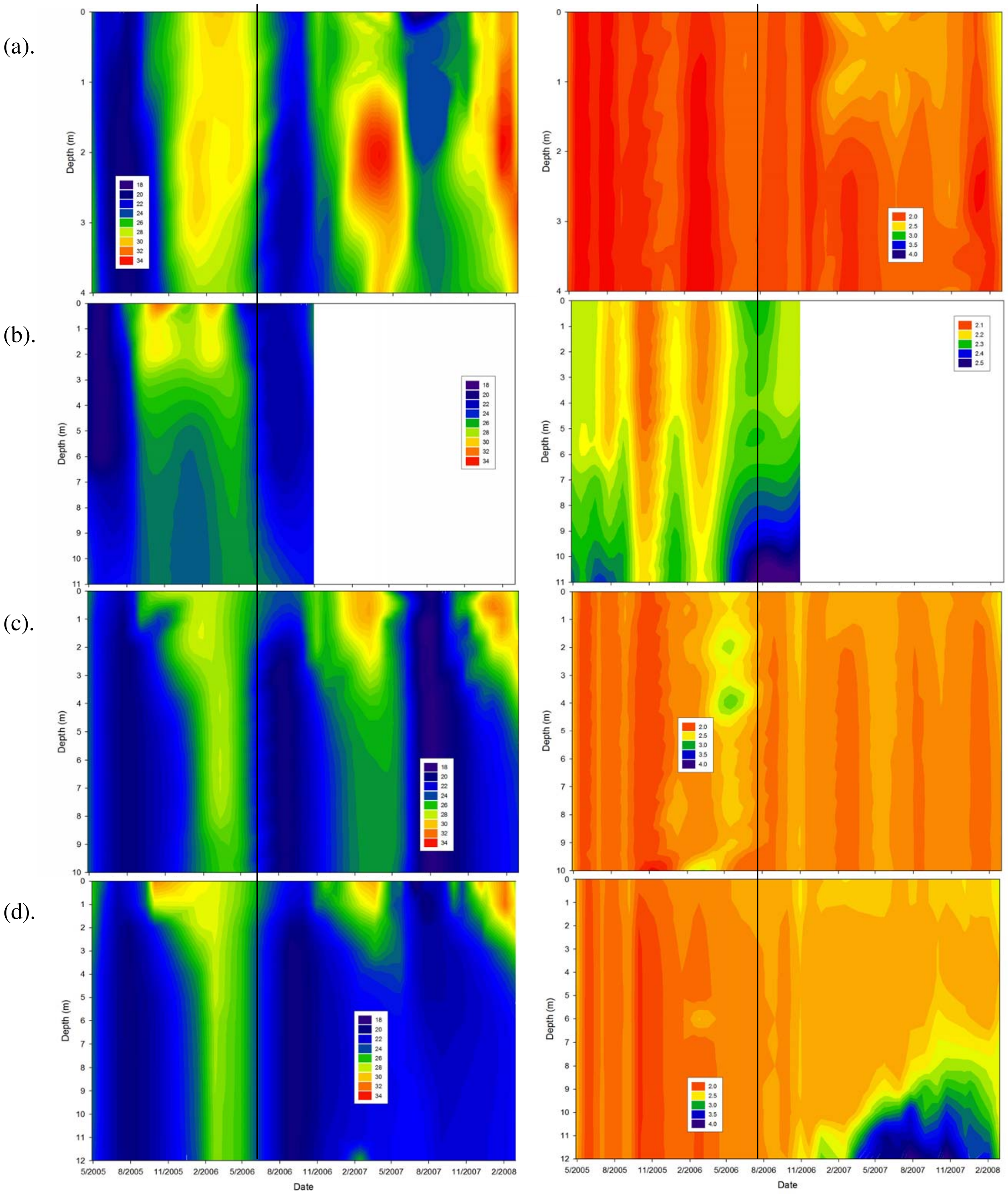
Methods

The study site for this experiment was at the Collinsville Coal Project, 150 km inland in North Queensland, Australia. This region is tropical with a warm Dry season and a hot Wet season where heavy rains occur due to coastal cyclonic activity. Three acid mine pit lakes on the Project were used as controls, while GAEW lake was treated in a before-after/control-impact (BACI) design. Prior to organic dosing, water chemistry of these pit lakes was monitored monthly for 15 months. This monitoring included vertical profiling of the water column physico-chemistry. From July 2006 to January 2007, the 70 ML GAEW lake was filled with dried primary digested sewage sludge (60 t), liquid primary digested sewage sludge (3,190 t) and municipal green waste (980 t). Limitations as to availability of organic materials meant organic dosing of GAEW represented an AMD:sewage:greenwaste ratio of 73:3:1. Monthly monitoring of the treated lake and control lakes then continued for another 19 months following dosing. Groundwater wells on western, eastern and southern sides of the pit lake were also monitored for groundwater depth and chemistry during these times.

Results

Throughout the monitoring period, groundwater levels were consistently above GAEW water surface levels and pH was circum-neutral across all monitoring bores. Pit lake water temperatures were warm and well-mixed during the Dry season (June to September; Figure 1). Lake warming began at the end of the Dry season with thermal stratification occurring in both sample years around December and lasting to June. However, water temperatures were warmest in the centre of the water column of Ramp 8 pit lake During stratification, indicating a salinity gradient effect or a warm water intrusion. Pit lake pH was stable around 2.2 in control lakes except for Ramp 5a during a cyclonic heavy rain event and for Ramp 8 where bottom waters increased to pH 2.7 during 2005 and 2006 Dry seasons (Figure 1). GAEW pH was as low as control lakes prior to dosing, however following dosing in August, GAEW hypolimnion pH began to increase from December 2006 to a maximum of 4.1 in July 2007. By November 2007, the bottom 4 m of GAEW, representing 1/3 of the volume of this pit lake, was above pH 3. However, following this peak, pH rapidly declined to the end of February 2008. Overall water column pH was then slightly but evenly elevated to pH 2.4 at the surface to 2.8 above the benthos. Following organic addition a strong sulfide smell was also evident in hypolimnion samples of GAEW, but not from the control lakes.

Figure 1 *Temperature (left) and pH (right) profiles of pit lakes (a) Ramp 5a, (b) Ramp 8, (c) GAEE, (d) GAEW from May 2005 to February 2008. The black line indicates the beginning of organic dosing*



Discussion

Depth and duration of stratification is an important factor in the efficacy and viability of sulfate reduction for *in situ* pit lake remediation (Martin, Crusius et al. 2003). Generally, CCP pit lakes appear to thermally stratify in the late Dry season although this stratification is broken when Wet season wind and rains arrive. Consequently, water column stratification is expected to occur in the treatment pit lake for at least 6 months of the year. Although heavy cyclonic rainfall events, as encountered in early 2007 and 2008, may destabilise thermal stratification earlier than usual, the halocline that forms as a result of this low conductivity water overlaying the denser high conductivity pit lake water may be more stable and effective at preventing mixing and consequently reduced metal oxidation than thermal stratification during warmer months (Wen, Poling et al. 2006). However, the unpredictable climate of this site makes prediction of such field remediation experiments difficult as heavy rains and mixing events can still occur at any time of the year.

The pit lakes do not appear to be thermally stratified over the cooler Dry season. Inversion of the GAEW thermocline during cooler months of the mid-Dry season is not likely to be due to exothermic re-oxidation of reduced sulfides in the hypolimnion as has been observed with cool-temperate AMD pit lakes (Gammons and Duaine 2006) as the hypolimnion ORP did not markedly change at this time under the continuing influence of the halocline. Rather, this inverted thermocline is more likely due to exothermic biological activity of anaerobic bacteria decomposing dosed organic material.

Other control pit lakes have seen a slight decrease in epilimnion pH due to cyclonic rain waters, and Ramp 8 has seen a significant increase in hypolimnion waters due to groundwater ingress. However, GAEW demonstrated an increasing pH developing in the hypolimnion since late 2006, only a few months after organic dosing began.

Conclusions

In addition to the strong sulphidic smell from hypolimnion waters, the pH increase in GAEW is indicative of alkalinity-producing sulfate reduction. However, unlike previous microcosm experiments (McCullough, Lund et al. 2006; McCullough, Lund et al. 2008) the sudden decline in hypolimnion pH may indicate that remediation processes have now slowed or even reversed. This difference between microcosm and field experimental results may be due to incoming surface water acidity or water column overturning by heavy rainfall causing addition of acidity or re-oxidation of iron and iron monosulphides/pyrites. The use of liquid primary digested sludge in this field experiment instead of dried primary digested sludge used in previous microcosm experiments may also have lead to reduced density of organic material and consequent biochemical oxygen demand and redox over the GAEW benthos. In much the same way as straw has been postulated to change mixing in other studies (Koschorreck, Frömmichen et al. 2002) greenwaste is likely to act as a mixing barrier for the water body directly above the sediment surface. Scaling differences from microcosm to field experiment has also meant that the small greenwaste twigs used in the microcosm experiment may not have not scaled up to an equivalent surface area for bacterial communities and bed roughness resistant to mixing in this large-scale field experiment. Alternatively, the sudden slowing in pH increase may be due to exhaustion of organic carbon due to the low AMD:organic mass ratio in the experiment. Sediment analyses to examine sediment chemistry and mineralogy are planned over the 2008 Dry season (two years after organic dosing began) to test for this latter hypothesis of carbon limitation of sulfate reduction processes.

Much other research published to date has occurred in cool temperate areas of Europe and North America e.g., (Gammons, Drury et al. 2000; Frömmichen, Wendt-Potthoff et al. 2004). This study has demonstrated promise for remediation of AMD water pH and high metal concentrations by sulfate reduction with green waste and sewage as organic substrates in warmer climes such as Australia. However, treatment timescale may be long and a remaining challenge is for researchers and regulatory agencies need to maintain an open-mind to the variety of treatment options available across different geographical and economical regions for this large and growing environmental issue.

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