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# Mercury in vegetation and soils at abandoned mercury mines in southwestern Alaska, USA

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## ABSTRACT:

We collected vegetation (willow and alder) and soil samples at three abandoned mercury (Hg) mines and at regional background sites in southwestern Alaska and compared Hg concentrations, speciation, and distribution. Total Hg and methylmercury (MeHg) concentrations were higher in vegetation and soil samples from all the mine sites compared to samples from the background sites but there was no correlation between total Hg concentrations in vegetation and total Hg concentrations in soil or between total Hg and MeHg concentrations. However, the percent MeHg of the total Hg was higher in samples from the background sites compared to samples from the mine sites and is higher in vegetation samples than in corresponding soil samples. The percent MeHg is an order of magnitude higher in the willow samples than in corresponding alder or soil samples. The percent of ionic Hg [Hg(II)] is highest in soil samples from the retort and background areas. The higher percent MeHg in vegetation and soil in samples from background sites may be explained by the higher proportions of reactive Hg species, such as gaseous Hg(0) and Hg(II) at these sites compared to the surface mined and tailings areas where most of the Hg is in the elemental and cinnabar forms. Dissolved gaseous

Hg species are more readily accumulated in vegetation and are more readily methylated than solid phases like HgS and liquid Hg.

**KEYWORDS:** *mercury, methylmercury, plants, soil, Alaska*

# INTRODUCTION

Mercury (Hg) is a heavy metal with no known biological function in any organism. It occurs in several forms, all of which are toxic to some degree (ATSDR 1999). When certain forms of Hg are ingested or inhaled by humans, the mercury adversely affects the central nervous system, the liver, and the kidneys; it can also cross the placental membrane in pregnant women, causing damage to the fetus (Clarkson 1994; NRC 2000). Mercury in the environment can be converted from inorganic forms such as HgS (cinnabar), Hg(0) (liquid Hg or Hg vapor), or Hg(II) (ionic, "reactive" Hg) to organic forms, such as methylmercury (MeHg), by aerobic and anaerobic bacteria (Wilson & Brown 2000). Organic Hg compounds, especially MeHg, are the most toxic forms of mercury (NRC 2000). Methylmercury accumulates to a greater extent in biological tissues than do inorganic forms of mercury because it is lipid soluble and has an affinity for the sulfhydryl (—SH) groups of some proteins (ATSDR 1999). Mercury concentrations tend to increase in organisms that are higher in the food chain, a process called biomagnification. Concentrations of the organic forms of Hg in the terrestrial environment are generally at least one order of magnitude less than concentrations of inorganic forms of Hg, but Hg accumulation in terrestrial food webs is not well studied. (Gnamus *et al.* 2000) suggest that significant amounts of MeHg can be transferred to herbivores from vegetation and that food intake of Hg in primary herbivores such as roe

deer (*Capreolus capreolus*) is more important than inhaled Hg near the Idrija (Slovenia) Hg mine.

Mercury mines and deposits are found throughout southwestern Alaska (fig. 1); they represent significant sources of Hg that could potentially damage surrounding environments and may pose a potential health risk to residents and wildlife. The objectives for this study were to determine the concentration and distribution of total Hg and MeHg in vegetation and soil samples collected from the Red Devil, Cinnabar Creek, and Red Top Hg mines and to compare these values with regional background sites. Assessment of the effects to the aquatic food chain was the focus of previous environmental studies in this region that evaluated Hg in stream-sediments, stream-waters, and fish collected downstream from Hg mines (Gray *et al.* 1994; Gray *et al.* 1995; Gray *et al.* 2000). However, little attention has been given to Hg in soils or vegetation in this area. Data from Lindberg (1996) indicates that the largest Hg fluxes in forested areas may be gas exchange at the air/vegetation interface suggesting that forests could act as sinks or sources for atmospheric Hg.

Some of the common vegetation species in this region have a variety of uses for local residents and are important food sources for residents and wildlife (Viereck 1987; Schofield 1989). Alder is commonly used for smoking fish; spruce logs are used in construction. Birch is also used for construction and for making baskets while the sap can be made into syrup. Humans and bears consume blueberries; ptarmigan and songbirds eat alder and willow buds and twigs (Schofield 1989). Willow and cottonwood shoots, leaves, and flowering parts are an important food for moose and rabbit (Viereck 1987). Bears, ptarmigan, moose, and rabbits are all harvested by residents for food.

# *GEOLOGY*

## Red Devil mine

The Red Devil mine is located about 10 km northwest of the village of Sleetmute (fig. 1) along the Kuskokwim River. Red Devil was Alaska's largest Hg mine; it operated intermittently from 1933 to 1971 and produced about 36,000 flasks (1 flask = 34.5 kg) of mercury (Miller *et al.* 1989). The deposit is found in rocks of the Cretaceous Kuskokwim Group (Cady *et al.* 1955), a thick sequence of interbedded graywacke and shale. These sedimentary rocks are cut locally by late Cretaceous and early Tertiary dikes. Ore at Red Devil occurs in discontinuous open-space veins, vein breccias, and massive replacements localized along and near intersections between northeast-trending altered dikes and northwest-trending bedding plane faults (MacKevett & Berg 1963). Principal metallic minerals are cinnabar (HgS) and stibnite ($Sb_2S_3$), with minor amounts of realgar (AsS), orpiment ($As_2S_3$), and pyrite ($FeS_2$). Quartz, carbonate, dickite, and sericite are common gangue minerals (MacKevett & Berg 1963).

The Red Devil mineralized zone covered a surface area of about 150 m by 270 m and extended about 190 m vertically downward (MacKevett & Berg 1963). Mineralized veins are mostly small, discontinuous, and less than 1 cm wide, but veins as much as 1 m wide and 10 m long were observed (MacKevett & Berg 1963). High-grade ore contained as much as 30% Hg, but most ore averaged about 2–5% Hg (Webber *et al.* 1947); (MacKevett & Berg 1963). Workings consisted of about 2,900 m of shafts, adits, drifts, crosscuts, and stopes (MacKevett & Berg 1963) although most of the shafts and adits have been filled in recently as part of ongoing remediation efforts by the Bureau of Land

Management (Schmetzer & Ramert 2001). Numerous sloughed trenches, where surface mining took place, total several hundred meters in length; most are heavily overgrown with alder and other vegetation. Several tailings and calcine piles (remains from ore retorting) on the site lie near the small Red Devil Creek that drains the mine area. The largest of these piles is adjacent to the creek and is about 60 m wide and 75 m long (fig. 2). Abundant cinnabar as well as lesser amounts of stibnite and a few beads of liquid Hg are visible in Red Devil Creek.

## Cinnabar Creek mine

The Cinnabar Creek mine is located about 100 km southeast of Aniak in the southern Kuskokwim Mountains (fig. 1). Bedrock in the area consists of interbedded graywacke, massive siltstone, volcanic rocks, and minor chert and limestone of the Triassic and Lower Cretaceous Gemuk Group (Cady *et al.* 1955; Sainsbury & MacKevett 1965). Gemuk Group rocks are locally cut by late Cretaceous- and early Tertiary-altered mafic dikes near the Cinnabar Creek mine.

The mine is located near the headwaters of Cinnabar Creek (fig. 3) and consists of a small open-pit about 50 m by 15 m and 10 m deep. Cinnabar was first discovered at Cinnabar Creek in 1941, and the mine operated intermittently from then until 1960 (Sainsbury & MacKevett 1965). Liquid Hg, stibnite and minor amounts of pyrite are also found in this deposit. Mercury ore consists of massive replacements, disseminations, and vug fillings of cinnabar in quartz-carbonate veins (Sainsbury & MacKevett 1965). Ore averaging about 3–4% Hg was processed nearby in a small retort built on the south side of Cinnabar Creek (fig. 3). About 525 flasks of mercury were recovered (Nokleberg *et*

*al.* 1987). Veins containing cinnabar and liquid Hg in the open pit and small ore piles at the mine site are sources of mercury that have eroded into Cinnabar Creek. Abundant cinnabar and a few beads of liquid Hg have been observed in stream-sediment samples collected from Cinnabar Creek (Gray *et al.* 1991).

## Red Top mine

The Red Top mine is located near the Wood River, about 27 km north of Dillingham (fig. 1). Placer cinnabar was first discovered in a nearby creek in 1941 and the lode source was discovered in 1943. The deposit is hosted in sedimentary rocks of the Triassic and Lower Cretaceous Gemuk Group locally cut by an altered mafic dike of Late Cretaceous or early Tertiary age (Sainsbury & MacKevett 1965). Cinnabar is the only ore mineral at this deposit and it is found in quartz-carbonate veins and vein breccias that cut altered sedimentary rocks replaced with calcite, dolomite, limonite, dickite, and minor amounts of disseminated pyrite (Sainsbury & MacKevett 1965).

Cinnabar was first mined from about 3 000 m of surface trenches that were excavated with a bulldozer. Later, a few hundred meters of underground workings were driven in through two adits (Sainsbury & MacKevett 1965). Ore containing as much as 70% Hg was reported from the surface trenches but most of the ore contains less than 2% Hg (Sainsbury & MacKevett 1965). The ore from the surface trenches and the underground workings was processed on-site in a small retort built adjacent to the Wood River (fig. 4). About 60 flasks were produced before the mine ceased operating in 1959 (Sainsbury & MacKevett 1965). A few small beads of liquid Hg were observed in the sediments of Wood River when we collected our samples at this site.

# METHODS

## *Field*

In the summer of 1996 we collected willow (*Salix* spp.) and alder (*Alnus crispa*) along with soil samples from several areas at the Red Devil, Cinnabar Creek, and Red Top abandoned Hg mines (figs. 2, 3, 4). Most of the willow samples were identified as *Salix planifolia* (diamondleaf willow); however, this species could not always be found at each site. When it was not available other species such as *Salix alaxensis* (feltleaf willow) or *Salix glauca* (grayleaf willow) were collected. For this study we assumed that all the willow species we collected accumulated Hg in a similar matter but it is possible there could be interspecies variation in the Hg accumulation mechanism. Mercury data from soil samples collected in the summer of 1998 for another study, and from soil samples collected in 1994 and reported in Bailey & Gray (1997) are also included in this report. Sample sites used for regional baseline determinations are shown on figure 1. These sites were selected based on similar bedrock geology, soil development, and vegetation to the mine sites but have no Hg deposits.

For each vegetation sample, the material identified as current-year's growth was collected—generally, the outermost 15 to 20 cm. At each sample site the vegetative material was composited from several branches on the same plant or from nearby plants of the same species. Soil samples were typically composited from the upper 15–20 cm material just below the surface litter layer. This is the zone of most active root development and is generally the primary zone of mercury accumulation in forest soils

(Godbold 1994). In the study area, soils are usually poorly developed below about 15 cm.

## *Analytical*

Vegetation samples were washed with distilled-deionized water, air-dried, and pulverized prior to analysis. Soil samples were air-dried before analysis. Sample drying took place in a Hg-free atmosphere at the laboratory. The 1996 vegetation samples along with the all of the soil samples were analyzed for total Hg using the method of Bloom & Fitzgerald (1988). The samples were digested in $HNO_3$ + $H_2SO_4$ and quantified by cold vapour atomic fluorescence spectrometry (CVAFS); the soil samples were digested in aqua regia followed by CVAFS detection. This technique for total Hg analysis has a precision of about ±20%. Methylmercury in vegetation samples was liberated through a distillation procedure (Horvat *et al.* 1993) then measured by CVAFS using the methods described in Bloom & Fitzgerald (1988), Bloom (1989), and Liang *et al.* (1994). Because of the high total Hg concentration in some of the soil samples, MeHg in the soils was solvent extracted rather than distilled to minimize potential artifact MeHg formation prior to detection by CVAFS (Bloom *et al.* 1997). A subset of the 1996 soil samples were also analyzed for inorganic (ionic or "reactive") Hg [Hg(II)] using the CVAFS method described in Bloom & Fitzgerald (1988). Ionic (reactive) Hg is operationally defined as the fraction of the total Hg in a sample that can be reduced by $SnCl_2$ after 1N HCl leaching (Bloom & Fitzgerald 1988). Precision of the Hg speciation methods is about ±10%. Standard reference materials, site duplicate, analytical duplicate, and matrix spike

samples were analyzed for each sample media using the same analytical methods for quality control purposes. The vegetation and soil data are reported on a dry weight basis.

Soils at the study sites are slightly acidic to nearly neutral, with values from 4.1 – 6.9 and averaging about 5.2. The pH of the soil samples was determined in the field by mixing equal volumes of soil and distilled deionized water to a 100-mL beaker. When most of the material had settled, the pH of the mixture was measured using a digital pH meter. Total organic carbon (TOC) content, as determined by difference after measuring total carbon (Curry 1990) and carbonate carbon (Brandt *et al.* 1990), varies considerably. Values from 0.1 – 12% were measured and the average value was about 2.3%.

# RESULTS

## *Vegetation samples*

Previous data from Bailey & Gray (1997) indicate that, in general, leaf tissue samples of a variety of shrub and tree species from Red Devil and Cinnabar Creek Hg mines, have the highest concentrations of total Hg compared to the other plant parts analyzed (fig. 5a, b). Total-Hg concentrations in vegetation samples collected from this study were as much as 970 ng/g in leaves, 210 ng/g in stems, and 370 ng/g in flowering parts. Total-Hg concentrations in background samples were as much as 210 ng/g in leaves, 110 ng/g in stems, and 240 ng/g in flowering parts. Only a few samples from that study were analyzed for MeHg; these concentration values range from about 0.45 ng/g to 2.8 ng/g and represent about 0.1-2.7% of the total Hg in the samples analyzed. The highest concentrations of MeHg were measured in leaf samples but the highest percentage of MeHg was found in two stem samples. For this reason we analyzed a composite of leaf,

stem, and flowering tissues, rather than separate plant parts for this present study. The 1997 data also show that willow and alder samples have higher total Hg concentrations (fig. 5a, b) than any of the other species analyzed and as a result we only collected these two species for this investigation.

The more recent vegetation sample data show that samples collected near the mine sites contain higher concentrations of total Hg and MeHg than samples collected from the background sites (table 1). The concentrations of total Hg in background vegetation samples are similar to those reported for other areas in Alaska with no known Hg deposits (Gough *et al.* 1991; Crock *et al.* 1999, 2000) or in other regions (Rasmussen & Mierle 1991). Generally, mercury concentrations were highest in samples from the tailings and from the mined areas at the three mines studied. Methylmercury values vary in different areas of the mine sites and overall, MeHg concentrations are higher in the willow samples than in the alder samples. Methylmercury concentrations as much as 11 ng/g were measured in willow from the mined area at the Red Devil mine and from the tailings area at the Cinnabar Creek mine.

### *Soil samples*

Soil Hg data from 1994, 1996, and 1998 have been combined and are shown in table 2; the 1994 data are from Bailey & Gray (1997). Similar to the vegetation data, total Hg concentrations are higher in soil samples from the mine sites than in samples from background locations. Overall, the highest Hg values are found in samples from the tailings areas at the three mines. Methylmercury concentrations in the soil samples from the mines and from the background sites are higher than MeHg concentrations in the corresponding vegetation samples. Values as much as 41 ng/g were measured in samples

from the mined area at Cinnabar Creek. Ionic Hg [Hg(II)] concentrations vary considerably but are higher in samples from the mine sites than in background samples. The highest Hg(II) value (438 mg/kg) is from a tailings sample from the Cinnabar Creek mine.

# DISCUSSION

Although none of the Hg mines in southwestern Alaska have operated for at least 30 years, Hg concentrations in vegetation and soil are all high compared to background samples (Tables 1, 2). While some studies (Huckabee *et al.* 1983; Cocking *et al.* 1995) have shown that total Hg concentrations in vegetation are correlated to total Hg concentrations in soil the relation in our data is not so clear (fig. 6). Although there appears to be no linear relationship between total Hg concentration in plants and total Hg concentration in soils (fig. 6), plots of the ratio of total Hg concentration in plants vs. soil (fig. 7) reveal higher ratios in the retort and background areas. Mercury accumulation in vegetation is the result of Hg volatilization (evasion) from soil and subsequent absorption through the stomata (Lindberg *et al.* 1979; Du & Fang 1982; Siegal *et al.* 1987; Godbold & Hütterman 1988), dry deposition of reactive Hg species from the atmosphere (Lindberg *et al.* 1991; Lindberg 1996), or transpiration of dissolved gaseous Hg species [(Hg(0) or Hg(II)] from the soil solution (Lindberg 1996; *S. Lindberg, pers. comm.* 2001; Hanson *et al.* 1995). The higher concentration ratios in the background and retort areas can probably have the highest proportion of reactive Hg species, such as gaseous Hg(0) and Hg(II), that likely resulted from retort emissions and atmospheric deposition. Beckwith (1965) reports that during the period 1940-1963, Hg recovery was only about 65-75 percent efficient and about 11-20 kg of Hg per ton retorted was lost to the

atmosphere. Ionic (reactive) Hg represents about 16-18% of the total Hg in retort and background soil samples while only about 4-7% of the total Hg in soil samples from the mined areas or the tailings (fig. 8). In the most contaminated areas at the three Hg mines we studied, such as the tailings or the mined areas where the concentration ratios are low (fig. 7), much of the mercury is in the form of cinnabar (HgS) that does not readily dissolve or volatilize under ambient conditions.

Our data also show little correlation between MeHg concentration in vegetation and MeHg concentration soil (fig. 9); it is unclear how MeHg accumulates in vegetation. Gay (1976) suggests that MeHg in plants is related to the age of the plant; younger plant tissues accumulate more MeHg than older tissues; this author provides no clear evidence on how MeHg is taken up by plants. Since we collected only the current year's growth and that may explain some of the high levels of MeHg we observed. The percent MeHg of the total Hg is high in the willow and alder samples from background compared to most of the mine site samples (figs. 10a, b). One exception is in willow samples from the surface mined areas (fig. 10a). Overall the vegetation samples have a higher percentage of MeHg (figs. 10a, b) than do the soil samples (fig. 10c) from all locations near the mines and the background sites (willow >> alder > soil). We observed in our initial study (Bailey & Gray 1997) that willow samples accumulated higher levels of total Hg than any other species tested. In this study we found that willow samples also accumulate higher levels (table 1) and higher percentages of MeHg than the alder samples (figs. 10a, b). Further studies on the accumulation mechanism or the formation of MeHg in vegetation are needed. Measuring the MeHg concentration in local herbivores to determine if MeHg

from vegetation bioaccumulates in these animals would also be an important contribution to assessing the impact of the Hg mines on the local environment.

Previous reports from Bailey and others (Bailey & Gray 1997; Bailey *et al.* 1997) showed that while MeHg concentrations in soil samples are high compared to background concentrations in soil samples from the most contaminated areas of the Hg mines, the percent MeHg of the total Hg is highest in soil samples from the background areas (fig. 10c) – similar to what we observed in the vegetation samples. Other researchers (Bloom & Katon 2000; Gnamus *et al.* 2000; Barkay 2001) have recently reported similar results in various environmental media; however, the reason for this is unclear. Since much of the Hg in soils at the mine sites we investigated is present as cinnabar (HgS) or liquid Hg [Hg(0)] it is not readily available for methylation. Most of the Hg at the background sites is likely deposited from the atmosphere in forms that are more readily methylated. A component of Hg in the atmosphere is gas-phase water-soluble Hg(II) (Lindberg 1996), a form that is reactive and can be methylated through microbial and abiotic pathways (Hobman *et al.*, 2000). Visible cinnabar in the tailings and calcines at both mines suggests that ore processing or retorting was inefficient. A considerable amount of gaseous Hg(0) and Hg(II) was emitted from the retort stacks (Beckwith 1965) and was likely deposited in the immediate vicinity. The higher percentage of Hg(II) found in the retort and background soil samples (fig. 8) than in the mined area or tailings samples may explain the higher percentages of MeHg observed in these samples.

# CONCLUSIONS

Total Hg and MeHg concentrations are elevated in willow and alder samples from the three mines we investigated compared to regional background sites. The willow samples generally have higher concentrations of Hg than the alders. Concentrations of total Hg, MeHg, and Hg(II) in soil are elevated in samples from the mine sites compared to background samples. No direct relationship between vegetation Hg concentration and soil Hg concentration is apparent. Although MeHg concentrations are higher in the mine site vegetation and soil samples than in the background samples, the percent MeHg of the total Hg is higher in the background samples. The higher percentages of MeHg in background vegetation and soil may be explained by the higher proportions of reactive Hg species, such as gaseous Hg(0) and Hg(II) at these sites, compared to the surface mined and tailings areas where most of the Hg in the form of cinnabar and liquid Hg. Dissolved gaseous Hg species are more readily accumulated by vegetation and are more readily methylated than solid phases like HgS and liquid Hg. Lindberg (1996) suggests that less than 2% of the Hg in soils is exported to streams but emissions of gas-phase Hg from terrestrial systems may be significant. Detailed studies of Hg vapor fluxes along with measurements of atmospheric Hg concentrations and speciation would be helpful in understanding Hg cycling at these mines and aid in estimating how much Hg is being contributed to the atmosphere and transported globally from past Hg producing sites.

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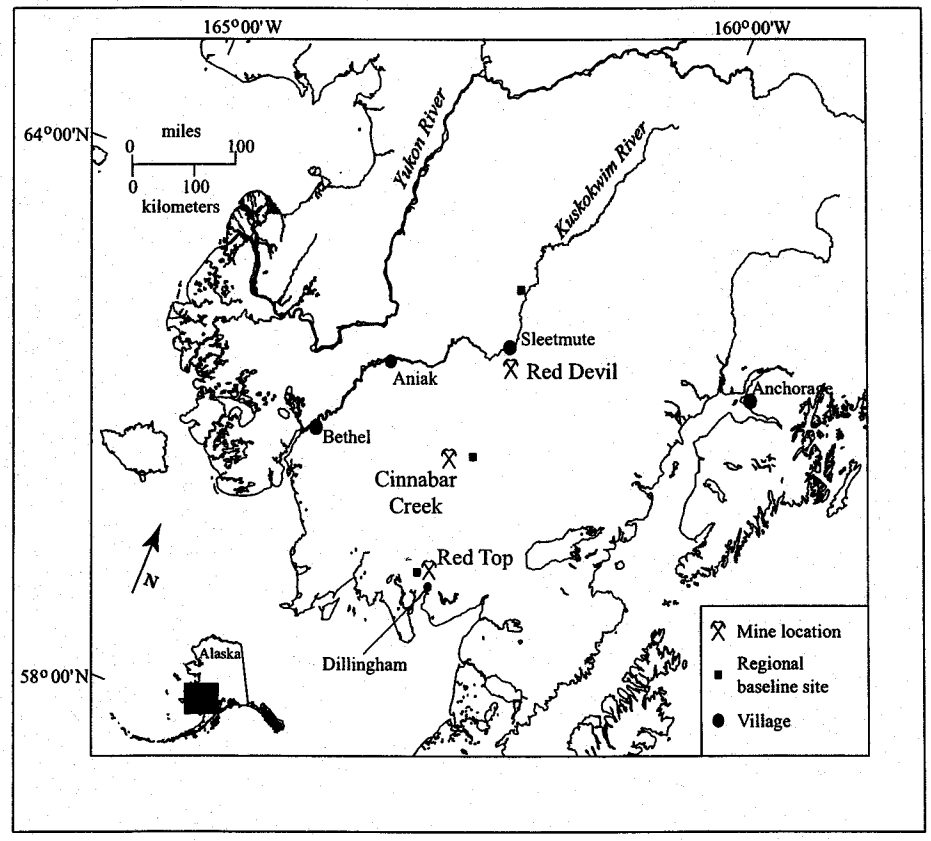



Fig. 1. Map showing the locations of the Red Devil, Cinnabar Creek, and Red Top abandoned Hg mines in southwestern Alaska, USA and regional baseline sample sites.

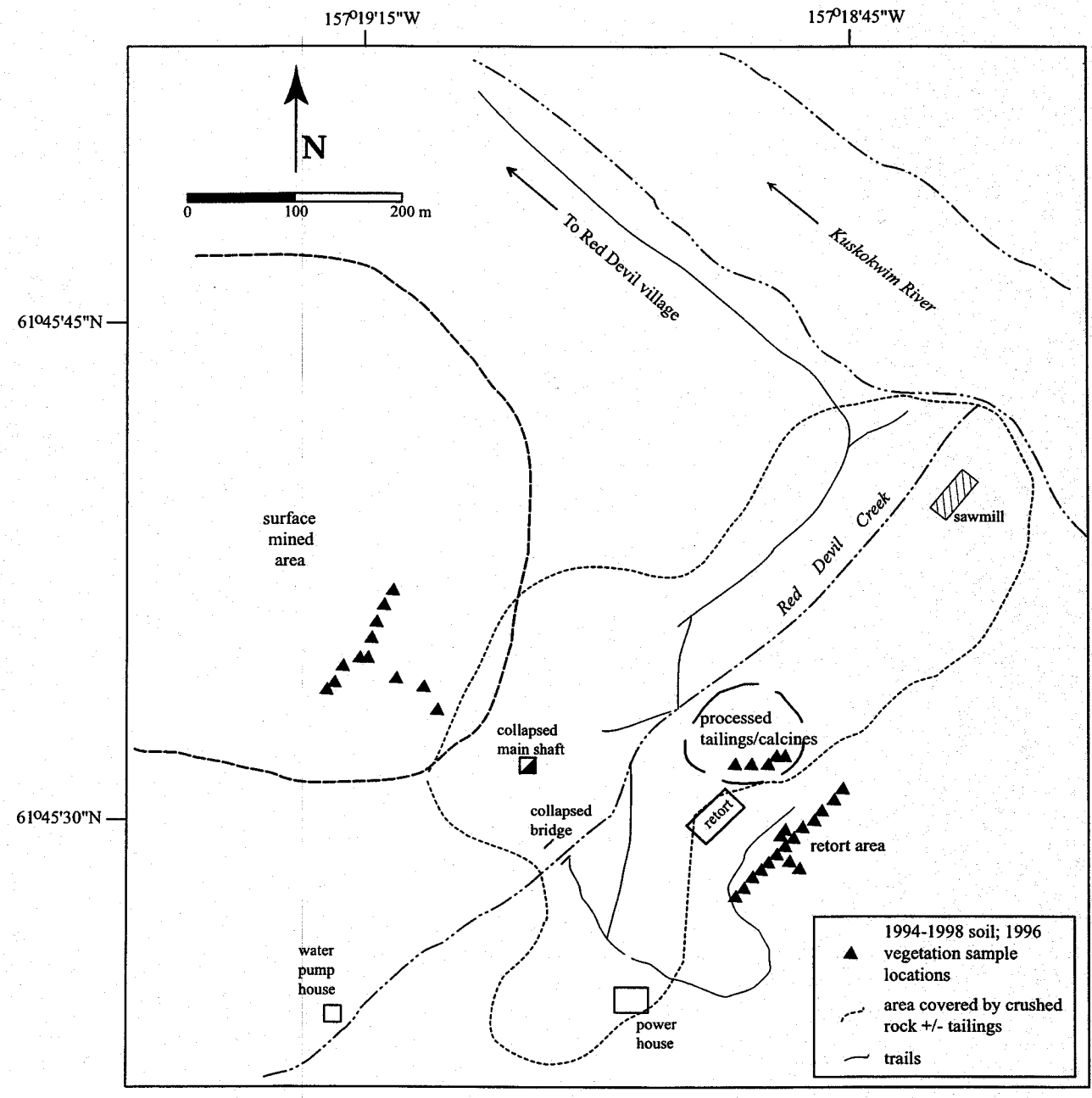



**Fig. 2.** Plan view of the Red Devil mine showing the location of sample sites for this study.

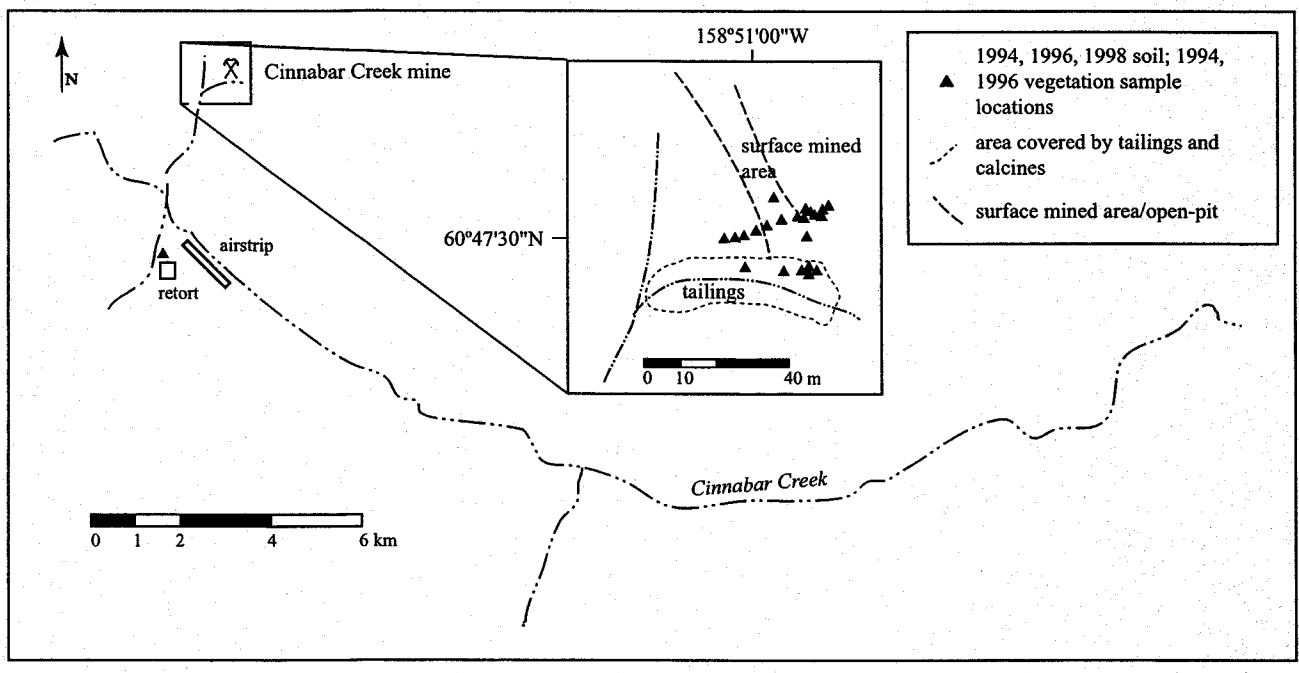



**Fig. 3** Plan view of the Cinnabar Creek mine showing location of samples sites used in this study. (modified from Gray *et al.* 2000).

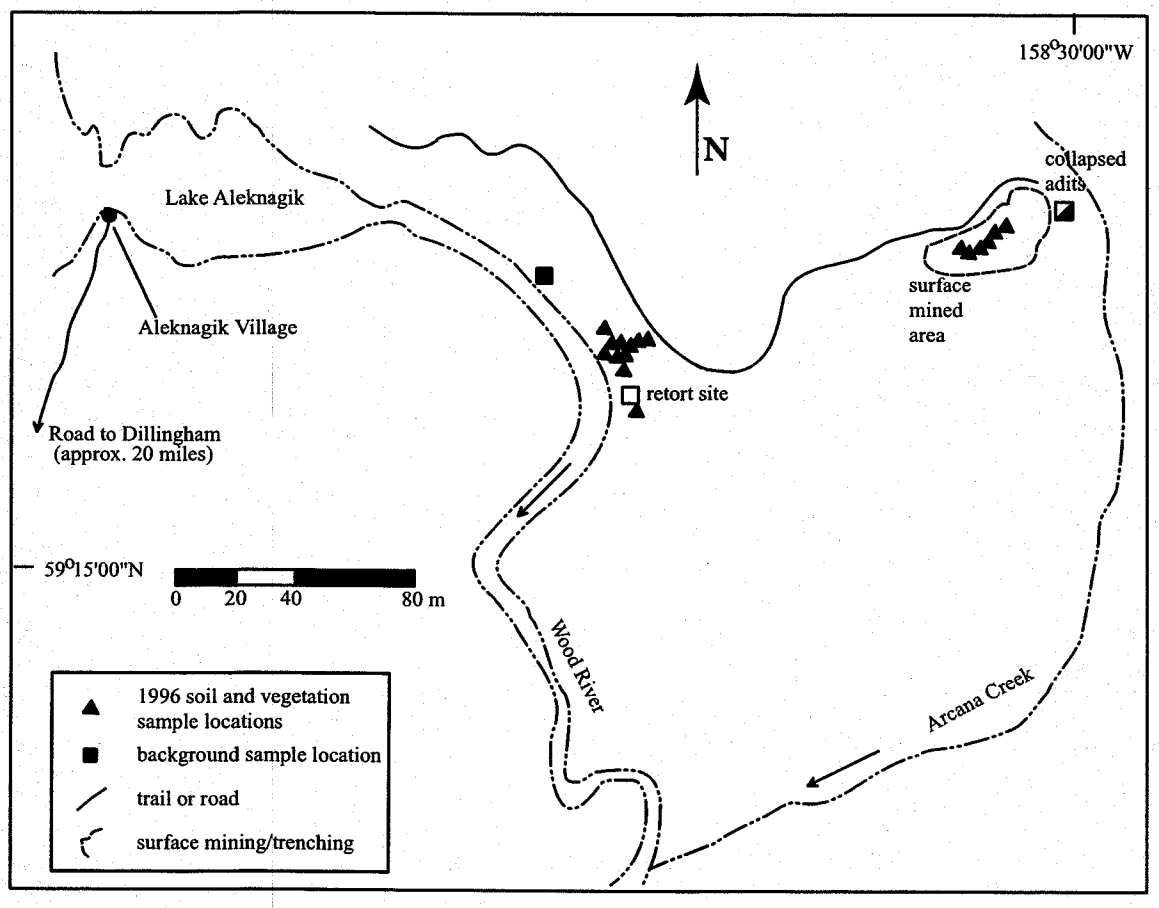



**Fig. 4.** Plan view of the Red Top Hg mine showing sample site locations and background site location.

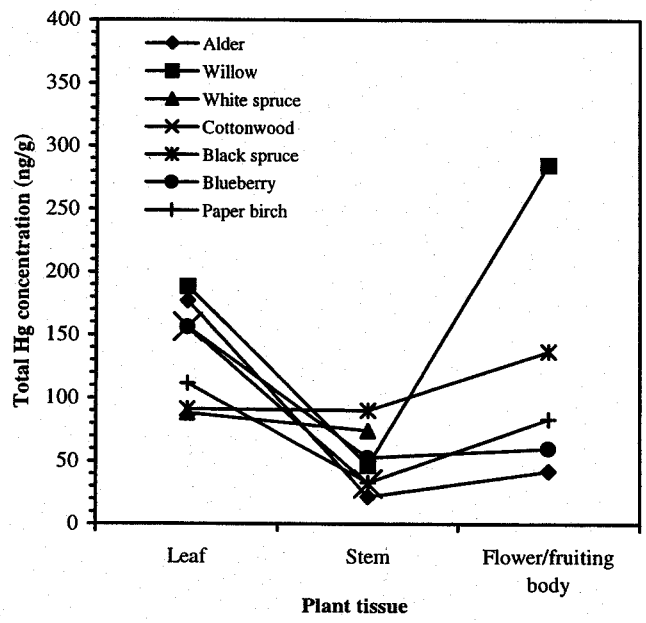



**Figure 5a**. Plot showing mean total Hg concentration in leaf, stem, and flower/fruiting body tissue in vegetation samples collected a the Red Devil Hg mine in 1994 (Data from Bailey and Gray, 1997)

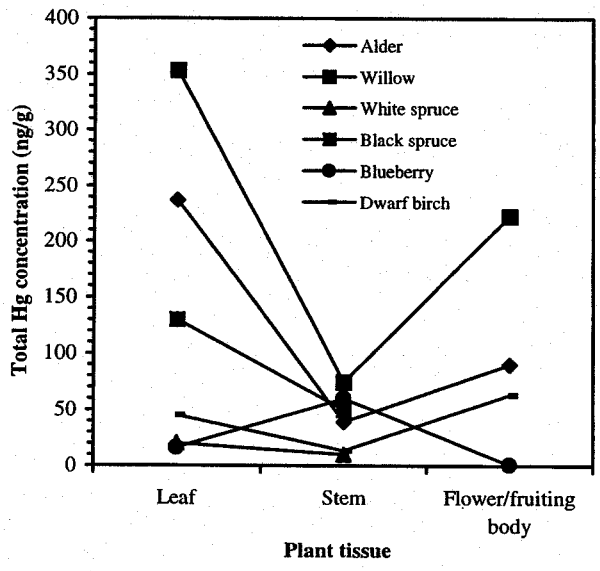
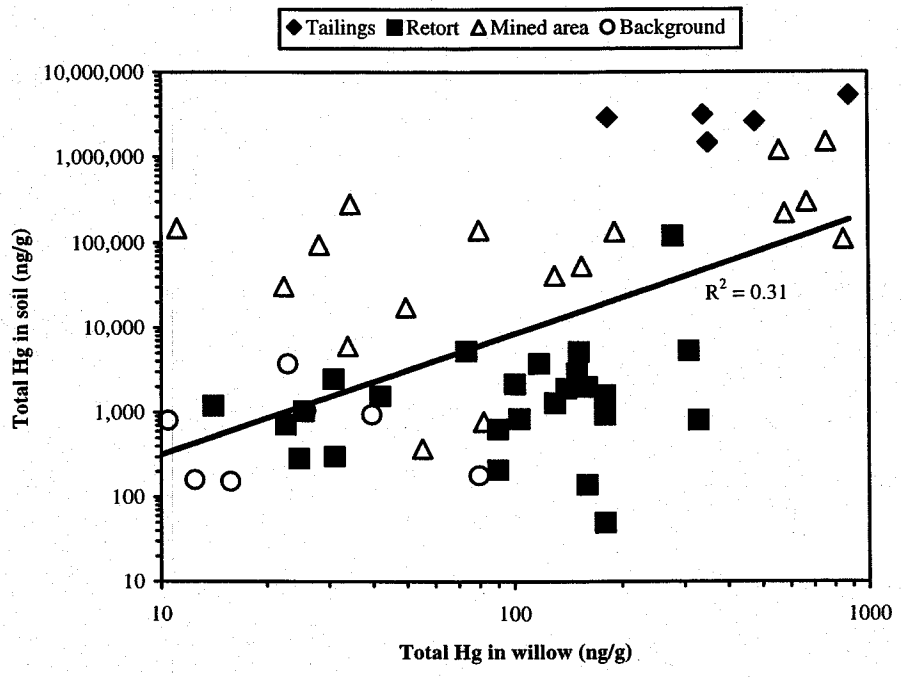



**Figure 5b**. Plot showing mean total Hg concentration in leaf, stem, and flower/fruiting body tissue in vegetation samples collected at the Cinnabar Creek Hg mine in 1994 (Data from Bailey and Gray, 1997)



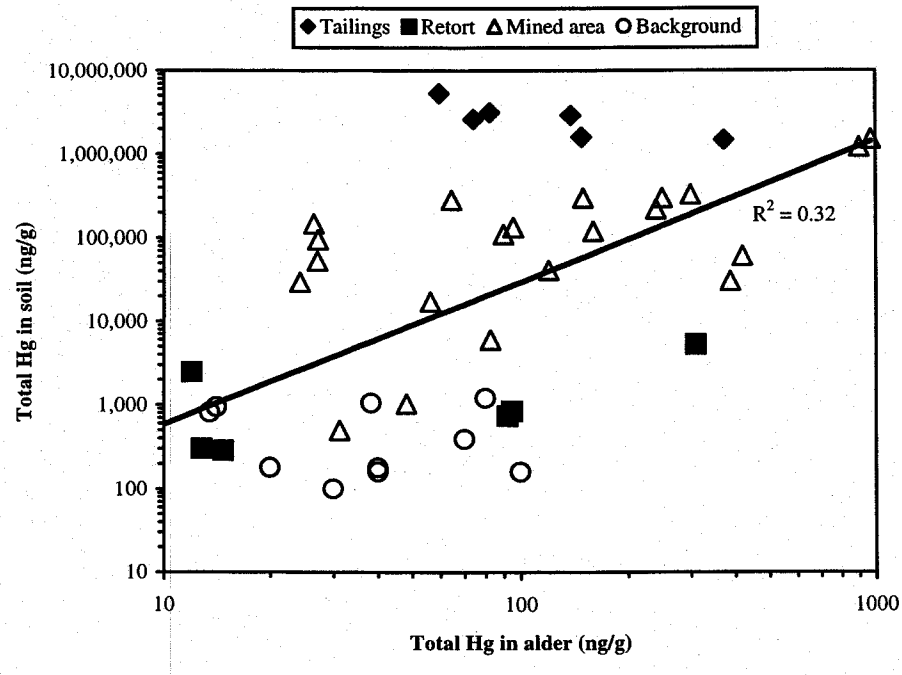




**Fig. 6.** Plot showing little correlation between total Hg concentration and total Hg concentration in willow (a) and alder (b).

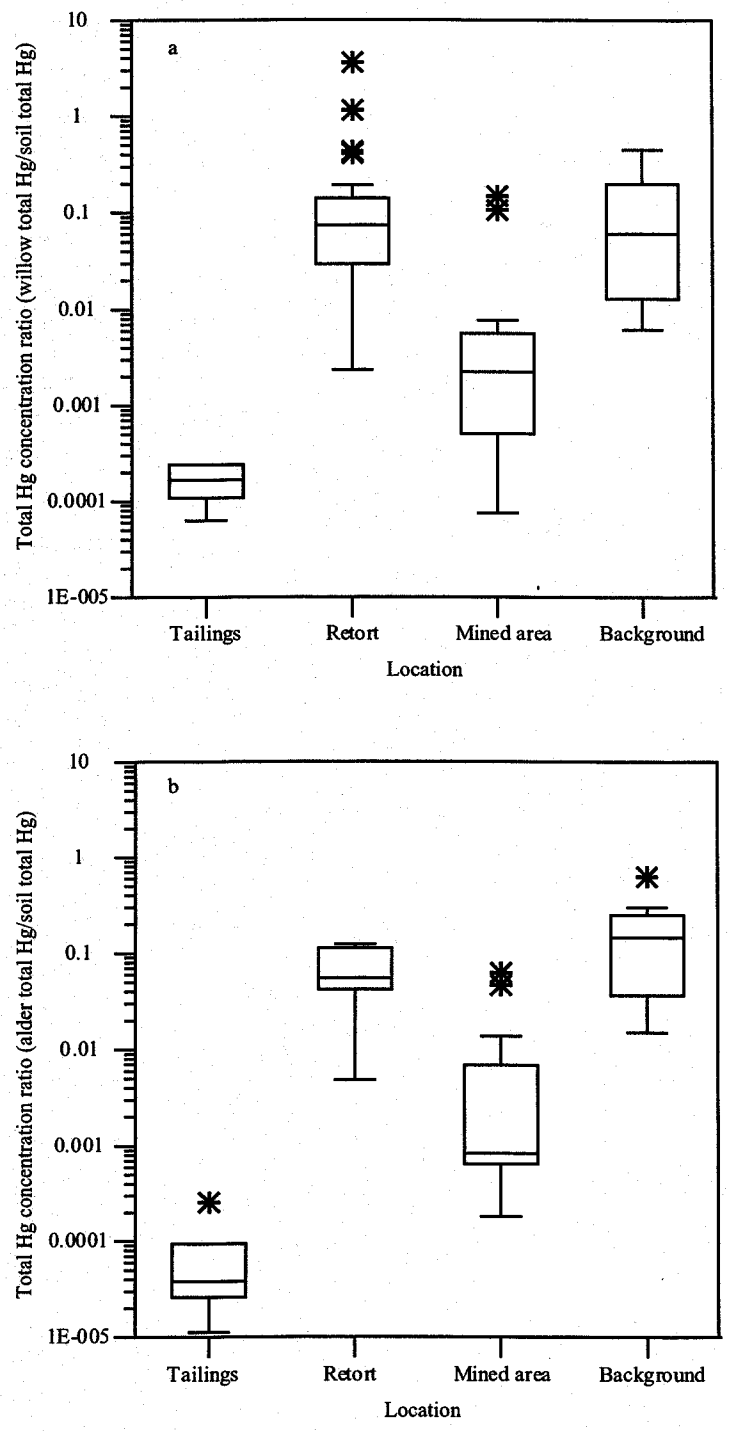



**Fig. 7.** Box and whisker plots showing plant total Hg concentration/soil total Hg concentration for willow (a) and alder (b) at the Red Devil, Cinnabar Creek, and Red Top Hg mines in southwestern Alaska, USA. The top line on each whisker represents the maximum value of the data set and the bottom line represents the minimum value. The middle line on each box represents the median value; the bottom and top lines represent the 25th and 75th percentile values respectively. The asterick symbols (*) indicate outlying values.

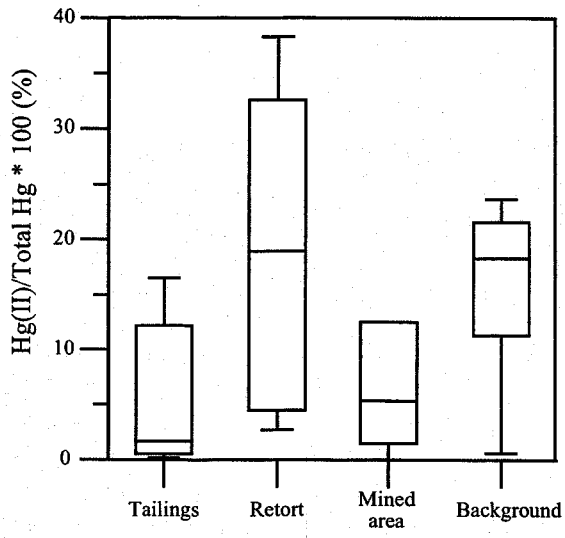
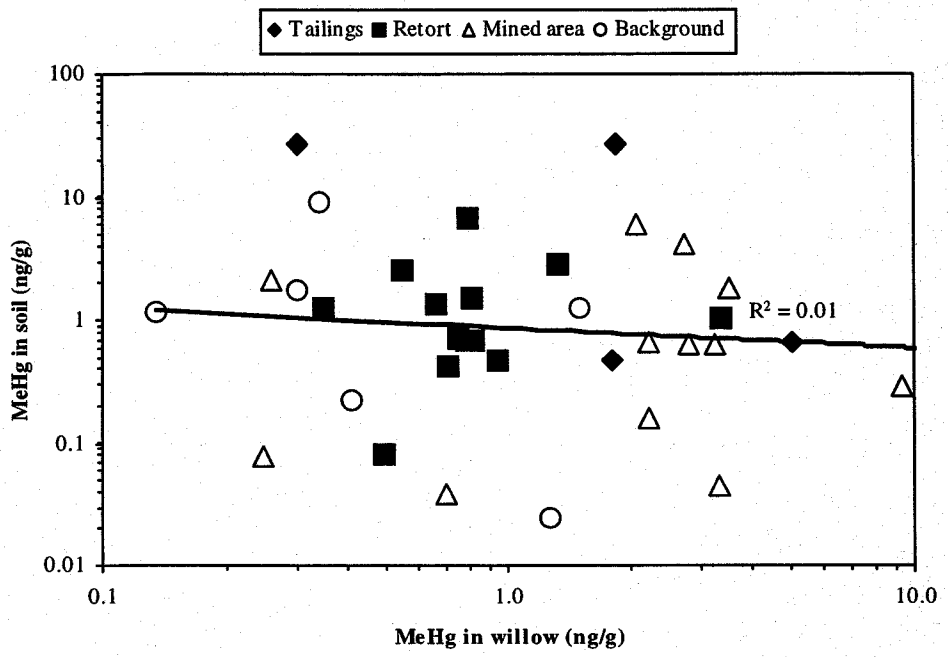



**Fig. 8.** Plot showing percent Hg(II) in soils samples from the Red Devil, Cinnabar Creek, and Red Top Hg mines in southwestern Alaska, USA.



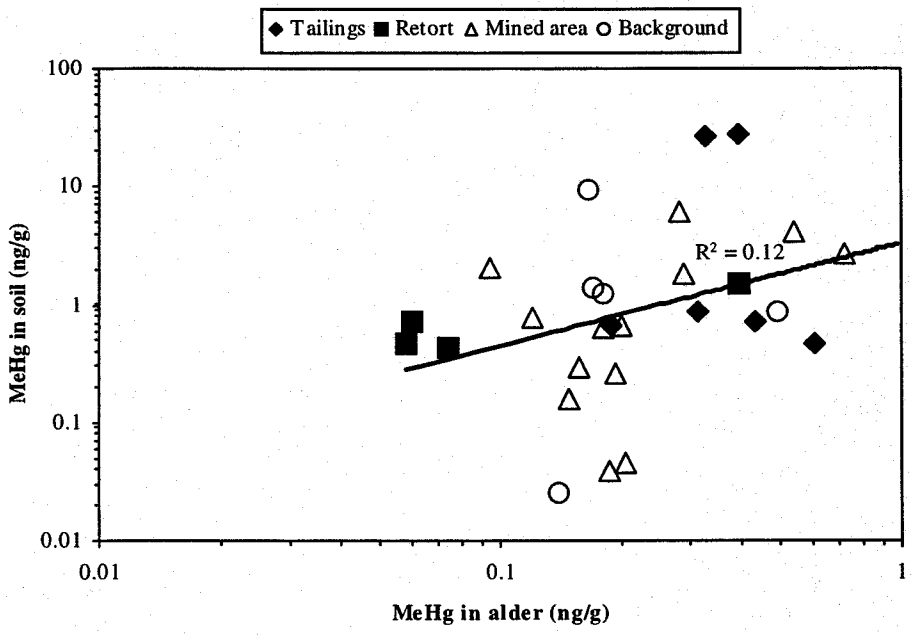




**Fig. 9.** Plot showing little correlation between MeHg concentration in soil and MeHg concentration in willow (a) and alder (b).

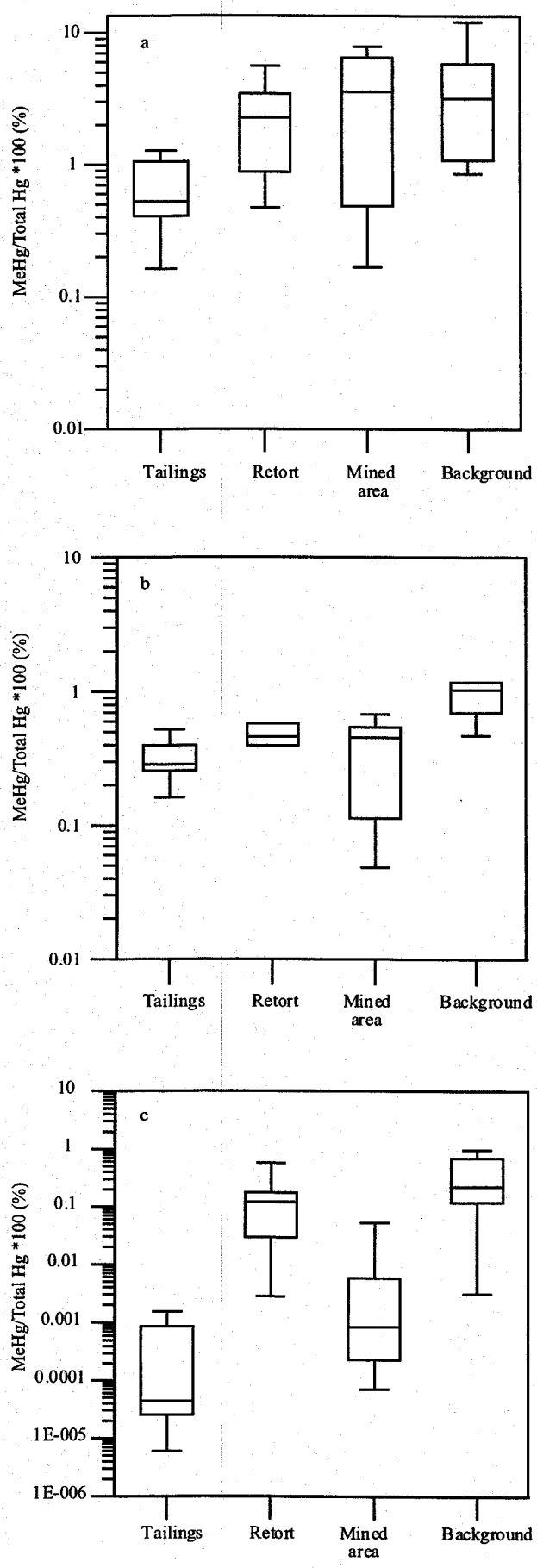



**Fig. 10.** Plots showing the percent MeHg of the total Hg in willow (a), alder (b), and soil (c) samples from the Red Devil, Cinnabar Creek, and Red Top Hg mines in southwestern Alaska, USA.

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**Table 1.** Summary Hg analytical data for vegetation samples collected at the Red Devil, Cinnabar Creek, and Red Top abandoned Hg mines in southwestern Alaska, USA

(n = 3, number of samples analyzed; where n = 0, no samples were available for collection)

| | | Alder | | | | | | Willow | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Mine | Sampling area | Total Hg (ng/g) | | | MeHg (ng/g) | | | Total Hg (ng/g) | | | MeHg (ng/g) | | |
| | | Mean | Range | n | Mean | Range | n | Mean | Range | n | Mean | Range | n |
| Red Devil | Tailings | 226 | (149 - 374) | n = 3 | 0.48 | (0.41 - 0.61) | n = 3 | 350 | (346 - 353) | n = 2 | 1.6 | (1.4 - 1.8) | n = 2 |
| | Retort | 310 | | n = 1 | | | n = 0 | 166 | (74 - 330) | n = 19 | 1.8 | (0.35 - 3.4) | n = 6 |
| | Mined area | 211 | (24 - 900) | n = 10 | 0.30 | (0.1 - 0.7) | n = 7 | 136 | (11 - 560) | n = 7 | 5.0 | (0.26 - 11.2) | n = 6 |
| | Background | 45 | (10 - 100) | n = 6 | 0.28 | (0.17 - 0.49) | n = 3 | 26 | | n = 1 | 1.5 | | n = 1 |
| Cinnabar Creek | Tailings | 89 | (60 - 139) | n = 4 | 0.31 | (0.19 - 0.40) | n = 4 | 473 | (185 - 884) | n = 4 | 4.6 | (0.3 - 11.3) | n = 4 |
| | Retort | | | n = 0 | | | n = 0 | | | n = 0 | | | n = 0 |
| | Mined area | 232 | (31 - 970 | n = 9 | 0.42 | (0.20 - 0.98) | n = 4 | 409 | (55 - 850) | n = 8 | 2.0 | (0.25 - 3.3) | n = 5 |
| | Background | 63 | (14 - 190) | n = 6 | 0.25 | (0.14 - 0.45) | n = 3 | 38 | (11 - 80) | n = 4 | 0.8 | (0.3 - 1.3) | n = 4 |
| Red Top | Tailings | | | n = 0 | | | n = 0 | | | n = 0 | | | n = 0 |
| | Retort | 45 | (12 - 95) | n = 5 | 0.15 | (0.4 - 0.06) | n = 4 | 37 | (14 - 104) | n = 8 | 0.7 | (0.5 - 0.9) | n = 9 |
| | Mined area | 160 | (27 - 388) | n = 3 | 0.18 | (0.15 - 0.2) | n = 3 | 28 | (22 - 35) | n = 3 | 1.7 | (0.7 - 2.2) | n = 3 |
| | Background | | | n = 0 | | | n = 0 | 12 | (8 - 16) | n = 3 | 0.3 | (0.14 - 0.4) | n = 2 |

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**Table 2.** Summary Hg analytical data for soil samples collected at the Red Devil, Cinnabar Creek, and Red Top abandoned Hg mines in southwestern Alaska, USA

(n = 3, number of samples analyzed; where n = 0, no samples were available for collection)

| Mine | Sampling area | Total Hg (µg/g) | | | MeHg (ng/g) | | | Hg(II) (µg/g) | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | Mean | Range | n | Mean | Range | n | Mean | Range | n |
| Red Devil | Tailings | 970 | (12.4 – 1,587) | n = 5 | 0.4 | (0.08 - 0.71) | n = 5 | 2.8 | (1.5 - 4.5) | n = 3 |
| | Retort | 8.5 | (0.05 - 120) | n = 21 | 3.3 | (0.69 - 8.2) | n = 8 | 0.26 | (0.12 – 0.40) | n = 2 |
| | Mined area | 210 | (6.0 - 1200) | n = 12 | 2.2 | (0.29 - 7.2) | n = 10 | 9.2 | (2.5 - 16) | n = 2 |
| | Background | 0.4 | (0.03 - 1.1) | n = 8 | 0.8 | (0.04 - 1.4) | n = 3 | 0.05 | (0.01 - 0.12) | n = 3 |
| Cinnabar Creek | Tailings | 2,750 | (165 – 5,326) | n = 6 | 11 | (0.66 - 27.4) | n = 5 | 186 | (12 – 483) | n = 3 |
| | Retort | 59 | | n = 1 | 1.7 | | n = 1 | 3.4 | | n = 1 |
| | Mined area | 330 | (0.38 – 2,539) | n = 16 | 5.4 | (0.05 - 41) | n = 3 | 318 | | n = 1 |
| | Background | 0.8 | (38 - 3743) | n = 10 | 1.7 | (0.02 - 9.2) | n = 7 | 168 | (7.0 - 368) | n = 4 |
| Red Top | Tailings | | | n = 0 | | | n = 0 | | | n = 0 |
| | Retort | 1.1 | (0.29 – 2.48) | n = 11 | 2.0 | (0.1 - 6.9) | n = 11 | 0.2 | (0.1 - 0.4) | n = 4 |
| | Mined area | 370 | (30 – 1,419) | n = 6 | 0.2 | (0.04 - 0.7) | n = 5 | 1.6 | | n = 1 |
| | Background | 0.1 | (41 - 159) | n = 3 | 0.9 | (0.2 - 1.3) | n = 3 | 0.2 | | n = 1 |

## DECISION RECORD AND DECLARATION

## RED DEVIL MINE

**I. Decision:**

It is my decision to implement the Red Devil Mine EE/CA Amendment at the Red Devil Mine, near the community of Red Devil, Alaska, at T. 19 N., R. 44 W., SE 1/4, Section 6, Seward Meridian, starting July 1, 2001. Comments received will be considered and incorporated into the final work plan as appropriate. This plan discusses the installation of an impermeable cap over the contaminated retort area soils/tailings, during the construction of a hazardous waste repository for disposal of mining debris, and construction of a solid waste monofill for uncontaminated debris. Characteristic hazardous waste will be treated to reduce contaminant mobility to render it non-hazardous. Other activities in this process and plan include the demolition and disposal of the remaining site buildings, the disposal of the fuel storage tanks, and additional sampling necessary to fill data gaps.

**II. Declaration/Rationale for Decision:**

The Proposed Action removes contaminants from the environment and protects human health and the environment from future contamination. An Engineering Evaluation/Cost Analysis and a Risk Assessment was completed as part of the development of this plan. The EE/CA and Risk Assessment showed that on-site disposal of contaminated and solid waste is an implementable, effective, and cost efficient remedy. This plan conforms to requirements of the Comprehensive Environmental Response, Compensation and Liability Act (CERCLA), the Resource Conservation and Recovery Act (RCRA), associated Alaska statutes, and site specific guidance by the U.S. Environmental Protection Agency and Alaska Department of Environmental Conservation.

**Field Manager** | **Date** April 3, 2002