POTENTIAL ECOLOGICAL IMPACTS OF THE TWIN METALS MINE

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SUMMARY

This report addresses ecological issues related to the development of the proposed Twin Metals mine, with the purpose of informing the EIS process. **Section I** examines the ecological sensitivity of the region and its importance. The potential impacts of the mine are high because this is a very large mine located in an ecologically sensitive area. The Kawishiwi River is an important canoe travel route that flows through the heart of the Boundary Waters Canoe Area Wilderness (BWCAW), which is visited by 250,000 people annually. The Kawishiwi River flows out of the BWCAW through Birch Lake, re-enters the BWCAW through Fall and Basswood Lakes, and then flows into Ontario's Quetico Provincial Park. In addition, several protected research sites are located in the watershed, and the watershed is protected by an 1854 Treaty for Native Americans.

The State of Minnesota has designated many of these waters as "outstanding resource waters", with high levels of protection from pollution. Rivers and lakes in this region are poorly buffered, making them highly sensitive to acidic mine drainage (AMD). Additional AMD would be a cumulative impact, on top of bacterial contamination, widespread Hg contamination of fish (there are now more than 300 waters in Lake and St. Louis counties that have fish consumption advisories based on Hg), and residual AMD from an abandoned mining operation.

Fish in this region are sensitive to acidification that could be caused by AMD (**Section II**). Even small reductions in pH by AMD would result in losses of some species. Most species would be lost if the pH were to decline below 5.0. In addition to pH effects, AMD also impairs fish through metal toxicity.

Elevated levels of sulfate in AMD would likely affect wild rice (**Section III**). Minnesota has a 10 mg/L sulfate standard in wild rice waters (>70 in Lake County, the proposed site for the Twin Metals mine and 135 in nearby St. Louis County) to protect wild rice. Elevated levels of sulfate caused by AMD would probably affect other rooted aquatic plants. Waters in the region have a background level ~ 6 mg/L, so there is very little capacity of these waters to receive additional sulfate loading from AMD without causing impact on wild rice.

The proposed mine may also exacerbate Hg poisoning of fish, and of people who consume fish, mainly via the effect of sulfate (from AMD) on Hg methylation, which increases biomagnification (**Section IV**).

Inputs of AMD and elevated sulfate levels would also affect the cycling of phosphorus (P) between sediments and waters, because elevated levels of sulfate would alter the cycling of binding of P with iron (**Section V**). It is fairly clear that the short-term response of lake sediments to inputs of AMD would result in release of P to the water, causing eutrophication. There is less research regarding long-term response of lake sediments to sulfate inputs, but a likely effect may be less net retention of P in sediments, which would lead to increased algae growth and reduced clarity.

Section VI examines the potential of leaching of AMD from the Twin Metals mine tailings and waste rock on domestic drinking water wells that serve about half of the population of Lake County, including all residents in the vicinity of the proposed mine, who withdraw water from shallow, readily contaminated wells.

Finally (**Section VII**), there would be a serious risk for downstream ecosystems if the Twin Metals mine were to build a dammed tailings pond because of the potential for dam failure. The risk level for failure of tailings dams around the world is on the order of 0.1% per year, based on the historical record. The future risk may be much higher than historical records suggest, because climate change in this region has increased the intensity of precipitation events, a trend that is likely to continue. Because tailings dam failures are often triggered by heavy rainfall, the likelihood of failures may increase, unless designs of tailing dams are adapted to future climatic conditions. If a tailings dam failure were to occur, the distribution of sediments, acidic waters, and metals would likely contaminate the Kawishiwi River for tens of kilometers. While the initial flush would result in short-term death of fish and other biota, it would be of short duration. Damage caused by metal-laden sediments would likely persist for years.

I. BACKGROUND CONDITIONS THAT RENDER THIS AREA ESPECIALLY SENSITIVE

A. Outstanding resources waters. Many of the lakes and rivers in the proposed mining region are deemed “outstanding resource value lakes” under Minnesota statute 7050.0180, which recognizes that “ the maintenance of existing high quality in some waters of outstanding resource value to the state is essential to their function as exceptional recreational, cultural, aesthetic, or scientific resources. To preserve the value of these special waters, *the agency will prohibit or stringently control new or expanded discharges from either point or nonpoint sources to outstanding resource value waters*”.

Specifically, there are 347 outstanding resource waters in Lake County, both within the BWCAW (111) and outside (236), plus another 64 outstanding resource waters in Cook and St. Louis Counties, which also contain parts of the Kawishiwi River watershed (WICOLA 2013).

Moreover, the Kawishiwi River is an important canoe travel route that flows through the heart of the Boundary Waters Canoe Area Wilderness (BWCAW), which is visited by 250,000 people annually. The Kawishiwi's waters flow out of the BWCAW through Birch Lake and re-enter the BWCAW through Fall and Basswood Lakes, and then flow into Ontario's Quetico Provincial Park. The River has high recreational and economic value. South Kawishiwi River is a popular entry point to the BWCAW. The significance of the River is underscored by the number of protected research sites in the area, including: South Kawishiwi Protected Lands, Kawishiwi Pines Scientific and Natural Area, Kawishiwi Experimental Forest, and Keeley Creek Research Natural Area. The River and its lakes are protected for Native American cultural values and use (hunting, fishing, gathering wild rice) under an 1854 Treaty.

B. Waters near the proposed mine have low buffering capacity. Many waters in the Cu-Ni mining region have low buffering capacity and many are susceptible to acidification. Thingvold et al. (1979) reported that both streams and lakes had average alkalinities of 19 mg $CaCO_3$/L and specific conductance values (uS/cm) of 55 (streams) and 65 (lakes). The average stream sulfate concentration was 6.6 mg/L.

Lakes in this region were included in a statistically-based survey of the National Lake Survey, a study conducted by EPA in the 1980s to ascertain the sensitivity of lakes to acidification (Baker et al. 1990). Lakes in the region that would be potentially affected by the Twin Metals mine would be mainly in subregions 2A (Northeastern Minnesota) and 2B (East Central Minnesota) (see map, Figure I-1). None of the lakes in these two subregions was "acidic", defined as having an acid-neutralizing capacity (ANC) < 0, but many had pH $\leq$ 6.5 (14% of the lakes in 2A and 13% in 2B) and had ANC values $\leq$ 100 ueq/L (20% in 2A and 13% in 2B). To put this in perspective, *it would take only a 1:10 mixture of pH 3 acidic mine drainage to acidify lakewater with an ANC = 100 ueq/L and decrease the pH to < 5.* Hence, for some lakes in this region, it would take very small volumes of acidic mine drainage seepage to cause serious ecological impact.

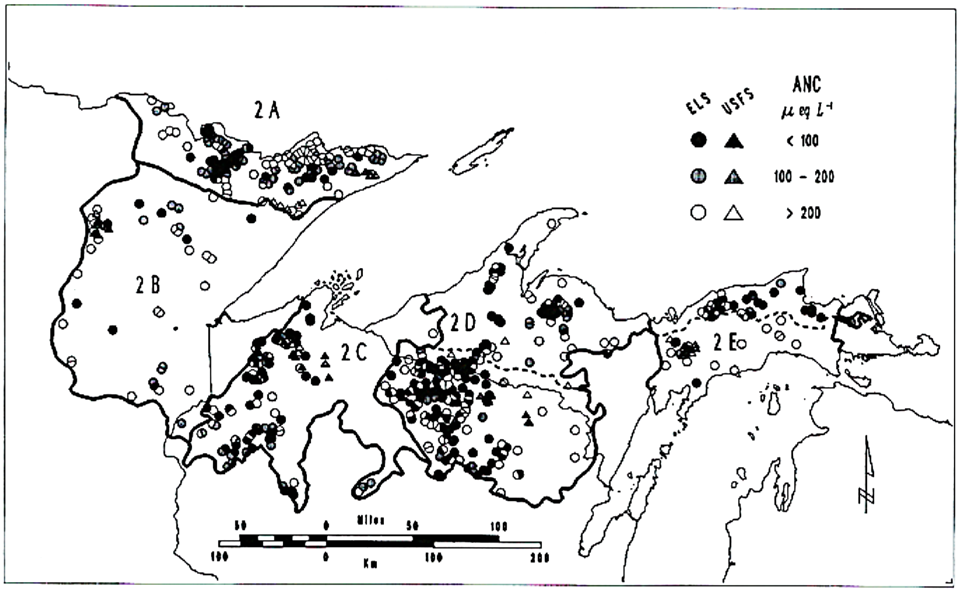

Figure I-1. Location of Midwest lakes sampled during the Eastern Lake Survey.

C. Any release of AMD from the Twin Metals Mine would be a cumulative impact. Impacts from the proposed Twin Metals mine would be a cumulative impact, exacerbating current water quality problems. These include:

Septic contamination. Waters adjacent to and near the proposed mine are already stressed. Most residents along the Kawishiwi River also depend on subsurface sewage treatment systems (SSTS). A SSTS survey of residences in the watershed revealed that only 35% were compliant, and 60% failed to protect groundwater (Associates 2012).

Table 1-1. Status of SSTS in the Kawishiwi River watershed (Associates 2012).

Status	Estimated Number	Estimated Percentage Range
Failure to Protect Groundwater	1173	55-65%
Compliant Not Meeting Setbacks	9	0-5%
Compliant Holding Tank Only	34	0-5%
Compliant SSTS	693	30-40%
No System (vacant/shared lot)	37	0-5%
Total	1946	100%

As one might expect from these findings, about 10% of water sampled by the White Iron Chain of Lakes Association's Beneficial Use Study in 2012 and 1013 were contaminated with fecal coliform (WICOLA 2013).

Existing mine drainage. There is also evidence of acidic mine drainage (AMD) to the river from previous mining in the immediate vicinity of the proposed Twin Metals operation. The Cu-Ni Regional Study (Thingvold et al. 1979) reported elevated levels of Cu (360-1000 ug/L), Zn (190-5300 ug/L) and Ni (10,000-13,000 ug/L) in a "small seep" at the "foot of a bulk sample site" (old mine rock from an exploration site). (Siegal and Ericson 1980) report copper levels in a groundwater discharge near Filson Creek of 700 ug/L and copper and nickel concentrations of 370 and 3,800 ug/L at well H2, nearby. More recently (Maccabee and Johnson 2011) report additional data for seeps from Dunka Pit, noting, for example, that sulfate is routinely discharged at concentrations > 1000 mg/L.

Mercury (Hg) accumulation in fish. A third issue of cumulative impacts is Hg poisoning of fish. This issue is discussed in greater detail in Section IV, but here it is noted that 116 waters within Lake County *already* have fish consumption advisories based on Hg contamination, as are 190 in nearby St. Louis County (MDH 2013). For several reasons discussed in Section IV, mining may exacerbate this problem.

Summary. The region where the Twin Metals mine proposes to locate has legally protected "outstanding resource waters" that are sensitive to acid inputs. New AMD would have to be considered as a cumulative impact, on top of septic system contamination, Hg contamination of fish, and existing AMD.

II. POTENTIAL EFFECTS OF AMD ON FISH

Modest quantities of AMD generated by the Twin Metals mine could easily impact fish and other aquatic organisms living downstream. This section first discusses the vulnerability of fish in this region to acidification in general, and then the specific impacts of AMD, which is characterized not only by low pH, but also high concentrations of toxic metals and sulfate.

A. Effects of low pH on fish and aquatic organisms. Most fish species in the Upper Midwest are very susceptible to declines in pH and disappear at pH levels far higher than pH levels typical of AMD (Baker 1990, Cook and Jager 1990). Figure V-2 (from (Baker 1990) shows the lower pH limit at which many common fish species can live. As the pH drops from 7 to 6, many minnow species disappear, including the blacknose shiner, bluntnose minnow, blacknose dace, and fathead minnow, but few game species are lost. As the pH declines from 6 to 5, 17 species are lost, including most important game species: walleye, smallmouth bass, lake trout, northern pike, and brook trout. Below pH 5, only a handful of species survive. Gamefish are reduced to largemouth bass and yellow perch.

With greater geographic specificity, Cook and Jager (1990) reviewed the literature on the disappearance of fish species with declining pH across lakes in northern Michigan and the Upper Peninsula of Michigan. Ten species disappeared as the pH declined from 7 to 6; 13 were lost between pH 6 and 5, and 9 as the pH declined from 5 to 4. Below pH 4, only the central mudminnow remained.

Findings from a long-term, experimental acidification of Lake 223 in Ontario, verify the progressive loss of fish species as pH declines (Mills et al. 1987). As the pH dropped to 5.64 in 1979, fathead minnows declined; pearl dace increased in 1980 (pH 5.59), but declined rapidly in 1982, when the pH was dropped to 5.89. All recruitment of new fish ceased by 1982. Lake trout survived down to pH 5.19 (1983), but became emaciated as food organisms succumbed to acidification.

Terrestrial piscivorous organisms would also be affected by drops in pH. In a major, long-term study of common loons in Canada, loon reproduction ceased when the pH declined to 6.4 (5.8-7.1) in small lakes and 5.5 (4.1-6.6) in large lakes, probably due to lack of forage fish (Tozer et al. 2013a, b).

In summary, even modest declines in pH of streams and lakes in this region would result in major loss of important fish species.

B. Additional impact of AMD. In addition to low pH alone, AMD also contains elevated levels of several toxic metals and high levels of sulfate and salinity. As AMD moves downstream, it undergoes numerous reactions, including chemical precipitation of Fe, Mn, and sulfate minerals, coating rocks, and bioaccumulation of metals in aquatic organisms (Moore et al. 1991). In their study of the Blackfoot River, MT, where AMD was generated by abandoned mines in the headwaters. Moore et al. (1991) found that pH and alkalinity were restored with a few kilometers, whereas elevated levels of some metals extended downstream 25 km or more. Persistence downstream was determined by dilution as well as reaction. Benthic fauna were absent just below the main AMD source, and impacts on occurrence persisted for 10s of kilometers. Many metals bioaccumulated, in the order (highest to lowest): Cd>Zn>Cu>As, Ni. Additional support for the role of direct metal toxicity comes from Farag et al. (2003), who used three approaches – survival, condition of populations, and physiological experiments – to understand the biotic impacts of AMD in several streams in the Boulder Creek watershed in Montana. They concluded that metals were a major mechanism of toxicity to trout.

As with acidification in general, the biotic impacts of AMD increase as pH is lowered. In a study of 166 species of fish in AMD streams in Pennsylvania, Butler et al. (1973) reported that 68 required a pH > 6.4, 38 tolerated pHs from 5.6-6.4, but only 10 tolerated pH < 5.5, with “severe degradation” occurring between pH 4.5-5.6.

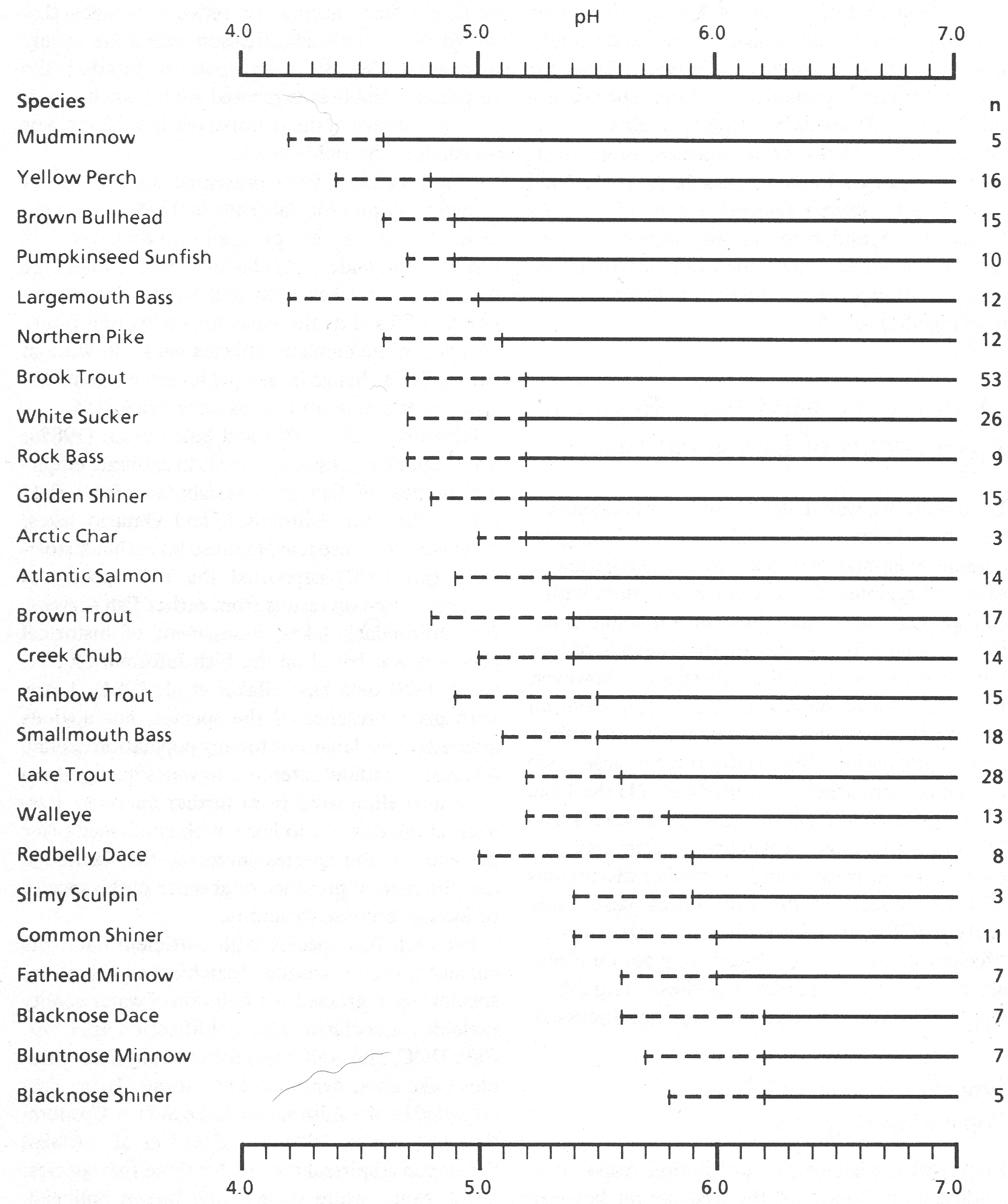

Figure II-1. Sensitivity of fish to low pH. From J. Baker (1990).

AMD impacts are very widespread throughout mining regions of the U.S. The U.S. Forest Service has estimated that 20,000-50,000 mines currently generate AMD, affecting 8,000-16,000 kilometers of streams (Jennings et al. 2008). In the mid-Atlantic and southeastern regions of the U.S., Herlihy et al. (1990) reported 4,590 km of AMD impacted streams (average pH = 4.3) and another 5,780 km of non-acidic but mine impacted streams.

C. Unpredictability of AMD effects. Finally, it is notable that predictions of AMD effects from previous EIS projects have been dismal. Kuipers et al. (2006) reviewed

25 EIS projects associated with major hardrock mines in the U.S. since 1975, mostly in western states. Of the 25, 19 had mining-related exceedences of water quality for either surface or groundwater. For 8 of these, EISs predicted low potential for contamination; for 8 others, EIS projects predicted moderate contaminant leaching; and for 3 others, high contaminant leaching was predicted. Thus, *for nearly half of the mines (8/19), exceedences of water quality standards occurred even though low impact was predicted.*

In a review of environmental releases from mines, EPA (1997) confirms the prevalence of toxic releases. In a review of 62 cases, not meant to be exhaustive, they identified 95 environmental releases. Of these, 49 involved inadequate containment of tailings, clay ponds, waste rock, process water, process solution (e.g., cyanide), wastewater, acid mine drainage, and stormwater.

At a local level, the problem of post-production AMD contamination is demonstrated by the Flambeau Mine near Ladysmith, WI (Chambers and Zamzow 2009). This mine started operation in 1993 and closed in 1997. The site contains 4.5 million tons of waste rock, but no tailings, because processing was done off site. The open pit was backfilled with waste rock, to which 30,000 tons of limestone was then added. Since then, there have been water quality problems exiting the mined area. Specifically, there have been repeated exceedences of the Cu standard in "Stream C", which drains the site, with measured Cu levels running 2-10 time the chronic Cu standard. Limited sampling in the Flambeau River downstream of the site has resulted in one exceedence of the Cu standard. Moreover, there is some evidence that groundwater from the mine pit is bypassing (or passing through) the "slurry wall" designed to isolate groundwater in the mine pit. Hence, the region has two cases in which AMD from mine pits has already caused water quality problems.

Predictions of AMD generation potential are further complicated by increasing temperature. Street (2013) noted that the rate of reactions that generate AMD from sulfide minerals will likely increase as temperatures warm with climate change. Hence, models of AMD generation used in mining environmental assessments must adjust temperatures to match expected temperatures in the future.

Summary. Fish in this region are very sensitive to pH changes. However, the potential for damage from AMD goes beyond its capacity to lower pH. AMD contains metals, and these metals cause biological damage. Moreover, AMD is also typically very high in sulfate, which impacts wild rice (Section III), may affect phosphorus cycling (Section IV), and affects Hg cycling in ways that probably increase Hg levels in fish (Section V). Finally, the track record at predicting the impacts of AMD has been poor; and predictive capacity may be reduced even further as climate changes.

III. POTENTIAL IMPACTS OF ELEVATED SULFATE LEVELS ON WILD RICE.

A. Sulfate standard for wild rice. The state of Minnesota has a 10 mg/L sulfate (as SO4) standard for protection of wild rice. The mechanism for toxicity is probably through conversion of sulfate to sulfide in the root zone, with sulfide causing the actual toxicity. There are more than 70 wild rice lakes in Lake County, with a total of more than 3,500 acres of wild rice, and another 135 wild rice lakes in nearby St. Louis County, with a total of 8,944 acres (Appendix 1). Moreover, the average sulfate concentration in waters in this region is ~6 mg/L (as SO4), so it would take very little additional AMD to reach the 10 mg/L standard.

The State of Minnesota is currently evaluating the 10 mg/L sulfate standard, but the mechanism by which sulfate causes toxicity to rooted aquatic plants is well known. Briefly, sulfate diffuses into sediments in plant beds and is converted to sulfide through a process called sulfate reduction. This occurs only in anoxic conditions, and provides sulfide. Toxicity is reduced if there is sufficient iron (Fe) in sediments to bind with the sulfide. Also, to varying degrees, oxygen diffusing from the roots of aquatic plants creates oxic zones around their roots; this is believed to be a mechanism for combating sulfide toxicity.

B. Research on sulfate cycling impacts on wild rice and other plants. The importance of the sediment environment for wild rice is seen in a study of 39 wild rice sites in Wisconsin (24 productive and 5 non-productive (Painchaud and Archibold 1990). A major metric was the "redox potential" (Eh), an indication of the potential for sulfate reduction. In both years of the study, the productive sites consistently had higher Eh values than the less productive sites, indicating less potential for generating sulfide. Preliminary data from an ongoing study of wild rice beds throughout Minnesota also suggest that sediment sulfide is related to the presence of wild rice (Swain 2013).

Baker (2012) hypothesized that the threshold for sulfate toxicity may be related to the density of wild rice beds, because decaying wild rice itself may provide the organic substrate that fuels the process of sulfate reduction, which provides sulfide. If this were true, the level of sulfate in the water column that causes toxicity to wild rice might depend on the density of the bed – lower levels of sulfate would be needed to cause toxicity in a dense wild rice bed than in a sparse wild rice bed. An ongoing study by MPCA, due to be completed in December 2013, should provide greater understanding of the impact of sulfate on wild rice.

Although the focus of the sulfate standard has been wild rice, other species of rooted aquatic plants might also be affected by increased sulfate concentrations in water, again, through the mechanism of sulfide toxicity. The sensitivity of rooted aquatic plants to sulfide toxicity varies, and can be an important factor in determining the distribution of aquatic plants in some environments. For example, using lab incubations and mesocosm (field enclosure) studies in the Everglades, Li et al. (2009) concluded that sulfide toxicity inhibited growth of sawgrass (*Cladium*), whereas cattails (*Typha*) could tolerate high sulfide levels and could therefore outcompete the sawgrass. Van der Welle et al. (2007) reported a similar finding in the Everglades: common rush (*Juncus effuses)* was more tolerant of sulfide than marsh marigold (*C. palustris*). In both cases, tolerance to sulfide was related to differences in "radial oxygen diffusion" from the roots into the sediment, which causes oxidation of sulfide, thereby protecting the plant. Also in both cases, this adaptation mechanism altered competition between the two species to favor the more invasive, less desirable species.

Summary. Because wild rice, and very likely other aquatic plants, is affected by sulfate, probably at very low levels, waters in this region may be especially vulnerable to additions of sulfate via AMD. AMD added to streams and lakes would likely be harmful to wild rice, and might also alter the distribution of other rooted aquatic plants, via differential capacity to adapt to sediment sulfide. Research to refine this knowledge is needed to support an EIS for the Twin Metals mine.

IV. POTENTIAL AMD EFFECTS ON EUTROPHICATION

None of the lakes in the Kawishiwi watershed are currently nutrient impaired (Anderson and Baratono 2011). Elevating concentrations of sulfate (via AMD) can increase P release from sediments of rivers, lakes, and wetlands (Caraco et al. 1993, Lamers et al. 1998, Zak et al. 2006), at least for short periods. The most likely mechanism by which sulfate causes P release is that sulfate is reduced to sulfide in anaerobic sediments; the sulfide then reacts competitively with Fe-bound phosphate, forming iron sulfide in the sediment and releasing phosphate to the overlying water. This phenomenon has been repeatedly demonstrated in short-term studies, usually in sediment-water microcosms or mesocosms. However, it is unclear whether this reaction is important in natural systems (Katsev et al. 2006).

A reasonable hypothesis regarding long-term inputs of AMD is that a rapid increase in sulfate (e.g., via addition of AMD) to a lake would result in a pulse of P release from sediments, resulting in a temporary increase in P concentrations in lakewater and a replacement of Fe-bound P in sediments with Fe-S compounds. This may cause a short-term eutrophication effect. However, over time, most of the Fe-bound P would be displaced from sediments. At this point, the release of P from sediments would decline, but there would be little sequestering of new P inputs to the lake by sediments, because most of the Fe would be now bound with sulfide. This would reduce the natural capacity of lakes to retain P in sediments, resulting in increased P concentrations in lakewater, with a concomitant increase in algal abundance and reduction in clarity. Alternatively, if AMD also supplied "new" iron, it is possible that P sequestration would continue, or perhaps even be enhanced.

With regard to a mine-related EIS, the role of AMD in regulating P cycling is not sufficiently well understood, so new research would have to be undertaken. Ideally, this would be done under near-natural conditions over the period of several years, perhaps in a whole-lake experiment analogous to the types of experiments conducted to evaluate the impacts of eutrophication (Schindler 195) and acidification (Mills et al. 1987) on lakes.

Summary. Increasing sulfate via AMD would very likely result in release of P from sediments to lakes over short periods. The longer-term picture is more vague, but a reasonable hypothesis is that P retention in lakes would decline, resulting in greater abundance of algae and decreased clarity. It is not possible at this time to predict this effect with confidence.

V. POTENTIAL IMPACTS OF TWIN METALS MINE ON HG TOXICITY

A. Current impact of Hg poisoning in Minnesota. Hg poisoning is already a serious problem in Minnesota, affecting fish, wildlife, and humans. Hg is by far the most common water quality impairment, with more than 1,200 Hg-impaired waters (MPCA 2007). The State of Minnesota now has a state-wide "TMDL" (total maximum daily load) plan to reduce Hg in its waters (MPCA 2007).

Within the Northeast Region of the state, the target for Hg in a "standard walleye" is 0.2 ug/g. The average in 1990 was 0.57 ug/g, which means that a 65% reduction will be needed. Within Lake County alone (the proposed site for the Twin Metals mine), there are 116 Hg-impaired lakes (MDH 2013). Within the Kawishiwi River watershed there are 72 Hg-impaired waters. More broadly, Evers et al. (2011) concluded that Hg contamination is widespread in the Great Lakes region, and that Hg in fish and wildlife exceed human and ecologic risk thresholds in many areas.

Hg bioaccumulates, so it becomes more toxic as it moves up the food chain to organisms, such as loons, mink, and humans, which consume fish. The threat to humans in the Lake Superior basin is very serious. At present, humans near Lake Superior are already affected by Hg poisoning. A recently study of 1,465 infants found that 8% had blood Hg levels > 5.8 ug/L, the EPA reference dose, and 1% had Hg levels > 58 ug/L, the "benchmark dose limit"(McCann 2011). Blood Hg levels were higher in summer, suggesting fish consumption as a major source of Hg. Nationally, 6-8%% of U.S. women of childbearing age have blood Hg levels that exceed the EPA standard, which translates to 300,000-600,000 children with impaired risk of neurological health. As in the McCann report, elevated blood Hg levels were associated with higher consumption of fish.

In Minnesota, there also is considerable concern about Hg poisoning of the iconic common loon, our state bird. In a study of loons in Central Ontario, Scheuhammer et al. (1998) found that loon reproduction decreased dramatically in lakes where the Hg content of small (prey) fish was $\geq$ 3 ug/g wet weight. Their study showed that the Hg content of fish was directly related to the dissolved carbon content (DOC) of the lake water, and inversely related to pH ($r^2 = 0.44$). There was also a direct relationship between Hg levels in fish flesh and loon blood. Three of the 26 lakes they studied had small fish with a mean Hg level of 0.3-0.4 ug/g. Loons were absent in several lakes in which small fish had average Hg levels > 4 ug/g.

B. Sulfate from the Twin Metals mining operation could increase Hg exposure. Inputs of sulfate from AMD to lakes, or increased sulfur emissions from power production associated with mining, could increase Hg methylation, increasing biomagnification of Hg in the food chain.

Experimental evidence that added sulfate stimulates Hg methylation is found in numerous studies, including Gilmour et al. (1992) (reservoir sediments), Branfireun et al. (1999) and Jeremiason et al. (2006) (peatlands), and St. Louis et al. (1996)(boreal forest and wetlands). Within Minnesota, an ongoing, multi-year study of the Hg cycling in peatlands in the Marcell Experimental Forest has led to better understanding of Hg cycling in in Minnesota peatlands (Kolka et al. 2011). One key experiment was an ecosystem-scale experiment to simulate increased atmospheric deposition of sulfate (using sprinklers). This experiment found that increased sulfate deposition increased Hg methylation and export of methylated Hg (MeHg) from the watershed. Although some added sulfate enhances Hg methylation, the effect may diminish at higher sulfate concentrations. Gilmour et al. (1992) estimated that optimal concentrations for Hg methylation may be in the range of 10-20 mg SO4/L, with rates declining at higher concentrations.

Several studies conducted at the regional level support findings from experiments. Hurley et al. (1995) found that watershed yields of MeHg in 34 Wisconsin watersheds increased in direct relation to the % of watershed that was wetlands. Similarly, Evers et al. (2007) examined factors that cause Hg "hotspots" in in landscapes of the U.S. Northeast and Canada. In addition to Hg deposition rates, two key factors were landscape characteristics and water level fluctuation. Landscape characteristics that promoted biomagnification included shallow flowpaths, the presence of wetlands, unproductive surface waters, added sulfate, and acidification. In addition, water level fluctuations, such as occur in reservoirs with managed water levels, was associated with high Hg concentrations in fish. Most of these characteristics describe lakes in the vicinity of the proposed Twin Metals mine, though none have been acidified yet.

Summary. The Twin Metals mine might exacerbate the Hg problem in this region in two ways. First, it may release AMD to surface waters, causing sulfate levels to rise, spurring Hg methylation. Second, the mining operation may produce sulfur emissions from the power production needed to power the mine. Because sulfur emissions affect atmospheric sulfate deposition for hundreds of kilometers, the influence of power production should be considered in an EIS, even if these operations occur at some distance from the proposed Twin Metals mine. By either mechanism, elevated sulfate levels in surface waters would likely enhance Hg methylation in sediments, resulting in increase Hg levels in fish and organisms that eat fish, including humans.

VI. POTENTIAL EXPOSURE OF HUMANS FROM CONTAMINATED GROUNDWATER.

About half of the residents in Lake County (5,256 individuals in 2005) obtain their water from domestic wells (Table VI-1). This includes nearly all of the residents that live near the proposed location of the Twin Metals mine.

Domestic wells in this region would likely be especially susceptible to contamination for three reasons:

(1) Geologic materials in this region provide little buffering capacity; hence AMD seepage into aquifers would not be neutralized.

(2) Many residences using well water are located near lakes and rivers, so there is a strong potential for contamination of domestic wells by upstream mine discharges.

(3) The proposed mine site is immediately upstream from numerous residences, hence there would be little opportunity for dilution.

Finally, although the USGS water withdrawal data indicate no direct withdrawals of surface water for domestic use, it is well known that some residents do withdraw water directly from lakes and rivers for domestic use.

The concern regarding AMD contamination of groundwater would include both aesthetics (for example, elevated iron levels, which are aesthetically unpleasant but generally not harmful) and toxicity, mainly from metals.

VII. POTENTIAL HUMAN AND ECOLOGICAL IMPACT OF CATASTROPHIC FAILURE OF TAILINGS PONDS

One of the major potential hazards of large mining operations like Twin Metals is the potential for catastrophic failures of tailings dams. To estimate the potential impact of a tailings pond failure, two estimates of total tailings production were used. The first is from an early analysis of the development (Cox et al. 2009), which estimated that the mine would produce 282 million tons of tailings over its 22-year life (Cox et al., 18-24) corresponding to 97 million cubic meters. A second analysis, conducted by Edison Investment Research (2013) indicated a total mineable resource of 745 million tons, and a mining rate of 80,000 tons per day.

A. Local context. For both scenarios, tailings (in addition to waste rock) would have to be stored indefinitely at or near the mining site. There is considerable uncertainty as to how this would be accomplished. Cox et al. indicated that the Dunka Pit could potentially receive 113 million tons before a dam would have to be built. The Cox report estimated a total production of 282 million tons, which would yield about 280 million tons of tailings. The report also noted a "potential" for paste backfill of 55-80% of tailings to the underground mine shafts, corresponding to 154-224 million tons. If Dunka Pit were used with no underground storage, there would be a need to store an additional 167 million tons of tailings, presumably in a dammed, above ground tailings pond. If we also assume that Dunka Pit could also be used, there would be no need for a dam. Hence, the potential need for tailings storage could be as high as 228 million tons (if Dunka Pit and backfill options were not selected) to as little as zero.

Table VI-1. Water withdrawals in Lake County, 2005.
Source: USGS data files.

Total population, 1000s	11.156
Population on public water, 1000s.	5.814
Groundwater, MGD	0.0
Surface water, MGD	1.00
Total, MGD	1.00
Total delivered for domestic use, MGD	0.59
Population on domestic water, 1000s	5.256
Groundwater, MGD	0.37
Surface water, MGD	0.00
Total, MGD	0.37
Irrigation, acres	0.02
Surface withdrawal, MGD	0.02
Total, MGD	0.02
Mining	
Surface water, MGD	129.35
Groundwater, MGD	0
Total, MGD	129.35
Thermoelectic power plants, once through	
Surface water, MGD	0.60
Groundwater, MGD	0
Total, MGD	0.60
Total groundwater, MGD	0.37
Total surface water, MGD	130.52
Total, MGD	130.89

The updated estimate of total production by Edison (2013) would result in 741 million tons of tailings (assuming 0.5% copper is extracted from the mined ore). If the same amount of backfill paste were used (155-225 million tons), this would now leave 516 to 586 million tons to be disposed above ground. If in addition, an additional 113 million tons were disposed in Dunka Pit, this would leave 403-473 million tons that would have to be stored in a dammed, above ground tailings pond

Because no decision has been made on either the total quantify of mine resource (hence quantity of tailings) or disposal options, the EIS should assume that that *up to 741 million tons of tailings would have to be stored behind some type of dam.* This is a very important consideration with regard to potential impacts to human health and downstream ecosystems. Tailings dam failures are common throughout the world, including the U.S., and the consequences to both humans and ecosystems if such a failure does occur tend to be catastrophic.

B. Impacts of historical tailings dam failures. Several recent catastrophic tailings dam failures have motivated several reviews on the subject. Rico et al. (2008) compiled a database for 146 tailings dam failures, of which 57 (39%) were in the U.S. and 26 (18%) were in Europe, a total of 57% of all global tailings dam failures. An important observation here is that tailings dam failures are very much a “first world” problem. Although failures are often caused by multiple factors, extreme meteorological events were the most common single identifiable factor, accounting for 25% of dam failures worldwide. Most failures worldwide (83%) occurred when the dam was active. Davis (2002) concluded that *there are normally 2-5 major failures per year over the past 30 years. For an estimated 3500 tailings dams worldwide, corresponding to risk levels of 1 in 700 to 1 in 1750* ***per year***. Davis notes that this is a far higher rate of failure that for conventional dams, which have a failure rate of 1 per 10,000 per year. For an even larger database of 18,401 mines, Assam (2010) reported a failure rate over the past 100 years of 1.2%.
For recent failures (since 2000), unusual weather and management were the major causes of failures. Chambers and Higman (2011) note that our experience of the long-term stability of tailings dams after closure is still limited. Some reasons for the high failure rates include inherent geotechnical issues, the need to last “in perpetuity”, and the perception of owners that tailings dams are an annoying cost of doing business, from which they derive no revenue. Davis and Martin (2009) observed that tailings dam incidents tend to occur 2-3 years after a mining boom, suggesting that hasty design and environmental review may contribute to the problem.

Tailings dam failures have major human and ecological consequences. In a review of tailings dam failures over the past 100 years, Assam and Li (2010) enumerated the following impacts:

- loss of human life (44 cases),
- infrastructure damage (33 cases),
- environmental pollution (83 cases),
- public health impact (7 cases).

Macklin et al. (2006) note that there have been 59 major failures of tailings dams since 1970, which have resulted in loss of 700 human lives and long-term ecosystem damage.

The following examples briefly illustrate the types of human and environmental damage caused by tailings dam failures:
Porco, Bolivia. In 1996, 235,000 m^3 tailings were released to the Pilaya River, killing fish 500 km downstream and contaminating drinking water. Most sediments were deposited within 60 miles (Macklin et al. 2006).

Bozanta, Romania. Two failures occurred, one in January 2999 due to high snowfall, and another in March 2000, caused by heavy rain, releasing 100,000 m^3 of water contaminated with cyanide and heavy metals, causing fish kills downstream. A second tailings dam failure occurred in the same county in March, releasing 120,000 m^3 of contaminated water and sediment (Macklin et al. 2006).

Aznalcollar, Spain. In 1998, dam failure at a mixed metal mine released 4 million m^3 of water and two million m^3 of toxic mud, causing flooding along 400 m one both sides of the Guadiamar River, with mud accumulating 40 km downstream (Grimalt et al. 1999). The spill released 16,000 tons of zinc and lead, 10,000 tons of arsenic, 4,000 tons of copper, and various other metals, decreasing the pH downstream to 3 and killing fish. Thirty-seven tons of dead fish were removed. To prevent catastrophic damage to Donoras Park, a major wildlife refuge located downstream, [most likely earthen]] dams were immediately constructed to divert water from the park. The mud, spread over more than 4000 hectares, was removed; agricultural land that had been flooded was remediated, and fishing was prohibited.

Stava, Italy. In 1985, two tailings dams upstream from the village of Stava, Italy, failed, releasing 180,000 m^3 of mud, which flowed through the village of Stava, killing 168 people and injuring 20 more and virtually destroying most of the village

(Luino and De Graff 2012).

Figure VII-1. Photos from the Stava mine tailings disaster.

A general temporal sequence of the impact of tailings dam failures includes the following:

1. Immediately after the spill. This phase is typified by gross pollution of mud and highly toxic water, causing death by burial or acute toxicity. The extent of metal contamination is typically many tens of kilometers, depending on the quantities of sediment released, the flow of the river, and the geomorphic structure of the riverbed.

2. The most toxic period, involving soluble metals, ends quickly, as the plume of contaminated water flows downstream.

3. Metal-laden sediments are deposited in the stream channel and the flood plain. The metal-laden sediments continue to release soluble metals.

4. The deposited tailings are transported downstream during subsequent flooding events.

With regard to environmental analysis, the recommendations of Hudson-Edwards et al. (2003) are relevant. These include the (1) the need for pre-mining geomorphic study, including sediment analysis; (2) the potential contribution of pollutants from other sources; and (3) the need to set aside funds to respond to tailings accidents.

For perspective, the incidents cited above released relatively small volumes of tailings relative to the volume that will have to be stored at the Twin Metals mine.

For consistency, the values reported below are *all reported in millions of cubic meters* and including only the volume of tailings, not the water:

Twin Metals:	296 (upper estimate of total tailings to be stored)
Aznalcollar, Spain	1.3 (solids released during dam failure)
Porco, Bolivia	0.24 (solids released during dam failure)

C. Uncertainties associated with climate change. Climate change will add to the difficulties of assuring that tailings ponds are designed to withstand increasingly intense precipitation events (ICAT 2013). The frequency of "very heavy" precipitation events in our region has already increased by 45% from 1958-2000. In one notable event that illustrates the potential of increasingly intense precipitation events to cause infrastructure damage, the nearby City of Duluth received 10" of rain in a single 48-hour period on June 20, 2012. The flooding caused by the storm wrecked roads and sewers, and destroyed 1,700 homes, with repairs estimated to cost $39 million.

From the period 1971-2000 to 2041-2070, annual precipitation in the Ely region is projected to increase by 1.4-1.6 inches/yr. Perhaps of greater significant is that precipitation during the "wettest 5 days" is projected to increase by 0.2-0.4"/yr. Adding to the difficulties of predicting hydrology for tailings dam design, temperature is projected to increase by about 5 ^{o}F, the frost free season is projected to increase by 21-24 days, and flooding will likely increase. Climate change is also projected to convert the vegetation regime in the region, adding to the complexity of predicting future impacts from mining.

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Appendix A: Wild Rice Waters in Lake and St. Louis Counties.(MNDNR 2008)

County	Lake	Location size, acres	Est. wild rice coverage, acres
Lake	Bald Eagle	1,243	
Lake	Basswood	14,610	485
Lake	Bluebill	44	11
Lake	Bonga	138	138
Lake	Cabin	71	55
Lake	Campers	56	56
Lake	Charity	26	
Lake	Christianson	158	
Lake	Clark		
Lake	Clark	49	
Lake	Cloquet	176	
Lake	Cloquet River		
Lake	Comfort	42	
Lake	Cougar	71	1
Lake	Cramer	69	55
Lake	Crooked		
Lake	Crooked		
Lake	Crown	69	
Lake	Driller	24	
Lake	Dumbbell	476	48
Lake	Ella Hall	372	1
Lake	Fall	2,322	23
Lake	Farm	1,292	
Lake	Flat Horn	52	
Lake	Fools	14	14
Lake	Gabbro	927	
Lake	Garden	4,236	212
Lake	Gegoka	174	14
Lake	Greenwood	1,469	15
Lake	Harris	121	18
Lake	Hjalmer	109	2
Lake	Hoist	117	
Lake	Horse River		
Lake	Hula	121	121
Lake	Isabella	1,318	
Lake	Isabella River		
Lake	Island River	49	49
Lake	Kawishiwi	468	

Lake	Kawishiwi River		
Lake	Little Gabbro	151	
Lake	Little Wampus		
Lake	Lobo	132	99
Lake	Manomin	455	23
Lake	Middle McDougal	104	
Lake	Moose	201	
Lake	Mud	164	
Lake	Muskeg	178	71
Lake	Newton		
Lake	Nine AM	27	14
Lake	North McDougal	273	
Lake	Papoose	54	3
Lake	Phantom	70	
Lake	Railroad	11	1
Lake	Rice	206	206
Lake	Roe	76	
Lake	Round Island	58	58
Lake	Sand	506	51
Lake	Sand	38r3	
Lake	Scott	52	
Lake	Silver	38021900	1,239
Lake	Slate	293	
Lake	Snowbank	4,819	50
Lake	Source	35	1
Lake	Sourdough	17	17
Lake	South McDougal	277	3
Lake	Stony	409	245
Lake	Stony	38r6	
Lake	Upland	74	1
Lake	Vera	26275	
Lake	Wampus	146	
Lake	Wind	952	10
Lake	Wood	587	125

County	Location Name (i.e. Lake or River)	Location size (acres)	Estimated wild rice coverage (acres)
St. Louis	??		15
St. Louis	Alden	190	
St. Louis	Anchor	316	32

St. Louis	Angell Pool	500	80
St. Louis	Artichoke	306	
St. Louis	Balkan	36	2
St. Louis	Bear	125	125
St. Louis	Bear Island River		
St. Louis	Bear Trap	131	
St. Louis	Big	2,049	20
St. Louis	Big Rice	416	416
St. Louis	Big Rice	2,072	1,700
St. Louis	Birch	7,628	381
St. Louis	Black	118	
St. Louis	Blueberry	130	13
St. Louis	Bootleg	352	
St. Louis	Breda	137	135
St. Louis	Burntside	7,314	
St. Louis	Canary	22	1
St. Louis	Caribou	569	3
St. Louis	Cloquet River		
St. Louis	Comet	28	
St. Louis	Cranberry	69	
St. Louis	Crane	3,396	600
St. Louis	Deadmans	5	
St. Louis	Dollar	51	51
St. Louis	Duck	126	
St. Louis	Eagles Nest #3	1,028	
St. Louis	East Stone	92	24
St. Louis	East Twin		
St. Louis	Echo		
St. Louis	Ed Shave	90	
St. Louis	Elliot	393	20
St. Louis	Embarrass River		
St. Louis	Five Mile	106	10
St. Louis	Four Mile	86	1
St. Louis	Gafvert	33	1
St. Louis	George	42	
St. Louis	Gill	18	
St. Louis	Grand	1,742	10
St. Louis	Grass	49	1
St. Louis	Grassey		
St. Louis	Grassy		
St. Louis	Grassy		
St. Louis	Gull	196	20
St. Louis	Hay	47	
St. Louis	Hay	78	78

St. Louis	Hay	32	1
St. Louis	Hay	114	114
St. Louis	Hay	42	1
St. Louis	Hay	82	45
St. Louis	Hockey	139	70
St. Louis	Hoodoo	252	252
St. Louis	Horseshoe	39	10
St. Louis	Indian	57	
St. Louis	Jeanette		
St. Louis	Johnson	473	24
St. Louis	Joker	46	5
St. Louis	King	320	39
St. Louis	Kylen	16	2
St. Louis	La Pond	176	176
St. Louis	Leeman	284	90
St. Louis	Lieung	476	10
St. Louis	Little Birch	58	
St. Louis	Little Cloquet R.		
St. Louis	Little Indian Sioux		
St. Louis	Little Mesaba		
St. Louis	Little Rice	266	266
St. Louis	Little Sandy	89	89
St. Louis	Little Stone	163	
St. Louis	Little Vermillion	558	
St. Louis	Long (Butterball)	442	400
St. Louis	Low	353	71
St. Louis	Lower Pauness	162	1
St. Louis	Martin	71	
St. Louis	Moose	82	62
St. Louis	Mud	51	
St. Louis	Mud	71	18
St. Louis	Mud		
St. Louis	Mud Hen	165	
St. Louis	Myrtle	876	
St. Louis	Nels	200	2
St. Louis	Nichols	444	22
St. Louis	One Pine	369	37
St. Louis	Oriniack	748	
St. Louis	Papoose	16	16
St. Louis	Pelican&River	11,944	119
St. Louis	Perch	79	32
St. Louis	Petrel Creek		
St. Louis	Picket	78	7
St. Louis	Pike River		

St. Louis	Prairie	807	16
St. Louis	Rainy	220,800	
St. Louis	Rainy (Grassy Narrows)		
St. Louis	Rat		
St. Louis	Rat		
St. Louis	Rice	41	41
St. Louis	Rice		
St. Louis	Round	336	
St. Louis	Ruth	47	9
St. Louis	Sandpoint		
St. Louis	Sandy	121	121
St. Louis	Seven Beaver	1,508	1,282
St. Louis	Shannon (& River)	135	108
St. Louis	Side	25	15
St. Louis	Simian Lake	81	5
St. Louis	Sioux River		
St. Louis	Six Mile	103	1
St. Louis	St. Louis River		
St. Louis	Stone	230	173
St. Louis	Stone	160	24
St. Louis	Sturgeon	2,050	243
St. Louis	Sunset	309	6
St. Louis	Susan	305	
St. Louis	Tommila	87	85
St. Louis	Trettel Pool	30	3
St. Louis	Turpela	76	61
St. Louis	Twin	18	1
St. Louis	Twin		
St. Louis	Unnamed	101	20
St. Louis	Unnamed (Camp 97)	25	
St. Louis	Upper Bug	23	
St. Louis	Upper Pauness	215	1
St. Louis	Vang	126	3
St. Louis	Vermilion	49,110	250
St. Louis	Vermilion River	1,125	562
St. Louis	Wabuse	64	51
St. Louis	Washusk #1	51	40
St. Louis	Watercress	43	43
St. Louis	Watercress (Mud)	30	
St. Louis	Wheel	11	6
St. Louis	Whitchel	71	53
St. Louis	White Iron		
St. Louis	Wild Rice	2,133	1
St. Louis	Wolf	456	