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LETTER

The influence of mining on hydrology and solute transport in the Elk Valley, British Columbia, Canada

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Abstract

Previous research has established that surface mining affects downstream water quality. However, there remains limited information and understanding about the interaction of hydrology and geochemistry in surface mined areas. This paper presents an analysis of a multi-year dataset of geochemistry across a gradient of surface mining in an alpine environment. This work formed part of an R&D program examining the influence of mining on hydrological and water quality responses in the Elk Valley, British Columbia, Canada, aimed at informing effective management responses. Results indicate that water from waste rock dumps has a consistent ionic profile that is distinct from reference catchments. The export of weathering solutes did increase with the degree of mine affected area, and was consistently limited more strongly by transport capacity than by supply. Geographical location of waste rock within the catchment (headwaters or outlet) did not affect chemical concentrations or the timing with which chemically distinct waters reported to basin outlets. The dominance of transport capacity over source as limiting to solute export highlights the importance of limiting water input to waste rock piles. However, results strongly suggest that lateral water inputs do not mobilize significant amounts of weathering solutes.

Introduction

Surface mining involves the removal of overburden soil and rock to access deep minerals. Surface mining has increased in frequency globally over recent decades (Palmer et al 2010), and has become the dominant technique for coal mining in North America. Previous research has demonstrated increased concentrations of weathering solutes in streams draining surface mined areas (Lindberg et al 2011, Bernhardt et al 2012, Griffith et al 2012). These solutes originate from the large volumes of waste rock produced during mining (Vengosh et al 2013) that are flushed by water flowing through waste rock (Griffith et al 2012, Murphy et al 2014, Nippen et al 2017). The concentrations of these solutes have been shown to increase linearly with the extent of upstream area impacted by surface coal mining (Lindberg et al 2011, Bernhardt et al 2012).

A key knowledge gap in surface mined systems in mountainous areas is the relationship between hydrology and geochemistry, which interact to determine the quantity of weathering solutes that are transported to streams (Godsey et al 2009, Griffith et al 2012). Watershed scale loss of solutes may be limited by solute availability or by hydrologic transport capacity. Transport limitation often arises from chemostasis, a phenomenon that can be inferred when variability of concentrations of chemicals is significantly less than variability in discharge (Godsey et al 2009, Basu et al 2011, Thompson et al 2011). While previous studies have identified changes in concentration-discharge relationships in single watersheds before and after surface coal mining (Bonta 2005), after long periods of mine inactivity (Murphy et al 2014), and at the scale of single waste rock dumps (Villeneuve et al 2017), no study has systematically assessed hydrochemistry across a spatial gradient of mine impact.
Previous work in surface mined watersheds has found that basin-wide export of selenium (Se), a water quality constituent of interest released by waste rock, is greater in wet years than dry years (Wellen et al., 2015), and that waste rock with vegetative cover releases less Se than bare waste rock (Lindberg et al., 2011; Wellen et al., 2015). These results would suggest that release of Se is limited by transport capacity rather than Se availability. However, waste rock piles with greater upstream area and hence more water throughput do not release more Se than waste rock piles with no upstream area (Wellen et al., 2015). It has been proposed that vertical flowpaths through a waste rock pile have more water-rock contact than horizontal flowpaths along the base of a waste rock pile that receive water from the surrounding watershed and/or transmit water through waste rock (Wellen et al., 2015). This hypothesis is supported by the construction practices of waste rock piles in British Columbia, which use coarse rock at their bottoms to allow the transmission of flows of 20 or 30 m$^3$ s$^{-1}$ unimpeded (Piteau, 1991; Piteau, 1997).

The goal of this study is to assess the validity of this explanation.

An understanding of the relationship of hydrology and chemistry requires an understanding of hydrological flowpaths. Previous research into the effects of surface mining on hydrology have suggested that waste rock stores substantial volumes of water (Dickens et al., 1989), elevate baseflow, and depend on site specific factors may attenuate or increase peak flows (Ferrari et al., 2009, Zégre et al., 2014, Miller and Zégre, 2014). Recently, Nippgen et al. (2017) used a comparative framework to show that across scales, catchments with large volumes of waste rock increase baseflow and strongly elevate stream salinity during low flows. Despite this work, there is only limited knowledge of how surface mining and reclamation practices alter hydrological processes at basin scales (Miller and Zégre, 2014). Recent reviews have called for integration of geochemical data into what to date has been a largely hydrometric literature of hydrological impacts of surface mining in mountainous areas (Miller and Zégre, 2014). Previous studies have focused on using metrics of total ionic strength (e.g. specific conductivity) or single tracers (e.g. sulfate) to infer flowpaths of water (Rikard and Kunkle, 1990; Murphy et al., 2014). Ionic profiles of multiple solutes are needed to infer the existence of geologically distinct flowpaths (Freeze and Cherry, 1979). Understanding flowpaths in both reference and mined catchments is necessary if reclamation can be designed which restores or mimics hydrological processes in pre-disturbance landscapes after mining has ceased.

This study quantifies the degree of source or transport limitation in surface mined watersheds using a multi-year geochemical dataset collected across a wide gradient of mine affected headwater watersheds (0%–80% waste rock coverage). This dataset is complemented by covariates not typically available, including waste rock volumes, reclamation practices, and waste rock age. In addition, this is one of the few studies on surface mining for coal in a cold, snowmelt dominated region, and one of the few outside the Appalachian region. Three hypotheses are tested concerning the hydrochemistry of mountainous watersheds influenced by surface coal mining: (i) water from waste rock dumps has a single, distinct, and consistent ionic signature; (ii) the degree of mine affected area affects the extent to which export of chemicals is limited by supply or transport capacity, while waste rock placement does not; and (iii) placement of waste rock within the basin affects timing with which chemically distinct waters report to the outlet.

**Methods**

The Elk River in Southeastern British Columbia drains 4450 km$^2$ of land area into the trans-boundary Lake Koocanusa as part of the Columbia River basin (figure 1). Total annual precipitation averages 613 mm, 264 mm of which falls as snow, although there is considerable variability in temperature and precipitation timing and phase (Environment Canada Climate Normals, http://climate.weather.gc.ca/climate_normals, Sparwood station). The runoff response of the Elk River is characteristic of a nival system, with peak flows in late May through early June, flow recessions through the summer and early fall, and baseflow conditions throughout the remainder of the year. Daily average temperatures range from $-7.3\, ^{\circ}\mathrm{C}$ in December to $15.8\, ^{\circ}\mathrm{C}$ in July. Teck Resources (Teck) operates five surface mine sites in the watershed of the Elk River. Fifteen headwater catchments were selected to represent a gradient of surface coal mining impact across a number of metrics (table 1, figure 1). Five watersheds had waste rock placed at the outlet, and six had waste rock in their headwaters. One watershed (CM_MC2) could not be classified but was retained for analyses. Three catchments were not impacted by surface mining. At least one watershed at each mine site was selected. Water samples were also taken from one waste rock seep in the Greenhill Creek catchment (GH_GH1). Available covariates at each catchment included waste rock coverage, waste rock volume by year, waste rock weighted age, length of pre-existing streams buried, and proportion of waste rock reclaimed with vegetative cover.

Water samples were taken weekly during freshet (April–June) and once monthly during the remainder of the year. The period of record was generally from 2010–2014, but did vary by watershed, as detailed in table 1. The samples per watershed averaged 79, with an interquartile range from 58–106. The total number of samples analyzed was 1258. Samples were taken in 500 ml PETG bottles, kept at roughly $4\, ^{\circ}\mathrm{C}$, and field preserved with acid. Samples were shipped in cooled HDPE bottles and analyzed by ALS Environmental in
Figure 1. Map of study watersheds. Inset boxes illustrate the positions of waste rock. Black areas in inset boxes indicate waste rock, while grey areas in inset boxes indicate watershed areas.

Table 1. Study watershed attributes.

| Name    | Area (km²) | Waste rock coverage (%) | Waste rock volume (million BCM) | Waste rock volume per area (million BCM km⁻²) | Waste rock position | Buried streams (km) | Reclaimed waste rock area (%) | Waste rock weighted age in 2010 (yr) | Total samples | Year range       |
|---------|------------|-------------------------|---------------------------------|-----------------------------------------------|---------------------|---------------------|-------------------------------|---------------------------------------|---------------|-----------------|
| CM_MC1  | 13.6       | 0                       | 0                               | NA                                            | NA                  | NA                  | NA                            | NA                                    | 38            | 2011–2014       |
| FR_UFR1 | 38.8       | 0                       | 0                               | NA                                            | NA                  | NA                  | NA                            | NA                                    | 63            | 2010–2014       |
| GH_DC1  | 26.3       | 0                       | 0                               | NA                                            | NA                  | NA                  | NA                            | NA                                    | 16            | 2008–2010       |
| EV_EC1  | 33.0       | 25                      | 400                             | Headwaters                                    | 15.9               | 18.0                | 19                           | 112                                   | 104           | 2010–2014       |
| EV_HC1  | 34.3       | 5                       | 557                             | Headwaters                                    | 1.2                | 29.3                | 24                           | 104                                   | 2010–2014     |
| GH_CC1  | 5.8        | 80                      | 387                             | Headwaters                                    | 17.7               | 0.0                 | 13                           | 71                                    | 2010–2014     |
| GH_GH1  | 16.1       | 17                      | 117                             | Headwaters                                    | 2.9                | 16.6                | 13                           | 72                                    | 2010–2014     |
| GH_PC1  | 3.0        | 54                      | 79                              | Headwaters                                    | 2.9                | 35.0                | 16                           | 70                                    | 2010–2014     |
| GH_TC1  | 11.2       | 29                      | 88                              | Headwaters                                    | 11.6               | 28.0                | 6                            | 43                                    | 2010–2014     |
| CM_MC2  | 101.1      | 56                      | 242                             | Headwaters                                    | 11.4               | 16.2                | 11                           | 105                                   | 2010–2014     |
| EV_BC1  | 11.5       | 56                      | 242                             | Headwaters                                    | 10.2               | 24.0                | 9                            | 109                                   | 2003–2014     |
| GH_SC1  | 7.2        | 41                      | 214                             | Headwaters                                    | 8.1                | 9.0                 | 14                           | 32                                    | 2010–2014     |
| FR_HC1  | 48.8       | 7                       | 157                             | Outlet                                        | 7.4                | 23.0                | 9                            | 90                                    | 2010–2014     |
| FR_KC1  | 43.1       | 21                      | 1056                            | Outlet                                        | 29.1               | 8.2                 | 12                           | 111                                   | 2004–2014     |
| LC_WLC  | 8.0        | 20                      | 192                             | Outlet                                        | 7.5                | 0.0                 | 21                           | 117                                   | 2010–2014     |

a Cumulative in 2010.
b Bulk cubic metres.
Figure 2. Piper plot with all geochemical samples. Piper plot is presented in rectangular space after Ray and Mukherjee (2008).

Vancouver, British Columbia. Time between sample collection and analysis in the lab ranged from 1–21 days, and averaged 7 days. Samples were filtered through a 0.7 um glass fiber filter. Water was analyzed for magnesium, potassium, sodium, calcium (Mg, K, Na, Ca, EPA method SW646 6010B); sulfate (SO₄, APHA method 4110); nitrate (NO₃, EPA method 300.0); chloride (Cl, EPA method 325.1); total alkalinity (NEMI method SM 2320B); pH (APHA method 4500-H); aluminum, selenium (Al, Se, EPA method 6020A or EPA method 6010B); and specific conductivity (SC, APHA method 2510). Measurements of alkalinity were converted to carbonate (HCO₃) using methods detailed by Chapra (1997, chapter 39). Field measurements of conductivity were taken and calibrated to lab measurements via linear regression to account for sensor drift. Discharge was measured at the time water samples were taken using current meter techniques in the field.

Discharge data were normalized by watershed area and expressed in units of mm day⁻¹. Flow-weighted mean concentrations were computed to disambiguate changes due to fluctuations in discharge from changes in land use (Griffith et al 2012). Medians were tested for significant differences using a non-parametric Mann-Whitney U test.

Specific conductivity has been shown to be a reliable proxy of total ionic load and often increases dramatically downstream of surface coal mines (Rikard and Kunkle 1990, Vengosh et al 2013, Murphy et al 2014). The change of specific conductivity with discharge often displays patterns of hysteresis. These patterns have been used to reveal the timing with which different sources of water report to the basin outlet (Evans and Davies 1998, Murphy et al 2014), with clockwise loops indicating a flushing of near-stream sources or changing sources as discharge increases, and counterclockwise loops indicating a delayed release of waters high in solutes (Seeger et al 2004). Classification trees were used to predict the hysteresis type using the following candidate predictor variables: percent waste rock coverage, waste rock volume, waste rock volume per catchment area, and waste rock position. All statistical analyses were carried out in MATLAB.

Results

Changing ionic composition: The hydrochemical facies of all water samples are presented in figure 2. The linear pattern in figure 2 clearly indicates that the in-stream samples can be described as waters mixed from two distinct flowpaths, one through waste rock and one through the unimpacted portions of the catchment (Freeze and Cherry 1979). The unmined watersheds and the waste rock seep (site UNT) form the two ends of this linear distribution. Samples from each watershed are highly clustered, indicating that the hydrochemical composition of water varies more in space than in time across the study sites.

Waste rock coverage explained much of the variability of ionic composition of streamwater. The percent coverage of waste rock of the watershed was strongly correlated with the average percent of SO₄ of anions, regardless of waste rock location (% anions as SO₄ = 9.8*ln(%cover)+92.2, r² = 0.73), with SO₄ approaching 90% of anions asymptotically as waste rock coverage increased to a maximum of 80%. Waste rock coverage also had a marked effect on the variability of percent of SO₄ of anions, again regardless of position (range of % anions as SO₄ = 38.8*exp(−2.59*%cover), r² = 0.59). The percent composition of anions by SO₄ is presented in figure 3. The vertical axis of figure 3 is analogous to the line along which the chemical facies of figure 2 are arranged. For some watersheds with waste rock at the outlet, there is a distinct change in the ionic composition of the water during the spring freshet period. During freshet, the ionic composition resembles a mixture of the baseflow, which has the highest proportion of water from waste rock (Dickens et al 1989, Murphy et al 2014), and the unimpacted catchments, which have a very low proportion of SO₄. As waste rock coverage increased, this pattern was considerably muted. The
catchments with waste rock in their headwaters also had a notable decrease of percent anions as SO$_4$ in the spring, with this pattern becoming muted as percent coverage increases.

**Hysteresis of specific conductance**: figures 4(a) and (c) illustrate that hysteresis occurs on an annual timescale, that the direction varies from site to site and that the ionic composition of waters also changes. Annual hysteresis loops were manually classified as clockwise, anticlockwise, and incoherent for each year and for each watershed (table 2). The hysteresis direction at each site was stable through time, indicating that hysteresis reflects some aspect of watershed properties. The best fit classification tree model had two splitting levels and predicted the loop type correctly in 77% of cases (figure S1 available at...
Figure 4. Plots depicting the hysteresis of the relationship between specific conductivity and discharge (a) and (c), as well as the geochemical profile of waters taken at different points along the hysteresis loop (b) and (d). Watersheds are FR-KC1 (top) and GH-CC1 (bottom). Kilmarnock Creek geochemical profiles May 2 (1), May 24 (2), July 6 (2), and October 3, 2011 (4). Cataract Creek geochemical profiles April 4 (1), May 2 (2), July 5 (3), and November 1, 2011 (4). Note that the concentrations of Al and SeO4 are multiplied by 1000 for presentation purposes, i.e. are in units of ueq/l.

Table 2. Direction of annual hysteresis loops of specific conductivity. C = clockwise, A = anti-clockwise, and inc = inconclusive.

| Watershed        | 2006 | 2007 | 2008 | 2009 | 2010 | 2011 | 2012 | Overall classification | Type |
|------------------|------|------|------|------|------|------|------|------------------------|------|
| CM_MC1           | –    | –    | –    | –    | C    | inc  | C    | C                     | C    |
| FR_UFR1          | –    | –    | –    | –    | inc  | inc  | inc  | inc                   | Reference |
| GH_DC1           | –    | –    | –    | –    | –    | –    | –    | –                     | Reference |
| EV_EC1           | –    | –    | –    | –    | inc  | inc  | inc  | inc                   | Reference |
| EV_HC1           | C    | C    | –    | –    | –    | –    | –    | –                     | Headwaters |
| GH_CC1           | –    | –    | –    | –    | –    | –    | inc  | inc                   | Headwaters |
| GH_GH1           | inc  | inc  | inc  | inc  | inc  | inc  | inc  | inc                   | Headwaters |
| GH_PC1           | –    | –    | –    | inc  | inc  | inc  | inc  | inc                   | Headwaters |
| GH_TC1           | –    | –    | –    | –    | inc  | inc  | inc  | inc                   | Headwaters |
| CM_MC2           | –    | –    | –    | inc  | inc  | –    | –    | –                     | Middle |
| EV_BC1           | inc  | inc  | inc  | inc  | inc  | inc  | inc  | inc                   | Outlet |
| GH_HC1           | –    | –    | –    | inc  | inc  | inc  | inc  | inc                   | Outlet |
| FR_HC1           | inc  | inc  | inc  | inc  | inc  | inc  | inc  | inc                   | Outlet |
| FR_KC1           | –    | –    | –    | –    | –    | –    | –    | –                     | Outlet |
| LC_WLC           | –    | –    | –    | inc  | inc  | inc  | inc  | inc                   | Outlet |

Summary

| Spoil Position | C | A | inc |
|----------------|---|---|-----|
| Headwaters     | 1 | 2 | 3   |
| Outlet         | 4 | 0 | 1   |
| Middle         | 1 | 0 | 0   |
Catchments with waste rock deposits covering greater than 47% of their area were predicted to have anti-clockwise loops, while for catchments with less than 47% waste rock the loop direction depended on waste rock position. Catchments with waste rock at their outlet were predicted to have clockwise loops, while catchments with waste rock in their headwaters were predicted to have incoherent loops. At proportional coverage rates below roughly half of the catchment, locating waste rock at the watershed outlet results in water rich with weathering ions reporting to the basin outlet first, while location of waste rock did not play a role in loop direction when the catchment was more than roughly half covered with waste rock.

Concentration-discharge relationships: figure 5 presents a scatterplot of the slopes of the concentration-discharge relationships in the log space for all of the catchments against the percent waste rock coverage for SO$_4^-$ and Se. While the slopes varied significantly, most watersheds for most constituents had slopes closer to 0 than to −1. This suggests that export of SO$_4^-$ and Se were limited more by transportation capacity (discharge) than by availability of these constituents (Godsey et al 2009). When only examining catchments with waste rock, there was a moderate correlation between the concentration-discharge slope and proportion of waste rock coverage for SO$_4^-$ (slope = 0.57, $r^2 = 0.47$, $p = 0.01$), while this relation for Se was similar and nearly statistically significant (slope = 0.53, $r^2 = 0.31$, $p = 0.06$). These relationships imply that as the proportion of waste rock coverage increases, the export of weathering ions is less limited by material availability and more limited by transport capacity. No significant correlation was found between concentration-discharge slope and waste rock volume per catchment area, percent waste rock reclaimed, length of buried streams, or waste rock age ($p > 0.05$). A Mann-Whitney U test was performed on the concentration-discharge slopes to determine differences in functioning between watersheds with waste rock at their outlets and in their headwaters. No significant differences were found for any analytes ($p > 0.05$).
For watersheds impacted by mining, flow weighted concentrations of HCO$_3$, Ca, Mg, Se, SO$_4$, and total dissolved solids were strongly correlated with the percent coverage of waste rock (figure S2; $p < 0.05$, $0.52 < r^2 < 0.73$), and more strongly correlated with the volume of waste rock per watershed area (figure 6, figure S3; $p < 0.05$, $0.77 < r^2 < 0.83$). This positive correlation is in agreement with previous studies (Lindberg et al 2011, Bernhardt et al 2012, Nippgen et al 2017), yet it is interesting to note that the inclusion of estimates of volume improved concentration estimates. The location of waste rock at the catchment outlet or headwaters did not affect the flow weighted concentrations of any of the geochemicals (figure 6, figure S2, S3). When the one extreme site is omitted from the regression, the fit does decline but is still statistically significant ($p < 0.05$, $0.56 < r^2 < 0.58$). We also tested whether the proportion of the watershed upstream of the waste rock piles affected flow weighted concentrations. No regressions were significant for any analyte ($p > 0.05$).

Discussion

Hypothesis 1—water from waste rock dumps has a single, distinct, and consistent ionic signature: previous research has used stable isotopes of strontium and selenium to identify the sources of contaminants originating from waste rock generated by surface coal mining (Vengosh et al 2013), yet most research into the watershed scale hydrochemistry of surface and mountain top mined systems has focused on using a single tracer, typically SO$_4$ or specific conductivity (SC) (e.g. Murphy et al 2014). We found that the geochemical profile of watersheds influenced by surface mining was high in SC and SO$_4$, and also that the dominant anions were SO$_4$ and HCO$_3$, with other anions present in very low proportions (figure 2). Previous hydrogeologic work in the Elk Valley indicated that groundwater from the coal-bearing Mist Mountain formation, which would have supplied the waste rock to all sites across the study with one exception, is characterized by relatively low SO$_4$ and high HCO$_3$ concentrations (Harrison et al 2000). Other work has found that minerals commonly found in coal-bearing rocks such as pyrite release SO$_4$ and metals (Fe, Se) when exposed to oxygen (Diehl et al 2012). It is important to note that quantitative mixing models use absolute concentrations, and not relative ones, before concluding that the ratio of SO$_4$ to HCO$_3$ can be used to quantify the proportion of runoff generated from waste rock dumps.

Figure S4 presents a mixing diagram for Greenhills Creek (GH_GH1) and Kilmarnock Creek (FR_KC1). Greenhills has waste rock in its headwaters, while Kilmarnock has waste rock at its outlet. Two endmembers per creek are sufficient to explain nearly all the variability in streamwater concentrations of Se and SO$_4$. The endmember at the low end of the range is estimated as the median of the streamwater concentrations of the unmined catchments. The endmember at the high end of the range at Greenhills Creek was estimated with samples taken from a waste rock seep (UNT) in the Greenhills catchment, while the high concentration endmember from Kilmarnock was estimated from streamflow data taken from Cataract Creek (GH_CC1), the watershed with the highest percent coverage of waste rock (80%). The excellent bounding of the streamflow by these end members substantiates that the ionic profile of water from waste rock is fairly consistent in space, albeit with some local variations, likely due to variations in composition of the source geologic material.
A comparison to work focused on Appalachian sites indicates comparable levels of $\text{SO}_4$ but much higher pH in surface mine influenced tributaries. The values of $\text{SO}_4$ concentration in GH_CC1, a watershed which is 80% covered by waste rock, averaged 1217 mg L$^{-1}$ $\text{SO}_4$. Bernhardt et al. (2012) estimated mean $\text{SO}_4$ concentrations in Appalachian streams with 80%–89% waste rock coverage ranged from 700 mg L$^{-1}$–1200 mg L$^{-1}$. Water from GH_CC1 had pH values ranging from 7.4–8.5. Rikard and Kunkel (1991) report median pH between 4.4 and 6.6 in surface mined tributaries. This difference is likely due to limestone bedrock in the Elk Valley.

**Hypothesis 2**—the degree of mine affected area affects the extent to which export of chemicals is limited by supply or transport capacity, while waste rock placement does not: previous work suggests that watersheds affected by surface mining are fairly chemostatic (Murphy et al. 2014, Wellen et al. 2015), implying that chemical export is limited by transport capacity more than supply. Interestingly, watersheds tended to behave more chemostatically as the proportion of coverage increased, indicating that waste rock is able to supply available geochemicals in excess of the ability of water to exhaust this supply. Watersheds displayed relatively chemostatic selenium dynamics even when unmined. Thus while the degree of sulphate chemostasis is weakly influenced by the degree of waste rock coverage, there is no evidence from this data that the available supply of selenium is exhausted by increased discharge through the watershed.

Waste rock placement did not affect the flow weighted concentrations of any geochemicals, providing more evidence that while watersheds with waste rock at their outlets may be characterized by strongly negative concentration-discharge (CQ) slopes, this is most likely not due to even temporary exhaustion of solute sources.

**Hypothesis 3**—placement of waste rock within the basin affects timing with which chemically distinct waters report to the outlet: in montane environments characterized by a nival runoff regime, hysteresis presents at an annual timescale, and has been linked to contributions of water through different flowpaths reporting to the basin outlet at different times through analysis of isotopic and chemical tracers (Zhang et al. 2018). Waste rock placement did not appear to have a dominant impact on the timing with which chemically distinct waters reported to the basin outlet as timing was predominantly controlled by total waste rock coverage. As catchments surpassed 50% waste rock coverage, the geochemical profile became remarkably consistent, suggesting that water from waste rock forms a relatively constant proportion of the overall flows (figure 3). The primary variable in a decision tree classifying hysteresis of SC was not waste rock position but rather the total coverage (figure S1). However, for catchments with waste rock coverage below the 50% value, placing waste rock at the basin outlet usually resulted in clockwise hysteresis of SC, while placing waste rock in the headwaters typically resulted in much less discernable patterns of SC hysteresis.

**Implications for management:** limiting the downstream transport of constituents of interest during and after mining operations is a priority in areas of surface mining. It is of considerable interest to operators to know whether the placement of waste rock in the catchment has an influence on downstream impacts. The export of Se and other geochemicals analyzed in this study is limited more strongly by transport capacity than by supply. This implies that more water moving through the waste rock mobilizes more constituents, highlighting the importance of limiting water input to piles. However, as waste rock placement does not affect flow weighted concentrations, water from the unimpacted part of the catchment must be traversing waste rock piles without becoming enriched with weathering ions. Hence, waste rock placement does not affect concentrations of constituents, and the effects of waste rock placement on CQ slopes is largely due to mixing. This suggests that water traversing the base of a pile moves along preferential flowpaths with low water-rock contact and does not mobilize many weathering solutes, while water vertically percolating through the waste rock piles does mobilize weathering solutes. This helps explain the paradoxical findings of a previous empirical model of Se export, which found that the area draining through a waste rock pile, which correlates strongly with annual discharge, had no effect on Se export despite more Se export appearing to be strongly transport limited (Wellen et al. 2015). If the water draining through a waste rock pile’s base does not mobilize constituents, the elevated constituents downstream from waste rock piles must be mobilized primarily through vertical percolation through a pile. Barbour et al. (2016) established the dominance of matrix flow over preferential flow through the use of stable isotopes. While preferential flow is ubiquitous, the movement of seasonal percolation cycles was tracked vertically to considerable depth. A volume accounting suggests that that the vast majority of the water movement occurs through the matrix in waste rock piles with a residence time of 5–20 years for 100 m high waste rock dumps. A modelling study by Appels et al. (2018) emphasizes the dominant role of matrix flow in large waste rockumps. In contrast, water moving through piles laterally is typically conducted through channels made with large boulders with minimal surface area (Piteau 1997) and has residence times of a few months at most.

Future research is necessary to evaluate whether reducing percolation through waste rock (e.g. through geotextiles, vertical rock drains, and reclamation) is more effective than approaches that divert streams away from waste rock piles for reducing elevated concentrations of weathering solutes in streams.
Our results suggest that should increases in precipitation result in increased streamflow (Environment Canada 2016), greater chemical release from waste rock is likely. However, more research is needed to determine whether more precipitation will result in more streamflow, and whether possible increased release rates will be offset by the documented decreased release of chemicals due to waste rock reclamation (Wellen et al. 2015).

In conclusion, this paper used long term geochemical and hydrological datasets to understand the hydrochemistry of watersheds subject to surface coal mining in an alpine region. This paper shows that water from waste rock dumps has a single, consistent ionic profile; that waste rock is fairly chemostatic across a range of metrics of mine impact; and that waste rock placement influences neither the timing with which water from distinct sources reports to the basin outlet nor the flow weighted concentrations of constituents. Taken together, these results strongly suggest that losses of constituents of interest from waste rock are due to water vertically percolating through a waste rock pile, not from lateral inputs of water.

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