



Mineralogical and geochemical characterization of the Old Tailings Dam, Australia: Evaluating the effectiveness of a water cover for long-term AMD control



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ABSTRACT

Establishing a shallow water cover over tailings deposited in a designated storage facility is one option to limit oxygen diffusion and retard oxidation of sulfides which have the potential to form acid mine drainage (AMD). The Old Tailings Dam (OTD) located at the Savage River mine, western Tasmania contains 38 million tonnes of pyritic tailings deposited from 1967 to 1982, and is actively generating AMD. The OTD was constructed on a natural gradient, resulting in sub-aerial exposure of the southern area, with the northern area under a natural water cover. This physical contrast allowed for the examination of tailings mineralogy and geochemistry as a function of water cover depth across the OTD. Tailings samples ($n = 144$, depth: ≤ 1.5 m) were collected and subjected to a range of geochemical and mineralogical evaluations. Tailings from the southern and northern extents of the OTD showed similar AMD potential based on geochemical (NAG pH range: 2.1 to 4.2) and bulk mineralogical parameters, particularly at depth. However, sulfide alteration index (SAI) assessments highlighted the microscale contrast in oxidation. In the sub-aerial zone pyrite grains are moderately oxidized to a depth of 0.3 m (maximum SAI of 6/10), under both gravel fill and oxidized covers, with secondary minerals (e.g., ferrihydrite and goethite) developed along rims and fractures. Beneath this, mildly oxidized pyrite is seen in fresh tailings (SAI = 2.9/10 to 5.8/10). In the sub-aqueous zone, the degree of pyrite oxidation demonstrates a direct relationship with cover depth, with unoxidized, potentially reactive tailings identified from 2.5 m, directly beneath an organic-rich sediment layer (SAI = 0 to 1/10). These findings are broadly similar to other tailings storage facilities e.g., Fox Lake, Sherritt-Gordon Zn–Cu mine, Canada and Stekenjokk mine, Sweden where water covers up to 2 m have successfully reduced AMD. Whilst geotechnical properties of the OTD restrict the extension of the water cover, pyrite is enriched in cobalt (up to 2.6 wt%) indicating reprocessing of tailings as an alternative management option. Through adoption of an integrated mineralogical and geochemical characterization approach for tailings assessment robust management strategies after mine closure can be developed.

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1. Introduction

Uncontrolled sulfide oxidation can lead to the generation of acid mine drainage (AMD) typically characterized by metal-laden acid-sulfate waters (Jambor et al., 2003). Impacts of AMD on the environment can occur throughout the life-of-mine with many examples published in the scientific literature; e.g., Harris et al.

(2003), Edraki et al. (2005), Parbhakar-Fox et al. (2014), Candeias et al. (2014), Staebe (2015) and Myers (2016). In order to prevent sulfide oxidation and AMD generation in tailings storage facilities (TSFs), appropriate control strategies are needed. Widely used strategies aim to exclude one or more of the constituents required for oxidation; e.g., water, iron, oxygen, bacteria and sulfide minerals. Lottermoser (2010) suggested that the most effective control on oxidation rate is specifically reducing oxygen availability. This can be achieved using wet covers (e.g., Wilkinson et al., 1999; Eriksson et al., 2001; Adu-Wusu et al., 2001; Mian and Yanful, 2004; Yanful et al., 2004; Kachhwal et al., 2010) or dry covers

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(e.g., Nicholson et al., 1989; Johnson and Hallberg, 2005; Meer and Benson, 2007; Higgins, 2012) with a summary given in the *GARD Guide* (2014). However, the use of a single technology may prove ineffective, and in many cases a combination of technologies offer the best chance of success (Lottermoser, 2010).

When dealing with historic/legacy or abandoned tailings sites, a cost-effective management strategy is favored as the funds available to the site owner (either the mine operator, or the government) are likely limited (Pepper et al., 2014). In such cases, the use of a water cover should be considered. Recent studies by Yanful et al. (2004), Bjelkevick (2005), Kachhwal et al. (2010) and Moncur et al. (2015) demonstrate the use of shallow water covers in North America and Sweden. They are effective as the maximum concentration of dissolved oxygen in water is three orders of magnitude lower than the atmosphere with AMD formation retarded (Guskek and Figueroa, 2009). However, a minimum water cover depth must first be established. Generally, deep-water covers (>10 m; Yanful et al., 2004) are unfavorable due to concerns regarding the high capital expenditure, long-term engineering stability of the containment walls and the need for dike maintenance (Yanful et al., 2004). Therefore, determining the minimum depth of water will retard sulfide oxidation is critical to the design and success of such aqueous TSFs.

Many laboratory and field studies have investigated the short-term storage of tailings under shallow water covers e.g., ≤30 years (Li et al., 1997; Holmström et al., 1999; Yanful et al., 2000; Vigneault et al., 2001; Elberling and Damgaard, 2001; Romano et al., 2003; Samad and Yanful, 2005; Oxley et al., 2008; Awoh et al., 2013), and demonstrated reduction of sulfide oxidation at 0.5 m–2 m, based on the examination of oxygen diffusion and consumption, vertical mixing due to wind-induced wave activity, erosion and resuspension. Comparatively fewer studies have examined the reactivity of tailings submerged for several decades e.g., Jacob and Otte (2004), Moncur et al. (2015) who instead report shallower effective depths (i.e., from 0.5 m to 1 m). However, in these examples, tailings stored at water depths >1 m were not examined. Therefore, our understanding of how effective sub-aqueous tailings disposal is over time and at depth is limited. Only additional data from established TSFs will help build this knowledge. Motivated by this, our study presents results from the Old Tailings Dam (OTD), Savage River mine, Tasmania. Pyritic tailings were deposited into this sub-horizontal repository since the 1960s. Consequently, tailings in the northern portion have been permanently submerged underwater for at least three decades (i.e., on its closure). Specifically, this study focussed on: (i) assessing the geochemical and mineralogical characteristics across the OTD; and (ii) determining the effectiveness of the existing long-term water cover in restricting sulfide oxidation.

2. Site description

The Savage River mine is located in northwest Tasmania, Australia (Fig. 1a and b), approximately 400 km from Hobart (latitude and longitude of 41.49°S and 145.21°E respectively; 100–350 m elevation). A cool temperate rainforest climate is experienced, with mean maximum and minimum temperatures of 14.3 °C and 6.3 °C respectively (BOM, 2016). Mean annual rainfall for the Savage River area is approximately 1925 mm with minimum rainfall experienced in January (range: 7–200 mm). Pan evaporation rates during the summer months (December to February) are approximately 125 mm (BOM, 2016). The Savage River itself runs through the middle of the site before continuing south through the Savage River National Park (Fig. 1d).

Whilst Savage River magnetite deposits were first discovered in 1877, the first shipment of pellets was not made until 1968, as the

location was previously considered isolated and the magnetite sub-economic grade (Kent, 2013). Approximately 74 Mt of magnetite concentrate was produced from 179 Mt of ore averaging 41.8% Fe₃O₄ in 2006 (Calver et al., 2014). Approximately 2.4 Mt/year of pelletized iron ore are produced by this mine (Kent, 2013). These operations target a group of Early Paleozoic magnetite-rich lenses within a series of highly metamorphosed rocks of marble, schist and metabasic rocks (Hutchison and Brett, 2006; Kent, 2013). Mineralization is hosted by deformed Mg-rich sequences, comprising tremolite-actinolite ± talc ± serpentine ± Mg-chlorite ± dolomite zones, with minor magnetite occurring in mafic rocks and massive carbonates (Berry and Holm, 2014). The mineralization comprises magnetite >> pyrite ± apatite ± chalcopyrite and is massive to disseminated (Berry and Holm, 2014).

These pits, oriented north-south; comprise a strike length of approximately 6 km. The operation currently aims to produce an average of 2.5 Mt of magnetite concentrate per annum, which generates on average 11.6 Mt of waste rock and 2.8 Mt of tailings (Kent, 2013). Two tailings facilities are located on site, the decommissioned OTD and the active Main Creek Tailings Dam. Approximately 38 Mt of pyritic tailings were deposited into the OTD by end-of-pipe spilling into the southwest corner of the dam (Hassell, 2005). Subsequently, tailings were sprayed from a single spigot point from the dam wall allowing it to be raised in lifts (Thompson and Brett, 1996). This has concentrated coarser, heavier and more permeable material against the southern dam wall resulting in extensive tailings oxidation (Kent, 2013). However, as the geometry of the repository is sub-horizontal, the northern portion of the dam (termed the Northern Pond) is sub-aqueous, with water depth continually increasing towards the north (to approximately 10 m depth). In proximity to the Northern Pond the water table is close to the surface, with fluctuations related to seasonal variation in rainfall, however water level saturation data scarce (Hassell, 2005). Previous hydrogeological observations identify a drainage divide, with meteoric water in the southwest draining towards the south, and the remainder towards the northeast (Hassell, 2005). Water levels at the OTD were not recorded during or after tailings deposition had ceased; though reference has been made to permanent saturation of the Northern Pond since 1980 (Kent, 2013).

Anecdotal information suggests evidence of AMD as early as 1977, with high Fe and Cu concentrations measured. However, this was only confirmed in 1996 when the mine ceased operation with the OTD contributing 50% of the overall AMD load (pH 2.3 to 3.7) from the site, with the remainder from waste rock dumps (Kent, 2013). Water quality assessments in the Savage River and Main Creek downstream of the site showed that median Cu concentrations exceeded ANZECC (2000) guideline values by over 25 times over the value for soft waters (i.e., contains low concentrations of ions, e.g., Ca and Mg). Elevated Cu concentrations were the cause of aquatic ecosystem degradation downstream of the mine (Koehnken, 2007; Kent, 2013). It was estimated that AUD \$120 million was required to collect and treat AMD (Kent, 2013). However, later studies indicated higher rehabilitation costs, as this assessment only accounted for <50% of the site's Cu load (Ray, 2001; Koehnken, 2007). The state government established the Savage River Rehabilitation Project in 1997, a co-operative project between itself and the current mine operator (Grange Resources), to manage remediation of the legacy pollution (i.e., pre-1996) with a budget of AUD \$24 million. Their objective is to restore the Savage River and surrounding national park to that of a modified, but healthy ecosystem. The reduction of AMD from waste rock has been successful to date (Hutchison and Brett, 2006). However, the management of the OTD has proved challenging, therefore development of a feasible long-term strategy is critical to this environmentally sensitive area. Therefore, evaluating the effectiveness of

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