History of Eagle Mine Water Quality Degradation
May 2023

Predicted Pollution

Eagle Mine discharges treated water into groundwater; that water then makes its way to surface seeps that form the headwaters of the East Branch of the Salmon Trout River. According to permitting analysis by the Michigan Department of Environmental Quality (MDEQ), this discharged water was predicted to be higher than background (existing conditions) for several substances.

- Cobalt: the potential concentration of cobalt in discharged water was predicted to be **45** ug/L
    - the applicable Michigan Water Quality Standard was 100, meaning that a mine could pollute water to the level of 100 ug/L
    - the background (existing) level of cobalt in water was **0**
    - Note: Minnesota's WQS for cobalt is 5 ug/L; the Eagle Mine's permit allows the mine to discharge cobalt at nine times the allowable standard in Minnesota
- Nickel: the potential concentration of nickel in discharged water was predicted to be **30.38** ug/L
    - the applicable Michigan Water Quality Standard was 32, meaning that a mine could pollute water to the level of 32 ug/L
    - the background (existing) level of nickel in water was **3**
- Zinc: the potential concentration of zinc in discharged water was predicted to be **73.78** ug/L
    - the applicable Michigan Water Quality Standard was 137, meaning that a mine could pollute water to the level of 137 ug/L
    - the background (existing) level of zinc in water was **13**

These figures are from a worksheet prepared by Michigan Department of Environmental Quality (MDEQ) staff to decide whether a NPDES (federal Clean Water Act surface water discharge) permit would be required for the mine. S. Wolf, GLEAS NPDES Permit Review Summary (Oct. 16, 2006). Re: Kennecott Minerals Eagle Project venting groundwater, pp. 2 and 3. (Contested case hearing Respondent exhibit no. 186)

The MDEQ staff person who prepared the worksheet wrote two memoranda, S. Wolf (Oct. 26, 2006) and (Nov. 6, 2006). MDEQ Interoffice communication Re: Kennecott Minerals Eagle Project - Venting Groundwater Review. (Contested case hearing Respondent exhibits no. 188 and 192) The MDEQ staff person states that "there is reasonable potential for the pollutants in Table 1 (cadmium, copper, selenium, silver) to exceed Michigan's water quality standards (WQS) in the seeps (i.e., the headwaters of the east branch of the Salmon Trout River)." Despite the staff findings, no permit was required for surface discharge at the seeps.

There is no record that would indicate that the seeps are being monitored. *See* Kennecott Eagle Minerals (Feb. 2006). Eagle Project Mining Permit Application Vol. I, Fig. 6-2 (showing proposed surface water monitoring points) (Contested case hearing

Respondent exhibit No. 25); <u>State of Michigan Depart of Environmental Quality (Dec. 14, 2007). Nonferrous metallic mineral mining permit MP 01 2007, pp. 22-23, provision L28</u> (requiring monitoring as shown in Fig. 6-2). (Contested case hearing Respondent exhibit No. 117). Publicly available monitoring data is available online at https://swpcemp.org/monitoring-results/ No data appears on that site that corresponds with the seeps.

Conclusion

Water quality degradation of groundwater by the Eagle Mine was foreseen by MDEQ permitting staff. The Eagle Mine's permit allows levels of water quality degradation (pollution) far greater than existing conditions, although within allowable levels per MDEQ standards. Potential exceedances of Michigan's surface water quality standards were of concern to MDEQ staff. However, there are no permit limits or monitoring requirements for the relevant location, so the actual impact on water quality is unknown.

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Analysis

Analysis of proposed 20-year mineral leasing withdrawal in Superior National Forest☆

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ABSTRACT

The Rainy River Watershed on the Superior National Forest is home to the Boundary Waters Canoe Area Wilderness (BWCAW). It also contains deposits of copper, nickel, and trace metals, and copper-nickel mining has been proposed adjacent to and upstream of the BWCAW. In 2017, the US Department of Agriculture proposed withdrawing land in the Rainy River Watershed within the Superior National Forest from mineral leasing, a position it reversed in 2018. These developments highlight the potential tradeoff between economic benefits from mining and concerns about its negative economic consequences for the local recreational and amenity-based economy. Previous studies of mining in the Superior National Forest focus on static effects on a single industry (e.g., mining) at some unspecified point over a medium-run horizon. We draw on these studies and the economics literature to provide a unified analysis of the effect of the proposed mining development on income and employment over time. Our results suggest that the proposed mining would lead to a boom-bust cycle that is typical of resource extraction economies, exacerbated by the likely negative effect on the recreation industry.

1. Introduction

The Boundary Waters Canoe Area Wilderness (BWCAW), located within the Superior National Forest in northeastern Minnesota along the Canadian border, consists of more than one million acres of connected lakes and rivers. The BWCAW is one of the most visited wilderness areas in the United States, with 150,000 visitors in 2015 (US Forest Service, 2016). Those visitors support a varied outdoor recreation industry in gateway communities, primarily Ely, Minnesota (Hjerpe, 2018). The lakes and rivers outside the BWCAW also attract recreational visits and both seasonal and permanent residents who locate there for the outdoor and lakes amenities.

The region also has rich mineral deposits. The Mesabi Iron Range, the largest iron mining district in North America, extends for nearly 100 miles to the southwest of the BWCAW, with its most northeasterly portion within ten miles of the wilderness boundary (Minnesota

☆ An earlier draft of this study was submitted on August 6, 2018, in letter form, as a comment on the U.S. Forest Service's proposed withdrawal of Superior National Forest land within the Rainy River Watershed from mineral leasing. This revision reflects several updates to the 2018 letter. The most significant of these is that the 2018 letter considered only direct and indirect employment and income. In response to comments received on the original letter, the current revision now includes estimates of induced (spillover) employment and income. This revision also incorporates several other changes. For internal consistency, multipliers are now taken solely from University of Minnesota-Duluth, (2012) for mining and from Hjerpe (2018) for recreation. Additionally, wage rates are all for the Arrowhead county region whereas, in the 2018 letter, some wage rates were statewide. The discussion of related academic literature has been expanded, and procedural recommendations to the US Forest Service made in the 2018 letter have been removed from this version. Taken together, the revisions affect numerical values in the 2018 letter but do not change the conclusions. We thank Steve Polasky, Cathy Kling, John Hinderaker, Tom Landwehr, and two referees for their comments.

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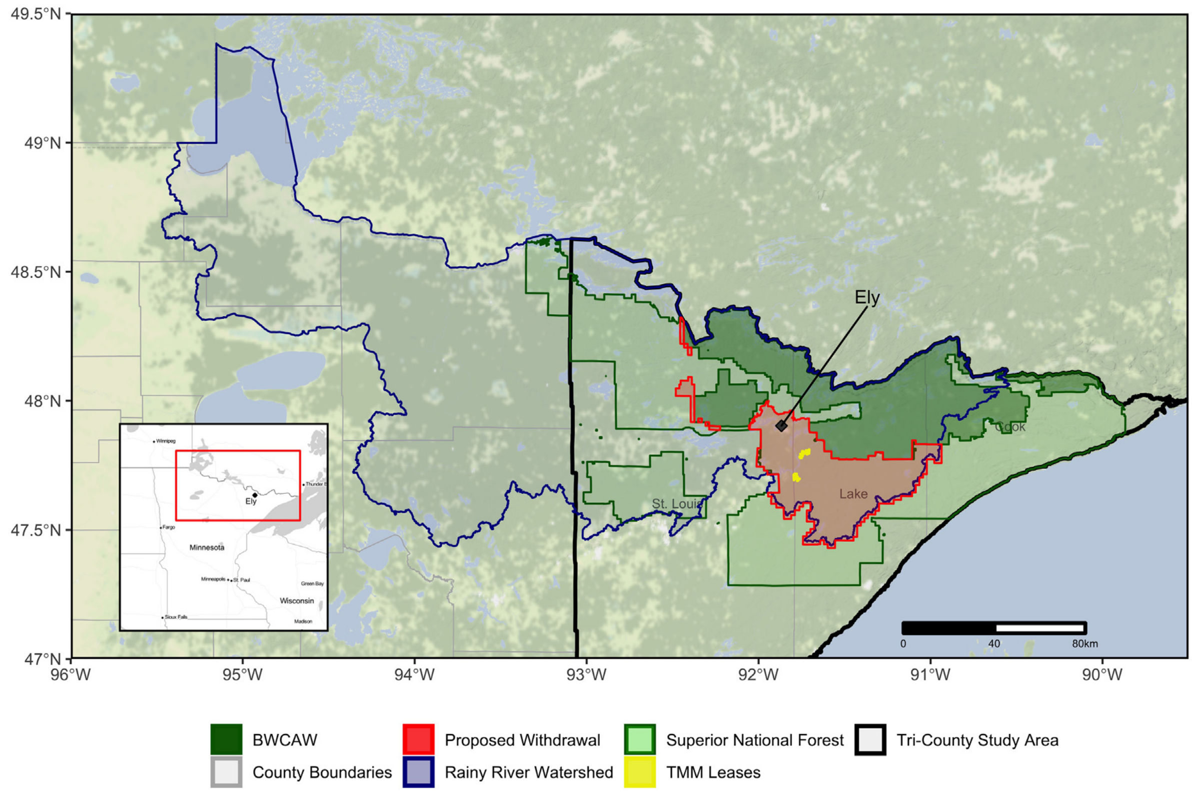

Fig. 1. Map of key hydrological and administrative features of the study area. The area of the proposed withdrawal of mineral rights within the Superior National Forest drains north into the BWCAW and encompasses the TMM mineral leases.
Sources: U.S. Forest Service, 2017, Minnesota Geospatial Information Office, 2020.

Department of Natural Resources, 2017). Although its high-grade iron ore (hematite) has been mined out, taconite mining continues, and taconite mining operations employed 3440 workers in 2016 (Minnesota Department of Natural Resources, 2017). In addition, there is commercial interest in developing copper-nickel mines in deposits both in and out of the Superior National Forest.

In particular, a copper-nickel mine, the proposed Twin Metals Mine (TMM), would be located in a site bordering and immediately upstream of the BWCAW (see Fig. 1).[2] The legal and jurisdictional setting is complex. The proposed site, along with much of the copper-nickel deposit, is on federal land for which mineral leasing rights are administered by the Bureau of Land Management (BLM), though the project also includes several state leases administered by the Minnesota Department of Natural Resources (DNR). The land is within the Superior National Forest, the surface of which is administered by the U.S. Department of Agriculture (USDA), and the USDA must consent to discretionary mineral leasing to ensure compatibility with the resource management objectives of the National Forest System. The USDA also administers the BWCAW, where federal law prohibits mining. Federal mining rights for the TMM project were originally granted in 1966 for a 20-year period with up to three 10-year renewals. The first two of these renewals were granted by the federal government. However, in December 2016 the USDA withheld its consent to renewal, citing concerns about negative impacts on the BWCAW, so the BLM denied the third renewal request.[3] In January 2017 under the Obama Administration, the U.S. Forest Service further proposed to withdraw from mineral leasing approximately 234,000 acres of federal lands within the Rainy River Watershed, which flows north into the BWCAW and which contains the TMM project, and it also initiated the preparation of an environmental impact statement (EIS) to assess the proposed withdrawal.[4]

In September 2018, the Trump Administration cancelled the withdrawal application and EIS,[5] and in May 2019 the BLM reversed its 2016 denial and renewed the TMM leases. TMM submitted its mine application to regulatory agencies in December 2019. The mine now awaits federal and state permitting, even as the federal lease renewal is currently being litigated.[6] Despite its cancellation, the proposed withdrawal remains a key part of the debate over mining near the BWCAW, and legislation that would permanently enact the Obama

[2] The Twin Metals Mine is a copper-nickel mine proposed by Twin Metals Minnesota, a subsidiary of Antofagasta PLC, one of the top ten copper producers by volume in the world. The project is located approximately nine miles southeast of the city of Ely, MN and proposes to mine sulfide-ore from the Maturi deposit of the Duluth Complex geologic formation. The Duluth Complex is one of the world's largest polymetallic deposits and in addition to copper and nickel includes cobalt and platinum group metals. Twin Metals Minnesota anticipates processing 20,000 tons of ore per day from the proposed sub-surface mining operation at the Twin Metals Mine (TMM, 2019a).

[3] Thomas Tidwell, Chief, BLM, Memorandum "Lease Renewal Application Rejected," December 15, 2016 at https://www.blm.gov/download/file/fid/7652.

[4] 82 FR 4282.

[5] https://www.usda.gov/media/press-releases/2018/09/06/usda-removes-roadblock-mineral-exploration-rainy-river-watershed.

[6] The TMM project is subject to multiple regulatory requirements. The project is subject to an environmental review process under the National Environmental Policy Act (NEPA) and Minnesota Environmental Policy Act (MEPA). In December 2019, TMM submitted its Mining Plan of Operations to state and federal agencies for review, thereby initiating the formal NEPA/MEPA Environmental Impact Statement process (TMM, 2019c) (see https://www.dnr.state.mn.us/input/environmentalreview/twinmetals/index.html). Following completion of the EIS, the TMM project must receive permits by various federal, state, and local regulatory agencies to commence construction and operation. These permitting requirements regulate various aspects of the project construction and operation, including hazardous waste management, the disposal of wastewater, and maintenance of air quality standards (TMM, 2019b).

Administration's withdrawal has been introduced in the House of Representatives.[7]

The proposed TMM mine raises a classic conflict between recreational use and conservation on the one hand and mining development on the other, a conflict made more stark by the unique attributes and wilderness status of the BWCAW. Proponents of the mine point to the jobs and income it will create (University of Minnesota-Duluth, 2012; Orr et al., 2018). Opponents point to the risks to the watershed because of potential acid mine drainage and toxin release (Myers, 2016; Pearson et al., 2019), noise and light pollution that would disrupt the wilderness experience and negatively impact the local recreational industry (US Forest Service, 2016), and potential reductions in amenity-based in-migration (Sungur et al., 2014).[8]

The environmental risks associated with sulfide-ore copper mining within the watershed of the BWCAW are potentially economically consequential. Mining and beneficiation processes for underground copper ore generate large volumes of tailings. In a watershed hydrology model of possible mining locations in northeastern Minnesota, Myers (2016) finds that even relatively short-term leaks of tailing materials on the surface at mining locations in the region could cause substantial loads of sulfate, a major product of acid mine drainage, in the rivers and downstream resources of the BWCAW. The economics literature provides some insights concerning the economic costs associated with these adverse environmental impacts. In a study of acid mine drainage-impaired lakes in rural Ohio, Mishra et al. (2012) find a negative relationship between sulfate levels in impaired lakes and recreational use. The literature (reviewed below) documenting the transition of amenity-rich communities from reliance on extractive industries to tourism-based growth suggests a link between the two: were sulfide-ore copper mining to proceed at the TMM site, a contraction in tourism and recreation-based economic activity could plausibly occur, depending on the extent of mining disamenities that diminish the wilderness experience as well as on the severity of spills, breaches, and/or drainage.

While there have been reports issued on both sides of the issue,[9] those reports tend to look at snapshots in time, use different assumptions, and do not provide an integrated comparison of the economic costs and benefits of the proposed withdrawal.

Our study aims to fill this gap by providing an accounting of the impacts over time of the potential development of copper-nickel mining adjacent to the BWCAW on regional employment and income. We focus on the proposed TMM project because it is the sole copper-nickel mine currently proposed for the Rainy River watershed. We consider a 20-year horizon, which is the horizon of the Obama Administration's proposed mineral rights withdrawal. Because the focus is on the TMM project, the economic analysis focuses on the greater Ely region including usage of the Boundary Waters Canoe Area Wilderness (BWCAW) and nearby non-BWCAW lakes and forests. The study area is shown in Fig. 1. Our analysis draws on relevant regional and industry data, modeling in previous economic studies of the withdrawal, and the related economics literature. Our employment concept is employment in industries directly affected by the project (so-called direct employment), plus employment in the directly affected industry's supply chain (indirect employment), plus employment created by spending the earnings from direct and indirect employment (induced employment). Our income concept is total earnings from those direct, indirect, and induced jobs, taking into account differences in wages across sectors.

One of the challenges in this undertaking is the uncertainty around each of the many assumptions needed for this calculation. Although historical data inform distributions for some of our parameters, for others there is no evident way to calibrate a distribution, and moreover some of the parameters could covary and no data are available to quantify those covariances. As a result, a textbook treatment of uncertainty, for example Bayesian or Monte Carlo methods, is not practical in this situation. We therefore use a multiple scenario approach, which (as we explain) results in 72 different scenarios which in turn generate 72 different time paths for income and employment over the 20 years.

We find that, in all our scenarios, mining would produce an initial but temporary net growth of employment and income. Over time, however, the economic benefits of mining tend to be outweighed by the negative impact of mining on the recreational industry and on in-migration, leading to a boom-bust cycle. The preponderance of our scenarios indicates negative net present values of income resulting from the mining project. The primary drivers of the longer-run decline in incomes are increasing productivity in mining (estimated using historical data), reduced amenity-based in-migration, and reduced recreational demand. This boom-bust finding is consistent with recent papers on boom-bust cycles in extractive resource development.

The scope of this study – incomes and employment – is intentionally narrow, and we have omitted multiple factors which are likely important. These omitted factors include: effects on real estate values in the region; proprietors' income and profits; the value of the BWCAW and Superior National Forest as a regional attractor of talent in the Duluth area and elsewhere; and the employment and income driven by the BWCAW and Superior National Forest elsewhere in the state. We also do not consider non-market benefits such as non-market ecosystem services and wilderness existence values.

Although our focus is on the proposed TMM project, our impression is that the challenges confronting our study arise more generally in other natural resource extraction cases. These challenges include competing advocacy studies based on input-output models (or no models) that focus on a specific, unspecified date in the future and which potentially make mutually inconsistent assumptions; a lack of dynamic analysis that incorporates (for example) productivity growth and economic trends; a relative paucity of local data; and considerable uncertainty about key parameters. We hope that the methods used here for reconciling studies and estimating a range of dynamic impacts might be useful in other applications.

We first develop our scenarios and present the net present value calculations. We then discuss factors omitted from this analysis and discuss our results in the context of the relevant academic literature.

2. Computing costs and benefits over a 20-year horizon

We compute annual employment and income for the 20 years of the proposed withdrawal under two cases: the base case of the status quo in which there is no mining, which corresponds to a withdrawal of mineral rights, and the alternative in which the TMM mine is developed. In addition, we compute the net present values of the differences in income between the two cases.

We consider direct, indirect, and induced employment and income effects of the TMM case, relative to the base case. Direct employment is in the industries under study (mining and recreation). Indirect employment is in industries that serve the industry or project under study, for example in the case of mining, the change in employment in industries that provide mining services such as equipment repair. Induced

[7] Boundary Waters Wilderness Protection and Pollution Prevention Act, H.R. 5598, 116th Cong. (2020), https://www.congress.gov/116/bills/hr5598/BILLS-116hr5598ih.pdf.

[8] The conflict over the proposed TMM project is but the latest phase in a history of tension between mining and environmental concerns in Northern Minnesota, which has generated scholarly as well as public interest. Baeten et al. (2016) document the waste footprint of iron ore and taconite mining in the Mesabi Iron Range. Sutherland (2015) catalogs the challenges of economic transition from mining to tourism faced on the Cuyuna Iron Range, just south of the Mesabi, where production peaked in the 1950s. Bergstrom (2019) examines media coverage of copper-nickel mining in northern Minnesota. Liesch and Keweenaw, (2016) and Thistle and Langston (2016).

[9] University of Minnesota-Duluth (2012), Hjerpe and Phillips (2013), Sungur et al. (2014), Barber et al. (2014), Minnesota DNR (2015), Phillips and Alkire (2017), Helmberger (2017), Hjerpe (2018), Ward (2018), and Orr et al. (2018).

employment is the employment resulting from the spending of direct and indirect income on local goods and services. The direct and indirect income effects of the TMM counterfactual in a given year are the net effect on incomes from direct and indirect employment in mining and recreation of the TMM project, relative to the withdrawal case, plus the net direct effect on income from those attracted to the region by amenity values. This latter term captures the income spent in the region by those who choose to live in the region because of its amenity effects, and whose decision to live in the region might be affected by the withdrawal/no withdrawal decision.[10]

Induced income and employment are "spillover" effects of direct and indirect earnings which operate through a Keynesian multiplier channel.[11] Because mining jobs are better-paying than recreation jobs, a job in mining will result in more induced employment and income than a job in recreation. Whether induced employment and income effects actually materialize depends on the availability of unemployed or underemployed resources locally. If there is economic slack, then the direct and indirect earnings can create new local jobs. If, however, the economy is already at full employment, then what is calculated as induced employment either substitutes for other employment as workers change jobs or creates local jobs by expanding the work force as out-of-region workers move into the area. Recent empirical evidence in Auerbach et al. (2019) suggests that on average over periods of recession and expansion, there are nonzero induced local income and employment multipliers. The induced multipliers we use, which are taken from the IMPLAN studies of the Arrowhead economy, fall in the range estimated by Auerbach et al. (2019).[12]

The construction of our scenarios entails developing benchmark assumptions for employment and income under the case of the withdrawal, then considering alternative assumptions under the TMM counterfactual. To capture uncertainty, we vary key parameters to generate a total of 72 scenarios.

For our employment calculations, we make the following assumptions. For the case of the withdrawal, absent extant third-party growth forecasts of recreational employment in the greater Ely area, we rely on two sources of growth in employment related to recreation. In the Arrowhead region (St. Louis, Lake, and Cook counties), employment in the tourism and hospitality industries from 2012 to 2016 grew by 1.4% per year (Minnesota Department of Employment and Economic Development, 2017). USDA (2016) provides projections of increased recreational usage by category for 2008–2030; for the category "Backcountry/challenge" the annualized growth rate of user-days is 1.2%. We use this latter, lower value as the baseline in the withdrawal scenario because it is more directly relevant to BWCAW usage rather than outdoor recreation generally. Although Arrowhead region employment in recreational industries is available, we are unaware of data on the recreational employment base potentially specifically affected by the TMM project. Full Arrowhead region recreational employment (tourism and hospitality) in 2016 was 13,616, however that includes activity not likely to be directly impacted by the mining, such as hotels and restaurants serving University of Minnesota-Duluth and Duluth hospitals. Using the IMPLAN model and a survey of actual user expenditures, Hjerpe (2018) estimates that BWCAW visits from in-season out-of-region overnight visitors alone supports 879 direct jobs. Canoe camping in the BWCAW is just one way that recreational users take advantage of the outdoors in the region, so jobs potentially affected include more than just those supported by BWCAW out-of-region users. We therefore approximate the narrow direct employment definition from Hjerpe (2018) as accounting for one-fourth of potentially affected jobs. The full Superior National Forest area extends well to the east of Ely (see Fig. 1). For this reason, the assumption of 3516 (= 879 × 4) affected direct jobs could be an underestimate. We therefore consider an alternative case in which the number of affected direct jobs in tourism and recreational is 50% greater, 5274, which is still less than two-fifths the number of recreational and tourism jobs in the tri-county area.

Under the TMM counterfactual, in our high-mining scenario, we assume that TMM direct employment starts at 650 jobs, a figure taken from TMM materials (Twin Metals Minnesota, 2019a; Barber et al., 2014). We consider this estimate to reflect the high end of direct mining employment. The UMD-Duluth (2012) study projected 427 direct employment jobs in non-ferrous mining. In addition, in May 2018 TMM announced that it would scale back the planned mining from 50,000 tons per day to 20,000 tons per day, the figure in its December 2019 proposed Mining Plan of Operations. A proportional employment reduction of the TMM 650 jobs at 50,000 tons/day yields 260 direct employment jobs. We therefore consider two additional mining scenarios, intermediate, at 427 direct jobs, and low, at 260 direct jobs.

As shown in Fig. 2, non-ferrous mining generally, and copper mining specifically in the US, has exhibited substantial gains in productivity. Using the data in Fig. 2, we consider three mining productivity growth scenarios.[13] In all, this generates nine paths for annual

[10] We compute indirect employment from direct employment using indirect/direct proportionality factors from the IMPLAN model results reported for non-ferrous mining by UMD-Duluth (2012, Table 25) (Arrowhead region plus Douglas County, Wisconsin) and for recreation/hospitality by Hjerpe (2018, Table 5) (northeast Minnesota). Direct and indirect labor incomes are computed from direct and indirect employment using wage rates for 2016 for the Arrowhead region as discussed below.

[11] Of the four studies related to the withdrawal that use the IMPLAN model (UMD (2012), Minnesota DNR (2015), Hjerpe (2018), and Orr et al. (2018)), only Hjerpe (2018) and Orr et al. (2018) report labor income. The labor income multipliers (induced/(direct + indirect)) computed from results in Hjerpe (2018) and Orr et al. (2018) are respectively 0.214 and 0.347. With constant marginal propensities to consume out of labor income, the induced income multiplier should be the same for income earned regardless of its source (e.g., mining or recreation). One difference between the two studies that could account for these different multipliers is that Orr et al. (2018) consider state-wide effects whereas Hjerpe (2018) restricts effects to the northeast Minnesota region. Because the focus of our analysis is regional, not state-wide, we use the multiplier 0.214. This induced income multiplier is in line with the (induced/(direct + indirect)) value added multiplier of 0.18 in University of Minnesota-Duluth (2012), which is for the Arrowhead region plus Douglas County, Wisconsin. We compute induced employment from induced labor income using Arrowhead tri-county average wages for 2016. If the larger, state-wide induced multiplier of 0.347 is used, the numerical results change but the qualitative results, both for incomes and employment, do not.

[12] Auerbach et al. (2019) use US Department of Defense spending at the local region level and find that, for each $1 of US DOD spending in a locality, GDP in that state goes up by $1.50, so that the GDP multiplier ((indirect + induced)/direct) is 0.50. Their data covers 1997–2016 so includes both the strong labor markets of the late 1990s and mid-2000s and the long period of slack during and recovering from the financial crisis recession. Their estimate of an induced GDP multiplier of 0.50 is consistent with IMPLAN output multipliers. Hjerpe's (2018, Table 5) IMPLAN output multiplier ((indirect + induced)/direct) is 0.59 regionally (not state-wide) for recreation income. UMD's (2012, Table 25) regional GDP IMPLAN multiplier ((indirect + induced)/direct) is 0.43 for non-ferrous mining. The PolyMet FEIS (2015, Table 5.2.10-2) output multiplier is ((indirect + induced)/direct) is 0.55. Orr et al.'s (2018, Table 1) state-wide GDP IMPLAN multiplier ((indirect + induced)/direct) is 0.48.

[13] Fig. 2 shows an overall positive trend in output per employee in the Arizona copper industry from 1970 to 2016, across all hard rock metal mining from 1987 to 2017, and in underground coal mining separately in the three major U.S. coal producing regions from 2001 to 2016. The declines in copper mining output per employee in the mid- to late-2000's are associated with temporary changes in global commodity prices, and the decline in Appalachian underground coal productivity reflects the contraction in the industry and depletion of the higher productivity mines. The average growth rate of output per employee in the Arizona copper industry, 1970–2016, is 2.1% per year. We incorporate uncertainty using low and high productivity growth scenarios of 1.4%, and 2.7%, which are the end points of a 95% confidence interval for productivity growth estimated from the Arizona data. We assume a constant

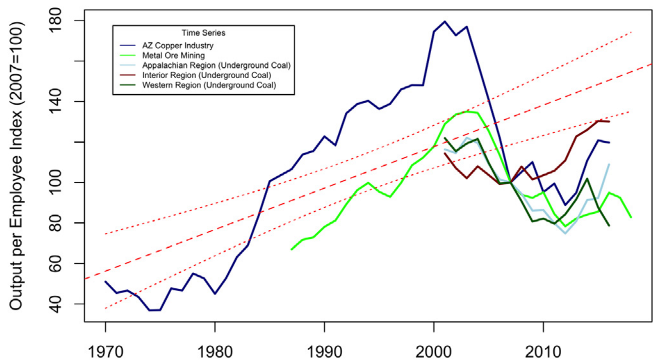

Fig. 2. Output per employee in copper, metal ore, and underground coal mining (index, 2007 = 100).
Sources: Energy Information Administration, 2016, Arizona Department of Mines and Mineral Resources, 1970, U.S. Geological Survey, 2015, Bureau of Economic Analysis, 1970-2017, Bureau of Labor Statistics, 1987-2016.

mining employment (three initial levels, three productivity growth rates).

Under the TMM counterfactual, we consider two paths for recreational employment, a low-impact path and a high-impact path. Because we are not aware of a directly comparable project (large-scale copper-sulfide-ore mining proximate to a water-based wilderness area) for which there are historical data, we consider a scenario in which recreational employment contracts at the rate of 1.2% per year and another in which it contracts at the rate of 2.4% per year. The first of these rates reverses the growth projected under the USDA baseline (USDA, 2016). The second of these rates is a reversal of twice the growth projected under the USDA baseline (USDA, 2016).[14] These counterfactuals are in line with previous studies of growth of other US amenity-based regional economies.[15]

We consider the high-impact scenario conservative in the sense that the impact on tourism over the long run of a major spill or acid mine drainage event are plausibly substantially more consequential.

For the income scenarios, the incomes associated with direct mining and recreational employment are computed using average local wage rates in those industries (Bureau of Labor Statistics, 2018; Minnesota Department of Employment and Economic Development, 2017). Employment in indirect and induced jobs are assumed to earn the average wage for the tri-county region in 2016 (Bureau of Labor Statistics, 2018; Minnesota Department of Employment and Economic Development, 2017).

The remaining component of income is the direct effect from those who move away from the region because of the mining and the related direct effect of those deterred from moving to, or retiring in, the region because of the mining (the "in-migration direct income"). To estimate this component, we used as a baseline the 2016 Census Bureau American Community Survey (U.S. Census Bureau, 2016) total income of the five-township Ely region (Ely, Eagles Nest, Fall Lake, Morse, and Stony River). We projected withdrawal baseline income growth as the sum of per-capita income growth and population growth. Our per-capita income growth projection is the historical per-capita income growth from 1970 to 2016 for the Arrowhead counties (Headwaters Economics, Economic Profile System, 2018). There is a large literature that documents increased population growth in amenities-rich areas (see Rickman and Rickman (2011) and Holmes et al. (2016) for surveys). We adopt the population growth rate from Rickman and Rickman (2011) for counties with USDA amenity rank equal to the average Arrowhead amenities rank (McGranahan, 1999). For the TMM counterfactual, we considered two scenarios for in-migration direct income. Polling by Sungur et al. (2014) found that 23% of residents would consider moving from the region in the event that the TMM project were undertaken. This estimate strikes us as high and many of those who would consider moving might not actually move. We therefore consider two scenarios one in which population growth slows to zero and a second in which in-migration population for amenity values declines by 10% over the 20-year period, less than half of the estimate in (Sungur et al., 2014).[16]

3. Results

In all, these assumptions generated 72 employment and income paths under the various scenarios. The employment paths are plotted in Fig. 3, and the income paths are plotted in Fig. 4.

All the scenarios in Figs. 3 and 4 show a similar pattern. Initially, mining is economically beneficial because of the new mining jobs, the income they produce, and their spillovers to the local economy. Over time, however, the net effect of the mining jobs erodes because of the growth of productivity in mining, the stagnation or decline of amenity-based in-migration, and the decline in wilderness-based recreation as a result of impacts of mining on the recreation industry. The magnitude and timing of the effect on employment and incomes varies across scenarios.

We computed the net present value for each of the income paths, using a 3% real discount rate (Office of Management and Budget, 2003). A histogram of these net present values is presented in Fig. 5. In 89% of the cases, the net present value of the TMM counterfactual is negative, that is, the income benefits of mining are outweighed by the income costs on recreation and in-migration. The cases for which the net present value of the TMM project are positive are those in which mining employment starts at the highest level (650 jobs, despite the

(*footnote continued*)
annual extraction rate, so that employment falls by the rate of growth of productivity for the three productivity scenarios.

[14] Rasker and Hackman (1996) examine employment and income trends in northwestern Montana and find that from 1969 to 1992, employment in counties characterized by pristine wilderness grew by 93%, an annualized rate of 2.9%. In contrast, resource-extractive counties observed employment growth of 15% over the same period, an annualized rate of 0.6%, a difference of roughly 2.3 percentage points. The scenario in which recreational employment contracts at the rate of 1.2% represents a difference of roughly 2.4 percentage points with respect to the withdrawal scenario. Thus, our rate of a 1.2% contraction in hospitality employment is reasonable assuming a reversal of Rasker and Hackman's (1996) estimate and is perhaps conservative given the degree to which hospitality and tourism employment is amenity-dependent.

[15] Rasker and Hansen (2000) examine rural counties in Idaho, Montana, and Wyoming and find that ecological and natural amenity variables are correlated with population growth in these areas. Deller et al. (2001) find similar results, finding a positive relationship between population growth and publicly owned land resources related to tourism. Winkler et al. (2007), find that "New West" communities, areas typically characterized by amenity migration, see anywhere from 38% to 195% higher employment in the tourism industry when compared to "Old West" communities. According to Winkler et al. (2007), this transition from "Old" to "New West" economic models has occurred over a 30-year period, which would imply an annual growth rate of between 1.2% and 6.5%. Empirical evidence supports the assertion that amenity-driven growth has supplanted extractive industries as the foundation of many amenity-rich, rural western counties (Lorah and Southwick, 2003). Rasker et al. (2013) find a positive relationship between growth in employment and proximity to protected public lands using data on federal lands in non-metropolitan Western counties. Henderson and McDaniel (2005) study sector-level employment growth and USDA natural amenity indices in more than 2000 rural U.S. counties, and find a statistically significant, positive relationship between landscape amenities and service sector employment growth.

[16] In-migrants are treated as bringing income to the economy but are not a business so do not undertake direct hiring, so there is no direct or indirect employment from this channel. That income is spent in part in the community, so it does generate induced employment and income, which are computed in the same way as induced employment and income from mining and recreation.

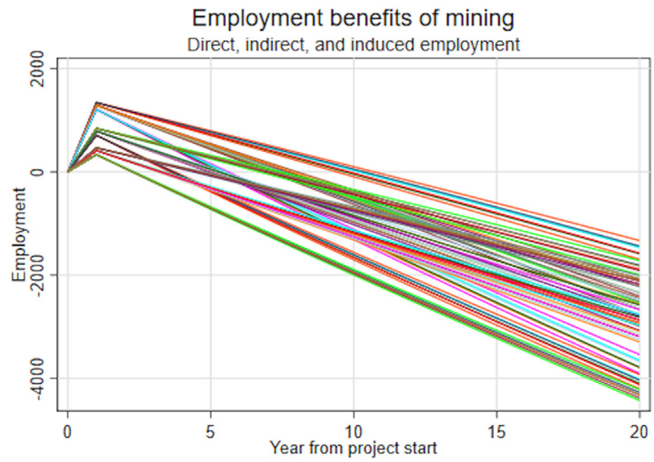

Fig. 3. Net annual employment effects (direct, indirect, and induced) of the TMM counterfactual over time on the Arrowhead economy. A positive employment value means that, under that scenario, the number of jobs in the TMM mining case exceeds the number of jobs in the no-mining baseline.
Notes: the horizontal axis denotes time, starting with the commencement of production at the TMM site.
Source: Authors' calculations.

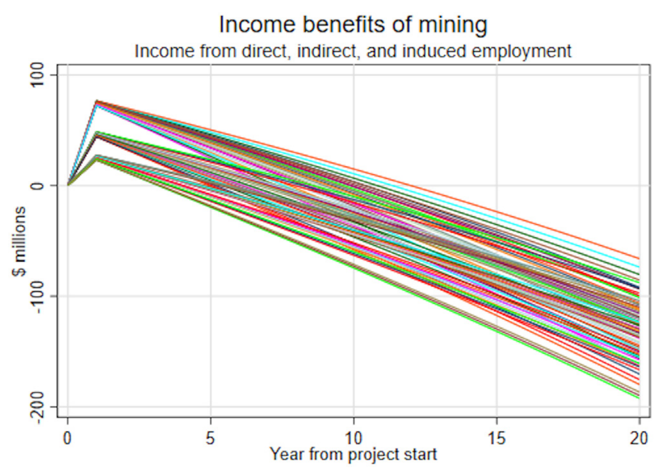

Fig. 4. Net annual income effects (direct, indirect, and induced) of withdrawal over time on the Arrowhead economy. A positive income value means that, under that scenario, the annual income in the TMM mining case exceeds the annual income in the no-mining baseline.
Notes: the horizontal axis denotes time, starting with the commencement of production at the TMM site.
Source: Authors' calculations.

2018 announcement and 2019 Mining Plan of Operations in which the project is scaled back) and impacts to tourism jobs and amenity-based in-migration are low.

4. Our estimates in the context of other studies

4.1. Other studies of rural economic growth and amenities

Multiple studies conclude that outdoor recreation and recreational amenities, especially wilderness amenities, have been the basis for strong and sustainable economic growth in rural communities with those attributes over the past two decades. This literature looks at a variety of measures including income, job growth, population growth in wilderness-abutting regions, willingness-to-pay, and property values. In early influential research, Deller et al. (2001) studied rural U.S. counties and concluded that "the empirical results provide strong evidence

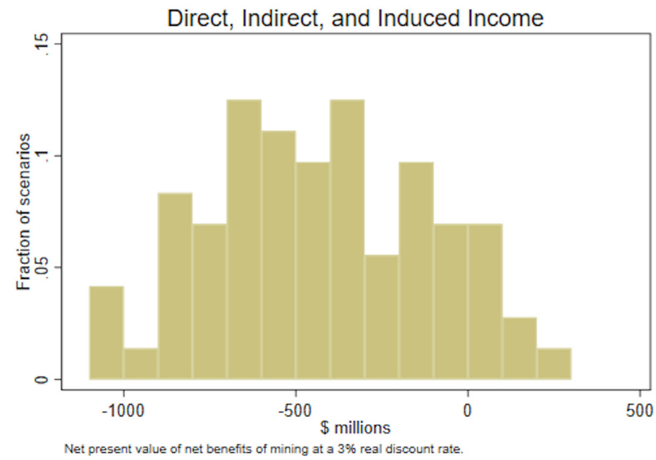

Fig. 5. Histogram of the net present value (NPV) of income in the 72 scenarios.
Source: Authors' calculations.

that rural areas which can be characterized as endowed with high levels of key natural resource amenity endowments and overall quality of life experience higher overall levels of growth" (p. 363). Rickman and Rickman (2011) examine population trends and measures of outdoor and recreational amenity in nonmetropolitan counties across the U.S.; they establish a positive relationship between amenity values and population growth. Lorah and Southwick (2003) look at the role of protected federal lands, which hold an intrinsic natural amenity value, on rural population growth in western counties and find that counties with protected federal lands within 50 miles of their center grew approximately 12 times faster than nonmetropolitan western counties without protected federal lands within 50 miles of their center. Poudayal et al. (2008) analyze nationwide county-level data on the role of natural resource amenities in attracting retiree in-migration; they find that the percentage of a county under forest, the quantity of high-quality water resources, and the presence of federally-protected National Parks are all statistically significant drivers of retiree in-migration. Winkler et al. (2007) finds similar demographic trends. McGranahan et al. (2011) study the underlying mechanism whereby sustainable growth is linked to amenity values and find that this growth has an endogenous element through the channel of entrepreneurs being attracted to rural locations with high outdoor amenity value.

Holmes et al. (2016) provide a recent survey of the literature on valuation of proximity to wilderness areas. In addition to reviewing estimates of the local economic effects (or "onsite" values) examined here, they include a discussion of "offsite" values on which we have not relied. These "offsite" values include both "use" values (e.g., residential property values; see below) and three so-called "passive use" values: existence value, option value, and bequest value. They argue that these passive use values can be large, a point that is relevant to the withdrawal proposal because they attempt to estimate directly the value of pristine wilderness.

These studies validate the inclusion of in-migration effects that are supported by the withdrawal and are potentially at risk if the withdrawal does not occur. In addition, these studies support a broader interpretation of the value of the BWCAW and Superior National Forest as an attractor of non-tourism, non-retirement jobs to the area because of the proximate wilderness. This latter category of job is not included in our study, and by excluding such jobs our study is conservative and understates the economic benefits of the withdrawal.

4.2. Resource extraction and sustainable growth

The question of resource extraction and economic growth has long been of interest in the economics literature at the country level (e.g., oil export economies), regional level, and local level. Although we are not aware of any recent hard-rock mining studies on sustainable local growth, the boom in nonconventional oil and gas development has stimulated recent research on extractive resource growth cycles.

Jacobsen and Parker, (2016) study county-level data for the American West and examine the consequences of oil and gas well drilling arising from the oil price increases of the 1970s and early 1980s. They find "that the boom created substantial short-term economic benefits, but also longer-term hardships that persisted in the form of joblessness and depressed local incomes.... In the longer run, after the full boom-and-bust cycle had concluded, we find that local per capita income was about 6% lower than it would have been if the boom had never occurred." (p. 1093).

Allcott and Keniston, (2018) study US county-level manufacturing data in connection with oil and gas booms and conclude that "while county-level population, employment, wages, and revenue productivity are all procyclical [i.e. all go up in the initial extractive stage], the booms are cancelled out by the busts. By the end of the 1990s, we see no significant remaining long-term effects of the boom and bust cycle of the 1970s and 1980s (p. 697)".

There is also some work on the economic impacts of nonconventional oil and gas extraction, however the scope for dynamic analysis is limited because that development is new and insufficient time has elapsed to observe a full cycle. One set of limited dynamic estimates is provided, however, by Feyrer et al. (2017). They use local geographic data to provide some estimates of the dynamic effect of nonconventional oil and gas extraction in the 2000s; they find that it has large employment effects, but that those employment effects are transitory at the local level. They only estimate dynamics over the first two years following the initial local extraction shock and find that wage income gains, including direct, indirect, and induced, dissipate by 1/3 within two years (the dissipation is faster if only direct and indirect wages are considered, see their Fig. 4). Because the technology for nonconventional oil and gas extraction has a shorter life cycle than hard rock mining or conventional oil and gas extraction, the findings of these studies are all qualitatively consistent with an extractive boom-bust cycle.

These studies are designed to estimate the effects of these booms on counties with average amenity values. Thus, these estimates capture the boom-bust effect on resource extraction and related jobs but do not include any special effects that resource extraction disamenities or environmental damage would have on employment and in-migration related to high-amenity regions like the area surrounding the BWCAW. Such effects would exacerbate the boom-bust nature because of the deterioration in environmental conditions and amenity values that would reduce non-mining amenity-related incomes.

4.3. Property values and mining disamenities

There is substantial evidence that mining disamenities reduce housing values. In their study of acid mine drainage (AMD) from coal mining in the Cheat River Watershed of West Virginia, Williamson et al. (2008) find that location near an AMD-impaired stream has an implicit marginal cost of $4783 on housing, or nearly 12.2% of a home's value. Kim and Harris (1996) examine the broader suite of possible mining disamenities and their effect on property values near a copper mine in Green Valley, AZ and find that parcels closest to the mining site lost 5.74% of their value with homes further away losing 0.66% of their value. In their study of sulfide-ore copper mining in the Arrowhead region, Phillips and Alkire (2017) use Kim and Harris' (1996) findings to estimate that the total loss in property value due to sulfide-ore copper mining would be approximately $508 million (2016 USD), or roughly 1.9% of the total property value of the three Arrowhead region counties. This is a large value which, if added to the NPVs in Fig. 5, would make all the NPVs negative.

Phillips and Alkire's (2017) estimate of a decline of 1.9% is in the range of those in related studies. Boxall et al. (2005) examine the impact of oil and gas facilities on rural residential property values in central Alberta, Canada using hedonic regression methods for property valuation.[17] They find that location within four km. of industry facilities leads to a four to 8% decrease in property value. Leggett and Bockstael (2000) use a hedonic property model to show that water quality has a significant effect on property values along the Chesapeake Bay, an amenity-rich, non-metropolitan setting with high recreational value. Poor et al. (2007) find a similar result in the Chesapeake Bay watershed examining non-point source pollutants, including suspended solids and nitrogen. In a study of the impact of lake water clarity on New Hampshire lakefront properties, Gibbs et al. (2002) find that water clarity—a measure of the degree of eutrophication—has a significant effect on prices paid for residential properties. More recent research linking local water quality to higher property values includes Keiser and Shapiro (2019) and Kuwayama et al. (2019).

In the case of the proposed withdrawal, these negative effects on housing values would be compounded by the downward pressure on housing values from reduced in-migration or, possibly, out-migration. Consistent with the boom-bust literature, one could see an initial rise in housing values as mine and associated industry workers buy or rent in the greater Ely area, however that increase would be temporary as mining employment, recreational employment, and in-migration housing demand subsequently decline. By omitting this effect, our analysis is conservative and likely understates the benefits of the proposed withdrawal.

5. Conclusion

We find that, over the 20-year time horizon of the proposed withdrawal, introducing copper-nickel mining in the Superior National Forest is likely to have a negative effect on the regional economy. Our calculations omit some factors, notably the negative effect of mining on real estate values, that would strengthen this conclusion. We reviewed the relevant literature and conclude that our findings are consistent with the literature, most notably the history of boom-bust economies associated with resource extraction that leave the local economy worse off.

In addition to adding to the debate over copper-nickel mining in the Superior National Forest, our study contributes to the broader literature on the tradeoffs between resource extraction and natural amenity-based economic growth. Our findings highlight the importance of considering the long-term effects of resource extraction in natural amenity rich areas. While estimates of the employment effects of the TMM project are positive in the short run, accounting for the well-documented boom-bust cycle that characterizes resource extraction results in negative estimates of the overall effect of allowing mining in the Superior National Forest. This analysis also demonstrates the importance of modeling dynamic responses to resource extraction in amenity-based income, for example through decreased in-migration and reduced demand for amenity-driven recreation.

Our study points to opportunities for future research. As noted previously, we omit several factors which are likely important to fully understand the impacts of allowing mining near the BWCAW, including both market values (such as housing) and non-market values (ecological services). Future work examining the effects of copper-nickel mining in this region should examine the long-run effects of mining on these additional values. More broadly, the prospective modeling approach of our study, which is shared by many other studies in this field, would ideally be complemented by empirical analysis of historical data. Additional work is needed on ex-post evaluation of the economic effects

[17] Hedonic regression is a method for estimating the value of a characteristic of a good when that characteristic is not sold separately but instead is part of a bundle of characteristics embodied in the good; see for example Haab and McConnell, 2002.

of resource extraction in comparable, ideally quasi-experimental, settings.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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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.

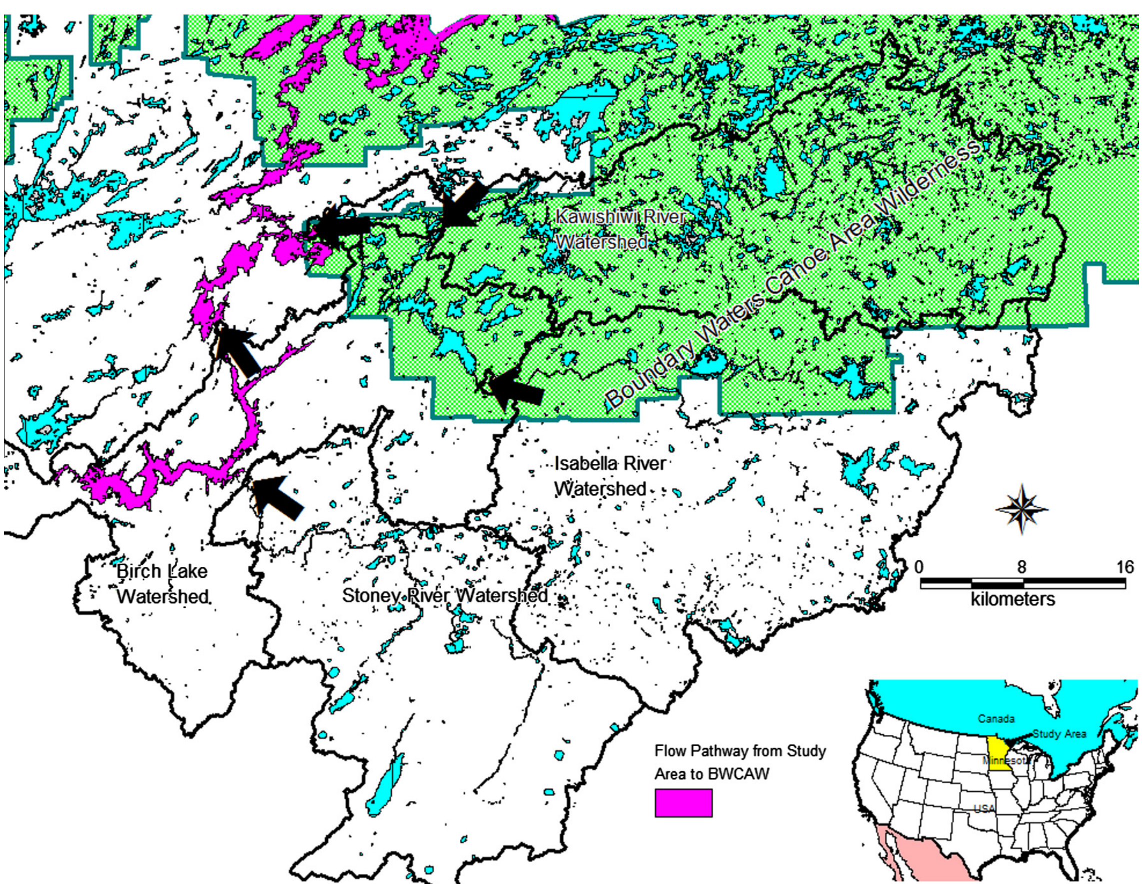

Fig. 1. Rainy Headwaters watershed and study area, showing subwatersheds, rivers, and lakes. Arrows are flow direction from watersheds. Watershed boundaries from Dnr100kwatersheds, www.mngeo.state.mn.us/chouse/metalong.html.

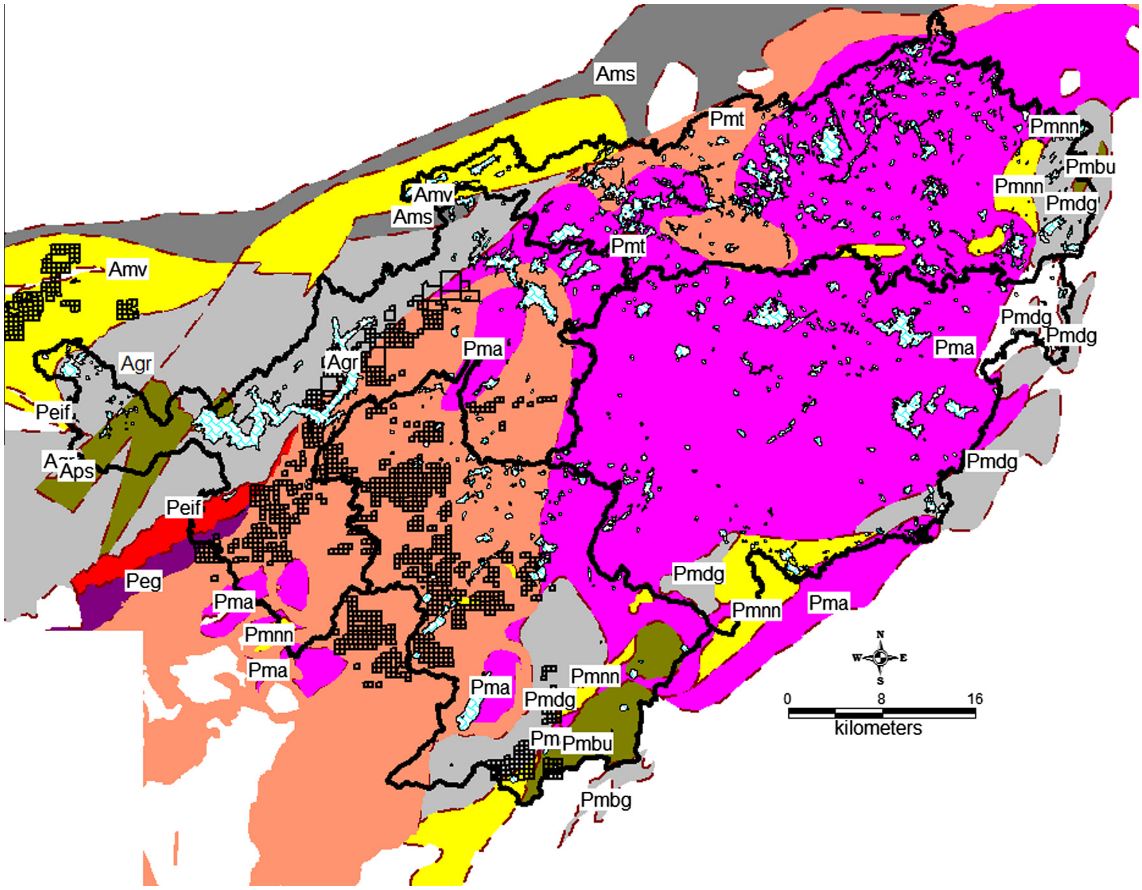

Fig. 2. Bedrock geology and watersheds of the study area. See Table 1 for a description of the formations. See Fig. 1 for the watershed names. The black rectangles are mining leases and approximate location of the mineral deposits.

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

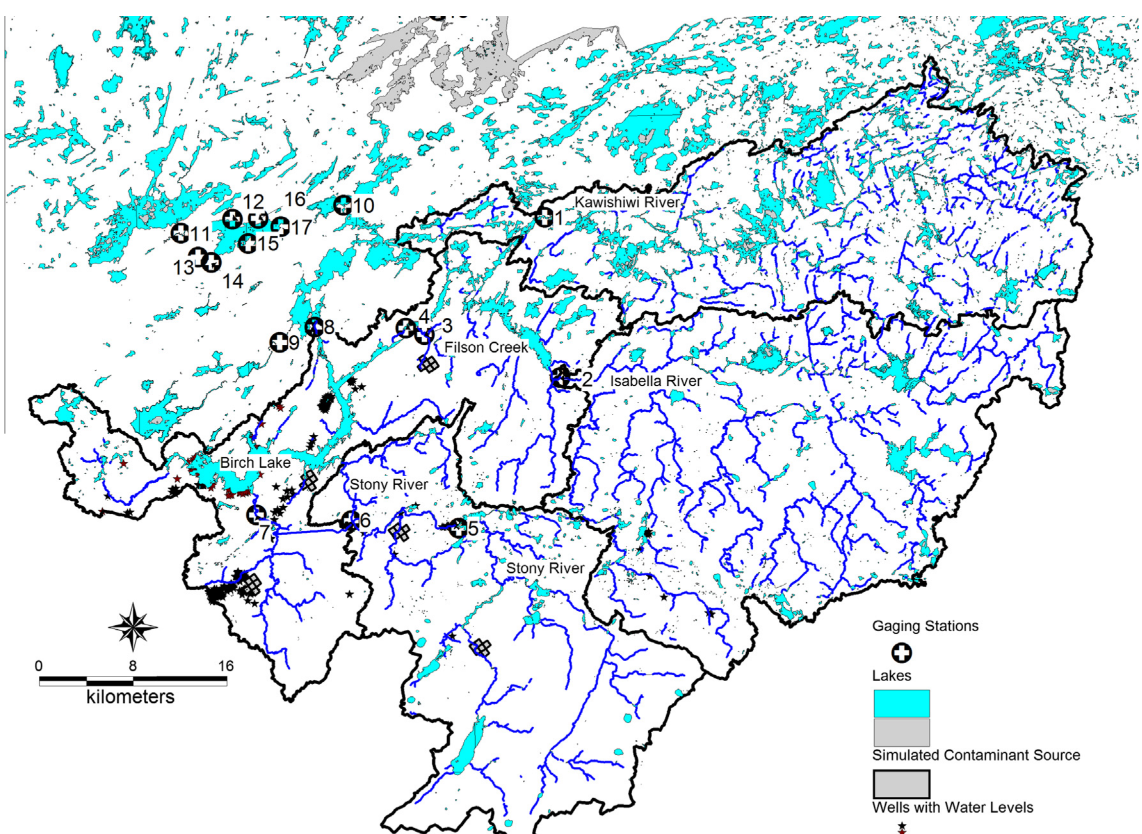

Fig. 3. Location of wells with water levels, gage stations, perennial streams, and lakes in the study area. See Table 2 for a list of gage stations. *Source:* http://mcc.mn.gov/gis.html#state_minerals_head.

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

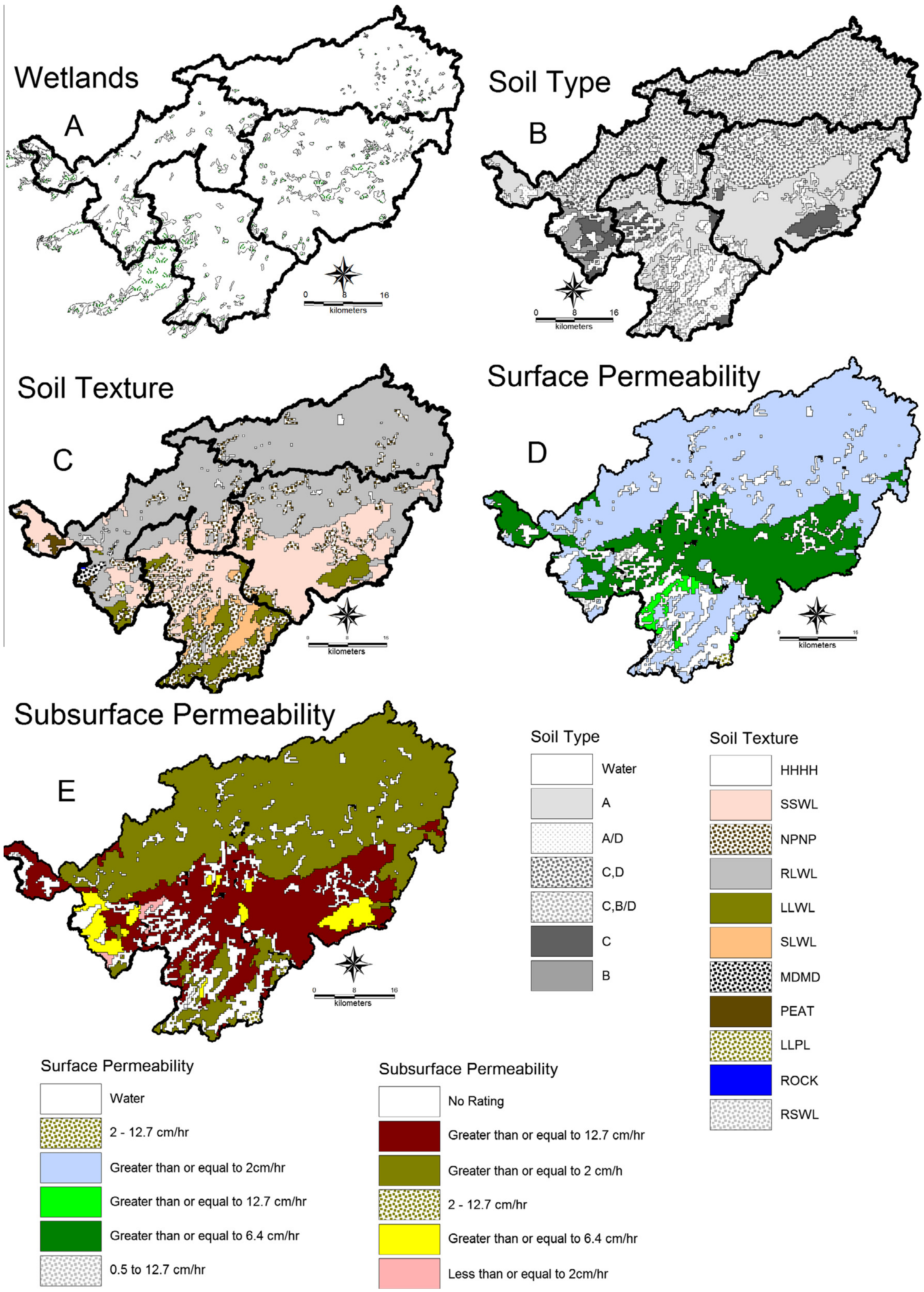

Fig. 4. Distribution of wetlands and soil types across the study area (Cummins and Grigal, 1980). Stream file Strm_baseln3, lakes and wetlands from Dnr100khydrography, from www.mngeo.state.mn.us/chouse/metalong.html. Soil type: 1234; Factor 1: Texture of soil below 5 feet, S is sandy, L is loamy, C is 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.

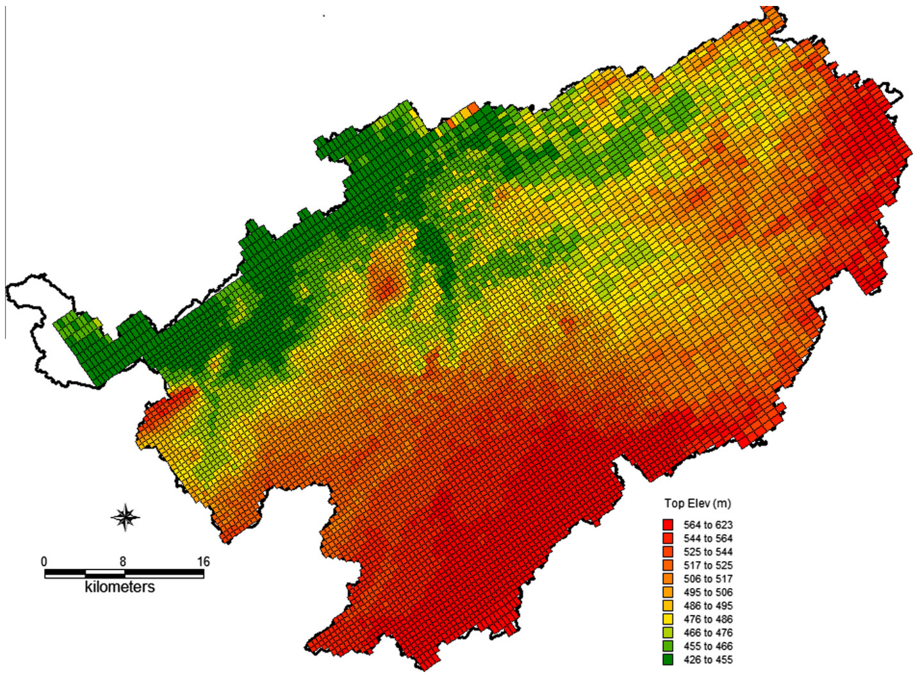

Fig. 5. Model grid and layer 1 top elevation by cell in meters.

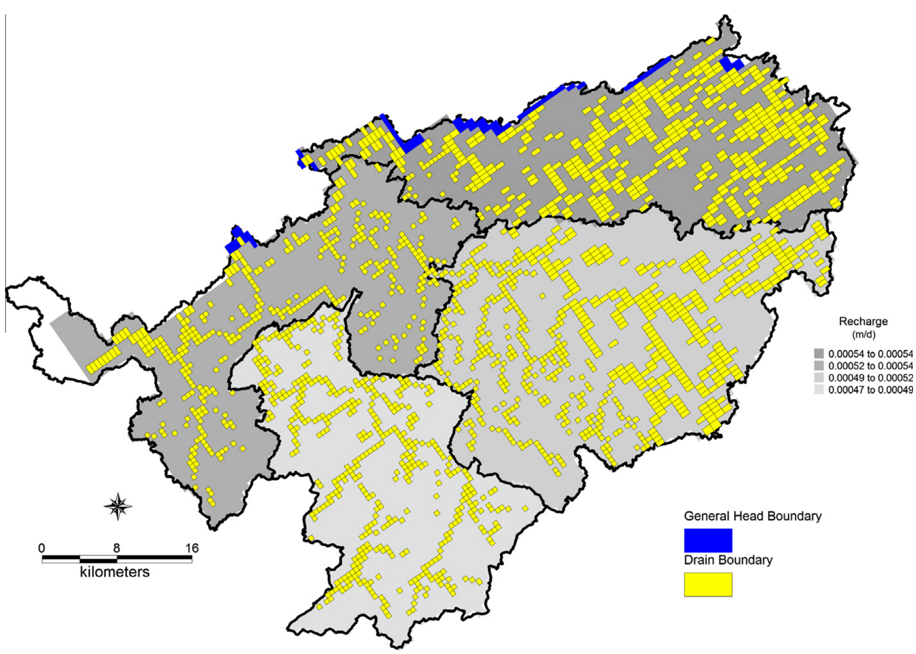

Fig. 6. Location of DRAIN and recharge boundaries in layer 1 and general head boundaries (GHBs) in layer 2. Final recharge rates equal 0.00054, 0.00049, 0.00047, and 0.00052 m/d for zones 2 through 5, respectively.

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.

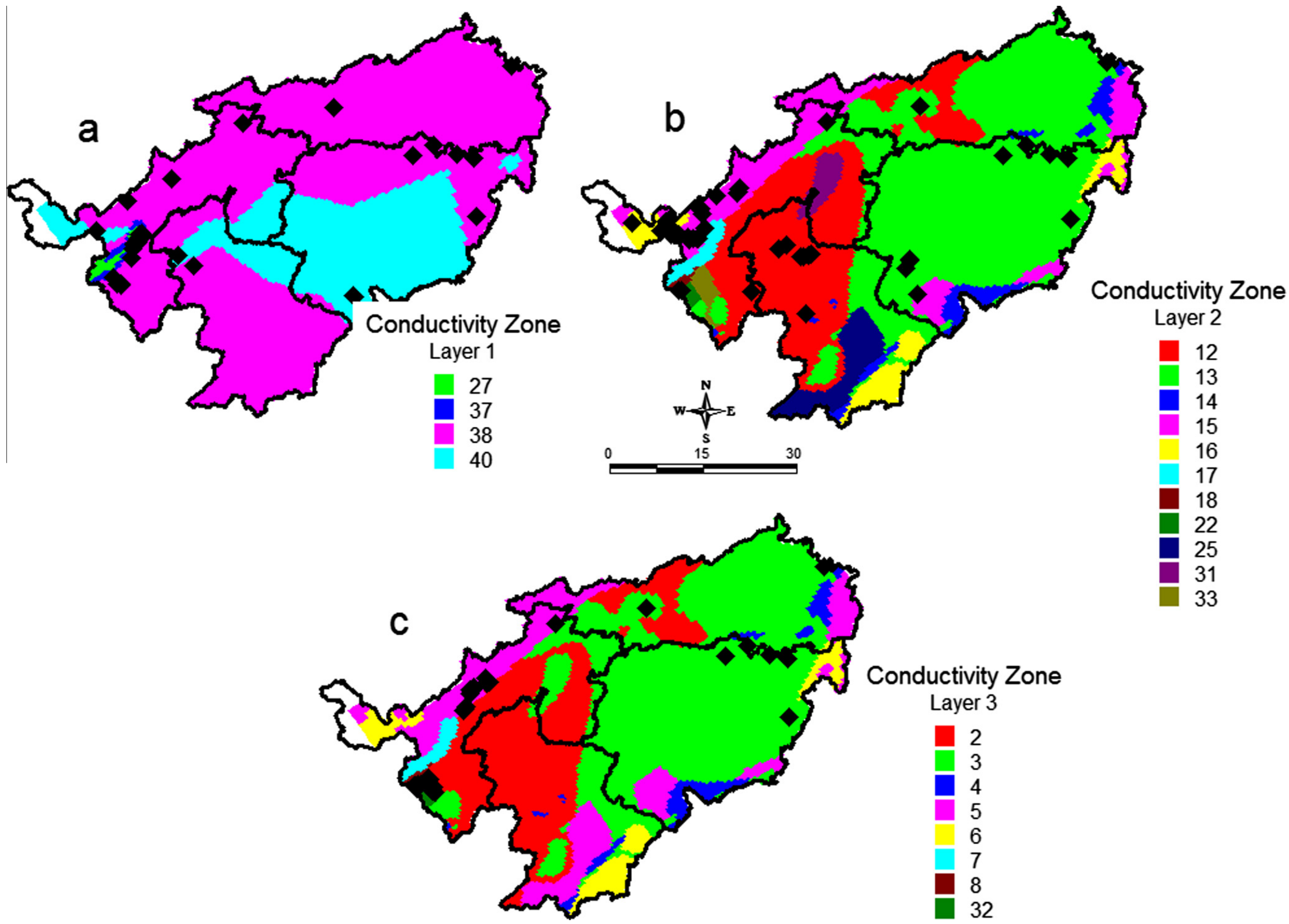

Fig. 7. Hydrogeology zones for three layers and steady state head target locations (black squares). Table 1 shows final conductivity values.

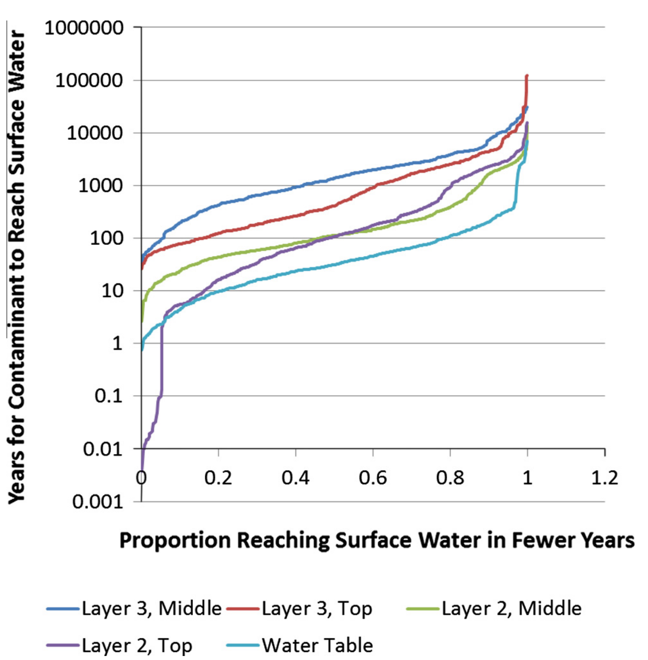

Fig. 8. Proportion of particles reaching surface water sources in a number of years for any location in the specified layer.

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.

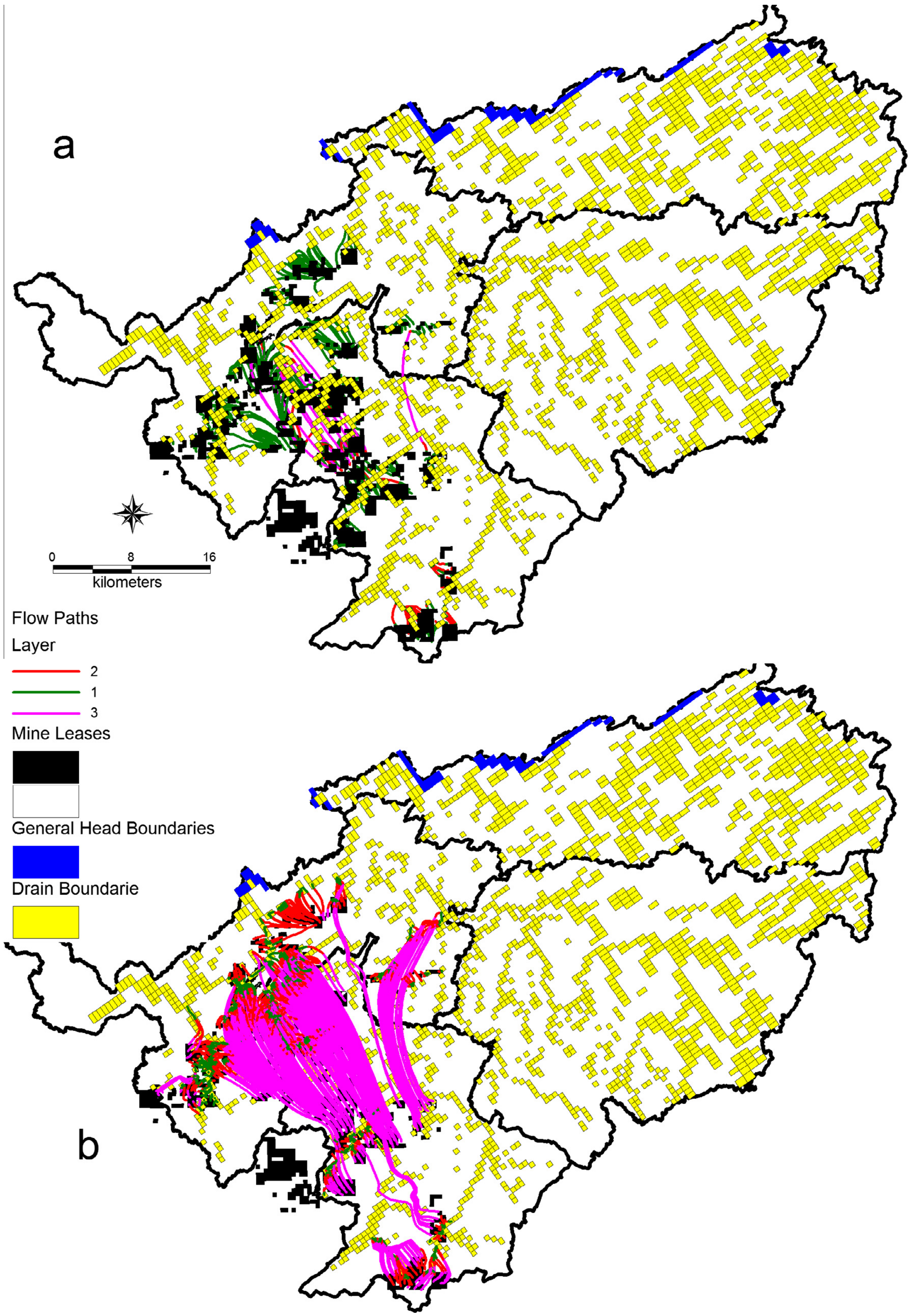

Fig. 9. Particle tracking for particles introduced at model cells near the mineral leases at the water table (a) or the top of layer 3 (b).

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

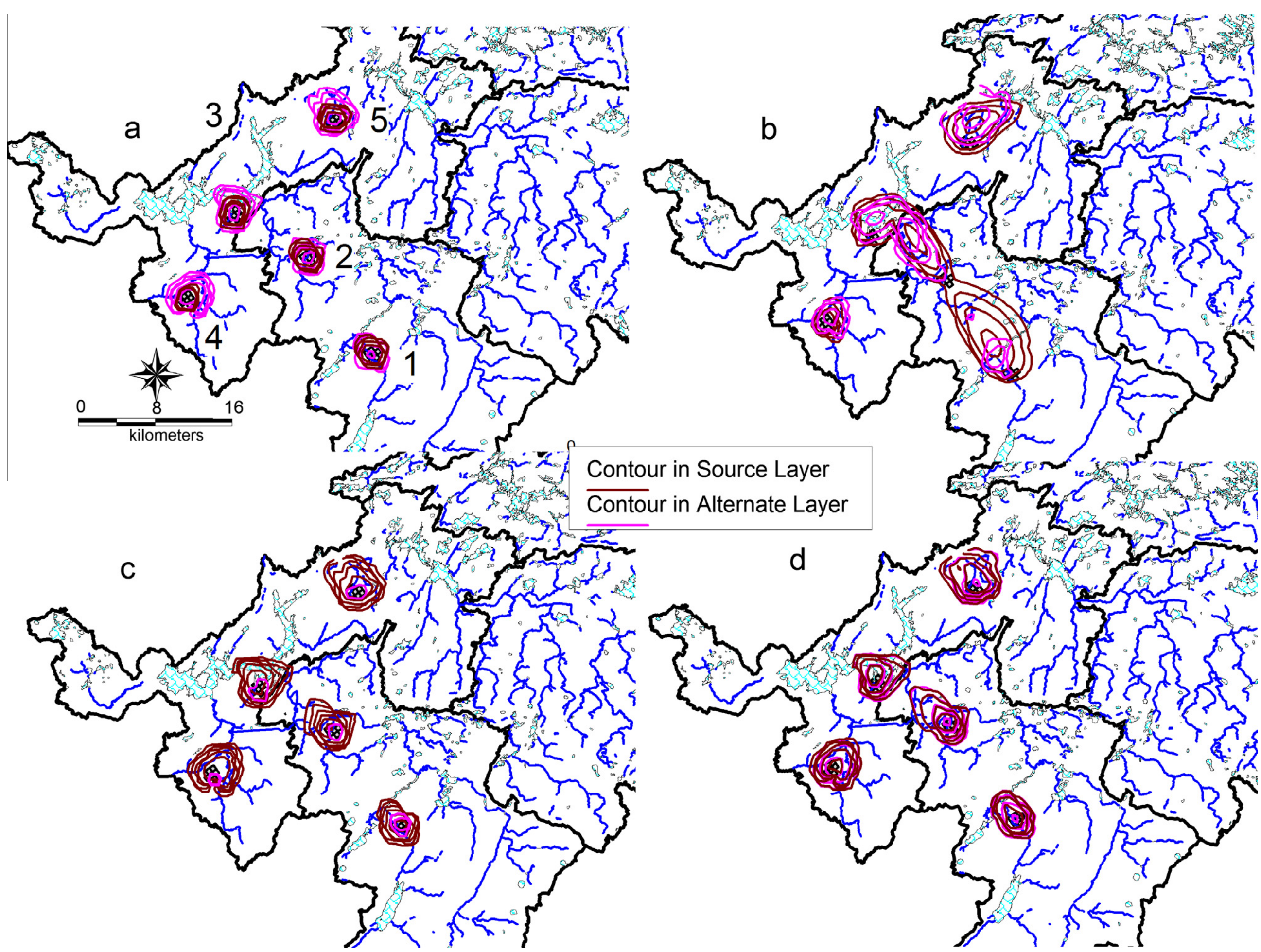

Fig. 10. Concentration contours for (a) underground source after 10 years, (b) underground source after 1000 years, (c) surface source after 10 years, and (d) surface source after 100 years. Underground source is layer 3, surface source is layer 1. Alternate layer is layer 1 in (a) and (b), layer 3 in (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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