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RESEARCH ARTICLE

Five-year performance monitoring of a high-density polyethylene (HDPE) cover system at a reclaimed mine waste rock pile in the Sydney Coalfield (Nova Scotia, Canada)

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Abstract Cover systems are commonly placed over waste rock piles (WRPs) to limit atmospheric water and oxygen ingress and control the generation and release of acid mine drainage (AMD) to the receiving environment. Although covers containing geomembranes such as high-density polyethylene (HDPE) exhibit the attributes to be highly effective, there are few, if any, published studies monitoring their performance at full-scale WRPs. In 2011, a HDPE cover was installed over the Scotchtown Summit WRP in Nova Scotia, Canada, and extensive field performance monitoring was conducted over the next five years. A range of parameters within the atmosphere, cover, waste rock, groundwater and surface water, were monitored and integrated into a comprehensive hydrogeochemical conceptual model to assess (i) atmospheric ingress to the waste rock, (ii) waste rock acidity and depletion and (iii) evolution of groundwater and surface water quality. Results demonstrate that the cover is effective and meeting site closure objectives. Depletion in oxygen influx resulted in slower sulphide oxidation and AMD generation, while a significant reduction in water influx (i.e. 512 to 50 mm/year) resulted in diminished AMD release. Consistent improvements in groundwater quality (decrease in sulphate and metals; increase in pH) beneath and downgradient of the WRP were observed. Protection and/or significant improvement in surface water quality was evident in all surrounding watercourses due to the improved groundwater plume and elimination of contaminated runoff over previously exposed waste rock. A variably saturated flow and contaminant transport model is currently being developed to predict long-term cover system performance.

Keywords Acid mine drainage · Contaminant remediation · Geomembrane liner · Hydrogeochemistry · Environmental monitoring

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Introduction

Mining activities produce large volumes of waste rock which are typically deposited in large, partially water-saturated porous heaps or piles on the ground surface. Mine waste rock piles

(WRPs) can contain significant quantities of sulphidic minerals such as pyrite and pyrrhotite, and exposure of these reactive minerals to oxygen and water can cause a complex sequence of oxidation-reduction reactions that produce acid mine drainage (AMD) (Nordstrom et al. 2015). Characterized by low pH levels, high acidity, high sulphate and high concentrations of heavy metals and other toxic elements (Nieto et al. 2013; INAP 2014), AMD emanating from WRPs can cause severe environmental impacts, particularly on soil, water resources and aquatic communities (e.g. Amos et al. 2015; Galhardi and Bonotto 2016). The impacts of AMD can be minimized at three levels: primary prevention of the AMD process, secondary control involving prevention of AMD migration and tertiary control or collection and treatment of effluents (Kefeni et al. 2017).

AMD prevention techniques, aiming to minimize the influx of atmospheric oxygen and water percolation to the sulphidic mass, typically involve the installation of covers over the waste rock (e.g. Johnson and Hallberg 2005; Aubertin et al. 2016). The design of a cover system is typically site-specific, depending on climatic conditions, WRP surface geometry (slope angle and length) and site closure objectives. A wide range of cover system configurations exist, from single to multi-layer systems consisting of earthen materials, organic materials and/or geosynthetic materials (MEND 2004). Single-layer 'moisture store-and-release' cover systems, which are typically used in semi-arid and arid regions, act to minimize water percolation by maximizing near-surface storage of moisture with subsequent release by evapotranspiration (e.g. Scanlon et al. 2005).

Multi-layer cover systems can be used in all climates, utilizing either the capillary barrier concept to maintain a high degree of saturation (at least 85%) in one or more layers (e.g. Yanful et al. 1999), or a resistive "barrier" layer with high moisture retention properties (Benson et al. 2007). These covers act to minimize moisture influx, where stored water is released in the form of runoff or evapotranspiration (e.g. Yanful et al. 2003), and provide an effective oxygen-diffusion barrier as a result of the low diffusivity of oxygen in water (Yanful 1993). While some soil layers such as natural compacted till have been unable to maintain a high degree of saturation throughout the year, "enhanced" layers such as till ameliorated with bentonite and compacted clay liners have significantly increased water retention capacity (e.g. Ayres et al. 2003). However, the performance of these layers can be strongly affected by desiccation and freeze-thaw cycling (e.g. Albright et al. 2006).

Alternative cover designs that have been used to act as effective physical and chemical barriers against oxygen diffusion include low-sulphide tailings (e.g. Bussière et al. 2004) and organic wastes such as municipal sludge, paper mill sludge and wood waste (e.g. Peppas et al. 2000). Geosynthetic clay liners (GCLs), consisting of bentonite clay supported by geotextiles and held together by needling, stitching, or chemical adhesion, have also been used in cover systems. However, while GCLs have performed well as a moisture barrier (e.g. Benson et al. 2007), their effectiveness as an oxygen-diffusion barrier remains questionable due to their short diffusion path length and susceptibility to desiccation (e.g. Rowe 2012).

Geomembranes (GM) such as high-density polyethylene (HDPE) are widely used as highly effective barriers to liquids and gases with the prime example being their extensive application in landfill liner systems. At mining sites, GMs have been successfully used for liquid containment, as basal liners in heap leach pads, and for tailings impoundments (Lupo and Morrison 2007). Despite this widespread use, HDPE liners have seen limited use in WRP cover systems. While pristine HDPE liners exhibit excellent attributes, in situ performance can be affected by various elements in the short term, such as wrinkles and/or defects (e.g. Rowe 2012), and in the long term by ageing (e.g. Rowe et al. 2009, 2014).

HDPE liner installation has a significant impact on the development of wrinkles, defects and deformations. Chappel et al. (2012) demonstrated that wrinkles develop rapidly (i.e. 1 day) in HDPE exposed to solar heating, suggesting that an overlying layer should only be placed over the HDPE at certain times/temperatures (Rowe et al. 2012). All installed HDPE liners have some level of defects with numerous analytical methods established to estimate leakage rates through defects (e.g. Giroud et al. 1992; Rowe 2012). However, even assuming as many as 15–20 defects per hectare, HDPE liners still provide a highly effective barrier layer (Rowe 2012; Meiers and Bradley 2017). Deformations and punctures can occur following placement and compaction of overlying mineral particles; while geotextile fabrics can be used to protect the HDPE (e.g. Thiel and Smith 2004; Rowe and Rimal 2008), they can weaken the interface shearing resistance on top of the HDPE (Datta 2010). Therefore, cover systems containing different geosynthetic layers should be assessed and designed to prevent pile instability, particularly along steep slopes (Briançon et al. 2002).

Significant research has been conducted to address questions regarding the long-term performance and service life of HDPE (e.g. Gulec et al. 2004; Rowe and Ewais 2015). Site-specific environmental elements, particularly service temperature, strongly influence long-term HDPE ageing, with higher temperatures resulting in shorter service life. The service life of covered (unexposed) HDPE was shown to be 450, 265 and 70 years at 20, 25 and 40 °C, respectively, which is significantly longer than exposed HDPE (Sangam and Rowe 2002; Koerner et al. 2011). Rowe et al. (2009) estimated that the service life of HDPE immersed in leachate was at least 700 years at 20 °C and 150 years at 35 °C, noting that the service life would be even greater if exposed to unsaturated soil instead, as shown by Sangam and Rowe (2002).

In terms of in situ performance, good installation practices and quality control will minimize the development of wrinkles, defects and punctures to maximize short-term performance. Based on the aforementioned studies, HDPE liners can maintain performance over a long service life, even when used as liners at the base of solid waste landfills and heap leach facilities where they are exposed to leachates and high pressures. In WRP cover systems, HDPE liners would be exposed to more moderate conditions and overlain with geotextile fabric and unsaturated soil. Therefore, it is evident that HDPE liners can provide an effective, long-term barrier in WRP cover systems, particularly in more humid climates where service temperatures are typically less than 30 °C. Despite this potential, very few HDPE cover systems have been applied at WRPs (e.g. Meiers et al. 2011) with no published studies on in situ cover performance monitoring in relation to site remedial objectives.

Field monitoring is a critical component of successful remediation to ensure the cover is meeting design expectations and remedial goals, particularly in the early stages where changes in in situ properties can be significant. This collected data can then be used to develop numerical models to predict the long-term geotechnical, hydrogeological and geochemical behaviour of the reclaimed WRP (e.g. Molson et al. 2005). While some studies have monitored the variation of a single parameter such as moisture content within the cover and shallow waste rock (e.g. Weeks and Wilson 2005), others have monitored a range of parameters, including water percolation, soil temperature, matric suction, gas concentrations and moisture content (e.g. O'Kane et al. 1998; Adu-Wusu and Yanful 2006). To a lesser extent, water quality evolution in environmental receptors has also been monitored to obtain an indirect indicator of remedial performance (e.g. Gibert et al. 2013). It is noted that a number of the aforementioned studies only conducted monitoring on small-scale test cover plots (e.g. Ayres et al. 2003; Adu-Wusu and Yanful 2006). In addition, few, if any, studies have simultaneously monitored parameters within the WRP and the surrounding environmental receptors.

The objective of this study was to conduct comprehensive in situ monitoring of the reclaimed Scotchtown Summit WRP, which was overlain with an engineered cover system containing a HDPE liner. An extensive range of meteorological, hydrogeological and geochemical data were collected over a five-year period and integrated into a hydrogeochemical conceptual model to assess (i) atmospheric water and oxygen influx, (ii) waste rock acidity, (iii) AMD generation and release rates from the WRP and (iv) evolution of downgradient groundwater and surface water receptors.

Materials and methods

Study site

Waste rock pile

The Sydney Coalfield in Nova Scotia, Canada, is the oldest mined coal field in North America, with underground mining occurring from the early 1700s to the early 2000s (Shea 2009). These historic mining operations included more than 50 underground mines, producing over 500 million tonnes of coal. In 2001, Enterprise Cape Breton Corporation (ECBC) implemented a programme for remediating and closing former mine sites throughout the coal field with Public Works and Government Services Canada (PWGSC) providing project management (PWGSC 2013). This programme included the remediation of numerous WRP sites, including the Summit WRP located in Scotchtown, Nova Scotia, approximately 15 km north of Sydney. Figure 1 presents a plan view of the WRP and surrounding area which is referred to as the Scotchtown Summit Study Area (SSSA).

The Summit WRP was used for the placement of coal waste rock from nearby collieries between 1911 and 1973 (no mining activity took place in the SSSA). Between 1981 and 1987, the waste rock was reprocessed by Selminco to recover residual coal. This increased the footprint of the WRP threefold with the waste rock spread thinly over 44 ha. Reclamation and remediation of the pile occurred between late 2009 and late 2011. Prior to cover system installation, northern and southwestern areas of the waste rock were excavated and placed on top of the pile resulting in a more uniform shape and a footprint reduction to 37 ha. The total volume of waste rock is approximately 2,550,000 t (1.5 million m^3). The slopes range between 1 and 10% on top of the pile and between 4 and 20% on the sides, indicating a relatively flat WRP.

Climate

The SSSA is located in a seasonally humid region and is classified as humid continental under the Köppen climate classification. Based on long-term meteorological data recorded at the nearby Sydney Airport, mean annual precipitation (PPT) and potential evaporation (PE) are approximately 1500 and 650 mm, respectively (Environment Canada 2017). PPT exceeds PE for the majority of the year, creating a surplus of water that would need to be managed by the cover system.

Geology and hydrogeology

The SSSA comprises a number of hydrostratigraphic units, as illustrated by the cross section in Fig. 2a (A-A in Fig. 1). The waste rock fill consists of iron-stained shales, sandstone and siltstone, mixed with coal fines, and ranges in thickness from 1.5 to 10 m with the thickest deposits near the centre of the WRP. Glacial till underlying the WRP is grey or brown sand to silty sand and ranges in thickness from 2 to 7 m. In some areas, there is a thin layer of discontinuous organic soil at the base of the WRP, between the waste rock and till. The till is underlain by bedrock of the Lower Morien Group which primarily consists of fine- to medium-grained sandstone, with some interbedded siltstone and thin coal seams. The key geotechnical properties of each unit are shown in Table 1.

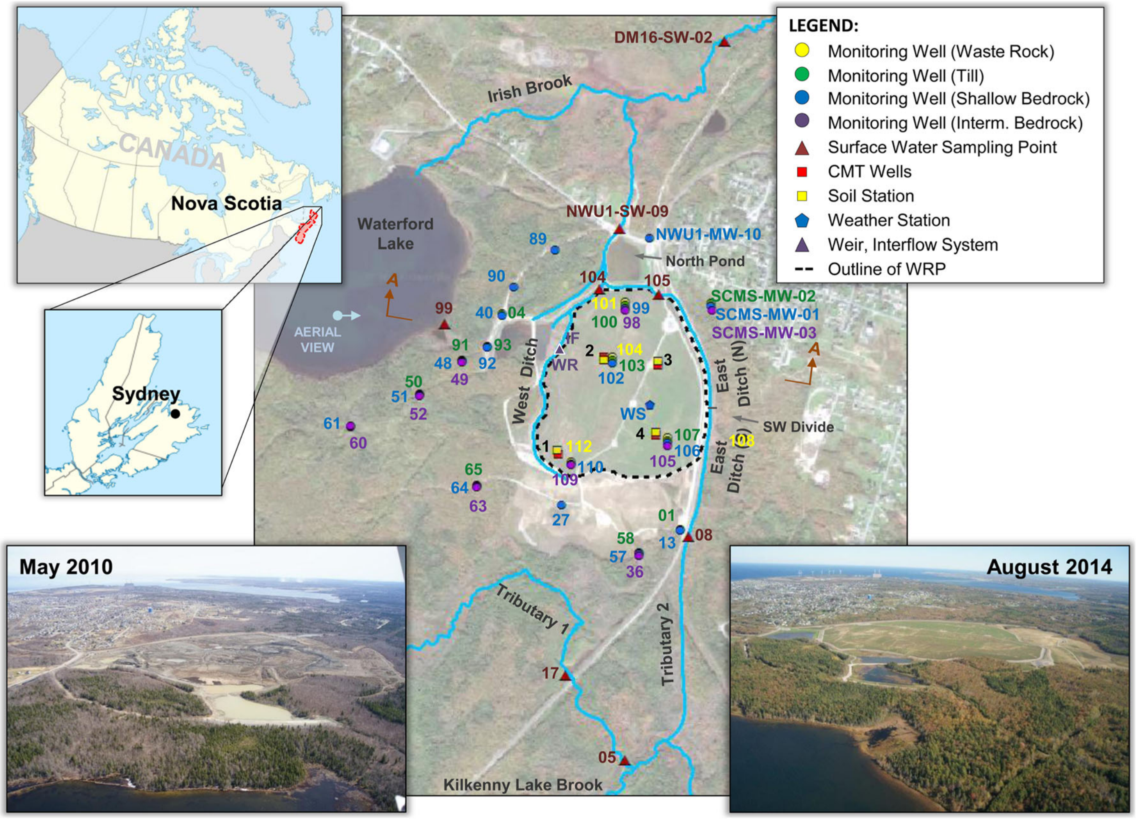

Fig. 1 Plan view (main) of the Scotchtown Summit WRP site, indicating the location of key environmental receptors and performance monitoring components (unless indicated, all monitoring well and surface water sampling point IDs are preceded with SSSA-MW and SSSA-SW, respectively). Oblique aerial photographs of the pile show the WRP before (May 2010) and after (August 2014) cover installation (photo source: Stantec Consulting)

The SSSA straddles a drainage divide between Irish Brook watershed to the north and Kilkenny Lake Brook (KLB) watershed to the south (Fig. 1). The WRP west, north and east(north) perimeter ditches drain into North Pond and subsequently northwards into Irish Brook, while the east(south) perimeter ditch, Tributary 1 and Tributary 2 drain southwards into KLB. Groundwater flows from the topographic high northeast of the WRP towards and through the WRP, and then flows predominantly towards regions to the west, northwest and south of the WRP, corresponding to the locations of the primary surface water receptors: Waterford Lake, Irish Brook and KLB (see Fig. 1). Waterford Lake is the primary municipal drinking water supply for Scotchtown and adjacent areas so it is a critical environmental receptor.

Site closure

Closure objectives

The key closure design criteria and objectives for the site were: (i) protection of water quality in Waterford Lake (a drinking water supply), (ii) protection of the two tributaries

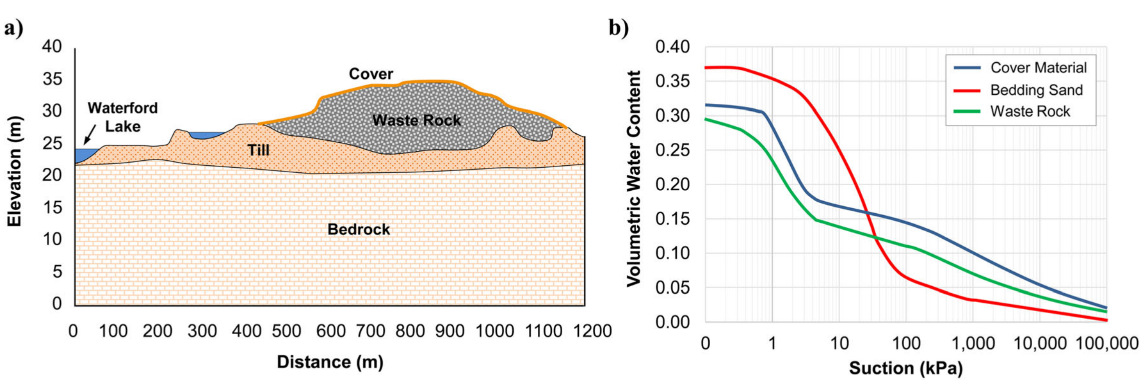

Fig. 2 **a** Geological cross section A-A through the site. **b** Moisture retention curves for the till cover material, bedding sand and waste rock

Table 1 Geotechnical properties of each hydrostratigraphic unit at the Summit WRP site

Unit	Thickness (m)	Hydraulic cond. (m/s)	Porosity (−)	Particle size dist. (%)[a]	Atterberg limits		
					LL	PL	PI
Till cover	0.5	4.6×10^{-7}	0.37	32/41/27	21	16	5
Bedding sand	0.15	1.0×10^{-7}	0.30	–	–	–	–
Waste rock	1.5–10	1.0×10^{-6}	0.35	46/38/16	24	20	4
Till	2–7	6.5×10^{-6}	0.25	30/39/31	19	14	5
Bedrock	–	3.0×10^{-5}	0.15	–	–	–	–

LL liquid limit, *PL* plastic limit, *PI* plasticity index

[a] Particle size distribution percentages are listed as 'gravel'/'sand'/'silt/clay'

to KLB with respect to fish habitat and (iii) improvement in the quality of the SSSA waters discharging to Irish Brook.

Closure options

A number of potentially viable closure options were identified that would satisfy all design criteria. While some options included the installation of various AMD treatment systems around the perimeter of the WRP to protect the three key watercourses, other options included the complete removal of the AMD source by relocating the waste rock to an alternative site. In addition to being extremely costly, these options do not remove the AMD problem, instead treating subsequent AMD contamination or transferring the problem elsewhere. The best option involved the prevention of AMD by significantly reducing water infiltration and limiting oxygen ingress.

Installed cover system

Between 2010 and 2011, an engineered cover system was installed over the WRP. A 0.15 m thick layer of uniform bedding sand was first placed over the rough waste rock surface. This bedding layer was overlain by a 60 mil (1.5 mm thick) HDPE GM, which is the most commonly used GM at mine sites (Thiel and Smith 2004). A protective layer of geotextile fabric was placed over the HDPE; the impact due to reduced interface shear resistance was not expected to be significant due to the low pile slopes (e.g. Briançon et al. 2002). The geotextile was overlain with a 0.5 m thick layer of till material. As shown in Table 1, the till is classified as a material with only slight plasticity (plasticity index < 7) and is expected to have a reasonable resistance to shear sliding and failure of the slope stability.

The in situ saturated hydraulic conductivity was measured with a Guelph permeameter which indicated a geometric mean of 4.6×10^{-7} m/s. The till material was hydro-seeded to establish a sustainable vegetative canopy and provide a geomorphically stable landform. This layer was constructed to 0.5 m thickness to allow sufficient space for vegetation rooting above the HDPE liner. Moisture retention curves (MRCs) for the cover, bedding sand and waste rock materials are presented in Fig. 2b. The MRC for the cover material and the waste rock are bi-modal and relatively similar; however, the cover material has more moisture retention in both the coarser and finer ends of the curve. The MRC for the bedding sand is relatively clean, marked by a sharp decrease in moisture content with an increase in suction above the air entry value.

Figure 1 presents oblique aerial photographs of the pile before (May 2010) and after (August 2014) installation of the cover system. This cover system was designed to significantly reduce the water influx to ~ 50 mm/year (assuming 20 defects per hectare in the cover), although predictive modelling demonstrated that larger water influx rates would still satisfy design criteria. The estimated cost of the installed cover was \$33 per square metre of the WRP (bedding sand \$2/m^2; HDPE \$15/m^2; geotextile \$4/m^2; till cover \$12/m^2). In comparison, a simple vegetative cover that has been installed at a nearby WRP cost \$12/m^2.

Field monitoring programme

Various field instrumentation was installed to enable performance monitoring under site-specific climatic conditions. All instrumentation was supplied by Campbell Scientific. Table 2 presents a summary of the various monitoring system elements and parameters measured.

Meteorological monitoring

A meteorological station measures rainfall, air temperature, relative humidity (RH), wind speed and direction, barometric pressure, net radiation and snowpack depth at the pile. Rainfall is recorded with a CS700 tipping bucket rain gauge (± 0.2 mm), while air temperature and RH are measured with a HMP45C probe (± 1% RH; ± 0.5 °C). A 05106-10 wind monitor is used to measure wind speed and direction (speed ± 0.3 m/s; direction ± 3°). Net radiation is measured with a NR-LITE2 net radiometer, while a 61302V sensor is used to measure barometric pressure (± 30 Pa). In addition, snowpack depth is recorded with a SR50A sonic ranging sensor

Table 2 Summary of the Summit WRP site monitoring elements and parameters measured

Monitoring element	Number	Parameters	Material
Soil monitoring station	4	In situ temperature, matric suction, volumetric moisture content, O_2/CO_2 pore-gas concentrations, pore-water pressure	Cover and waste rock
CMT wells	4	O_2/CO_2 pore-gas concentrations, differential pressure, temperature, groundwater level, water chemistry[a]	Waste rock and groundwater
Meteorological station	1	Rainfall, air temperature, relative humidity, wind speed and direction, barometric pressure, snowpack depth, net radiation	Atmosphere and cover
Weir	1	Surface run-off from WRP	Cover
Interflow collection system	1	Lateral flows through cover	Cover
Groundwater monitoring wells	42	Groundwater level, water chemistry[a]	Groundwater
Surface water sampling points	8	Surface water flow rate, water chemistry[a]	Surface Water

[a] Includes pH, acidity, alkalinity, sulphate, total/dissolved metals, EC, turbidity, TIC/TOC, TDS and TSS

(± 1 cm). Snow surveys were conducted at designated cover locations throughout the winter to determine the depth, density and water equivalent of the snowpack on the cover. Meteorological parameters are measured every 60 s, with hourly and daily averages stored for data collection.

Waste rock pile

Four soil monitoring stations (SMS) were installed across the WRP to continuously monitor moisture and temperature conditions within the cover and shallow waste rock. CSI Model 229-L thermal conductivity sensors measure temperature and matric suction (± 0.5 °C; ± 1 kPa), while CSI Model CS616-L time domain reflectometry (TDR) sensors measure volumetric moisture content (± 0.1%). These sensors were installed along single-depth profiles at each station, with sensor depths of 0.05, 0.10, 0.20, 0.30, 0.40, 0.49, 0.60, 0.90 and 2.15 m, as illustrated in Fig. 3. Each parameter is automatically recorded every 3 h. Pore-gas sampling ports were also installed at three depths, corresponding to the till cover (0.40 m), just below the HDPE (0.60 m) and waste rock (0.90 m). A Nova portable gas analyzer (± 0.1%) was connected to the sampling line for each port to manually measure pore-gas concentrations every month. A OTT PLS sensor was used to record pore-water pressure in the till above the HDPE liner at SMS-4 (note in this area, till cover is ~ 0.6 m thick). A CSI Open Path Eddy Covariance system was used on the cover to measure actual evapotranspiration (AET).

Four continuous multi-channel tubing (CMT) wells were installed through the entire depth of the waste rock and completed in the shallow bedrock. Each CMT well is instrumented with six CSI 109B-L temperature probes and CMT. The CMT provides six sampling depths, ranging from 1.4 to 14.9 m, with differential pressure automatically monitored at two depths with Setra Model 264 sensors and pore-gas concentrations manually measured at the remaining depths (see Fig. 3). Groundwater levels were collected monthly at the base of the CMT wells, along with water samples for geochemical analysis.

A 60° V-notch weir was used to continuously monitor surface water runoff (R). A sonic ranger and water conductivity and temperature probe were used to obtain hourly stage, ambient temperature and discharge water temperature and conductivity measurements. An interflow collection system quantifies lateral percolation (LP) throughout the cover. The location of the weir and interflow system are indicated in Fig. 1.

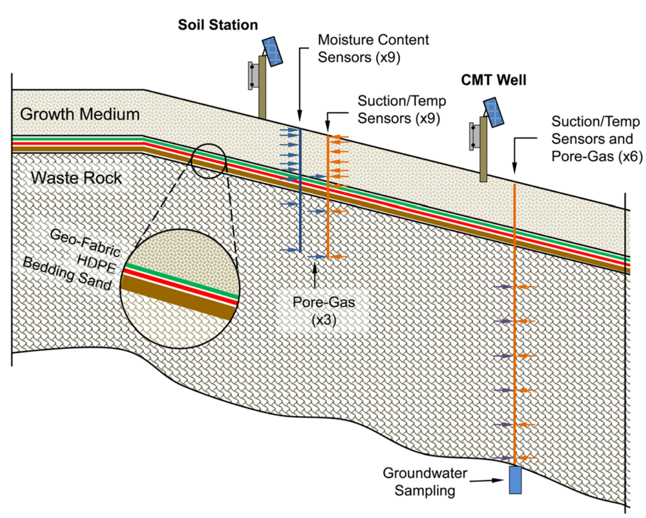

Fig. 3 Cross section showing the cover system profile and the Soil Monitoring Stations and CMT wells and associated parameters measured (reproduced from Meiers et al. 2011)

Environmental receptors

A total of 42 monitoring wells, screened within each geologic unit, are located throughout the site for monitoring the spatial and temporal distribution of groundwater levels and water geochemistry. Figure 1 indicates the location of the monitoring wells, with three wells located upgradient of the pile, 14 wells located within the pile footprint and 25 wells located in downgradient regions. The monitoring wells located outside the pile footprint were installed and monitored at various times between March 2003 and September 2011. The 14 wells within the pile footprint were installed alongside the cover system in 2010/2011. All 42 monitoring wells were sampled each Spring, Summer and Fall between August 2014 and July 2016. Surface water sampling was conducted at eight locations encompassing each watercourse in the SSSA: WRP perimeter ditches, Irish Brook, Waterford Lake, Tributary 1 and Tributary 2 which drains to Kilkenny Lake Brook (see Fig. 1).

Landform stability and vegetation

Site inspections, including erosion surveys, vegetation surveys and aerial inspections, were conducted at the site annually to monitor the stability of the landform and evolution of the vegetative cover.

Hydrogeochemical conceptual model

The collected hydrogeological and geochemical data were used to develop a conceptual model to assess atmospheric ingress, AMD generation within the pile and the evolution of AMD impacts at environmental receptors.

Atmospheric ingress

The rate of atmospheric ingress controls the generation and release of acidity from the WRP.

1. Water flux

The primary mechanism for generating water flux (net percolation, NP) through a cover system containing a GM layer is leakage through defects. Leakage through a GM is influenced by the head of water, size and number of defects, hydraulic conductivity of the underlying material and quality of the contact between the GM and underlying material (Meiers and Bradley 2017). The following analytical solution was established by Giroud et al. (1992) to estimate leakage through a circular defect:

$$Q = C_{qo} \cdot A^{0.1} \cdot h^{0.9} \cdot K_{sat}{}^{0.74}, \tag{1}$$

where Q is the leakage rate [m^3/s], C_{qo} is the contact quality factor (0.21 for good and 1.15 for poor) [−], A is the area of the defect [m^2], h is the pressure head [m] and K_{sat} is the saturated hydraulic conductivity of the underlying material [m/s]. Equation (1) was used in this study to estimate NP through the HDPE liner. The pressure head was determined by the OTT PLS pore-water pressure sensor at SMS-4. From Table 1, the K_{sat} for the bedding sand is 1×10^{-7} m/s. Poor contact was assumed between the HDPE liner and the underlying bedding sand. The frequency of defects was assumed to be 20 per hectare with a diameter of 10 mm which is similar to previous studies (e.g. Giroud et al. 1992; Meiers and Bradley 2017).

Annual water balances were developed to evaluate water dynamics in the cover system and were composed of the following components (e.g. Meiers et al. 2009):

$$\mathrm{PPT} = \mathrm{R} + \mathrm{AET} + \Delta \mathrm{S} + \mathrm{LP} + \mathrm{NP}, \tag{2}$$

where ΔS is the change in storage (water and snow) within the cover material [mm]. NP is calculated as the residual of Eq. (2) and is used to validate NP rates estimated in Eq. (1).

2. Oxygen flux

Oxygen ingress into the waste rock controls the oxidation rate of sulphide minerals that generate additional acidity in the WRP. Ingress of oxygen across the cover system to the underlying waste can occur through molecular diffusion and dissolved oxygen within percolating water. Advective flux due to barometric pumping through the two-exhaust vents installed at the pile is negligible. Diffusive oxygen flux, J_{diff}, which is driven by the concentration gradient across the HDPE liner, was estimated by Fick's Law which has been widely used to evaluate flux through various covers (e.g. Aubertin et al. 2000):

$$J_{diff} = -D^{\Delta C}/_{\Delta x}, \tag{3}$$

where D is the diffusivity coefficient [m^2/s], ΔC is the change in oxygen concentration [mol/m^3] and Δx is the thickness of the HDPE [m]. The diffusivity coefficient specific to oxygen and HDPE is 1.7×10^{-11} m^2/s (Hedenqvist 2005). Dissolved oxygen will infiltrate into the waste rock through NP. The solubility of dissolved oxygen [mg/L] in water is estimated by (Muralikrishna and Manickam 2017)

$$C_{O_{2_{diss}}} = {}^{(P-p)\times 0.678}/_{35} + t, \tag{4}$$

where P is the measured barometric pressure [torr], t is the measured water temperature [°C] and p is the saturated vapour pressure [torr].

The total oxygen flux can be converted to an acidity load on the basis of H_2SO_4 equivalent (mol/m^2/year). It is assumed that (i) all sulphide is present as pyrite; (ii) oxygen is available for pyrite oxidation and (iii) no oxygen is consumed by the oxidation of organic material in the cover system or carbonaceous material in the waste rock, thus creating the maximum amount of acidity per mole of oxygen. The overall summary reaction of pyrite weathering to form AMD is shown in Eq. (5), where oxidation of 1 mol of pyrite (FeS_2) generates 2 mol of sulphuric acid (H_2SO_4) per 3.75 mol of oxygen (O_2) present (Nordstrom et al. 2015):

$$FeS_2 + {}^{15}/_{4}O_2 + {}^{7}/_{2}H_2O \rightarrow Fe(OH)_3 + 2H_2SO_4 \tag{5}$$

Waste rock acidity

Acid base accounting (ABA), which is the most commonly used static test method for characterizing waste rock, was performed on waste rock samples extracted from the pile. In November 2006, 18 rock samples were collected with a stainless steel trowel from 18 test pits distributed across the WRP landform and excavated to depths ranging from 3.6 to 6.2 m, thereby providing high spatial resolution. In February 2011, two samples (one shallow and one deep) were collected during the drilling of each CMT well.

The waste rock samples were characterized with the widely used Modified Sobek Method (e.g. Downing 2014). Total sulphur is determined with a LECO sulphur analyzer, while sulphate-sulphur is determined by hydrochloric acid (HCl) extraction (sulphate-sulphur equals total sulphur minus acid extractable sulphur). Sulphide-sulphur was determined by difference. Acid generation potential (AGP) is based on the sulphide-sulphur content (wt%). Acid neutralization potential (ANP) uses fizz test and HCl that dissolves carbonates and reactive silicate minerals. The most important acid-producing minerals in this respect are iron-containing sulphides, especially pyrite, while the carbonates, especially calcite ($CaCO_3$), are the main rapid neutralisers. It is assumed that stored acidity is mainly related to sulphate-sulphur since the contribution of other sources like gypsum and hexahydrite minerals is very low due to their low dissolution.

AMD impacts to the receiving environment

The impact of AMD seepage on the receiving environment was evaluated by continuously monitoring key geochemical parameters at all groundwater monitoring wells and surface water sampling locations. These parameters included sulphate, pH, total dissolved solids (TDS) and key dissolved metal concentrations: iron (Fe), aluminium (Al) and manganese (Mn). Sulphate is a key AMD indicator due to its high persistence in groundwater, often being used as a geochemical tracer.

In AMD studies, an estimate of acidity can be obtained from pH and the sum of the milliequivalents of the dissolved metals as follows (e.g. Kirby and Cravotta 2005; Park et al. 2015):

$$Acidity_{calc} = 50 \cdot \left\{ 2 \cdot [\mathrm{Fe}]/56 + 3 \cdot [\mathrm{Al}]/27 + 2 \cdot [\mathrm{Mn}]/55 + 1000 \cdot 10^{(-\mathrm{pH})} \right\} \tag{6}$$

where 50 is the equivalent weight of $CaCO_3$, which converts the acidity in milliequivalents per litre into milligrams per litre of $CaCO_3$ equivalent. Studies have demonstrated that the acidity from Eq. (6) is in good agreement with measured acidities over a broad range of pH values (e.g. Kirby and Cravotta 2005).

The evolution of water quality emanating from the WRP is determined from wells within the pile footprint. Concentrations of calculated acidity are then integrated with the basal seepage rate to determine the loading of AMD from the WRP to groundwater. Periodic monitoring of groundwater and surface water at downgradient locations indicates the evolution of AMD impacts on key environmental receptors. In particular, surface water quality at DM16-SW-02, SSSA-SW-99 and SSSA-SW-05 is used to assess whether key closure objectives are being met at Irish Brook, Waterford Lake and KLB, respectively.

Results and discussion

Meteorological conditions

Figure 4a presents the daily PPT between January 2012 and December 2016. The WRP experienced a range of conditions, with the highest PPT in the fall and winter periods. The annual cumulative amount of PPT for 2012–2016 was 1299, 1507, 1616, 1531 and 1675 mm, respectively. Snow accounted for 16, 26, 18, 29 and 23% of PPT each year.

Atmospheric ingress

Water influx

The mean volumetric moisture content measured at the four SMSs between January 2012 and December 2016 are presented in a two-dimensional contour profile in Fig. 4b. As shown, moisture content in the cover material is highest (blue regions) in spring and fall and correlates to periods of high PPT and low PE. Moisture contents are lowest in the summer due to lower PPT and higher PE. Frozen soil conditions were indicated by soil temperature measurements (not shown) with corresponding moisture content measurements considered inaccurate and blanked in Fig. 4b. While frozen soil conditions occurred throughout the cover depth each winter between

2012 and 2015, only the top 5 cm of soil was frozen in 2016 as shown in Fig. 4b.

The head of water continuously measured by the OTT PLS sensor at SMS-4 is presented in Fig. 4c. As expected, variations in pressure head correspond to variations in moisture content, with periods of pressure head occurring during periods of increased moisture content (i.e. spring and fall). Extended periods (> 3 months) occur in the late fall and early winter where the pressure head is equal to or near the thickness of the cover system profile. This demonstrates the need for a drainage layer above the HDPE liner. The pressure head is low in winter and summer periods each year, except for the winter in 2016 where the pressure head remains high. This corresponds to the warmer soil conditions and high moisture contents in Fig. 4b.

Assuming 20 defects/ha in the HDPE liner, was estimated by integrating the measured pressure head with Eq. (1). The annual cumulative NP is also plotted in Fig. 4c. NP increases during periods of pressure head, with significant increases occurring in spring, late fall and early winter. Annual NP is relatively similar for 2012–2015, but the warmer cover conditions in winter 2016 resulted in a significant increase in NP and a larger annual NP.

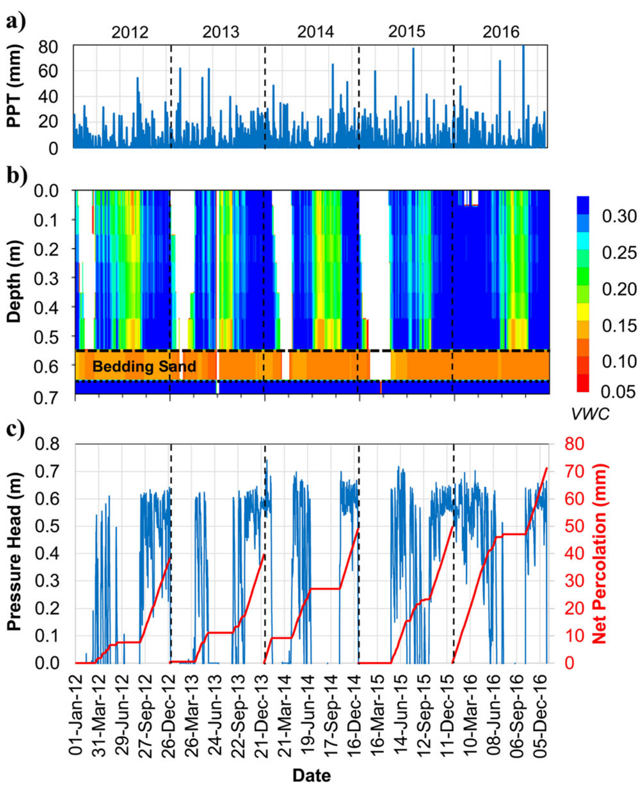

Fig. 4 **a** Daily PPT at the Summit WRP. **b** Mean volumetric moisture content at the four SMSs. **c** Pressure head above the HDPE liner and estimated NP (assuming 20 defects/ha) between January 2012 and December 2016. The TDR sensor detects only unfrozen water, so frozen conditions are indicated by the blank (white) regions in **b**

Annual water balances were developed between 2012 and 2016 using Eq. (2). To illustrate a developed water balance, Fig. 5 plots the cumulative flux for each component in 2012. Total PPT is 1299 mm with 3 mm of water stored in the snow (ΔSS) over the winter months and 17 mm of water stored in the cover (ΔWS). R was estimated as 725 mm with another 39 mm leaving the WRP through LP within the cover. AET was estimated as 477 mm, leaving a residual (NP) of 47 mm which is similar to the NP of 38 mm estimated by assumed defect leakage. Table 3 summarizes the results of each annual water balance. The absence of a drainage layer above the HDPE resulted in high runoff volumes from the cover and limited lateral drainage above the HDPE.

The mean infiltration rate, equal to ~ 3% PPT, demonstrates that the cover is effective at reducing water ingress. It is substantially lower than NP rates between 15 and 25% through single-layer soil covers and similar to NP rates through multi-layer soil covers maintaining a high degree of saturation (i.e. 90% or higher) (e.g. Yanful 1993; O'Kane et al. 1998). Furthermore, this 'post-cover installation' NP exhibits a substantial reduction from the 'pre-cover installation' NP estimate equal to 34% PPT (King et al. 2003).

The evolution of water levels since cover installation suggests that drain-down of water perched within the pile has already ceased. Therefore, it is assumed that the seepage rate of water transporting acidity from the pile is equal to NP.

Oxygen flux

The mean pore-gas concentrations recorded at each SMS between 2012 and 2016 are presented in Fig. 6a, with the error bars indicating one standard deviation. Mean oxygen concentrations are near atmospheric above the HDPE liner (ranging from 17 to 20%) and significantly depleted at depths below the HDPE liner (0.8 to 4.8%). As expected, carbon dioxide concentrations have increased at depths below the liner. The

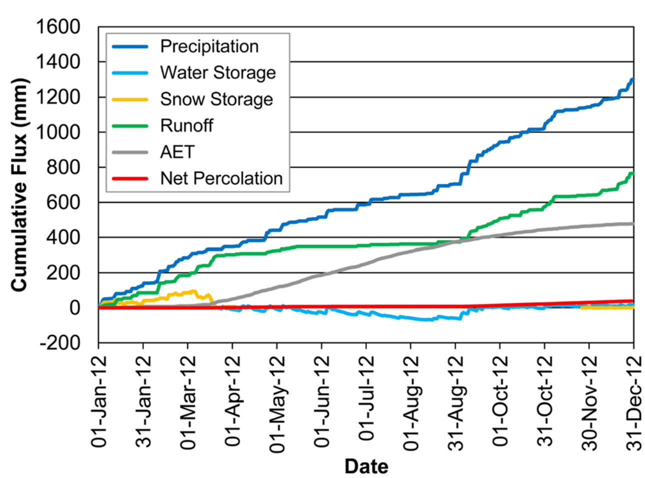

Fig. 5 Cumulative flux of each water balance component 2012

depletion of oxygen within the waste rock demonstrates that oxygen flux is being limited by the cover as desired. Integrating the measured oxygen concentration gradient across the HDPE liner with Eq. (3), the diffusive oxygen flux is determined.

The barometric pressure and soil temperature continuously monitored at the WRP were integrated with Eq. (4) to estimate the dissolved oxygen concentration in meteoric water infiltrating the cover system. This dissolved oxygen concentration was combined with NP to obtain the flux percolating into the waste rock. Figure 6b plots the variation in diffusive and dissolved flux between January 2012 and December 2016. The diffusive flux exhibits a similar trend to the oxygen concentration difference across the HDPE, while the dissolved flux varies similar to NP. The mean annual diffusive and dissolved fluxes between 2012 and 2016 are summarized in Table 3, with the total mean flux equal to 2.52 mol/m^2/year. This limited oxygen ingress indicates that the cover system is an effective oxygen barrier (e.g. Yanful 1993; Demers et al. 2008).

The total oxygen flux is converted to an acidity load on the basis of H_2SO_4 equivalent (mol/m^2/year) using Eq. (5). Since 2 mol of acid is generated per 3.75 mol of oxygen, then the calculated mean flux of 2.52 mol O_2/m^2/year will generate 1.34 mol H_2SO_4/m^2/year. Using this acidity rate, the total extrapolated catchment acidity load over the area of the cover system is 49.58 t of acidity per year (as $CaCO_3$).

Waste rock acidity

A summary of the ABA results is shown in Table 4. Paste pH for the 2011 waste material samples ranged from 3.0 to 4.1, with a mean value of 3.44. Total sulphur concentrations ranged from 0.16 to 1.61 wt%, with sulphate-sulphur ranging from 0.08 to 1.01 wt% and sulphide-sulphur ranging from 0.08 to 0.60 wt%. ANP ranged from − 14.71 to 0.25 kg $CaCO_3$ equivalent acidity per tonne of waste rock (kg $CaCO_3$/t), AGP ranged from 2.44 to 18.88 kg $CaCO_3$/t, while all net neutralization potential (NNP) values were negative. Due to the low paste pH values, negative ANP values and very low ANP/AGP ratios, the waste rock is considered highly acid generating. Figure 7 presents a plot of ANP/AGP versus paste pH, highlighting that all waste rock samples are potentially acid generating (ANP/AGP ratio << 1) (INAP 2014). Comparison of the 2006 and 2011 data, with no cover system over the waste rock between these sampling times, demonstrates how quickly the waste rock oxidized over time. The volume of sulphide-sulphur decreased from 0.51 to 0.35 wt%, and the sulphate-sulphur increased from 0.27 to 0.54 wt%.

Waste rock acidity exists in two forms: stored acidity or potential acidity. Stored acidity is acidity readily available for transport from the pile, while potential acidity first requires oxidation of sulphide minerals to create generate additional stored acidity. The mean AGP in 2011 was 10.95 kg $CaCO_3$/t, resulting in 27,925 t of potential acidity. Based on the mean sulphate-sulphur content of 0.54 wt% in 2011, 16.90 kg $CaCO_3$/t resulted in 43,088 t of stored acidity.

AMD impacts to the receiving environment

Groundwater flow system

Using the mean groundwater levels measured at the 42 monitoring wells and four CMT wells, the piezometric surface and flow directions in the till, shallow bedrock and intermediate bedrock units were developed as shown in Fig. 8. In all units,

Table 3 Water and oxygen flux components between 2012 and 2016

Component	2012		2013		2014		2015		2016		Mean	
	mm	%[a]	mm	%	mm	%	mm	%	mm	%	mm	%
Water												
PPT	1299	–	1507	–	1616	–	1531	–	1675	–	1526	–
R	725	56	822	55	1165	72	865	57	1034	62	922	61
LP	39	3	44	3	46	3	45	3	49	3	45	3
AET	477	37	473	31	460	28	460	30	464	27	467	31
ΔWS	17	1	2	0	− 10	0	5	0	9	1	5	0
ΔSS	3	0	126	8	− 94	0	107	7	48	3	38	2
NP	38	3	40	3	49	3	50	3	71	4	50	3
	mol/m^2		mol/m^2		mol/m^2		mol/m^2		mol/m^2		mol/m^2	
Oxygen												
Diffusion	2.79		2.83		2.62		2.00		1.92		2.43	
Dissolved	0.07		0.08		0.09		0.08		0.10		0.09	
Total	2.86		2.91		2.71		2.09		2.02		2.52	

[a] Each water balance component is indicated as a percentage of PPT

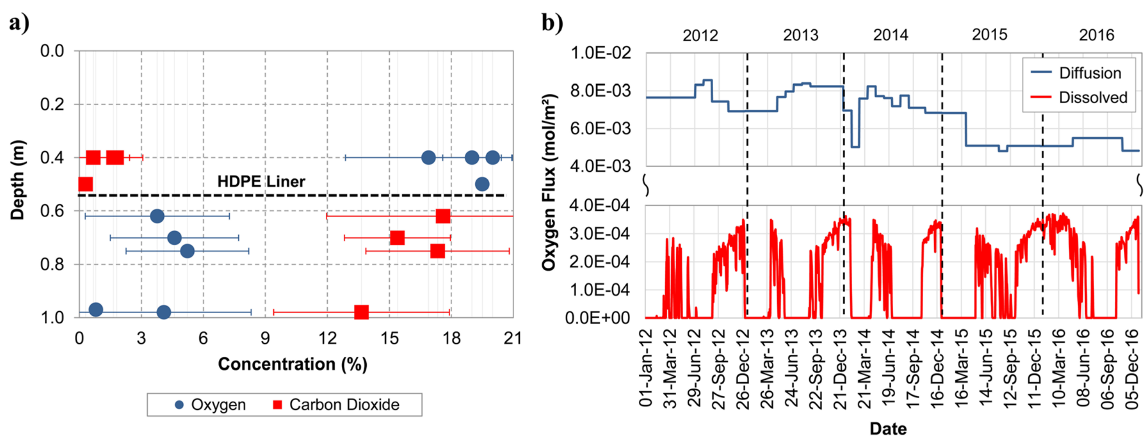

Fig. 6 **a** Mean pore-gas concentrations measured within and below the cover system between January 2012 and December 2016 (error bars depict one standard deviation). **b** Variation in diffusive and dissolved oxygen flux between January 2012 and December 2016

groundwater flows from the northeast at the topographic high in Scotchtown to the WRP. In the till and shallow bedrock, a groundwater divide exists in an east-west trend through the southern portion of the WRP which generally corresponds with the surface water divide between the Irish Brook and KLB watersheds. Therefore, as illustrated in Fig. 8a, b, groundwater flowing through the WRP is deflected by the groundwater divide, with groundwater in the northern portion flowing in a western to northwestern direction towards Waterford Lake and Irish Brook, while groundwater in the southern portion flows in a south to southeastern direction towards Tributary 1 and Tributary 2. In the intermediate bedrock flow system in Fig. 8c, all groundwater flows towards the south and southeast.

AMD impacts to groundwater

The nest of three upgradient monitoring wells located to the northeast of the WRP, each one screened in a different unit, provide background water quality prior to mixing with WRP seepage. The 14 monitoring wells and four CMT wells installed within the WRP footprint indicate water quality in each unit underlying the pile. Samples taken from the four monitoring wells screened in the waste rock—SSSA-MW-101, SSSA-MW-104, SSSA-MW-108 and SSSA-MW-112—would provide the most representative seepage water quality from the WRP; however, all four wells were dry during all sampling events. Samples taken from wells screened in each underlying unit are used to investigate source terms associated with each unit. In the till unit, two wells are located in the northern portion of the WRP, with one well located in the southern portion (see Fig. 8). In shallow bedrock, two monitoring wells and two CMT wells are located in both the northern and southern portions, while in the intermediate bedrock, three monitoring wells exist. The 25 monitoring wells downgradient of the WRP indicate the quality of the AMD contaminant plume that consists of upgradient groundwater and WRP seepage water. Corresponding to the flow regime, downgradient groundwater in till and shallow bedrock is defined as areas to both the west and northwest, and south and southeast of the pile, while downgradient groundwater in the intermediate bedrock is defined as areas to the south and southeast.

AMD is characterized by low pH, elevated concentrations of dissolved metals (primarily iron, aluminium and

Table 4 Summary of ABA tests, including paste pH, total sulphur, ANP, AGP and NNP

Sample	Paste pH –	Total S wt%	Sulphide wt%	Sulphate wt%	ANP kg $CaCO_3$/t	AGP kg $CaCO_3$/t	NNP kg $CaCO_3$/t
2006 (*N* = 18)							
Mean	4.13	0.79	0.51	0.27	− 2.54	16.1	− 18.61
Min–max	3.5–4.8	0.09–1.55	0.0–1.21	0.08–0.85	− 7.3–2.6	0.1–37.7	− 42.6 to − 6.2
Std dev	0.38	0.42	0.37	0.2	2.59	11.53	11.32
2011 (*N* = 8)							
Mean	3.44	0.89	0.35	0.54	− 5.55	10.95	− 16.5
Min–max	3.0–4.1	0.16–1.61	0.08–0.6	0.08–1.01	− 14.71–0.25	2.44–18.88	− 33.6 to − 2.19
Std dev	0.43	0.43	0.19	0.28	4.94	5.97	9.52

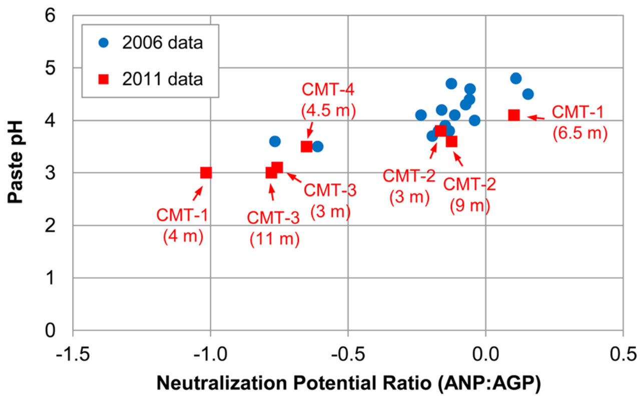

Fig. 7 Classification of waste rock from samples extracted in 2006 and 2011

manganese) and sulphate, high fluid electrical conductivity and high TDS. Figure 9 presents the evolution of sulphate, TDS and pH at the four CMT wells between January 2012 and December 2016. Sulphate and TDS concentrations exhibit similar trends with both gradually decreasing over time. The most significant reductions occur at CMT-2 (red circles) and CMT-4 (magenta triangles) which are located along the prominent groundwater flows and pathways to the northwest and southeast of the WRP, respectively. Figure 9c indicates that the pH is increasing over time. All parameters at each well are within (CMT-2, CMT-3, CMT-4) or approaching (CMT-1) the Canadian Drinking Water Quality (CDWQ) guidelines.

Figure 10 presents the mean sulphate, TDS and pH at each downgradient well before and after cover system installation (error bars based on one standard deviation). The pre-cover installation water quality (darker bars) is based on the mean concentrations sampled between February 2009 and September 2011, while the post-cover installation water quality (lighter bars) is based on samples between August 2014 and July 2016 (nine samples in total). Downgradient water quality has improved at most well locations, with decreases in sulphate and TDS and increases in pH. The most prominent downgradient reductions occurred along the groundwater and plume flow paths to the west (e.g. SSSA-MW-91 to SSSA-MW-93) and southeast of the pile (e.g. SSSA-MW-01 and SSSA-MW-13). Although not shown, pre-cover installation and post-cover installation concentrations of aluminium, iron and manganese exhibit similar reduction trends over time.

Sulphate is used as a geochemical tracer to represent the AMD plume in each unit before and after cover system installation, as shown in Fig. 11. Only the 28 monitoring wells existing during both time periods, as in Fig. 10, are included for a comparison of pre-cover installation and post-cover installation sulphate plumes. Reductions in sulphate concentrations are evident in each unit, particularly in the till and shallow bedrock. Significant reductions are evident in the west near Waterford Lake and in the southeast towards Tributary 2.

AMD impacts to surface water

Figure 12 presents the evolution of sulphate, TDS and pH in surface water at each sampling location between September 2009 and December 2016. In Fig. 12a, sulphate concentrations flowing north in the west perimeter ditch at SSSA-SW-104 (black squares) and east perimeter ditch at SSSA-SW-105 (red circles) exhibit similar trends, which is expected as surface runoff is no longer interacting with the previously exposed waste rock. All surface water flowing north in the perimeter ditches discharges to North Pond which is an alkaline treatment pond. The resulting off-site surface water quality at NWU1-SW-09 (orange x's) demonstrates that the water treatment is effective, showing substantial improvement over time since cover system installation. Further downstream in Irish Brook at DM16-SW-02 (blue diamonds) also highlights improved water quality flowing northwards from the site. For instance, the most recent sulphate concentration at DM16-SW-02 was 27 mg/L (July 2016), which indicates a significant reduction from the baseline concentration of 114 mg/L (mean concentration between November 2009 and February 2012). The evolution of parameters at SSSA-SW-99 demonstrates that Waterford Lake has maintained consistently high water quality over time.

Surface water quality flowing to the south of the WRP also shows marked improvement. Sulphate concentrations in surface water flowing south in the east perimeter ditch at SSSA-SW-08 (black diamonds) are very similar to water quality in the other perimeter ditches at SSSA-SW-104 and SSSA-SW-105. Improved water quality is also evident at SSSA-SW-17 (orange diamonds) and SSSA-SW-05 (blue circles) in Tributary 1, with SSSA-SW-05 being the sampling location immediately prior to off-site discharge to KLB. The most recent sulphate concentration measured at SSSA-SW-05 was 3.4 mg/L compared to the baseline concentration of 9.6 mg/L. These improvements at all sampling locations were also demonstrated by increases in TDS and decreases in pH, as shown in Fig. 12b, c, respectively.

The surface water quality is compared to the established performance criteria for the three key sampling locations in Irish Brook, Waterford Lake and KLB. The criteria for surface water quality at DM16-SW-02 in Irish Brook are compared to the water quality before remediation to confirm that water quality in Irish Brook is improving. Table 5 presents the measured values for key parameters (based on the mean values collected in 2016) and the developed criteria.

Criteria for Waterford Lake are based on threshold levels developed in the site closure plan by environmental consultants AMEC. Mean values collected in 2016 at SSSA-SW-99 are compared to the threshold values as shown in Table 5. At the discharge location to KLB (SSSA-SW-05), CCME Freshwater Aquatic Life (FAL) Guidelines are used to assess if the water quality is becoming suitable for fish habitat.

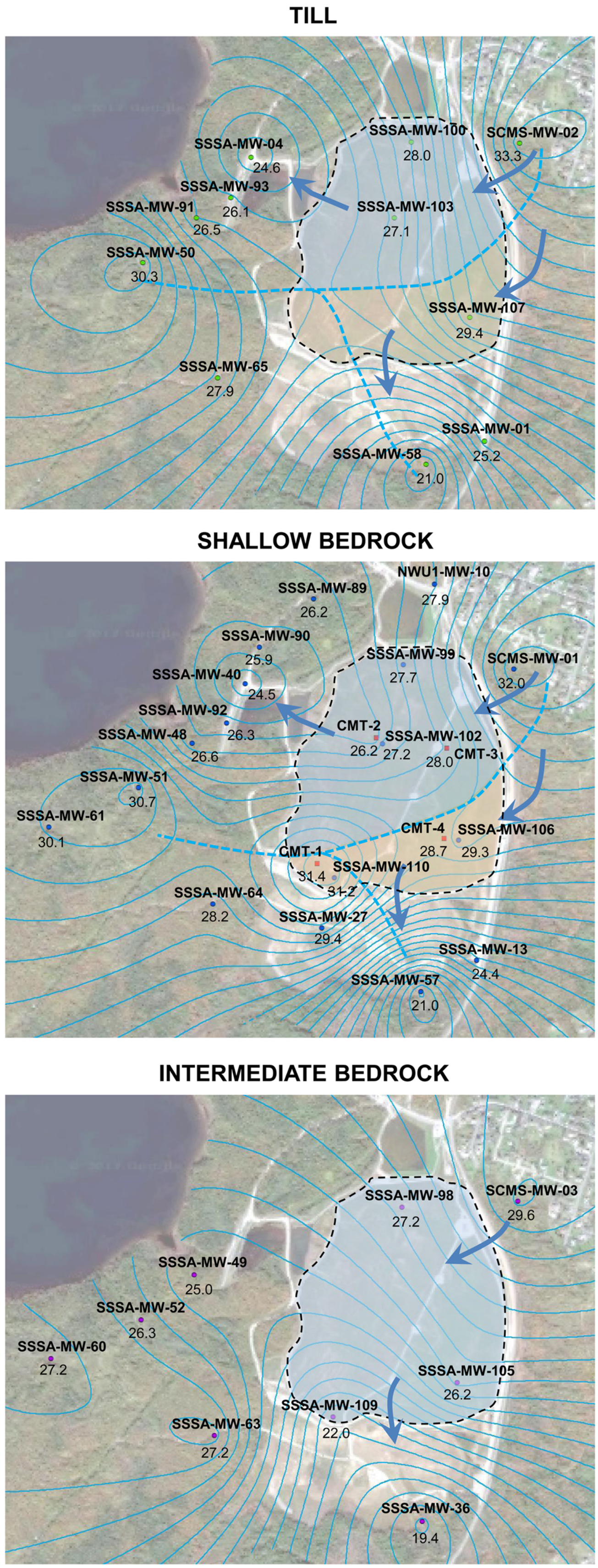

Fig. 8 Groundwater piezometric surface and flow directions in the till, shallow bedrock and intermediate bedrock units. The shaded blue and shaded yellow WRP areas in represent the north and south portions of the WRP, respectively

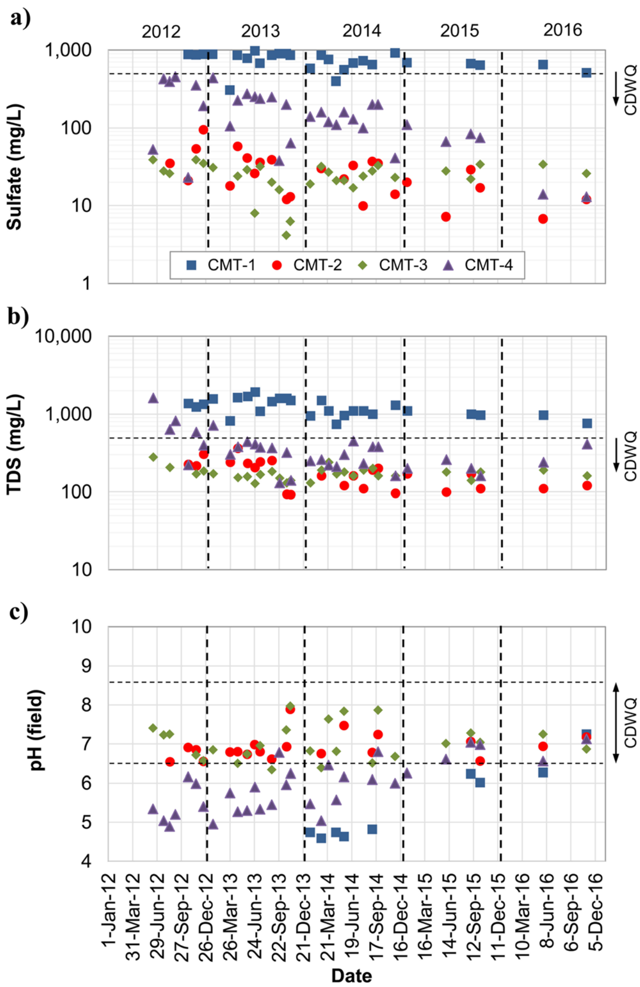

Fig. 9 Evolution of **a** sulphate, **b** TDS and **c** pH at each CMT well between January 2012 and December 2016. Canadian Drinking Water Quality (CDWQ) guideline values are indicated by the horizontal black dashed lines

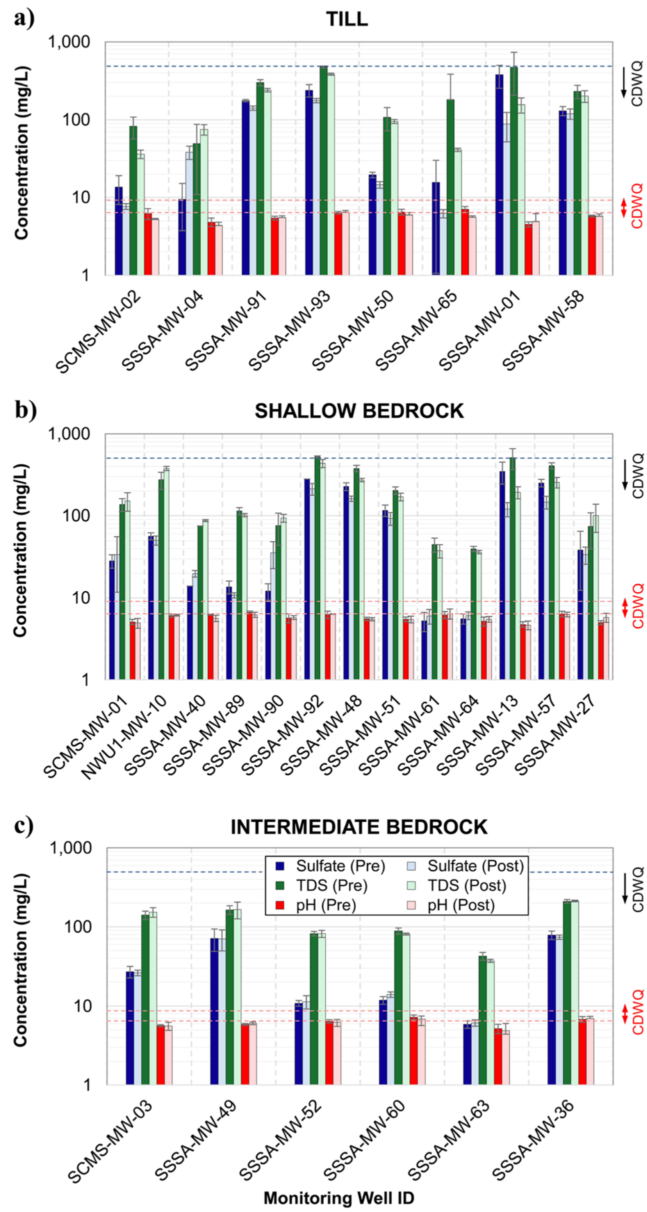

Fig. 10 Mean 'pre-cover installation' and 'post-cover installation' concentrations of sulphate, TDS and pH at each upgradient and downgradient monitoring well in **a** till, **b** shallow bedrock and **c** intermediate bedrock. Canadian Drinking Water Quality (CDWQ) guideline values are indicated by the horizontal black (sulphate, TDS) and red (pH) dashed lines

Acidity load depletion

Since water quality was not available within the waste rock, acidity in the till unit was used to estimate loading from the WRP. Using Eq. (6), the mean acidity concentrations between 2012 and 2016 at the monitoring wells in the till—SSSA-MW-100, SSSA-MW-103, SSSA-MW-107—are 343.75, 11.10 and 264.06 mg/L, respectively, to provide a mean WRP seepage acidity of 206.30 mg/L. This acidity was combined with the seepage rate (equal to NP) of 0.59 L/s to provide an acidity loading of 3.82 t/year. Although alkalinity contained within the waste rock would neutralize some of the acidity, it is ignored to obtain a conservative estimate of acidity depletion.

While 43,088 t of stored acidity is currently in the WRP, additional stored acidity will be generated by pyrite oxidation. With a total potential pyrite (sulphide) acidity of 27,925 t and an oxidation rate of 49.58 t/year, it will take approximately 563 years to oxidize all the potential acidity. Since the annual oxidation rate is larger than the acidity seepage rate, stored acidity in the pile will actually increase by 45.76 t each year for the 563 years to complete oxidation. At that time, the original stored acidity of 43,088 t will have increased to 68,863 t. Assuming the long-term seepage rate of 3.82 t/year remains constant, it will take an additional 18,043 years (total of 18,606 years) to deplete all acidity from the pile.

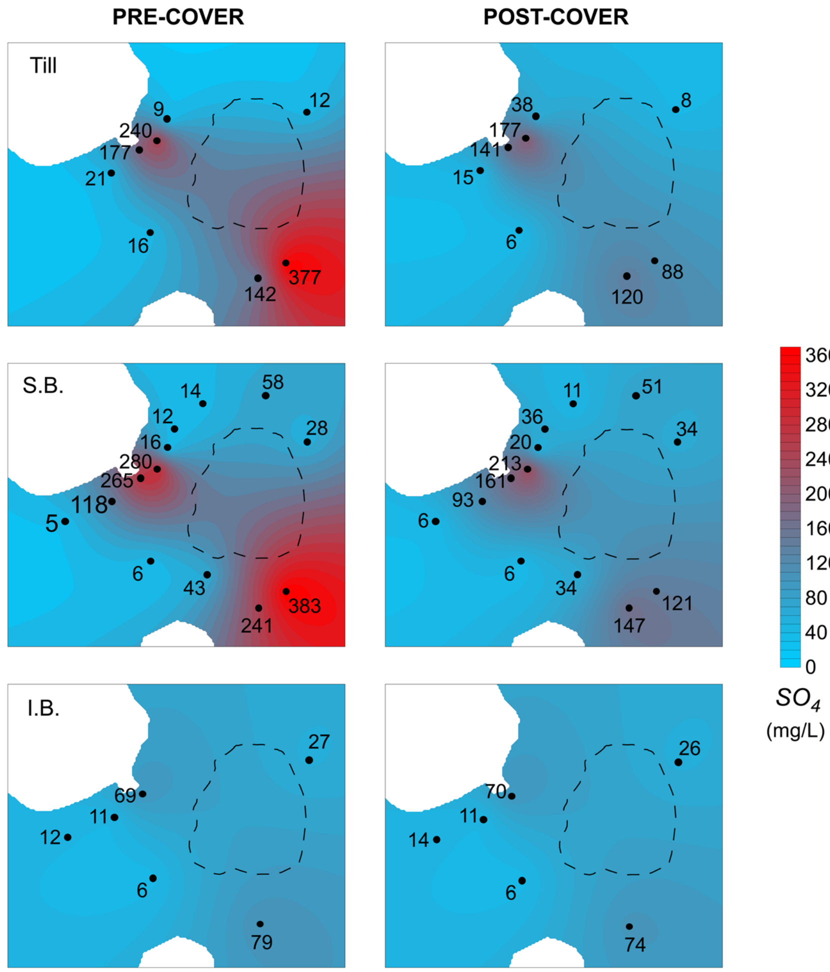

Fig. 11 Distribution of downgradient sulphate concentrations before and after cover system installation

Landform stability

Landform stability is a critical requirement for successful cover system performance. Although the Summit WRP has relatively low slopes, erosion can occur due to the absence of a drainage layer above the HDPE GM. Pore-water pressures can be high in the cover and lead to high saturation and a loss in shear strength. This was confirmed during the site surveys with ponding observed at some locations. The highly saturated cover led to low lateral percolation and high surface runoff. As a result, localized discharge areas were formed leading to visible erosion at some locations. Aside from these sparse locations, the cover and vegetation are thriving.

Limitations

A number of simplifying assumptions were used in this study. It is acknowledged that WRPs are typically complex and heterogeneous and the assumption of homogeneity is simplistic. This assumption simplifies water flow and distribution through the waste rock along with geochemical characterization. However, although mean values from 26 ABA waste rock samples were used to estimate potential and stored acidity volumes in the WRP, these volumes were only used as part of a coarse estimation of WRP acidity depletion. It is acknowledged that this estimate, using mean values of water acidity and seepage rates from the WRP and assuming they remain constant in the long-term, is highly simplistic.

It was not possible to obtain pore-water samples within the waste rock so groundwater samples from the underlying till unit were used to represent seepage water from the WRP. The AMD loading is represented by acidity which is based on the concentrations of iron, aluminium, manganese and pH and calculated from Eq. (6). This calculated acidity and associated acidity loading from the WRP does not account for the neutralization capacity of alkaline minerals such as carbonates and silicates in the waste rock. A net alkaline environment would exist if alkalinity exceeds acidity, which may be enhanced with the lower NP and increased residence time of water in the pile to enhance the dissolution of alkaline minerals such as silicates and carbonates. However, in this study, acidity is used to provide a conservative estimate of AMD loading from the WRP, particularly since acidity is based on

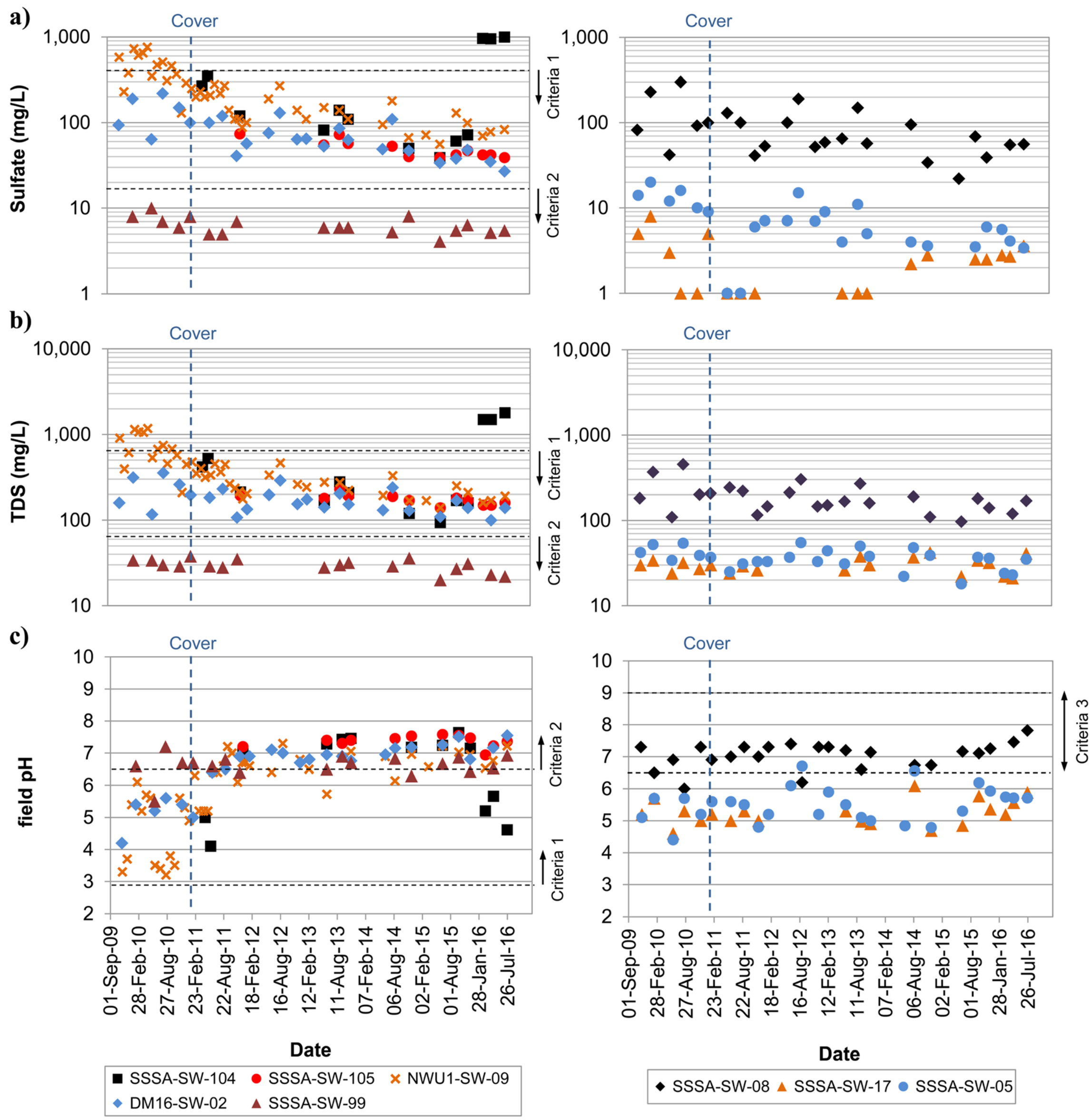

Fig. 12 Evolution of **a** sulphate, **b** TDS and **c** pH at each surface water sampling location in regions west-to-northwest (left column) and south-to-southeast (right column) of the WRP. The different guideline values for DM16-SW-02, SSSA-SW-99 and SSSA-SW-05 are indicated by criteria 1, criteria 2 and criteria 3, respectively

groundwater beneath the WRP and is likely to have already been reduced by mixing with upgradient groundwater.

Kinetic aspects need to be incorporated in future studies to more accurately access the annual generation and release of acidity. The sulphide oxidation rate is assumed to be equal to the oxygen flux into the waste rock. This oxygen flux was based on diffusion, driven by the measured oxygen concentration gradient and diffusivity of the HDPE, and dissolution through percolating water. Oxygen concentrations already stored in the waste rock were not considered along with thermal advection and convection. Therefore, the effects of pre-oxidation on the waste rock (e.g. Pabst et al. 2017) were not included in the determination of sulphide oxidation, along with oxidation that may be catalyzed by bacteria.

Current work involves laboratory tests with humidity cells and leaching columns on waste rock samples to assess various kinetic aspects such as the velocity of sulphide oxidation and metal leaching rates (e.g. Plante et al. 2014). Furthermore, a variably saturated contaminant flow model with FEFLOW (Diersch 2014) is being developed and validated to collected field data, and will incorporate long-term climatic variations, to examine long-term spatial and temporal evolution of AMD plumes and the impacts to environmental receptors. Future work will incorporate the PHREEQC geochemical reaction model (Charlton and Parkhurst 2011) into the FEFLOW

Table 5 Comparison of measured surface water quality at key locations to closure objectives

Parameter	Irish Brook		Waterford Lake		Kilkenny Lake Brook	
	Criteria[a] (mg/L)	SW-02 (mg/L)	Criteria[b] (mg/L)	SW-99 (mg/L)	Criteria[c] (mg/L)	SW-05 (mg/L)
Sulphate	400	31	17	5.4	–	4.4
TDS	630	120	67	23	–	27
pH	2.9	7	6.4	6.8	6.5–9	5.6
Aluminium	1.6	0.01	0.07	0.04	0.1	0.16
Iron	25	1.3	0.16	0.05	0.3	0.39
Manganese	7	0.53	0.13	0.02	–	0.12
Chloride	40	21	14	8.2	–	8.8
Magnesium	26	4.1	2.1	0.9	–	0.77
Zinc	0.056	0.010	0.058	0.007	0.03	0.007

[a] Based on data collected before remediation (mean values plus 2 times the standard deviation)

[b] Waterford Lake threshold values

[c] Canadian Council of Ministers for the Environment Guidelines for Protection of Freshwater Aquatic Life

model to include kinetic and multiphase transfer processes for long-term AMD predictions.

While the cover is performing well and meeting key closure objectives in the first five years, it is acknowledged that the cover system will have to maintain its current physical stability over the long-term. The HDPE GM may encounter some level of deterioration over time. Differential settlements may occur that could fracture the HDPE liner while ageing may damage the plastic seals along the joints. The absence of a drainage layer above the HDPE and the subsequent water ponding has already led to some eroded and bare vegetation areas on the pile. This deterioration will lead to increased NP and oxygen influx to the waste rock, thereby increasing AMD generation and release. Periodic erosion surveys will continue along with vegetation surveys to ensure any excessive root growth that may affect cover integrity is prevented.

Conclusions

Cover systems containing HDPE geomembranes are expected to provide a highly effective and durable method to inhibit the AMD generation in mine WRPs; however, the installation of this cover system type over WRPs has been very limited. Furthermore, no published studies exist that have monitored in situ HDPE cover system performance over WRPs. In this study, a HDPE cover system was installed at the reclaimed Summit WRP with a comprehensive field monitoring programme demonstrating its effectiveness to meet closure objectives. Field monitoring data collected over a five-year period demonstrated significant reductions in (i) water and oxygen influx to the waste rock, (ii) acidity loading from the WRP and (iii) AMD contamination in environmental receptors.

Depletion in oxygen influx to the waste rock resulted in reduced sulphide oxidation and AMD generation. A significant reduction (~ 90%) in water influx was also demonstrated, decreasing from 512 to 50 mm/year of annual precipitation. This resulted in a significant reduction in water seepage and AMD release from the WRP to groundwater, which was confirmed by the improvements in evolving groundwater quality directly beneath the WRP and downgradient of the WRP. The placement of the cover over the previously exposed waste rock removed the source of contaminated surface runoff to the perimeter ditches. This was confirmed by the significant improvements in surface water quality in each perimeter ditch. Surface water quality monitored at downstream locations demonstrated that the cover system is meeting the key site closure objectives: (i) significant improvement in Irish Brook relative to pre-remediation conditions, (ii) protection of the municipal drinking water supply in Waterford Lake and (iii) improvement in Kilkenny Lake Brook for fish habitat.

While the conclusions of this study are based on extensive field data collected during the first five years following remediation, it is acknowledged that the performance of the cover system will evolve over time in response to climatic conditions and site-specific physical, chemical and biological processes. Future integration of comprehensive numerical model predictions with in situ monitoring and a physical study of freeze-thaw, vegetation and erosion cycles will constitute a rigorous physically based evaluation of the long-term performance of the HDPE cover system.

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