

The Effect of Forest Fire on Soil Mercury Burden

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Abstract

Atmospheric deposition of mercury results in an increased load of Hg to organic soil horizons. Forest fires can generate soil temperatures sufficient to cause Hg volatilization and consumption of organic soil. The possible effect of forest fire on organic soil Hg burden was investigated. Organic soil samples from burned and unburned areas (1947), were collected from a watershed in Acadia National Park, Maine, U.S.A.. The burned site is characterized by a beech-dominated hardwood forest, and the unburned site is a mature spruce-fir forest. Samples were analyzed for total mercury. Organic soils from the burned site were thinner and significantly lower in Hg concentration (burned mean = 231 ng Hg/g dry wt., unburned mean = 550 ng Hg/g dry wt., $P = 0.004$). The lower Hg concentration in the soils from the burned sites probably result from a combination of the following: low or zero initial Hg burden, reduced Hg retention ability, lower dry deposition collection efficiency of the hardwood canopy, and possibly the orientation of the slope at the sampling site.

Introduction

The majority of mercury in soil is bound in the organic soil horizon (Lindquist et al., 1991; Steinnes, 1990, Abbott, 1994). This study investigated the possible role that forest fire may play in mercury burden reduction in organic soil horizons. The study site is in the catchment of the Eagle Lake in Acadia National Park, Maine, U.S.A.. The watershed was partially burned by a severe forest fire in October, 1947 (Patterson et al., 1983). The pre-fire forest is mature spruce-fir. The post-fire vegetation consists of a beech-dominated, northern hardwood forest. These two distinct forest types provided a means of identifying fire-exposed soil and unburned soil. The watershed is located entirely within the park boundaries and thus the possibility of other post fire disturbances, e.g. logging, are minimal. There have been no other substantial fires within this watershed from 1947 to 1983 (Patterson et al., 1983). Personal observations of sampling sites indicate that no other fires have occurred in this area from 1983 to present.

Mercury deposition occurs through wet and dry processes (Iverfeldt, 1991). Ionic forms of mercury are bound tightly to soil by forming complexes with organic matter in the upper soil

horizon (Lindqvist et al., 1991). Schuster, (1991), found that under acidic conditions, which prevail in podzolic soils, typical of the study site, organic matter sorbs mercury. Furthermore, solid organic matter in acidified soil columns decreases leaching of mercury 300% (Lodenius et al., 1987). The dominant mercury species in soil are gaseous elemental mercury (Hg^0) and the mercuric ion (Hg^{2+}), with small amounts of monomethyl and dimethylmercury (CH_3Hg^+ , $(\text{CH}_3)_2\text{Hg}$) (Revis et al., 1989; Steinnes, 1990; Schuster, 1991). The mercuric ion rarely occurs in the free ionic form under natural conditions due to its strong complexing (Steinnes, 1990), with organic matter, which is typically humic and fulvic acids (Mierle and Ingram, 1991).

Mercury in wet precipitation in Maine in 1993 was measured at 7.88 ng/liter in rain, and in snow was 7.76 ng/liter (Abbott, 1994). In 1995, mercury in wet precipitation at Acadia National Park was measured at 4.13 ng/liter (Webber, personal communication). The annual rain and snowfall in Acadia National Park averages 123 cm and 150 cm respectively (Patterson et al., 1983). Considering the additional input from dry deposition, which may contribute one-third of total annual input in other areas (Fitzgerald et al., 1991), a substantial annual mercury deposition may occur. Soils retain 80 to 90% of annual mercury deposition to terrestrial landscapes (Mierle, 1989; Aastrup et al., 1991). Thus, a substantial mercury burden may occur in

forest soils of Maine.

Because a typical forest fire generates soil surface temperature exceeding 510°C (Wells et al., 1979), and the majority of sorbed mercury volatilizes between 100° and 200°C (Landa, 1978), the mercury burden in the soil at the burned site should be significantly less than the burden in the unburned soil. In addition, substantial organic matter is lost to combustion during forest fires (Viro, 1974). This humus would lose any ionic forms of unvolatilized mercury that were previously humus-bound. Dissolved organic carbon input to surface waters rises dramatically for a period of 1 to 2 years following watershed burning (Schindler et al, 1992). He also found that after this initial rise, dissolved organic carbon declines to considerably lower concentrations than before burning. Furthermore, this loss of organic matter may have prevented the subsequent retention of deposited mercury until a layer of organic soil had accumulated. Viro (1974) found that it took 12 years for the formation of new humus to exceed decomposition of old humus after an intense forest fire, and fifty years for humus to return to pre-fire thickness. This long process of organic soil renewal results from the rapid decomposition of the successional deciduous tree litter, which results in a very slow accumulation of humus (Viro, 1974). Reductions of organic matter by fire are greatest on unproductive or submarginal forests where

there is only a thin layer of organic matter over parent material (Wells et al., 1979). This forest type is characteristic of the pre-fire vegetation that dominated the study site.

Material and Methods

The sampling sites were located in the southern part of the Eagle Lake watershed (Figure 1). The hardwood sites were located on the northeastern slope of The Bubbles, to the west of the Park Loop Road. The softwood sites were located on the northwestern slope of Pemetic Mountain, to the east of the Park Loop Road. The road approximately coincides with the burned area boundary and thus separates the two forest types. The softwood sites have a thick canopy of mature spruce trees and moist, moss dominated ground cover. The hardwood sites have a thick beech canopy with little ground cover. Three samples were collected from each site, approximately 10 meters equidistant from a center point. Soil samples were collected using an acid-washed, PVC corer, with a 40 mm inside diameter, from two softwood sites and two hardwood sites of similar elevation. The organic horizons were collected and placed directly in acid-washed glass jars with teflon-lined lids. Samples were weighed wet, freeze-dried and reweighed, and moisture concentration

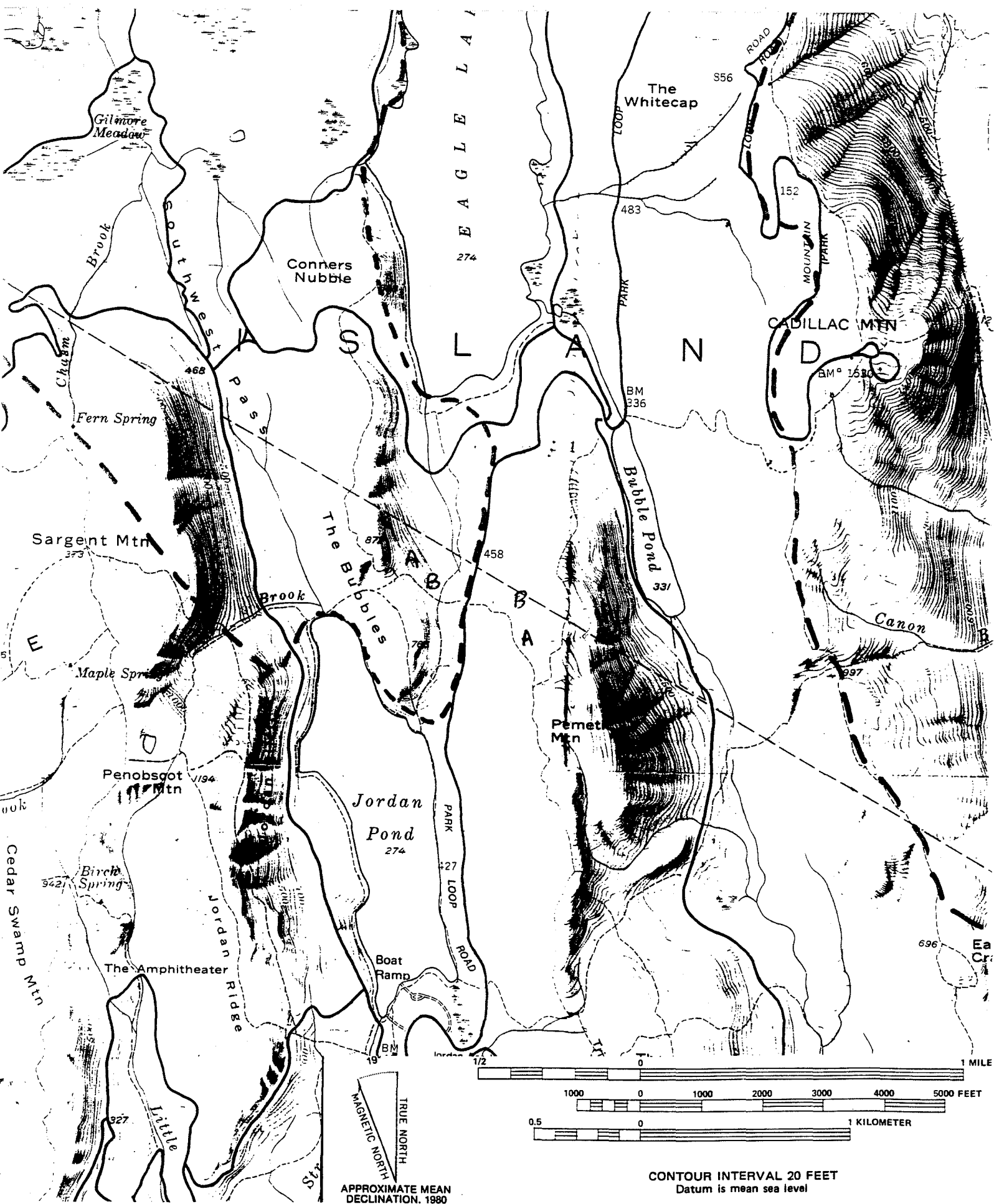


Figure 1. Map of study area, Eagle Lake watershed, Acadia National Park. Sampling locations A and B (burned and unburned), are also shown. The forest fire boundary is shown by the dashed line.

was calculated.

Soil samples were homogenized by grinding with an acid-cleaned mortar and pestle. For mercury determination, 0.25 grams of dry, homogenized soil was digested with 6 ml of 2:1 sulfuric acid : nitric acid and 0.4 ml of hydrochloric acid in a 120 ml teflon digestion vessel (Savillex Corp. #578). The digestates were allowed to stand overnight, then microwaved for 3 minutes at high power, allowed to stand for 2 minutes, then microwaved for 2 minutes at high power, allowed to stand for 10 minutes and then cooled for 30 minutes in a refrigerator. After cooling, the digestates were microwaved for 10 minutes at medium power, then cooled for 30 minutes. To oxidize any remaining organic material, 6% potassium permanganate in deionized water was added until a pink color persisted, followed by the addition of 0.5 ml of 5% potassium persulfate. The digestates were allowed to stand overnight. I then added 0.5 ml of 20% hydroxylamine hydrochloride. The digestates were transferred to 25 ml volumetric flasks and brought to volume with deionized water. Digestates were centrifuged in acid-cleaned centrifuge tubes to remove undigested material, decanted, and stored in acid-washed polyethylene bottles.

Mercury was determined by stannous chloride reduction, followed by cold vapor atomic fluorescence spectroscopy (PS

Analytical LTD model PA 20.100 random access autosampler). The method was validated by the use of calibration standards, reagent blanks, and National Bureau of Standards reference material (#1575 Pine Needles).

Results

Total mercury concentration varied considerably among samples; however, total mercury in unburned soils was significantly higher than in burned samples (Table 1), both in concentration (ng Hg/g dry wt., $P=0.004$), and on a total mercury per area basis (ng Hg/cm², $P=0.016$). The unburned soils had thicker organic horizons and a much higher percent moisture (Table 1).

Sample concentrations were corrected for background mercury concentration measured in a procedural blank (8.1 ng Hg/50 ml). Analyses of National Bureau of Standards reference material (#1575 pine needles) were within their confidence interval (mean = 171.7 ng Hg/g , $n=2$, NBS = 150 +/- 50 ng Hg/g).

RESULTS:

Burned Samples:

ID	dry wt. (g)	moisture percent	horizon thick. cm.	Hg (ng / g) dry wt.	Hg (ng / cm2)
A-1	14.104	33.2	4	300	337
A-2	22.093	35.5	5	159	280
A-3	15.397	36.4	4	154	189
B-1	9.904	36.9	3	236	186
B-2	21.915	38.2	5	307	535
B-3	7.632	44.5	3	232	141
mean				* 231	** 278
std. dev.				66	148

Unburned Samples:

ID	dry wt. (g)	moisture content	horizon thick. cm.	Hg (ng / g) dry wt.	Hg (ng / cm2)
A-1	14.474	64.3	8	515	594
A-2	10.289	59.8	6	665	544
A-3	13.334	48.5	7	468	496
B-1	17.699	56.3	8	348	490
B-2	8.97	58.7	6	768	548
B-3	9.779	47.4	7	535	416
mean				* 549	** 515
std. dev.				148	61

* Significant at P = 0.004

** Significant at P = 0.016

(Kruska-Wallis one-way analysis of variance; Systat for Windows 5.0 statistical software)

Discussion

These results suggest that forest fire decreases the total mercury burden of soil. However, a possible artifact in the data stems from comparing soil from different canopy types. This was unavoidable because of the spruce-fir climax forest that dominated the pre-fire vegetation. Soils under hardwood canopies are generally lower than at softwood sites in total mercury concentration (Abbott, 1994). Although, the differences in the literature are not as great as these results suggest. Another possible source of bias is the different slope orientations at sampling sites. The slope difference may result in varying mercury deposition rates based on the prevailing wind direction. The impact of this difference needs to be evaluated through further sampling. The large difference between organic horizon thicknesses suggests that the fire was severe enough to substantially consume part if not all of the original organic layer. Because the total mercury per area basis was significantly higher in the unburned soils than in the soils at the burned sites, I conclude that the forest fire probably reduced the original mercury burden considerably. However, spruce trees are more efficient than beech trees at collecting particulate matter (Nihlgard, 1970), and the majority of mercury deposition is associated with particulate matter. Thus, one would expect higher mercury concentrations in spruce forest

soils. In addition, deciduous trees are bare of leaves approximately one-half of the year, during which their collection of dry deposition is more inefficient. In conclusion, the lower mercury burden in the burned soils probably results from a combination of the following: reduced initial Hg burden, reduced Hg retention ability, lower dry deposition collection efficiency of the hardwood canopy, and possibly the sampling site slope orientation. The very high softwood soil Hg levels warrant further study of coastal Maine soils.

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