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Acid mine drainage risks – A modeling approach to siting mine facilities in Northern Minnesota USA

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SUMMARY

Most watershed-scale planning for mine-caused contamination concerns remediation of past problems while future planning relies heavily on engineering controls. As an alternative, a watershed scale groundwater fate and transport model for the Rainy Headwaters, a northeastern Minnesota watershed, has been developed to examine the risks of leaks or spills to a pristine downstream watershed. The model shows that the risk depends on the location and whether the source of the leak is on the surface or from deeper underground facilities. Underground sources cause loads that last longer but arrive at rivers after a longer travel time and have lower concentrations due to dilution and attenuation. Surface contaminant sources could cause much more short-term damage to the resource. Because groundwater dominates baseflow, mine contaminant seepage would cause the most damage during low flow periods. Groundwater flow and transport modeling is a useful tool for decreasing the risk to downgradient sources by aiding in the placement of mine facilities. Although mines are located based on the minerals, advance planning and analysis could avoid siting mine facilities where failure or leaks would cause too much natural resource damage. Watershed scale transport modeling could help locate the facilities or decide in advance that the mine should not be constructed due to the risk to downstream resources.

1. Introduction

Acid mine drainage (AMD) is a problem associated with mines throughout the world (Jacobs and Testa, 2014). In the United States, promoting mining on public lands has been a priority since the 1800s (Leshy, 1987), with little consideration for the waste other than getting it away from the mine site being the practice prior to about 1970 (Church, 1996; Ferderer, 1996). Mines were developed with little concern regarding AMD (Crews, 1973; Williams, 1975).

That is no longer the situation. Mining-caused contamination is a global problem and few sites are isolated or sufficiently underused that potential contamination can be ignored. One example of global cooperation among the mining industry, conservation groups, and stakeholders to set a higher standard for mining, including the prevention of AMD and promotion of it remediation is the Initiative for Responsible Mining Assurance (IRMA) (http://www.responsiblemining.net/). The goal of IRMA is to promote responsibility by certifying the most responsible mines.

Watershed-scale planning is necessary to avoid the most serious problems. However, much watershed-scale research focuses on remediation (Church et al., 2007; Crews, 1973; Kimball et al., 2006; Nimick and von Guerard, 1998; Skousen et al., 1999), often with the perspective of optimizing treatment (Crews, 1973; Kimball et al., 1999). Conceptual and numerical modeling at various scales can aid in prioritizing sites for remediation (Myers, 2013; Runkel et al., 2013). Herr et al. (2003) developed a watershed-scale model of AMD entering a river to show the contaminant sources and potentially demonstrate the effectiveness of remediation of specific sites. Runkel and Kimball (2002) simulated flow and equilibrium chemistry along a stream heavily impacted by AMD to demonstrate the effects of remediation. Related modeling showed that simulation results are most affected by model parameters affecting a nearby stream reach or watershed (Gooseff et al., 2005). Myers (2013) suggested priorities for remediating phosphate mines based on a groundwater model of a large western watershed contaminated with selenium. Statistical models also can show the mining features or geology that best explain the variability in salinity discharging from a mined watershed (Evans et al., 2014). These studies however do not suggest a means of avoiding AMD or other contamination issues as part of the planning process.

Preventing future mines from becoming AMD problems is often considered an engineering issue at the mine site (Jacobs et al., 2014; Buxton et al., 1997; EPA, 1994), although failures occur often

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(Caldwell and Charlebois, 2010; Kuipers et al., 2006; Rico et al., 2008). The level of damage caused by these failures can depend on their location in the watershed. Missing from the literature and generally from mine planning is research showing methods designed to site mines and mine facilities to avoid large-scale AMD problems when leaks occur.

The objective of this study was to use watershed-scale groundwater flow and transport modeling to predict which mine sites in a sulfide rich watershed would be more likely to cause downstream AMD problems if engineering controls fail. It demonstrates how watershed-scale modeling prior to the actual development of mines can improve mine planning to facilitate future remediation when engineering failures occur, a topic currently not substantially addressed in the literature. The setting is the Birch Lake watershed, located within the larger Rainy Headwaters watershed in northern Minnesota, USA (Fig. 1). The area has no current mining and one historic mine. Mining companies hold leases on at least six different copper/nickel deposits (MNDNR, 2014) within the watershed. The Boundary Waters Canoe Area Wilderness (BWCAW), a high value and one of the most-visited wilderness areas in the United States (Heinselman, 1996), lies directly downstream of the potential mining (Fig. 1).

The model could help to optimize mining and waste disposal locations or to decide whether the risks of mining are too high as well as providing information on where more information is needed for decision making. The model could be an example for countries and companies around the world contemplating entering relatively pristine watersheds currently valued for resources that could be damaged by mine pollution.

2. Method of analysis

2.1. Study area

The study area is in the Birch Lake watershed south of the South Kawishiwi River in northeastern Minnesota, USA (Fig. 1). The middle two thirds of the study area overlies the Duluth Complex while the north end abuts Granite Range granite (Fig. 2, Table 1). The Duluth Complex hosts nickel–copper–platinum sulfide deposits in the basal portion of the South Kawishiwi intrusion as much as 1200 m below ground surface (Miller et al., 2002; Parker and Eggleston, 2014). The deposits are potentially significant acid-producers (EPA, 1994; Lapakko, 1988; Lapakko and Olson, 2015; Polymet Mining, 2013b; Polymet Mining, 2012; Severson et al., 2002). The sulfide content of the Spruce Road deposit is 2–5% by volume and 3–4% by weight (Parker and Eggleston, 2014), which may on the high end of the range for the Duluth Complex (Seal et al., 2015). The host mineralized zone has previously produced AMD (EPA, 1994; Lapakko, 1988; Lapakko and Olson, 2015).

Most mining leases lie south of the South Kawishiwi River in the Birch Lake and Stone Creek watersheds (Fig. 2). Proposed mines are expected to initially be underground (Cox et al., 2009; Parker and Eggleston, 2014), including some underground waste rock and tailings disposal (Twin Metals, 2014). Waste rock is rock and overburden removed to reach the ore and tailings are the processed ore from which the valuable mineral has been removed. Waste rock and tailings are considered contaminant sources for this paper because mine planning as to the placement of either material is not sufficiently advanced to distinguish among the properties of either type.

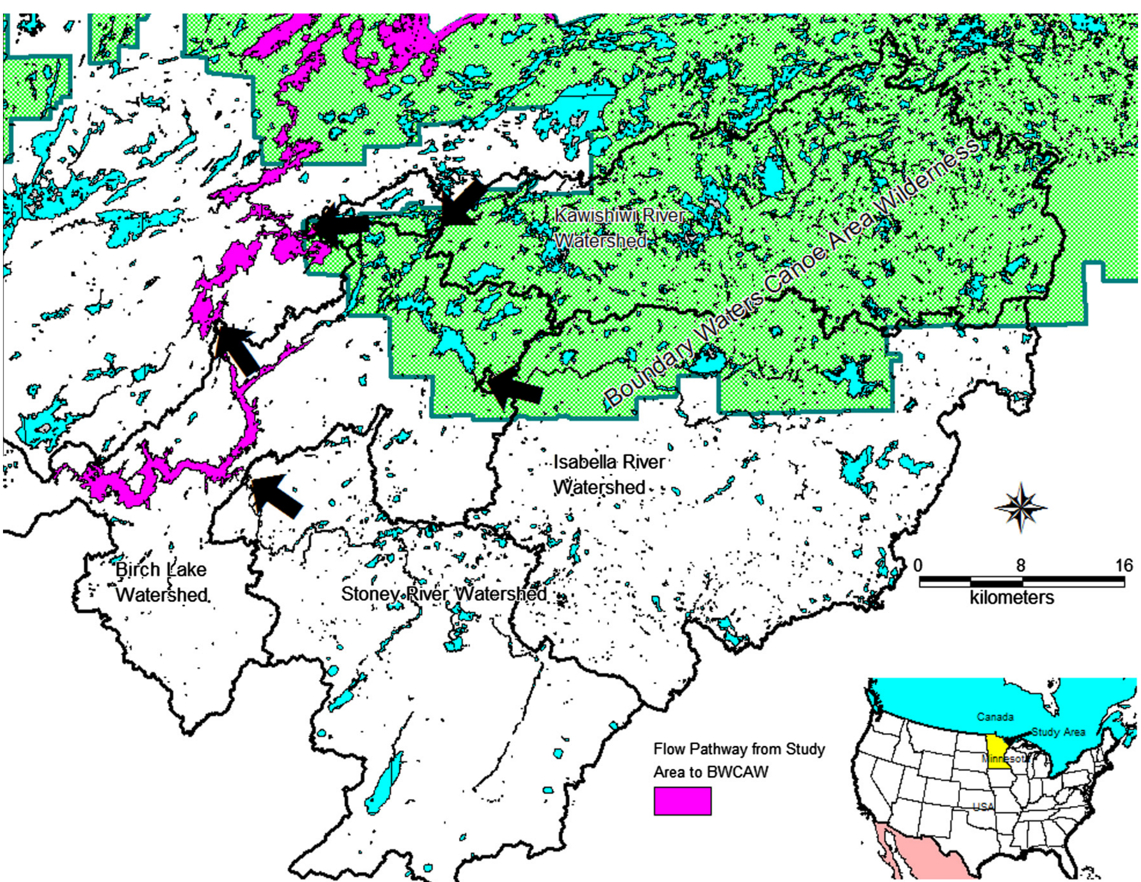

Fig. 1. Rainy Headwaters watershed and study area, showing subwatersheds, rivers, and lakes. Arrows are flow direction from watersheds. Watershed boundaries from Dnr100kwatersheds, www.mngeo.state.mn.us/chouse/metalong.html.

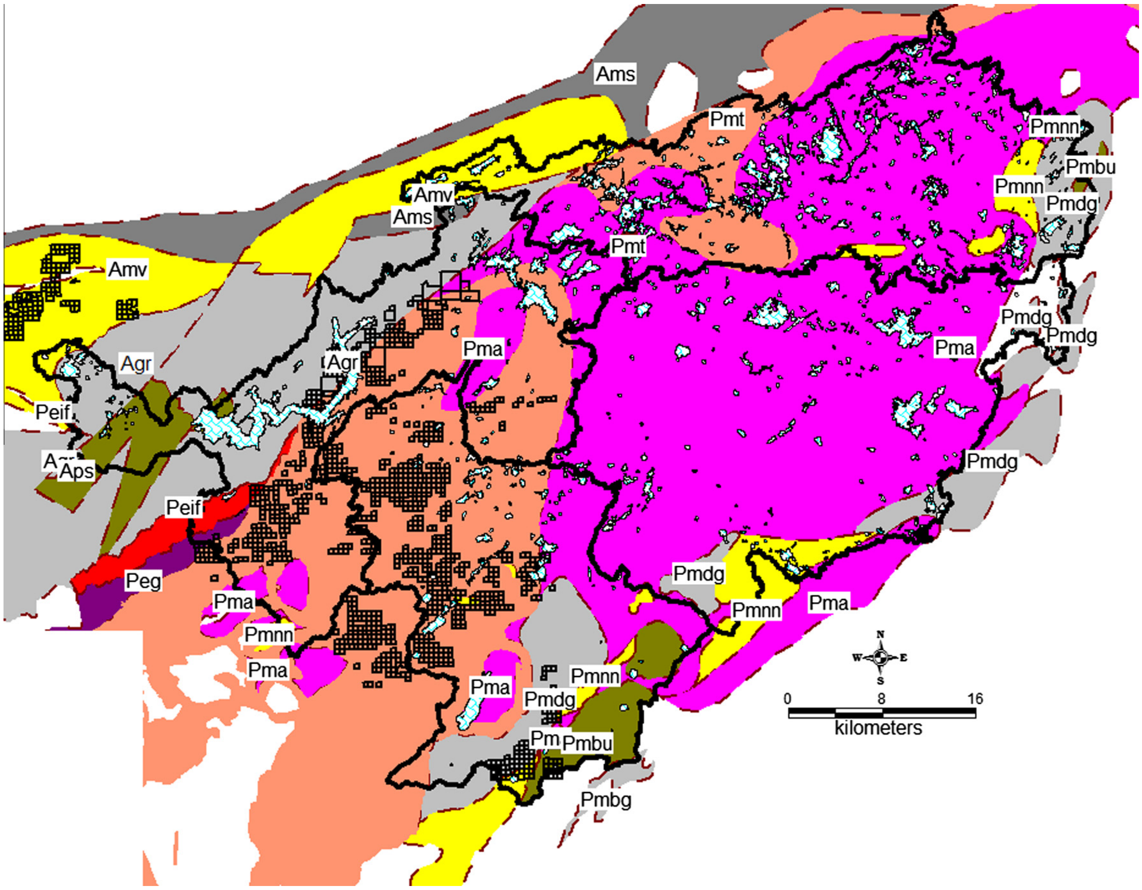

Fig. 2. Bedrock geology and watersheds of the study area. See Table 1 for a description of the formations. See Fig. 1 for the watershed names. The black rectangles are mining leases and approximate location of the mineral deposits.

Table 1
Geologic formations (Nicholson et al., 2007) along with model zone, layer and final calibrated horizontal and vertical conductivity (Kh and Kv, respectively) in meters/day. *n* is effective porosity, Sy is specific yield, and Ss is storage coefficient.

Formation/lithology		Zone	Kh	Kv	Sy	Ss	*n*	Layer
Duluth Complex, troctolite/gabbro	Pmt	2	0.307	0.01008	.07	.000001	.007	3
		12	0.342	0.137	.12	.00001	.012	2
		22	0.102	0.114	.12	.00001	.012	2
		32	0.00014	0.0182	.07	.000001	.007	3
		33	0.035	0.0145	.12	.00001	.012	2
Duluth Complex, anorthosite/gabbro	Pma	3	0.025	0.2	.07	.000001	.007	3
		13	2.9	0.4	.12	.00001	.012	2
		31	0.26	0.002	.12	.00001	.012	2
Basalt/rhyolite	Pmnn	4	0.05	0.025	.07	.000001	.012	3
		14	0.1	0.06	.12	.00001	.012	2
Giants range Granite	Agr	5	0.0015	0.0015	.02	.000001	.002	3
		15	0.214	0.2	.05	.00001	.005	2
		25	0.8	0.5	.05	.00001	.005	2
Gabbro/troctolite	Pmbu	6	0.27	0.01	.08	.000001	.008	3
		16	2	0.09	.14	.00001	.014	2
Biwabik iron formation	Peif	7	0.16	0.001	.3	.00001	.3	3
		17	0.36	0.0075	.4	.0001	.4	2
		27	0.3	0.001	.4	.001	.4	1
		37	26	0.02	.4	.001	.4	1
Shale/siltstone	Peg	8	0.1	0.01	.03	.000001	.003	3
		18	2	0.1	.05	.00001	.005	2
Surficial aquifer		38	7.4	0.16	.15	.01	.015	1
		39	1	0.05	.15	.01	.015	1
		40	5.2	0.1	.15	.01	.015	1

2.2. Conceptual flow model

Four HU10-scale (USGS et al., 2013) watersheds form the study area: Birch Lake, Stony River, Isabella River, and Kawishiwi River (Fig. 1). Surface water flows north and west from Birch Lake and the Kawishiwi River watershed through the Kawishiwi River and several lakes to the BWCAW. Rivers from the Stony River and Isabella River watersheds flow into the Birch Lake watershed (Figs. 1 and 3). Lakes and wetlands connected by low-gradient rivers cover much of the study area which generally has relief less than 10 m from a divide to nearby lakes and rivers.

A surficial aquifer consisting of glacial till or sand and gravel generally less than 3–6 m thick covers the area (Mast and Turk, 1999). Hydraulic conductivity (*K*) of the sand/gravel surficial aquifer ranges from 0.000003 to 1070 m/d, or over nine orders of magnitude, for the surficial aquifer (Siegel and Ericson, 1981; Stark, 1977; Winter, 1973). Well yields in the Kawishiwi watershed are less than 54 m^3/d (Siegel and Ericson, 1981) reflecting the very thin to nonexistent surficial aquifers.

The Duluth Complex (Fig. 2) is a low-permeability intrusive formation with a very low *K* (Table 1) except possibly near some of the infrequent faulting, on which there is little available hydrogeologic data (Miller et al., 2002; Thorleifson, 2008), and in the upper 30 m which is relatively fractured with well yields from 27 to 82 m^3/d. The plutonic rocks have primary porosity up to 3%, but the effective permeability is very low because the pores are isolated (Stark, 1977). The specific capacity of wells in the Duluth Complex ranges from 0.36 to 1.97 m^3/d/m. In bedrock, fractures control permeability and secondary porosity, and also flow paths. Porosity in the fractured Biwabik formation is as high as 50%.

Total flow, or total runoff, from the watershed above the gage for US Geological Survey gaging stations in the area (Fig. 3), was divided into direct runoff and baseflow using methods of Lim et al. (2005, 2010) (Table 2). Yield is total stream flow per area and ranges from 21.6 to 31.2 cm per year (cm/y). Baseflow varies from 14.4 to 24.2 cm/y, although for watersheds with more than 160 square km, it varies from 17.0 to 20.8 cm/y. Assuming recharge is baseflow distributed over drainage area (Cherkauer, 2004; Scanlon et al., 2002), the average recharge is 19.2 cm/y, or 0.00052 m/d for the study area.

Based on gages 7, 5, 2, and 1 (Table 2), subwatersheds Birch Lake, Stony River, Isabella River, and Kawishiwi River (Fig. 3) yield recharge equal to 0.00052, 0.00047, 0.00049, and 0.00054 m/d, respectively. However, the low rates for the small watersheds, Dunka River near Babbitt and Filson Creek gages, 0.00044 and 0.00046 m/d, respectively, illustrate the heterogeneity of the recharge distribution.

Many factors control the recharge distribution, including wetlands, soil types, and soil landforms, including whether the soil is well drained, whether the soils contain substantial peat (Siegel et al., 1995), and whether the bedrock is shallow. Using several statewide GIS soils databases (Land Management Information Center, 1996), maps of wetland coverage, hydrologic soil classification (NRCS, 2007), soil type, surface and subsurface permeability were developed to illustrate the variability (Fig. 4a through e) (Cummins and Grigal, 1980).

A commonly-used method for estimating regional-scale recharge in Minnesota for areas less than 5000 km^2 based on precipitation, growing degree days, specific yield (based on Rawls et al. (1982)) and baseflow recession indices (Delin et al., 2007; Lorenz and Delin, 2007) yielded a recharge estimate for the

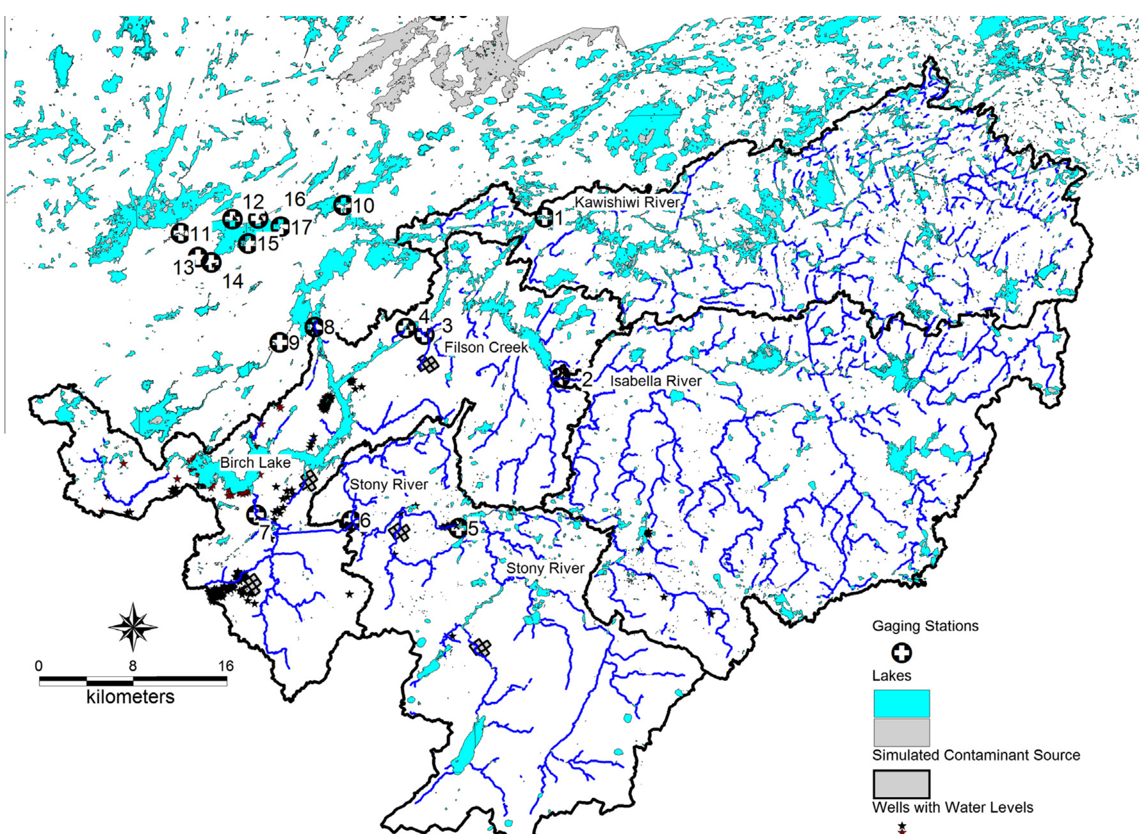

Fig. 3. Location of wells with water levels, gage stations, perennial streams, and lakes in the study area. See Table 2 for a list of gage stations. *Source:* http://mcc.mn.gov/gis.html#state_minerals_head.

Table 2
U.S. Geological Survey gaging stations and station parameters. See Fig. 3 for the location. BFI is baseflow index, the proportion of total flow that is baseflow.

No	USGS site no	USGS site name	Area (km^2)	Avg flow (m^3/d)	Avg runoff (m^3/d)	Base flow (m^3/d)	BFI	Rech (m/d)
1	05124480	KAWISHIWI RIVER NEAR ELY, MN	657.9	461,164	108,226	352,938	0.77	0.00054
2	05124500	ISABELLA RIVER NEAR ISABELLA, MN	883.2	686,970	256,400	430,571	0.63	0.00049
3	05124990	FILSON CREEK IN SESW SEC. 24 NEAR WINTON, MN	25.0	18,642	7211	11,431	0.61	0.00046
4	05125000	SOUTH KAWISHIWI RIVER NEAR ELY, MN	0.05	991,271	248,969	742,301	0.75	0.00000
5	05125500	STONY RIVER NEAR ISABELLA, MN	466.2	307,811	89,744	218,067	0.71	0.00047
6	05126000	DUNKA RIVER NEAR BABBITT, MN	138.3	94,826	34,147	60,679	0.64	0.00044
7	05126210	SOUTH KAWISHIWI R ABV WHITE IRON LAKE NR ELY, MN	2167.8	1,573,205	443,911	1,129,295	0.72	0.00052
8	05126500	BEAR ISLAND RIVER NEAR ELY, MN	177.4	105,117	28,451	76,666	0.73	0.00043
9	05127000	KAWISHIWI RIVER NEAR WINTON, MN	3185.7	2,419,154	746,217	1,672,937	0.69	0.00053
10	05127205	BURNTSIDE RIVER NEAR ELY, MN	178.7	145,526	34,005	111,521	0.77	0.00062
11	05127207	BJORKMAN'S CREEK NEAR ELY, MN	3.5	2626	1239	1388	0.53	0.00039
12	05127210	ARMSTRONG CREEK NEAR ELY, MN	13.7	11,186	4516	6670	0.6	0.00049
13	05127215	LONGSTORFF CREEK NEAR ELY, MN	22.9	18,984	7403	11,581	0.61	0.00051
14	05127219	SHAGAWA RIVER Trib AT ELY, MN	1.8	259	155	103	0.4	0.00006
15	05127220	BURGO CREEK NEAR ELY, MN	7.9	7977	3477	4499	0.56	0.00057
16	05127230	SHAGAWA RIVER AT ELY, MN	256.4	219,331	49,307	170,025	0.78	0.00066
17	05127500	BASSWOOD RIVER NEAR WINTON, MN	4506.6	3,284,031	724,319	2,559,712	0.78	0.00057

Kawishiwi watershed of 20–30 cm/y. This estimate is similar to the total runoff yield and exceeds the recharge estimates made herein by approximately 20–30%. Surface storage in wetlands and small lakes and subsurface storage in the unsaturated zone and groundwater, which support long-term baseflow (Sophocleous, 2002), could explain the differences among estimates. Microscale topography embedded within larger flow systems connected by small surface drainages and interflow causes large variability in flowpath length (Winter, 1998). This may cause surface runoff hydrographs to have long receding legs which resemble groundwater discharge and are difficult to separate from the runoff hydrograph which may cause errors in estimates based on statistical analyses using baseflow recession (Delin et al., 2007; Lorenz and Delin, 2007).

Baseflow follows a seasonal pattern. Average monthly river flow at the Kawishiwi River near Ely gage peaks at more than 80 cm/y during May just two months after the low flow of less than 8 cm/y recorded in March. Much recharge would occur during this snowmelt freshet flood because water runs on the ground surface and river and stream levels are higher than the water levels in the streambanks. After reducing to less than 20 cm/y by September, the baseflow fluctuates between 10 and 20 cm/y until reaching its low in March.

2.3. Conceptual transport model

Waste rock and tailings developed from ore bodies in the study area would likely become sources of AMD-related contaminants because of accelerated oxidation of the ore's high sulfide contents (Bain et al., 2000; EPA, 1994; Jacobs et al., 2014; Johnson and Hallberg, 2005; Lapakko, 1988; Lefebvre et al., 2001; Nash and Fey, 2007; Polymet Mining, 2013b, 2012). Oxidation within a surface waste rock dump, the most common means of disposal (Lottermoser, 2010; Nash and Fey, 2007), is complex due to multiphase flow within the rock (Lefebvre et al., 2001). Pathways are either across the ground surface (Nordstrom, 2011) or through poorly-buffered groundwater (Bain et al., 2000; Jones et al., 2014; Mayes et al., 2007; Siegel, 1981; Siegel and Ericson, 1981) to streams. Often, the existence of groundwater seepage containing a contaminant load is found only through tracers or synoptic sampling that finds a load at a certain location not accounted for by surface samples (Kimball et al., 2002).

Waste may be backfilled underground to submerge the waste more quickly and decrease the discharge of contaminants (Johnson and Hallberg, 2005). This means of disposal can be a significant short-term source of contaminants (Kohfahl et al., 2004; Neal et al., 2005; Runkel et al., 2013) as the recovering groundwater submerges backfilled waste and leaches oxidation products into the groundwater.

Sulfate transport in the Rainy Headwaters is conservative because of a lack of buffering in the watershed and in general is conservative at typical concentrations (>100 mg/l) (Nordstrom, 2011, 2008). Most transport through bedrock in this watershed is through fractures that have limited surface area limiting contact time with any carbonate rock. Sulfate transport in groundwater flow to streams has responded conservatively in other similar situations (Neal et al., 2005; Nordstrom, 2008; Runkel et al., 2013). A good example is Straight Creek in the Red River Valley of New Mexico; groundwater flowed through an unconfined debris-fan aquifer without attenuation of metals and with sulfate being diluted by fresher groundwater inflows (Nordstrom, 2008), as modeled herein.

2.4. Numerical flow and transport model

A reconnaissance-level numerical flow and transport model using MODFLOW-2000 (Harbaugh et al., 2000) and MT3DMS (Zheng and Wang, 1999) was developed to simulate the conceptual models described above. Model flowpaths and contaminant travel times were estimated using the MODPATH code (Pollock, 1994). The dominant model cell size is a 500-m square which approximates the 16.2-hectare mining leases, expanding to 1000-m squares away from the leases in the upper Kawishiwi and Isabella River watersheds (Fig. 5). Three model layers represent the general stratigraphy, with layer 1 being the surficial till and sand/gravel layer and layers 2 and 3 being bedrock, with layer 2 bedrock being more fractured with higher K (Table 1). The top elevation was based on 30-m and 10-m digital elevation models (DEMs) (Fig. 5). Layer 1 was 15-m thick, based on the median and mean depth to bedrock being 14 and 17.4 m. Layer 2 thickness was set so that the total thickness of layers 1 and 2 equaled 140 m. The bottom of layer 3 was set at elevation −1000 m.

MODFLOW DRAIN boundaries, head-controlled flux boundaries that only allow water to leave the model domain, were specified for larger lakes and rivers which cover most of the area due to close surface–groundwater connections (Fig. 6). The lake boundary head was set one m below the top elevation of the model cells so that lakes would receive inflow only when the groundwater level is close to the ground surface. The river head was set five m below the average top elevation of each model cell to simulate discharge to rivers with embedded channels. General head boundaries (GHB), with head and distance to head based on lake water levels just across the boundary, allow groundwater to cross the northern

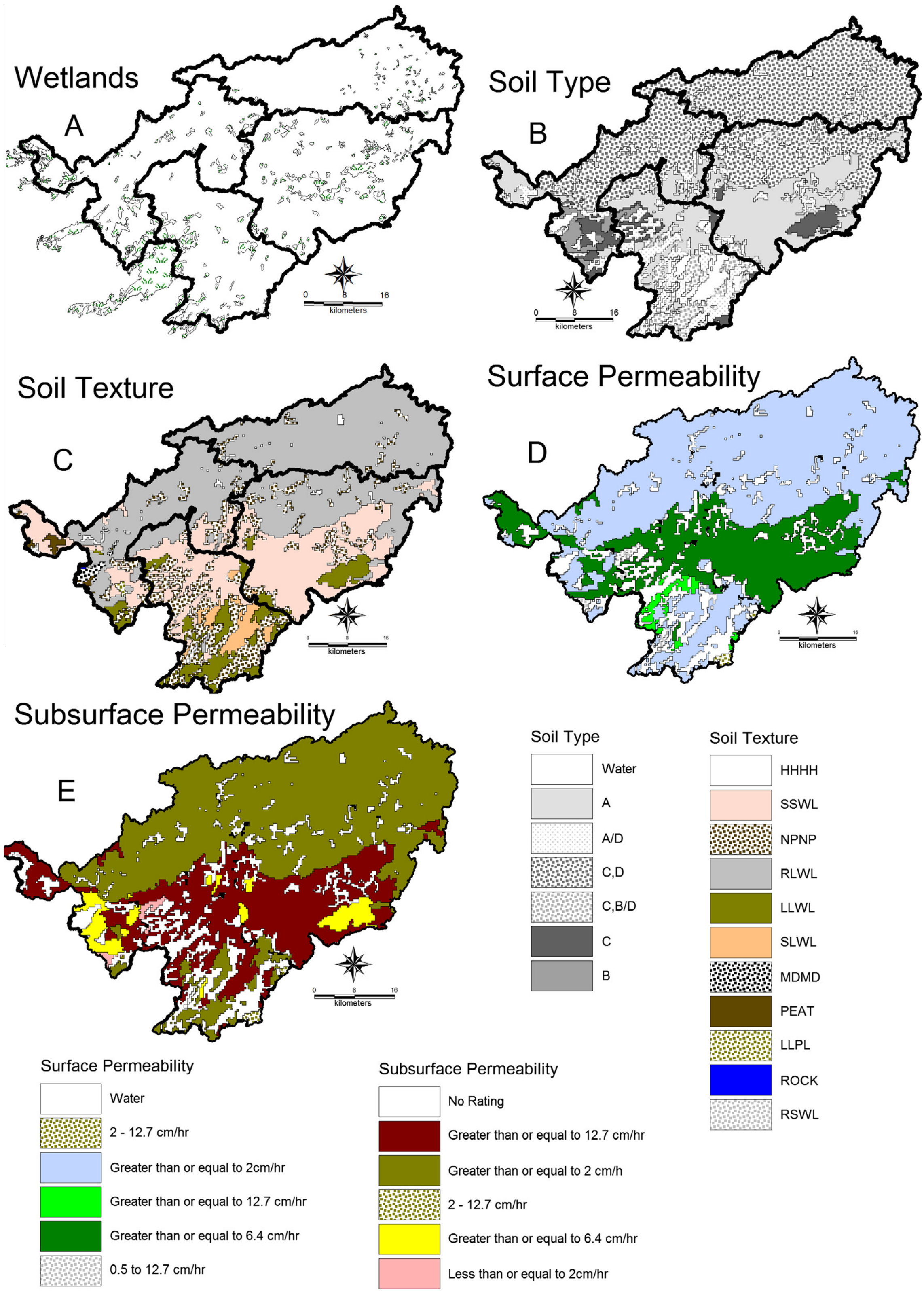

Fig. 4. Distribution of wetlands and soil types across the study area (Cummins and Grigal, 1980). Stream file Strm_baseln3, lakes and wetlands from Dnr100khydrography, from www.mngeo.state.mn.us/chouse/metalong.html. Soil type: 1234; Factor 1: Texture of soil below 5 feet, S is sandy, L is loamy, C is clayey, X is mixed sand and loam, Y is mixed silt and clay, R is bedrock; Factor 2: Texture of soil in top 5 feet, as for factor 1; Factor 3: Drainage, W means well-drained, P means poorly drained; Factor 4: Color, D is dark, L is light.

Kawishiwi watershed boundary (Fig. 6) at topographic low points. Recharge zones were specified based on subwatershed (Fig. 6), with rates set so that recharge equals the measured baseflow in the primary rivers.

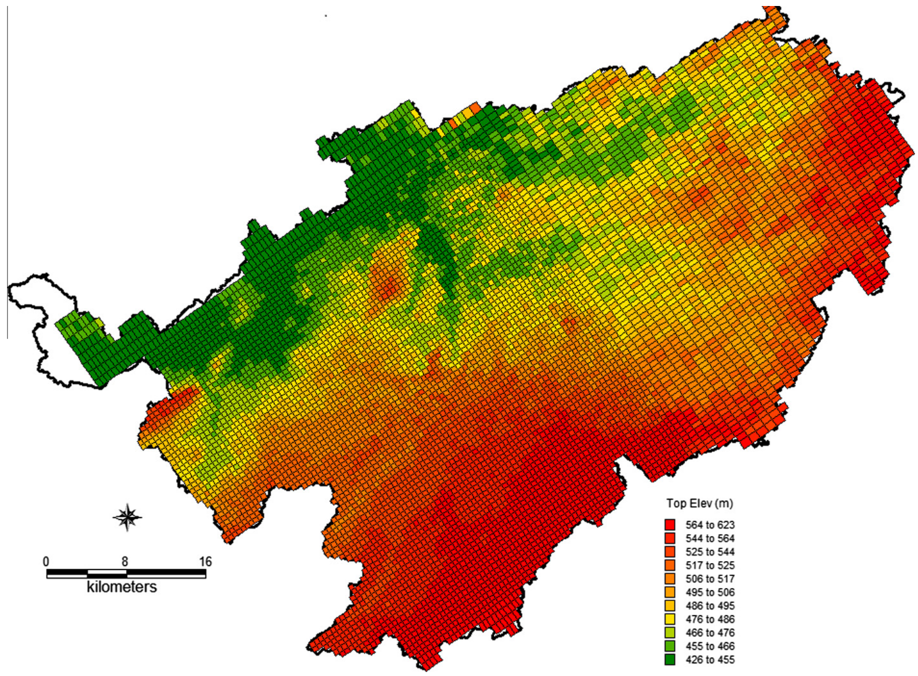

Fig. 5. Model grid and layer 1 top elevation by cell in meters.

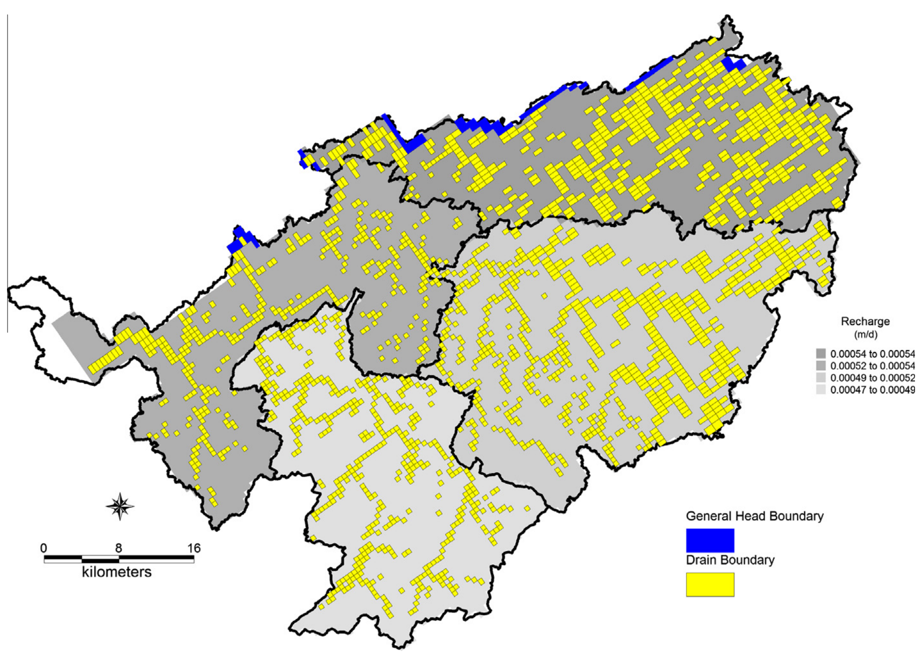

Fig. 6. Location of DRAIN and recharge boundaries in layer 1 and general head boundaries (GHBs) in layer 2. Final recharge rates equal 0.00054, 0.00049, 0.00047, and 0.00052 m/d for zones 2 through 5, respectively.

Transport modeling of sulfate was completed using MT3DMS (Zheng and Wang, 1999). In addition to advection controlled by the flow model and effective porosity (Table 1), the variation of which would simply proportionally increase or decrease contaminant arrival time, dispersion affects concentration by spreading the contaminants along and transverse to the flow path (Fetter, 1999). Dispersivity is a function of length of the flow path from source to sink (Fetter, 2002; Xu and Eckstein, 1995). The longest paths emanate from particle placement at mid-level in layer 3, and are approximately 33,000 m, which is much longer than the length at which further increases in dispersivity with length became negligible, 1000 m (Xu and Eckstein, 1995). Because the flow paths in this model domain vary from less than 100 m to as much as 33,000 m, setting dispersivity for 1000 m is reasonable and avoids changing D for each source. The apparent longitudinal dispersivity therefore is 11.8 m. The transverse and vertical dispersivity equals 0.2 and 0.1 times the longitudinal dispersivity (Schulze-Makuch et al., 1999). This analysis treats sulfate as conservative to estimate the sources which could have the most significant impacts without relying on estimates of reactivity to attenuate the risk.

2.5. Flow model calibration

Steady state calibration is the process of adjusting *K* and boundary reach conductance so that simulated steady state groundwater levels match observed target groundwater levels and that simulated boundary fluxes equal measured fluxes (Anderson and Woessner, 1992). Within the four study area watersheds, the groundwater level database (MN Geological Survey and MN Department of Health, http://www.mngeo.state.mn.us/chouse/metadata/wells.html) contained 1238 wells with 362 having depth to water, water level elevation, well depth, and depth to bedrock (Fig. 3). There are few groundwater level measurements in the headwaters of the Isabella River and Kawishiwi River watershed, so nine artificial targets weighted 0.3 were set there in each model layer with head set equal to ground surface elevation minus 4 m, similar to methods of Halford and Plume (2011).

Calibration was deemed sufficiently reliable for comparative testing of the siting of contaminant sources (Nordstrom, 2012) when continued parameter estimation yielded composite scale sensitivity (CSS) within a few orders of magnitude for all parameters (Hill and Tiedeman, 2007), the parameter estimates ceased changing during automated calibration, and the sum of squared residuals (SSR) and actual mean residual was at a minimum.

2.6. Flow path simulation

Advective pathways from the mineral leases (Fig. 2) to respective discharge points, a DRAIN boundary, were determined and timed using MODPATH (Pollock, 1994). Contaminant particles were placed in approximately 630 model cells coincident with mineral leases (Fig. 2) at five different levels – the middle and top of layer 3 and the middle and top of layer 2 to represent contaminants leaching from underground, and the top of layer 1, or the water table, to represent surface leaks through the vadose zone.

2.7. Transient model scenarios

The modeling scenarios are generic but representative of mining which could occur in this area (Parker and Eggleston, 2014; Polymet Mining, 2013a) with relatively stringent and well-enforced regulations. The simulated contaminant loads are similar to values expected for waste at the nearby proposed Polymet mine (Polymet Mining 2013a, 2013b, 2013c, 2012) because the ore is of similar sulfide content and the hydrogeology is also similar. The two model scenarios include one for which waste is backfilled into underground workings and one for waste piled onto the ground surface where leaching can occur. Model simulations were transient, with a one-year period (20 time steps with 1.2 multiplier) of waste input, as described below, and a 1000-year period (60 time steps, 1.1 multiplier) of long-term transport. The one-year period of contaminant injection is conservative because if engineering plans go wrong or a leak goes undetected, the contaminant source could continue for much longer. Groundwater fluxes continue as simulated in steady state except for the small amount of injection used to simulate the underground waste. Simulations do not consider mine dewatering or other water management activities.

Waste in underground workings oxidizes, but the rate decreases manyfold after the water level recovers and saturates the waste (Demchak et al., 2004; Kohfahl et al., 2004). As the water level recovers, it flows through the waste leaching a contaminant load into the surrounding groundwater. To simulate this leaching as a sulfate load to groundwater, a low-flow (60 m^3/d), high-concentration (10,000,000 μg/l, based on concentrations expected at the nearby proposed Polymet mine (Polymet Mining, 2013a, 2013c)), injection well was placed within each of five model cells in model layers 2 and 3 at five locations representative of the mineral leases (Fig. 2). The placement of a sulfate source at different levels and locations allows consideration of the sensitivity of the flow paths emanating from the different locations of backfilled waste. The one-year period accounts for the probable short-term cessation of oxidation as groundwater levels recover.

Above-ground sources, waste dumps, are simulated as a 10,000,000 μg/l concentration added to the natural recharge over six cells located as for the underground sources. This concentration is an order of magnitude higher than observed in the field for similar ore (Lapakko and Olson, 2015), but justified because the samples in that paper were taken downstream of the source after some dilution. The one-year simulation period is the equivalent of the operator developing a waste rock storage area and covering it after discovering a leak, completing reclamation over a one-year time period, or moving the waste to a different location. Total load varies from 2,847,000 kg to 2,573,250 kg depending on cell size.

3. Results and discussion

3.1. Calibration

The final SSR was 4405 and 2173 for unweighted and weighted targets, respectively. The standard deviation is 4.2% and 3.0% of the 150-m range in observations, from the lowest to highest groundwater elevation. There is no detectable trend with observed groundwater elevation and simulations should yield no bias. Final Ks for parameter zones (Fig. 7) are shown in Table 1.

Some of the wells cluster so closely (Fig. 3) they represent essentially the same information, so they were thinned, first by keeping just one well per layer within 200 m of each other, and second by keeping just one well per model cell with the head target equal to average head of the wells remaining after the first thinning (Wellman and Poeter, 2006).

Zones with few head observations were not sensitive so the final parameter values were selected on formation type and on values necessary to generate reasonable head values (ASTM, 1998). Horizontal conductivity (Kh) values in the surficial aquifer are high but within the observed values and vertical conductivity (Kv) values reflect highly stratified till. Each bedrock formation *K* varies over at least two orders of magnitude. Conductivity decreases with depth for most formations as expected due to compaction occurring due to overburden and less weathering with depth. In zones 32 and 3, Kh was much less than Kv (Table 1) which reflects a tendency for vertical flow in the upthrust Duluth Complex (Miller et al., 2002).

Simulated heads generally show groundwater movement from southeast to the north and northwest. The water table in the upper layer follows the irregular topography while in the deep layer contours reflect a consistent slope to the northwest with an upward gradient from layer 3 toward Birch Lake in the northwest portion of the study area near its primary surface water outlet (Figs. 3 and 8).

Recharge (Fig. 6) and simulated discharge were nearly equivalent in the Isabella and Stony River watersheds but varied by from 10% to 20% in the other watersheds due to interbasin groundwater flow. Percent differences among watersheds are small and indicate that the distribution of recharge and discharge through the model domain is accurate. All ten river reaches gain more flow in their lower reaches near their outlet due to converging flow. Ten simulated lakes received zero discharge because their bottom was above the water table which reflects their location in the upper recharge portions of the watersheds.

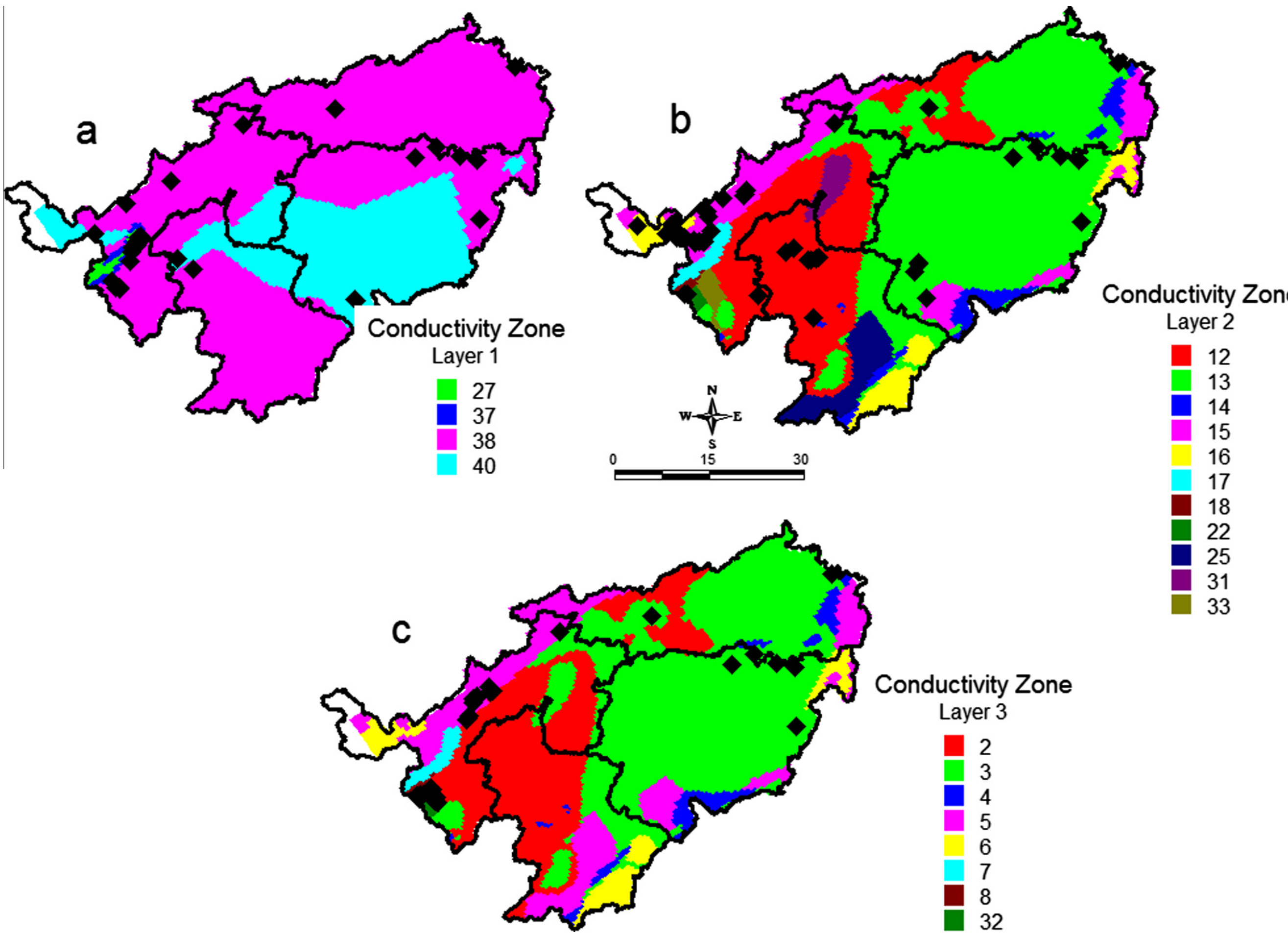

Fig. 7. Hydrogeology zones for three layers and steady state head target locations (black squares). Table 1 shows final conductivity values.

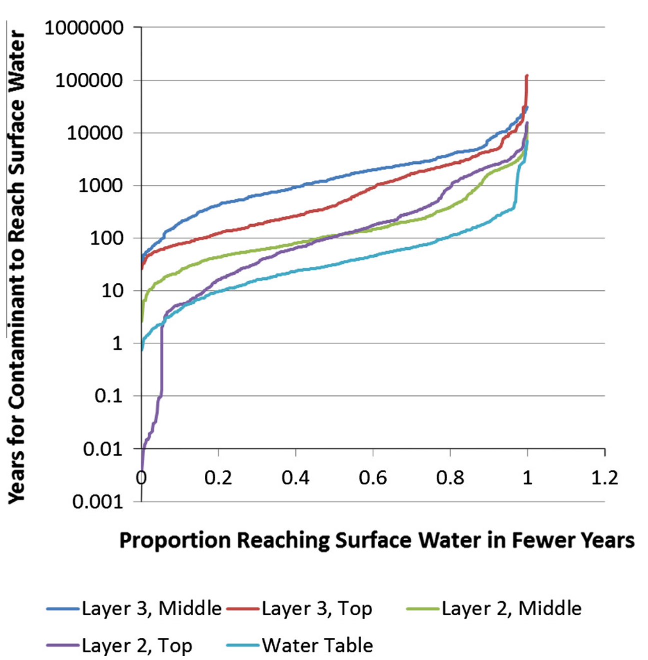

Fig. 8. Proportion of particles reaching surface water sources in a number of years for any location in the specified layer.

3.2. Particle tracking

Particles introduced at the middle level of layer 3, or about 750 m bgs, required the longest and those introduced at the water table required shortest time to reach surface water (Fig. 8), with minimum times varying from 26 years to less than a year, depending on layer. About 2.8% and 23.5% of the particles released at the top of layer 3 and middle of layer 2 reached surface water in less than 50 years, respectively (Figs. 8 and 9). Flow paths are longest through layer 3 because of the layer's thickness. The shortest pathways occur where a river boundary is close and there is an upward gradient, such as near Birch Lake (Figs. 3 and 9).

The shortest pathways, requiring less than two years, were from water table sources starting close to rivers (Fig. 9). Most particles reached surface water quickly, with 21% reaching surface water within 10 years and 62.9% in 50 years (Fig. 8). Longer transport times were for particles being transported deeply into layer 2 or 3 (Fig. 9).

The primary control on transport time, other than distance from the sink, is whether the particle sinks deeper into the bedrock (Gburek and Folmar, 1999), which would result from normal groundwater circulation. Contaminants released where they sink would present a less substantial risk to downstream resources. However, long pathways could result in contamination remaining a risk long after mining has ceased if it does not attenuate.

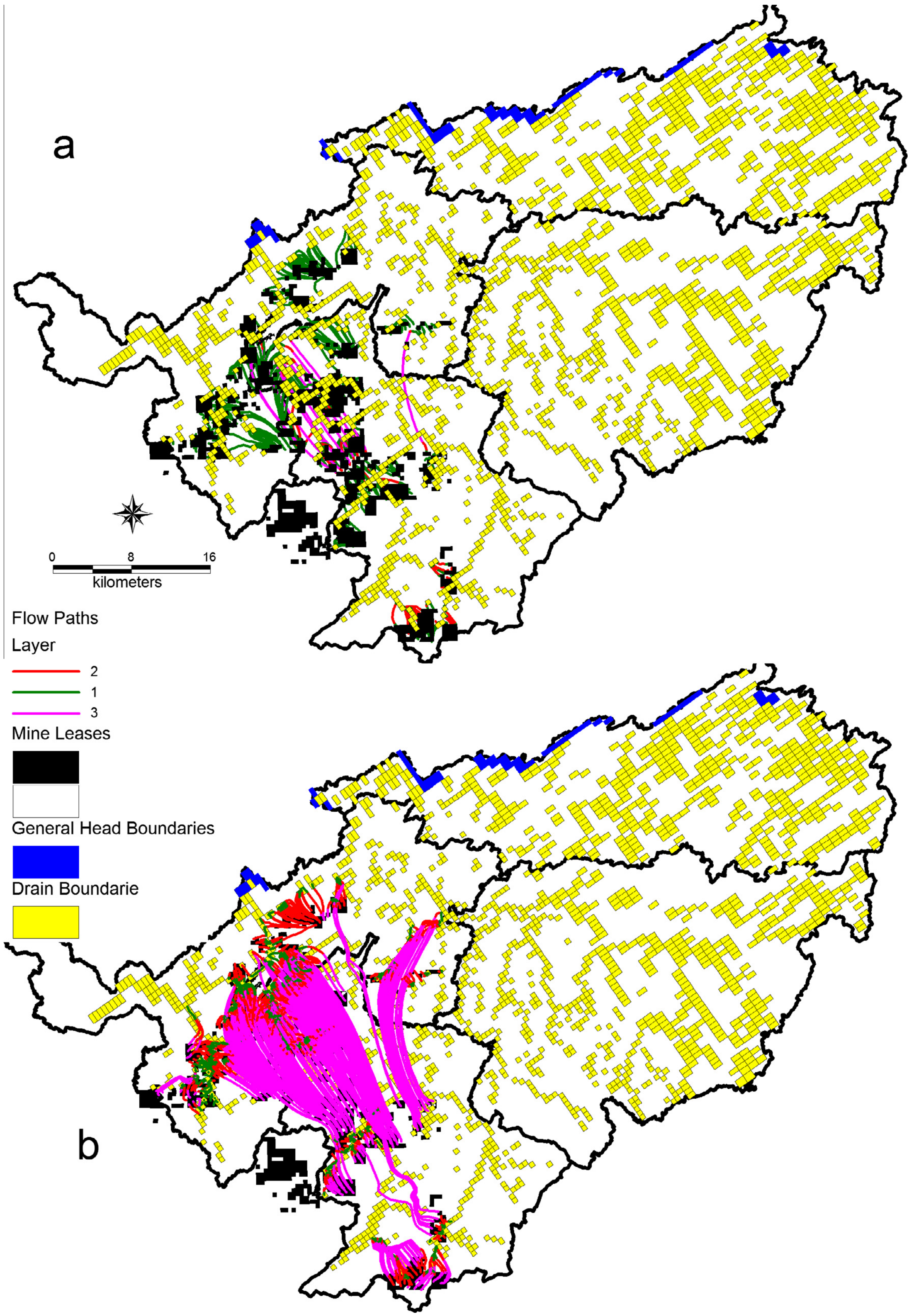

Fig. 9. Particle tracking for particles introduced at model cells near the mineral leases at the water table (a) or the top of layer 3 (b).

3.3. Contaminant transport modeling

Detectable sulfate, at 1 μg/l, expanded through the groundwater domain variously depending on the location of the source (Fig. 10). Sulfate from deep sources flows to the northwest with various amounts of lateral and vertical dispersion (Fig. 10a and b). Detectable sulfate reached up to 2.5 km from the source with up to 1.5 km lateral dispersion (Fig. 10a). Near sources 1 and 2 (source numbers specified in Fig. 10a), vertical dispersion caused contours in the surface layer to almost mirror those in the source layer. Sulfate originating at depth transports further in a thousand years than does sulfate originating on the surface which is limited in extent by discharge to rivers (Fig. 10b).

Sulfate originating from surface sources disperses to the northwest from headwaters and radially from near-river sources (Fig. 10c). Sulfate from source 4 expanded to the northeast but was constrained from expanding to the south by a steep topographic slope (Fig. 10c). Between ten and a hundred years, the sulfate contours did not spread significantly (Fig. 10c) due to discharge to surface waters while concentrations near the source decreased by an order of magnitude.

The amount of groundwater affected depends on the source, flow paths and dispersion. Contaminants eventually reach surface water, but at widely varying travel times (Fig. 8). The load is most important with respect to discharge to the rivers, and peak loads reach the various river reaches at times depending on distance and whether the source is surface or underground (Table 3). Peak loads from underground sources reached rivers in from ten to forty years and from surface sources in less than five years (Table 3 and Fig. 8).

The highest sulfate loads from surface sources were up to two orders of magnitude higher than those from underground sources and reach their peak at rivers in the Stony River watershed at the end of the first year, reflecting their close proximity of the source to the rivers. Filson Creek receives the highest load which translates to a concentration of near 120,000 μg/l. Peak loads reach Birch Lake and Dunka River after two to five years but are lower than for Stony River and Filson Creek, due to dilution over the longer flow path. Surface leaks reach the streams quicker and have higher concentration due to there being much less attenuation due to the shorter flow paths.

Baseflow makes up as much as 70 percent of the flow in rivers in this area, so the simulated loads (Table 3) would not be significantly diluted during low flows. During critical low flow periods (Winterstein et al., 2007) the sulfate concentrations would equal that determined from the groundwater load and flux discharging to the rivers (Mayes et al., 2007; Runkel et al., 2013).

The sulfate loads reaching the rivers vary substantially based on the location and depth of the sources. Surface sources contribute load to rivers much sooner and with a higher peak than do underground sources. The load reaching the relatively close-by Birch Lake or Dunka River is much higher but also much shorter-lived

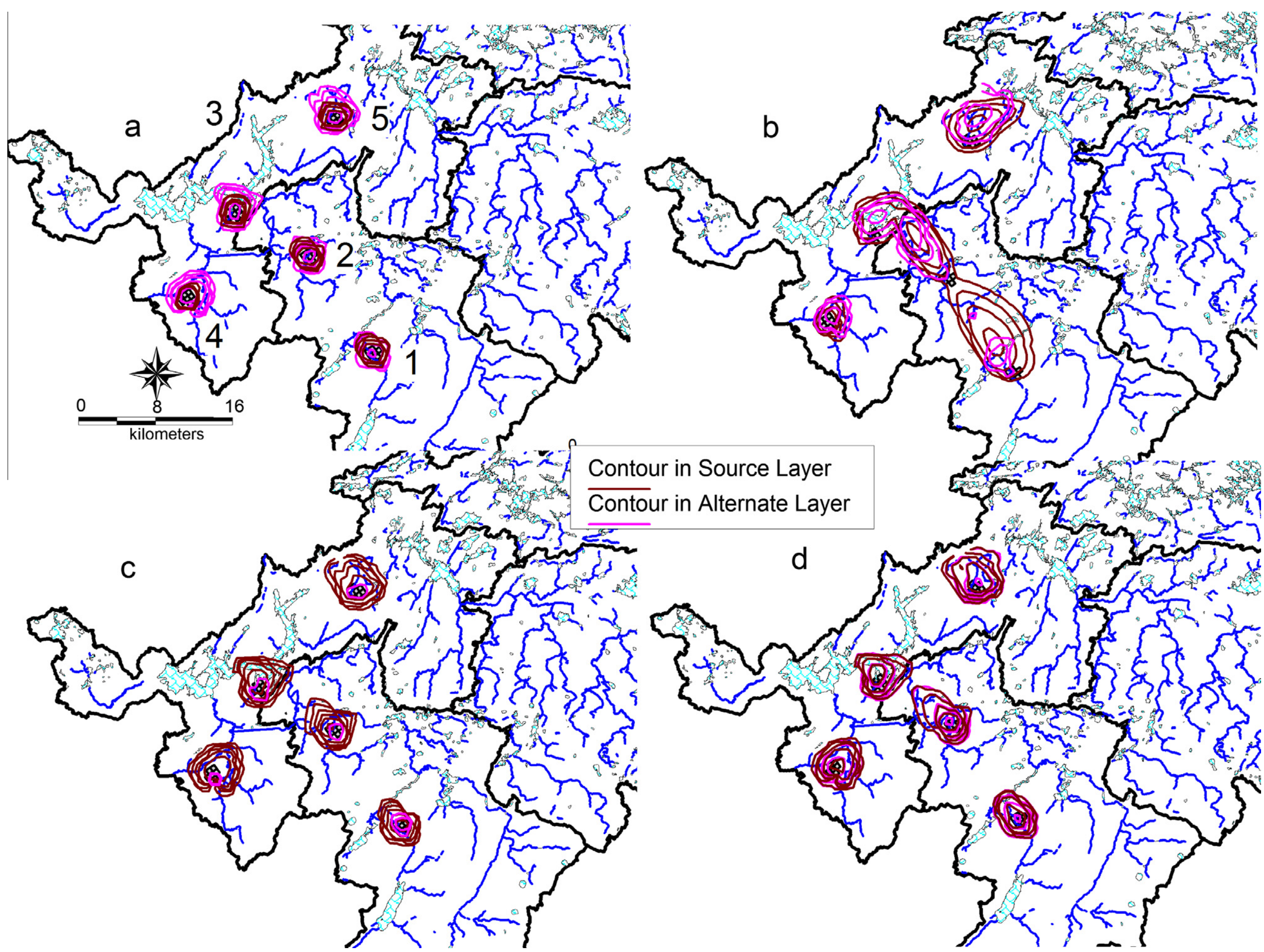

Fig. 10. Concentration contours for (a) underground source after 10 years, (b) underground source after 1000 years, (c) surface source after 10 years, and (d) surface source after 100 years. Underground source is layer 3, surface source is layer 1. Alternate layer is layer 1 in (a) and (b), layer 3 in (c), and layer 2 in (d). The contaminant source number is shown in (a). Contours range from 1 to 100,000 μg/l from outer to inner.

Table 3
Sulfate load (kg/day) discharging to reaches 1, 71, 72, 73, and 75 at various times corresponding to stress periods (1 or 2) and time step (number in parentheses). Steady state discharge to reach 1, 71, 72, 73, and 75 is −74,916, −73,295, −31,383, −168,050, and −11,439 m^3/d, respectively. The discharges are stated as a negative because they represent a loss from the groundwater domain. Reach 1 is Birch Lake, Reach 71 is Dunka River, Reach 72 is Stony River between Babbitt and Isabella, Reach 73 is Stony River above Isabella, Reach 75 is Filson Creek nr Ely, as shown in Fig. 3. Other reaches did not receive substantial load.

Period	Years	Reach				
		1	71	72	73	75
Deep sources						
1(20)	1	0.1	0.3	10.5	18.8	28.7
2(14)	10	12.4	14.9	38.7	43.0	45.6
2(21)	21.8	16.7	17.7	36.1	37.0	36.3
2(24)	30.2	17.5	18.1	32.7	32.9	31.3
2(27)	40.9	17.2	18.3	28.3	28.4	26.4
2(36)	99.4	9.9	15.3	12.2	12.4	11.9
2(60)	1000	1.3	0.3	0.0	0.2	2.2
Surface source						
1(20)	1	28.7	56.8	1033.5	1118.2	1378.2
2(1)	1.3	54.4	101.3	955.0	909.0	929.2
2(3)	2.1	115.2	184.2	782.6	589.4	524.5
2(5)	3	159.3	222.6	612.2	390.4	377.1
2(9)	5.5	161.7	192.8	313.1	187.7	211.1
2(11)	7.1	131.8	151.4	203.4	136.0	150.8
2(14)	10	86.9	94.3	105.0	90.2	93.9
2(21)	21.8	37.2	39.9	30.5	40.5	45.7
2(36)	99.4	8.2	25.4	10.1	15.0	15.2
2(60)	1000	0.0	0.0	0.0	0.0	0.0

than the load reaching the other rivers because distance slows the transport time assuring it will continue further into the future. Burying waste or placing it further from the resources to be protected will decrease the load reaching those rivers and substantially decrease the potential impacts of mining.

4. Conclusion

The reconnaissance-level fate and transport model presented herein simulates groundwater flows and estimates where, when, and at what concentration sulfate would discharge for various mine development scenarios in the Rainy Headwaters watershed. The model allows a comparison among sources to assess where mines and their associated waste facilities would cause less risk from spills and alternatively where mines could be riskier. Similar models should be developed for watersheds throughout the world that have substantial mineral deposits to prioritize development or alternatively to decide development is too risky. This type of model also shows where additional data should be collected, such as along the predicted pathways to reduce the uncertainty in advective flow rates and dispersion, a common need in most watersheds undergoing development (Caruso et al., 2008).

Groundwater with substantial contaminant concentrations discharges to streams whether sourced from deep underground or the ground surface. Even relatively short-term leaks on the surface could cause substantial loads to reach the rivers and valuable downstream resources. Longer-term leaks could cause peak concentrations reaching the rivers to be much higher than simulated herein. Underground sourced contaminant discharges last longer but have lower concentrations and are recommended for use in sensitive watersheds globally. In the Birch Lake watershed, leases trending southwest to northeast would discharge to surface water relatively quickly. Leases in the headwaters of the Stony River watershed would discharge to nearby surface water. These discharges would eventually coincide with critical low flow periods and cause potentially significant damage to rivers and the BWCAW.

Leaks into groundwater commence a long-term process in which contaminants travel to surface waters for a long time after the leaks have ceased discharging. Contamination may not be obvious until after a mine closes and impacts can continue for decades, with substantial concentrations still reaching rivers for hundreds of years even if the leaks cease. These factors should be considered when establishing bonds for long-term water quality remediation and modeling such as presented herein can be used to estimate the potential for future remediation.

Although mines are located based on the minerals, advance planning and analysis could avoid siting mine facilities where failure would cause too much natural resource damage. Reconnaissance-level modeling can provide the basis for more complete watershed-level studies as suggested to assess a watershed (von Guerard et al., 2007) and to determine where additional geochemical and hydrogeologic data should be collected (Caruso et al., 2008). Unless there is a clear geochemical sink for the contaminant, treating the transport as conservative will allow better decision making. Mine facilities should be located based on the potential for a leak or spill to damage downstream resources, as predicted with watershed-scale transport modeling. Such planning could lead to certification under responsible mining standards such as IRMA. Some areas should not be mined at all due to the risk to downstream resources.

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