# Forest and terrestrial ecosystem impacts of mining

Lee E. Frelich, Director, The University of Minnesota Center for Forest Ecology

September 22, 2014

## Introduction

Boreal forests are one of the world's most important purveyors of ecosystem services including carbon storage and clean water (Schindler and Lee 2010). Boreal forests have a large impact on climate at local, regional and global scales. They also harbor globally significant wildlife populations. Forests of Minnesota's Boundary Waters Canoe Area Wilderness (BWCAW) and the surrounding Quetico-Superior Ecosystem including Superior National Forest, Voyageurs National Park, and Quetico Provincial Park (Ontario) are in the southern boreal (or near boreal) forest zone of central North America (Heinselman 1996).

Mining has been identified as one of the major environmental impacts on the boreal forest worldwide, due to widespread presence of ancient rock formations that contain metallic ores (Frelich 2013). Although most of the studies done to date have concentrated on aquatic impacts, there is a modest but still significant body of knowledge of mining impacts on terrestrial ecosystems. Well-known cases of heavy metal pollution of forests via aerial deposition from smelting have occurred in North America and Europe (Freedman and Hutchinson 1980, Salemaa et al. 2001), and although smelting is not going to occur in northern Minnesota, these historic episodes of smelting provide basic information about ecosystem response to heavy metals. Studies of impacts of mining on various measures of environmental quality and reclamation, for cases with acid mine drainage, open-pit and underground iron mining, have been published for northern forests (Rayfield et al. 2005, Northwatch and Mining Watch Canada 2008, DeLong et al. 2012, Anawar et al. 2013). In addition, there is a much larger body of ecological knowledge that can be synthesized to develop issues related to mining that have previously received little attention.

The ecological footprint of mining activity extends well beyond the area directly impacted. We can divide the footprint of mining into primary and secondary areas. The primary footprint is the area directly impacted by the mine, processing/rock crushing facilities, tailings areas, buildings, roads, parking lots, and energy transmission network built to accommodate the mine and workers. The secondary footprint comprises adjacent areas affected through mining activities and changes in the landscape that can propagate ecological changes for various distances; this includes such items as fragmentation, changes in forest type within the primary footprint, changes in wildlife migration and habitat use patterns, noise, light, windblown dust, dispersal of invasive species established on the mine site, and watershed areas affected by water withdrawals and mine drainage. The effects of the secondary footprint gradually decline with distance from a mine, and the various types of impacts should always be defined in terms of ecological impacts judged to be significant and the distance and spatial pattern within which those effects are estimated to occur. Distances and spatial pattern will vary by type of impact, and the spatial pattern could be directed by flow of water and air, animal movements, and seed dispersal away from the mine site.

The purpose of this document is to point out potential impacts to terrestrial ecosystems that could occur due to copper-nickel exploration and mining in the primary and secondary footprints in the Quetico-Superior Ecosystem, and in particular, impacts likely to affect the BWCAW. Magnitude of these impacts is unknown and cannot be evaluated here, although all impacts identified are certain to occur to some extent. In addition, little is said here about mitigation and reclamation. Magnitudes of impacts and mitigation issues should be addressed in an EIS.

## Potential Terrestrial Impacts in the Quetico-Superior Ecosystem

### Forest and other vegetation acreage and composition

Mining will directly displace forests and potentially change the composition of any forest remnants within the primary footprint, and some effects will extend into the secondary footprint. Magnitude of the impacts will depend on number of acres of timberland (i.e. forests on sites productive enough to have the capability of producing commercial timber) by forest type removed or fragmented in such a way that it would no longer be suitable for harvest (e.g. Minnesota Generic Environmental Impact Statement on Timber Harvesting and Forest Management in Minnesota (Jaakko Pöyry Consulting, Inc. 1994)).

Mining could also contribute to change in forest type. For example, fragmentation and disturbance could favor early successional species such as aspen and generalist species such as red maple. This will interact with climate change, which will occur at the same time as mining and is also predicted to change forest types and/or reduce forest acreage, since fragmentation effects could contribute to a warmer local environment within forests of the primary and portions of the secondary mining footprints. Therefore, analyses of the current and future area and composition of forest cover for various mining scenarios will be basic information to help with analyses of other impacts, for example, impacts on biodiversity and wildlife (e.g. Jaakko Pöyry Consulting, Inc. 1992a, 1992b).

Vegetation types other than productive forests would also have similar concerns, including rock outcrops, dry shrub lands, and especially wetlands including shrub carrs (shrub-dominated wetlands of willow, alder and dogwood), sedge meadows, bogs, and marshes. The Field Guide to Native Plant Communities of Minnesota (MNDNR 2003), Northern Superior Uplands Section, can be used as a guide for vegetation types relevant to the mining area.

### Old growth forest, old forest, and primary forest

Old growth and primary forests represent irreplaceable baselines of ecosystem function and species diversity, with a number of ecological, genetic, cultural, and spiritual values (Frelich and Reich 2003). For example, red pine forests only occupy about 1% of their former acreage

(Anand et al. 2013). In the Great Lakes Region, only a low percentage of the original unlogged forest remains, which elevates the importance of unlogged stands both inside and outside the BWCA wilderness (Frelich 1995). There are many definitions of old growth; Minnesota Department of Natural Resources considers stands at least 120 years old to be old growth and in northern Minnesota, this generally also means that the stand has never been logged, since European settlement occurred relatively late compared to the rest of the state (Frelich 1995). Old forests (80-120 years old), which represent future old growth, are also important to consider, due to the dynamic nature of boreal forests. Primary forests are those that have never been logged, and may be any age, but have a continuous legacy of natural disturbance (fire, wind, native insect infestation) rather than harvesting (Frelich and Reich 2003). Such forests of fire-dependent species like jack pine are common in the Ely area. Some old-growth, old and primary forest remnants outside the BWCAW have been or will be impacted by exploration and eventually by mining. The existence of any such forests outside the BWCAW contributes to the ecological integrity of forests inside the BWCAW, since they reduce many types of edge effects that occur along the wilderness boundary.

**Fragmentation**

Fragmentation affects forests through increased edge to interior ratio, leading to increased light and temperatures in a buffer zone that extends into the forest from artificial edges (Fischer and Lindenmayer 2007). Edge adapted species, especially weedy species of native and non-native plants and animals, respond to the change in environment at the expense of forest interior species, and these impacts can extend well into otherwise undisturbed forests.

Fragmentation would be brought by roads and transmission lines, as well as the mines, tailings areas, pipelines to tailings areas, buildings and parking lots built to support the mines. Additional fragmentation effects would include increased residential and commercial development and associated traffic. While there are no inventoried roadless areas in the proposed exploration and mining zones, and logging is common, logging roads tend to be temporary while many of the roads built for mining will exist for a several decades, be wider, and have larger edge effects for temperature and sunlight, and greater potential for spread of invasive species and native edge species. Because of lag time effects in ecosystems, fragmentation effects are likely to persist for a few decades after mines have closed, so that area sensitive species of plants and animals will continue to decline for a long time—during mining itself and a few decades after.

There is a strong interaction between fragmentation and invasive species (Hawbaker and Radeloff 2004). A large zone of fragmentation many miles in width could lead to a large invasion front of invasive (and native edge-adapted) species at the edge of the wilderness (see discussion of 'mass effect' under '**Invasive species**' below). Definition for a zone of vigilance and no tolerance of invasive species around the mining/BWCAW interface is needed, even for exploration and hydrogeological study activities.

Fragmentation of wetlands and forests will likely lead to local warming in addition to warming associated with global warming. This is basically due to creation of non-vegetated surfaces that absorb sunlight and do not have the cooling effects caused by evapotranspiration that occurs inside a forest. Large industrial complexes with many roads, buildings, and other interruptions in the local vegetation can therefore create a local zone with warmer temperatures than the surrounding landscape, similar to an urban heat island. Decay and release of stored carbon to the atmosphere and effects on species composition are likely to occur in areas directly disturbed as well as some distance into undisturbed forests.

Road salt from increased traffic will have impacts on surface and soil water, wetlands, and forest (Kaushal et al 2005, Jull 2009), extending along drainage ways that are crossed by roads, and into all forests along all stretches of roads. Two pathways for road salt damage occur: (1) accumulation in snow windrows along the sides of roads and parking lots that melts in spring, briefly saturating roots with salt; and (2) aerial salt particles that are deposited on twigs and kill buds and foliage of trees. Large trucks will require bare pavement and large application of salt, also will loft salt higher into the air than regular automobile traffic, and thus dispersal distances into the forests and waterways will be relatively large. Also, the road edge to forest area ratio (e.g. number of miles of heavily used roads per square mile of forest) for roads that generate a lot of salt could potentially increase over a large primary footprint of mining. Soils in the area of the proposed mines are highly susceptible to salt damage because they are shallow, so that root contact with salty water flowing along the bedrock-soil interface, root damage, and tree death, is a likely occurrence. Several tree species in the area are susceptible to salt damage, principally white pine, basswood, red oak, bur oak and red maple, while fir, spruce and cedar are relatively resistant to salt damage. White pine damage and mortality from salt has been noted in the Ely area (Frelich personal observation).

**Wildlife**

Wildlife species living within the primary mining footprint would be directly displaced, due to loss of forest and other vegetation. For example, loss of trees will directly lead to less nesting habitat for birds (Van Wilgenburg et al. 2013). A number of wildlife species respond to changes in road density, especially permanent roads that would be created in the primary footprint of the proposed mines. Road kill due to increased traffic and more artificial edges that wildlife species will have to cross would affect some wildlife populations. Noise and light pollution can impact songbirds at variable distances from road networks and other human activities (Kociolek et al 2011). Amphibians such as salamanders and frogs that live in forests, especially swamp forests, and wetlands are sensitive to pH and could be affected by acid dust from rock crushing areas as well as movement of polluted soil and surface water.

Certain species require large tracts of unfragmented forest (Schmiegelow et al. 1997). Pine marten, fisher, and area sensitive (interior forest) bird species, including many warbler species, are likely to experience direct displacement of habitat within the primary mining footprint, with impacts possibly extending some distance into the secondary footprint. Edge species such as deer, raccoons, cowbirds, and invasive species (plant and animal) will increase in abundance, compete with area sensitive species, and could even work to change habitat value of forest remnants within the primary footprint and possibly extending into the secondary footprint.

Many landscapes in the area potentially affected by mining have a characteristic pattern of dry upland forest, rock outcrop, wetland forest, wetlands and aquatic habitats. Many wildlife species move among these habitats on daily and seasonal time scales, and this mosaic of habitat types is partly responsible for the diversity of mammal, amphibian and bird species present. This pattern could be disrupted in the primary footprint of mining activity, so that impacts on habitat use spread across a larger landscape, due to increased travel distance for wildlife to use different habitat types.

High deer populations responding to fragmentation and other changes in the environment have a potential negative impact on moose because deer carry the deer brainworm, which does not kill deer, but which can kill moose and be a significant negative factor for moose populations when deer densities are high (Lankester 2010).

Roads and associated changes in human activity are a negative factor for wolves (Mech et al 1988). If wolves are displaced from the primary mining footprint, then further concentration of deer in addition to that caused by fragmentation could occur, due to predator avoidance by deer. This would result in a trophic cascade that would change the rest of the ecosystem including the plant community (Frelich et al. 2012). Such a cascade with a low wolf, high deer, low plant diversity regime could lead to loss of plant species that are sensitive to deer browsing, such as white cedar, yew, yellow birch, trillium and certain orchids (Callan et al. 2013).

Wildlife corridors that cross the primary footprint of mining could be disrupted. For example, lynx travel tens of miles during the year (Burdett et al. 2007), would be subject to road kill by increased traffic, and do not prefer to den in areas with the high road densities (Bayne et al. 2008) that would be created in the primary footprint of the proposed mines. Many examples of effects on wildlife species that regularly cross the wilderness boundary and travel through the primary footprint of mining will occur. Finally, there may be stop overs within the primary footprint for migrating species that eventually stay within the BWCAW or beyond.

## Rare species

Rare species lists need to be compiled for all taxa (plant and animal) for the primary and secondary footprints, including rare species that do not live in the area but migrate across the footprints. This should include state and federal species of concern, threatened and endangered species, as well as Forest Service sensitive species. To determine the impact, a survey of the potential mining area within which it is likely that rare species will be directly displaced, should be carried out. There is a high likelihood that some rare species populations will be locally extirpated, which may have an effect on the overall viability of a given species at the ecoregion, state or federal level. Many rare species have scattered, somewhat isolated populations, which nevertheless form an interdependent network (metapopulation, Moilanen et al. 1998) that allows exchange of individuals and genetic material and maintenance of the individual populations.

Disruption of populations outside a wilderness areas can have negative consequences for populations inside a wilderness area, since the regional population structure is established in response to natural factors without regard to artificial wilderness boundaries.

Soil dwelling species, including insects, worms, bacteria and fungi that have not yet been discovered probably exist in the mining area. For example, Schlaghamersky et al. (2014), found possibly as many as nine new species of native worms (Enchytraieds) in two days of field work in northern Minnesota and Wisconsin. Therefore, there is a significant chance of losing native biodiversity within the primary footprint, but possibly also lesser effects in the secondary footprint of mining. These soil dwelling organisms, while not charismatic to the public, are important for running ecosystem processes such as nutrient and water cycles. They are also the taxonomic groups most likely to yield new drugs or chemicals that could be used in industrial processes.

**Invasive species**

Numerous points of introduction will occur due to exploration and mining at the landscape scale, since human disturbances such as clearing vegetation and road building are well known to introduce invasive species (Hansen and Clevenger 2005, Cameron et al. 2007). Even with the procedures put in place by Superior National Forest for preventing invasive species (such as cleaning equipment when moving between sites and removing visible individuals that appear near work sites), they are likely to invade in the fragmentation edge zones that are beyond the primary footprint of mining activities. This is because many invasive species have long-distance dispersal abilities and can take advantage of changes in the environment (amount of light, temperature, soil disturbance) that may occur hundreds of feet into the forest along roads and other openings created. There can be a large 'mass effect' for invasive species, whereby the larger the area fragmented by roads, transmission lines, buildings and parking lots, the more locations where invasive species can become established. Larger masses of invasive species within human-dominated areas lead to larger numbers of seeds or other propagules spreading into adjacent natural areas, and because establishment is a chance process, the likelihood of establishment within secondary footprints is increased in proportion (or possibly exponential proportion) to the size of the population. Such mass effects can exceed critical mass for establishment of invasive species and their spread into the secondary footprint. Current guidelines for invasive species management on the Superior National Forest are not adequate to address potential invasive species problems at the scale of the proposed mines.

Canada thistle and other exotic thistles, hawkweed, buckthorn, purple loosestrife, leafy spurge, garlic mustard, spotted knapweed, exotic honeysuckles, hybrid and narrow leaf cattail, and reed canary grass will likely be among the invasive plant species. Several of these are already reported within the Superior National Forest and BWCAW (Superior National Forest 2014), and minimizing seed sources around the edge of the wilderness is key to maintaining the pristine condition of plant communities within the wilderness. In addition, new invasive species not currently being watched will likely appear. Therefore, analyses of mining impacts should look at exotic species that have traits of invasive species that are locally present and likely to become invasive, or species that have become invasive elsewhere that are present in the vicinity of the mine, but have not become invasive in Minnesota. There are potential interactions with climate change; i.e. some species that are present and limited in extent and success under the current climate could 'take off' in a warmer climate and invade the wilderness from the mining sites.

Numerous underground invaders exist, including earthworms—the invasive species group with the largest known impacts on ecosystems in terms of types, magnitudes and spatial extents of impacts in terrestrial ecosystems worldwide. In northern temperate and southern boreal forests, earthworms consume the forest floor (litter or duff layer), making soils warmer, drier and more nutrient poor, lead to lower plant diversity, and reduced tree growth (Frelich et al. 2006, Larson et al. 2010). Thus, soil movement during mining becomes an issue since it is easily possible in the Ely area (which currently has a spotty distribution of soil invaders) to bring in new earthworm species as well as spread existing species via soil movement. In addition, earthworms can alter post-mining succession of plant species (Mudrak et al. 2012), so that novel successional sequences not familiar to ecologists in the Quetico-Superior Ecosystem may occur due to earthworm presence. Soil movement within the primary footprint could easily lead to spread of underground invasive species into the secondary footprint within the wilderness.

## Soil disruption and recovery time

Soils that took thousands of years to develop will not recover quickly from mining. Interruption in continuity of soils during mining occurs in ways that do not happen after fires or logging. Reclamation procedures, while better than nothing, restore minimal soil function, and do not restore the same function as before mining within a human time frame. It is possible to develop an index of soil quality for reclaimed sites, although unique aspects of each region need to be taken into account, and any such index can create a standard by which to judge any treatments that may be developed (Asensio et al 2013). Simple communities such as shortgrass prairies recover faster than complex communities such as forests (Frouz et al 2013).

Windblown pyrite dust or other metallic dust can remain a problem for several decades after mining ceases on Cu-Ni mine tailings (Bagatto and Shorthouse 1999) and can also cause acidification of soils outside the mine site (Bussinow et al. 2012). Cu and Ni deposition can occur for distances of 2-3 miles around contemporary mining sites, such as the Eagle Mine in Upper Michigan (Conestoga-Rovers 2007). Once in the soil system, heavy metals are recycled by plants through litterfall, litter decomposition and root uptake (Johnson et al. 2003). They are also subject to volatilization and redeposition during forest fires, for example the effects of forest fires on Mercury (Hg) in the forest floor that originated from fossil fuel burning has been studied in northern Minnesota (Witt et al. 2009, Woodruff and Cannon 2010). Acidification of soils can mobilize Aluminum and the resulting cascade of chemical changes can lead to Calcium deficiencies that affect tree and animal growth, especially on shallow soils with poor nutrient supply. The ability of the ecosystem to fix nitrogen can also be reduced (Lorenc-Plucinska et al

2013). Mycorrhizae—fungi essential to tree growth—would also be impacted by any changes in soil chemistry within the primary and secondary footprints of the mine.

Microtopographical complexity includes tip-up mounds and pits, rotten wood at various stages of decay, and variable build-up of humus, and is essential for maintenance of biodiversity of tree seedlings, herbaceous plants, fungi, mosses and microorganisms that in turn are needed for ecosystem function. Microtopography can take centuries to form, or to rebuild on reclaimed soils. Some microtopographical features can persist for 6,000 years (Samonil et al. 2013). Microtopography can be enhanced artificially, speeding up the recovery process after mining, such as by adding woody debris (Brown and Naeth 2014).

Surprisingly, the problem with soil reclamation for the existing iron mines in northern Minnesota is high pH (8.0-8.1), rather than acid soils (Norland and Veith 1995, Felleson 1999). However, the situation should be different, with low pH around 4 or less for copper-nickel reclamation (Bagatto and Shorthouse 1999). Although reclamation is only an issue for the primary footprint, reclamation there would determine what future types of vegetation and wildlife habitat would occur just outside the BWCAW, and therefore would impact the wilderness through many types of spatial ecological processes described elsewhere in this report. These processes include seed rain, chemical processes in soil and water that depend on species-dependent leaf litter chemistry, and use of habitats by wildlife species that move across the landscape and use habitats within and outside of the BWCAW.

## Terrestrial-aquatic linkages

There are strong interactions between wetland forests and streams and groundwater, as well as terrestrial forest systems. Biogeochemical processing of leaf litter and other ecosystem function in headwater streams can be negatively affected by mining (Berkowitz et al. 2014). The Quetico-Superior Ecosystem as a whole has an exceptionally large magnitude of forest-aquatic linkages compared to many of the world's ecosystems. Streams (intermittent and permanent), wetlands of spatial extents from 1/100$^{th}$ acre to thousands of acres, and lakeshores produce literally tens of miles of aquatic-terrestrial interfaces per square mile of landscape. The entire biosphere over most of the Quetico-Superior ecosystem is like a veneer of soil and all of the organisms within, lying on top of bedrock, with water flowing through an intricate web of tree roots and associated mycorrhizae. Repeating patterns of dry, mesic and wet forests occur across the landscapes of the ecosystem, so that the entire ecosystem consists of dry to wet ecological gradients (Heinselman 1996).

The wetland vegetation types exist along a gradient of water chemistry, especially pH, and oxygen content (MNDNR 2003). The high pH, high nutrient and high oxygen group includes ash swamps, cedar swamps, sedge meadows, and shrub swamps (alder, dogwood, willow), while tamarack swamps tend to be intermediate and black spruce swamps and open sphagnum/cranberry bogs occur in low pH, low nutrient and oxygen poor conditions. Changes to water flow and/or chemistry caused by mining could upset the balance among these vegetation types, which are interlinked and intergrade with each other along natural chemical gradients across the landscape. This could lead to changes in decomposition and nutrient cycling, as well as tree mortality. These effects could extend well into the secondary footprint of mining, and if they occurred, would last for centuries.

Large areas of disruption of water flow, such as installation of roads, pipelines, and transmission lines; draw down of water for mine use during periods of low flow; as well as cumulative impacts of hundreds of small disruptions of ground water flow and quality across the landscape could have significant additive and cumulative effects. Changes in water table—both up and down—caused by disruption of water flow when roads are installed can lead to death of swamp forests. Use of water by the mine during times of low flow, as well as runoff from hard surfaces in the primary footprint during periods of high flow, could exaggerate high and low water effects. These effects propagate variable distances away from the primary footprint, depending on the configuration of a given watershed, and last for decades to centuries.

The potential for the tailings location(s) or associated pipeline to fail in an extreme weather event, such as the heavy rainfall in and around Duluth in June of 2012, could also lead to large impacts that would propagate across the landscape, and this potential could last throughout the period of mining and many years beyond.

Accelerated ecosystem aging (acidification, Ford 1990) of wetland forests is possible if any acid drainage occurs, or from wind-blown deposition. Acidification due to production of organic acids in conifer leaf litter and mosses on the forest floor occurs progressively over several thousand years under natural conditions, converting lakes and relatively nutrient rich wetlands (cedar swamps, shrub swamps) to acid bogs, which then fill with floating mats of Sphagnum moss. Note that fill in of lakes can occur from either high nutrient conditions and eutrophication, or acidification, since the acid-loving *Sphagnum* moss species can form floating mats that fill increasingly acidic waters. Many of the watersheds have been and will continue to be (under natural conditions), buffered by natural chemical processes against acidification for the next several millennia. However, a pulse of acidic material in windblown dust or water flow could upset the natural buffering system with large consequences for swamp forests, wetlands, and lakes. Acidification that would have taken thousands of years to occur naturally could occur within a few years.

Beaver, moose, bear, insects and other species can move any toxic element (heavy metals) that may be present across aquatic and terrestrial ecosystem boundaries, by moving vegetation, eating vegetation or fish and depositing the elements elsewhere (Sarica et al. 2004, Mogren et al. 2013). Also, disturbances such logging, wind or fire may influence movement of toxic elements across terrestrial and aquatic ecosystems in complex ways (Gabriel et al. 2012, Mitchell et al. 2012).

## Cumulative impacts

Synergy among mining impacts, invasive species and climate change could exceed the current resilience of the forest, reduce the current level of resilience, and affect resilience of the forest well into the secondary footprint, and inside the wilderness. Resilience is defined as the ability of an ecosystem to recover to its initial state after being disturbed. For example, forests of the BWCAW are resilient to fire, at the stand and landscape scales. Stands of trees killed are immediately reseeded with native species after fire, and at the landscape scale, fires create a mosaic of forests in different stages of succession that is ideal for maintaining wildlife (Heinselman 1996). In contrast, human activity can speed up forest change to a level too rapid for recovery, or the intensity of human disturbance may be higher than natural disturbances, and may overwhelm the forest's ability to recover. The many types of changes brought about by mining combined with climate change are likely to reduce resilience. Furthermore, the combination of individual impacts lasting for several decades could potentially create a whole new trajectory in future development of the forest for the primary and secondary mining footprints, which would differ from the trajectory without mining to a greater degree as time goes on. Generalist tree species like red maple will be able to take advantage of the changes, and create new alternate forest states. Loss of moose could be locally accelerated, and a part of the moose core habitat in Minnesota displaced.

Cumulative effects of many small changes in water flow and quality across the landscape could lead to large changes over time. Increased evaporation from a warmer climate combined with changes in water flow due to mining in some areas will lead to change or replacement of wetland forests. Changes in flow of water including change in the variability of flow, caused by the large primary footprint of the mines, combined with increased deer grazing, will inevitably kill some wetland forests and cause conversion to other vegetation types, such as shrub carr, reed canary grass, and hybrid cattail in places not acidified, and increased ecosystem aging in any areas with low natural pH that may receive additional acidification.

The extended or secondary footprint of mining activities will have some impact within the wilderness, due to air and water flow issues, movement of invasive species and fragmentation, all of which can have effects that propagate beyond the primary footprint of mining.

Invasive species propagule pressure from numerous test sites and roads outside the wilderness could have significant impacts. It is inevitable that a rising edge to interior ratio just outside the wilderness will change the composition of the forest there, and that will propagate into the wilderness. Such spatial cascades could also affect aquatic terrestrial linkages into the wilderness. Size of the secondary footprint within the wilderness for various environmental effects remains to be estimated, as well as how severely the nearby portion of the wilderness would be affected, what proportion of the wilderness will have some impacts, and what absolute mileage and proportion of the wilderness boundary would be affected.

## Summary of BWCAW Wilderness impacts

There are many reasons to protect wilderness integrity. However, the science-related reasons are seldom discussed. The BWCAW is internationally known for ground-breaking research on forest fires, landscape patterns, biodiversity, wildlife, soils, nutrient cycles and other ecosystem processes, lakes, climate change, and recreational use of wilderness. Furthermore, the BWCAW provides the baseline for the rest of the landscape which is manipulated by logging, mining, roads, housing and other human activities. Having a baseline is the most basic principle of science; even in the context of global climate change, the BWCAW shows us how the forests, lakes and landscape patterns would develop in the absence of direct human manipulation. This role of wilderness and other natural areas as a scientific baseline has become critical in the last few decades, to assess the overall impacts of human activity at local, regional and global scales. Without these baselines, we are essentially ‘flying blind’ in our ability to manage ecosystems to provide the many types of services needed by humanity.

The BWCAW is a pristine area where zero impacts are expected, and therefore any potential impacts are a concern. Of the individual and cumulative impacts listed in the table below, and given the current knowledge of interconnections in the field of landscape ecology, it is legitimate to state that most (25 of 39, all those with a ‘2’ in the second column of the summary table) will impact the wilderness to some degree. The living portion of the BWCAW ecosystem is like a thin membrane with many fine-scale interconnections among paths of water flow, lying on top of undulating bedrock. A large primary footprint of mining activity at the top of the watershed can cause many effects related to water flow and chemistry (including aerial deposition), that will affect everything lower in the watershed. Given the linkages between aquatic and terrestrial components of the ecosystem in the BWCAW, these effects will also extend into terrestrial vegetation. Changes in forest type, soils, and fragmentation, within the terrestrial primary footprint will also impact invasive species, and all of these will affect vegetation, wildlife, and rare species of plants and animals within the BWCAW.

**Summary table of mining impacts on forest and terrestrial ecosystems.** Footprint column shows whether a given impact would be in the primary or secondary footprint, or both. Note that all impacts with a '2' can occur within the BWCAW wilderness. The last two columns show whether a given impact would occur from exploration and mining.

| **Impact** | **Footprint** | **Exploration** | **Mining** |
|---|---|---|---|
| Baseline vegetation impacts | | | |
| Loss of forest acreage by type | 1 | | X |
| Forest composition change by forest type | 1,2 | X | X |
| Loss of non-forest vegetation by type | 1 | | X |
| Non-forest vegetation change by veg type | 1,2 | X | X |
| Loss of old-growth forest remnants, acres by forest type | 1 | X | X |
| Loss of old forest (80-120 years), acres by forest type | 1 | X | X |
| Loss of primary forest remnants, acres by forest type | 1 | X | X |
| Fragmentation | | | |
| Edge to area ratio due to roads, transmission lines, parking, tailings, buildings, residential and commercial development | 1,2 | X | X |
| Environment effects in remaining forest within primary footprint | 1 | | X |
| Changes in native edge versus interior plant and tree species | 1,2 | X | X |
| Road salt effects on trees and water | 1,2 | | X |
| Water flow effects on vegetation | 1,2 | X | X |
| Wildlife, all impacts are per species for the relevant species group, a number of fragmentation effects are also listed here as 'frag effect'. | | | |
| Area sensitive mammals, marten, fisher (frag effect) | 1 | X | X |
| Area sensitive birds, warblers, etc. (frag effect) | 1 | X | X |
| Loss of nesting habitat by forest type and bird species | 1 | X | X |
| Loss of habitat acres by wildlife species and vegetation/forest type | 1 | | X |
| Effects on species sensitive to aquatic and aerial chemistry (amphibians) | 1,2 | X | X |
| Effects on wolves and trophic cascade (frag effect) | 1,2 | | X |
| Effects on deer and deer-moose relationships (frag effect) | 1,2 | | X |
| Road kill effects (frag effect) | 1,2 | | X |
| Road salt effects (frag effect) | 1,2 | | X |
| Corridor disruption for mobile but non-flying species | 1,2 | | X |
| Loss of critical stopovers for migrating species | 1,2 | | X |

| | | | |
|---|---|---|---|
| Disruption of landscape pattern of vegetation/habitat | 1 | | X |
| Noise, light and vibration effects | 1,2 | X | X |
| Rare species | | | |
| Direct habitat loss per species | 1 | X | X |
| Impacts on local populations and regional stability per species | 1,2 | X | X |
| Invasive species | | | |
| Transport by equipment and soil movement per species | 1 | X | X |
| Response to fragmentation per species | 1,2 | X | X |
| Soils and productivity | | | |
| Acidification by water and air movement | 1,2 | | X |
| Movement and effects of heavy metals in the soil | 1,2 | | X |
| Loss of complexity | 1 | X | X |
| Terrestrial-aquatic linkages | | | |
| Accelerated ecosystem ageing | 1,2 | | X |
| Water chemistry effects on landscape arrangement of marshes, sedge meadows, peatlands, bogs, shrub cars and wetland forests | 1,2 | | X |
| Changes in water flow effects on landscape arrangement of wetland vegetation types | 1,2 | | X |
| Heavy metal movement across aquatic-terrestrial boundaries | 1,2 | | X |
| Cumulative impacts | | | |
| Spatial cascade of fragmentation effects including deer, moose, forest type, invasive species interactions | 2 | | X |
| Sensitivity of future trajectory of forest and wildlife impacts to number of exploration sites and total size of primary footprint | 1,2 | X | X |
| Synergy among climate change, invasive species and mining impacts | 1,2 | X | X |

## Literature cited


Anand, M., M. Leithead, L.C.R. Silva, C. Wagner, M.W. Ashiq, J. Cecile, I. Drobyshev, Y. Bergeron, A. Das and C. Bulger. 2013. The scientific value of the largest remaining old-growth red pine forests in North America. *Biodiversity and Conservation* 22: 1847-1861.

Anawar, H.M., N. Canha, I. Santa-Regina, and M.C. Freitas. 2013. Adaptation, tolerance, and evolution of plant species in a pyrite mine, in response to contamination leveland properties of mine tailings: sustainable rehabilitation. *Journal of Soils and Sediments* 13: 730-741.

Asensio, V., S.D. Guala, F.A. Vega, and E.F. Covelo. 2013. A soil quality index for reclaimed mine soils. *Environmental Toxicology and Chemistry* 32: 2240-2248.

Bagatto, G. and J.D. Shorthouse. 1999. Biotic and abiotic characteristics of ecosystems on acid mettaliferous mine tailings near Sudbury, Ontario. Canadian Journal of Botany 77: 410-425.

Bayne, E.M., S. Boutin, and R.A. Moses. 2008. Ecological factors influencing the spatial pattern of Canada lynx relative to its southern range edge in Alberta, Canada. *Canadian Journal of Zoology* 86: 1189-1197.

Berkowitz, J.F., E.A. summers, C.V. Noble, J.R. White, and R.D. DeLaune. 2014. Investigation of biogechemical functional proxies in headwater streams across a range of channel catchment alterations. *Environmental Management* 53: 534-548.

Brown, R.L. and M.A. Naeth. 2014. Woody debris amendment enhances reclamation after oil sands mining in Alberta, Canada. *Restoration Ecology* 22: 40-48.

Burdett, C.L., R.A. Moen, G.J. Niemi, and L.D. Mech. 2007. Defining space and movements of Canada lynx with global positioning system telemetry. *Journal of Mammalogy* 88: 457-467.

Bussinow, M., B. Sarapathka, and P. Dlapa. 2012. Chemical degradation of forest soil as a result of polymetallic ore mining activities. *Polish Journal of Environmental Studies* 21: 1551-1561.

Callan, R., N.P. Nebbelink, T.P. Rooney, J.E. Wiedenhoeft, and A.P. Wydeven. 2013. Recolonizing wolves trigger a trophic cascade in Wisconsin (USA). Journal of Ecology 101: 837-845.

Cameron, E.K., E.M. Bayne, and M.J. Clapperton. 2007. Human-facilitated invasion of exotic earthworms into northern boreal forests. *Ecoscience* 14: 482-490.

Conestoga-Rovers and Associates. 2007. Expert report review of air permit application and draft air permit, Kennecott Eagle Minerals, Eagle Project, Marquette Michigan. Available from author on request.

DeLong, C., J. Skousen, amd E. Pena-Yewtukhiw. 2012. Bulk density of rocky soils in forestry reclamation. *Soil Science Society of America Journal* 76: 1810-1815.

Felleson, D.A. 1999. Iron ore and taconite mine reclamation and revegetation practices on the Mesabi Range in northeastern Minnesota. Restoration and Reclamation Review 5, No. 5.

Fischer, J., and D.B. Lindenmayer. 2007. Landscape modification and habitat fragmentation: A synthesis. *Global Ecology and Biogeography* 16: 265-280.

Ford, M.S. 1990. A 10,000-yr history of natural ecosystem acidification. *Ecological Monographs* 60: 57-89.

Freedman, B. and T.C. Hutchinson. 1980. Long-term effects of smelter pollution at Sudbury, Ontario, on forest community composition. *Canadian Journal of Botany* 58: 2123-2140.

Frelich, L.E., R.O. Peterson, M. Dovciak, P.B. Reich, J.A. Vucetich, and N. Eisenhauer. 2012. Trophic cascades, invasive species, and body-size hierarchies interactively modulate climate change responses of ecotonal temperate-boreal forest. *Philosophical Transactions of the Royal Society-B* 367: 2955-2961.

Frelich, L.E. 2013. "Boreal Biome" *Oxford Bibliographies in Ecology*. Ed. David Gibson. New York: Oxford University Press, May 2013. http://www.oxfordbibliographies.com/view/document/obo-9780199830060/obo-9780199830060-0085.xml

Frelich, L.E., C.M. Hale, S. Scheu, A.Holdsworth, L.Heneghan, P.J. Bohlen, and P.B. Reich. 2006. Earthworm invasion into previously earthworm-free temperate and boreal forests. *Biological invasions* 8: 1235-1245.

Frelich, L.E. and P.B. Reich. 2003. Perspectives on development of definitions and values related to old-growth forests. *Environmental Reviews* 11: S9-S22.

Frelich, L.E.  1995.  Old forest in the Lake States today and before European settlement. *Natural Areas Journal* 15: 157-167.

Frouz, J., V. Jilkova, T. Cajthami, V. Pizl, K. Tajovsky, L. Hanel, A. Buresova, H. Simackova, K. Kolarikova, J. Franklin, J. Nawrot, J.W. Groninger, and P.D. Stahl. 2013. Soil biota in post-mining sites along a climatic gradient in the USA: Simple communities in shortgrass prairie recover faster than complex communities in tallgrass prairie and forest. *Soil Biology and Biochemistry* 67: 212-225.

Gabriel, M., R. Kolka, T. Wickman, L. Woodruff, and E. Nater. 2012. Latent effect of soil organic matter oxidation on mercury cycling within a southern boreal ecosystem. *Journal of Environmental Quality* 41: 495-505.

Hansen, M.J., and A.P. Clevenger. 2005. The influence of disturbance and habitat on the presence of non-native plant species along transport corridors. *Biological Conservation* 125: 249-259.

Hawbaker, T.J. and V.C. Radeloff. 2004. Roads and landscape pattern in northern Wisconsin based on a comparison of four road data sources. *Conservation Biology* 18: 1233-1244.

Heinselman, M.L. 1996. The boundary waters wilderness ecosystem. University of Minnesota Press, Minneapolis, MN.

Jaakko Pöyry Consulting, Inc.  1994.  *Generic Environmental Impact Statement study on timber harvesting and forest management in Minnesota*. 500+ pages.

Jaakko Pöyry Consulting, Inc. 1992a. *Forest Wildlife: A technical paper for a generic environmental impact statement on timber harvesting and forest management in Minnesota*. 210 pp.

Jaakko Pöyry Consulting, Inc. 1992b. *Biodiversity: A technical paper for a generic environmental impact statement on timber harvesting and forest management in Minnesota*. 111 pp.

Johnson, D., W. MacDonald, W. Hendershot and B. Hale. 2003. Metals in northern forest ecosystems: role of vegetation sequestration and cycling, and implications for ecological risk assessment. *Human and Ecological Risk Assessment* 9: 749-766.

Jull, L.G. 2009. Winter salt injury and salt tolerant landscape plants. University of Wisconsin Cooperative Extension, Madison, WI.

Kaushal, S.S., P.M. Groffman, G.E. Likens, K.T. Belt, W.P. Stack, V.R. Kelly, L.E. Band, and G.T. Fisher. 2004. Increased salinization of fresh water in the northeastern United States. *PNAS* 102: 13517-13520.

Kociolek, A.V., A.P. Clevenger, C.C. St.Clair, abd D.S. Proppe. 2011. Effects of road networks on bird populations. *Conservation Biology* 25: 241-249.

Lankester, M. 2010. Understanding the impact of meningeal worm, *Paralaphostrongylus tenuis*, on moose populations. *Alces* 46, 53-70.

Larson, E., L.E. Frelich, P.B. Reich, C.M. Hale, and K. Kipfmueller. 2010. Tree rings detect earthworm invasions and their effects in northern hardwood forests. *Biological Invasions* 12: 1053-1066.

Lorenc-Plucinska, G., M. Walentynowicz, and A. Niewiadomska. 2013. Capabilities of alders (Alnus incana and A. glutinosa) to grow in metal-contaminated soil. *Ecological Engineering* 58: 214-227.

Mech, L.D., S.H. Fritts, G.L. Radde and W.J. Paul. 1988. Wolf distribution and road density in Minnesota. *Wildlife Society Bulletin* 16: 85-87.

Mitchell, C.P.J., R.K. Kolka, and S. Fraver. 2012. Singular and combined effects of blowdown, salvage logging, and wildfire on forest floor and soil mercury pools. *Environmental Science and Technology* 46: 7963-7970.

Mogren, C.L., W.E. Walton, D.R. Parker, and J.T. Trumble. 2013. Trophic transfer of arsenic from an aquatic insect to terrestrial insect predators. *Plos One* 8, article number e67817, DOI: 10.1371/journal.pone.0067817.

Moilanen, A., A.T. Smith, and I. Hanski. 1998. Long-term dynamics in a metapopulation of the American pika. *The American Naturalist* 152: 530-542.

MN DNR. 2003. Field guide to the native plant communities of Minnesota: the Laurentian Mixed Forest Province. Ecological Land Classification Program, Minnesota County Biological Survey, and Natural Heritage and Nongame research Program. MNDNR St. Paul, MN.

Mudrak, O., K. Uteseny and J. Frouz. 2012. Earthworms drive succession of both plant and Collembola communities in post-mining sites. *Applied Soil Ecology* 62: 170-177.

Norland, M.R. and D.L. Veith. 1995. Revegetation of coarse taconite iron ore tailing using municipal solid waste compost. *Journal of Hazardous Materials* 41: 123-134.

Northwatch and Minning Watch Canada. 2008. The boreal below. Mining Issues and Activities in Canada's Boreal Forest. Accessed June 23, 2014: http://www.miningwatch.ca/sites/www.miningwatch.ca/files/Boreal_Below_2008_web.pdf

Rayfield, B., M. Anand, and S. Laurence. 2005. Assessing simple versus complex restoration strategies for industrially disturbed forests. *Restoration Ecology* 13: 639-650.

Salemaa, M., I. Vanha-Majamaa, and J. Derome. 2001. Understorey vegetation along a heavy-metal pollution gradient in SW Finland. *Environmental Pollution* 112: 339-350.

Šamonil, P., R.J. Schaetzl, M. Valtera, V. Goliáš, P. Baldrian, I. Vašičová, D. Adam, D. Janik, and L. Hort. 2013. Crossdating of disturbances by tree uprooting: Can treethrow microtopography persist for 6000 years? *Forest Ecology and Management* 307: 123-135.

Sanderson, L.A., J.A. McLaughlin and P.M. Antunes. 2012. The last great forest: a review of the status of invasive species in the North American boreal forest. *Forestry* 85: 329-340.

Sarica, J., M. Amyot, L. Hare, M.R. Doyon, and L.W. Stanfield. 2004. Salmon-derived mercury and nutrients in a Lake Ontario spawning stream. *Limnology and Oceanography* 49: 891-899.

Schindler, D.W. and P.G. Lee. 2010. Comprehensive conservation planning to protect biodiversity and ecosystem services in Canadian boreal regions under a warming climate. *Biological Conservation* 143: 1571-1586.

Schmiegelow, F.K., C.S. Machtans, and S.J. Hannon. 1997. Are boreal birds resilient to forest fragmentation? An experimental study of short-term community responses. *Ecology* 78: 1914-1932.

Schlaghamersky, J., N. Eisenhauer, and L.E Frelich. 2014. Earthworm invasion alters enchytraied community composition and individual biomass in northern hardwood forests of North America. *Applied Soil Ecology*, in press.

Superior National Forest. 2014. Non-native invasive plant management on the Superior National Forest. Online, http://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsm91_048658.pdf. Accessed August 8, 2014.

Van Wilgenburg, S.L., K.A. Hobson, E.M. Bayne, and N. Kopper. 2013. Estimated avian nest loss associated with oil and gas exploration and extraction in the Western Canadian Sedimentary Basin. *Avian Conservation and Ecology* 8(2): 9. http://www.ace-eco.org/vol8/iss2/art9/

Wardle, D.A., M. Jonsson, S. Bansal, R.D. Bardgett, M.J. Gundale, and D.B Metcalfe. 2012. Linking vegetation change, carbon sequestration and biodiversity: insights from island ecosystems in a long-term natural experiment. *Journal of Ecology* 100: 16-30. [doi: 10.1111/j.1365-2745.2011.01907.x]

Witt, E.L., R.K. Kolka, E.A. Nater, and T.R. Wickman. 2009. Forest fire effects on mercury deposition in the boreal forest. *Environmental Science and Technology* 43: 1776-1782.

Woodruff, L.G. and W.F. Cannon. 2010. Immediate and long-term fires effects on total mercury in forest soils of northeastern Minnesota. *Environmental Science and Technology* 44: 5371-5376.