APPLIED ECOLOGY

Risks of mining to salmonid-bearing watersheds

Christopher J. Sergeant[1,2]*, Erin K. Sexton[1], Jonathan W. Moore[3], Alana R. Westwood[4], Sonia A. Nagorski[5], Joseph L. Ebersole[6], David M. Chambers[7], Sarah L. O'Neal[8], Rachel L. Malison[1], F. Richard Hauer[1], Diane C. Whited[1], Jill Weitz[9], Jackie Caldwell[10], Marissa Capito[11], Mark Connor[10], Christopher A. Frissell[1,12], Greg Knox[13], Erin D. Lowery[14], Randal Macnair[15], Vicki Marlatt[16], Jenifer K. McIntyre[17], Megan V. McPhee[2], Nikki Skuce[18]

Mining provides resources for people but can pose risks to ecosystems that support cultural keystone species. Our synthesis reviews relevant aspects of mining operations, describes the ecology of salmonid-bearing watersheds in northwestern North America, and compiles the impacts of metal and coal extraction on salmonids and their habitat. We conservatively estimate that this region encompasses nearly 4000 past producing mines, with present-day operations ranging from small placer sites to massive open-pit projects that annually mine more than 118 million metric tons of earth. Despite impact assessments that are intended to evaluate risk and inform mitigation, mines continue to harm salmonid-bearing watersheds via pathways such as toxic contaminants, stream channel burial, and flow regime alteration. To better maintain watershed processes that benefit salmonids, we highlight key windows during the mining governance life cycle for science to guide policy by more accurately accounting for stressor complexity, cumulative effects, and future environmental change.

INTRODUCTION

Mining for metals and coal provides resources used by humanity but has the capacity to harm aquatic ecosystems. Mining can alter water and sediment chemistry, water cycling, physical habitat, and the health of organisms ranging from microbes to mammals, including humans (*1*–*5*). Mining impacts span vast scales of time and space. For example, in the Rio Tinto in Spain, pollution from primarily copper mining has persisted for over 5000 years (*6*). Pollution can extend tens to hundreds of kilometers downstream from mining operations (*1*, *7*, *8*). Globally, extracted mining wastes now cover ~1 million km^2 (*9*), and on the basis of publicly available data, mine waste reservoirs currently store 44.5 billion m^3 of tailings, enough to bury 59 km^2 Manhattan Island under 750 m (*10*).

From 2008 to 2017, the U.S. government spent 2.9 billion U.S. dollars (USD) addressing hazards posed by approximately 22,500 abandoned hardrock mine features, and many billions more USD are required to continue mitigation and cleanup (*11*). In the Canadian province of British Columbia (BC), the estimated reclamation liability for current major mine projects is 2.8 billion Canadian dollars (CAD) (*12*). At the same time, the social pressure to increase metal mining in North America is forecast to greatly increase, especially to support low-carbon technologies that reduce greenhouse gases (*13*). Considering that mining activities can have impacts that are long-lasting, spatially extensive, and costly to mitigate, there is a clear need to effectively link the science and known complexity of mining impacts to risk assessment and decision-making, particularly in ecosystems that support species of cultural and economic importance.

Here, we review how metal and coal mining can affect Pacific salmonid fishes (specifically, the genera *Oncorhynchus* and *Salvelinus*) and the watersheds that support them in northwestern North America. We define this region as extending from the eastern edge of the Columbia River Basin, west to the Washington State coastline, and north through BC and Yukon Territory and the state of Alaska (Fig. 1). We focus on salmonid-bearing watersheds for several reasons. First, salmonids are ecologically, culturally, and economically important species, including for Indigenous communities and rights holders. Salmonids are often the focus of environmental concerns related to mining impacts (*14*). Second, northwestern North America holds substantial coal and metal ore reserves and encompasses thousands of historical, current, and proposed mines (Fig. 1) yet still has some of the most productive and least disturbed salmonid habitat remaining on Earth (*15*, *16*). Therefore, this region represents a convergence of valuable mining reserves underlying watersheds supporting cultural keystone species, some of which are legally protected by treaties and legislation such as the U.S. Endangered Species Act and Canada's Species at Risk Act (*17*). Third, salmonids migrate across a wide range of habitats during their life cycles and can be exposed to many different pathways of impacts. In other words, if mining policies and regulations can be designed to protect salmonids, then it is likely that they are also protective of many aspects of watershed health.

We integrate and synthesize knowledge from multiple disciplines of the natural sciences including hydrology, river ecology, aquatic toxicology, and salmonid biology as well as components of mining policy such as environmental impact assessment. Wherever possible, we cite peer-reviewed studies conducted within northwestern

[1]Flathead Lake Biological Station, University of Montana, Polson, MT 59860, USA. [2]College of Fisheries and Ocean Sciences, University of Alaska Fairbanks, Juneau, AK 99801, USA. [3]Earth2Ocean Research Group, Simon Fraser University, Burnaby, BC V5A 1S6, Canada. [4]School for Resource and Environmental Studies, Dalhousie University, Halifax, NS B3H 4R2, Canada. [5]Environmental Science Program, University of Alaska Southeast, Juneau, AK 99801, USA. [6]U.S. Environmental Protection Agency, Corvallis, OR 97333, USA. [7]Center for Science in Public Participation, Bozeman, MT 59715, USA. [8]School of Aquatic and Fishery Sciences, University of Washington, Seattle, WA 98105, USA. [9]Salmon Beyond Borders, Juneau, AK 99801, USA. [10]Lands, Resources, and Fisheries, Taku River Tlingit First Nation, Atlin, BC V0W 1A0, Canada. [11]Juneau, AK 99801, USA. [12]Department of Hydrology, Salish Kootenai College, Pablo, MT 59855, USA. [13]SkeenaWild Conservation Trust, Terrace, BC V8G 1M9, Canada. [14]Environment, Land, and Licensing Business Unit, Seattle City Light, Seattle, WA 98104, USA. [15]Wildsight, Kimberley, BC V1A 1Z6, Canada. [16]Department of Biological Sciences, Simon Fraser University, Burnaby, BC V5A 1S6, Canada. [17]School of the Environment, Puyallup Research and Extension Center, Washington State University, Puyallup, WA 98371, USA. [18]Northern Confluence Initiative, Smithers, BC V0J 2N0, Canada.
*Corresponding author. Email: christopher.sergeant@umontana.edu

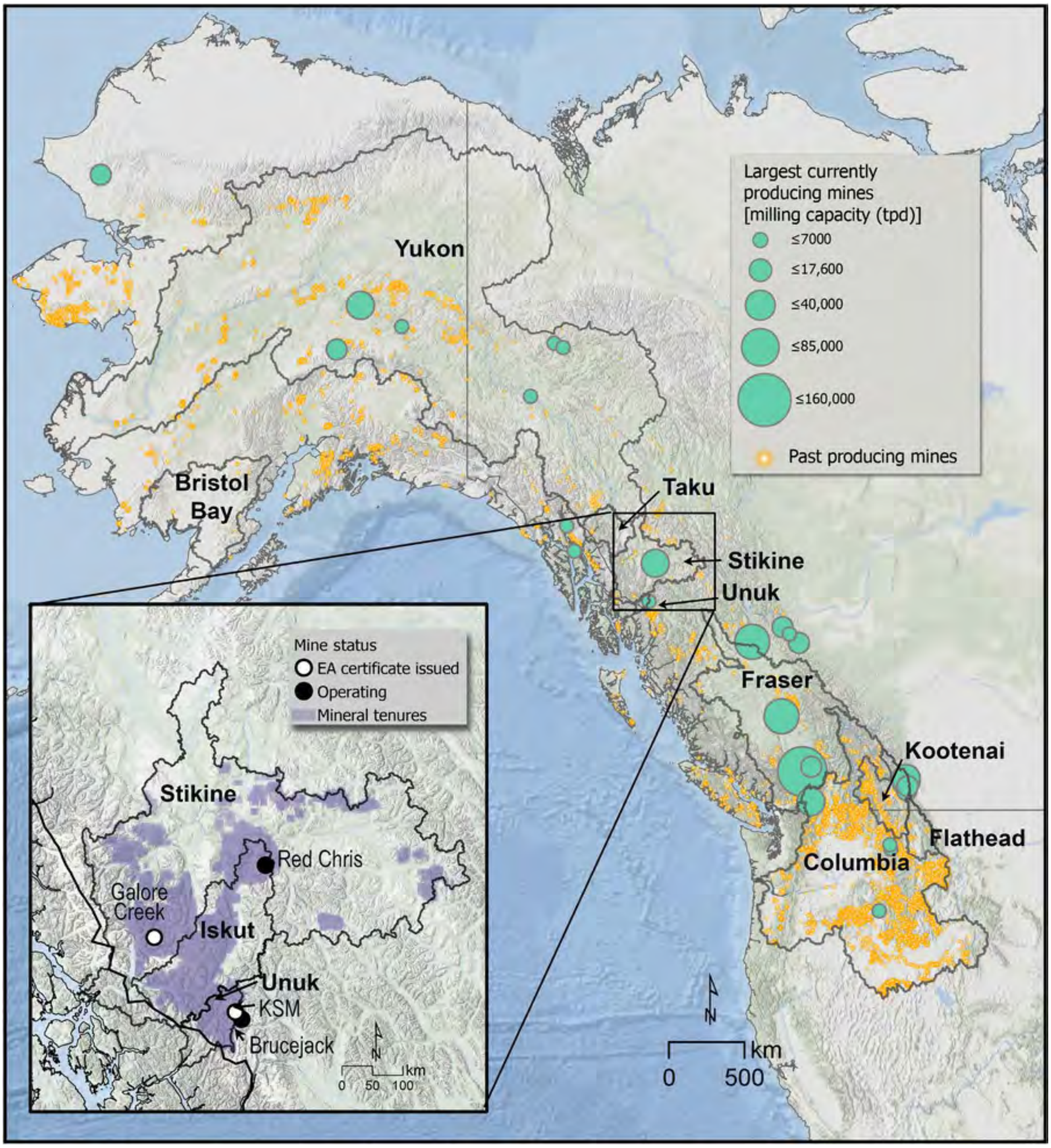

Fig. 1. Current and past producing metal and coal mining locations in northwestern North America. Outlined watersheds are referenced in the text. Teal circles represent the largest currently operating mines in the region ($n = 26$), where sizes are proportional to daily milling rate in metric tons per day (tpd). The inset illustrates the high density of mineral tenures (purple polygons) in the BC extent of the Stikine, Iskut, and Unuk Rivers. Data sources and definitions of "producer" and "past producer" are found in Supplementary Text.

North America. When necessary, we cite general textbooks and peer-reviewed studies outside of the focal region but with transferable and relevant knowledge. Information related to mining operations, current and historical production, case studies of impacts, and regulation and policy are often found in gray literature. Therefore, to provide a more robust assessment of the mining landscape of northwestern North America, we combine information from sources such as agency reports (e.g., British Columbia Chief Inspector of Mines Annual Report), federal/state/provincial-hosted databases [e.g., U.S. Geological Survey (USGS) Mineral Resources Data System], formal disclosure documents (e.g., legal filings with the Canadian Securities Administrators at sedar.com), and technical documentation posted on company websites (e.g., mining project overview descriptions). Our objectives are to (i) describe the extent of mining in northwestern North America, (ii) provide an overview of large-scale mining techniques and how they interact with salmonid-bearing watersheds, (iii) summarize pathways of impacts to salmonid-bearing watersheds, and (iv) highlight key windows during the mining governance life cycle where science can be used to better guide mining policy.

SALMONID-BEARING WATERSHEDS

As context for considering the risks of mining in northwestern North America to salmonid-bearing watersheds, we provide an overview of key attributes of these systems and salmonid life histories. Salmonids are a unique group among freshwater taxa in our study region due to their large home ranges and inclination to permeate all accessible reaches of a watershed's stream network during all seasons. Northwestern North America includes some of the longest remaining stretches of predominantly free-flowing rivers on the continent, such as in the Yukon and Fraser Rivers (*18*), ecologically important unconstrained river valleys that originate from glaciated mountains (*19*), and large intact forests, such as the boreal and coastal rainforests of BC and Alaska. As salmonids from the same river system move throughout a watershed, their exposure and sensitivity to potential mining impacts vary in a complex manner across time and space. Pacific salmonid species have adapted to thrive in dynamic and varied aquatic habitats that drain into the Pacific Ocean (*15*, *20*, *21*). Geological processes such as glacier advance and retreat (*22*), bedrock weathering, mass wasting of slopes, soil evolution, and fluvial geomorphic forces continue to shape these systems (*23*). In some cases, salmonids rapidly colonize new habitat formed by processes such as retreating glaciers (*22*, *24*). These cross-scale processes drive slow and rapid shifts in the locations, types, and amounts of freshwater habitats (*25*, *26*). Seasonal patterns of river flows and water temperatures along with the shifting physical distribution of habitats collectively define the amount, location, and suitability of productive salmonid habitat, which tend to shift within and across watersheds from year to year (*27*, *28*). These watershed dynamics not only drive system complexity and resilience (*15*, *29*) but also pose challenges for human infrastructure and attempts to assess and mitigate risks of development, including mining activities.

Within the family Salmonidae, we focus on native salmonids in the genera *Oncorhynchus* and *Salvelinus*, which include freshwater-resident trout such as cutthroat trout (*Oncorhynchus clarkii*), char such as bull trout (*Salvelinus confluentus*) and Dolly Varden (*Salvelinus malma*), and anadromous Pacific salmon such as Chinook, coho, sockeye, and pink (*Oncorhynchus* spp.) that perform extensive migrations between marine and freshwater habitats. When salmonids migrate, spawn, and die in high numbers in freshwater habitats, they import marine-derived nutrients (*30*, *31*) and provide a critical source of nutrients and energy to local consumers, ranging from grizzly bears (*32*) to resident fishes and aquatic invertebrates (*33*–*35*).

The population status of salmonids varies across northwestern North America. Watersheds in BC and Alaska still contain many diverse, resilient, and productive salmon stocks (*36*). However, especially toward the southern extent of their range, many populations of anadromous salmonids have been extirpated by human activities or are of conservation concern (*37*). Resident salmonids are also threatened in many regions; for example, the Flathead River watershed is one of the last remaining strongholds in the United States for nonhybridized native westslope cutthroat trout (*Oncorhynchus clarkii lewisi*) (*38*). Habitat degradation and loss, with the additional challenge of ongoing climate change (*39*), are threatening the productivity and resilience of salmonid-bearing watersheds and the benefits that they provide (*40*).

Different salmonid species and locally adapted populations have distinct life histories and habitat requirements [reviewed in (*41*)] that determine the duration and magnitude of their potential exposure to freshwater stressors. Spawning generally occurs once annually, when a single female may deposit hundreds to thousands of eggs in a gravel nest (redd) buffered by cool, flowing water. Depending on the species, individuals may spawn once during their lifetime (semelparous) or multiple times (iteroparous). After incubating as eggs in gravel for several months, larval fish emerge and rapidly grow into fry. Many species occurring in watersheds that connect to the ocean will migrate to the ocean after several weeks to several years in fresh water and eventually return to their natal freshwater stream or lake to spawn (anadromous). Some of these species will stay in fresh water their entire life, migrating between streams and lakes (adfluvial) or remaining in streams and/or large rivers (fluvial), resulting in one or more life stages overlapping in river habitats. This creates a high potential for exposure to acute stressors. Alternatively, for anadromous salmon species that immediately go to the ocean following emergence, such as pink salmon (*Oncorhynchus gorbuscha*), the time frame for exposure to acute stressors in fresh waters is seasonally narrow. Given that salmonids use different habitats across their life cycle, they can be exposed to cumulative stressors across multiple life stages and habitat types (*42*).

Salmonids are a cultural keystone species to many people in northwestern North America (*43*). Indigenous peoples have harvested migratory anadromous salmon for millennia, and this reliable source of food contributes to the cultural stability of their communities (*44*, *45*). Salmon fisheries are critically important to the food security and identity of coastal peoples (*46*–*48*). Salmon consumption represents an estimated 5.3% of protein and 45.5% of vitamin D intake by some contemporary First Nations peoples in BC (*46*). About one-third of Alaska-wide subsistence diets, as measured by weight, consists of salmon (*49*). Anadromous salmon also support globally important commercial fisheries. Millions of sockeye salmon are harvested in coastal commercial fisheries each year in Bristol Bay, Alaska, and these fisheries have sustained high harvests for over a century (*15*). The nearly 100,000 km^2 comprising the Tongass National Forest of southern Alaska supports an annual average of 48 million salmon for commercial fisheries, with a dockside value of 88 million USD (*50*). Similarly, recreational fisheries for salmonids support robust economies, with anglers and guide outfitters investing in gear, travel, and other costs in pursuit of a diversity of salmonids, from salmon in the ocean to anadromous steelhead to inland westslope cutthroat trout (*51*).

THE MINING LANDSCAPE OF NORTHWESTERN NORTH AMERICA

Below, we describe the density, types, and sizes of mining operations that overlap with salmonid-bearing watersheds in northwestern North America. We focus on metallic mineral and coal extraction because these mining activities represent some of the largest operations in terms of earth moved, ore processed, and economic impact (Figs. 1 and 2, Supplementary Text, fig. S1, and table S1) (*52*). Using data maintained by U.S. and Canadian governments, we conservatively estimate that, at a minimum, 3654 mines existed as past producers at least as far back as 1857 (Fig. 1; additional data source details are found in Supplementary Text). The USGS Mineral Resources Data System includes underground, surface, and placer mines. Data to determine mine size are often lacking from individual records, but our database query was targeted to minimize the number of small placer mines represented (see Supplementary Text). In

Fig. 2. Representative mining operations in northwestern North America. (**A**) Open-pit operations with a wet tailings impoundment facility beginning to take shape in the background (Red Chris Mine, BC; Garth Lenz). (**B**) Open-pit placer operations with a pit lake used for recirculating sluicing water (Atlin, BC; Jackie Caldwell). (**C**) Legacy underground operations adjacent to a glacial river (Tulsequah Chief Mine, BC; Christopher Sergeant). (**D**) Mountaintop removal coal mining (Elk Valley, BC; Garth Lenz).

contrast to hardrock mining, which removes nonfuel metals and minerals from solid ore beneath the ground, placer mining relies on water and gravity to concentrate valuable minerals such as gold that have been mobilized from their original deposits and now lie in surface sediments. The BC and Yukon MINFILE Mineral Inventories only include underground and open-pit operations. We found that data on active placer mining in BC and Yukon locations are not currently accessible in public databases. Additional mechanized placer mining operations in Alaska that are regulated by the U.S. Army Corps of Engineers are also not fully accounted for in our estimates. Considering these limitations, it is likely that the density of past producing mines in the southern portion of the Columbia Basin appears higher than in other portions of our study region (Fig. 1) because historical documentation was more broadly available in comparison to more northern areas. Currently active mining operations vary greatly in their styles of operation, capacities, and spatial footprints. The Highland Valley Copper Mine in south-central BC is the largest open-pit copper mine in Canada (and in our focal region) and, in 2017, mined nearly 119 million metric tons of earth and milled more than 52 million metric tons of ore (see Supplementary Text). In BC and the Yukon Territory, an emerging demand for minerals and precious metals has led to 41 major projects planned or under construction as of 2020, which collectively represent investments of 28 billion CAD (*53*).

Canadian mineral and coal "tenures"—which are land use agreements such as leases, licenses, or claims—provide individuals and companies the rights to explore and develop specific ore deposits over stipulated periods of time, but further permitting is needed to begin full-scale operations. Some watersheds contain such high densities of mining tenures that considerable portions of these watersheds are already staked for potential mining. For example, 59% of the Unuk River Basin is covered by mineral tenures, equaling approximately 88% of the BC portion of the watershed (Fig. 1). In the Iskut River, the largest tributary to the Stikine River, nearly the entire riparian corridor and 54% of the lower river's watershed are covered by tenures that overlap with rearing, migrating, and spawning habitat for salmonids (Fig. 1). Thus, many major salmonid-bearing watersheds have potentially high exposure to future impacts from mineral and coal mining operations.

Our review of publicly available data found little systematically collected information related to the processing rates and value of placer mining operations (Fig. 2B). These typically occur in valley bottoms and riparian areas and affect the hydrology, water quality, and channel morphology of fish-bearing rivers. While these operations are relatively small and tend to have low acid-generating potential (*54*), studies of specific watersheds suggest that the cumulative biological and physical impacts of placer mining may be substantial. Heavy metals such as arsenic and mercury can be released through the excavation process (*55*) and become toxic to salmonids (*56*). Extensive placer mining in the Fraser River greatly modified the physical habitat by altering natural sediment composition and transport rates (*57*). The State of Alaska has listed more than 193 km of streams impaired by placer monitoring activities that lead to excessive turbidity levels (*58*). Despite this evidence of the potential for environmental harm, BC and Yukon appear to not have any publicly available data on the numbers of placer mines. Thus, there appears to be less regulatory oversight of placer mining.

MINING OPERATIONS

In this section, we provide general descriptions of mining practices to provide context for the possible pathways of impacts on salmonid-bearing watersheds discussed later. In northwestern North America, most mining operations extract hardrock minerals (primarily metals) or coal by creating underground tunnel complexes or excavating large open pits at the earth's surface (Figs. 2 and 3). Mining

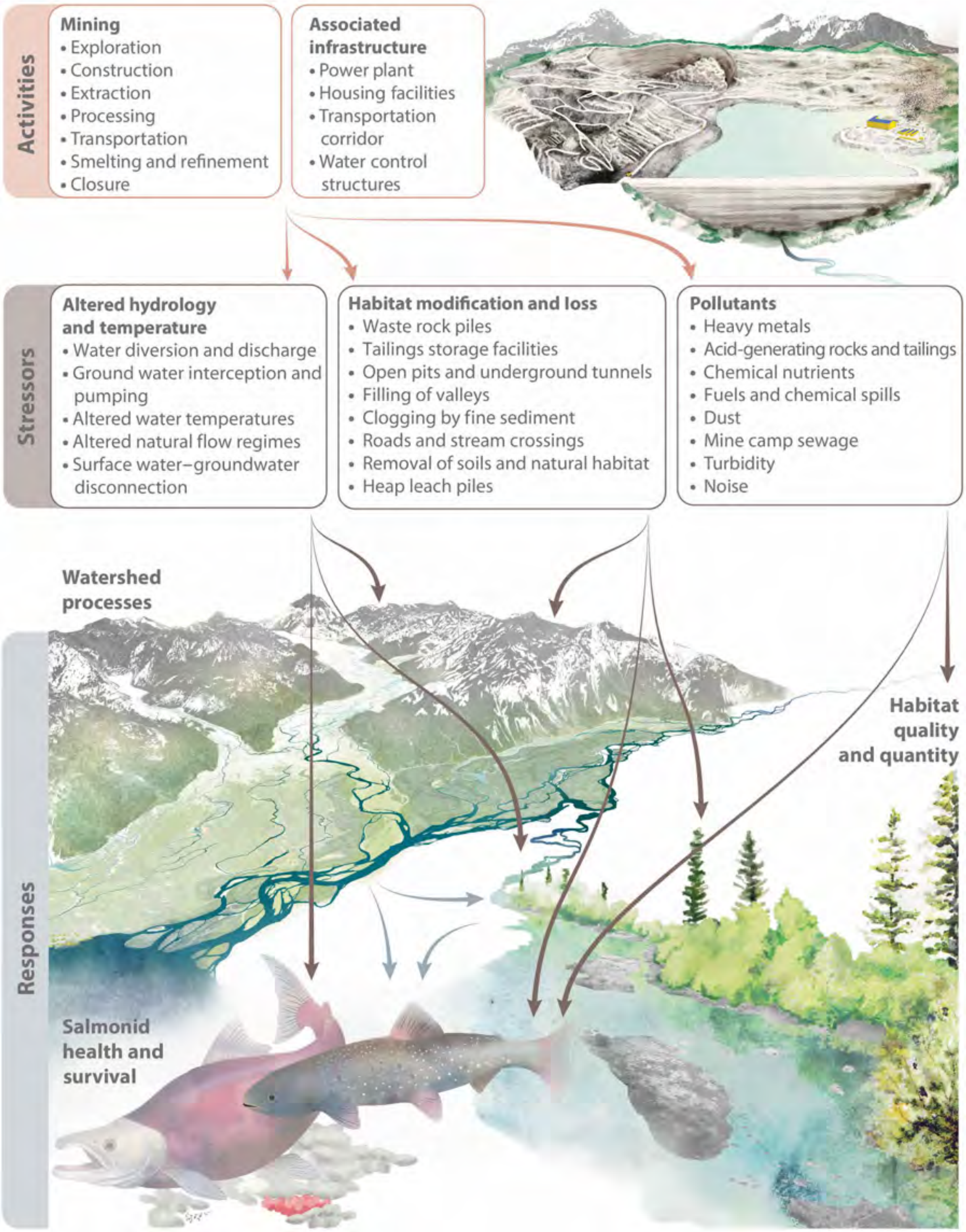

Fig. 3. Mining activities and pathways of impacts to salmonid-bearing watersheds. The different stages of mining activities and associated infrastructure can result in combinations of stressors that, in turn, influence the watershed processes that shape and define salmonid-bearing watersheds from headwaters to outlet, alter habitat quality and quantity, and directly influence salmonid health and survival (brown arrows). These pathways of impacts can have internal feedbacks and connections (gray arrows). Illustration by Cecil Howell.

typically produces ore, tailings, and waste rock. While coal mined for energy generation is sometimes washed before shipment, it does not always produce tailings. Mining generally consists of seven stages (with some differences between hardrock and coal operations): (i) **Exploration** locates and identifies potential mineral resources; (ii) **construction** involves a multiyear effort to prepare the land and build infrastructure before mining occurs; (iii) **extraction** (also known as production) removes the overburden and isolates rock containing metals or coal; (iv) **processing** pulverizes rock and uses metallurgical separation to isolate the target ore concentrate from waste material, which for metal mining is typically >99% of the total material mined (*59*); (v) **transportation** conveys intermediate and target products, fuel and chemical supplies, and waste material; (vi) **smelting and refining** heat or chemically process ore concentrate to remove the target metals; in northwestern North America, this stage is typically outsourced to China, which hosts the largest proportion of world smelter production and capacity (*52*); and (vii) **closure** occurs after a mine ceases to produce ore, and the site is either abandoned or reclaimed, maintained, and monitored for long-term water quality, dust, and visual impacts. It is outside the scope of this review to provide in-depth descriptions of each type of mining operation in northwestern North America. Therefore, we concentrate on commonalities across metal and coal mining and refer readers to more detailed operational descriptions in books such as the work of Whyte and Cumming (*60*). While not a focus of this review, we also note that phosphate mining is a large industry in parts of our study region such as southeastern Idaho. Similar to coal mining, phosphate mines use strip mining and open-pit techniques and can potentially elevate selenium concentrations to levels that create adverse effects to aquatic ecosystems (*61*, *62*).

Exploration involves a range of technologies and approaches. Standard geologic mapping augmented by geochemical analysis of soils is commonly used to determine mineral composition within a watershed (*63*). Remote sensing by aircraft or satellite can provide hyperspectral imagery of the gross geologic structure of potential deposits. Gravimetric, magnetic, seismic, electromagnetic, and electrical surveys are also used for mineral exploration. Test bore holes must be drilled to refine locations of valuable ore and support mine design and economic feasibility analysis. Habitat disturbance resulting from activities such as drilling (*64*) and frequent helicopter landings can range from minimal to long-lasting impacts on the land.

Throughout the life cycle of a mine, the associated infrastructure needed for operation includes plants for electrical generation and transmission, housing facilities, roads and potentially ports for transportation, and pipelines for conveying water and other substances (Fig. 3). There are two primary methods for accessing and extracting metallic ore or coal: surface and underground mining. Surface mining methods include placer mining, strip mining, mountaintop removal, and open-pit mining (Fig. 2). Strip and mountaintop removal methods involve dragging and/or blasting overburden to uncover subsurface coal seams or relatively shallow minerals. Strip mines sequentially backfill their excavations with part of the excess material, while mountaintop removal deposits waste rock in adjacent valleys. Open-pit designs use blasting and earthmoving equipment to excavate terraced depressions tens to hundreds of meters deep, usually requiring commensurate water table drawdown and groundwater management. Placer mining mechanically sorts target minerals out of alluvial deposits via gravity settling and, for gold mining, sometimes requires the addition of elemental mercury as a chemical amalgam. Underground mining also relies on blasting and earthmoving equipment but, in contrast to open-pit designs, creates a system of tunnels and underground rooms. Underground mining generally has a smaller aboveground footprint and produces less waste rock than open-pit mining, but it can lead to sinkholes and land subsidence.

Metallic ore bodies, rocks containing economically valuable concentrations of minerals such as gold and copper, typically host very low percentages of the targeted mineral (*9*). In a large mine operation, ore is transported to a processing plant, normally on-site, where it is crushed and ground to fine particles (clay to sand, 2 μm to 2 mm), sometimes physically separated or concentrated, and then chemically treated to concentrate target metals for refinement or smelting, which is typically conducted off-site. Low-grade ore—containing, for example, concentrations of metal less than 1 g of gold per metric ton of rock—may also be processed using chemical leaching on large piles of uncrushed ore. Ore is heaped onto large open-air pads with a synthetic liner and irrigated with a cyanide or acid solution that dissolves the metal. The resulting leachate is collected and processed for the target metals.

The concentrate produced by grinding and chemical treatment leaves a slurry of fine particles and chemical additives called tailings. Tailings are composed of a mix of liquid and solid particles that is piped away for storage in tailings impoundment facilities contained by embankment dams (*9*). Tailings dams are usually constructed with mine waste rock or, alternatively, with the coarser fraction of the tailings themselves. Less commonly, tailings are dewatered, filtered, and stored in an unsaturated form in engineered piles called dry stacks. Over the past four decades, only 3 to 6% of new tailings facilities use this dry-stack technology (*10*). All tailings impoundments, dams, and associated liners leak to some extent over time (*9*). Waste rock, uneconomic grade rock that occurs alongside the target ore, is broken up and stacked in large piles adjacent to aboveground or underground mine operations and generally lacks any sort of underlying liner. Such waste rock accumulations are often the largest sources of contaminants at mine sites (*65*, *66*). Where ore bodies are disturbed by mining and contain substantial concentrations of sulfide minerals, both tailings and waste rock can react with water, air, and bacteria to generate acidic and metal-laden effluent, known as acid mine drainage or acid rock drainage (*67*).

Transportation of fuel, consumable reagents, extracted minerals, ore concentrate, and other mining products to and from the mine site typically requires substantial investment in transportation corridors such as pipelines, roads, culverts, railroads, tramways, ferry terminals, and associated ports and storage facilities. Trucking, shipping, or piping of concentrated slurries may depend on seasonal conditions that allow transport over, through, and around mountain passes, along river and stream corridors, or around lakes and wetlands. Construction of transportation corridors requires dredge and fill activities in, around, and upslope of waterways.

Following closure, mine sites continue to be chemically and physically active over geologic time scales (*6*, *7*). Large mine sites in particular are so profoundly and irreversibly altered from their natural state that even after reclamation efforts (e.g., recontouring, revegetation, and infrastructure removal) are complete, active water treatment may be needed in perpetuity. In some cases, sites are abandoned without reclamation. This can be a consequence of insufficient bonding to finish reclamation, lack of regulatory enforcement, financial hardships experienced by the project owner, or extensive environmental damages. Some abandoned mine sites are also legacies of old mining laws before any financial assurances were required (*68*). Abandoned and partially remediated sites leave local communities or governments with an indefinite financial and ecological burden (*11*).

PATHWAYS OF MINING IMPACTS ON SALMONID-BEARING WATERSHEDS

Across the seven stages of mining described above (exploration, construction, extraction, processing, transportation, smelting and refining, and closure), mining activities and their associated infrastructure introduce stressors that present risks to watersheds and the salmonids that they support (*1*, *4*). These stressors can directly and indirectly affect all freshwater life stages of salmonids. We categorize impacts to salmonids using three interrelated categories of stressors (*69*): (i) altered hydrology and temperature, (ii) habitat modification and loss, and (iii) pollutants (*7*, *14*, *70*, *71*). These stressor categories modify important watershed processes, habitat quality and quantity, and the health and survival of individual fish and populations (Fig. 3).

Altered hydrology and water temperature

Mining alters the natural flow patterns of ground and surface waters by dewatering open pits, filling streams and wetlands with waste rock dumps and tailings impoundments, and intercepting or rerouting stream channels around mine infrastructure. While water treatment and storage facilities provide options for managing water quality and quantity in the short term, treating wastewater to match the natural flow regime "in perpetuity" creates an expensive post-mining legacy that can be challenging to maintain. In North America, these issues have been well studied in the coal mining regions of the eastern United States (*72*, *73*). In northwestern North America, little published information exists regarding the alteration of flow regimes by surface and underground mining, but there is evidence that (i) waste rock piles from coal mining in southern BC dampen flow regime response to precipitation events and increase dissolved ion loads (*74*) and (ii) open-pit mines with acid-generating rock have the potential to overflow after closure and threaten downstream salmonid habitat (*75*). This is a critical area for continued research, because in parallel with mining activities, climate change is shifting the seasonal and spatial patterns of precipitation, air temperature, streamflow, and water temperature. These changes are exacerbated by rapid glacier retreat, warming air temperature, less precipitation falling as snow, and more frequent extreme precipitation events such as those brought about by atmospheric rivers (*22*, *76*–*78*).

In addition to modifying streamflow patterns, water and tailings impoundment facilities modify natural thermal regimes of river valleys, either cooling or warming surface waters depending on the timing and method of releasing water (*79*). At northern latitudes, groundwater plays a critical role in salmonid growth and survival—especially for eggs incubating in gravels—by warming waters, providing refugia, slowing the onset of freezing during winter, and cooling waters during summer (*80*, *81*). Open pits, water and tailings impoundments, diversion channels, and roadways alter natural connections between surface water and groundwater (*82*), reducing the ability of streams to buffer extreme temperatures during periods of low discharge. Deviations from the streamflow and water temperature patterns to which local fish populations adapt can influence the timing of key life history events such as spawning and migration or alter growth and survival via direct (e.g., stream drying and exceedance of thermal tolerances) or indirect (e.g., alterations to food webs and reductions in available habitat) pathways (Fig. 3). Complex groundwater–surface water connections and the variety of pathways to organismal responses make translating impacts to fish populations challenging. Impact assessments and mitigation plans may rely on flow-habitat models [e.g., Physical Habitat Simulation System (PHABSIM)] (*83*) to translate risks to fish populations, but these require assumptions that are difficult to evaluate and can underestimate the water needs of fish (*84*).

Habitat modification and loss

The footprint of mines and their associated infrastructure can modify or eliminate salmonid physical habitats through the displacement, filling, rerouting, or permanent burial of stream channels and wetlands (*85*). We consider salmonid habitat to consist of physical attributes such as the arrangement of substrate and cover, as well as chemical and biological attributes that control salmonid growth and survival, such as the concentrations of trace metals in water and the availability of suitable invertebrate prey. Aquatic habitat can be altered directly from the construction of mine infrastructure or indirectly via modified streamflow and sediment regimes. Tailings and other fine sediments from mined areas can be transported into streams by erosion, potentially resulting in clogging of coarse bed material and even stream blockage, flooding, and/or channel entrenchment (*86*). Tailings impoundments are often one of the largest components of a mine's footprint and displace streams and land surfaces that would otherwise support aquatic and terrestrial life. For example, the Thompson Creek molybdenum mine in the Salmon River watershed of Idaho is currently inactive but maintains an approximately 240-m tailings storage dam that impounds a 130-ha reservoir (U.S. Army Corps of Engineers National Inventory of Dams; https://nid.usace.army.mil/). These reservoirs can fail with catastrophic consequences (*7*). On 4 August 2014, a failure of the 40-m tailings dam at the Mount Polley Mine released 7.3 million m^3 of metal-laden mine waste into Quesnel Lake, an important sockeye salmon (*Oncorhynchus nerka*) nursery lake in the upper Fraser River watershed of BC (*87*). Before reaching Quesnel Lake, the tailings slurry scoured, deforested, and buried 9.2 km of the Hazeltine Creek riparian zone and mainstem, which was a known salmonid spawning and rearing habitat (*88*, *89*). Although much of this discharge was deposited into lake sediments greater than 100 m in depth, mine waste resuspends in surface waters during spring and fall mixing of the water column (*89*), and the potential for long-term effects to the lake food web remains unknown. Researchers conservatively estimate that more than 130 tailings dam failures have occurred in the United States and Canada since 1910, accounting for 43% of all such failures globally during the past 100 years (*90*). Tailings dams, which must be maintained in perpetuity, are generally more prone to failure than water-retaining dams due to their unconsolidated earthen material construction that is typically built in stages over the course of many years as the impoundment facilities grow (*7*, *90*).

While tailings impoundments are conspicuous and receive attention due to their high potential impact, other mining structures such as waste rock piles, open pits, underground tunnels, and electrical transmission and transportation corridors also contribute to physical habitat modification and loss. Electrical transmission and transportation–related impacts can not only be direct, such as poorly constructed culverts creating barriers to movement, but also indirect by facilitating increased human access to remote areas, enabling the formation of mining districts or other industrial development. The BC Northwest Transmission Line was built at a cost of $746 million CAD and includes 2100 km of wires to increase the feasibility of mining projects and attract more exploration in remote portions of northern BC (*91*). Access roads built for new mine projects may hinder fish passage via stream crossings, bridges, and culverts (*92*). They may also promote the erosion of fine sediments into aquatic habitats, undercut slopes and increase landslide risk, restrict floodplain and channel migration, intercept groundwater, simplify habitat, mobilize methylmercury and other atmospherically deposited pollutants from disturbed soils, modify animal behavior, and contribute vehicle-related pollutants (*93*). Access by rail or road to and from ports, where concentrates are shipped elsewhere for smelting, poses additional threats when large vehicles filled with ore concentrate and/or mining-related chemicals are transported over sensitive landscapes and waterbodies. Construction and use of ports for ore concentrate loading may pose risks to coastal environments, including estuaries of salmonid-bearing watersheds. Mining community infrastructure may stress adjacent stream systems with issues related to sewage, garbage, loss of vegetation and shade, noise and air pollution, and invasive species introductions (*94*, *95*).

Pollutants

Mining for metals and coal alters the physical attributes and the geochemical stability of the disturbed geologic materials, often leading to pollution of downstream receiving waters. Chemical pollution can range from chronic, low-level metal leaching at the river-reach scale to catastrophic, sudden failures with watershed-scale impacts. Metal contamination in stream waters or sediments can be detected up to hundreds of kilometers from their source (*8*, *96*), and their presence can impose direct and indirect deleterious health effects on salmonid-bearing watersheds. In addition to metals, pollutants leaching from disturbed mine operation areas can include sulfate, nutrients, and nitrates from nitrogen-containing explosives (*97*–*99*). Leaching also occurs on road systems and power corridors from exposed soils, fossil fuel combustion, and spilled haul materials.

Pollution can continue long after mine closure, especially where acid-generating rock is present and tailings impoundment facilities exist. Long-term metal pollution results largely from oxidative chemical reactions acting upon sulfide minerals in the exposed metalliferous ore or coal seams, tailings, and waste rock (*4*). Acid mine reactions in sulfide-bearing metal ores and coal deposits are common, largely unavoidable, and can persist for millennia if they are not proactively managed (*67*, *100*). Increasing the surface area of the ore body by multiple orders of magnitude, as is done in the milling process where rock is broken and crushed, greatly accelerates and sustains acid rock drainage and other reactions that release trace elements (*101*). Acidic conditions dissolve trace metals, allowing them to be easily transported downstream, where shifts in redox conditions can cause them to precipitate and sorb to streambed sediments (*102*). Tailings may also contain processing chemicals such as petroleum by-products, acids, and cyanide (*4*). While modern smelting operations are typically outsourced to Asia, atmospheric circulation patterns return some pollutants to northwestern North America. Industrial emissions from eastern Asia contribute to global pollution associated with acid rain, heavy metal fallout, and carbon pollution (*103*). They can also travel back across the Pacific Ocean and contribute to increased atmospheric deposition of trace metals within sections of northwestern North America such as Alaska (*104*) and Oregon (*105*).

Direct impacts to salmonids resulting from elevated concentrations of metals from mining have included the interruption of upstream migration [Atlantic salmon (*Salmo salar*) in New Brunswick, Canada] (*106*) and the extirpation of local populations (Chinook salmon in Idaho, USA) (*107*). Olfaction and antipredatory behavior may be impaired by metal-rich water (*108*–*111*), and the ability of salmonids to use spawning gravels may be degraded because of iron hydroxides precipitating and coating the streambed (*112*). In heavily polluted waters, acute exposure of salmonids such as rainbow trout (*Oncorhynchus mykiss*) to elevated metal concentrations can result in death within hours to days (*113*, *114*). Sublethal concentrations of copper may reduce the migration success and seawater adaptability of anadromous salmonids such as coho salmon (*115*). In the Coeur d'Alene River basin in Idaho, elevated levels of arsenic, cadmium, lead, and zinc created by a high density of hardrock mining operations were correlated with less abundant native fish assemblages and decreased aquatic insect diversity and abundance, even 70 years or more after cessation of mining (*116*, *117*). These correlations may in part reflect that highly mobile salmonid species such as cutthroat trout may be able to avoid habitat with high metal loads relative to more sedentary fishes with small home ranges such as sculpin (*Cottus* spp.) (*116*).

Pollutants from mining-disturbed areas can propagate across food webs and affect salmonid food sources. Altered water chemistry downstream of mines can result in corresponding decreases in benthic invertebrate richness and abundance, changing community composition to favor pollutant-tolerant species (*97*, *99*). Selenium is a common element found in metal and coal geology that is essential for life in trace amounts but tends to bioaccumulate in the food chain (*118*). When chronically leached into downstream surface and groundwaters from mine sites, selenium can reach concentrations that are toxic to fish and all aquatic life, potentially resulting in deformities and ultimately reproductive failure (*99*, *119*). Fish are also directly affected because of ingestion of contaminated prey (*120*).

In summary, cumulative stressors resulting from mines can cause direct and indirect harm to salmonid-bearing watershed health via multiple pathways of impact. Evidence of direct impacts on salmonids exists and speaks to the importance of effective mining governance.

THE SCIENCE OF MINING POLICY

Mining in northwestern North America is governed by regulations, laws, and policies that vary by jurisdiction. In addition to analyzing potential environmental impacts, mining governance also considers other factors such as economics, human values, and community well-being. While science is only one of several dimensions of mining decision-making, it plays a foundational role in the accurate characterization of environmental impacts. In this section, we highlight key windows for science to guide mining policy. This is not intended to be a comprehensive review of mining policy, which is beyond the present scope.

The following regulatory processes and policies define the mining governance life cycle: (i) **Preproject** policies can include land-use designations or plans that govern whether a region is deemed appropriate for resource extraction; (ii) **impact assessment** informs project permitting, including the approval or rejection of the project, and associated mitigation strategies; (iii) **operations** consist of regulation, monitoring, enforcement, and mitigation of mining operations and their potential impacts; and (iv) **closure** of operations transitions the mine from being active to inactive and can govern abandonment, remediation, or reclamation. Depending on individual mining projects, these phases may not occur in order and may overlap in time. Even when these four general categories of mining policy occur at discrete stages of an individual mine's operations, there are strong cross-dependencies. For example, mitigating project impacts is a key activity during mining operations, but the efficacy of these mitigations is mainly considered during the impact assessment phase.

Preproject

Before the impact assessment of a specific mining project, forward-looking planning processes at the regional or watershed scale can establish a collaborative conservation and long-term development vision for the area. Such efforts avoid the pitfalls of single-project cumulative effect assessments (*121*, *122*) and identify specific areas where mining poses risks that cannot be mitigated, are not in the public interest, and should not proceed.

There are various policy tools that could be implemented to advance regional planning. For example, in the Taku River watershed, the Taku River Tlingit First Nation established the Wóoshtin Yan TOO.AAT Land Use Plan with BC, which defined 13 protected areas covering 560,000 ha and established resource management zones, cultural areas of significance, salmon ecosystem management areas, and critical aquatic habitat areas. The Nation and BC also have a Shared Engagement Agreement that outlines the way both parties will engage on land development projects. In addition, Canada's federal Impact Assessment Act (2019) allows for the use of regional assessments as a planning tool to guide the protection or development of regions under pressure. Both the Impact Assessment Act and BC's Environmental Assessment Act (2018) (*123*) were recently updated to include provisions for early engagement among proponents, regulators, other governments, Indigenous Peoples, and the public. Incorporating the values and priorities of local stakeholders and Indigenous rights holders may allow people who bear the immediate burden of the environmental impacts or benefits of mining to shape the vision of their place. However, other applicable legislation in these watersheds [e.g., the National Environmental Policy Act (NEPA)] (*124*) and the Yukon Environmental and Socio-economic Assessment Act (*125*) do not have these provisions.

Given the many cumulative risks associated with mining in a large region and across administrative boundaries, it is important to ensure that scientific predictions of impacts are undertaken at the appropriate scale. Ideally, major mining project proposals—especially those that cross international jurisdictions—would automatically trigger federal-, regional-, and/or watershed-scale planning and assessment. Project-specific permitting should consider plans that integrate current and future additional projects across the entire watershed or region, ecological values of the region, goals and values of rights holders and stakeholders (including those across international boundaries), and potential cumulative effects. These considerations could be used to develop scenarios for future social-ecological alternative states of the ecosystem based on the complete development of natural resources in that watershed.

Impact assessment

Across northwestern North America, the process of assessing the potential environmental impacts of a proposed project and approving its construction may be overseen by federal, municipal, provincial, state, territorial, and/or First Nations and Tribal entities. The lead entities for each assessment depend on the project location, its size, and the types of permits and approvals required. The predominant modern legal tool for evaluating and/or approving proposed mines is impact assessment. Throughout the review, we use this term broadly to cover other jurisdiction-dependent terms such as environmental assessment (EA), environmental impact statement, or risk assessment. Impact assessment is intended to weigh predicted impacts against the public interest and likelihood of significant adverse effects to inform decision-making and ensure the development of proper mitigation measures (*123*, *126*–*128*).

There is general scientific concern that impact assessments do not always meet internationally accepted standards for environmental review and decision-making, including scientific rigor, open data and methods, and independent review (*123*, *129*). A recent study on the role of science in Canada's impact assessment processes concluded that proponent-collected data for a single project do not and cannot capture systemic cumulative effects (*123*). These flaws can result in assessment reports that neither accurately weigh environmental risks nor provide realistic predictions of economic benefits, thus compromising decision-making and environmental protection (*123*, *129*–*132*). Although there have been recent efforts in Canada, for example, to provide more publicly available data related to cumulative effect estimation, data and impact prediction models associated with specific project assessments are consistently unavailable to the public. Project assessments that often rely on proprietary and non–peer-reviewed data stand in contrast to the global expectation in the research community for scientific data and methodologies to be open, freely available, and meeting standards of interoperability, reuse, and peer review within the constraints of applicable data privacy laws (*133*, *134*).

Considering the foundational importance of impact assessment to mining governance, it is critical to determine whether assessments provide accurate estimates of risks. While there are many examples of mines causing harm to freshwater ecosystems via a variety of direct and indirect pathways, these examples do not reveal whether harm is commonplace or rare. Ideally, to determine the extent to which assessed impacts are comprehensive and accurate, researchers would undertake studies that systematically compare the predicted impacts outlined during the impact assessment process with observed project impacts over the life of the mine. To our knowledge, there is only one such study in North America. The authors found that 16 of 25 hardrock mines exhibited poorer water quality than predicted in the environmental impact statements (EISs), representing clear failures in water quality mitigation (*135*). Thus, measured impacts exceeded predicted impacts for the majority of mines studied. Kuipers *et al.* (*135*) concluded that additional similar studies have not happened because (i) impact assessment predictions, along with baseline and operational water quality data, are sometimes unavailable or proprietary; (ii) data that are available can be spread across multiple repositories using combinations of microfiche, paper, and digital records; and/or (iii) available data do not have sufficient temporal, spatial, or methodological replication to facilitate robust comparative statistics (*135*, *136*). In summary, while there are many examples of mines causing harm to freshwater ecosystems via a variety of direct and indirect pathways, a lack of transparency and access to data throughout the mining governance cycle currently prohibits a robust and systematic analysis of predicted versus observed impacts.

In light of these challenges, we outline four key scientific questions intended to promote a transparent discussion of whether impact assessment processes are sufficiently considering risk and uncertainty in complex and dynamic salmonid-bearing watersheds: (i) To what extent is stressor complexity acknowledged and analyzed? (ii) Are cumulative effects sufficiently inventoried and quantified? (iii) Are long-term mitigation strategies based on proven technology and robust to future change? (iv) Are climate change risks incorporated into impact assessment and mitigation strategies?

To what extent is stressor complexity acknowledged and analyzed?

Our understanding of the pathways of mining impacts on salmonid-bearing watersheds will continue to evolve. Therefore, science-based mining policy must strive to minimize lags in applying new knowledge and, when necessary, acknowledge the uncertainty presented by the complex interactions of multiple stressors. Mixtures of metals leaching into rivers via mining projects provide a useful illustration. The current regulation of mining pollution is typically based on water quality standards developed from acute or chronic dose-response relationships for single stressors, evaluated for a limited number of organisms, usually in laboratory settings. However, we note that under the National Pollutant Discharge Elimination System, the U.S. Environmental Protection Agency recommends whole effluent toxicity testing with sensitive aquatic organisms to better assess potential problems caused by mixtures of pollutants (https://epa.gov/npdes/permit-limits-whole-effluent-toxicity-wet). Relying solely on acute and chronic water quality criteria overlooks the indirect effects and multiple interacting pathways of contaminant exposure, which can alter individual behavior or ecological interactions with directly affected species. In addition, the toxicity of some metals to aquatic organisms is controlled by other components of water quality that affect metal speciation (e.g., dissolved organic carbon or pH) or competition for biotic ligands [e.g., calcium (Ca^{2+})] (*137*). Factors such as dissolved organic carbon that can reduce the toxicity of metals such as copper tend to occur at low levels in the steep-sloped and thin-soil mountain environments found throughout northwestern North America (*138*). In addition to water quality conditions, additive or synergistic effects of multiple metals are not considered when establishing water quality criteria (*139*). Metal concentrations are time-consuming and expensive to monitor (*140*), can be difficult or impossible to reduce at large legacy sites with preexisting contamination (*71*, *141*, *142*), and their effects on aquatic organisms can be complex to quantify (*139*, *143*). While postmining pollution trajectories can be reversed even in severely degraded watersheds, restoration activities often begin many years after mining operations cease, can cost tens of millions of dollars for individual projects, and may not demonstrate ecosystem benefits for one to several decades after restoration begins (*107*, *144*).

Are cumulative effects sufficiently inventoried and quantified?

Current environmental legislation in the United States and Canada typically requires the assessment of cumulative effects relative to the scale of an individual proposed project rather than taking a regional multiproject approach.

Previous studies from the past two decades have noted the tendency for cumulative effect analyses to underestimate impacts and be overly narrow in scope, which can collectively introduce considerable scientific uncertainty (*121*, *123*, *145*, *146*). Underpredictions of risk and impact are exacerbated when multiple mines and other resource extraction activities such as logging occur within a single watershed yet are evaluated in isolation (*147*–*149*). The additive or synergistic amplification of mining activities (Figs. 3 and 4) (*150*) may put salmonid-bearing watersheds at risk when mine assessment, permitting, and development occur within one jurisdiction but impacts extend far downstream and span multiple jurisdictions. Narrow scoping of the spatial scale of impacts can exclude downstream governments and communities from the processes governing mine assessment, permitting, and regulation (*151*). Riverine transport of mining pollution and its associated risks can extend far downstream. For example, selenium and nitrate contamination from the Elk Valley metallurgical coal mines in southeastern BC have been measured over 250 km downstream, crossing the international boundary into U.S. and Tribal territories (*8*). The long-distance migration of salmonids, which can exceed hundreds of kilometers, potentially exposes individual fish to multiple mines or other development projects throughout their lifetime. The spatial and temporal extent of accounting for environmental risks should be aligned with the true scale of impact, which can often stretch from headwaters to estuary (*152*).

Are long-term mitigation strategies based on proven technology and robust to future change?

A critical source of uncertainty in predicting mining impacts is verifying the efficacy of long-term mitigation, including infrastructure such as water treatment facilities, tailings reservoir liners, and water control structures. Despite the consideration of mitigation measures in modern impact assessment processes, mining continues to harm watersheds. Recent publicized examples of unforeseen impacts within the salmonid-bearing watersheds of northwestern North America include the following: (i) a catastrophic tailings dam collapse at the Mount Polley Mine in BC (*87*); (ii) excessive and continuous discharge of polluted water at the Buckhorn Mine in Washington State (*153*); (iii) filling of open pits and stalled water treatment due to unforeseen permafrost thaw at the Red Dog Mine in Alaska (*154*, *155*); (iv) extreme rains leading to untreated mine-contact water discharge to the Yukon River from the Minto Mine in the Yukon Territory (*156*); and (v) a salmonid fish kill at Line Creek Coal Mine in BC due to water treatment plant malfunction (*157*). There is evidence that the water quality values predicted during the impact assessment process and the mitigations needed to properly treat water are overly optimistic and often fail (*135*), but as we note above, formal studies on this are exceedingly rare.

Mitigation technology for projects that move into operational phases should be fully funded, proven, and scalable before mine production begins, rather than based on theoretical or laboratory-tested technologies that lack validation at the scale of the operating mine. This not only is limited to wastewater management but also extends to mitigation and compensation for degraded physical salmonid habitat. Many projects result in an overall loss of important habitat when mitigations fall short of predicted effectiveness (*158*). In general, there is a need to develop consistent, quantifiable milestones that rely upon empirical data and verified methods for evaluating and adaptively correcting mitigation technologies when they fail to meet performance expectations (*159*). When mitigation for large-scale projects may not be feasible because of a lack of proven technology or the practical challenges of remote settings, this should be accurately conveyed and considered during impact assessments.

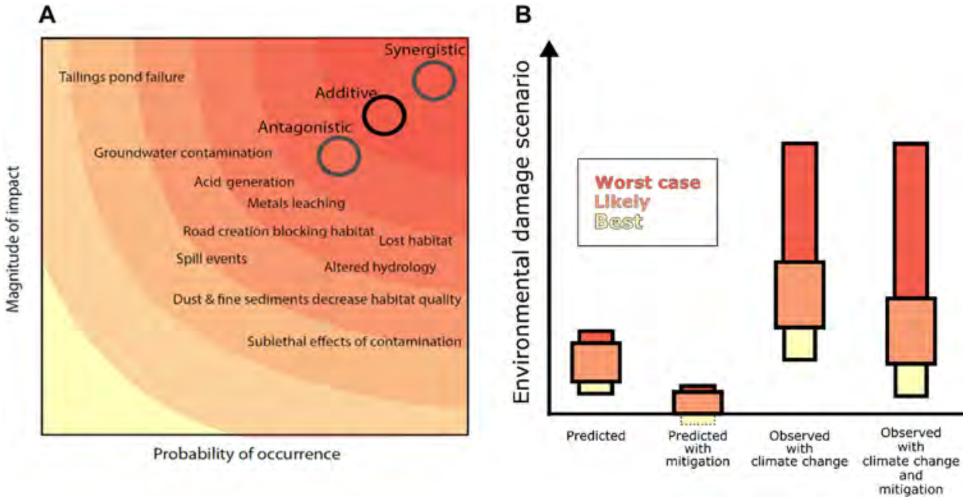

Fig. 4. Conceptual diagrams of cumulative and assessed risks resulting from mining activities. (**A**) Mining activities pose risks that vary in magnitude of impact × probability of occurrence. Yellow, lower risk; red, higher risk. Activities are placed for illustration purposes only, and the actual placement of individual activities relies on specific project details. As reviewed in (*150*), combined risks, which are represented by circles, can be antagonistic (combined effect of multiple stressors is less than the sum of individual effects), additive (combined effect is the sum of individual effects), or synergistic (combined effect is greater than the sum of individual effects). (**B**) Scenarios of environmental damage predicted during the impact assessment process and the proposed mitigation strategies can have unacknowledged uncertainty introduced by poorly quantified cumulative effects and climate change. In some cases, project proponents may assert that the proposed mitigation will improve environmental conditions (light yellow bar below the horizontal axis).

Are climate change risks incorporated into impact assessment and mitigation strategies?

Climate change and associated natural hazards intensify environmental risks and pose direct challenges to the performance of mining infrastructure and mitigation technology (Fig. 4) (*156*, *160*, *161*). As noted earlier, climate change is shifting the patterns of extreme precipitation events and the resulting riverine flow regimes. The steady transition from mainly ice- and snow-fed runoff patterns to mixed snow- and rain-fed runoff patterns will challenge engineers to design adaptive facilities that can withstand environmental changes occurring over decades to centuries. In southeastern Alaska and northern BC, up to 97% of extreme precipitation events occur because of tropic-originating atmospheric rivers (*162*). The frequency of these events is expected to increase through the 21st century (*78*), resulting in a greater number of rain-on-snow runoff events. Mine infrastructure has typically been built under the assumption that the current variability of the physical environment will not change (*156*, *161*). For example, infrastructure built to withstand an extreme precipitation event with a recurrence interval of 1 in 200 may wrongly assume that this magnitude will remain static over time (*163*, *164*). Robust infrastructure is especially important for projects in northwestern North America such as Kerr-Sulphurets-Mitchell, a proposed mine crossing the Unuk and Nass River watersheds (Fig. 1) that proposes to store and treat water for at least 200 years after mine closure (*165*).

It is impossible to predict with certainty whether water storage and treatment infrastructure will be able to withstand the unknown envelope of environmental variability and unforeseen extreme weather events and earthquakes over two centuries. Climate shifts are already affecting operations at the Red Dog Mine near Kotzebue, Alaska, one of the world's largest zinc mines that began in 1989; accelerated permafrost thaw due to increasing air temperatures has overwhelmed wastewater treatment and water management facilities and led to tens of millions of USD in infrastructure upgrades (*154*, *155*). Discharge from Red Dog Mine eventually drains to the Wulik River, an important salmonid watershed for the people of Kivalina, Alaska (*14*, *166*). When mining projects are confronted with climate change–induced uncertainty, scenario analysis could be a key tool for illuminating future problems that are difficult to estimate in the present with statistical certainty. Scenario analyses are a form of visioning exercises that use a structured process for exploring the potential opportunities, risks, and decision-making necessary to adapt to alternative visions of future environmental conditions (*167*). In some cases, climate change may create an especially complex future by improving the suitability of watershed habitat for salmonids. For example, glacier retreat is opening up hundreds of kilometers of new salmon habitat in the mountainous regions of northwestern North America over the coming decades; thus, mines may compromise the viability of habitat that is not important for salmon now but will be in the future (*26*). Actual mining risks could be much greater than assessed (Fig. 4B), and future-looking analyses of habitat potential could clarify these unassessed risks.

The intrinsic complexity of salmonid-bearing watersheds under climate change pressures suggests that impact assessments should adhere to precautionary approaches and use ongoing environmental effects monitoring during all stages of mining activities to allow for adaptation of reclamation efforts when environmental conditions change. To design infrastructure that accounts for the environmental variability brought about by climate change and the dynamic nature of watersheds, rigorous baseline data collection is critical for properly capturing system variability. For example, river discharge data should be collected consistently (e.g., at the 15-min or hourly time scale) with few temporal gaps for a minimum of 5 to 10 years, but as much as 15 years of initial data collection may be required until hydrologic metrics can be accurately calculated (*168*) and used for infrastructure design. The Alaska Highway Drainage Manual (*169*) recognizes the importance of surface water variability when designing bridges and culverts, stating, "A complete [discharge] record is usually defined as one having at least 10 years of continuous record. Twenty-five years of record is considered optimal." Hourly to daily water quality measurements are often necessary to accurately define extreme conditions (*170*), but mine monitoring programs typically prescribe weekly to quarterly measurement intervals that are unlikely to efficiently detect trends and the true range of water quality variability.

Project approval or denial

After completing the impact assessment, which can take several years, decision-makers render a decision on the fate of the project. When faced with substantial uncertainty or lack of robust baseline data, impact assessment and permit processes for proposed mines are increasingly considering "no-go" as a valid response. Mechanisms for this exist in both the United States and Canada. For example, in the United States, an Environmental Assessment (EA) is guided by NEPA. If significant project impacts are expected, then a broader EIS will follow. Under an EA process facilitated by a lead federal agency, all project assessments must include a "no action alternative" to provide reasoned context for understanding the significance of the negative environmental impacts of a proposed project (Canadian laws require a "no project" option). In some cases, the analysis of potential project impacts generated by the EA process supports the decision of a federal agency to deny the issuance of individual permits. For example, at the conclusion of the EA and Clean Water Act review processes for the Pebble Mine in 2020, the U.S. Army Corps of Engineers did not issue the Section 404 Clean Water Act permit. This decision delayed the potential construction of the mine, located within the greater Bristol Bay watershed of Alaska (Fig. 1), where it was determined that construction would result in adverse impacts to wetlands that could not be adequately mitigated (*171*). While politics can undoubtedly play a role in these types of decisions (*172*), we make the point here that mechanisms are in place that allow for the denial of key permits, but this is not the case for all agencies. In the United States, for example, the Bureau of Land Management and Forest Service may require modifications to a mining plan, but they cannot deny it outright. In Canada, recent rejections include the New Prosperity Gold-Copper Mine Project (*173*), Grassy Mountain Coal Project (*174*), and the Morrison Copper-Gold Project. For the Morrison Project, BC officials specifically stated that "there remain uncertainties and risks to fish and water quality," which were deemed not in the public interest (*175*).

Operations

During operations, mining projects generally monitor for environmental impacts that exceed regulatory thresholds. If monitoring detects environmental harm or a failure of mitigation technology, then mining operations and mitigations should, in theory, be adjusted to maintain performance. As noted in the "Impact assessment" section above, there are several major scientific challenges with this in practice. There are issues with data transparency in some mining sectors. Mitigation approaches can fail. Climate change and associated natural hazards are changing. Monitoring programs may not be designed to capture the true scope of impacts, especially as scientific knowledge evolves. To illustrate this point, as methods of toxicity determination increase in sensitivity and sophistication, there is growing evidence that some contaminants have impacts at lower concentrations than previously assumed. For example, toxicity thresholds for selenium have been revised downward over time (*176*).

We recommend that working groups across all levels of affected governments be formed to consolidate basic mining information into publicly available, user-friendly, and annually updated data portals that transcend political boundaries. Many data sources on mine locations, reclamation costs, and other basic operational details are unavailable or diffuse (see "The mining landscape of northwestern North America" section above, the "Closure" section below, and the Supplementary Materials). Before consideration of a new mining operation begins, all potentially affected jurisdictions should agree to consistent protocols that lead to a collaborative, watershed-scale monitoring and evaluation program. This program should include agreement on specific monitoring objectives and define the final reporting based on those objectives. The envisioned final reporting products would guide monitoring program design, including defined roles and responsibilities, identification of reference sites, sufficient sampling frequency, and a high likelihood to detect changes to the environment due to potential mining impacts (*177*). Trade-offs in impact assessment and monitoring design are expected for any monitoring program, but it is important for all potentially affected jurisdictions to explicitly acknowledge potential funding gaps and formally agree upon compromises made during permitting and monitoring program development.

Collison *et al.* (*178*) recently highlighted a regulatory loophole that may enable harm to freshwater systems from mining operations once the impact assessment process has concluded. Their systematic examination of approved and operating mines in BC found that 65% requested amendments after approval, with 98% of requests approved. Almost half of the amendments were assessed as having the potential to harm aquatic ecosystems, such as increasing the authorized amount of harm to fish habitat or increasing water extraction. Most amendments were issued within less than 2 years of mine approval and were not subject to the same level of scientific and public scrutiny as the impact assessment process. Although the first documented case of amendment-related "scope creep," this regulatory challenge likely applies to impact assessment laws in other jurisdictions.

Closure

The reclamation and closure of mines can be expensive, and there can be challenges with financial liability. Bonds based on reclamation estimates are intended to guarantee that mining companies will bear the cost of standard mine reclamation and closure (*179*). Small placer operations may be exempt from bonding. Of the 26 largest operating metal and coal mines in our study region (teal circles in Fig. 1), 21 provide publicly available bond amounts or company-estimated reclamation and closure costs. At the time of our research, bonding and financial assurance costs ranged from 95,000 USD for the Golden Chest Mine in northern Idaho to nearly 586 million USD for the Red Dog Mine in northwestern Alaska (Fig. 1, fig. S1, and table S1). Individual bond amounts are not publicly available for the five mines creating the Teck Coal Elk Valley complex (Elkview, Fording River, Line Creek, Coal Mountain, and Greenhills), but together, their reclamation liability amounts to 1.4 billion CAD, and the current bond amount is approximately 900 million CAD, representing an approximately 500 million CAD shortfall (*12*). Although reclamation bond amounts are subject to high uncertainty (*180*), available information indicates that it will take billions of dollars to reclaim northwestern North America mine sites.

Intuitively, bond amounts should increase with mine size and environmental risk. We found that although bond amounts tend to increase with ore milling rate, there was no clear correlation between bond amount and mine size (fig. S1). Therefore, it is difficult to evaluate the consistency across bond estimates and whether they represent an accurate financial estimate of potential reclamation (fig. S1 and table S1). We were able to extract consistent estimates of milling rate across northwestern North America's large mines, but that is only one indicator of a mine's environmental footprint and potential liability. Other mine characteristics such as disturbed area, acid-generating potential, water quality treatment needs, and equipment removal are included in the overall calculation (*181*), but we did not find sources for consistently extracting this additional information. The financial liability of mining companies for their environmental legacy warrants further attention and supports the notion that the development of financial assurance at each mine site should include a transparent review process with consistent reporting listing how each variable adds up to the final amount.

Policy in transboundary watersheds

Mining policy is complicated in our study region by watersheds that span international boundaries between the United States, Canada, and Indigenous territories. These "transboundary" watersheds represent complicated sociopolitical landscapes, where governance of water, fisheries, and resource extraction are often conflicting or inadequately defined (*182*). This can cause fragmented and inconsistent decision-making regarding the siting of mines, EA, permitting, and regulatory enforcement. For example, water quality criteria can differ across adjacent segments of the same watershed, with associated inconsistencies in the methodologies for calculating, monitoring, and regulating exceedances (*183*, *184*). Likewise, the inherent downstream transport of mine effluent complicates effective permitting and oversight of mines because the assessment of risks in one jurisdiction may not adequately account for the consequences of impacts realized in another jurisdiction (*151*, *184*). Although downstream jurisdictions may be invited to provide public comments during the assessment process, they are often excluded from formal decision-making and have limited avenues for legal recourse.

One avenue for improving transboundary impact assessment is the International Joint Commission (IJC), which exists to oversee the Boundary Waters Treaty of 1909 and to prevent and resolve disputes regarding U.S.-Canada transboundary lakes and rivers (*185*). The IJC set precedent in our study region when they intervened on behalf of the United States and Canada in 1985 to evaluate the potential impacts of a proposed open-pit coal mine in the BC headwaters of the transboundary Flathead River. Following 3 years of impact assessment undertaken by a binational team of scientists, the IJC ruled against the approval of the mine based on the potential impacts to water quality and critical spawning and rearing habitat for transboundary bull trout populations (*186*). This precautionary ruling by the IJC is an example of a watershed-scale impact assessment process that relied upon binational, transparent, and objective science to inform preservation of the Flathead watershed's endangered salmonid populations (*187*).

LOOKING AHEAD

In this review, we have linked current scientific understanding of watershed ecology and salmonid biology with the pathways of mining impacts to salmonids and their habitats. The body of knowledge presented here supports the notion that the risks and impacts of mining have been underestimated across the watersheds of northwestern North America. To facilitate future transparent discussions of risk and scientific uncertainty, we posed four questions related to watershed stressor complexity, cumulative effects, long-term risk mitigation, and climate change. Considering these existing uncertainties, the application of the precautionary principle would help to ensure the protection of salmonid-bearing watersheds and the benefits that they provide for diverse peoples. There are many existing opportunities throughout the mining governance life cycle to improve the science behind mining policies, such as with regional planning, strengthened impact assessment, independent research and monitoring, and harmonization of data collection. Given that mining plays a role for the needs of society, there is an urgent need for current and future mining projects to be operated in such a way that protects our last remaining healthy watersheds and abundant salmonid populations.

SUPPLEMENTARY MATERIALS

Supplementary material for this article is available at https://science.org/doi/10.1126/sciadv.abn0929

REFERENCES AND NOTES

1. J. N. Moore, S. N. Luoma, Hazardous wastes from large-scale metal extraction: A case study. *Environ. Sci. Technol.* **24**, 1278–1285 (1990).
2. A. M. Farag, D. F. Woodward, J. N. Goldstein, W. Brumbaugh, J. S. Meyer, Concentrations of metals associated with mining waste in sediments, biofilm, benthic macroinvertebrates, and fish from the Coeur d'Alene River Basin, Idaho. *Arch. Environ. Contam. Toxicol.* **34**, 119–127 (1998).
3. K. L. Barry, J. A. Grout, C. D. Levings, B. H. Nidle, G. E. Piercey, Impacts of acid mine drainage on juvenile salmonids in an estuary near Britannia Beach in Howe Sound, British Columbia. *Can. J. Fish. Aquat. Sci.* **57**, 2032–2043 (2000).
4. K. Spitz, J. Trudinger, *Mining and the Environment: From Ore to Metal* (CRC Press, 2019).
5. M. Simonin, J. D. Rocca, J. R. Gerson, E. Moore, A. C. Brooks, L. Czaplicki, M. R. V. Ross, N. Fierer, J. M. Craine, E. S. Bernhardt, Consistent declines in aquatic biodiversity across diverse domains of life in rivers impacted by surface coal mining. *Ecol. Appl.* **31**, e02389 (2021).
6. M. Olías, J. M. Nieto, Background conditions and mining pollution throughout history in the Río Tinto (SW Spain). *Environments* **2**, 295–316 (2015).
7. D. Kossoff, W. E. Dubbin, M. Alfredsson, S. J. Edwards, M. G. Macklin, K. A. Hudson-Edwards, Mine tailings dams: Characteristics, failure, environmental impacts, and remediation. *Appl. Geochem.* **51**, 229–245 (2014).
8. C. A. Mebane, C. G. Schmidt, Selenium and mercury in the Kootenai River, Montana and Idaho, 2018–2019 (2019); https://doi.org/10.5066/P9YYVV7R.
9. B. G. Lottermoser, *Mine Wastes: Characterization, Treatment, and Environmental Impacts* (Springer, 2010).
10. D. M. Franks, M. Stringer, L. A. Torres-Cruz, E. Baker, R. Valenta, K. Thygesen, A. Matthews, J. Howchin, S. Barrie, Tailings facility disclosures reveal stability risks. *Sci. Rep.* **11**, 5353 (2021).
11. U.S. Government Accountability Office, Abandoned hardrock mines: Information on number of mines, expenditures, and factors that limit efforts to address hazards (2020); https://gao.gov/assets/710/705146.pdf.
12. BC Chief Inspector of Mines, Chief Inspector of Mines 2019 annual report (2021); https://gov.bc.ca/assets/gov/farming-natural-resources-and-industry/mineral-exploration-mining/documents/health-and-safety/ci-annual-reports/112869_2019_ci_annual_rpt.pdf.
13. B. K. Sovacool, S. H. Ali, M. Bazilian, B. Radley, B. Nemery, J. Okatz, D. Mulvaney, Sustainable minerals and metals for a low-carbon future. *Science* **367**, 30–33 (2020).
14. C. A. Woody, R. M. Hughes, E. J. Wagner, T. P. Quinn, L. H. Roulson, L. M. Martin, K. Griswold, The mining law of 1872: Change is overdue. *Fisheries* **35**, 321–331 (2010).
15. R. Hilborn, T. P. Quinn, D. E. Schindler, D. E. Rogers, Biocomplexity and fisheries sustainability. *Proc. Natl. Acad. Sci. U.S.A.* **100**, 6564–6568 (2003).
16. X. Augerot, D. N. Foley, C. Steinback, A. Fuller, N. Fobes, K. Spencer, *Atlas of Pacific Salmon: The First Map-Based Status Assessment of Salmon in the North Pacific* (University of California Press, 2005).
17. R. S. Waples, M. Nammack, J. F. Cochrane, J. A. Hutchings, A tale of two acts: Endangered species listing practices in Canada and the United States. *Bioscience* **63**, 723–734 (2013).

18. G. Grill, B. Lehner, M. Thieme, B. Geenen, D. Tickner, F. Antonelli, S. Babu, P. Borrelli, L. Cheng, H. Crochetiere, H. E. Macedo, R. Filgueiras, M. Goichot, J. Higgins, Z. Hogan, B. Lip, M. E. McClain, J. Meng, M. Mulligan, C. Nilsson, J. D. Olden, J. J. Opperman, P. Petry, C. R. Liermann, L. Sáenz, S. Salinas-Rodríguez, P. Schelle, R. J. P. Schmitt, J. Snider, F. Tan, K. Tockner, P. H. Valdujo, A. Soesbergen, C. Zarfl, Mapping the world's free-flowing rivers. *Nature* **569**, 215–221 (2019).
19. F. R. Hauer, H. Locke, V. J. Dreitz, M. Hebblewhite, W. H. Lowe, C. C. Muhlfeld, C. R. Nelson, M. F. Proctor, S. B. Rood, Gravel-bed river floodplains are the ecological nexus of glaciated mountain landscapes. *Sci. Adv.* **2**, e1600026 (2016).
20. D. E. Schindler, X. Augerot, E. Fleishman, N. J. Mantua, B. Riddell, M. Ruckelshaus, J. Seeb, M. Webster, Climate change, ecosystem impacts, and management for Pacific salmon. *Fisheries* **33**, 502–506 (2008).
21. R. S. Waples, G. R. Pess, T. Beechie, Evolutionary history of Pacific salmon in dynamic environments. *Evol. Appl.* **1**, 189–206 (2008).
22. K. J. Pitman, J. W. Moore, M. R. Sloat, A. H. Beaudreau, A. L. Bidlack, R. E. Brenner, E. W. Hood, G. R. Pess, N. J. Mantua, A. M. Milner, V. Radic, G. H. Reeves, D. E. Schindler, D. C. Whited, Glacier retreat and Pacific salmon. *Bioscience* **70**, 220–236 (2020).
23. D. R. Montgomery, J. M. Buffington, Channel-reach morphology in mountain drainage basins. *Geol. Soc. Am. Bull.* **109**, 596–611 (1997).
24. A. M. Milner, A. L. Robertson, L. E. Brown, S. H. Sønderland, M. Mcdermott, A. J. Veal, Evolution of a stream ecosystem in recently deglaciated terrain. *Ecology* **92**, 1924–1935 (2011).
25. D. C. Whited, M. S. Lorang, M. J. Harner, F. R. Hauer, J. S. Kimball, J. A. Stanford, Climate, hydrologic disturbance, and succession: Drivers of floodplain pattern. *Ecology* **88**, 940–953 (2007).
26. K. J. Pitman, J. W. Moore, M. Huss, M. R. Sloat, D. C. Whited, T. J. Beechie, R. Brenner, E. W. Hood, A. W. Milner, G. R. Pess, G. H. Reeves, D. E. Schindler, Glacier retreat creating new Pacific salmon habitat in western North America. *Nat. Commun.* **12**, 6816 (2021).
27. J. A. Falke, B. M. Huntsman, E. R. Schoen, Climatic variation drives growth potential of juvenile Chinook salmon along a subarctic boreal riverscape, in *Advances in Understanding Landscape Influences on Freshwater Habitats and Biological Assemblages* (American Fisheries Society, 2019).
28. S. R. Brennan, D. E. Schindler, T. J. Cline, T. E. Walsworth, G. Buck, D. P. Fernandez, Shifting habitat mosaics and fish production across river basins. *Science* **364**, 783–786 (2019).
29. J. W. Moore, M. P. Beakes, H. K. Nesbitt, J. D. Yeakel, D. A. Patterson, L. A. Thompson, C. C. Phillis, D. C. Braun, C. Favaro, D. Scott, C. Carr-Harris, W. I. Atlas, Emergent stability in a large, free-flowing watershed. *Ecology* **96**, 340–347 (2015).
30. S. M. Gende, R. T. Edwards, M. F. Willson, M. S. Wipfli, Pacific salmon in aquatic and terrestrial ecosystems. *Bioscience* **52**, 917–928 (2002).
31. D. E. Schindler, M. D. Scheuerell, J. W. Moore, S. M. Gende, T. B. Francis, W. J. Palen, Pacific salmon and the ecology of coastal ecosystems. *Front. Ecol. Environ.* **1**, 31–37 (2003).
32. T. Levi, C. T. Darimont, M. MacDuffee, M. Mangel, P. Paquet, C. C. Wilmers, Using grizzly bears to assess harvest-ecosystem tradeoffs in salmon fisheries. *PLoS Biol.* **10**, e1001303 (2012).
33. J. W. Moore, D. E. Schindler, C. P. Ruff, Habitat saturation drives thresholds in stream subsidies. *Ecology* **89**, 306–312 (2008).
34. J. J. Verspoor, D. C. Braun, M. M. Stubbs, J. D. Reynolds, Persistent ecological effects of a salmon-derived nutrient pulse on stream invertebrate communities. *Ecosphere* **2**, art18 (2011).
35. J. C. Walsh, J. E. Pendray, S. C. Godwin, K. A. Artelle, H. K. Kindsvater, R. D. Field, J. N. Harding, N. R. Swain, J. D. Reynolds, Relationships between Pacific salmon and aquatic and terrestrial ecosystems: Implications for ecosystem-based management. *Ecology* **101**, e03060 (2020).
36. J. R. Griffiths, D. E. Schindler, J. B. Armstrong, M. D. Scheuerell, D. C. Whited, R. A. Clark, R. Hilborn, C. A. Holt, S. T. Lindley, J. A. Stanford, E. C. Volk, Performance of salmon fishery portfolios across western North America. *J. Appl. Ecol.* **51**, 1554–1563 (2014).
37. R. G. Gustafson, R. S. Waples, J. M. Myers, L. A. Weitkamp, G. J. Bryant, O. W. Johnson, J. J. Hard, Pacific salmon extinctions: Quantifying lost and remaining diversity. *Conserv. Biol.* **21**, 1009–1020 (2007).
38. V. S. D'Angelo, C. C. Muhlfeld, Factors influencing the distribution of native bull trout and westslope cutthroat trout in streams of western Glacier National Park, Montana. *Northwest Sci.* **87**, 1–11 (2013).
39. L. G. Crozier, M. M. McClure, T. Beechie, S. J. Bograd, D. A. Boughton, M. Carr, T. D. Cooney, J. B. Dunham, C. M. Greene, M. A. Haltuch, E. L. Hazen, D. M. Holzer, D. D. Huff, R. C. Johnson, C. E. Jordan, I. C. Kaplan, S. T. Lindley, N. J. Mantua, P. B. Moyle, J. M. Myers, M. W. Nelson, B. C. Spence, L. A. Weitkamp, T. H. Williams, E. Willis-Norton, Climate vulnerability assessment for Pacific salmon and steelhead in the California Current Large Marine Ecosystem. *PLOS ONE* **14**, e0217711 (2019).
40. J. Lichatowich, *Salmon without Rivers: A History of the Pacific Salmon Crisis* (Island Press, 2001).
41. T. P. Quinn, *The Behavior and Ecology of Pacific Salmon and Trout* (University of Washington Press, 2018).
42. P. S. Ross, C. J. Kennedy, L. K. Shelley, K. B. Tierney, D. A. Patterson, W. L. Fairchild, R. W. Macdonald, The trouble with salmon: Relating pollutant exposure to toxic effect in species with transformational life histories and lengthy migrations. *Can. J. Fish. Aquat. Sci.* **70**, 1252–1264 (2013).
43. A. Garibaldi, N. Turner, Cultural keystone species: Implications for ecological conservation and restoration. *Ecol. Soc.* **9**, 1 (2004).
44. D. Lepofsky, K. Lertzman, D. Hallett, R. Mathewes, Climate change and culture change on the southern coast of British Columbia 2400-1200 cal. BP: An hypothesis. *Am. Antiq.* **70**, 267–293 (2005).
45. S. K. Campbell, V. L. Butler, Archaeological evidence for resilience of Pacific Northwest salmon populations and the socioecological system over the last ~7,500 years. *Ecol. Soc.* **15**, 17 (2010).
46. L. Chan, O. Receveur, D. Sharp, H. Schwartz, A. Ing, C. Tikhonov, *First Nations Food, Nutrition and Environment Study (FNFNES): Results from British Columbia (2008/2009)* (University of Northern British Columbia, Université de Montréal, and Assembly of First Nations, 2011).
47. H. K. Nesbitt, J. W. Moore, Species and population diversity in Pacific salmon fisheries underpin indigenous food security. *J. Appl. Ecol.* **53**, 1489–1499 (2016).
48. W. I. Atlas, N. C. Ban, J. W. Moore, A. M. Tuohy, S. Greening, A. J. Reid, N. Morvin, E. White, W. G. Housty, J. A. Housty, C. N. Service, L. Greba, S. Harrison, C. Sharpe, K. I. R. Butts, W. M. Shepert, E. Sweeney-Bergen, D. Macintyre, M. R. Sloat, K. Connors, Indigenous systems of management for culturally and ecologically resilient Pacific salmon (*Oncorhynchus* spp.) fisheries. *Bioscience* **71**, 186–204 (2021).
49. J. A. Fall, A. Godduhn, G. Halas, L. Hutchinson-Scarbrough, B. Jones, B. McDavid, E. Mikow, L. A. Sill, T. Lemons, "Alaska subsistence and personal use salmon fisheries 2017 annual report" (Technical Paper no. 451, Alaska Department of Fish and Game, Division of Subsistence, 2020).
50. A. C. Johnson, J. R. Bellmore, S. Haught, R. Medel, Quantifying the monetary value of Alaska national forests to commercial Pacific salmon fisheries. *N. Am. J. Fish. Manag.* **39**, 1119–1131 (2019).
51. J. W. Brownscombe, S. D. Bower, W. Bowden, L. Nowell, J. D. Midwood, N. Johnson, S. J. Cooke, Canadian recreational fisheries: 35 years of social, biological, and economic dynamics from a national survey. *Fisheries* **39**, 251–260 (2014).
52. U.S. Geological Survey, "Mineral commodity summaries 2020" (2020); https://pubs.er.usgs.gov/publication/mcs2020.
53. Energy and Mines Ministers' Conference, "Natural resources: Major projects planned or under construction — 2020 to 2030" (2020); www.nrcan.gc.ca/science-data/data-analysis/major-projects-inventory/22218.
54. R. J. Goldfarb, C. D. Taylor, A. L. Meier, W. M. D'Angelo, R. M. O'Leary, in *Geologic Studies in Alaska by the US Geological Survey, 1995* (United States Government Printing Office, 1995), pp. 3–18.
55. J. D. LaPerriere, S. M. Wagener, D. M. Bjerklie, Gold-mining effects on heavy metals in streams, circle quadrangle, Alaska. *J. Am. Water Resour. Assoc.* **21**, 245–252 (1985).
56. K. J. Buhl, S. J. Hamilton, Comparative toxicity of inorganic contaminants released by placer mining to early life stages of salmonids. *Ecotoxicol. Environ. Saf.* **20**, 325–342 (1990).
57. R. I. Ferguson, M. Church, C. D. Rennie, J. G. Venditti, Reconstructing a sediment pulse: Modeling the effect of placer mining on Fraser River, Canada. *J. Geophys. Res. Earth.* **120**, 1436–1454 (2015).
58. Alaska Department of Environmental Conservation, "Impaired waterbody listing" (2018); https://mywaterway.epa.gov/state/AK/water-quality-overview.
59. B. A. Wills, J. A. Finch, *Wills' Mineral Processing Technology: An Introduction to the Practical Aspects of Ore Treatment and Mineral Recovery* (Butterworth-Heinemann, 2015).
60. J. B. Whyte, J. Cumming, *Mining Explained: A Layman's Guide* (Northern Miner, 2007).
61. S. J. Hamilton, K. J. Buhl, Selenium in water, sediment, plants, invertebrates, and fish in the Blackfoot River drainage. *Water Air Soil Pollut.* **159**, 3–34 (2004).
62. L. M. Zinsser, C. A. Mebane, G. C. Mladenka, L. R. Van Every, M. L. Williams, "Spatial and temporal trends in selenium in the upper Blackfoot River watershed, southeastern Idaho, 2001–16" (USGS Scientific Investigations Report 2018-5081, 2018).
63. G. J. S. Govett, *Rock Geochemistry in Mineral Exploration* (Elsevier, 2013).
64. D. M. Chambers, K. Zamzow, Documentation of acidic mining exploration drill cuttings at the Pebble Copper–Gold Mineral Prospect, Southwest Alaska. *Environments* **6**, 78 (2019).
65. U.S. Environmental Protection Agency, "Damage cases and environmental releases from mines and mineral processing sites" (EPA Report A530-R-99-023, EPA Publication, 1997).
66. U.S. Environmental Protection Agency, "Nationwide identification of hardrock mining sites" (EPA Report 2004-P-00005, EPA Publication, 2004).
67. P. C. Singer, W. Stumm, Acidic mine drainage: The rate-determining step. *Science* **167**, 1121–1123 (1970).
68. G. A. Tremblay, C. M. Hogan, Managing orphaned and abandoned mines–a Canadian perspective, paper presented at the Dealing with Derelict Mines Summit organized by CRC-Care, Singleton, NSW, 4 to 6 December 2016.

69. J. van Sickle, S. G. Paulsen, Assessing the attributable risks, relative risks, and regional extents of aquatic stressors. *J. North Am. Benthol. Soc.* **27**, 920–931 (2008).
70. R. Eisler, S. N. Wiemeyer, Cyanide hazards to plants and animals from gold mining and related water issues, in *Reviews of Environmental Contamination and Toxicology* (Springer, 2004), vol. 183, pp. 21–54.
71. S. N. Luoma, P. S. Rainbow, *Metal Contamination in Aquatic Environments: Science and Lateral Management* (Cambridge Univ. Press, 2011).
72. J. R. Ferrari, T. R. Lookingbill, B. McCormick, P. A. Townsend, K. N. Eshleman, Surface mining and reclamation effects on flood response of watersheds in the central Appalachian Plateau region. *Water Resour. Res.* **45**, W04407 (2009).
73. A. J. Miller, N. P. Zégre, Mountaintop removal mining and catchment hydrology. *Water* **6**, 472–499 (2014).
74. N. J. Shatilla, thesis, McMaster University, Hamilton, Ontario (2013).
75. A. Maest, R. Prucha, C. Wobus, Hydrologic and water quality modeling of the Pebble Mine Project pit lake and downstream environment after mine closure. *Minerals* **10**, 727 (2020).
76. I. T. Stewart, D. R. Cayan, M. D. Dettinger, Changes toward earlier streamflow timing across western North America. *J. Climate* **18**, 1136–1155 (2005).
77. N. Mantua, I. Tohver, A. Hamlet, Climate change impacts on streamflow extremes and summertime stream temperature and their possible consequences for freshwater salmon habitat in Washington State. *Clim. Change* **102**, 187–223 (2010).
78. V. Radić, A. J. Cannon, B. Menounos, N. Gi, Future changes in autumn atmospheric river events in British Columbia, Canada, as projected by CMIP5 global climate models. *J. Geophys. Res. Atmos.* **120**, 9279–9302 (2015).
79. J. D. Olden, R. J. Naiman, Incorporating thermal regimes into environmental flows assessments: Modifying dam operations to restore freshwater ecosystem integrity. *Freshw. Biol.* **55**, 86–107 (2010).
80. G. Power, R. S. Brown, J. G. Imhof, Groundwater and fish—Insights from northern North America. *Hydrol. Process.* **13**, 401–422 (1999).
81. L. Wirth, A. Rosenberger, A. Prakash, R. Gens, F. J. Margraf, T. Hamazaki, A remote-sensing, GIS-based approach to identify, characterize, and model spawning habitat for fall-run chum salmon in a sub-arctic, glacially fed river. *Trans. Am. Fish. Soc.* **141**, 1349–1363 (2012).
82. K. L. Webster, F. D. Beall, I. F. Creed, D. P. Kreutzweiser, Impacts and prognosis of natural resource development on water and wetlands in Canada's boreal zone. *Environ. Rev.* **23**, 78–131 (2015).
83. S. F. Railsback, Why it is time to put PHABSIM out to pasture. *Fisheries* **41**, 720–725 (2016).
84. M. J. Bradford, P. S. Higgins, J. Korman, J. Sneep, Test of an environmental flow release in a British Columbia river: Does more water mean more fish? *Freshw. Biol.* **56**, 2119–2134 (2011).
85. M. A. Palmer, E. S. Bernhardt, W. H. Schlesinger, K. N. Eshleman, E. Foufoula-Georgiou, M. S. Hendryx, A. D. Lemly, G. E. Likens, O. L. Loucks, M. E. Power, P. S. White, P. R. Wilcock, Mountaintop mining consequences. *Science* **327**, 148–149 (2010).
86. J. Lewin, B. E. Davies, P. J. Wolfenden, Interactions between channel change and historic mining sediments, in *River Channel Changes* (Wiley, 1977), pp. 353–367.
87. E. L. Petticrew, S. J. Albers, S. A. Baldwin, E. C. Carmack, S. J. Déry, N. Gantner, K. E. Graves, B. Laval, J. Morrison, P. N. Owens, D. T. Selbie, S. Vagle, The impact of a catastrophic mine tailings impoundment spill into one of North America's largest fjord lakes: Quesnel Lake, British Columbia, Canada. *Geophys. Res. Lett.* **42**, 3347–3355 (2015).
88. L. Nikl, B. Wernick, J. Van Geest, C. Hughes, K. McMahen, L. Anglin, Mount Polley mine embankment breach: Overview of aquatic impacts and rehabilitation, in *Proceedings Tailings and Mine Waste 2016*, Keystone, Colorado (UBC Studios, 2016), pp. 1–13.
89. A. K. Hamilton, B. E. Laval, E. L. Petticrew, S. J. Albers, M. Allchin, S. A. Baldwin, E. C. Carmack, S. J. Dery, T. D. French, B. Granger, K. E. Graves, P. N. Owens, D. T. Selbie, S. Vagle, Seasonal turbidity linked to physical dynamics in a deep lake following the catastrophic 2014 Mount Polley mine tailings spill. *Water Resour. Res.* **56**, e2019WR025790 (2020).
90. Z. Lyu, J. Chai, Z. Xu, Y. Qin, J. Cao, A comprehensive review on reasons for tailings dam failures based on case history. *Adv. Civ. Eng.* **2019**, 4159306 (2019).
91. BC Hydro, "New transmission line ready to power Northwest B.C." (2014); www.bchydro.com/news/press_centre/news_releases/2014/new-transmission-line-ready-to-power-northwest-bc.html/.
92. D. M. Price, T. Quinn, R. J. Barnard, Fish passage effectiveness of recently constructed road crossing culverts in the Puget Sound region of Washington State. *N. Am. J. Fish. Manag.* **30**, 1110–1125 (2010).
93. S. C. Trombulak, C. A. Frissell, Review of ecological effects of roads on terrestrial and aquatic communities. *Conserv. Biol.* **14**, 18–30 (2000).
94. L. E. Richards, Environmental management in the minerals industry: Theory and practice. *Bus. Strategy Environ.* **5**, 87–93 (1996).
95. G. Hilson, B. Murck, Sustainable development in the mining industry: Clarifying the corporate perspective. *Resour. Policy* **26**, 227–238 (2000).
96. E. V. Axtmann, S. N. Luoma, Large-scale distribution of metal contamination in the fine-grained sediments of the Clark Fork River, Montana, U.S.A. *Appl. Geochem.* **6**, 75–88 (1991).
97. K. J. Hartman, M. D. Kaller, J. W. Howell, J. A. Sweka, How much do valley fills influence headwater streams? *Hydrobiologia* **532**, 91–102 (2005).
98. T. T. Lindberg, E. S. Bernhardt, R. Bier, A. M. Helton, R. B. Merola, A. Vengosh, R. T. D. Giulio, Cumulative impacts of mountaintop mining on an Appalachian watershed. *Proc. Natl. Acad. Sci. U.S.A.* **108**, 20929–20934 (2011).
99. U.S. Environmental Protection Agency, "The effects of mountaintop mines and valley fills on aquatic ecosystems of the central Appalachian coalfields" (EPA Report EPA/600/R-09/138E, EPA Publication, 2011).
100. D. K. Nordstrom, C. N. Alpers, C. J. Ptacek, D. W. Blowes, Negative pH and extremely acidic mine waters from Iron Mountain, California. *Environ. Sci. Technol.* **34**, 254–258 (2000).
101. S. O. Helgen, J. N. Moore, Natural background determination and impact quantification in trace metal-contaminated river sediments. *Environ. Sci. Technol.* **30**, 129–135 (1995).
102. D. W. Blowes, C. J. Ptacek, J. L. Jambor, C. G. Weisener, The geochemistry of acid mine drainage, in *Environmental Geochemistry (Treatise on Geochemistry)* (Elsevier, 2005), vol. 9, pp. 149–204.
103. X. Zhang, L. Yang, Y. Li, H. Li, W. Wang, B. Ye, Impacts of lead/zinc mining and smelting on the environment and human health in China. *Environ. Monit. Assess.* **184**, 2261–2273 (2012).
104. C. Pearson, D. Howard, C. Moore, D. Obrist, Mercury and trace metal wet deposition across five stations in Alaska: Controlling factors, spatial patterns, and source regions. *Atmos. Chem. Phys.* **19**, 6913–6929 (2019).
105. D. Jaffe, E. Prestbo, P. Swartzendruber, P. Weiss-Penzias, S. Kato, A. Takami, S. Hatakeyama, Y. Kajii, Export of atmospheric mercury from Asia. *Atmos. Environ.* **39**, 3029–3038 (2005).
106. R. L. Saunders, J. B. Sprague, Effects of copper-zinc mining pollution on a spawning migration of Atlantic salmon. *Water Res.* **1**, 419–432 (1967).
107. C. A. Mebane, R. J. Eakins, B. G. Fraser, W. J. Adams, Recovery of a mining-damaged stream ecosystem. *Elementa* **3**, 000042 (2015).
108. A. Moore, An electrophysiological study on the effects of pH on olfaction in mature male Atlantic salmon (Salmo salar) parr. *J. Fish Biol.* **45**, 493–502 (1994).
109. A. O. H. C. Leduc, E. Roh, M. C. Harvey, G. E. Brown, Impaired detection of chemical alarm cues by juvenile wild Atlantic salmon (Salmo salar) in a weakly acidic environment. *Can. J. Fish. Aquat. Sci.* **63**, 2356–2363 (2006).
110. J. K. McIntyre, D. H. Baldwin, D. A. Beauchamp, N. L. Scholz, Low-level copper exposures increase visibility and vulnerability of juvenile coho salmon to cutthroat trout predators. *Ecol. Appl.* **22**, 1460–1471 (2012).
111. J. M. Morris, S. F. Brinkman, R. Takeshita, A. K. McFadden, M. W. Carney, J. Lipton, Copper toxicity in Bristol Bay headwaters: Part 2—Olfactory inhibition in low-hardness water. *Environ. Toxicol. Chem.* **38**, 198–209 (2019).
112. K. L. Hogsden, J. S. Harding, Consequences of acid mine drainage for the structure and function of benthic stream communities: A review. *Freshw. Sci.* **31**, 108–120 (2012).
113. J. M. Morris, S. F. Brinkman, M. W. Carney, J. Lipton, Copper toxicity in Bristol Bay headwaters: Part 1—Acute mortality and ambient water quality criteria in low-hardness water. *Environ. Toxicol. Chem.* **38**, 190–197 (2019).
114. C. A. Mebane, F. S. Dillon, D. P. Hennessy, Acute toxicity of cadmium, lead, zinc, and their mixtures to stream-resident fish and invertebrates. *Environ. Toxicol. Chem.* **31**, 1334–1348 (2012).
115. H. W. Lorz, B. P. McPherson, Effects of copper or zinc in fresh water on the adaptation to sea water and ATPase activity, and the effects of copper on migratory disposition of coho salmon (*Oncorhynchus kisutch*). *J. Fish. Res. Board Can.* **33**, 2023–2030 (1976).
116. T. R. Maret, D. E. MacCoy, Fish assemblages and environmental variables associated with hard-rock mining in the Coeur d'Alene River Basin, Idaho. *Trans. Am. Fish. Soc.* **131**, 865–884 (2002).
117. H. Lefcort, J. Vancura, E. L. Lider, 75 years after mining ends stream insect diversity is still affected by heavy metals. *Ecotoxicology* **19**, 1416–1425 (2010).
118. S. J. Hamilton, Review of selenium toxicity in the aquatic food chain. *Sci. Total Environ.* **326**, 1–31 (2004).
119. A. D. Lemly, Symptoms and implications of selenium toxicity in fish: The Belews Lake case example. *Aquat. Toxicol.* **57**, 39–49 (2002).
120. J. A. Hansen, J. Lipton, P. G. Welsh, D. Cacela, B. MacConnell, Reduced growth of rainbow trout (*Oncorhynchus mykiss*) fed a live invertebrate diet pre-exposed to metal-contaminated sediments. *Environ. Toxicol. Chem.* **23**, 1902–1911 (2004).
121. P. N. Duinker, L. A. Greig, The impotence of cumulative effects assessment in Canada: Ailments and ideas for redeployment. *Environ. Manag.* **37**, 153–161 (2006).
122. A. L. Whitehead, H. Kujala, B. A. Wintle, Dealing with cumulative biodiversity impacts in strategic environmental assessment: A new frontier for conservation planning. *Conserv. Lett.* **10**, 195–204 (2017).
123. A. R. Westwood, M. Olszynski, C. H. Fox, A. T. Ford, A. L. Jacob, J. W. Moore, W. J. Palen, The role of science in contemporary Canadian environmental decision making: The example of environmental assessment. *Univ. B. C. Law Rev.* **52**, 243 (2019).
124. R. K. Morgan, Environmental impact assessment: The state of the art. *Impact Assess. Proj. A.* **30**, 5–14 (2012).

125. Government of Canada, "Yukon environmental and socio-economic assessment act" (2003); https://laws-lois.justice.gc.ca/eng/acts/y-2.2/index.html.
126. G. E. Beanlands, P. N. Duinker, *An Ecological Framework for Environmental Impact Assessment in Canada. Institute for Resource and Environmental Studies* (Dalhousie University and Federal Environmental Assessment Review Office, 1983).
127. P. Hackett, J. Liu, B. Noble, Human health, development legacies, and cumulative effects: Environmental assessments of hydroelectric projects in the Nelson River watershed, Canada. *Impact Assess. Proj. A.* **36**, 413–424 (2018).
128. Impact Assessment Agency of Canada, "Basics of environmental assessment under CEAA 2012" (2021); www.canada.ca/en/impact-assessment-agency/services/environmental-assessments/basics-environmental-assessment.html.
129. A. L. Jacob, J. W. Moore, C. H. Fox, E. J. Sunter, D. Gauthier, A. R. Westwood, A. T. Ford, Cross-sectoral input for the potential role of science in Canada's environmental assessment. *FACETS* **3**, 512–529 (2018).
130. L. A. Greig, P. N. Duinker, A proposal for further strengthening science in environmental impact assessment in Canada. *Impact Assess. Proj. A.* **29**, 159–165 (2011).
131. M. Doelle, CEAA 2012: The end of federal EA as we know it? *J. Environ. Law Pract.* **24**, 1–17 (2012).
132. A. J. MacKinnon, P. N. Duinker, T. R. Walker, *The Application of Science in Environmental Impact Assessment* (Routledge, 2018).
133. M. McNutt, Journals unite for reproducibility. *Science* **346**, 679 (2014).
134. M. D. Wilkinson, M. Dumontier, I. J. Aalbersberg, G. Appleton, M. Axton, A. Baak, N. Blomberg, J.-W. Boiten, L. B. da Silva Santos, P. E. Bourne, J. Bouwman, A. J. Brooks, T. Clark, M. Crosas, I. Dillo, O. Dumon, S. Edmunds, C. T. Evelo, R. Finkers, A. Gonzalez-Beltran, A. J. G. Gray, P. Groth, C. Goble, J. S. Grethe, J. Heringa, P. A. C. 't Hoen, R. Hooft, T. Kuhn, R. Kok, J. Kok, S. J. Lusher, M. E. Martone, A. Mons, A. L. Packer, B. Persson, P. Rocca-Serra, M. Roos, R. van Schaik, S.-A. Sansone, E. Schultes, T. Sengstag, T. Slater, G. Strawn, M. A. Swertz, M. Thompson, J. van der Lei, E. van Mulligan, J. Velterop, A. Waagmeester, P. Wittenburg, K. Wolstencroft, J. Zhao, B. Mons, The FAIR guiding principles for scientific data management and stewardship. *Sci. Data* **3**, 160018 (2016).
135. J. Kuipers, A. S. Maest, K. A. MacHardy, G. Lawson, "Comparison of predicted and actual water quality at hardrock mines: The reliability of predictions in environmental impact statements" (Earthworks, 2006).
136. B. Roach, T. R. Walker, Aquatic monitoring programs conducted during environmental impact assessments in Canada: Preliminary assessment before and after weakened environmental regulation. *Environ. Monit. Assess.* **189**, 109 (2017).
137. W. Adams, R. Blust, R. Dwyer, D. Mount, E. Nordheim, P. H. Rodriguez, D. Spry, Bioavailability assessment of metals in freshwater environments: A historical review. *Environ. Toxicol. Chem.* **39**, 48–59 (2020).
138. J. B. Rasmussen, L. Godbout, M. Schallenberg, The humic content of lake water and its relationship to watershed and lake morphometry. *Limnol. Oceanogr.* **34**, 1336–1343 (1989).
139. C. A. Mebane, T. S. Schmidt, J. L. Miller, L. S. Balistrieri, Bioaccumulation and toxicity of cadmium, copper, nickel, and zinc and their mixtures to aquatic insect communities. *Environ. Toxicol. Chem.* **39**, 812–833 (2020).
140. S. A. Nagorski, J. N. Moore, T. E. McKinnon, D. B. Smith, Scale-dependent temporal variations in stream water geochemistry. *Environ. Sci. Technol.* **37**, 859–864 (2003).
141. N. F. Gray, Environmental impact and remediation of acid mine drainage: A management problem. *Environ. Geol.* **30**, 62–71 (1997).
142. I. Park, C. B. Tabelin, S. Jeon, X. Li, K. Seno, M. Ito, N. Hiroyoshi, A review of recent strategies for acid mine drainage prevention and mine tailings recycling. *Chemosphere* **219**, 588–606 (2019).
143. L. S. Balistrieri, C. A. Mebane, Predicting the toxicity of metal mixtures. *Sci. Total Env.* **466**, 788–799 (2014).
144. W. H. Clements, D. B. Herbst, M. I. Hornberger, C. A. Mebane, T. M. Short, Long-term monitoring reveals convergent patterns of recovery from mining contamination across 4 western US watersheds. *Freshw. Sci.* **40**, 407–426 (2021).
145. M. L. Hunter Jr., M. J. Bean, D. B. Lindenmayer, D. S. Wilcove, Thresholds and the mismatch between environmental laws and ecosystems. *Conserv. Biol.* **23**, 1053–1055 (2009).
146. A. J. Sinclair, M. Doelle, P. N. Duinker, Looking up, down, and sideways: Reconceiving cumulative effects assessment as a mindset. *Environ. Impact Assess.* **62**, 183–194 (2017).
147. D. M. Franks, D. Brereton, C. J. Moran, The cumulative dimensions of impact in resource regions. *Resour. Policy.* **38**, 640–647 (2013).
148. G. G. Singh, J. Lerner, M. Mach, C. Clarke Murray, B. Ranieri, G. P. S. Laurent, J. Wong, A. Guimaraes, G. Yunda-Guarin, T. Satterfield, K. M. Chan, Scientific shortcomings in environmental impact statements internationally. *People Nat.* **2**, 369–379 (2020).
149. L. J. Sonter, S. H. Ali, J. E. Watson, Mining and biodiversity: Key issues and research needs in conservation science. *Proc. R. Soc. B.* **285**, 20181926 (2018).
150. I. M. Côté, E. S. Darling, C. J. Brown, Interactions among ecosystem stressors and their importance in conservation. *Proc. R. Soc. B.* **283**, 20152592 (2016).
151. E. K. Sexton, C. J. Sergeant, J. W. Moore, A. R. Westwood, D. M. Chambers, M. V. McPhee, S. A. Nagorski, S. L. O'Neal, J. Weitz, A. Berchtold, M. Capito, C. A. Frissell, J. Hamblen, F. R. Hauer, L. A. Jones, G. Knox, R. Macnair, R. L. Malison, V. Marlatt, J. McIntyre, N. Skuce, D. C. Whited, Canada's mines pose transboundary risks. *Science* **368**, 376–377 (2020).
152. J. W. Moore, C. Carr-Harris, A. S. Gottesfeld, D. MacIntyre, D. Radies, M. Cleveland, C. Barnes, W. Joseph, G. Williams, J. Gordon, B. Shepert, Selling First Nations down the river. *Science* **349**, 596 (2015).
153. Washington Department of Ecology, *Administrative Order Docket #13638 to Buckhorn Mountain Mine* (WADOE, sent 19 July 2016).
154. Alaska Public Media, "As Arctic warming accelerates, permafrost thaw hits Red Dog mine with $20 million bill" (2020); www.alaskapublic.org/2020/09/01/as-arctic-warming-accelerates-permafrost-thaw-hits-red-dog-mine-with-20-million-bill/.
155. Teck, "Reverse osmosis treatment plant start up" (2020); www.alaskapublic.org/wp-content/uploads/2020/08/RO-Anoucnment.pdf.
156. T. D. Pearce, J. D. Ford, J. Prno, F. Duerden, J. Pittman, M. Beaumier, L. Berrang-Ford, B. Smit, Climate change and mining in Canada. *Mitig. Adapt. Strateg. Glob. Chang.* **16**, 347–368 (2011).
157. Environment and Climate Change Canada, "Teck coal limited – Conviction information for 2017-10-05" (2017); https://environmental-protection.canada.ca/offenders-registry/Home/Record?RefNumber=198.
158. J. T. Quigley, D. J. Harper, Compliance with Canada's Fisheries Act: A field audit of habitat compensation projects. *Environ. Manag.* **37**, 336–350 (2006).
159. E. Zambello, L. Olander, E. Glidden-Lyon, E. Meza, J. Wilkinson, "Trends in measuring biodiversity and ecosystem function in mitigation quantification methodologies" (Working Paper NI-WP-19-01, Duke University, 2018).
160. W. Colgan, H. H. Thomsen, M. Citterio, Unique applied glaciology challenges of proglacial mining. *Geol. Surv. Den. Greenl.* **33**, 61–64 (2015).
161. S. D. Odell, A. Bebbington, K. E. Frey, Mining and climate change: A review and framework for analysis. *Extr. Ind. Soc.* **5**, 201–214 (2018).
162. A. R. Sharma, S. J. Déry, Contribution of atmospheric rivers to annual, seasonal, and extreme precipitation across British Columbia and southeastern Alaska. *J. Geophys. Res. Atmos.* **125**, e2019031823 (2020).
163. T. D. Pearce, J. D. Ford, F. Duerden, B. Smit, M. Andrachuk, L. Berrang-Ford, T. Smith, Advancing adaptation planning for climate change in the Inuvialuit Settlement Region (ISR): A review and critique. *Reg. Environ. Chang.* **11**, 1–17 (2011).
164. E. Mavrommatis, D. Damigos, S. Mirasgedis, Towards a comprehensive framework for climate change multi-risk assessment in the mining industry. *Infrastructures* **4**, 38 (2019).
165. H. Ghaffari, J. Huang, K. Jones, J. Gray, R. Hammett, R. W. Parolin, D. Kinakin, M. Lechner, J. G. Parkinson, P. Pelletier, N. Brazier, S. Allard, T. Lipiec, M. Ramirez, "2016 KSM (Kerr-Sulphurets-Mitchell) prefeasibility study update and preliminary economic assessment" (2016); www.miningdataonline.com/reports/KSM_PFS_PEA_112016.pdf.
166. P. J. Griffin, thesis, University of Washington, Seattle, WA (2019).
167. P. N. Duinker, L. A. Greig, Scenario analysis in environmental impact assessment: Improving explorations of the future. *Environ. Impact Assess. Rev.* **27**, 206–219 (2007).
168. M. J. Kennard, S. J. Mackay, B. J. Pusey, J. D. Olden, N. Marsh, Quantifying uncertainty in estimation of hydrologic metrics for ecohydrological studies. *River Res. Appl.* **26**, 137–156 (2010).
169. Alaska Department of Transportation, "Alaska highway drainage manual," Revised June 13, 2006 (2006); http://dot.alaska.gov/stwddes/desbridge/pop_hwydrnman.shtml.
170. C. J. Sergeant, S. A. Nagorski, The implications of monitoring frequency for describing riverine water quality regimes. *River Res. Appl.* **31**, 602–610 (2015).
171. U.S. Army Corps of Engineers, "Army finds Pebble Mine project cannot be permitted as proposed" (2020); www.army.mil/article/238426/army_finds_pebble_mine_project_cannot_be_permitted_as_proposed.
172. A. Wittenberg, J. Marshall, T. Cama, In reversal, "Trump kills huge Alaskan gold mine. Here's why," *Science* (2020).
173. Impact Assessment Agency of Canada, "Decision statement issued under Section 54 of the Canadian Environmental Assessment Act, 2012" (2014); www.ceaa-acee.gc.ca/050/evaluations/document/98458?culture=en-CA.
174. Impact Assessment Agency of Canada, "Decision statement issued under Section 54 of the Canadian Environmental Assessment Act, 2012" (2021); https://iaac-aeic.gc.ca/050/evaluations/document/140985?&culture=en-CA.
175. BC Ministry of Environment and Climate Change Strategy, "Ministers' reasons for decision: Morrison copper-gold project" (2022), p. 4.
176. U.S. Environmental Protection Agency, "Aquatic life ambient water quality criterion for selenium – Freshwater 2016" (EPA Report 822-R-16-006, EPA Publication, 2016).
177. C. J. Sergeant, B. J. Moynahan, W. F. Johnson, Practical advice for implementing long-term ecosystem monitoring. *J. Appl. Ecol.* **49**, 969–973 (2012).
178. B. R. Collison, P. A. Reid, H. Dvorski, M. J. Lopez, A. R. Westwood, N. Skuce, Undermining environmental assessment laws: Post-assessment amendments for mines in British Columbia, Canada, and potential impacts on water resources. *Facets* **7**, 611–638 (2022).
179. D. Gerard, The law and economics of reclamation bonds. *Resour. Policy.* **26**, 189–197 (2000).

180. A. H. Watson, in *Proceedings of the First International Seminar on Mine Closure* (Australian Centre for Geomechanics, 2006), pp. 493–497.
181. L. Cheng, J. G. Skousen, Comparison of international mine reclamation bonding systems with recommendations for China. *Int. J. Coal Sci. Technol.* **4**, 67–79 (2017).
182. E. S. Norman, *Governing Transboundary Waters: Canada, The United States, and Indigenous Communities* (Routledge, 2015).
183. M. A. Giordano, Managing the quality of transboundary rivers: International principles and basin-level practice. *Nat. Resour. J.* **43**, 111–136 (2003).
184. N. D. Hall, Transboundary pollution: Harmonizing international and domestic law. *U. Mich. JL Reform.* **40**, 681–746 (2007).
185. N. D. Hall, The centennial of the Boundary Waters Treaty: A century of United States-Canadian transboundary water management. *Wayne L. Rev.* **54**, 1417–1449 (2008).
186. International Joint Commission, "Impacts of a proposed coal mine in the Flathead River Basin" (International Joint Commission Digital Archive, 1988); https://scholar.uwindsor.ca/ijcarchive/369.
187. F. R. Hauer, C. C. Muhlfeld, Compelling science saves a river valley. *Science* **327**, 1576–1576 (2010).
188. BC Chief Inspector of Mines, "Chief Inspector of Mines 2018 annual report" (2019); www2.gov.bc.ca/assets/gov/farming-natural-resources-and-industry/mineral-exploration-mining/documents/health-and-safety/ci-annual-reports/2018_ci_annual_rpt.pdf.
189. State of Alaska, Department of Natural Resources, Division of Mining, Land and Water (2014); https://dnr.alaska.gov/mlw/mining/large-mines/fort-knox/pdf/fgmibond2014.pdf.
190. State of Alaska, Department of Natural Resources, Division of Mining, Land and Water (2021); https://dnr.alaska.gov/mlw/mining/large-mines/red-dog/pdf/rdfa2021.pdf.
191. Government of Yukon, Energy, Mines and Resources (2019); https://emr-ftp.gov.yk.ca/emrweb/COMM/major-mines/eagle-gold/mml_eg_financial_security_determination_summary_table.pdf.
192. Government of Yukon, Energy, Mines and Resources (2015); https://emr-ftp.gov.yk.ca/emrweb/COMM/major-mines/minto/mml_minto_determination_of_financial_security_jan2015.pdf.
193. State of Alaska, Department of Natural Resources, Division of Mining, Land and Water (2022); https://dnr.alaska.gov/mlw/mining/large-mines/pogo/.
194. State of Alaska, Department of Natural Resources, Division of Mining, Land and Water (2020); https://dnr.alaska.gov/mlw/mining/large-mines/greens-creek/pdf/2020-7-21-MOU-Greens-Creek-Mine-Final.pdf.
195. State of Alaska, Department of Natural Resources, Division of Mining, Land and Water (2021); https://dnr.alaska.gov/mlw/mining/large-mines/kensington/pdf/511336-Kensington-Reclamation-Bond-Swap.pdf.
196. Americas Silver Corporation, "Technical report on the Galena complex" (Report NI 43-101, Shoshone County, Idaho, USA, 2016); www.sec.gov/Archives/edgar/data/1286973/000106299317000127/exhibit99-73.htm.
197. Hecla Mining Company, "Form 10-K, annual report pursuant to Section 13 or 15(d) of the Securities Exchange Act of 1934" (2019); www.sec.gov/Archives/edgar/data/719413/000143774919003108/hl20181231_10k.htm.
198. Government of Yukon, Energy, Mines and Resources (2019); https://emr-ftp.gov.yk.ca/emrweb/COMM/major-mines/keno-hill/mml-keno-amendment-berminghamcover_letter-nov2019.pdf.
199. New Jersey Mining Company, "Form 10-K, Annual report pursuant to Section 13 or 15(d) of the Securities Exchange Act of 1934" (2015); www.sec.gov/Archives/edgar/data/1030192/000105291815000115/njmc10kmar3115final.htm.
200. J. E. Athey, M. B. Werdon, "Alaska's mineral industry 2018: Alaska Division of Geological & Geophysical Surveys Special Report 74" (2019); https://doi.org/10.14509/30227.
201. Usibelli Coal Mine, "Mining equipment" (2022); www.usibelli.com/mining/mining-equipment.
202. Alexco, "Keno Hill Silver District project page" (2022); www.alexcoresource.com/operations/overview/.
203. Idaho Strategic Resources, "Golden Chest Mine project page" (2022); https://idahostrategic.com/production/.

Acknowledgments: This review is based on an October 2019 workshop held at the University of Montana Flathead Lake Biological Station (more information at https://flbs.umt.edu/newflbs/research/working-groups/mining-and-watersheds/). We thank E. O'Neill and other participants for valuable contributions. A. Beaudreau, M. LaCroix, P. McGrath, K. Schofield, and L. Brown provided helpful reviews of earlier drafts. Three anonymous reviewers provided thoughtful critiques that greatly improved the manuscript. The views expressed in this article are those of the authors and do not necessarily represent the views or policies of the U.S. Environmental Protection Agency. Our analysis comes from a western science perspective and hence does not incorporate Indigenous knowledge systems. We acknowledge this gap and highlight that the lands and waters we explore in this review have been stewarded by Indigenous Peoples for millennia and continue to be so. **Funding:** The workshop was cooperatively funded by the Wilburforce Foundation and The Salmon Science Network funded by the Gordon and Betty Moore Foundation. **Author contributions:** C.J.S. led the review process, writing, and editing. C.J.S. and E.K.S. co-organized the workshop. E.K.S. and J.W.M. extensively contributed to all aspects of the review conceptualization, writing, and editing. A.R.W., S.A.N., J.L.E., D.M.C., S.L.O., R.L.M., F.R.H., D.C.W., and J.W. significantly contributed to portions of the review conceptualization, writing, and editing. J.C., M.Ca., M.Co., C.A.F., G.K., E.D.L., R.M., V.M., J.K.M., M.V.M., and N.S. provided writing and editing and are listed alphabetically. **Competing interests:** The authors declare that they have no competing interests. **Data and materials availability:** All data needed to evaluate the conclusions in the paper are present in the paper and/or the Supplementary Materials.

Submitted 30 November 2021
Accepted 11 May 2022
Published 1 July 2022
10.1126/sciadv.abn0929