

Draft of the USGS report to the DEQ, State of Idaho—May 26, 2006

**Evaluation of Trends of Mercury Deposition in Salmon Falls Creek Reservoir, Idaho, using
Reservoir Sediment Cores**

By John E. Gray

U.S. Geological Survey, P.O. Box 25046, Federal Center, MS 973, Denver, CO 80225 USA email: jgray@usgs.gov

USGS Administrative Report prepared for the State of Idaho, Department of Environmental Quality, 1410 North
Hilton, Boise, ID 83607

1. Introduction

Salmon Falls Creek Reservoir (SFCR) is one of five reservoirs in Idaho that has been listed with a mercury (Hg) advisory. This reservoir has a maximum storage capacity of 230,650 acre-ft and is located in south-central Idaho, southwest of the town of Twin Falls (location: 42° 13' N, 114° 44' W), and it is used for agricultural irrigation, fishing, and recreation (Fig. 1). Fish in SFCR have been found to contain Hg concentrations exceeding the 0.5 µg/g (wet weight, fish muscle) safe level recommended by the State of Idaho and the World Health Organization. The source of Hg contamination to SFCR is presently under study by the State of Idaho. Mercury contamination of lakes, reservoirs, and other water bodies is a widespread problem in the USA and worldwide. Presently, there are over 3,000 water bodies in the USA that contain fish with Hg concentrations exceeding recommended safe levels. Consumption of fish contaminated with Hg is the dominant pathway of Hg to humans. The greatest potential concern to humans is exposure to methyl-Hg (CH₃Hg⁺), which is a neurotoxin that damages the central nervous system in humans, and is the dominant form of Hg in fish as >95 % of the total Hg in fish is methyl-Hg.

Studies of lakes and reservoirs worldwide have shown that sources of Hg contamination may be natural, anthropogenic, or a combination of sources (USEPA, 1997; Fitzgerald and others, 1998; Ullrich and others, 2001). The most common natural sources include erosion from rocks and soils or discharge from hydrothermal waters such as hot springs. Common anthropogenic sources include discharge or atmospheric emissions from mines, atmospheric emissions from the burning of fossil fuels such as coal in power plants, and releases from industrial and medical incinerators. In the region near SFCR, there are several gold mines using sulfide-mineral roasting operations in northern Nevada, which have released significant emissions of Hg and are a potential source of Hg contamination to SFCR and other water bodies

in the region. For example, four of these gold roasters emitted 13,560 pounds of Hg into the atmosphere in 1998 (Kosich, 2005), which are very high Hg emissions from point sources, and about 90 percent of the total atmospheric Hg releases for all of Nevada. Several of these gold roasters have been in operation in this area since the late 1990's (Henetz, 2005). In addition, a Mercury Depositional Network (MDN) site in northern Nevada, located about 55 km southwest of SFCR (Fig. 1), and located between the northern Nevada gold mines and SFCR, has reported high concentrations of Hg in precipitation (in some instances exceeding 100 ng/L in 2003), which are among the highest reported by the MDN in the USA. There are no mines in Salmon Falls Creek basin, and thus, mine runoff is not a Hg issue locally. In addition to point sources of Hg in this region, it is possible the Hg entering lakes and reservoirs in the area is related to global cycling of atmospheric Hg, which is mainly due to worldwide burning of fossil fuels and is known to be a common Hg contaminant of many water bodies (Fitzgerald and others, 1998).

In order to trace a contaminant source through time in reservoir or lake sediment cores the contaminant must be in a form with a long-term stability. In the case of Hg, one of the most stable forms is HgS (mercury sulfide), which is highly resistant to physical and chemical weathering. The dominant form of Hg in the atmosphere and that emitted from mining roasters and from the burning of fossil fuels is Hg⁰, but there is also a much lesser proportion of reactive ionic Hg such as Hg²⁺ in the atmosphere (Krabbenhoft and Rickert, 1995; Fitzgerald and others, 1998; USEPA, 1997; Gray, 2003). These atmospheric forms of Hg enter watersheds or water bodies primarily through wet deposition (precipitation), but a minor component is dry deposition (dust and particulates containing Hg, Fig. 2). If the Hg entering a water body is Hg⁰, it is generally unreactive and may volatilize back into the atmosphere (Fig. 2). However, some portion of Hg⁰ may be oxidized to Hg²⁺ or converted to a stable form such as HgS and settled

out of the water column along with sediment. Ionic Hg^{2+} , or particulate Hg ($\text{Hg}_{(p)}$, Fig. 2) containing Hg^{2+} , that enters a water body is reactive and is generally converted to methyl-Hg by microbial activity. Methyl-Hg is bioavailable and is rapidly taken up by fish and other biota in the ecosystem (Fig. 2). Thus, when ecosystems such as lakes and reservoirs are exposed to elevated concentrations of bioavailable Hg, formation of methyl-Hg may result, which subsequently leads to high concentrations of Hg in fish and fish consumption advisories.

2. Objectives

The State of Idaho (Department of Environmental Quality, Boise) under a Joint Funding Agreement contracted the U.S. Geological Survey (USGS, John E. Gray, Denver) to collect sediment cores from SFCR, analyze these cores for Hg, and date sections of the core for time of deposition. The primary objective of the sediment coring was to evaluate historical depositional trends of Hg in the reservoir cores and relate any trends found to possible sources with high concentrations of Hg that have influenced Hg input into the reservoir over time. Three sediment cores were collected from SFCR with the stated objectives of (1) dating sections of the cores to provide a time distribution within sections of the cores, (2) analyze the sections of the cores for total Hg concentrations to understand the deposition of Hg throughout the time of sediment deposition in the reservoir, and (3) relate the patterns of Hg deposition in the past in SFCR to possible sources of Hg, which have potentially led to Hg contamination in this reservoir. SFCR is about 95 years old and sediment deposition in the reservoir over this time should provide the ability to evaluate patterns of Hg deposition through time. Sediment coring and geochemical analysis is a technique that offers the possibility to identify past Hg-bearing sediment loads into a reservoir as a function of time (depth in the cores). At USGS expense, total Hg and methyl-Hg

were measured in eight water samples collected from SFCR to evaluate the proportion of methyl-Hg/total Hg in water from the reservoir.

3. Methods

3.1 Sediment coring

Sediment cores were collected from three locations in SFCR using a push corer with a 10.1-cm diameter polycarbonate barrel (Fig. 3). Two cores (1SFCR and 2SFCR) were collected in June 2005 (Fig. 4; Tables 1 and 2). In October 2005, one additional core (14SFCR) was collected (Fig. 4; Table 3). All cores were kept vertical and anaerobic until they were extruded. All of these cores were extruded and sectioned at 2 cm intervals the day following collection. During extrusion, representative splits were obtained by quartering each core section using a plastic spatula and splits were obtained for (a) dating by the ^{137}Cs method (1/2 of the section), (b) total Hg analysis (1/4 of the section), and (c) other geochemical determinations (1/4 of the section). All samples were stored in sealed glass vessels with Teflon lined lids until analyzed. Samples for Hg analyses were frozen following core extrusion and remained so until analyzed. Samples for dating were weighed following core splitting, air dried, and reweighed to obtain bulk densities. Volumes used in bulk density calculations were obtained using the inside diameter of the core barrel and the increment of the core extruded for each sample. Dried samples for dating and other geochemical determinations were pulverized prior to analysis. Sediment ^{137}Cs activities were measured on dried and pulverized sediment samples in a fixed-geometry vessel with a high resolution, intrinsic germanium detector gamma-ray spectrometer. Radiometric counting errors were generally about 10%, but in the samples of the peak ^{137}Cs activity were about 2%. Samples were dated using isotopic methods by the USGS (Center for

Coastal Studies, St. Petersburg, FL). Particle size determinations were made using a Model LS 2000 Coulter Counter on each sample split prior to dating. These grain size determinations were used to relate core stratigraphy to hydrologic discharge records from Salmon Falls Creek; for instance, coarse grained sediment layers generally correlate with periods of high precipitation and high runoff.

The concentration of total Hg was determined in the sediment core samples following total digestion using aqua regia as outlined in EPA Method 1631, Appendix A (USEPA, 1996). The Hg ions in the digestate were reduced by acidic SnCl_2 to elemental-Hg and purged from the sample with argon. The released Hg was measured by cold vapor atomic absorption (USEPA, 1996). Quality control for total Hg analysis was established using method blanks, sample matrix spikes, standard reference materials (SRM's), and sample duplicates. Recoveries for total Hg on sample matrix spikes were 80-113 %. The relative percent difference in sample duplicates was \leq 20 % for total Hg. For IAEA 405, the SRM analyzed in this study, recoveries ranged from 96-113 % of the certified value for total Hg. The lower limit of determination for total Hg was 8.0 ng/g. Method blanks were below the lower limit of determination for total Hg. Total Hg analysis was carried out under a USGS contract with Battelle Marine Sciences Laboratory (Sequim, WA).

Sediment samples from cores 1SFCR and 2SFCR were also analyzed for 42 elements (including Nb, Sc, and V used in this study, Tables 1 and 2) by inductively coupled plasma-mass spectrometry (ICP-MS). These sediment samples were decomposed using a mixture of hydrochloric, nitric, perchloric, and hydrofluoric acids at low temperature. An aliquot of the digested sample was aspirated into the ICP-MS, where concentrations of the optimal elements from the ICP-MS were determined. The ICP-MS is calibrated with aqueous standards and

internal standards are used to compensate for matrix effects and internal drift. Data is deemed acceptable if recovery for all 42 elements is $\pm 15\%$ at five times the lower limit of determination and the calculated relative standard deviation (RSD) of duplicate samples is no greater than 15%. The ICP-MS analyses were carried out at the USGS (Denver, CO). Only data for Nb, Sc, and V determined by ICP-MS are discussed in this study.

3.2 Water samples

Unfiltered water samples were collected from SFCR in June 2005. Unfiltered water is the most important sample type for water because comparative reference standards for Hg such as national and international drinking water and aquatic life standards are established for unfiltered water (WHO, 1971; USEPA, 1992); there are no clearly established standards for Hg in filtered water. Thus, filtered water samples were not collected in this study. Eight unfiltered water samples were collected from the reservoir to evaluate the variation of total Hg and methyl-Hg concentrations. Unfiltered water samples for total Hg and methyl-Hg analysis were collected in Teflon bottles pre-cleaned by boiling in concentrated HNO_3 for 48 hours, and within eight hours of collection, the samples were acidified with ultra-pure HCl.

Measurement of total Hg in water samples followed EPA Method 1631 (USEPA, 1996; USEPA, 2002) and analysis of methyl-Hg followed EPA draft Method 1630 using cold-vapor atomic fluorescence. All samples were distilled to separate methyl-Hg from the water matrix (Horvat and others, 1993). An ethylating agent was added to each sample to form a volatile methyl-ethylmercury derivative, and then purged onto graphite carbon traps as a means of preconcentration and interference removal. The sample was then isothermally chromatographed, pyrolytically broken down to elemental Hg, and detected using cold vapor fluorescence (Bloom,

1989). Quality control for Hg and methyl-Hg determinations in water was addressed with method blanks, blank spikes, matrix spikes, standard reference materials (SRM's), and sample duplicates. Recoveries on blank and matrix spikes were 95-102 % for total Hg and 91-110 % for methyl-Hg. The relative percent difference in sample duplicates was ≤ 6 % for total Hg and 1 % for methyl-Hg. For the SRM, NIST 1641 analyzed in this study, total Hg was ± 13 % of the certified concentration. There is no SRM for methyl-Hg in water, but for samples of the tissue SRM-DORM-2 analyzed with the water samples, methyl-Hg was ± 6 % of the certified concentration. Field and method blanks were below the limits of determination for total Hg. Lower limits of determination were 0.2 ng/L for total Hg and 0.02 ng/L for methyl-Hg in water samples. Total Hg and methyl-Hg analysis of water was carried out by Battelle Marine Sciences Laboratory (Sequim, WA). Water parameters such as pH, conductivity, total dissolved solids, alkalinity, turbidity, temperature, Fe^{2+} , oxidation-reduction potential, and dissolved oxygen were also measured at each sample site using portable meters, field test kits, or a Hydrolab instrument.

4. Results and Discussion

Several sediment cores were collected from SFCR during fieldwork in 2005, but of these, only three were retained for analysis as required by the agreement of the USGS-DEQ contract. The first two cores (1SFCR and 2SFRC) were collected from the south end of the reservoir where the water was shallow and the locations were close to the inlet where stream water from Salmon Falls Creek feeds into the reservoir. The third core (14SFCR) was collected more centrally within SFCR, near the mouth of Corral Creek, to obtain a core in an area of the reservoir with consistent sedimentation through time (Table 3).

Core 1SFCR (location: 42.06962° N, 114.76476° W) was about 90 cm thick and consisted of mostly silt and clay, except for the bottom 1.5 cm, which was coarse sand and gravel that may be an old channel deposit of Salmon Falls Creek. Core 2SFCR (location: 42.07374° N, 114.76372° W) was about 86 cm thick and consisted of mostly silt for the top 18 cm, followed by 34 cm of silt and fine sand, and then 34 cm of mixed silt and clay. Core 14SFCR (location 42.09611° N, 114.75383° W) was 125 cm thick and consisted of numerous layers of silt and sand throughout the core.

For core 14SRCR, the ^{137}Cs data and stratigraphic variation (grainsize) can be correlated with time markers (Fig. 5) and this core has the most reliable time pattern of the three cores collected. In this core, important time markers are as follows (a) a prominent ^{137}Cs peak activity, which relates to the peak ^{137}Cs deposition of radioactive fallout in 1963 in the USA (Holmes, 1998), (b) the initial presence of ^{137}Cs activity that relates to the beginning of radioactive fallout in 1952, (c) the end of significant ^{137}Cs deposition that corresponds to about 1970, (d) several stratigraphic markers that consist of layers with abundant sand that correspond to periods of high precipitation and high annual discharge from Salmon Falls Creek in 1997, 1995, 1984, 1971, and 1943 (Fig. 5, Table 4), and (e) the top of the core, which obviously corresponds to the time of collection in 2005. Using these time markers, dates were assigned to the remainder of the samples sectioned from the core, sedimentation rates were calculated, and Hg profiles were generated that relate the deposition of Hg with time.

For cores 1SFCR and 2SFCR there were clearly significant time gaps in the chronology of these cores making them much more difficult to interpret. In both cores, the past 8 years (1997-2005) show periods of nondeposition of sediment, which is consistent with local observations indicating that the area where these cores were collected from SFCR was dry for most of this

time period, which was mostly due to drought and low average annual discharge from Salmon Falls Creek. In both of these cores, ^{137}Cs depositional markers (a-c described above for core 14SFCR) are also evident, but few other stratigraphic markers relating time to grainsize were found. The lack of stratigraphic markers was especially evident in core 1SFCR, where there is little variation in grainsize. In core 2SFCR, the ^{137}Cs peak depositional marker is found only 16 cm below the top of the core and very high in the core section (whereas in 14SFCR this marker is found 76 cm below the core top), further suggesting that there are periods of nondeposition in this core. These two cores have a less reliable chronological record than 14SFCR.

4.1 Variation of Hg concentration with time

The range of Hg concentration (8.5-58 ng/g) in samples of core 14SFCR is generally low to average when compared to Hg found in studies of lakes worldwide (Lockhart and others, 1998; Heyvaert and others, 2000; Yang and others, 2002; Yang and Rose, 2003). The average Hg concentration for samples from 14SFCR is 28 ng/g (Fig. 6). The two most prominent spikes in Hg concentration, which exceed the average Hg concentration for core 14SFCR, correlate to (1) the period from about 1955-1971, where Hg concentrations varied from 34-58 ng/g, and (2) the period between about 1931-1934, where Hg concentrations varied from 31-37 ng/g (Fig. 6). The average concentration of total Hg in 14SFCR is generally lower than that for the upper continental crust, which averages 56 ng/g (Wedepohl, 1995), and is also somewhat lower than the mean total Hg concentration of 58 ng/g in soil collected throughout the USA (Shacklette and Boerngen, 1984). There are few Hg data for rocks in the SFCR area, but 75 soil samples collected in the Twin Falls Quadrangle, Idaho, and the Elko Quadrangle, Nevada, contained Hg concentrations ranging from <20-100 ng/g, averaging about 22 ng/g (USGS, 2004). The Hg

concentrations in these soil samples are similar to those found in the SFCR sediment core samples collected in this study.

Cores 1SFCR and 2SFCR also show two prominent Hg concentration peaks. In core, 1SFCR, Hg concentration varies from 11-52 ng/g, averaging 34 ng/g (Fig. 7). This core shows higher than average Hg concentrations ranging from 37-52 ng/g for the period from 1963-1976, and there is a shorter duration spike from 1946-1948, where the Hg concentrations ranged 37-38 ng/g (although these concentrations are within analytical error of the average Hg concentration of this core). In core 2SFCR, Hg concentrations vary from 13-44 ng/g, averaging 23 ng/g (Fig. 8). This core shows higher than average Hg concentrations ranging from 39-47 ng/g for the period from 1958-2005, and from 1919-1921, there is short duration spike where the Hg concentration was 37 ng/g. It is not meaningful to relate these patterns of Hg concentrations to temporal periods without calculating Hg accumulation rates for these cores.

4.2 Mercury Accumulation Rates

Using core sedimentation rates, Hg concentrations, and the bulk density of each sample sectioned from each core, Hg accumulation rates or Hg fluxes ($\text{ng}/\text{cm}^2/\text{yr}$) were calculated. These Hg accumulation rates are used because it is difficult to evaluate historical Hg deposition based only on sediment Hg concentration from a given section of a sediment core. Sedimentation rates can vary significantly in various areas of a given reservoir and throughout time, thus, affecting the overall accumulation of Hg in any period of time. Therefore, Hg accumulation rates are calculated to more accurately evaluate Hg deposition in a given area in a certain time period. The Hg accumulation rate can be calculated as follows:

$$\text{Hg accumulation rate} = \rho (s) \text{Hg}_{(T)}$$

Where ρ is the sediment bulk density in g/cm^3 , s is the sediment accumulation rate in cm/yr , and $\text{Hg}_{(T)}$ is the concentration of total Hg in ng/g .

In core 14SFCR, the average Hg accumulation rate is $55 \text{ ng}/\text{cm}^2/\text{yr}$ (Fig. 6). This core shows three periods when the Hg accumulation rate significantly exceeded this average including (1) from 1964-1971 when Hg accumulation rates varied from $75\text{-}149 \text{ ng}/\text{cm}^2/\text{yr}$, (2) in 1955 and 1956 when the Hg accumulation rates varied from $66\text{-}76 \text{ ng}/\text{cm}^2/\text{yr}$, and (3) from 1932-1934 when it varied from $66\text{-}72 \text{ ng}/\text{cm}^2/\text{yr}$. These periods of increased Hg accumulation generally correspond to periods of above average Hg concentration and/or sedimentation rates, especially for the period from 1964-1971. There were several years of above average annual discharge from Salmon Falls Creek during these periods of time based on historical data (Table 4; USGS, 2006). This Hg accumulation pattern for 14SFCR suggests that the primary source of Hg in this case is lithologic (often referred to as a geogenic source, or geologically derived versus anthropogenically derived). The Hg accumulation pattern for 14SFCR is interpreted to be related to erosion from upstream soil and bedrock geologic units in Salmon Falls Creek basin during periods high precipitation and runoff. Such erosion has resulted in high sedimentation rates and high Hg accumulation rates during these periods. Periods of high Hg accumulation are unlikely to be related to deposition of Hg from a source outside of Salmon Falls Creek basin, such as atmospheric Hg, or if there is a such an external source, it is of low Hg concentration compared to that in sediments eroding into SFCR and cannot be resolved in these sediment core data.

Patterns of Hg accumulation in the other two sediment cores collected from SFCR (1SFCR and 2SFCR) show no consistent or diagnostic trends of Hg deposition in this reservoir with time (Figs. 6 and 7). In core 1SFCR, the average Hg accumulation rate is 48 ng/cm²/yr (Fig. 7). From 1982-1995, the Hg accumulation rate, ranging from 63-68 ng/cm²/yr, generally exceeds the average for 1SFCR. In addition, from 1952-1975 there are several short duration periods of increased Hg accumulation rates varying from 54-66 ng/cm²/yr. Core 2SFCR has a complete time section dating back to 1912, when the reservoir was constructed, but shows less variation of Hg accumulation rate with time. The average Hg accumulation rate is 36 ng/cm²/yr in core 2SFCR and the highest Hg accumulation rates were from 1912-1921, ranging from 45-77 ng/cm²/yr (Fig. 8). As mentioned above, both of these cores had several time gaps or periods of nondeposition from about 1980-2005, and therefore, the Hg accumulation rates for this period are incomplete and difficult to interpret. Patterns of increased Hg accumulation in cores 1SFCR and 2SFCR are inconsistent with any known point source Hg contaminant during these periods. Similar to core 14SFCR, periods of increased Hg accumulation rate in cores 1SFCR and 2SFCR are also interpreted to be related to upstream lithologic erosion in Salmon Falls Creek basin as a result of periodic high precipitation and high sedimentation rates. These Hg accumulation rates data do not suggest influence from an additional external contaminant source of Hg from outside of the Salmon Falls Creek basin.

The Hg accumulation rates determined for SFCR are generally higher than those reported for studies of lakes in the northern hemisphere, ranging from 0.2-38 ng-Hg/cm²/yr (Swain and others, 1992; Lockhart and others, 1998; Heyvaert and others, 2000; Yang and Rose, 2003). Many of these lakes are also known to contain fish with elevated Hg concentrations and atmospheric deposition is reported to be the most significant source of Hg. However, reservoirs

worldwide have been reported with higher Hg accumulation rates than natural lakes because sedimentation rates are generally much higher in reservoirs, resulting in much higher reservoir Hg accumulation rates (Gray and others, 2005). The Hg accumulation rates for SF CR are higher than those in most natural lakes as a result of the generally high sedimentation rates in SF CR, which range from 0.4-3.33 cm/yr.

4.3 Geochemical Enrichment Factors

Another geochemical method to evaluate enrichment or depletion of an element in an open system is to calculate enrichment factors (EF's). This method can be used to quantify element redistribution in a soil or sediment column and/or integrated over time to evaluate the enrichment or depletion of an element of interest (Kurtz and others, 2000). Using this approach, the element of interest (Hg in this case) is ratioed against an element that is generally insoluble and immobile (elements such as Nb, Zr, V, or Sc; Kurtz and others, 2000; Hissler and Probst, 2006). In this study, the immobile elements used for comparison were Nb, V, and Sc. The resultant EF's indicate the changes in concentrations relative to that in an uncontaminated baseline sample. For this study, the baseline chosen was sediment collected from deep within the core that is assumed to predate any significant local or regional Hg contaminant point source, and thus, represents the upstream basin lithologic geochemical signature (for core 1SF CR this sample corresponds to 1948; for core 2SF CR this sample corresponds to 1913). This calculation is as follows:

$$EF = (C_{Hg, x} / C_{Hg, b}) \times (C_{i, b} / C_{i, x}) - 1$$

Where EF is the enrichment (or depletion) of Hg, $C_{\text{Hg},x}$ is the concentration of Hg in the core sediment sample of interest, $C_{\text{Hg},b}$ is the concentration of Hg in the baseline core sediment sample, $C_{i,b}$ is the concentration of the immobile element (Nb, V, or Sc) in the baseline core sediment sample, and $C_{i,x}$ the concentration of the immobile element (Nb, V, or Sc) in the core sediment sample of interest.

The EF's calculated for Hg in cores 1SFCR and 2SFCR show little variation, all varying between 0.71 and -0.68 (Figs. 9 and 10). Studies have suggested that EF's < 1 are not significant, whereas EF's in excess of 2 have been related to anthropogenic contamination (Kurtz and others, 2000; Hissler and Probst, 2006). The EF's for both SFCR cores show similar patterns for Nb, V, and Sc. Similar to the Hg accumulation rates for SFCR, the EF's shown here suggest Hg in the reservoir core sediments is dominantly derived from upstream lithologic sources in the Salmon Falls Creek basin because they do not vary significantly from the EF calculated for the lithologic baseline.

4.4 Water samples

During fieldwork in June 2005, a relatively small number of samples ($n = 8$) were collected and analyzed for total Hg and methyl-Hg to evaluate the general concentration of these Hg species in the water column of SFCR. A number of on-site geochemical parameters were also collected with portable instruments at the sites of water collection. At one location (42.06962°N , $114.76476^{\circ}\text{W}$), a series of four water samples were collected with increasing depth (Table 5). These water samples show some noteworthy trends: (1) Total Hg concentrations (3.1-4.4 ng/L) in the water samples collected from SFCR are significantly below the international drinking water standard of 1,000 ng/L for Hg established by the World Health

Organization and also below the 2,000 ng/L drinking water standard for Hg used in the U.S. (WHO, 1971; USEPA, 1992). The total Hg concentrations in water from the reservoir are also below the 12 ng/L standard for total Hg recommended by the USEPA to protect against adverse chronic effects to aquatic life (Fig. 11). Although the total Hg concentrations in the SFCR water samples are lower than the stated standards, it is clear that there is enough bioavailable Hg to produce elevated Hg concentrations in fish. (2) Total Hg, but especially methyl-Hg concentrations, in these water samples is higher than those reported for many other reservoirs and lakes worldwide as well as those in uncontaminated global baselines (Fig. 11). (3) The ratio of methyl-Hg/total Hg varies up to about 40% and is highly elevated indicating a high proportion of methyl-Hg in the water column, whereas the ratio of methyl-Hg/total Hg in water typically ranges from 1-10% even for water bodies with Hg contamination (Ullrich and others, 2001; Gray and others, 2000). (4) There is increasing methyl-Hg concentration (0.76-1.8 ng/L) with increasing water depth, and in addition, dissolved oxygen, pH, temperature, and ORP (oxidation-reduction potential) all decrease with increasing water depth, which are conditions that tend to favor methylation of Hg (Table 5).

These water data are too few to draw any significant conclusions and such data cannot point to a source of the Hg affecting SFCR. However, the water data collected thus far indicate potentially high rates of Hg methylation in this reservoir either in the water column, or in the sediment column, which is then transferred to water. Methyl-Hg formed in an aquatic system such as a lake or reservoir is generally transferred rapidly to biota such as fish. These methyl-Hg water data suggest that Hg is entering SFCR in a bioavailable form, probably ionic Hg^{2+} , which is then rapidly converted to methyl-Hg. Clearly, more water data are needed from SFCR to more thoroughly understand Hg methylation and Hg cycling in this reservoir.

5. Conclusions

In the past few years, there has been considerable attention directed at Hg emissions from processing roasters at gold mines in northern Nevada, which have reported significant emissions of Hg since the late 1990's. The Hg accumulation rates presented here for sediment cores collected from SFCR do not indicate a correlation in time to Hg emissions from this point source. For example, in core 14SFCR for the period from about 1990-2005, the Hg accumulation rates are generally below average as compared to that for the remainder of this core. The Hg accumulation rates for SFCR suggests that the dominant contribution of Hg to the sediments in this reservoir is from lithologic sources, such as soil and bedrock, derived upstream from Salmon Falls Creek basin. Enrichment factors calculated for Hg/Nb, Hg/V, and Hg/Sr also indicate that the dominant portion of the Hg in SFCR is from a lithologic source and do not indicate any significant enrichment in Hg with time.

Periods of increased Hg accumulation rates, primarily from about 1964-1971, are interpreted to be related to periods of high precipitation and high discharge from Salmon Fall Creek resulting in high sedimentation rates and correspondingly high Hg accumulation rates. It is unlikely that increases in Hg accumulation rates are related to deposition from any significant external source of Hg, such as atmospheric Hg, or if there is an atmospheric component, it cannot be identified in the sediment core data for SFCR. If atmospheric Hg is affecting SFCR, it is possible that (1) such atmospheric Hg is in a highly reactive form such as Hg²⁺, which is then converted to methyl-Hg, rapidly taken up by fish, and as a result, this Hg is not removed with sediment during sedimentation in SFCR, (2) there is little atmospheric Hg contribution to SFCR or the atmospheric Hg contribution to SFCR is in a much lower Hg concentration relative to that

in the reservoir sediment, and thus, any atmospheric Hg is a lower contribution compared to the sediment load in SFCR, or (3) the atmospheric Hg contribution in this region is of low concentration and the source of Hg to SFCR is from global cycling of Hg. Regardless, a significant point source of Hg was not identified in the sediment cores collected from SFCR in this study.

References:

- Bloom, N.S., 1989, Determination of Picogram Levels of Methylmercury by Aqueous Phase Ethylation, Followed by Cryogenic GC with CVAf Detection, *Canadian Journal of Fish Aquatic Science*, v. 7, p. 1131
- Fitzgerald, W.F., Engstrom, D.R., Mason, R.P., Nater, E.A., 1998, The case for atmospheric mercury contamination in remote areas, *Critical Reviews Environmental Science and Technology*, v. 32, no.1, p. 1-7.
- Gray, J.E., 2003, Geologic studies of mercury by the U.S. Geological Survey, U.S. Geological Survey Circular 1248, 35 p.
- Gray, J.E., Fey, D.L., Holmes, C.W., and Lasorsa, B.K., 2005, Historical deposition and fluxes of mercury in Narraguinnep Reservoir, southwestern Colorado, USA, *Applied Geochemistry*, v. 20, p. 207-220.
- Gray, J.E., Theodorakos, P.M., Bailey, E.A., and Turner, R.R., 2000, Distribution, speciation, and transport of mercury in stream sediment, stream water, and fish collected near abandoned mercury mines in southwestern Alaska, U.S.A., *Science of the Total Environment*, v. 260, p. 21-33.
- Heyvaert, A.C., Reuter, R.E., Slotton, D.G., Goldman, C.R., 2000, Paleolimnological reconstruction of historical atmospheric lead and mercury at Lake Tahoe, California—Nevada, *Environmental Science Technology*, v. 34, p. 3588-3597.
- Henetz, P., 2005, Utah hopes Idaho's in-depth study sheds light on mercury problem. *Salt Lake City Tribune*, June 6, 2006, http://www.sltrib.com/search/ci_2779013.
- Hissler, C., Probst, J.L., 2006, Impact of mercury atmospheric deposition on soils and streams in a mountainous catchment (Vosges, France) polluted by chlor-alkali industrial activity: The import trapping role of the organic matter, *Science of the Total Environment*, v. 361, p. 163-178.
- Holmes, C.W., 1998, Short-lived isotopic chronometers—A means of measuring decadal sedimentary dynamics, U.S. Geological Survey Fact Sheet FS-073-98, 2 p.
- Horvat, M., Bloom, N., and Liang, L., 1993, A Comparison of Distillation with other Current Isolation Methods for the Determination of Methyl Mercury Compounds in Low Level Environmental Samples Part 2, Water, *Analytica Chimica Acta*, v. 282, p. 153-168.
- Kosich, D., 2005, Nevada blazes trail to control airborne mercury, *Mineweb*, http://www.mineweb.net/sections/whats_new/654844.htm
- Krabbenhoft, D.P., and Rickert, D.A., 1995, Mercury contamination of aquatic ecosystems, U.S. Geological Survey Fact Sheet FS-216-95.
- Kurtz, A.C., Derry, L.A., Chadwick, O.A., and Alfano, M.J., 2000, Refractory element mobility in volcanic soils, *Geology*, v. 28, p. 683-686.
- Lockhart, W.L., Wilkinson, P., Billeck, B.N., Danell, R.A., Hunt, R.V., Brunskill, G.J., Delaronde, J., St. Louis, V., 1998, Fluxes of mercury to lake sediments in central and northern Canada inferred from dated sediment cores, *Biogeochemistry*, v. 40, p. 163-173.
- Loseto, L.L., Lean, D.R.S., Siciliano, S.D., 2004, Snowmelt sources of methylmercury to high arctic ecosystems, *Environmental Science and Technology*, v. 38, p. 3004-3010.
- Lyons, W.B., Welch, K.A., Bonzongo, J.C., 1999, Mercury in aquatic systems in Antarctica, *Geophysical Research Letters*, v. 26, p. 2235-2238.

- Seiler, R.L., Lico, M.S., Wiemeyer, S.N., and Evers, D.C., 2004, Mercury in the Walker River basin, Nevada and California—Sources, distribution, and potential effects on the ecosystem, U.S. Geological Survey, Scientific Investigations Report 2004-5147, 24 p.
- Shacklette, H.T., and Boerngen, J.G., 1984, Element concentrations in soils and other surficial materials of the conterminous United States, U.S. Geological Survey Professional Paper 1270, 105 p.
- Swain, E.B., Engstrom, D.R., Brigham, M.E., Henning, T.A., Brezonik, P.L., 1992, Increasing rates of atmospheric deposition in midcontinental North America, *Science*, v. 257, p. 784-787.
- Ullrich, S.M., Tanton, T.W., Abdrashitova, S.A., 2001, Mercury in the aquatic environment: A review of factors affecting methylation, *Critical Reviews Environmental Science and Technology*, v. 31, no. 3, p. 241-293.
- USEPA, 1992, Water quality standards; establishment of numeric criteria for priority toxic pollutants; states' compliance; final rule, *Federal Register*, 40 CFR Part 131, 57/246, p. 60,847-60,916.
- USEPA, 1996, Test Methods for Evaluating Solid Waste, US Environmental Protection Agency, Office of Solid Waste and Emergency Response, Washington, D.C. (SW846)
- USEPA, 1997, Mercury study report to Congress, Volumes I-VIII, US Environmental Protection Agency Report EPA-452/R-97-003.
- USEPA, 2002, Method 1631e, Mercury in Water by Oxidation, Purge and Trap, and Cold Vapor Atomic Fluorescence Spectrometry, EPA 821-R-02-019, August 2002.
- USGS, 2004, The national geochemical survey—Database and documentation, U.S. Geological Survey Open-file Report 2004-1001, <http://tin.er.usgs.gov/geochem/doc/home.htm>.
- USGS, 2006, Calendar year stream flow statistics for Idaho, Salmon Falls Creek near San Jacinto, Nevada. U.S. Geological Survey, Water Resources Division, website accessed 5/15/2006, http://nwis.waterdata.usgs.gov/id/nwis/annual/calendar_year/?site_no=13105000&agency_cd=USGS
- Wedepohl, K.H., 1995, The composition of the continental crust. *Geochimica et Cosmochimica Acta*, v. 59, p. 1217-1232.
- WHO, 1971, International standards for drinking water, 3rd ed., World Health Organization, Geneva, Switzerland.
- Yang, H., and Rose, N.L., 2003, Distribution of mercury in six lake sediment cores across the UK, *Science of the Total Environment*, v. 304, p. 391-404.
- Yang, H., Rose, N.L., Battardee, R.W., and Boyle, J.F., 2002, Mercury and lead budgets for Lochnagar, a Scottish mountain lake and its catchment, *Environmental Science and Technology*, v. 26, p. 1383-1388.

Table 1. Dates and geochemical data for core 1SFCR.

Sample #	Date	Hg (ng/g)	Mean Grainsize (mm)	% Sand	Bulk Density (g/cm ³)	Sed. Rate (cm/yr)	Hg Flux (ng/cm ² /yr)	Nb ug/g	Sc ug/g	V ug/g
1-SFCR-1	2005	31	0.033	15.5	1.04	2	64	33	8.4	45
1-SFCR-2	1998	31	0.035	16.6	0.87	2	54	33	8.3	45
1-SFCR-3	1997	33	0.035	17.0	0.79	2	52	34	8.5	45
1-SFCR-4	1996	35	0.037	17.7	0.66	2	46	34	8.3	44
1-SFCR-5	1995	34	0.04	20.7	1.00	2	68	34	8.7	46
1-SFCR-6	1994	30	0.054	31.2	0.87	2	52	33	8.2	44
1-SFCR-7	1993	35	0.049	26.8	0.90	2	64	34	8.5	45
1-SFCR-8	1992	36	0.039	19.2	0.77	2	55	33	8.4	45
1-SFCR-9	1991	36	0.034	16.1	0.88	2	64	35	8.2	45
1-SFCR-10	1990	32	0.07	38.3	0.87	2	56	36	8.2	43
1-SFCR-11	1989	30	0.025	9.8	1.00	2	61	33	7.8	41
1-SFCR-12	1988	34	0.05	30.1	0.83	2	56	34	8.2	43
1-SFCR-13	1987	33	0.045	24.4	0.86	2	57	34	8.3	43
1-SFCR-14	1986	34	0.029	13.5	0.88	2	60	35	9.2	48
1-SFCR-15	1985	32	0.013	5.8	0.82	2	52	35	10	51
1-SFCR-16	1984	32	0.043	22.1	0.86	2	56	31	8.8	47
1-SFCR-17	1983	35	0.038	19.8	0.92	2	64	32	8.5	46
1-SFCR-18	1982	33	0.025	9.7	0.94	2	63	31	8.7	46
1-SFCR-19	1980	35	0.022	8.2	0.96	1.3	44	34	8.7	46
1-SFCR-20	1978	35	0.026	10.8	0.95	1.3	44	34	8.7	46
1-SFCR-21	1976	41	0.028	11.6	0.76	1.3	40	34	8.7	46
1-SFCR-22	1975	52	0.035	16.6	0.88	1.3	60	33	9.2	49
1-SFCR-23	1974	44	0.04	20.8	0.79	1.3	45	32	9.3	49
1-SFCR-24	1972	42	0.021	8.0	0.82	1.3	45	32	9.8	51
1-SFCR-25	1971	40	0.024	9.2	0.84	1.3	44	31	9.3	50
1-SFCR-26	1970	42	0.017	4.9	0.95	1.3	51	32	9.3	50
1-SFCR-27	1968	49	0.021	7.2	0.94	1.3	60	33	9.2	48
1-SFCR-28	1966	38	0.032	14.9	0.88	1.3	44	33	9.1	47
1-SFCR-29	1964	37	0.028	12.5	0.81	1.3	39	32	9.2	48
1-SFCR-30	1963	37	0.022	8.5	0.89	2	66	32	9.2	50
1-SFCR-31	1961	32	0.027	12.4	0.93	1	30	33	9.2	49
1-SFCR-32	1959	33	0.042	22.0	0.92	1	30	32	9.1	48
1-SFCR-33	1957	33	0.038	19.0	0.98	1	33	32	9.2	50
1-SFCR-34	1956	32	0.025	10.6	0.94	2	61	32	9.2	50
1-SFCR-35	1955	33	0.025	10.4	0.76	2	50	31	9.1	50
1-SFCR-36	1953	34	0.033	15.2	0.98	1	34	29	9	48
1-SFCR-37	1952	31	0.024	8.4	0.87	2	54	32	9	48
1-SFCR-38	1950	31	0.03	11.9	0.99	1	31	30	9.3	50
1-SFCR-39	1948	38	0.023	7.8	0.86	1	33	32	9.3	51
1-SFCR-40	1946	37	0.029	11.6	1.07	1	39	31	9.1	49
1-SFCR-41	1944	26	0.04	20.3	1.40	1	36	32	9	49
1-SFCR-42	1942	20	0.045	24.3	1.44	1	28	31	8.6	47
1-SFCR-43	1940	26	0.043	22.3	1.08	1.25	35	31	8.2	46
1-SFCR-44	1938	28	0.11	42.9	0.80	1.25	28	32	8.2	45
1-SFCR-45	1936	11	0.28	69.3	0.98	1.25	14	33	5.4	28

Table 2. Dates and geochemical data for core 2SFCR.

Sample #	Date	Hg (ng/g)	Mean Grainsize (mm)	% Sand	Bulk Density (g/cm ³)	Sed. Rate (cm/yr)	Hg Flux (ng/cm ² /yr)	Nb (µg/g)	Sc (µg/g)	V (µg/g)
2 SFCR-1	2005	39	0.045	20	0.46	2.0	36	33	8.9	47
2 SFCR-2	1997	37	0.049	22	0.89	1.0	33	30	8.9	46
2 SFCR-3	1984	40	0.043	23	0.79	1.0	32	36	11	57
2 SFCR-4	1980	47	0.043	22	0.80	1.0	38	30	11	58
2 SFCR-5	1975	39	0.035	17	0.90	1.0	35	37	12	61
2 SFCR-6	1971	42	0.074	39	0.89	1.0	37	32	11	59
2 SFCR-7	1965	44	0.043	23	0.93	1.0	41	30	11	60
2 SFCR-8	1963	44	0.036	18	0.95	1.0	42	39	11	60
2 SFCR-9	1958	44	0.038	19	0.80	1.0	35	40	12	63
2 SFCR-10	1952	25	0.146	63	0.98	1.0	24	38	12	61
2 SFCR-11	1951	12	0.133	65	1.17	1.0	15	31	9.8	54
2 SFCR-12	1947	12	0.086	41	1.22	1.0	14	30	7.2	36
2 SFCR-13	1946	11	0.128	64	1.15	2.0	25	40	8.8	41
2 SFCR-14	1945	10	0.137	66	1.33	2.0	28	36	7.7	37
2 SFCR-15	1943	13	0.121	65	1.21	1.0	15	35	7.6	36
2 SFCR-16	1942	13	0.105	60	1.28	2.0	32	37	8.8	43
2 SFCR-17	1940	11	0.088	55	1.36	1.0	15	34	8.3	41
2 SFCR-18	1938	13	0.145	74	1.36	1.0	17	36	8.5	44
2 SFCR-19	1936	13	0.121	57	1.25	1.0	16	36	8.0	41
2 SFCR-20	1935	10	0.187	84	1.08	2.0	22	40	7.0	35
2 SFCR-21	1934	13	0.184	82	1.14	2.0	30	29	6.6	33
2 SFCR-22	1933	12	0.102	47	1.31	2.0	32	25	7.6	39
2 SFCR-23	1932	12	0.133	63	1.09	2.0	25	28	7.3	36
2 SFCR-24	1931	12	0.072	47	1.08	2.0	26	29	7.6	38
2 SFCR-25	1930	14	0.068	41	1.21	2.0	34	28	7.5	37
2 SFCR-26	1929	16	0.071	41	1.07	2.0	35	32	7.5	38
2 SFCR-27	1928	13	0.073	40	1.11	2.0	30	31	7.5	38
2 SFCR-28	1927	13	0.063	34	1.20	2.0	32	33	7.8	39
2 SFCR-29	1926	14	0.051	28	1.17	2.0	33	30	7.8	40
2 SFCR-30	1925	16	0.070	38	1.30	2.0	43	29	8.3	44
2 SFCR-31	1924	14	0.044	23	1.14	2.0	33	28	8.4	45
2 SFCR-32	1923	19	0.041	21	0.89	2.0	33	32	8.4	44
2 SFCR-33	1922	20	0.046	22	1.00	2.0	39	32	9.0	49
2 SFCR-34	1921	37	0.038	20	0.96	2.0	71	31	9.0	51
2 SFCR-35	1920	21	0.033	15	1.07	2.0	45	29	9.2	50
2 SFCR-36	1919	37	0.042	22	1.01	2.0	75	29	9.4	49
2 SFCR-37	1918	23	0.032	15	1.06	2.0	48	28	9.2	49
2 SFCR-38	1917	23	0.033	15	1.05	2.0	48	28	9.6	53
2 SFCR-39	1916	24	0.031	13	1.05	2.0	51	28	9.6	52
2 SFCR-40	1915	21	0.035	16	1.12	2.0	47	24	9.1	50
2 SFCR-41	1914	28	0.031	12	0.93	2.0	53	26	8.7	52
2 SFCR-42	1913	28	0.030	12	1.21	2.0	68	26	8.6	55
2 SFCR-43	1912	29	0.036	17	1.30	2.0	77	24	8.6	58

Table 3. Dates and geochemical data for core 14SFCR.

Sample #	Date	Hg (ng/g)	Mean Grainsize (mm)	% Sand	Bulk Density (g/cm ³)	Sed. Rate (cm/yr)	Hg flux (ng/cm ² /yr)
14SFCR-1	2005	36	0.09	33.1	0.28	2.2	22
14SFCR-2	2004	22	0.09	37.5	0.53	2.0	23
14SFCR-3	2003	25	0.07	33.3	0.59	2.0	30
14SFCR-4	2002	30	0.06	26.6	0.78	2.0	47
14SFCR-5	2001	32	0.08	33.9	0.82	2.0	52
14SFCR-6	2000	35	0.04	22.0	0.61	2.0	43
14SFCR-7	1999	26	0.05	25.3	0.78	2.0	41
14SFCR-8	1998	22	0.09	40.1	0.87	2.0	38
14SFCR-9	1997	26	0.14	61.6	0.96	2.0	50
14SFCR-10	1996	21	0.08	36.0	0.88	2.0	37
14SFCR-11	1995	8.5	0.19	83.4	1.16	2.0	20
14SFCR-12	1994	14	0.15	76.1	1.01	2.0	28
14SFCR-13	1993	17	0.09	55.8	1.19	2.0	40
14SFCR-14	1992	13	0.11	53.2	1.07	2.0	28
14SFCR-15	1991	16	0.08	39.5	1.04	1.0	17
14SFCR-16	1989	28	0.05	23.1	0.91	1.0	25
14SFCR-17	1987	27	0.05	22.4	1.00	2.0	54
14SFCR-18	1986	8.5	0.21	92.7	1.15	1.0	10
14SFCR-19	1984	8.5	0.28	96.8	1.24	2.0	21
14SFCR-20	1983	8.5	0.23	93.8	1.38	2.0	23
14SFCR-21	1982	8.5	0.20	96.9	1.24	0.667	7
14SFCR-22	1979	17	0.20	44.8	1.22	0.667	14
14SFCR-23	1976	21	0.35	87.2	1.07	0.667	15
14SFCR-24	1973	8.5	0.33	90.1	1.36	2.0	23
14SFCR-25	1972	8.5	0.29	96.1	1.46	3.33	41
14SFCR-26	1971.5	8.5	0.34	95.8	1.28	3.33	36
14SFCR-27	1971	26	0.26	87.8	1.07	3.33	93
14SFCR-28	1970.5	42	0.02	5.8	0.91	3.33	127
14SFCR-29	1970	58	0.03	15.7	0.773	3.33	149
14SFCR-30	1969	46	0.07	23.8	0.861	2.0	79
14SFCR-31	1968	56	0.02	4.5	0.747	2.0	84
14SFCR-32	1967	56	0.02	5.6	0.701	3.33	131
14SFCR-33	1966.5	53	0.02	0.7	0.729	3.33	129
14SFCR-34	1966	53	0.01	2.8	0.818	3.33	144
14SFCR-35	1965	45	0.02	4.1	0.855	3.33	128
14SFCR-36	1964.5	41	0.02	9.4	0.849	3.33	116
14SFCR-37	1964	41	0.02	6.8	0.913	2.0	75
14SFCR-38	1963	38	0.02	7.2	0.799	2.0	61
14SFCR-39	1962	34	0.02	9.5	0.94	1.0	32
14SFCR-40	1960	29	0.03	11.8	0.947	1.0	27
14SFCR-41	1958	30	0.03	12.4	1.04	1.0	31
14SFCR-42	1956	31	0.03	10.6	1.06	2.0	66
14SFCR-43	1955	34	0.04	21.4	1.12	2.0	76
14SFCR-44	1954	29	0.05	24.6	1.01	2.0	59

14SFCR-45	1953	25	0.05	22.7	1.07	2.0	54
14SFCR-46	1952	25	0.03	14.6	1.03	0.4	10
14SFCR-47	1947	31	0.03	12.4	0.995	0.5	15
14SFCR-48	1943	23	0.14	67.5	1.13	2.0	52
14SFCR-49	1942	19	0.13	51.3	1.24	2.0	47
14SFCR-50	1941	20	0.03	16.3	1.18	2.0	47
14SFCR-51	1940	18	0.08	31.3	1.39	2.0	50
14SFCR-52	1939	23	0.03	14.1	1.17	2.0	54
14SFCR-53	1938	24	0.03	11.4	1.34	2.0	64
14SFCR-54	1937	30	0.04	17.7	1.17	2.0	70
14SFCR-55	1936	25	0.04	17.0	1.14	2.0	57
14SFCR-56	1935	27	0.03	16.0	1.13	2.0	61
14SFCR-57	1934	57	0.03	12.6	1.06	2.0	121
14SFCR-58	1933	32	0.04	17.0	1.03	2.0	66
14SFCR-59	1932	40	0.06	24.1	0.905	2.0	72
14SFCR-60	1931	37	0.06	22.6	0.853	2.0	63
14SFCR-61	1930	31	0.12	48.8	0.956	2.0	59
14SFCR-62	1929	14	0.09	44.7	1.26	2.0	35

Table 4. Annual mean streamflow data for Salmon Falls Creek.

Date	Streamflow (ft ³ /s)	Date	Streamflow (ft ³ /s)
2003	72	1957	174
2002	106	1956	153
2001	63	1955	78
2000	90	1954	60
1999	153	1953	110
1998	183	1952	212
1997	217	1951	191
1996	181	1950	162
1995	197	1949	160
1994	68	1948	125
1993	184	1947	87
1992	47	1946	143
1991	88	1945	181
1990	80	1944	162
1989	149	1943	234
1988	101	1942	213
1987	69	1941	86
1986	208	1940	77
1985	165	1939	116
1984	440	1938	146
1983	224	1937	81
1982	193	1936	125
1981	61	1935	108
1980	201	1934	45
1979	165	1933	99
1978	157	1932	152
1977	72	1931	55
1976	170	1930	71
1975	250	1929	117
1974	171	1928	120
1973	170	1927	135
1972	228	1926	75
1971	270	1925	148
1970	177	1924	106
1969	167	1923	138
1968	78	1922	167
1967	120		
1966	70		
1965	189		
1964	170		
1963	125		
1962	207		
1961	58		
1960	110		
1959	73		
1958	147		

Table 5. Data for unfiltered water collected from location 42.06962° N, 114.76476° W, Salmon Falls Creek Reservoir.

Water depth	Surface	1.2 m	2.4 m	3.3 m
T (°C)	15.9	15.8	14.1	13.9
pH	7.6	7.6	7.3	7.2
ORP (mV)	152	153	156	138
DO (mg/L)	6.4	6.2	4.4	3.7
Hg (ng/L)	3.1	3.3	3.6	4.4
Methyl-Hg (ng/L)	0.76	0.91	1.4	1.8

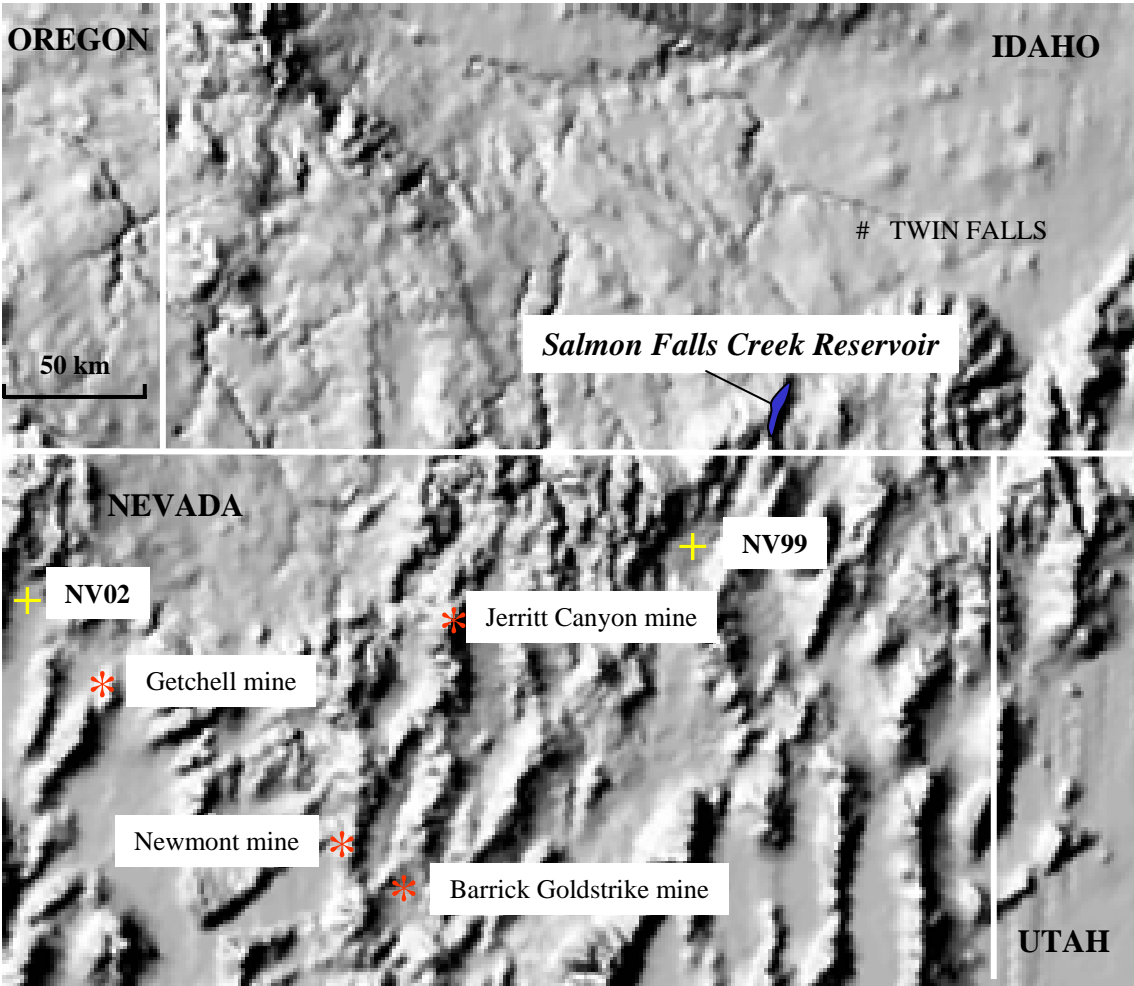


Figure 1. Location of Salmon Falls Creek Reservoir, Idaho. Location of several gold mines in northern Nevada are shown for reference. NV99 and NV02 are sites in the Mercury Depositional Network.

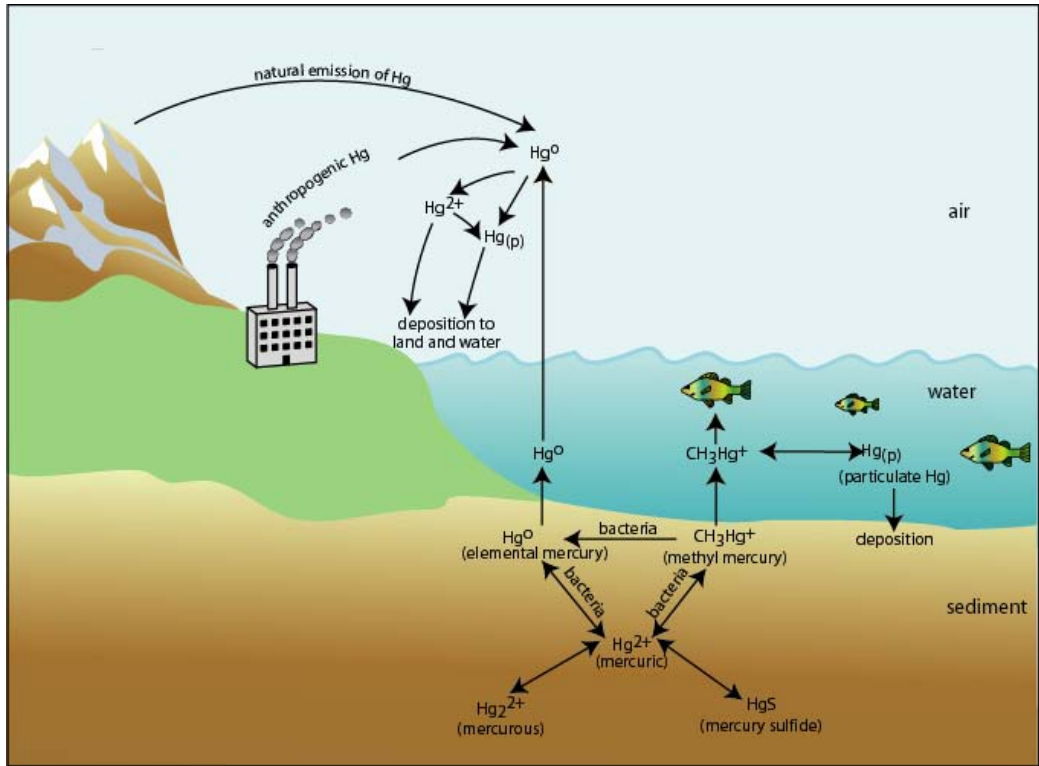


Figure 2. Major species and transformations of Hg (from Gray, 2003). $\text{Hg}(\text{p})$, particulate mercury. Conversion to methyl-Hg is most important because it is bioavailable and is transferred to water and biota.

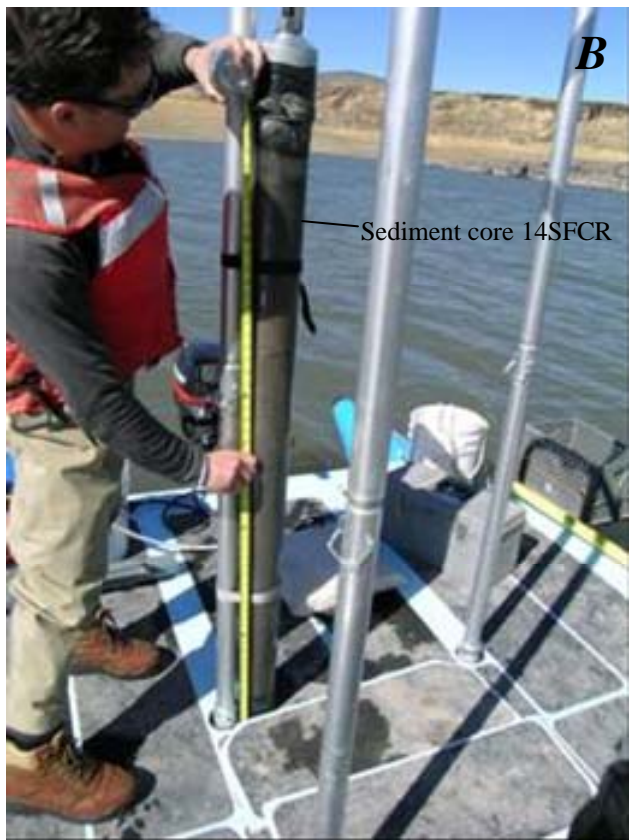


Figure 3. (A) Photo showing pontoon boat and sediment core. (B) Photo showing reservoir sediment core barrel containing core 14SFCR, October 6, 2005.

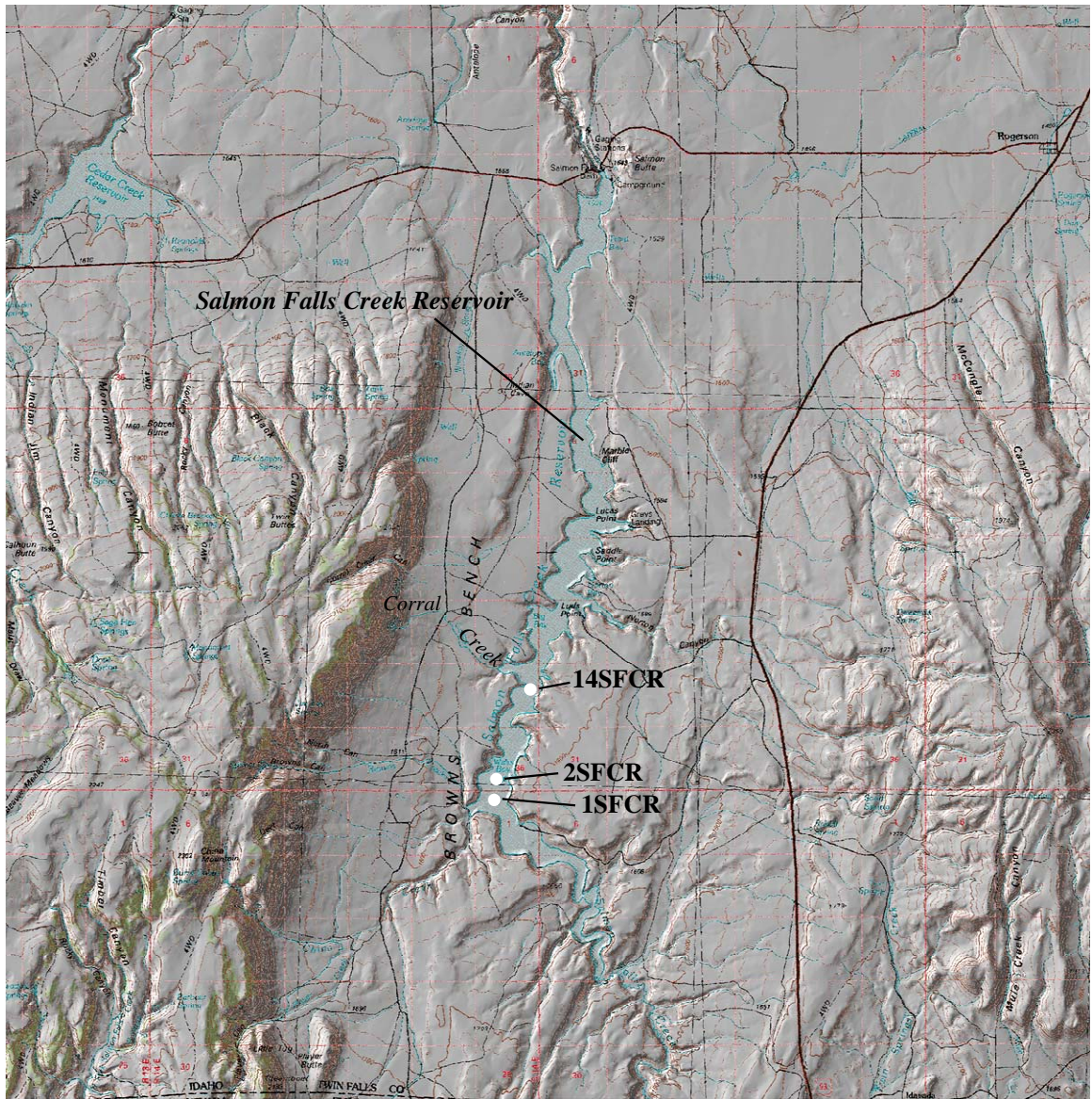


Figure 4. Location of sediment core samples collected from Salmon Falls Creek Reservoir.

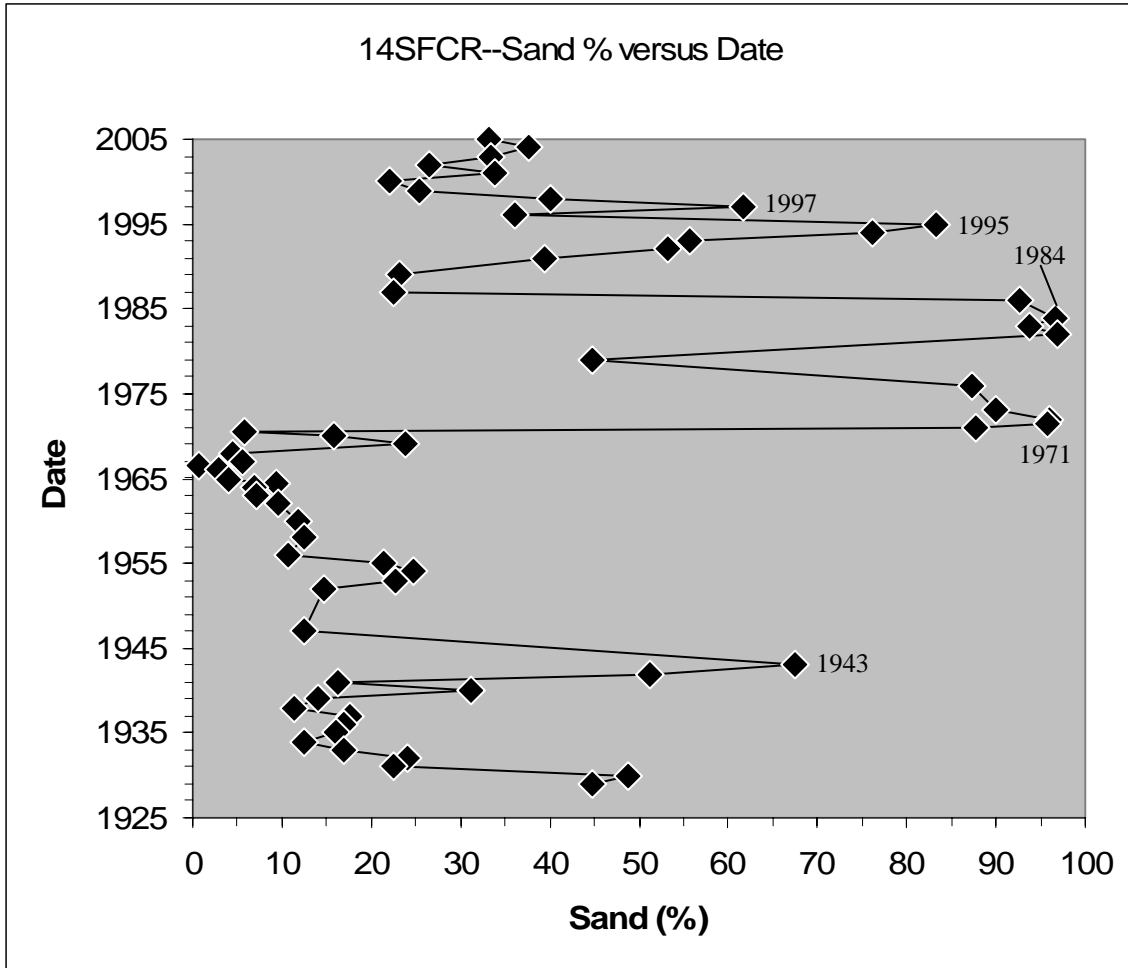


Figure 5. Variation of sand (%) versus date for core 14SFCR. Coarse-grained layers within this core correlate closely with years (1997, 1995, 1984, 1971, and 1943) of high discharge from Salmon Falls Creek.

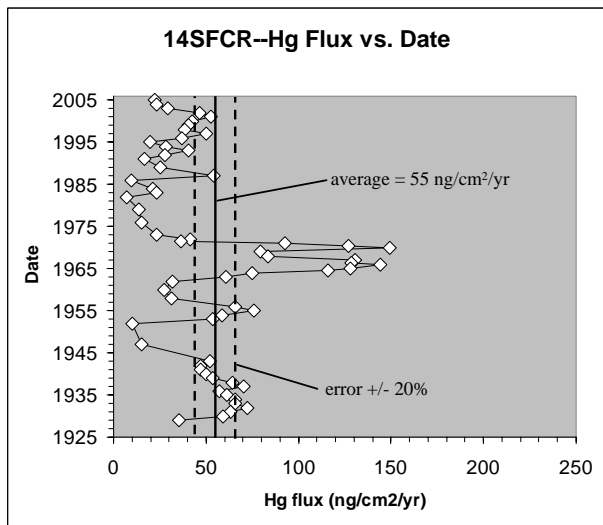
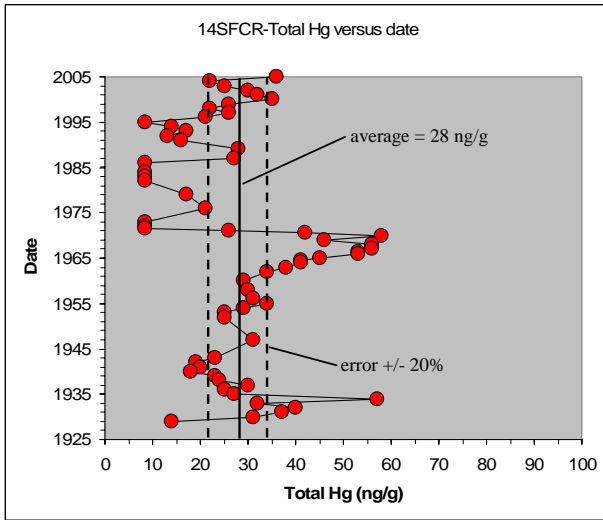
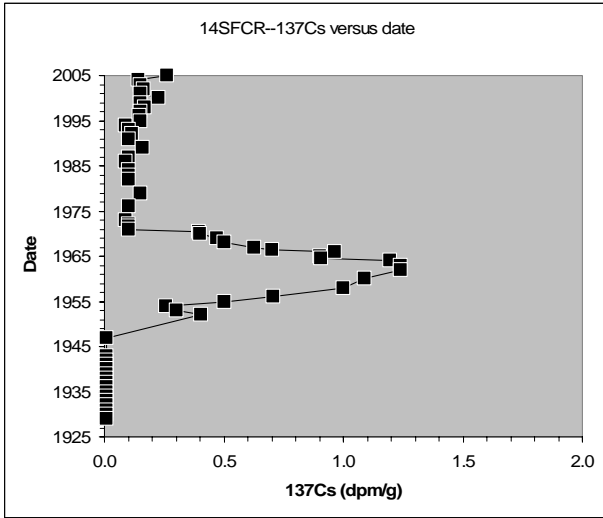


Figure 6. Variation of date with ¹³⁷Cs, total Hg concentration, and Hg flux for core 14SFCR.

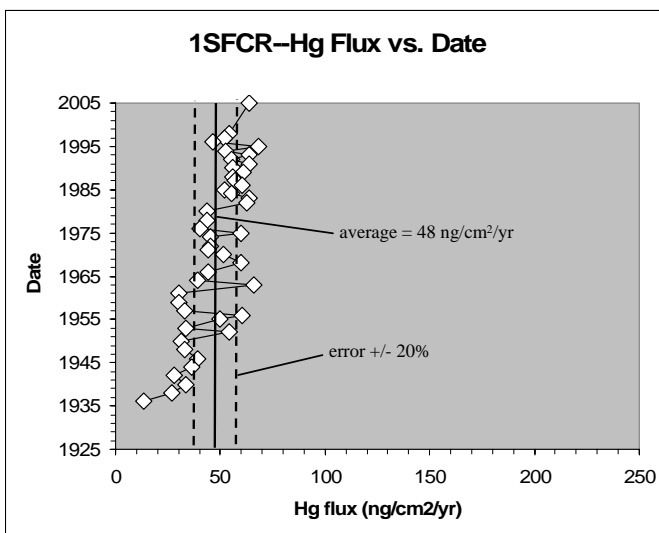
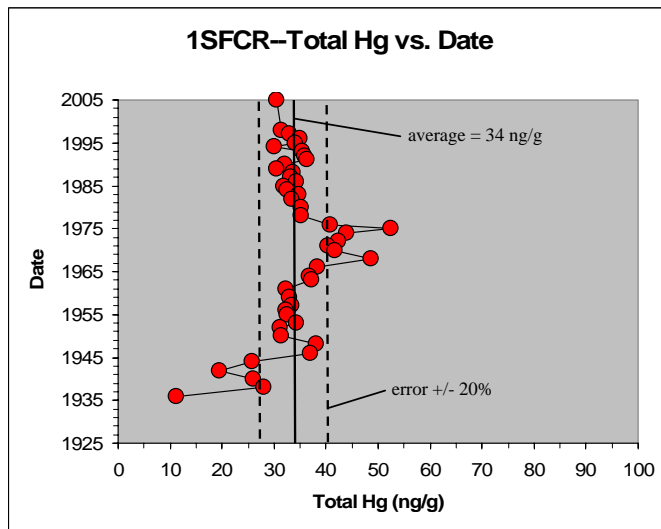
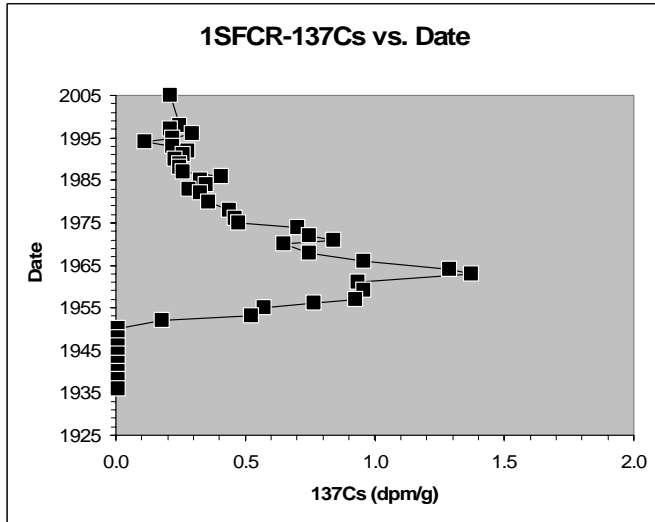


Figure 7. Variation of date with ¹³⁷Cs, total Hg concentration, and Hg flux for core 1SFCR.

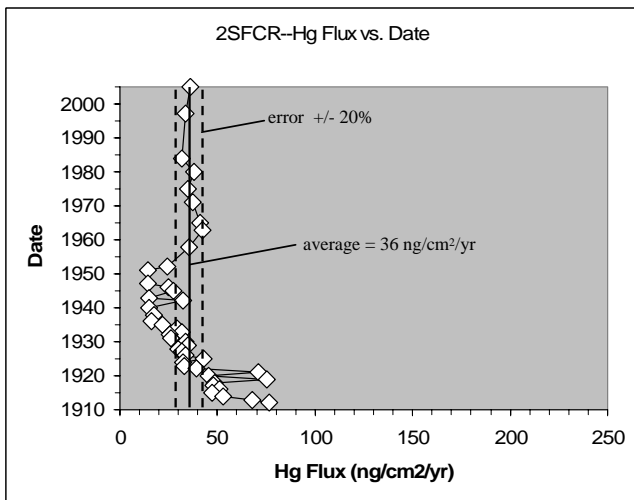
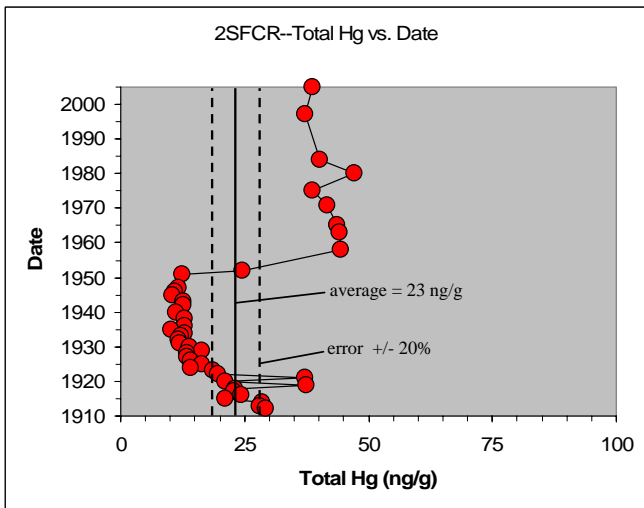
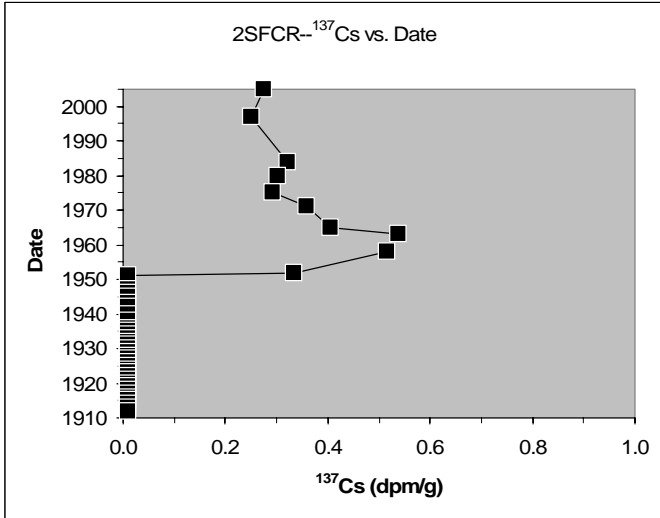


Figure 8. Variation of date with ¹³⁷Cs, total Hg concentration, and Hg flux for core 2SFCR.

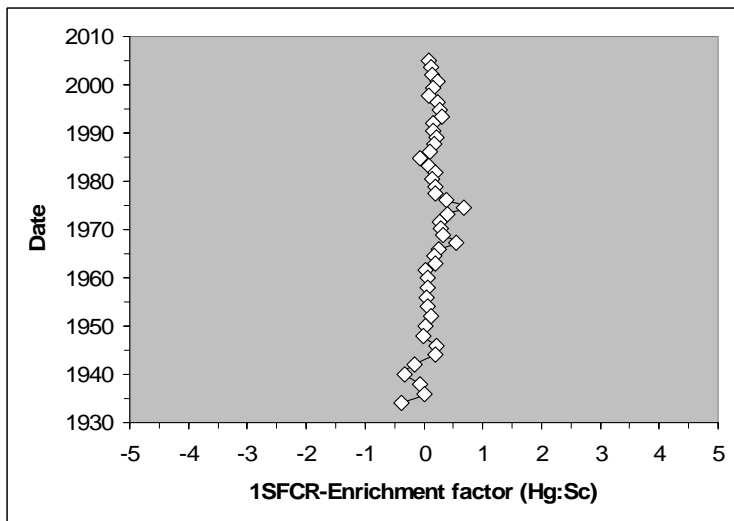
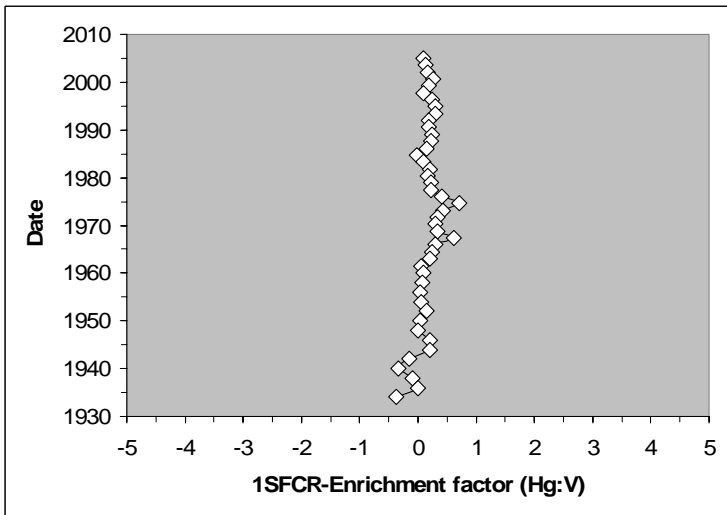
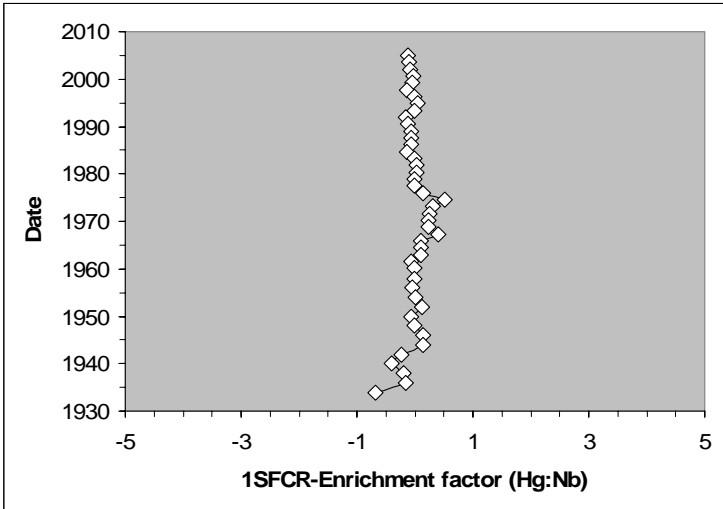


Figure 9. Enrichment factors versus date for core 1SFCR.

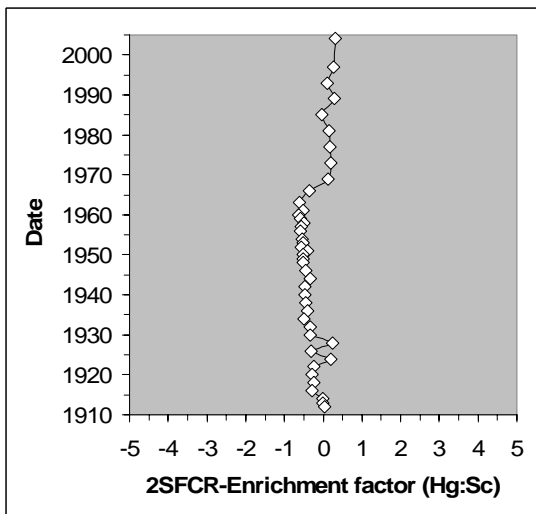
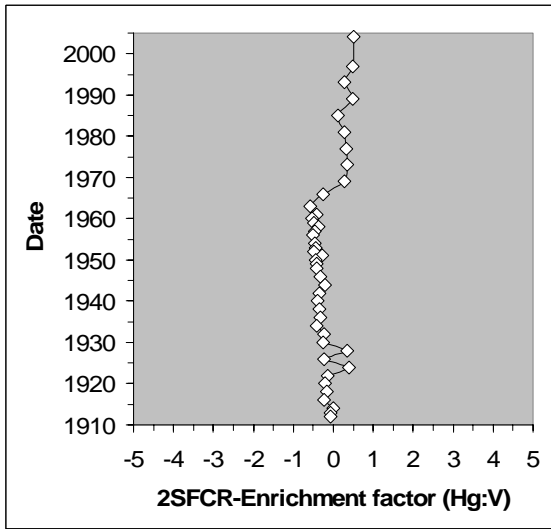
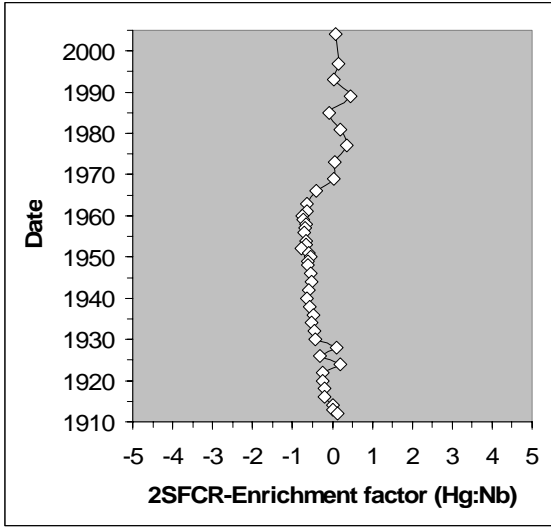


Figure 10. Enrichment factors versus date for core 2SFCR.

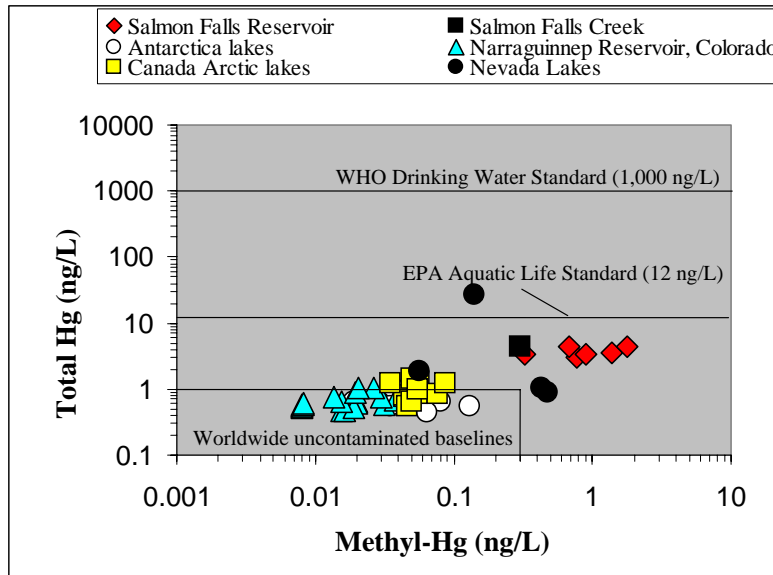


Figure 11. Total Hg versus methyl-Hg for unfiltered water collected from Salmon Falls Creek Reservoir, Idaho, June 15, 2005. Data shown for Antarctica lakes (Lyons and others, 1999), Canada arctic lakes (Loseto and others, 2004), Narraguinnep Reservoir, Colorado (Gray and others, 2005), and Nevada lakes (Seiler and others, 2004).