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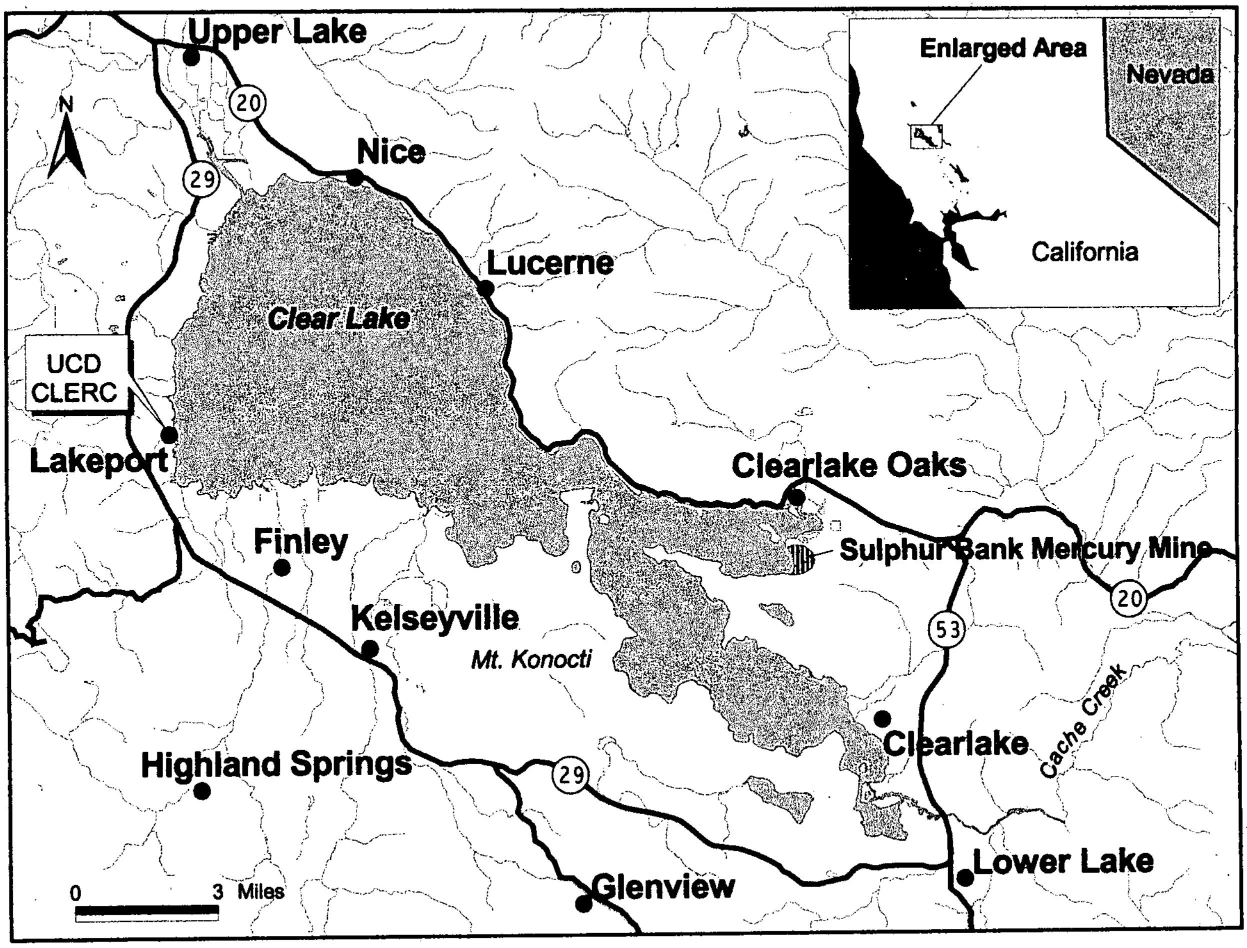

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Lakeport, CA

Affiliated with:

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SECOND ANNUAL
CLEAR LAKE SCIENCE AND MANAGEMENT SYMPOSIUM

October 24, 1998
Lakeport, CA

The U.C. Davis Clear Lake Environmental Research Center hosted the Second Clear Lake Science and Management Symposium on Saturday, October 24, 1998. It was held at the Lion's Club Community Hall, Lakeport, CA. The symposium opened at 8:00 am and concluded at 5:00 pm. The symposium offered a forum for the presentation and discussion of past, present and future studies associated with Clear Lake, CA and the surrounding watershed. Representatives from U.C. Davis and local, county, state and federal agencies presented on the topics of Clear Lake watershed processes and programs, physical and chemical processes, contaminants in Clear Lake and the surrounding watershed, and the Sulphur Bank Mercury Mine. Topics were presented in both oral and poster presentations.

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The Clear Lake Pollen Record: A Benchmark Profile for California Climate History

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ABSTRACT

Sediment cores recovered by the U.S. Geological Survey from Clear Lake in 1973 have yielded a detailed pollen record that spans the past 130,000 years. A series of informal geologic-climate units have been named and described as thermomers and cryomers (warm and cool intervals). A 115-meter core from the upper arm of the lake provided a 130,000-year record, and a 50-meter core from the Highlands arm provided a 50,000-year record as well. Calibration of the pollen record using modern surface samples suggests that temperatures during the coldest parts of the last glacial cycle in the Clear Lake region were up to 8°C cooler than at present, and that mid-Holocene temperatures were about 1°C warmer.

Plots of pine *vs.* TCT pollen percentages for the various geologic-climate units of the Clear Lake record indicate that vegetation dynamics varied significantly through time. Comparisons with similar plots from pollen records at other localities of different ages and at different elevations suggest that the changes in vegetation dynamics observed at Clear Lake through the last glacial cycle reflect plant communities that have existed in the western U.S. for at least the past 3 million years. Long, continuous stratigraphic records provide the historic context within which modern anthropogenic changes must be interpreted, and represent a valuable scientific resource for the evaluation of global change.

Keywords: Clear Lake, sediment cores, Quaternary, pollen, palynology, California, climate history

INTRODUCTION

Eight sediment cores were recovered from Clear Lake in 1973 as part of a program to establish a record of past seismic events through the study of deformed sedimentary structures (Sims, 1988). The record of past earthquakes proved elusive; the basin appears always to have been affected by enough wind stirring and bioturbation of the bottom sediments to prevent the preservation of laminated structures. However, the cores have proven themselves as a significant paleontological resource, with well-preserved records of fossil pollen, diatoms, and fish scales. Most of the work on the 1973 cores, as well as work on two longer cores taken in 1980, is summarized by Sims (1988) and by Adam (1988); the bibliographies in those works document the numerous USGS Open-File Reports that present the primary data. This paper presents a less technical summary of the work.

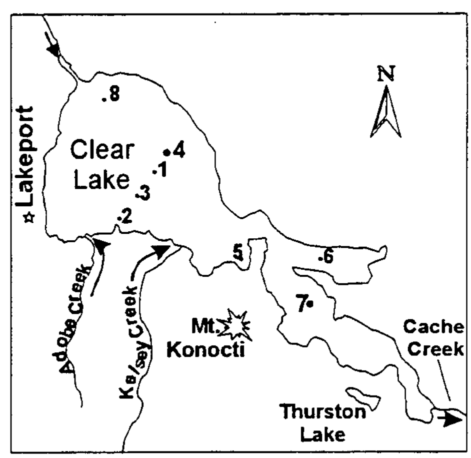

Figure 1.--Map Showing locations of cores (numbered dots) taken from Clear Lake by the USGS in 1973.

MATERIALS AND METHODS

The sediment cores were recovered using a truck-mounted drilling rig on a 3-section steel barge. The six-inch-diameter cores were taken in 30-inch sections using a Pitcher Barrel coring device that provided undisturbed cores. Recovery was generally quite good. Cores were extruded from the core barrel in the field and placed into PVC pipes sealed at the ends with paraffin for transport to the laboratory. Because refrigerated storage was not available, the cores had to be stored in an uncontrolled environment for several months before they were opened for examination in the laboratory.

When opened for examination, each core was split into a sample half and a voucher half, with a 1-cm-thick slice removed from the middle of the core for X-ray radiography. Samples were removed at 10-cm intervals for pollen, diatoms, geochemistry, plant macrofossils, weight loss, and mineralogy. Detailed sampling procedures are given by Beaver *et al.* (1975).

Two cores were analyzed for fossil pollen content. The longest core (Core 4) was 115 meters long and spanned the past 130,000 years. Dating was based on extrapolation of radiocarbon-dated sedimentation rates from the top of the core, and was consistent with correlations of the oak pollen percentage curve with deep-sea $\delta^{18}O$ curves (Adam, 1988; Robinson *et al.,* 1988). Core 7, from the Oaks Arm of the lake, was 49 meters in length and spanned approximately the past 50,000 years, based on radiocarbon dates.

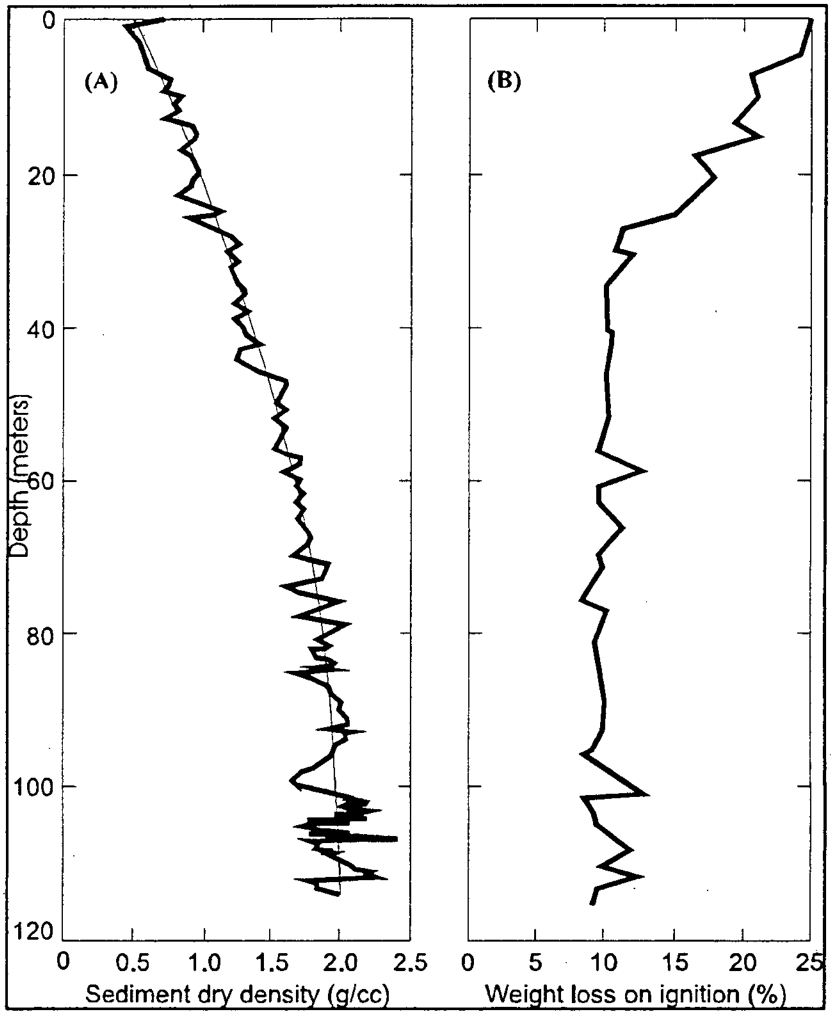

Figure 2.--Plots of (a) sediment dry density and (B) percent weight loss on ignition *vs.* Depth for Core 4 (Adam, 1988).

RESULTS

Sediment Properties

Core 4 was composed of fine-grained reduced clay throughout. Sediment density generally increased with depth, but systematic fluctuations with depth are evident on Figure 2. Organic content of the core, as shown by percent weight loss on ignition, varied from about 25% at the top of the core to about 10% below a depth of about 30 meters. The top 7 meters of Core 7 consisted of fine-grained, open-water lake muds similar to those observed at the top of core 4. Below 7 meters, however, the core was composed of interbedded peats and shallow-water lake muds.

Palynology

Both Core 4 and Core 7 were studied for fossil pollen content. The Core 4 record has produced the longest climate record, and is discussed here; the Core 7 record is generally consistent with the Core 4 record. The climatic interpretation of the pollen record, based in large part on the curve for oak pollen percentage *vs.* depth, was used to create a series of informal climatic intervals (Figure 3). Only the three dominant pollen types are shown here: oak (*Quercus* spp.), Pine (*Pinus* spp.), and TCT (plant families Taxaceae, Cupressaceae, and Taxodiaceae; probably mostly incense-cedar (*Calocedrus*) in the Clear Lake pollen record, but possibly also juniper (*Juniperus*) and cypress (*Cupressus*)).

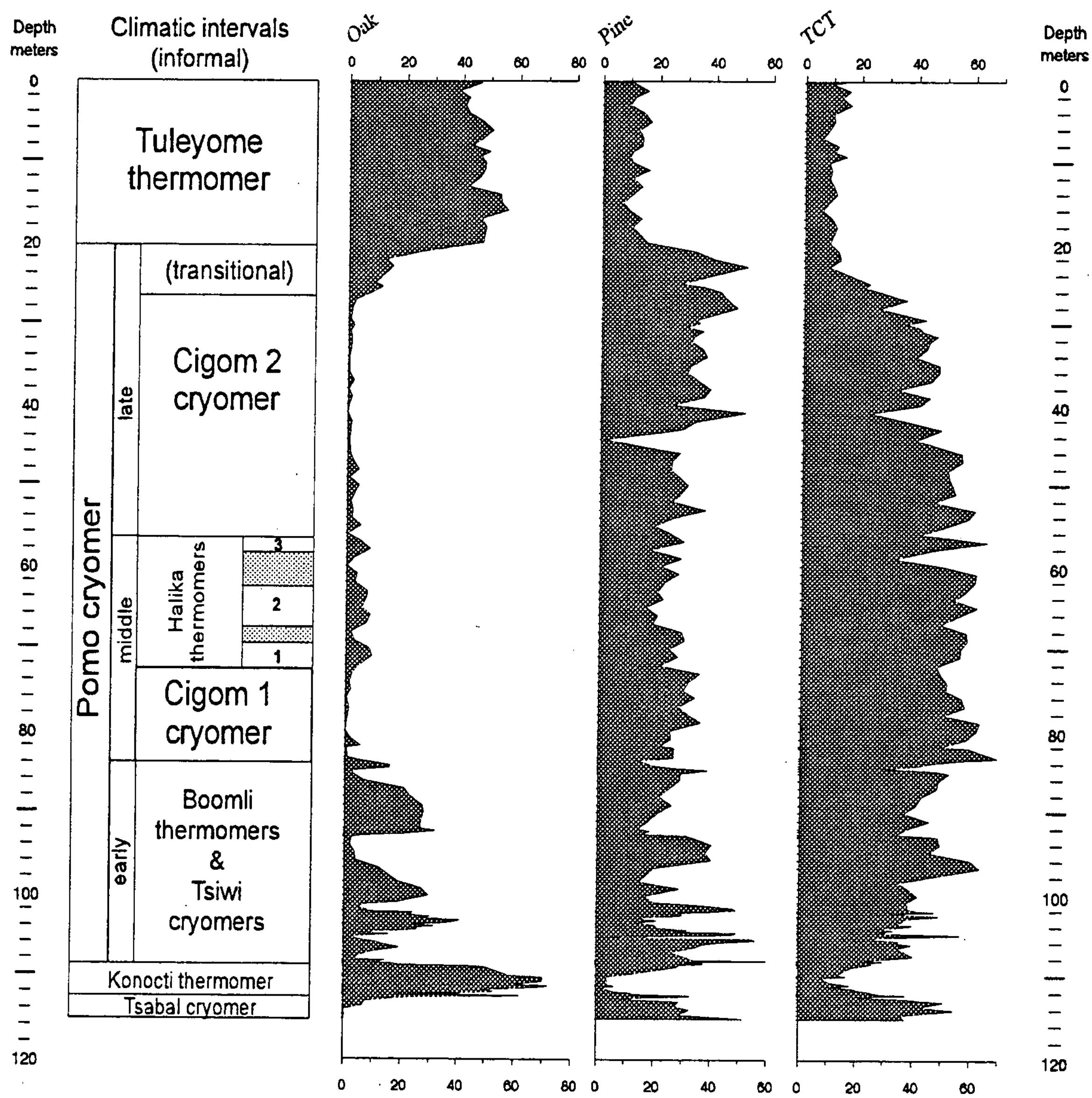

Figure 3.—Climatic intervals and major pollen types plotted against depth for Clear Lake Core 4. Increasing percentages of oak pollen correspond to warmer temperatures, whereas high pine percentages represent cooler intervals. Climatic intervals are described in terms of nested thermomers (relatively warm) and cryomers (cool). Summarized from Adam (1988).

The most important feature of the Core 4 pollen record is the curve for oak pollen, which fluctuates between values ≥50% at the top of the core(the Tuleyome thermomer) and near the bottom (the Konocti thermomer) to ≤10% (the Pomo and Tsabal cryomers) for much of the record. Radiocarbon dating of the top part of the core established that the Tuleyome thermomer represents the Holocene (postglacial) Period, and extrapolation of inorganic sedimentation rates below the radiocarbon-dated part of the core indicated that the Konocti thermomer corresponds to the last major interglacial, with an age of about 117,000 to 127,000 years. Comparison of the oscillations in the oak pollen curve with changes in $\delta^{18}O$ recorded in deep-sea cores showed that the climatic changes recorded in the deep-sea cores also affected the vegetation around Clear Lake in a major way.

The series of wide oscillations between 109 and 84 meters depth (Figure 3) are of particular interest, and indicate that temperature fluctuations caused major changes in the distribution of oak trees in the Clear Lake basin over an interval of about 50,000 years following the end of the Konocti thermomer (last interglacial). These changes represent temperature fluctuations on the order of 6°C (Adam and West, 1983).

DISCUSSION

Sediment Density

The relationship between carbon content and sediment depth shown in Figure 2B indicates that lakebottom sediment is depleted of nutrients only very slowly with time. Sediments buried less than 30 meters (100 feet) deep are still releasing carbon, and quite probably other nutrients as well. Any proposal to dredge Clear Lake to remove bottom sediments as a nutrient source must take a large volume of sediment into account.

The fluctuations of sediment density with depth (Figure 2A) suggest that the balance between sediment input from the Clear Lake drainage and sediment produced within the lake itself has varied through time, and it seems likely that variations in erosion rates within the basin have occurred in response to changes in environmental conditions. The extent to which future changes in erosion rates could affect the lake is not clear, but it is quite possible that

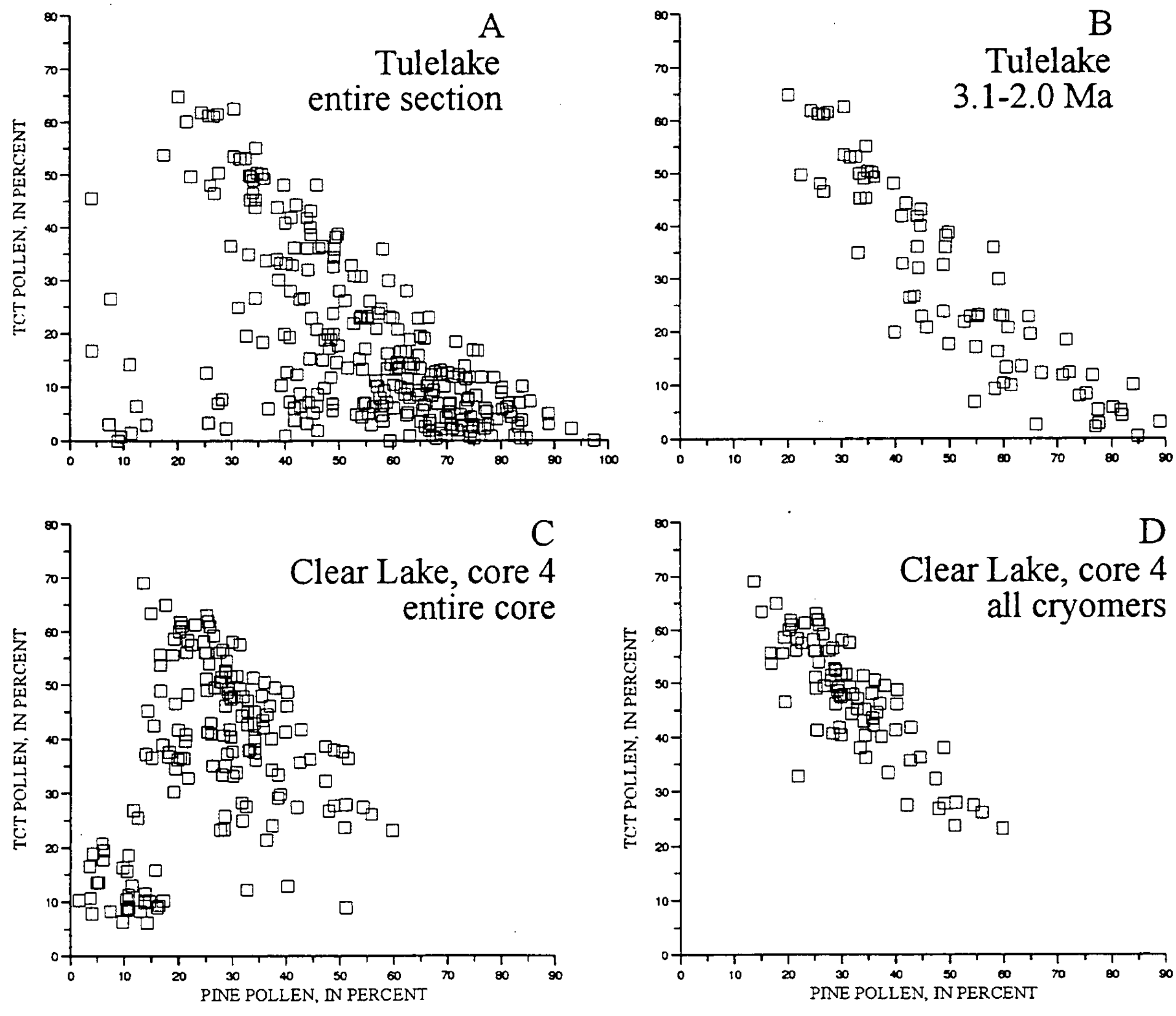

Figure 4.--Plots of percent TCT pollen *vs.* Percent pine pollen for (A) Tulelake entire core, (B) Tulelake samples older than 2 million years, (C) Clear Lake, Core 4, entire core, and (D) Clear Lake, Core 4, samples from cryomer intervals only. See text for discussion.

the effects of anthropogenic erosion rate changes could have deleterious effects on Clear Lake.

Comparison with other Pollen Records

The Clear Lake work established the validity of using long sedimentary records to develop the climatic history of California, and led to a subsequent drilling project at Tulelake, on the California/Oregon border, that yielded a pollen record estimated to span the past 3.2 million years (Adam et al., 1989). Comparison of that record with the Clear Lake record was hampered by the general scarcity of oak pollen at Tulelake, a result of the higher elevation and biogeographic factors. However, by comparing the records of pine and TCT pollen at the two sites, it was possible to establish broad comparisons between the vegetation dynamics at the two sites for various intervals in the past (Adam et al., 1990).

The approach taken was to plot percent TCT *vs.* percent pine for each sample from particular time intervals. When the Tulelake results were plotted for the entire record (Figure 4a), no clear pattern was present, but when only samples older than 2 million years (*i.e.*, Upper Pliocene) were included (Figure 4b), a much more structured cloud of points was apparent. Similar plots for the Clear Lake Core 4 data were prepared in an effort to better understand the Pliocene vegetation dynamics at Tulelake, with similar results: the plot for all of core 4 (Figure 4c) was not easy to interpret, but when only the samples from the cryomers were plotted (Figure 4d), the resulting cloud of points resembled that shown in Figure 4b. While it is not possible to draw detailed conclusions based on only two pollen types, the general match between Figures 4b and 4d suggests that the vegetation dynamics that prevailed at Tulelake during the late Pliocene were broadly similar to the conditions that prevailed at Clear Lake during the cooler parts of the last glacial cycle. A similar pattern was also found in Holocene deposits at Hodgdon Ranch, in an alluvial sequence at 4300 feet (1310 m) elevation on the western side of the Sierra Nevada (Adam, 1967; Adam *et al.*, 1990).

CONCLUSIONS

The Clear Lake pollen record is among the most detailed sequences yet recovered from sediments deposited during the last full glacial cycle. It correlates remarkably well with pollen sequences from northwestern Europe and with the deep-sea $\delta^{18}O$ record, and demonstrates that the climatic sequences developed there reflect global climate change, rather than local effects. The sedimentation rate at Clear Lake is relatively high, which has produced a 115-meter section capable of yielding a highly detailed record. Such sequences are unusual, and the Clear Lake basin thus contains a scientific resource of particular value for global change studies.

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Ticks and Tick-Borne Diseases of the Clear Lake (California) Watershed

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ABSTRACT

The possibility of contracting a tick-borne disease is a continuing public health threat in Lake County, California. Of the four common species of ticks in Lake County; *Ixodes pacificus, Dermacentor occidentalis, Dermacentor variabilis,* and *Ornithodoros coriaceus,* all but the latter are known to be capable of vectoring disease to humans. These diseases include Lyme disease, human babesiosis, human ehrlichosis, tularemia, and tick paralysis. Lyme disease is the most predominant locally transmitted tick-borne disease. In Lake County, *Ixodes pacificus,* the western black-legged tick is the primary vector of *Borrelia burgdorferi*, a corkscrew-shaped spirochete that causes Lyme disease. The LCVCD does tick surveys and collects ticks for laboratory tests that detect the presence of the causative agents of human disease. A reservoir host focused zoonoses control project was also undertaken in cooperation with Dr. Robert Kimsey of the University of California at Davis.

HABITAT PARTITIONING AMONG SOME CULICIDAE IN LAKE COUNTY, CALIFORNIA

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ABSTRACT

The 22 species of Lake County mosquitoes occupy various habitats at various times of the year. Peak abundances can occur during the winter (e.g., *Culiseta inornata*), spring (e.g., *Aedes sierrensis*), or summer (e.g., *Culex tarsalis*). Some species can be abundant in irrigated pastures (e.g., *Aedes nigromaculis*), water-filled treeholes (e.g., *Aedes sierrensis*), organically loaded waters (e.g., *Culex stigmatosoma*), or ricefields (e.g., *Anopheles freeborni*). Some species (e.g., *Aedes fitchii*) are restricted to high elevation snowmelt pools. Although many Lake County Culicidae are separated temporally or spatially, larvae of *Aedes bicristatus* and *Aedes increpitus* can occur together in some aquatic habitats. Various chemical, physical and biological factors were analyzed in twelve impermanent lentic habitats containing one or both of these species. Densities of *Aedes bicristatus* were positively correlated with dissolved oxygen concentrations, but *Aedes increpitus* densities were negatively correlated with this factor. Gut content analyses indicated *Aedes bicristatus* fed upon photosynthesizing Chlorophyta in areas of sunlit pools, and *Aedes increpitus* fed on decaying organic matter (e.g., leaves) and associated microorganisms in areas of lentic habitats with trees.
Keywords: Culicidae, water quality, *Aedes bicristatus, Aedes increpitus*

INTRODUCTION

The Culicidae (mosquitoes) of Lake County might be grouped according to genus, by host preference, by vector competency for the transmission of various diseases, by number of generations per year, by type of larval habitat, or by similarity of adult activity periods. Some species occur in similar habitats at different times (Woodward et al., 1998).

The present study examined spatial and temporal differences in the occurrences of six species (*Aedes bicristatus, Aedes increpitus, Aedes sierrensis, Anopheles freeborni, Culiseta inornata,* and *Culex tarsalis*) of mosquitoes in Lake County. All of these mosquitoes are known to bite humans. *Culex tarsalis* is the primary transmitter of Western equine encephalomyelitis to humans, and *Anopheles freeborni* is the most important potential vector of malaria in California (Bohart and Washino, 1978; Reeves,1990). *Aedes sierrensis* is an important vector of the canine heartworm, *Dirofilaria immitis* (Woodward et al., 1988). Some *Culiseta inornata* adults collected in California have been found to be naturally infected with California group encephalitis viruses (Reeves, 1990). The disease-transmission potentials of *Aedes bicristatus* and *Aedes increpitus* are unknown, although *Aedes bicristatus* is closely related to *Aedes provocans*, which is the most important vector of Jamestown Canyon virus in the Midwestern United States (Heard et al., 1990).

MATERIALS AND METHODS

The larval mosquitoes and other aquatic invertebrates were sampled by collecting standard (400 ml) aquatic dip samples (Colwell and Garcia, 1989) ca. 0.5 m from the shoreline. Some dip samples were concentrated in a 0.5 mm mesh net (Kramer et al., 1987), and the contents were identified (Bohart and Washino, 1978, Merritt and Cummins, 1996) and counted in the laboratory. The gut contents of some *Aedes increpitus* (= *Aedes washinoi* of Eldridge et al., 1998) and *Aedes bicristatus* larvae were analyzed in the laboratory according to Woodward et al. (1988).

Daily samples of adult mosquitoes were collected with a Fay-Prince trap baited with carbon dioxide (Garcia et al., 1989). Weekly monitoring of adult mosquitoes utilized a New Jersey light trap (Kramer et al., 1988). All adult mosquitoes were taken to the laboratory and identified (Bohart and Washino, 1978) at 15X magnification.

A 400 ml water sample was collected 0.1 m below the water surface at each aquatic location. Water samples were transported to the lab and analyzed (Greenberg et al., 1985; Colwell et al., 1995). The latitude, longitude and elevation of each site were also determined (DeLorme, 1993).

Plankton was sampled by means of vertical tows (Colwell and Schaefer, 1983). These samples were preserved and analyzed according to the methods of Anderson et al. (1986).

Diel studies of host-seeking females were conducted at Dry Lake during 1993 on April 21, 26, 27, and 28. A non-directional black and white trap (similar to a Fay trap without the wing panels) was baited with ten pounds of dry ice. An electronic bottle rotator (John W. Hock Company, Model 1512) pooled mosquitoes for each three hour collection period.

RESULTS AND DISCUSSION

Interspecific competition among the 22 species of Lake County mosquitoes might be reduced by temporal partitioning. Figure 1 indicates that some species do have differences in their seasonalities. *Culiseta inornata* mosquitoes were most abundant during winter months, *Aedes sierrensis* mosquitoes were active in the spring and early summer, and *Culex tarsalis* mosquitoes were abundant in late summer. The carbon-dioxide baited Fay trap data (Figure 2) indicated *Aedes bicristatus* was an early season univoltine mosquito. *Anopheles freeborni* mosquitoes had multiple generations which resulted in increased population densities during the summer.

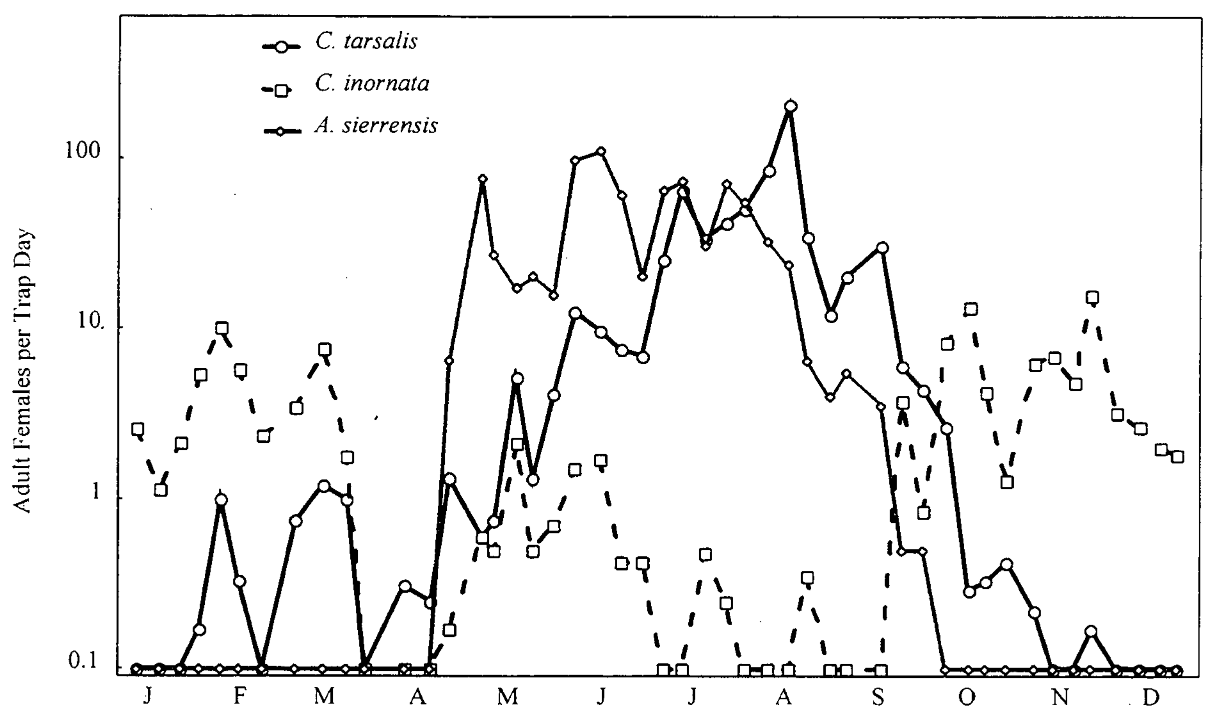

Figure 1. Numbers (X+0.1) of adult female mosquitoes per trap day collected by a standard New Jersey light trap (*Culex tarsalis* and *Culiseta inornata*) or by a CO_2-baited Fay trap (*Aedes sierrensis*) during 1996.

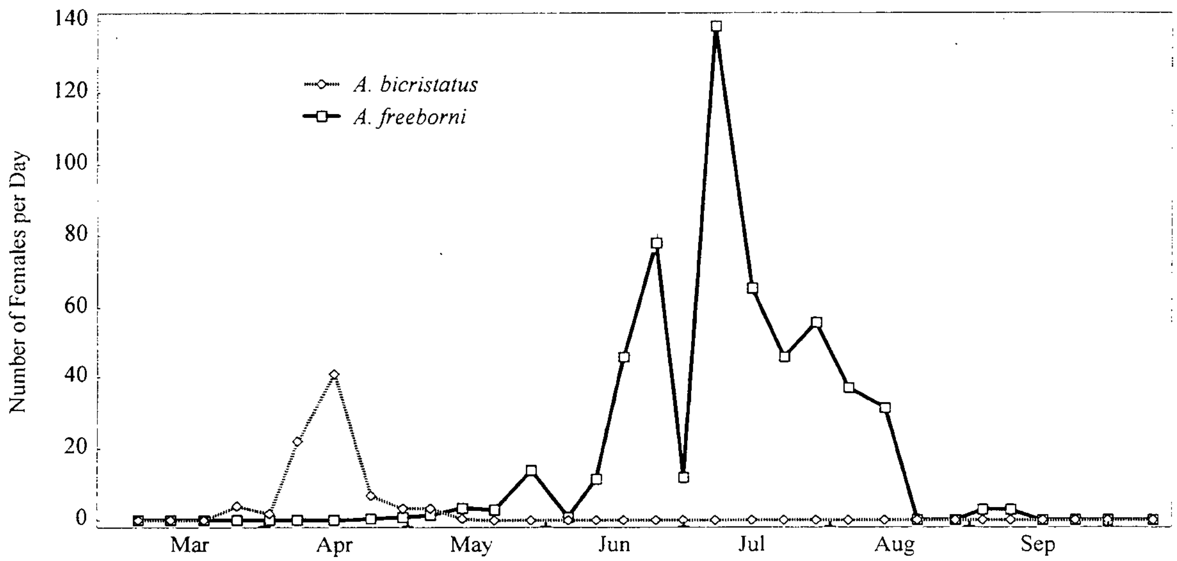

Figure 2. Numbers of adult female mosquitoes collected per trap day by Fay traps baited with carbon dioxide during 1997.

Dip sampling in Lake County indicated *Aedes increpitus* and *Aedes bicristatus* mosquitoes occurred together in some habitats. Figure 3 indicates there were even some similarities in the daily activity peaks of these two species.

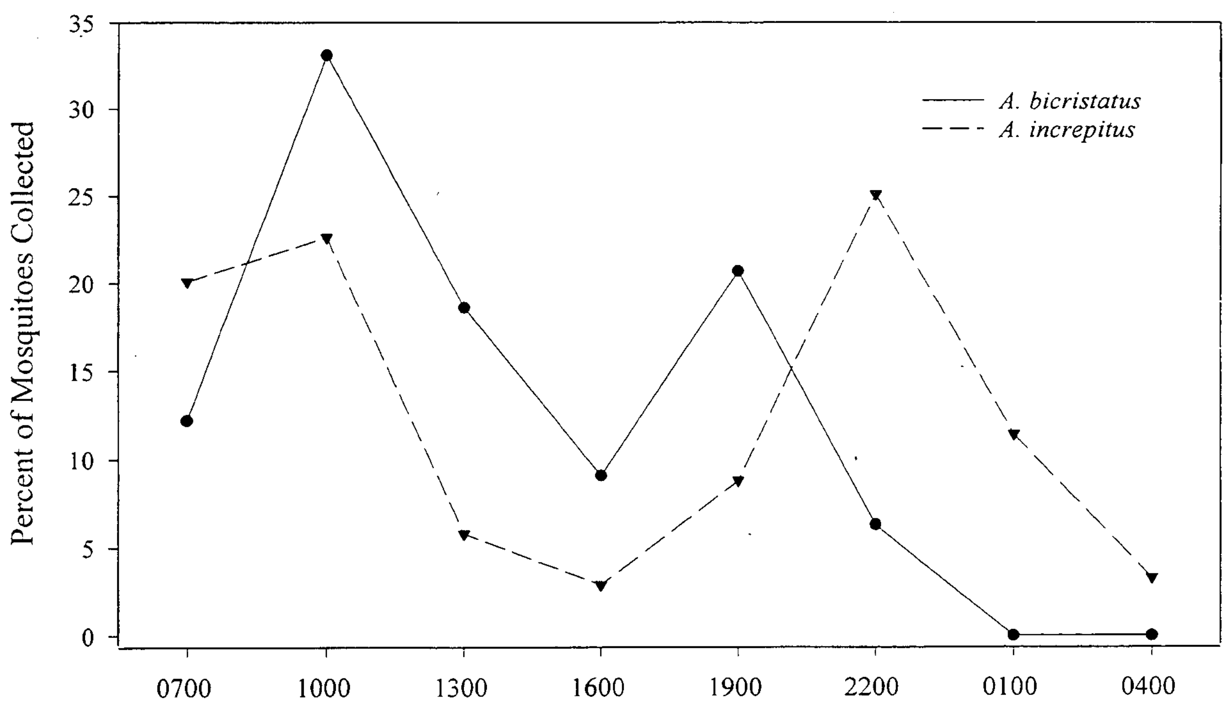

Figure 3. Percentages of mosquitoes collected by a CO_2-baited Fay trap connected to a collection bottle rotator at various times of day.

When wintertime rains raise the water level of a temporary lake or vernal pool, eggs of *A. bicristatus* hatch into first instar larvae. This can produce large numbers of early instar larvae in Lake County during February (Figure 4). These develop into third and fourth instars, and pupation can peak in March. Adult *A. bicristatus* then emerge and host-seeking peaks in April (Figure 4).

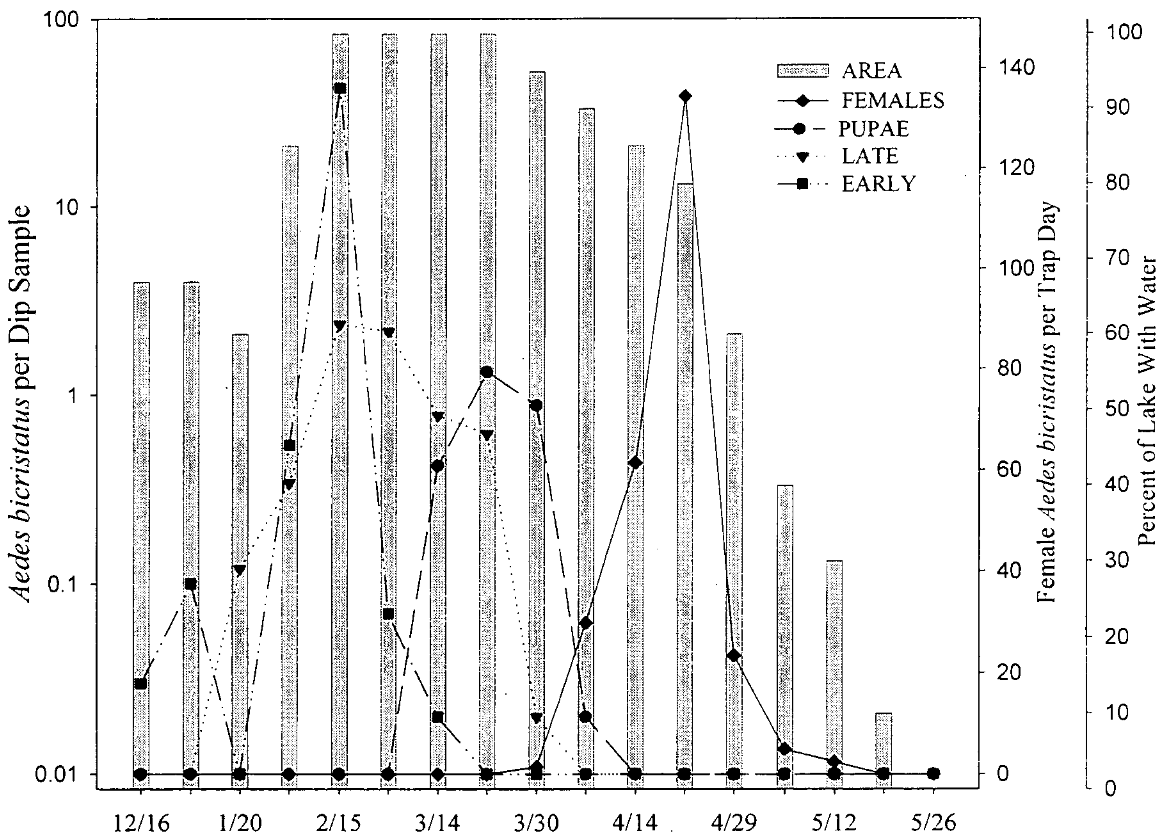

Figure 4. Numbers of immature mosquitoes per dip sample (X+0.01) collected from Dry Lake. The right Y-axes indicate the percentage of the lakebed area which was flooded during the 1993-1994 rainfall season, and the numbers of adult mosquitoes per trap day collected by a CO_2-baited Fay trap.

Since *Aedes bicristatus* and *Aedes increpitus* mosquitoes can occur at similar times (Figure 5) in similar lentic habitats, it was not clear which factors determined population abundances of these two species. Statistical analyses of the biological, chemical and physical factors in Tables 1-4 indicated only one factor, the dissolved oxygen concentration of the water, had a positive correlation for one species and a negative correlation for the other species. The habitats with higher dissolved oxygen concentrations tended to have higher concentrations of *A. bicristatus* and lower densities of *A. increpitus* larvae (Figure 6). Plankton sampling indicated that chlorophytes and chrysophytes were abundant in some habitats, and photosynthesis may have been contributing to high dissolved oxygen concentrations in areas where *A. bicristatus* larvae were abundant. However the aquatic habitats with *A. increpitus* often had shady areas or trees and decaying leaves in the water which may have reduced the dissolved oxygen concentrations. Gut content analyses of field collected larvae indicated that *A. bicristatus* larvae were feeding on green algae, whereas *A. increpitus* larvae were browsing on decaying organic matter and associated microorganisms

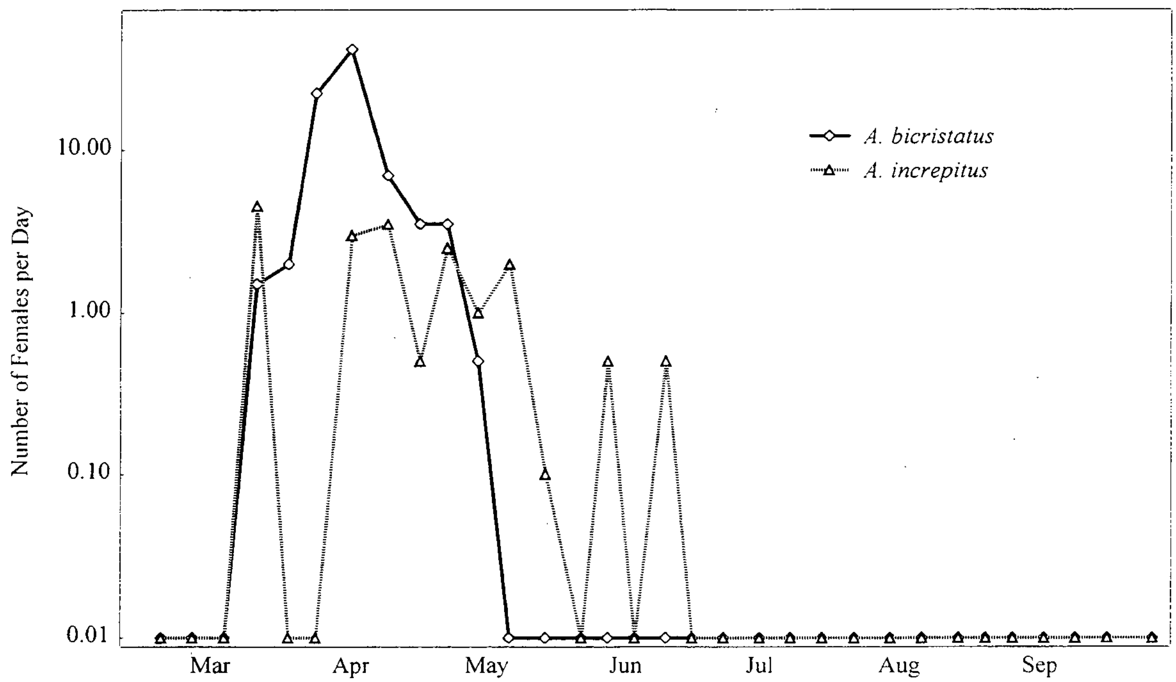

Figure 5. Numbers of female mosquitoes collected per trap day by CO_2-baited Fay traps near Dry Lake in 1997.

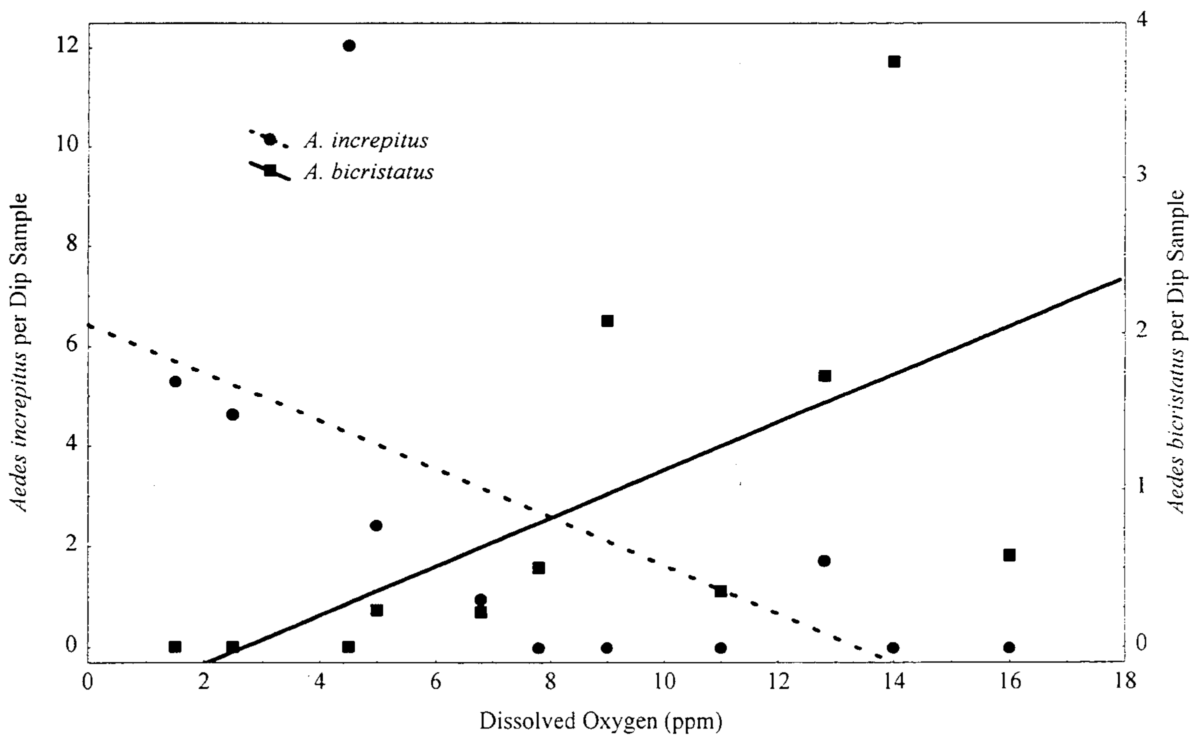

Figure 6. One-variable regressions of dissolved oxygen concentrations and numbers of immature mosquitoes per dip sample in twelve lentic habitats in Lake County.

Table 1. Mean numbers of mosquitoes and other organisms collected per standard dip sample during 1995.

Location	*Aedes increpitus*	*Aedes bicristatus*	Chironomidae	Dytiscidae	Hydro-philidae	Coen-agrionidae	Ephemeroptera	Turbellaria	Amphipoda	*Linderiella occidentalis*	Anura
Slater Island	4.63	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.03	0.00	0.00
Anderson Marsh	12.05	0.00	0.00	0.02	0.00	0.00	0.00	0.00	1.22	0.00	0.00
C.L. State Park	5.30	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Corinthian Bay	2.42	0.23	0.02	0.00	0.00	0.00	0.00	0.00	0.10	0.00	0.00
Wright Pasture	0.28	0.32	0.02	0.00	0.00	0.00	0.03	0.18	0.03	0.00	0.00
Dry Lake	0.00	2.08	0.00	0.10	0.10	0.05	0.00	0.99	0.00	0.02	0.02
Pinkeye Lake	0.00	0.50	0.00	0.07	0.05	0.00	0.00	0.07	0.00	0.02	0.00
Bradley Lake	0.00	3.75	0.00	0.02	0.00	0.00	0.10	0.13	0.00	0.52	0.00
Maya Lake	1.72	1.73	0.02	0.02	0.00	0.00	0.00	0.84	0.00	0.10	0.00
Boggs Lake	0.95	0.22	0.02	0.43	0.00	0.00	0.00	0.63	0.00	0.02	0.00
C. L. R. M.	0.00	0.58	0.02	0.03	0.00	0.00	1.50	1.20	0.00	0.00	0.41
Loch Lomond	0.00	0.35	0.00	0.02	0.00	0.00	0.00	0.03	0.00	0.00	0.77

Table 2. Mean numbers of organisms per liter collected in vertical tow plankton samples during 1995.

Location	*Bosmina*	*Daphnia pulex*	*Daphnia galeata*	*Diaptomus*	Cyclopoida	Ostracoda	*Brachionus*	*Aulacoseira*	*Stephanodiscus*	*Fragilaria*	Other Pennales
Slater Island	7.9	46.4	43.0	1.1	3.4	0.0	0.0	43527	435	69644	145
Anderson Marsh	0.0	1.1	0.0	0.0	5.7	0.0	145.1	106000	4643	0	1016
C. L. State Park	0.0	0.0	1.4	32.5	2.8	0.0	0.0	0	0	0	0
Corinthian Bay	18.9	1.9	0.0	0.0	703.6	0.0	241.8	24182	1451	0	1451
Wright Pasture	0.0	0.0	0.0	0.0	60.0	20.0	0.0	0	0	17948	58972
Dry Lake	4.5	4.5	0.0	0.0	31.7	11.3	0.0	11607	0	2321	871
Pinkeye Lake	0.0	0.0	1.4	4.2	2.8	2.8	0.0	16323	1632	0	907
Bradley Lake	0.0	0.0	0.0	9.4	2.8	0.9	0.0	0	9	3023	605
Maya Lake	0.0	0.0	0.0	0.0	30.6	0.0	145.1	0	0	91408	1016
Boggs Lake	0.6	0.0	1.1	2.3	1.7	0.0	72.6	0	73	0	0
C. L. R. M.	0.0	0.0	0.0	1.4	7.1	0.0	0.0	0	0	0	1995
Loch Lomond	0.0	0.0	0.0	20.0	60.0	0.0	0.0	0	0	0	0

Table 3. Water quality of some floodwater mosquito sources in Lake County during 1995.

Location	pH (units)	Total Alkalinity (ppm)	Total Hardness (ppm)	Iron (mg/l)	Nitrate (mg/l)	Boron (mg/l)	Chloride (mg/l)	Tannic Acid (mg/l)	Dissolved Oxygen (ppm)	Chemical Ox. Dem. (mg/l)	Bacteria (no./ml)
Slater Island	7.59	200	180	0.26	0.1	2.4	12.16	2.1	2.5	45	$1x10^4$
Anderson Marsh	7.49	200	185	1.16	0.3	1.8	9.71	1.1	4.5	22	$1x10^5$
C.L. State Park	7.15	125	110	1.08	2.5	2.0	23.38	14.5	1.5	207	$1x10^7$
Corinthian Bay	7.72	210	220	0.09	0.2	1.7	10.08	1.7	5.0	50	$1x10^4$
Wright Pasture	7.69	290	280	-	-	-	-	-	-	-	-
Dry Lake	7.43	40	35	0.65	0.0	0.3	4.25	0.6	9.0	5	$<1x10^2$
Pinkeye Lake	8.14	30	30	0.81	0.2	0.2	2.55	1.0	7.8	10	$<1x10^2$
Bradley Lake	7.29	30	40	0.47	0.3	0.3	7.47	0.8	14.0	9	$<1x10^2$
Maya Lake	7.41	30	40	0.05	0.1	0.2	3.81	1.5	12.8	32	$<1x10^2$
Boggs Lake	7.51	30	30	0.21	0.1	0.3	1.94	0.9	6.8	8	$1x10^3$
C.L. R. M.	8.46	210	230	0.47	0.1	0.0	17.57	1.2	16.0	29	$1x10^6$
Loch Lomond	7.69	10	10	0.13	0.5	0.4	1.70	4.0	11	95	$<1x10^2$

- = No data.

Table 4. Physical parameters of some sources of larval mosquitoes during 1995.

Location	Latitude	Longitude	Elevation (meters)	Depth (cm)	Temperature (°C)	Turbidity (NTU)	Conductivity (μmhos/cm)	True Color (Pt-Co Un.)	Area (hectares)
Slater Island	38° 56' 06"	122° 38' 12"	404.8	5-46	13	3.92	440	61	2.18
Anderson Marsh	38° 55' 10"	122° 37' 53"	406.6	8-25	14	2.63	420	43	1.76
C.L. State Park	39° 01' 07"	122° 48' 41"	405.4	0-15	14	1.27	285	257	0.10
Corinthian Bay	39° 01' 27"	122° 51' 18"	405.4	8-15	13	2.76	480	62	0.15
Wright Pasture	39° 00' 57"	122° 54' 14"	404.8	8-10	-	1.67	600	46	0.12
Dry Lake	38° 54' 40"	122° 38' 23"	486.7	8-20	18	11.40	110	86	0.85
Pinkeye Lake	38° 54' 33"	122° 39' 27"	547.7	8-10	17	22.20	58	162	5.56
Bradley Lake	38° 57' 07"	122° 47' 33"	550.8	8-20	18	23.10	130	142	17.23
Maya Lake	38° 56' 52"	122° 43' 55"	570.4	5-15	12	1.36	120	92	0.36
Boggs Lake	38° 53' 21"	122° 46' 36"	852.8	8-13	18	4.46	54	64	22.46
C. L. R. M.	39° 00' 34"	122° 54' 20"	413.9	0-20	18	2.21	500	63	0.25
Loch Lomond	38° 52' 01"	122° 43' 07"	861.9	5-25	24	2.03	32	40	1.70

Aedes bicristatus densities were positively correlated not only with dissolved oxygen concentrations but also with densities of the fairy shrimp *Linderiella occidentalis* (Figure 7), but this does not indicate there is any cause-effect relationship. Rather, both of these organisms are well adapted to ephemeral aquatic habitats.

z=-0.394+20.431*x+0.16*y-12.2*x*x-0.58*x*y-0.006*y*y

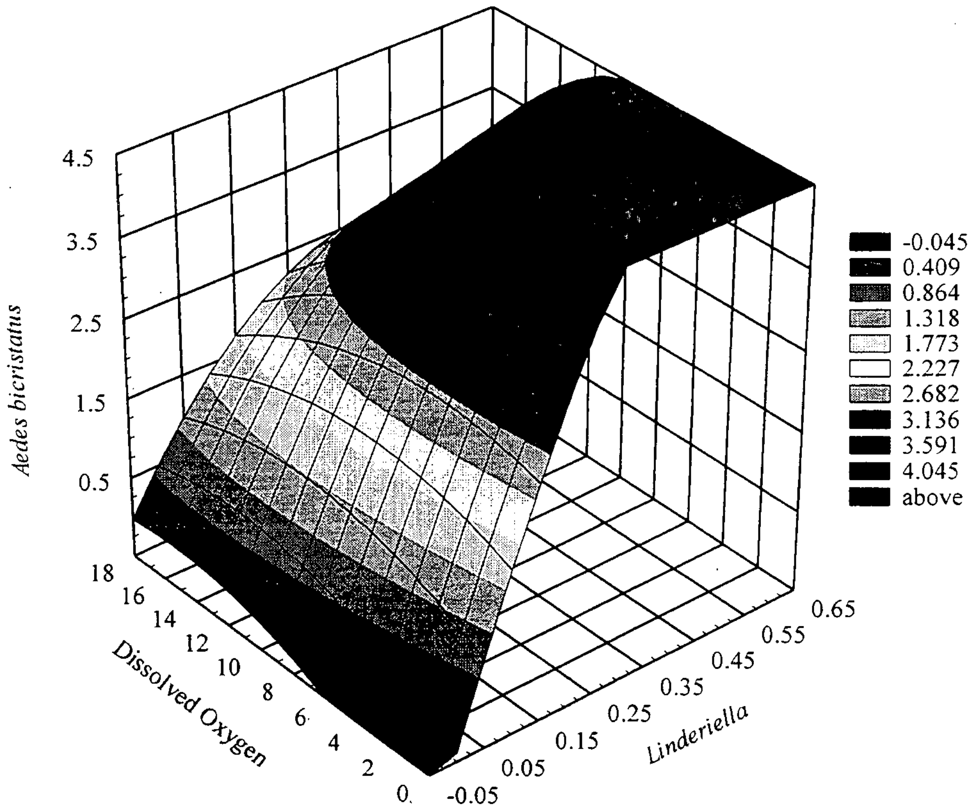

Figure 7. Dissolved oxygen concentration, and numbers of *Aedes bicristatus* and *Linderiella occidentalis* per dip sample.

Aedes increpitus densities were positively correlated with densities of Amphipoda (primarily the scud *Hyalella azteca*), although there probably was no cause-effect relationship involved. Both *A. increpitus* larvae and scuds thrive in lentic habitats with an abundance of decaying leaves.

Some Lake County mosquitoes occupy unique niches. For example, *Coquillettidia perturbans* is the only species with larvae which can remain submerged and attached to aquatic vegetation from which they obtain oxygen. However, many other mosquitoes have adaptations which result in temporal (e.g., Figure 2) or spatial (e.g., Figure 6) partitioning of lentic habitats.

ACKNOWLEDGMENTS

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HYDRILLA ERADICATION IN CLEAR LAKE

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Abstract - On August 1, 1994, hydrilla, a submersed noxious weed was found in Clear Lake during a routine detection survey. A multi agency cooperative program to eradicate hydrilla from Clear Lake was initiated immediately. Use of aquatic herbicides, Komeen™ and Sonar™ has significantly reduced the weed population that currently infests 600 acres of Clear Lake. Survey and detection, treatment, public information and awareness and monitoring are the major components of the eradication project.

Key Words - hydrilla, fluridone, copper, eradication, Clear Lake.

INTRODUCTION

Hydrilla (*Hydrilla verticillata* Royal L.F.), is a noxious submersed non-native aquatic weed. Two different forms of this weed have been found in the United States, indicating at least two separate introductions of this plant. The dioecious form of hydrilla was first identified in Florida in the 1960's, where it was believed to have been introduced in the 1950's. This infestation has since spread throughout the southeastern United States into Texas and California. The monoecious form was first detected in the Potomac River, near Washington D.C. in the 1980's, and has since spread south into a number of the southern states and has also been found in Washington State and California.

Hydrilla was first found in California in 1976, in a 35-acre lake near Marysville, Yuba County. Since then hydrilla has been found in seventeen counties in California. In 1993, monoecious hydrilla was found for the first time in California at an aquatic nursery in Tulare County. A second infestation of monoecious hydrilla was found in Clear Lake, Lake County, in 1994, and in Yuba County in 1997. Hydrilla can form dense mats on the water surface and fill entire water column with plants, impeding water deliveries in irrigation systems, affecting recreational uses and displacing native vegetation.

Various federal, state and local agencies are concerned about the potential negative impacts of hydrilla and contribute monies, manpower or other support to the eradication program. State agencies that contribute include the Department of Boating and Waterways, Department of Water Resources and the Department of Fish and Game. Federal agencies include the Bureau of Reclamation, Bureau of Land Management, United States Army Corps of Engineers and the United States Department of Agriculture - Animal Plant Health Inspection Service and Agricultural Research Service. Numerous local and county agencies are also involved with the program, including the Yolo County Flood Control and Water Conservation District, Yuba County Water District, Imperial Irrigation District and Lake County Public Works Department. When an infestation of hydrilla is detected the local lead agency is the County Agricultural Commissioner. The California Department of Food and Agriculture, Integrated Pest Control Branch is responsible for administering and conducting the statewide eradication program.

Of the seventeen counties in California that have had infestations of hydrilla, eradication has been achieved in nine of them; Los Angeles, Monterey, Riverside, San Bernardino, San Francisco, Santa Barbara, Sonoma, Sutter and San Diego.

Hydrilla was first found in Clear Lake on August 1, 1994, during a routine detection survey conducted by personnel from the California Department of Food and Agriculture and the Lake County Department of Agriculture. The result of the initial delimiting survey indicated 175 to 200 surface acres of infestation along the shoreline of the upper arm of the lake. Since then, areas of infestation have been detected. As of January 1, 1998, approximately 648 acres of Clear Lake were infested with hydrilla. The current level of infestation includes seven new locations found in 1997 totaling 1.5 acres. These sites contain single or scattered plants. A substantial portion of the infestation is still located in the upper arm of the lake. Eleven infested areas are located in the lower arms southeast of the Narrows.

PROJECT COMPONENTS

Survey and detection, treatment, public information and awareness, and monitoring are the major components of the eradication project. The following summarizes the efforts and results associated with these component activities in 1997

Survey Detection

Detection surveys outside the hydrilla eradication area were conducted by the Department's Associated Agricultural Biologist assigned to the district which includes Clear Lake. Surveys were conducted in cooperation with County Agricultural Commissioners and other cooperating agencies' staff. In Lake County, Indian Valley Reservoir (4,000 acres), Highland Spring Reservoir (80 acres), Lake Pillsbury (1,980 acres), Blue Lakes (150 acres) and Thurston Lake (300 acres) were surveyed. In addition, major reservoirs and lakes in Colusa, Napa, Mendocino, Yolo and Sonoma Counties were surveyed because movement of boats from Clear Lake to these bodies of water are a relatively common occurrence. No hydrilla was found during these detection surveys.

Surveys within the eradication area constitute a major portion (approximately 50%) of the staff's field activities. There are two major objectives associated with survey operations within the eradication area. The first objective is to ascertain and evaluate the status of hydrilla growth in the infested areas. This includes determining when plants start to emerge from vegetative propagules in the hydrosoil and monitoring the plant populations prior to and after treatment.

The second objective involves surveillance of non-infested areas of Clear Lake. These surveys are essential because timely detection of new incipient infestations within the eradication area is a critical element in the hydrilla eradication project.

Scheduled water surface and subsurface surveys conducted by project personnel in boats and from the shoreline are the primary methods for hydrilla detection. Surveys are conducted through visual inspection of the water to identify rooted plants or plant fragments floating on the surface. In addition, project staff utilize a multi-pronged grappling hook-type device to retrieve plants rooted in waters where depth and turbidity preclude visual inspections. Scuba divers conduct underwater surveys on a limited basis. These surveys are conducted to establish more accurate information on plant density within a given area. As progress toward eradication continues, underwater surveillance activities will increase to quantify reduction in plant population.

Initial surveys during the 1997 season concentrated on the infested area. The first plant was found on April 28. The initial find of hydrilla in 1996 and 1995 were on May 1 and May 8, respectively. The initial 1997 find was in the location designated as Area #78, on the west side of Quercus Point, whereas the 1996 and 1995 initial finds were in the location designated Area #5, adjacent to the Big Valley Rancheria.

Tuber germination and plant growth were relatively slow during May. However, as water temperatures increased in June, detection of hydrilla increased significantly. The littoral zones (shoreline) in the infested and non-infested areas were surveyed on a two to three week interval through November. Detection surveys in the deep water sections of the lake were initiated in September and continued on a monthly basis until the end of November. No hydrilla has been detected in deep water areas. In areas where hydrilla has been found in the past, one area has not had any plants detected since 1994, one area has had no regrowth since 1995, and four areas have had no regrowth since 1996. All infested areas have had significantly less plant density in 1997 than in the previous three years.

In March, April, May, October, and December, project personnel conducted detection surveys at various access points along Cache Creek, and in September the creek was surveyed by canoe starting at the confluence of Bear Creek and Cache Creek, ending approximately 28 miles downstream at the Capay Dam. Personnel from the Bureau of Land Management (BLM) conducted surveys of Cache Creek from the dam at Clear Lake to the Bear Creek-Cache Creek confluence. No hydrilla has been found in Cache Creek.

Treatment

Initially, in 1997, surface and subsurface applications of Komeen™ were utilized to control hydrilla. Komeen™ was applied at a rate of 16 gallons/acre. The first application was made on June 10, 1997. Infested areas were treated on a three to four week basis. The last treatment was completed on October 27, 1997. Except for seven infested areas which received only Sonar™ treatment, all infested sites were treated at least once with Komeen™. Of these, eleven were treated with Komeen™ only once. Of the eleven areas that required only single treatments of Komeen™, five areas had no subsequent plants found and no further treatments were required, the remaining six areas which received a single Komeen™ treatment were treated with Sonar™.

Of the areas treated with Sonar™, the slow release pelleted (Sonar™ SRP) formulation of Sonar™ was applied in all locations except two. The liquid formulation (Sonar™ A.S.) was applied in Holiday Harbor and Holiday Cove, two relatively small, enclosed areas with little water exchange. Sonar™ SRP was applied with a mechanical fertilizer spreader mounted on the front of the boats. Applications were generally made at a rate of 10 parts per billion (ppb) twice a week for seven weeks. In some cases, applications were initially made at 20 ppb followed by subsequent treatments at 10 ppb. Sonar™ was applied at 20 ppb on a weekly basis in five locations. Three Sonar™ treated areas received a maximum concentration of 100 ppb, two areas received 120 ppb, twenty-two areas received 140 ppb, and two areas received 150 ppb.

Complete control of all submersed aquatic weeds was obtained in all areas treated with Sonar™. Once an infested area was treated with Sonar™ no further use of Komeen™ was required. The first Sonar™ A.S. application was made on June 24, 1997, and the first Sonar™ SRP application was made on July 8, 1997, with treatments continuing throughout the remainder of the growing season. The last application was made on October 30, 1997. Applications were made during this time of the growing season to control plant biomass and stop production of tubers and turions, a major requirement for hydrilla eradication. Tuber and turion production in hydrilla is a response to changing photoperiod. As daylight hours decrease, generally starting around August 1 to 15, production of vegetative propagules increases significantly. From September through November, plant growth slows down and hydrilla transfers its resources and energy into tuber production. Sonar™ interrupts this process effectively and stops production of propagules.

The situation in Soda Bay is an excellent example of the effectiveness of Sonar™ to control hydrilla and reduce the need for additional Komeen™ applications. On July 29, 1996, approximately 20 plants were found scattered around several boat docks in Soda Bay. This 40- acre area was heavily infested with other aquatic weeds including coontail and pondweeds. The entire bay was treated with Komeen™ on August 1, 1996, to control all the plants. This treatment was necessary to facilitate detection of hydrilla once regrowth occurred. On August 12, 1996, Sonar™ was applied to a 14.7 acre section of Soda Bay and continued until October 1, 1996. Complete control of hydrilla was attained and no additional treatments of Komeen™ were required. Additionally, in 1997, there was no regrowth of hydrilla in the Sonar™ treated area.

Potable water intake areas designated as Area #17 (north of Rocky Point), Area #59 (east side of Baylis Point), and Area #67 (south part of Buckingham Peninsula) were monitored for residual fluridone concentrations in water prior to and twenty-four hours after each weekly treatment. Water sample results showed the highest concentration of fluridone detected in the three areas was in Area #59 at 5.95 ppb inside the treatment area after 60 ppb of Sonar™ SRP had been applied over a three week period. The highest fluridone concentration in Area #17 was 2.25 ppb inside the treatment area after 90 ppb of Sonar™ SRP had been applied over a five week period, and, in Area #67, 1.5 ppb inside the treatment area after 40 ppb had been applied over a two week period.

The use of Sonar™ in Clear Lake has significantly reduced the amount of Komeen™ applied to Clear Lake. In 1995, 47,580 gallons of Komeen™ were applied, 20,126 gallons were applied in 1996, and 12,205 gallons were applied in 1997, a 74.3 % reduction in use since 1995.

Public Information and Awareness

Public information and awareness regarding the identification of hydrilla and procedures boaters should follow to prevent its spread in Clear Lake and to other bodies of water is an essential component of the project. Since public access to the lake is not being restricted, this aspect of the project must be maintained throughout the duration of the project.

Informational signs warning the public about hydrilla and reminding them to clean their boats and trailers before leaving the lake have been established at 28 public boat launching facilities. In addition, the three major highways (20, 29, and 175) to Clear Lake area posted with prominent signs.

Informational pamphlets, produced by the Department of Boating and Waterways and the Department of Food and Agriculture, are distributed by project personnel to businesses in the area. Approximately 9,100 pamphlets were distributed to all motels, sporting goods stores, gas stations and many other retail establishments. Additionally, 681 homeowners with lakefront property in Sonar™ treated areas were notified prior to chemical applications with a letter explaining the program and treatment schedules.

Monitoring

A comprehensive monitoring program was continued in 1997 by a group of scientists from the California Department of Fish and Game, the USDA-Agricultural Research Service and the University of California evaluated the impact of eradication activities to non-target organisms. Results of the monitoring activities will be presented in a separate report and distributed to all stakeholders and interested parties.

Progress and Plans for 1998

The hydrilla eradication program has continued to significantly reduce the level of hydrilla in Clear Lake and prevented the spread to other bodies of water. Survey results indicate plant populations are extremely low and scattered in the infested area. Seven new locations of hydrilla were detected in 1997. Four of these locations contained only one or two plants. The remaining three locations contained four to six plants. Although finding additional new sites is disappointing, early detection was achieved and enabled project staff to respond quickly and effectively to prevent further spread within the infested area.

Applications of Sonar™ and Komeen™, when necessary, will continue in 1998. The use of Sonar™ will be expanded to more areas, and treatment protocols will be developed toward optimizing the use of Sonar™ at the lowest possible rate that effectively controls hydrilla.

Exploratory Analysis of the Clear Lake Gnat, *Chaoborus astictopus*

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ABSTRACT

The clear lake gnat, *Chaoborus astictopus,* has a long history of reaching nuisance levels, and being subject to various control strategies. Although recent abundance has been below nuisance levels, this is concern that future outbreaks could occur. For that reason we attempted to determine the factors controlling its population dynamics. We analyzed monthly data on gnat abundance, primarily from the upper arm of the lake. Analysis of the seasonality of abundance data indicated abundance was essentially constant from November through June. We defined the average abundance over these months to be the overwintering index. We examined correlations of the overwintering index and the ratio of sequential overwintering indices (i.e., summer growth) with environmental variables. We found no significant correlation with environment or density. It does not seem possible to formulate a predictive model at this time. We formulated a model for estimating the probability that the gnat population would reach nuisance levels based on the random variability in this population. The probability of the population reaching levels 50 times its current level is in the range of 0.1 to 0.2.

Timeline of Environmentally Important Events within Clear Lake and its Watershed

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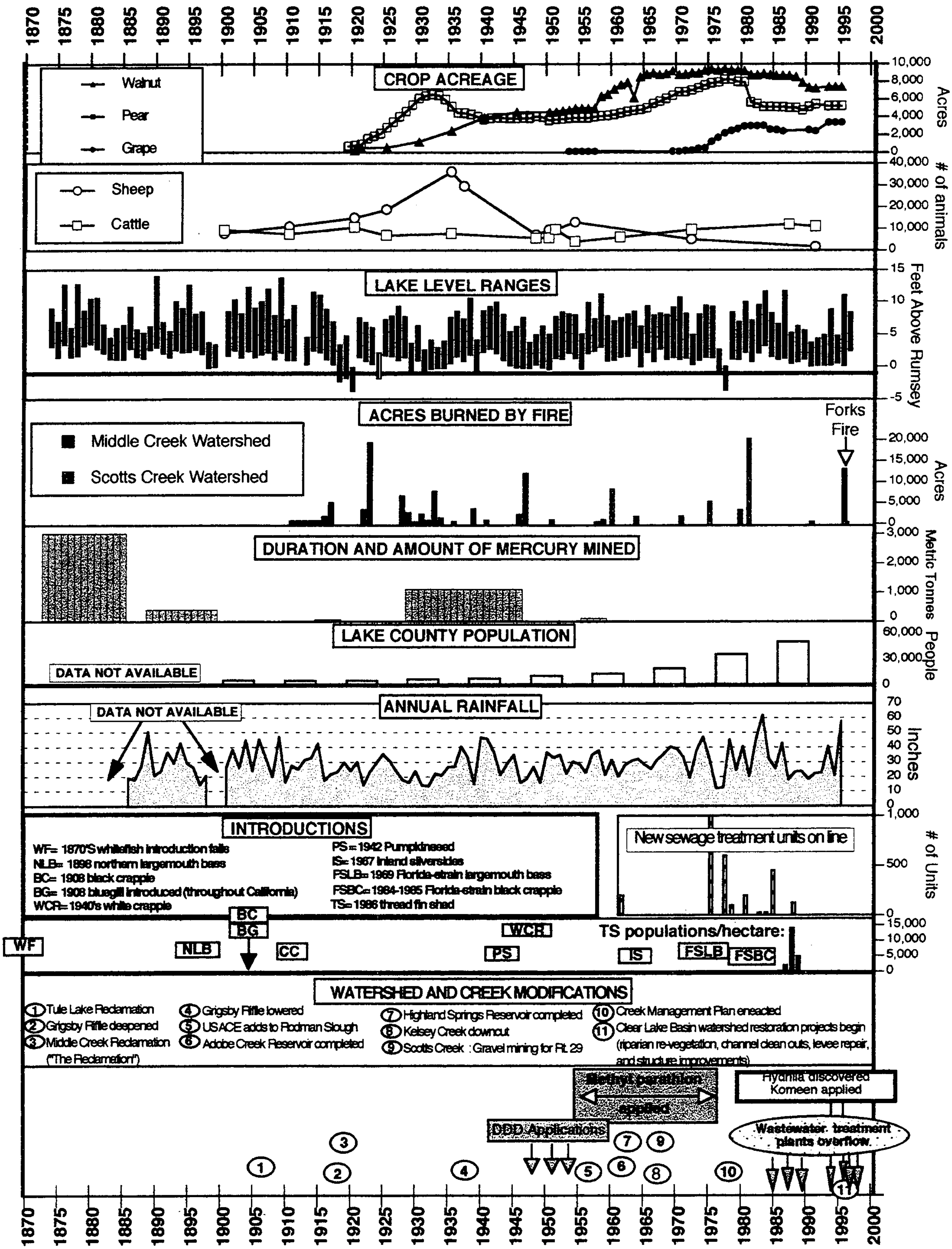
The events displayed here are only some of the potential stressors to Clear Lake's ecology. Studying the impacts of these events on Clear Lake's quality is necessary to gaining a better understanding of the entire watershed and assists managers in enhancing the quality of Clear Lake and it's resources. Human activity in the Clear Lake Basin undoubtedly impacts Clear Lake and the wildlife which utilize it. Shoreline dredge and fill projects, highway and road construction, housing development, wetland reclamation, agriculture, and mining are just some activities which may have had a significant impact on Clear Lake's water quality. Such activities cause increased erosion and decreased wetland acreage. Erosion causes increased nutrient input to the lake. Decreasing wetlands increases the amount of nutrients which actually reach the lake because wetlands trap nutrients before they reach the lake. Development, species introductions, and pesticide applications are some of the activities which impact wildlife which utilize Clear Lake. The introduction of new species can cause populations of native species to decline. The application of pesticides, such as DDD in the late 1940's and mid 1950's, had a significant impact on waterfowl health, thus affecting their population sizes.

Information on this poster was made available to the U.C. Davis Clear Lake Environmental Research Center by several sources and agencies including: Lake County Flood Control and Lakebed Management, Lake County Environmental Health, Lake County Department of Agriculture, Lake County Vector Control District, State of California Department of Fish and Game, Community Development Services and the Mauldin and Goebel archives at the Lake County Museum historical library.

Key Words: Clear Lake, anthropogenic impacts, multiple stresses, historical timeline

Clear Lake Timeline



Developing a Clear Lake Management Plan

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ABSTRACT

A wide range of issues affecting the health of Clear Lake and the surrounding natural resources confronts resource managers and decision-makers. Issues of concern range from effects of algal blooms and aquatic weeds on recreation and tourism to protecting drinking water quality, fisheries, and wildlife. Successful resolution of these issues will depend upon understanding their origin, nature, and interrelationships. The County is developing a Clear Lake Management Plan (CLMP) which will address these issues in a comprehensive document. The Plan will summarize and integrate the findings of the many studies already completed and identify and prioritize implementation measures. The Plan will utilize a collaborative process among the numerous agencies with jurisdictional, trustee, or research interest in the Clear Lake Basin. The process will provide opportunities for active public participation through the Clear Lake Advisory Subcommittee, during public workshops, and ultimately, in public hearings before the County Board of Supervisors. This presentation introduces some of the main themes the Plan will undertake to investigate.

Keywords: Clear Lake, water resources, watershed planning, resource management, general plans

OVERVIEW

The effects of mercury, tick-borne diseases, and erosion control are only some of the varied topics in papers presented at this years Clear Lake Science and Management Symposium. As California's largest natural freshwater lake and possibly the oldest lake in North America, Clear Lake has been the center of a number of prominent research programs. Lake County is developing a plan that will integrate the findings of the studies that have been undertaken, and those in-progress into a single document, the Clear Lake Management Plan (CLMP).

The idea of a Plan was first suggested by the Basin Committee, predecessor of the current Resource Management Committee as a means to resolve Clear Lake water quality and resource issues. While the various Clear Lake studies contain substantial information, the implications of this information are not clearly articulated anywhere. In addition, many of these issues are complex and there exist many different perspectives about how to resolve them. Therefore, a plan is needed to synthesize the available information in a way that will enhance our understanding concerning the nature of these issues and their interrelationships.

The Plan will provide a "framework" for public debate and the basis for developing funding priorities. An integral part of this planning process involves developing an implementation plan and financing strategy. The Plan also can facilitate discussion among the various local, state, and federal agencies with jurisdiction over resources within the county and focus their attention on critical issues and funding needs. A preliminary outline for the Plan is included in Table 1.

PURPOSE OF DEVELOPING A PLAN

Goals

The "Mission Statement" for the CLMP is to; "*Integrate findings and conclusions of various studies into a single document, provide a balanced perspective on issues, and recommend effective policies and implementation measures.*" The goals of this Plan are closely related and include the following:

1. To develop an issues framework, implementation plan, and financing scheme in response to priority water quality-related concerns to benefit Lake County residents.
2. Encourage environmentally sustainable land-use practices.
3. Improve quality-of-life and economic opportunities for county residents.

Objectives
The Plan objectives are an outgrowth of the goals and are listed below:
- Develop projections of future or desired conditions.
- Develop issues framework and priorities.
- Broaden awareness and participation.
- Develop on-going issue resolution process and communications network.
- Develop priorities for issues and implementation options.
- Develop an implementation strategy with specific measures.
- Pursue implementation and financing measures.

Scope
The CLMP will focus upon those issues that, through the process of developing the Plan, are determined high priority. The Water Resources Division is leading the effort to develop the Plan and thus the focus is upon water resource issues, in particular water quality. However, realizing that the majority of water quality issues are related to land-use practices, land-use will be a major focus of the Plan. And while the spotlight is on Clear Lake, many of these issues arise because of activities in the upper watersheds. Therefore, attention will focus upon lake and shoreline land-uses and activities, as well as those in the valleys, uplands and forest areas. The Plan will encompass the entire county, although it will concentrate primarily on the Clear Lake Basin (those areas draining to Clear Lake). Three major components are included in the Plan: *Synthesis and Analysis of Issues, Policy Recommendations, Implementation, and Financing.* Developing the Plan (report) is scheduled for six months. Public meetings, hearings, and workshops may extend this time period. A schedule for carrying out the adopted implementation measures will be developed once the Plan report is completed and financing arrangements are secured. Refer to the Outline (Table 1) for additional scoping information.

Themes
A number of obstacles may be confronted within the course of the plan-development process including, changing leadership, a changing political landscape and associated funding realities, and varying levels of public interest or participation. Therefore, it is essential to plan with these potential obstacles in mind. Efforts should be made to ensure the planning process is collaborative and participatory, and one that promotes a shared learning experience and vision for the future. To this end, the County should take steps to ensure the RMC and its subcommittee's represent the various interests affected by this Plan. Balancing interests and accepting tradeoffs will be a necessary part of this process in order to achieve consensus. Furthermore, it is imperative from the outset to secure a commitment from all parties to embrace project goals and objectives. Most critical, however, is developing an effective implementation and financing strategy.

ISSUES OF CONCERN

A broad range of land-use practices has affected water quality in Lake County, having implications for human health and quality of life. Decisions about how to resolve these issues will also most certainly have repercussions on the local economy. Seasonal proliferation of algae blooms in Clear Lake is an example of one high-profile issue. A variety of activities, such as road building, gravel mining, and floodplain development in the last half-century, have resulted in accelerated erosion and sedimentation.

Aquatic weeds is another issue that has generated considerable concern. During summer months, these plants cover large areas of the lake, thereby impeding swimming and boating. While it is neither feasible nor desirable to eradicate these, mostly naturally occurring, plants, the Greater Lakeport Chamber of Commerce and County are evaluating various approaches to managing aquatic weeds. The Pilot Project is limited to selected sites where geotextile, chemical, and mechanical methods are being tested.

Algae blooms and aquatic weeds not only affect lake ecology and aesthetics, but socioeconomic conditions-recreation, tourism, and the local economy, as well. A number of additional issues are similarly influenced by human factors, as well as by complex and dynamic natural conditions. These issues, which are listed below, will be further defined as the CLMP takes shape.

- Mercury transport and bioaccumulation
- Effects of past and present chemical pesticides, herbicides, and fertilizers
- Algal blooms
- Aquatic weeds
- Hydrilla
- Tick –borne diseases
- Clear Lake Gnat
- Erosion, nutrients, stream and lake water quality
- Drinking water quality
- Fisheries, recreational fishing, species introduction/stocking
- Lake dredging proposals
- Motor-boating and personal watercraft, MTBE
- Stormwater run-off, sewage overflows, wastewater effluent
- Wetlands protection/acquisition, mitigation, enhancement
- Shoreline development
- Wildlife (grebes, game species, endangered species)
- Forest Service and BLM land-uses (logging, fire management, ORVs, grazing)
- Conversion of oak woodlands
- Flood control and drainage facilities, Upper Lake levees
- Lake level management
- Groundwater use/quality
- Water supply
- Water conservation
- Integrating erosion control with community amenities (river-walks and bike trail corridors)
- Creating eco-tourism and educational opportunities along with watershed management.
- Habitat protection and conservation planning (vernal pools, oak woodlands, riparian corridors)
- Natural hazards and technical emergencies with water quality implications (earthquakes, floods, landslides, brown-outs, Y2K)

INFORMATION SOURCES

Numerous studies have been conducted on different aspects of the above issues. University of California, Davis, together with Lake County, produced the Clean Lakes Study in July 1994, investigating the causes and control of algal blooms in Clear Lake. UC-Davis is studying the Sulfur Bank Mercury Mine EPA Superfund site and has also completed a number of studies concerning the fate and effects of mercury in Clear Lake. UC-Davis CLERC, in cooperation with the County and other agencies, is compiling a meta-database for various water quality parameters, in addition to a GIS database.

California Department of Fish and Game is in the process of updating a Fisheries Management Plan for Clear Lake. The Land and Resource Management Plan, completed in February 1995, guides land-uses within the Mendocino National Forest, comprising a large portion of Lake County. The Army Corps of Engineers is developing a Feasibility Study for the Middle Creek Ecosystem Restoration Project and an EIS will follow.

The Lake County Water Resources Division has completed numerous water resources studies, including a countywide Watershed Assessment and Upper Lake Watershed Analysis both published recently. A Basins 2000 Report was completed in December 1997, proposing to treat wastewater utilizing constructed wetlands, after which the effluent would be pumped to the Geysers Geothermal Facility. The Lake County Vector Control District has published a number of studies. The Lake County General Plan and Plan Elements, Zoning Ordinance, Grading Ordinance, and Shoreline Ordinance (Chapter 23) all serve as important policy documents.

Concomitant with these studies, a number of programs are being developed which will be reflected in the CLMP development process. These programs include the Middle Creek Restoration and Basin 2000 Projects referenced above, the Hydrilla Eradication and Aquatic Weed Management Programs, the General Plan Update, and Rodman Slough Acquisition efforts. In addition to these programs, the various tribes in Lake County are addressing water resource issues through their respective environmental programs.

CONCLUSIONS

Community members and resource agency representatives alike have expressed concern about a wide range of water quality issues affecting Clear Lake. The quality of Clear Lake is closely tied to the economy of Lake County, as well as other values important to the quality of life enjoyed by county residents. Various aspects of these issues have been and continue to be studied by the Department of Public Works, Water Resources Division and other County departments. Some issues, such as mercury contamination, are the focus of federal attention, and are being investigated through a series of studies by UC-Davis, Clear Lake Environmental Research Center. Although there are many points of view about how to address these issues, there appears to be a general acknowledgement that these issues are significant and need to be resolved through an "action plan". The Clear Lake Management Plan seeks to achieve consensus through building upon these community concerns, developing an issues-framework, and setting in motion a process for prioritizing issues and implementing strategies.

TABLE 1: CLEAR LAKE MANAGEMENT PLAN DRAFT OUTLINE

I. ***Introduction***

- Goals
- Objectives
- Themes
- Scope
- Work Program

II. ***Review of Background Studies and References***

III. ***Public Review, Agency Coordination, and Approval Process***

- Advisory Committees (RMC, CLAS)
- Technical Task Force
- Board of Supervisors Meetings and Hearings
- Periodic Press Releases
- Public Workshops
- Internet WEB Site
- Presentations to Select Groups

IV. ***Synthesis and Analysis of Issues***

V. ***Implementation (Action Plan)***

- Adoption of Board policies and ordinance amendments
- Adoption of CLMP as General Plan Element
- Support land acquisition funding
- Establish and maintain community/resource agency communications network
- Expand public outreach, including publication of CLMP newsletter
- Develop MOUs with cities to address joint concerns

VI. ***Financing***

- Explore alternatives, including; bond issue, assessment district, and boating fees
- Identify/pursue state/federal grant funding

VII. ***Bibliography***

TABLE 2: PARTICIPATING AGENCIES AND PROGRAMS

Agencies

Lake County

- Public Works Department (DPW)
- Community Development Department (CDD)
- Air Quality Management District (AQMD)
- County Agricultural Commissioner
- Health Department
- Mosquito Abatement District and Vector Control
- Special Districts
- Private Water Companies

State

- UC-Davis Clear Lake Environmental Research Center (CLERC)
- State Department of Food and Agriculture
- UC Agricultural Extension
- CA Department of Fish and Game (CDFG)
- State Water Resources Control Board (SWRCB)
- Central Valley Regional Water Quality Control Board (RWQCB)
- Department of Water Resources (DWR)

Federal

- U.S. Forest Service (USFS), Mendocino National Forest
- Natural Resource Conservation Service (NRCS)
- Bureau of Land Management (BLM)
- Environmental Protection Agency (EPA)
- U.S. Fish and Wildlife Service (USFWS)

Tribal

- Big Valley Tribe
- Robinson Rancheria Tribe
- Scotts Valley Band of Pomo Indians
- Elem Tribe

Programs

- Lake County/USFS Upper Lake Watershed Analysis Partnership
- SWRCB (205j) Watershed Assessment Grant
- UC-Davis Clear Lake Environmental Research Center programs
- CDFG Fisheries Management Plan Program
- Lake County General Plan Update Program (developing Area Plans)
- County (CDD) Fee Exempt Permit trial program (re: agricultural conversions to vineyards)
- USDA/State Hydrilla Abatement Program
- Vector Control District Mosquito Abatement Program
- Lake County/State Boating and Waterways, Aquatic Weed Management Grant Pilot Project
- Middle Creek Ecosystem Restoration Project
- Rodman Slough Acquisition proposal
- Lake County Flood Control and Water Conservation District Projects (assessment district alternatives, flood control facility improvements, ...)
- Lake County Shoreline Protection Ordinance Revisions
- Lake County Groundwater Ordinance
- Tribal Environmental Programs and Pending Grant Project Proposals

Predicting Emergence of Hydrilla from Clear Lake Sediments

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ABSTRACT

Hydrilla *(Hydrilla verticillata)* was discovered in eastern Clear Lake, California, in 1994. The area infested by this weedy species increased from 172 to 261 hectares (425 to 646 acres) from 1994 to 1996. Previous research demonstrated that emergence of hydrilla from underground tubers could be predicted from accumulated degree-days (ADD). Our purpose was to use this information to estimate the timing of emergence for hydrilla plants from tubers in Clear Lake sediments. This knowledge may help managers direct their efforts. We deployed data loggers at Clear Lake sites inhabited by hydrilla and, recorded sediment temperatures once every 0.5 hr from July, 1997 to August, 1998. We calculated degree-days using the single triangulation method. The lower (7.8 C, 46 F, LT) and upper thresholds (27 C, 80.6 F, UT) for tuber sprouting were based on published reports. Mean daily sediment temperature was less than LT during mid-February and began to increase thereafter. Thus, we started calculating ADD from February 11. Predicted emergence of plants from tubers based on Clear Lake ADD indicated that young plants should begin emerging in mid-April, continuing through July. There was very good agreement between these predictions and the presence of hydrilla plants in a 1997 survey. These results indicate that "scouting" efforts should be initiated in early April as hydrilla plants are not likely to be found prior to that time.

Keywords: Clear Lake, Hydrilla, weed management, sediment temperature

Factors Affecting Habitat Segregation Among Treehole Mosquitoes in Lake County, California

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ABSTRACT

Larval surveys indicated *A. sierrensis* and *O. signifera* were the only species of mosquitoes largely restricted to treeholes in Lake County, California. Segregation of the species into treeholes that were temporarily (*A. sierrensis* only) or permanently (both species) filled with water largely resulted from the ovipositional periodicity of each species. Most *A. sierrensis* oviposition occurred from late spring to midsummer, when the largest numbers of treeholes with declining water levels were available as oviposition sites. Oviposition by *O. signifera* females was largely restricted to the hottest, driest months of late summer, which temporally excluded most of the eggs from drought-susceptible habitats. Female *O. signifera* also used cues associated with the permanency of the habitat when selecting oviposition sites. The distributions of the eggs of each species along horizontal and vertical transects did not indicate that interspecific competition affected oviposition site selection. Instead, limited larval food resources were partitioned temporally between the species.

INTRODUCTION

Seasonal flooding places an important environmental constraint on mosquitoes utilizing treeholes as breeding sites in Lake County, California. The area is characterized by a Mediterranean climate with mild, wet winters and hot, dry summers. More than 90% of treeholes dry completely each summer (Washburn and Hartmann, 1992), making most of these habitats seasonally unsuitable for larval development. As a result, the mosquito fauna lacks diversity relative to other areas of North America. *Aedes sierrensis*, the western treehole mosquito, and *Orthopodomyia signifera* are the only Culicidae in the Clear Lake watershed that are largely restricted to treeholes.

Aedes sierrensis is highly adapted for survival through the dry-summer conditions that characterize most California forests. Females oviposit drought-resistant eggs above the water in treeholes during spring and summer (Peyton, 1956). The eggs do not hatch until flooded by rain, usually in fall and winter. Comparatively, much less is known about the adaptive traits which allow *O. signifera* populations to survive the annual perturbation of summer drought. Previous studies (Chapman, 1964; Baerg, 1968) have shown *O. signifera* eggs hatch upon completion of embryonic development, a period of four days at 30°C. Since *O. signifera* eggs lack a resting state, larval populations are susceptible to desiccation in treeholes fated to dry out. This trait apparently restricts successful populations to the low percentage of treeholes that hold water year-round (Zavortink, 1968).

Further studies of the bionomics of *A. sierrensis* and *O. signifera* are needed. *Aedes sierrensis* females are biting pests of humans (Woodward, et al., 1996) and important vectors of *Dirofilaria immitis*, the canine heartworm (Weinmann and Garcia, 1974). *Orthopodomyia signifera* females are not known to bite man, but they readily take bloodmeals from birds (Zavortink, 1968). Some of the causative agents of human disease, such as Western Equine Encephalitis virus, are known to be maintained in nature among populations of native birds (Reisen, 1990). The role of *O. signifera* in the maintenance of medically important arbovirus cycles in sylvan bird populations is not currently known. Factors affecting habitat and resource partitioning between *A. sierrensis* and *O. signifera* are also poorly understood, but could be important considerations for control strategies directed against these mosquitoes.

In this study, the temporal and spatial distributions of the immature stages of *A. sierrensis* and *O. signifera* were determined through the use of ovitraps and larval surveys. The interrelationships among the distributions of the immature stages with climate and the seasonality of lentic habitats in treeholes were analyzed as factors affecting habitat and resource partitioning between *A. sierrensis* and *O. signifera.*

MATERIALS AND METHODS

Study Sites

Most of the study was conducted in a 1.2 ha northern oak woodland (Munz 1965) dominated by interior live oak (*Quercus wislizenii*) and Pacific madrone (*Arbutus menziessii*). The site was located 6.4 km west of the Lake and Mendocino County boundary near Potter Valley, CA. Study periods included the years 1991, 1995 and 1997 (seasonal distributions of oviposition) and 1994 (vertical distribution of oviposition). Weather data were obtained from a U.S. National Weather Service station located 10 km north of the site. Additional data were collected during 1998 from a single, permanent blue oak (*Q. douglasii*) treehole located near Highland Springs Reservoir in Lake County. Rainfall data for the Highland Springs site were obtained from a University of California Integrated Pest Management station located 11 km to the north.

Surveillance of Natural Treeholes for Water and Immature Mosquitoes

A total of 10 water-filled treeholes were located at the Potter Valley site. Polyethylene rulers were mounted in each of the treeholes to measure water depths. Immature populations of mosquitoes were monitored by pipette sampling according to Woodward et al. (1998). All immature mosquitoes collected in the samples were identified to species and returned to the treeholes. Treeholes were considered to be dry when no water could be removed with the pipette. During the seasonal study, water depths and mosquito populations were monitored weekly (April to November) or one to four times monthly (December to March) except during January and December of 1991 (no data were collected).

A census of immature mosquitoes in the permanent treehole at Highland Springs was completed once per month from March to November of 1998. All of the water was siphoned from the treehole and returned to the laboratory. All immature mosquitoes were identified to species and life stage with a dissecting microscope (10x magnification). Treehole water volume was measured and all of the water and organisms were returned to the treehole using a funnel and hose to refill the treehole from the bottom up.

Surveillance of the Oviposition of A. sierrensis and O. signifera

The temporal and spatial distributions of oviposition in the Potter Valley woodland were monitored with ovitraps. A polyethelene cup holding 380 ml of blue oak treehole water (pH =7.56 to 7.93) as an ovipositional attractant was lined with Terri-Wiper® towel strips, the substrate for oviposition. The cup was placed in a black plywood box (ca. 29 x 15 x 18 cm) with a screened (2.5 cm mesh) vertical entrance (ca. 11 x 11 cm) on its front (see Woodward et al., 1996). Seasonal distributions of oviposition were monitored using eight ovitraps located on the ground near the north sides of the bases of tree trunks. The ovitraps were located along two east-west transects of the woodland. Ovitraps were operated continuously from April to December during each year of the seasonal study. Treehole water and liners were replaced once per week. Mosquito eggs on the liners were identified using methods described by Woodward et al. (1996).

The vertical pattern of oviposition was examined with six ovitraps mounted on the north side of the primary stem of an interior live oak with entrances 0.2, 1.0, 3.0, 5.0, 7.5, and 10.0 m above ground. The ovitraps were operated continuously from May 9 to November 2, 1994. Oviposition was monitored weekly using the same methods described for other years of the study.

Statistical Analyses

Seasonal totals of *O. signifera* and *A. sierrensis* oviposition at each of the eight ovitrap stations were compared for each year of the study with a Spearman rank order correlation analysis. Oviposition site selection by *O. signifera* and *A. sierrensis* was further investigated by Chi-square analysis. The frequencies of ovitrap liners with and without the eggs of each species were analyzed for independence using a 2 x 2 contingency table. The vertical ovipositional distributions of each species were analyzed using Kruskal-Wallis ANOVA by ranks followed by multiple range tests. The numbers of eggs collected at each height were compared for the entire seasonal ovipositional periods of each species.

RESULTS

Weather Data

Rainfall and maximum temperature data for the seasonal study are shown in Figure 1A. Rainfall totals were lower in 1991 (57.9 cm) than in 1995 (136.8 cm) and 1997 (106.7 cm). Maximum temperatures peaked during the dry summer months each year. From June 1 to September 30 mean maximum temperatures averaged 32.1, 30.1 and 29.9°C in 1991, 1995 and 1997, respectively. The months of August and September lacked rain each year except 1997.

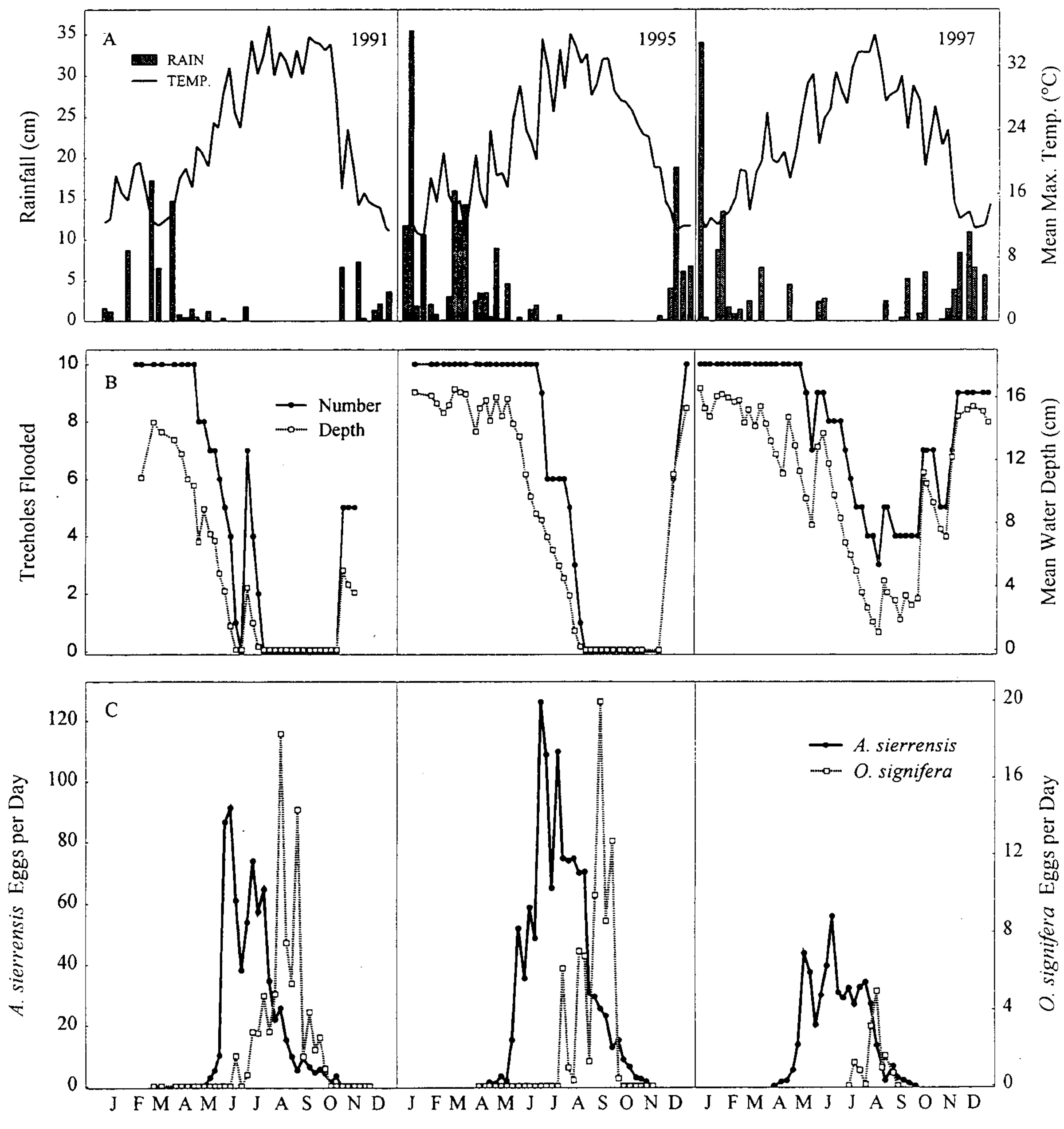

Figure 1. A) Mean maximum temperatures and total precipitation recorded at Potter Valley, CA during weekly sampling periods in 1991, 1995 and 1997. B) Mean water depth and the number of treeholes holding water. C) Mean numbers of *A. sierrensis* and *O. signifera* eggs collected during weekly trap periods.

Surveillance of Natural Treeholes for Water

All 10 of the temporary treeholes surveyed at Potter Valley had exposed horizontal openings at basal locations (mean height above ground ± std. dev. = 0.34 ± 0.38 m) on tree trunks and held light colored acidic to slightly basic water. The pH of the five deepest treeholes (those which held water into August of each year) ranged from 6.19 to 7.26.

Certain treeholes, for example those in some willows (*Salix sp.*) and cottonwoods (*Populus sp.*) remain full of water year-round because they are filled by stemflow from the plant (Zavortink, 1968). The data shown in Figure 1A and 1B indicate water levels in the 10 oak and madrone treeholes monitored in this study were rainfall-dependent. Maximum water depths in the treeholes ranged from 6.4 to 30.0 cm (mean ± std. dev. = 17.1 ± 8.2 cm). Winter rain kept water levels high in all of the treeholes but water depths declined with the onset of summer draught. All of the treeholes dried completely during 1991 and 1995, but following unusual August rain in 1997, three treeholes did not completely dry during summer.

The permanent treehole at Highland Springs held highly basic water (pH = 8.94) with a nearly opaque black color. This treehole was elevated 1.5 m above ground, had a large volume (2.4 l) and a small, vertical opening. Zavortink (1968) previously reported that a large volume and restricted opening were characteristic of permanent treeholes in California. These types of treeholes fill with rain during winter and do not dry completely during summer since the restricted opening reduces evaporation rates. Others (Copeland and Craig, 1992; Woodward et al., 1998) concluded these types of treeholes are usually restricted to above ground locations at the sites of wounds to the bases of secondary tree branches. In 1998, the Highland Springs treehole held >85% of its full water volume in August (Figure 4A) despite a lack of rainfall since early June. The treehole began to refill with water following rainfall in early November.

Temporal Distributions of A. sierrensis and O. signifera Oviposition

Yearly *A. sierrensis* ovipositional totals into the eight ovitraps were 38,725 in 1991, 65,054 in 1995 and 27,761 in 1997 (Figure 1C). Oviposition was detected from April to October, encompassing periods when both temporary and permanent treeholes were available as oviposition sites. Peak ovipositional activity occurred each year as treehole water levels declined with the onset of summer drought. During the three-year study, >83% of the *A. sierrensis* oviposition occurred before water levels in the monitored treeholes reached their yearly minimums.

Conversely, *O. signifera* oviposition was largely restricted to the hottest, driest months of summer when most, or all, of the temporary treeholes had dried completely. Female *O. signifera* laid totals of 4,247, 4,135 and 750 eggs into the ovitraps during 1991, 1995 and 1997, respectively. A single, peak period of ovipositional activity occurred during August or September of each year. During 1991 and 1995 more than 89% of *O. signifera* eggs were oviposited after all of the temporary treeholes had dried completely. In 1997, oviposition peaked on August 14, the same day water levels in the monitored treeholes reached their seasonal minimum.

During the entire three-year study, *A. sierrensis* oviposition was detected an average of 77 days before and 23 days after the yearly oviposition periods of *O. signifera.* The magnitude of *A. sierrensis* oviposition into the ovitraps was >14 times that of *O. signifera.* No other species of mosquito oviposited into the ovitraps during the study.

Spatial Distributions of the Oviposition of A. sierrensis and O. signifera.

All eight of the ovitraps were positive for the eggs of both species of mosquitoes during each year of the study (Figure 2). However female *O. signifera* laid more eggs at station 8 (44% of the total) than at any other location. Station 8 also had the lowest ovipositional total for *A. sierrensis*. However, when yearly ovipositonal totals of the two species were compared for all of the stations there was not a significant correlation for 1991 (R= -0.43, P= 0.29), 1995 (R= -0.52, P= 0.18) or 1997 (R= 0.00, P= 1.00), indicating females of each species chose oviposition sites independently of the other species. This conclusion was supported by an analysis of the distributions of eggs on individual ovitrap liners. The frequencies of liners with *O. signifera* eggs were not significantly different (χ^2 = 0.85, P>0.35) on liners with or without the presence of *A. sierrensis* eggs.

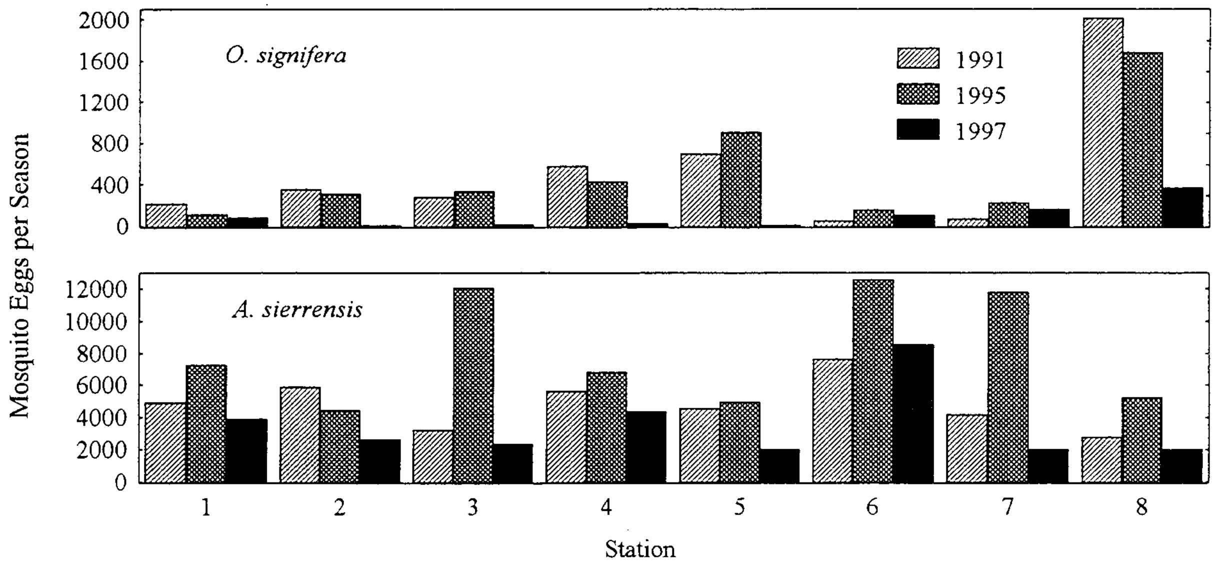

Figure 2. Seasonal totals of *A. sierrensis* and *O. signifera* oviposition at each of eight ovitrap stations during the years 1991, 1995 and 1997.

In the 1994 vertical distribution study, *O. signifera* females laid a total of 1,293 eggs into the ovitraps between June 6 and October 4. All of the eggs were oviposited between 3.0 and 10.0 m above ground (Figure 3), and most (74%) were oviposited in the 5.0 and 7.5 m ovitraps. Although the numbers of *O. signifera* eggs found at middle (3.0 and 5.0 m) and high (7.5 and 10.0 m) heights above ground were not significantly different from each other, all of the elevated ovitraps collected significantly ($P<0.05$) more eggs than ovitraps at low (0.2 and 1.0 m) levels above ground. In the same study, *A. sierrensis* females oviposited a total of 21,538 eggs into the ovitraps between May 24 and October 25. A Kruskal-Wallis ANOVA by ranks indicated there was no significant difference in the number of *A. sierrensis* eggs oviposited at any height ($P>0.65$).

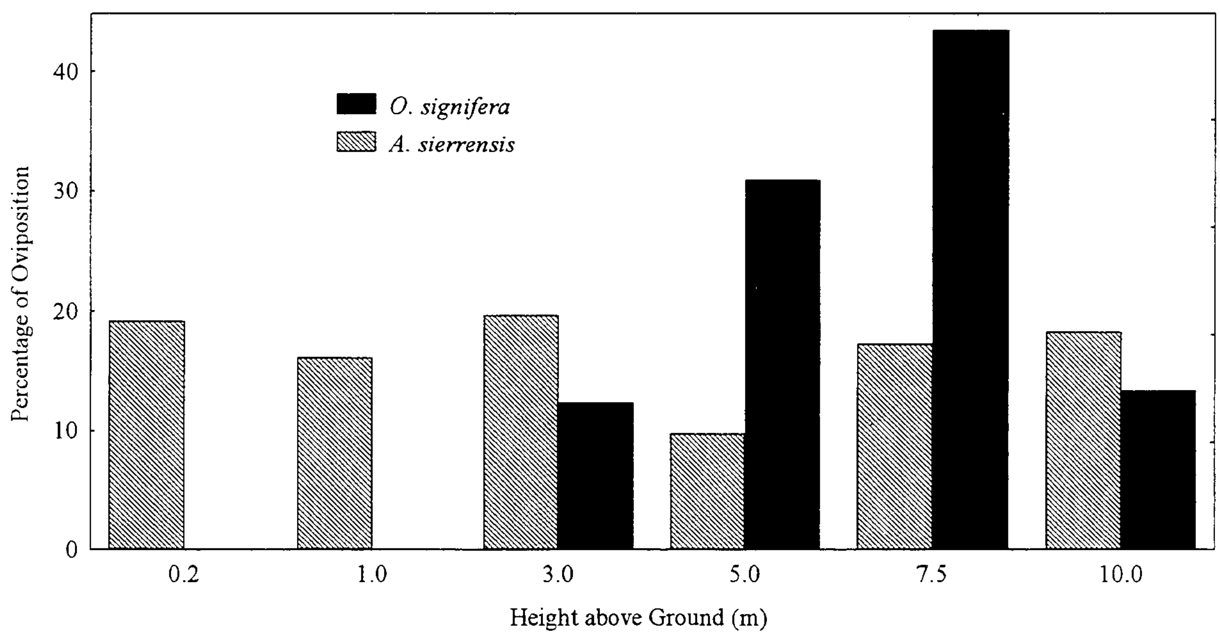

Figure 3. Percentage of O. *signifera* (n=1,293 eggs) and *A. sierrensis* (n=21,538 eggs) oviposition which occurred at each of six heights above ground during weekly trap periods in 1994.

Larval Surveys of Natural Treeholes

Most (>96%) of the larvae found in pipette samples from the 10 temporary treeholes at Potter Valley were *A. sierrensis*, the only species identified during surveys conducted from August until April. Eggs of this species hatched in all of the treeholes soon after they were flooded by rain each fall. Nearly all overwintering *A. sierrensis* immatures had emerged by the end of May each year but small summer generations occurred following rainfall in seven treeholes in 1991 (June 30 - July 18), three treeholes in 1995 (July 20 - August 9) and five treeholes in 1997 (May 28 - July 4 and August 17 - September 4). Two generalists (mosquito species attracted to a wide variety of habitats for oviposition) were also identified during the larval surveys. *Culiseta incidens* immatures were found in four treeholes in 1995 (May 23 - July 5) and five treeholes in 1997 (April 23 to July 16). *Culex stigmatosoma* immatures occurred in four treeholes during 1997 (June 12 - July 31). These data indicate that three species of mosquitoes utilized the treeholes as breeding sites during summer months that overlapped the oviposition periods of *O. signifera*. However, *O. signifera* larvae were never found in any of the treeholes during the study.

Censuses of the Highland Springs permanent treehole (Figure 4B) indicated *A. sierrensis* larval development occurred during winter and the entire population had pupated by May. *Orthopodomyia signifera* larvae exhibited a lack of development during winter and most did not pupate until June and July. Successive summer generations of *O. signifera* adults oviposited eggs which hatched and developed during August to November. *Aedes sierrensis* larvae hatched from resting eggs with the onset of fall rains. These data indicate *O. signifera* larval development was restricted to summer and early fall periods when *A. sierrensis* larvae were not present in the treeholes. This timing resulted in temporal partitioning of larval food resources between the species.

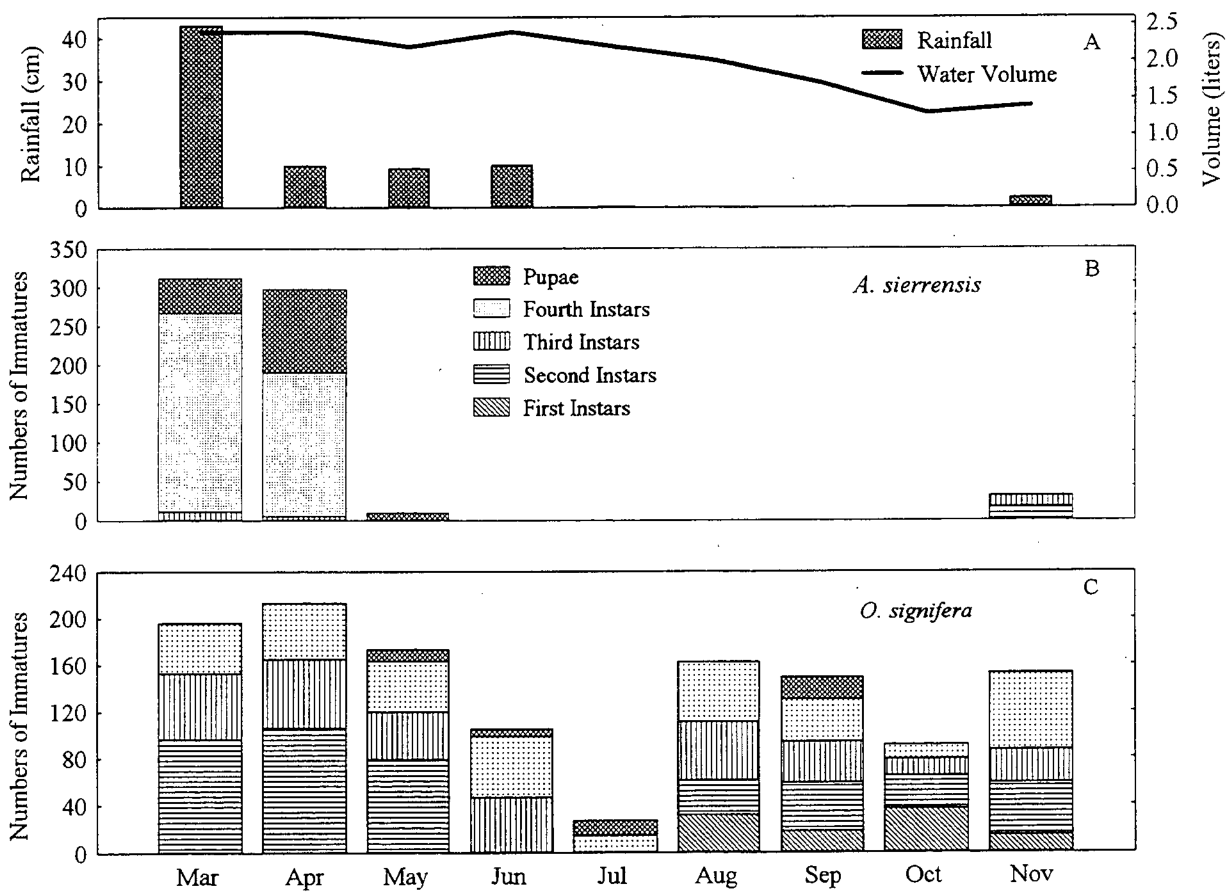

Figure 4. A) Total monthly rainfall recorded at Lakeport, CA from March 1 to November 8, 1998. B) and C) Total numbers of *A. sierrensis* and *O. signifera* immatures found during monthly censuses of a permanent blue oak treehole near Highland Springs, CA.

DISCUSSION

The data shown in Figure 1 indicated the oviposition periods of both Lake County species of treehole mosquitoes were adapted to weather patterns associated with the Mediterranean climate. Most *A. sierrensis* oviposition occurred from late spring to mid-summer when the largest numbers of treeholes with declining water levels were available as oviposition sites for their drought-resistant eggs. The eggs of *O. signifera* lack a resting state, and peak oviposition periods occurred during August and September, temporally excluding most of the eggs from drought-susceptible habitats. Segregation of the species into temporary (*A. sierrensis* only) and permanent (both species) habitats largely resulted from the periodicity of the oviposition of each species. The results also indicated that *O. signifera* females used cues associated with the permanence of the habitat to selectively avoid temporary habitats for oviposition. Three species of mosquitoes utilized temporary treeholes for oviposition during periods when *O. signifera* females were actively laying eggs into the ovitraps. However female *O. signifera* completely avoided temporary treeholes as oviposition sites during these periods. The ovitraps presented a number of cues which mimicked those presented by permanent treeholes. Each provided a darkened, sheltered cavity and held dark colored water with basic pH, traits that have all been associated with natural permanent water-filled treeholes (Zavortink, 1968; Woodward, 1998).

In the more diverse treehole mosquito communities of eastern North America, the permanence of the aquatic habitat (Bradshaw and Holzapfel, 1988) and interspecific competition (Copeland and Craig, 1990) are both considered to be important factors affecting habitat partitioning. For example, Copeland and Craig (1992) concluded that *A. hendersoni* used differential oviposition based on treehole water quality to avoid competition with *A. triseriatus*. Northern California treeholes are known to be resource limited (Colwell et al., 1995) and usually numerically dominated by larval populations of *A. sierrensis* (Woodward et al., 1988). However the horizontal distributions of the eggs of each species shown in Figure 2 did not indicate that *O. signifera* females avoided interspecific competition by ovipositing their eggs at locations that were unattractive to *A. sierrensis*. Data from Figure 3 indicated *O. signifera* females preferred to oviposit at above ground locations. However since *A. sierrensis* females showed no ovipositional preference relative to height above ground, *O. signifera* females would not avoid interspecific competition by ovipositing at particular heights. Traits which caused *O. signifera* females to prefer elevated locations for oviposition were more likely related to the natural spatial distributions of permanent rain-filled treeholes. Permanent treeholes are characterized by a large volume and a restricted opening, physical characteristics which usually occur only at the sites of wounds to the bases of elevated secondary tree branches.

The horizontal and vertical transect data show that *A. sierrensis* and *O. signifera* females choose oviposition sites independently of the other species, without regard to the effects of interspecific competition on their progeny. Instead, the data in Figure 4 indicate that in treeholes utilized by both species the limited food resources are partitioned temporally.

ACKNOWLEDGMENTS

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Micro-Meteorological Considerations, and a Program to Enhance the Number and Locations of Meteorological Monitoring Stations within the Lake County Air Basin

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ABSTRACT

The Lake County Air Basin encompasses the Lake County geographical borders and incorporates a mountainous system of lakes and valleys, which results in very unique and complex terrain. The interaction between mountain slopes and the land/lake interface causes a varied micro meteorological climate. An effort has been initiated by the fire agencies and the AQMD to implement a weather monitoring network throughout the air basin. These meteorological monitors measure rain, wind speed, wind direction, temperature, and relative humidity at 15 minute intervals. To date approximately 15 stations have been implemented. As the network continues to grow, quality assurance and storage programs are being developed and perfected. It is hoped through this extensive data gathering process that an air shed model may be developed for major sub-basins within the air basin, and at a minimum a greater understanding of transport winds will be gained. It is also hoped that by increasing the number of stations through the use of low cost equipment and volunteers, more meaningful data can be available for specific sub-basins. These data have proven valuable to agricultural management, air quality management, soils management, flood control, fire hazard prediction, etc. It is hoped the program will continue indefinitely. A map is provided of stations that are presently in use, and for which data are being collected.

INTRODUCTION

The Weather Information Network (WIN) is a low cost meteorological monitoring partnership between the Lake County Air Quality Management District, Lake County Fire Protection Districts, the California Department of Forestry and Fire Protection, and the cities of Lakeport and Clearlake. Each organization has independent and mutual interests in the collection of atmospheric data. WIN currently consists of 15 weather stations located at Fire Protection Districts and other locations throughout Lake County (see Table 1.). Stations were purchased by LCAQMD, built by LCAQMD staff and/or partner agency staff or volunteers, and are run and maintained by partner agencies, largely on a volunteer basis. A "station master" for each station oversees equipment and software maintenance. WIN is planned to expand with stations in new locations throughout the county, thus bringing in new partners, interests, and creating a more geographically diverse data set. The combination of mutual interests between participants, flexibility in recognizing the needs of each partner, volunteer participation, and the relatively low cost of the stations make this possible.

Table 1.

Station Name	Station Location	Station Master	Station Type	Station Elev.
Lakeport FPD	Lakeport	Al Moorhead	Monitor 2	1343'
Lakeport FPD Parkway Station	Parkway & Hwy 29	Al Moorhead	Wizard 3	1380'
LCAQMD	Lakeport	Taylor McKinnon	Monitor 2	1360'
Upperlake FPD	Upperlake	Mo Fitch	Monitor 2	1343'
Nice FPD	Nice	James Cleveland	Monitor 2	1340'
Lucerne FPD	Lucerne	James Robbins	Monitor 2	1332'
Clearlake Oaks FPD	Clearlake Oaks	Lou Dukes	Monitor 2	1335'
City of Clearlake	Clearlake	Jan Bosse	Monitor 2	1345'
Lakeshore FPD	Clearlake	Scott Gamba	Monitor 2	1350'
Lower Lake FPD	Lower Lake	Charlie Diener	Monitor 2	1372'
South County FPD	Middletown	Carol Grice	Wizard 3	1115'
Kelseyville FPD	Kelseyville	Howard Strickler	Monitor 2	1386'

Boggs Mtn CDF	Boggs Mtn.	Steve Sayers	Monitor 2	3040'
Northwest Plant	Hwy 29	Dean Eichelmann	Wizard 3	1420'
Reynolds Residence	Riviera Heights	Bob Reynolds	Monitor 2	2150'

MATERIALS AND METHODS

Two types of Davis Instruments stations are used; Weather Wizard 3's, which record inside and outside temperature, outside wind speed and direction, and rain; and Weather Monitor 2's, which record the parameters of the Weather Wizard 3's in addition to inside and outside relative humidity, dew point, and atmospheric pressure. Each station consists of the sensors listed in Table 2 (Davis Instruments 1998).

Table 2.

Sensor	Type	Range	Accuracy	Resolution
Temperature	Platinum Wire Thermistor	-50 to 140 Deg. F	± 1 Deg. F	.1 Deg. F
Relative Humidity	Film Capacitor Element	0 to 100 %	± 3 %	1%
Wind Speed	Wind Cups and Magnetic Switch	2-275 MPH	± 5 %	1 MPH
Wind Vane	Wind Vane and Potentiometer	0-360 Deg. 16 Compass Points	7 Degrees	1 Deg, 22.5 Deg Between Points
Barometer	NA	NA	NA	NA
Rain Gauge	Tipping Bucket with Magnetic Reed Switch	0-99"	± 2%	.01"

Instruments are mounted in locations free of wind obstruction and excessive temperature bias, usually atop buildings at least eight feet above roof crest. A few stations are located amidst forest canopies to fulfill particular needs or because of limited resources.

Weatherlink software allows each station to be interfaced with a personal computer. An optical isolator reduces the risk of bad data through radio or other interference. One week of data are stored in a datalogger (intermediate of the instrumentation and personal computer and contained in a small wall display unit) and can be accessed and downloaded from the software. The datalogger records data at fifteen-minute intervals. A small processor in the data logger samples and calculates weather data as follows (Davis Instruments 1997):

1. Temperature is sampled at an eight second sampling rate and the average temperature, high and low temperatures for the archive interval period are recorded in degrees F (and can easily be converted to C).
2. Wind speed is sampled at an eight second sampling rate and the average wind speed and peak wind speed for the archive interval period are recorded in MPH.
3. Wind Direction is sampled at an eight second sampling rate. If wind speed is greater than 0, one of sixteen direction categories is used to describe the wind direction. At the end of the interval, the category containing the most sampling is recorded as the prevailing wind direction. Direction categories can be expressed in degrees or compass direction.
4. Rainfall totals for the archive interval period are recorded at each interval in hundredths of an inch.
5. Relative Humidity is sampled at the time of the archive, every fifteen minutes.
6. Atmospheric Pressure is sampled at the time of the archive, every fifteen minutes, and is measured in inches (and can also be recorded in mb, mm, or hpa).

Downloaded data are compiled in monthly files in the personal computer, thus creating a seamless data set from month to month. Basic analytical and graphing functions are included as part of the software, and data can be exported in tab delimited format for import into spreadsheets, statistical packages, databases, or word processing programs.

Each WIN station is kept on a regular downloading schedule by participating volunteers and LCAQMD staff. Every other month, these data are exported (one month of data per export file) via floppy disk, zip disk or e-mail, and compiled in central databases by the Air Quality Management District. The central database filters all but select atmospheric parameters. Each station has its own file within this database. At the end of each one-year period, a new database using the same or improved format will be employed to store new data. Eventually, we hope to make each database available to public access via the internet. All monthly export files are retained (in tab delimited format) in folders for availability to data requests. Individual station masters may provide data directly to OES, fire agencies, and other organizations in times of emergency.

Several measures were taken for quality assurance in the building of these stations. Each instrument was tested before leaving stock. Wind direction sensors were aligned to 0 degrees along the mounting arms, and all station times were set according to an atomic clock such that they record and archive data within 5 seconds of each other. All wires were insulated from solar radiation with shrink tubing or drip tubing. Many stations have undergone on site outside temperature and humidity calibrations. Although LCAQMD staff have audited each station to ensure proper initial set up, a regular quality and assurance audit procedure is currently being developed but has yet to be implemented. The ongoing inspection of equipment and software by the station masters is crucial to continuous operation.

DISCUSSION

Giruex and Lawton (1980) compiled data from a micro-meteorological network study in conjunction with LCAQMD and environmental contractors ES&S that consisted of eighteen weather stations throughout the Known Geysers Resource Area (KGRA). These stations collected data for a period of two years from 1976-1978, at a cost of around $300,000. These data were used to generate windflow patterns throughout the KGRA (which includes most of the Lake County Air Basin) and identify "mini-airsheds" and their respective wind motion characteristics and patterns. Five "quasi-homogeneous" mini-airsheds were defined. Weather Information Network stations are located in two of these five mini-airsheds; Clear Lake (fourteen stations) and Putah Creek (one station).

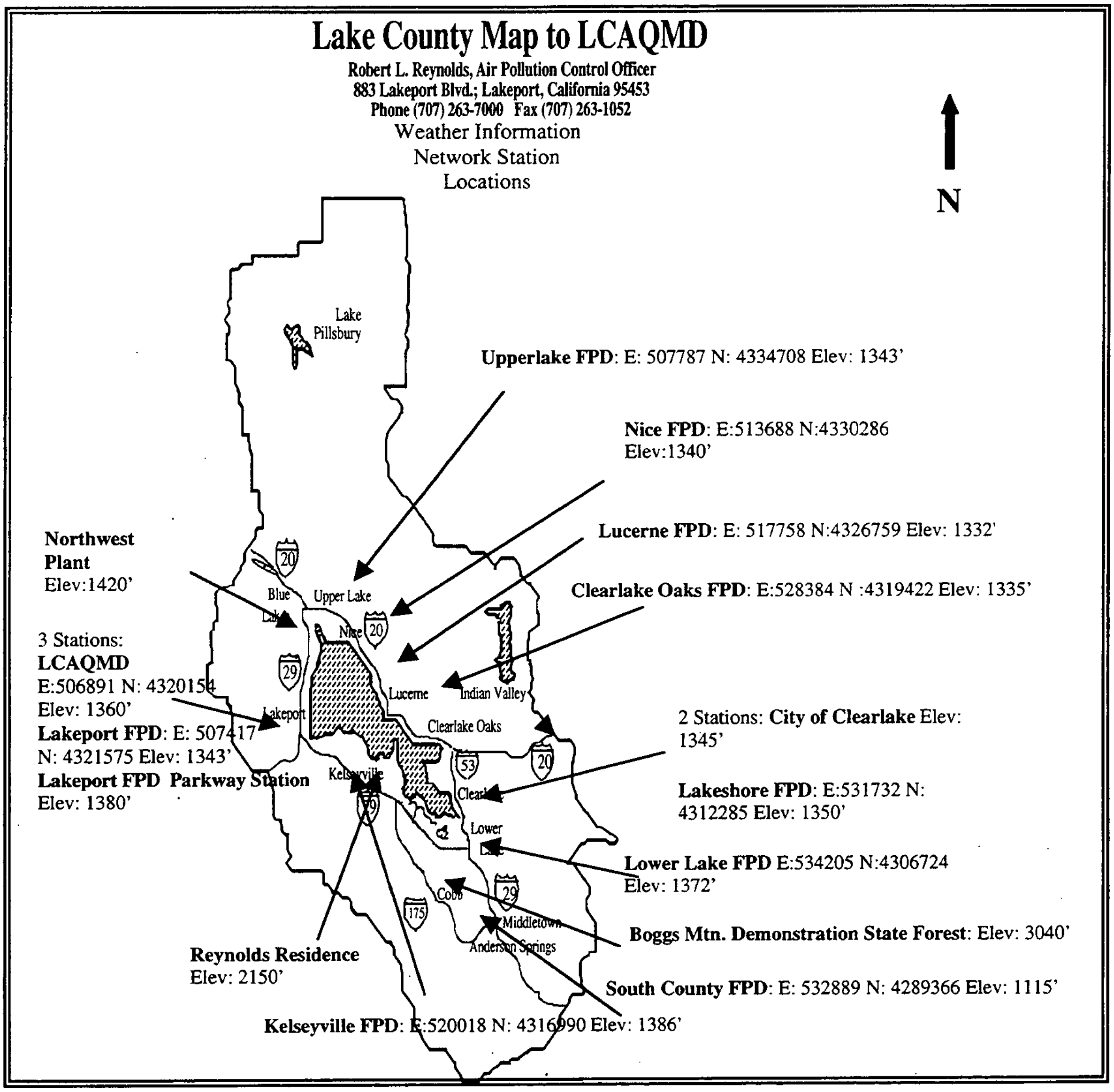
The station located in the Putah Creek mini-airshed (South County FPD) will not, by itself, provide enough data to further the characterization of wind flow patterns. However, historical data has been collected through GAMP on this mini-airshed and air flow patterns within it are well established. The other fourteen stations, however, may prove to be quite useful in furthering air flow pattern characterization and may be used to generate an air flow model for the Clear Lake mini-airshed.

The evolution of an air shed model or furthering air flow pattern understanding would aid resource managers. The prediction and monitoring efficiency of air pollutant movements in the Clear Lake mini-airshed would be greatly enhanced, possibly identifying areas of stagnation or pooling, both nocturnally and diurnally. Planning of agricultural burning, prescribed wildland vegetation burning, and variance events could be aided with added predictability, and their respective impacts upon population centers could be significantly reduced.

Rainfall data collected through WIN may assist in the short term prediction of flood events and flow rates entering Clear Lake. In addition, precipitation rate and distribution patterns throughout the region may be better understood and contribute to long term flood and flow forecasting, erosion studies, micro climate identification, and the generation of a fire prediction model.

Most importantly, the WIN data sets will provide for retrospective data analysis over long periods of time.

While the WIN data set may prove to be a useful resource in the future, real time data has already proven useful in a number of applications. Fire weather prediction and forecasting has been enhanced throughout Lake County with the availability of real time data at the participating Fire Protection Districts, and the ability of the Districts to communicate the information to their pertinent networks. The city of Lakeport has used real time WIN data to monitor potential odor movement in the management of their sewer treatment plants.



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A Whole-Lake Sulfur Budget of Clear Lake, California

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ABSTRACT

Despite the uncertainty of quantifying the inflow from the mine, a whole-lake S budget was estimated by examining all sources of SO_4^{2-} input to Clear Lake. Atmospheric input and spring loading of SO_4^{2-} are estimated to be very small portions of the S budget, while the effect of seasonal change on the S budget shows its significance under the low flow rate (1,635 m^3 day^{-1}) of seepage from the mine site. Stream loading and overflow/runoff from the mine site during the wet seasons is estimated to yield about 3,000 MT SO_4^{2-} per year into the lake. The SO_4^{2-} output from the outlet of Clear Lake is estimated at about 2,000 MT SO_4^{2-} per year.

Sulphur Bank Mercury Mine is predicted as the major source of sulfate and acidity loading when the seepage from the mine site is high (4.0×10^4 m^3 day^{-1}). Results under high throughflow rates indicate that 94 % of SO_4^{2-} input, as high as 43,200 MT per year, is from the Herman Pit. The net sulfate reduction rate is about 700 nmole cm^{-2} day^{-1}, and the calculated percentage of reduced sulfides re-oxidized back to sulfate is about 50 %. The magnitude of the sulfate input and removal by sulfate reduction demonstrates the specific importance of sulfate in the lake's ecosystem.

INTRODUCTION

Over the past 10 years, research in Clear Lake by issuing biogeochemistry of mercury, phosphate and iron, and paleo-processes in the sediments has led to an improved understanding of multiple stresses on the lake. More and more evidence has shown that sulfur may play some key roles in water and sediment chemistry. Several hypotheses of S effects have derived from the previous research (Richerson et al. 1994, 1997, Suchanek et al. 1997, Mack 1997 and Li 1998), which may explain some phenomena of high P concentration, iron limitation to algal growth, mercury precipitation, and other dramatic changes in the lake.

The purpose of this paper is to estimate a whole-lake S budget in Clear Lake. Construction of a whole-lake S budget has emphasized inputs and outputs of SO_4^{2-}, primarily because SO_4^{2-} is the most abundant and most commonly measured S species (Cook and Kelly 1992). Estimating SO_4^{2-} input from the Mine Site is challenged by uncertainty and difficulties of this complicated systems which consists of sulfide-rich waste rock, pit water, wetland, and throughflow underground. Estimating the whole-lake S budget is also critical to our hypotheses mentioned above. This paper examined all the possible SO_4^{2-} sources input into Clear Lake, and sketched the whole-lake S budget, which will enhance our understanding of the specific importance of sulfate in the lake's ecosystem.

ATMOSPHERIC LOADING

The SLAMS collected wet deposition at the Lakeport station and the data covered the period of 7/84 to 2/89. The estimate of annual SO_4^{2-} loading from rainfall is the following:

Rn.Yr.(Jul – Jun)	7/84 – 6/85	7/85 – 6/86	7/86 – 6/87	7/87 – 6/88	7/88 – 2/89
Rainfall (inch)	31.46	42.28	17.94	23.62	9.9
SO_4 Conc.(mg/l)	0.31	0.21	0.35	0.19	0.34
SO_4 Load (kg)	44127	40370	28283	19994	15009

Atmospheric sulfur input into Clear Lake was between 0.113 and 0.255 g (S) $m^{-2}a^{-1}$ during the period from 1984 to 1989. Compared with other northern temperate lakes (Cook and Kelly 1992), it is a small portion and may not be significant in the whole-lake S budget. It is noted that SO_4^{2-} from wet precipitation during the drought period was only a half of that in the normal years.

UCD measured SO_4^{2-} concentration from wet precipitation at SBMM site and Kelseyville on rain events of April 1998. Among three measurements, two were not detected at the detective level (2.5 mg/l) and one was 2.6 mg/l. Two matrix spikes were analyzed with samples and the recoveries were both 93 %. Based on SLAMS data, the probability of SO_4^{2-} >2.5 mg/l is less than 1 %, and > 2.0 mg/l less than 3 %. If atmospheric sulfur deposition does increase to the level UCD measured in 1998, it would have some impact on water and sediment chemistry of the lake.

STREAM LOADING

Table 1 and Figure 1 show the relationship of stream flow and sulfate concentration. Regression analysis indicates that sulfate concentrations of steams have a fairly linear relationship ($r = 0.6$) with log stream flow.

For February 1998, the average concentration of SO_4^{2-} in all streams is 5.4 mg/l, for March 1998, 8.5 mg/l, for May 1998, 9.4 mg/l.

Based on Richerson 1994, average annual streamflow of all streams (not including dowmnstream rivers) is 321.5 cfs. Average SO_4^{2-} concentration in the streams is estimated at 6.0 mg/l. The estimate of annual SO_4^{2-} loading from tributaries is calculated by the following formula:

$$SO_4^{2-} \text{ stream loading} = 0.0283 \times 321.5 \text{ (cfs)} \times 86400 \times 360 \times 6.0 \text{ } (SO_4^{2-} \text{ g/m}^3)$$
$$= 1,700 \text{ (MT/yr)}$$

Table 1. Streamflow Rate and SO_4^{2-} Loading

	SO_4^{2-} Conc.(mg/l)*			Average Annual	Streamflow (cfs)		
	Feb. 98	Mar. 98	May 98	Streamflow (cfs)#	High[1]	Medium[2]	Low[3]
Adobe Cr.	4.2-7.3	14.0	16	12.0	2502.0	78.9	15.7
Cole Cr.	4.6-6.3	7.7	4.3	--	449.1	93.7	39.4
Kelsey Cr.	3.2-4.6	3.0	6.1	76.0	1360**	92**	48.9
Middle Cr.	6.7-8.3	12.0	15	91.0	1500**	--	--
Schindler Cr.	4.9-5.6	14.0	21	--	--	15.6	3.3
Scotts Cr.	4.2-4.4	9.7	9.7	89.0	2420**	86**	--
Siegler		4.4				25.3	

*: UCD data, 1998. **: DWR data. #: Richerson, P.J. et al. 1994.
[1]: UCD data: Feb. 19, 1998 data; [2]: March 11, 1998; [3]: May 19 – 21, 1998.

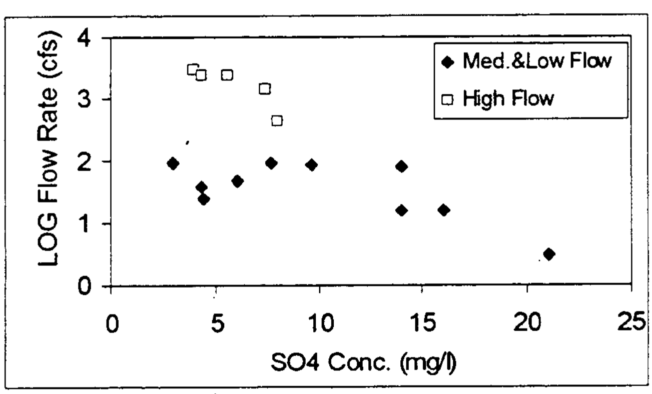


Figure 1. Relationship of SO_4^{2-} concentration to flow rate of streams in Clear Lake.

SPRING LOADING

Chemical composition of spring water along the shore of Clear Lake was analyzed by USGS in late 1970's (Thompson, et al. 1981). Table 2 gives the result of SO_4^{2-} concentrations in these springs.

A limited amount of data exists for spring and groundwater flows. Based on the estimate of Richerson 1994, annual contribution of groundwater is 1.355×10^6 m^3/yr. If the average SO_4^{2-} concentration is estimated at 3 mg/l, the annual SO_4^{2-} loading from springs and groundwater is

$$SO_4^{2-} \text{ groundwater loading} = 1.355\times10^6 \ m^3/yr \times 3.0 \ (g/m^3) = 4 \ (MT)$$

SULFATE IN THE MINE SITE

Sulphur Bank Mercury Mine has become major source of sulfate and acidity loading since it began to be operated by open pit methods in 1927. SO_4^{2-} in porewater of the sediments exhibits the clear signal of a point source sulfate originating from the Sulphur Bank Mercury Mine. Sulfate concentrations in the impoundment water, surface runoff from waste piles, and other pits and ponds in the mine site are shown in Table 4, groundwater in Table 4, and wetland in Table 5.

Herman Pit

Approximately 10^6 m^3 of sulfide rich overburden and waste rock were excavated in the course of open pit operation, leaving a flooded impoundment of 8.60×10^5 m^3, Herman Pit. UC Davis measured SO_4^{2-} concentrations at different depths in the pit and found the SO_4^{2-} concentration is almost uniform with depths, 2500 ~ 2800 mg SO_4^{2-}/L with pHs around 3. So, the storage of SO_4^{2-} in Herman Pit is estimated to be 2000 ~ 2500 MT.

Herman Pit surface water was about 11 to13 feet above Clear Lake during the period of May and June, 1996; and even more higher in wet seasons. The elevation differences make it possible that seepage of water from these pits may make a contribution of SO_4^{2-} to Clear Lake. pH and floc surveys on the early summer of 1996 by UC Davis found that high floc concentrations and conductivity, and low pH and redox values are characteristic of offshore of the mine, indicating strong outflow of water that originates from the mine site at this location.

A fluorometric dye trace study was conducted by Schladow and Massoudi (1997), suggested the possibility of direct hydraulic connection between the Herman Pit and Oaks Arm. Arguing against previous estimates (Columbia Geoscience's (1988) of 20 gpm, Veatch's (1883) of 300 gpm, and dewatering records of 140 gpm), their best estimate suggested a value as high as 8,600 gpm (4.7×10^4 m^3 day^{-1}), not below 7,000 gpm (4.0×10^4 m^3 day^{-1}), which was one to two orders of magnitude larger than previous estimates of throughflow.

SBMM Wells

Most SBMM wells contain high concentration of SO_4, about 3,000 mg/L in groundwater from the impoundment to the Oaks Arm, data being shown in Table 7. According to the Rhodamine measurement by Schladow and Massoudi (1997), the main direction of throughflow from the Herman Pit to the Oaks Arm may be northwest, towards along wells of SR-8, MW-7, MW-5, and MW-2, in which high dye concentrations were

Table 2. SO_4^{2-} concentrations in springs of Clear Lake

Location	Sample No.	Collection Date	SO_4^{2-} Conc.(mg/l)	Flow (L/min)
Moki Beach Dorn Bay	CLW 79	11/29/76	1	80
	LJ-78-10	9/6/78	7	
Big Soda Spring	GT27Rm74	9/10/74	1	
	LJ-78-11	9/6/78	2	
Horseshoe Spring	CLW 15	12/6/74	1	80
	LJ-78-12	9/7/78	1	
Riviera Beach	CLW 103	10/19/76	1	
	LJ-78-13	9/7/78	1	
Konocti Bay	CLW 17	12/6/74	<0.5	
	LJ-78-15	9/8/78	5	
Kono Tayee Well	CLW 83	12/4/76	31	--
Lower Lake off Lucerne	CLW 58	6/24/76	11	
	LJ-78-14	9/7/78	11	
Near Anderson Island	LJ-78-16	9/8/78	11	

detected. This is consistent with the observations of high Hg floc area with low pH along the shore of the Mine (Brister et. al., 1997).

Wetland

The wetland area north of the mine may represent another source of SO_4^{2-} to Clear Lake. Surficial sediments contain 20,000 to 60,000 mg SO_4^{2-}/kg. Table 6 shows the SO_4^{2-} concentrations of surface and porewater in the different locations of the wetland. SO concentrations in wells near the wetland also have values as high as 2,000 mg/L. If the area of the wetland is about 1 km^2, the storage of SO_4^{2-} in the water and sediments (top 10 cm) is about 1000 MT.

Although the wetland is separated from Clear Lake by a levee, during the wet season stormwater and seepage from the wetland may have some impacts on SO_4^{2-} budget of Clear Lake.

Table 3. SO_4^{2-} concentrations in Pits and Ponds of the Mine Site*

Map ID#	Location	Sample Description	Collection Date	SO_4^{2-} mg/l	pH
2	Herman Pit	Surface water	June 19, 1998	3,200	
2	Herman Pit	Surface water	Feb. 25, 1998	2,100	
2	Herman Pit	Surface water	Sept. 15, 1997	2,900	
2	Herman Pit	Surface water	May 8, 1996	1,800	2.9
2	Herman Pit	Surface water	June 27, 1976	4,430	3.5
3	Northwest Pit	Surface water	Feb. 17,1998	1,100	
3	Northwest Pit	Surface water	Sept. 15, 1997	3,800	
3	Northwest Pit	Surface water	May 8, 1996	1,400	2.5
6	Overflow channel at interface		Feb. 26,1998	1,800	
6	Overflow channel just below HP	Run-off water	March 5, 1996	2,500	3.1
7	Overflow channel just below lake	Run-off water	March 5, 1996	2,200	3.1
8	Mine beach 10m N along rip rap	Water	March 5, 1996	120	5.5

Table 4. SO_4^{2-} concentrations in SBMM wells*

	5/13/96	9/18/96	3/24/97	9/15/97	12/8/97	4/7/98	6/11/98
MW-1	407	445					
MW-2	3,490	3,030	3,500	2,700	3,600	4,500	3,000
MW-3	--	6,050					
MW-4	4,240	4,750			2,900		
MW-5	3,180	3,220					3,300
MW-6	2,840	2,740					
MW-7	2,790	2,910					
MW-8	179	171	180	490	520	200	330
MW-9	2,690	2,910					
SB-8d	3,040	3,890	4,100	3,300	320	3,800	4,200
SB-8s	2,530	--				2,600	2,700
SB-9	1,340	1,850	2,000	2,400	1,800	1,300	1,900

Table 5. SO_4^{2-} concentrations of surface and porewater in the wetland*

	NW-01 (mg/l)		NW-02 (mg/l)		NW-03 (mg/l)		NW-04 (mg/l)	
	PoreW.	Surf.W	PoreW.	Surf.W	PoreW.	Surf.W	PoreW.	Surf.W
4/15/98	4450	1050	170	190	120	260	200	200
2/17/98	--	170*	--	330**	--	--	--	--
4/2/97	5,200	1,350	330	320	2,200	660	270	330
5/27/97	5,600	3,400	420	720	2,600	860	560	430
12/8/97	1,500	460	1,400	330	3,000	960	660	250
6/28/96	4,000	2,900			3,800			

* Data from the Center of Clear Lake Research, UC Davis.

ESTIMATING SO_4^{2-} INPUT FROM THE MINE SITE

SO_4^{2-} Input by Overflow/Stormwater During Wet Seasons

Rainfall during the wet seasons causes a great input of SO_4^{2-} loading to the Oaks Arm near the mine. From Figure 2, which covers the period of forty months from May 1994 to August 1997, it is clear that rainfall is strongly associated with log concentration of SO_4^{2-} in porewater (A), and has some effects on SO_4^{2-} concentration in deep water of OA-01 (B).

Two wet seasons and increases of SO_4^{2-} concentration in the porewater on OA-01 can help us to estimate SO_4^{2-} input from the mine site. Table 6 gives rainfall data and SO_4^{2-} concentration increases during the wet seasons. Figure 3 show the increment of SO_4^{2-} from the mine site (OA-F) to the center of Upper Arm (UA-04).

Increment of SO_4^{2-} in porewater during the wet season can be integrated to estimate SO_4^{2-} input from the mine site including pit overflows and wetland stormwater. We assume that *f(x)* is a relationship function of porewater SO_4^{2-} gradient with distance from the mine site.

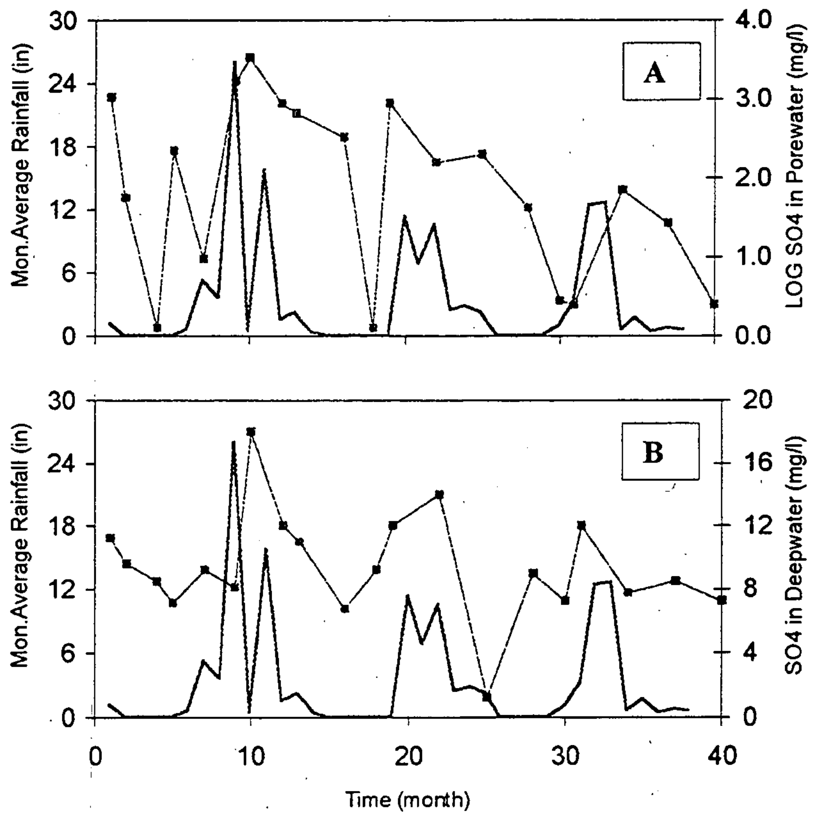
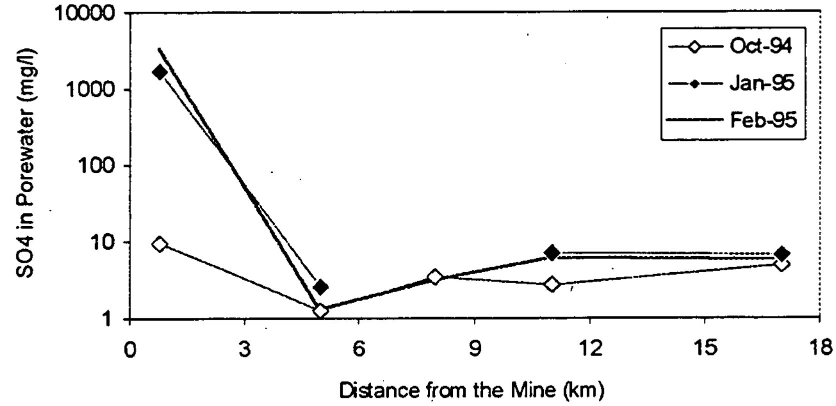


Figure 2. Relationships of rainfall and log concentration of SO_4^{2-} in porewater (A), and in deep water (B) at OA-01, Clear Lake. Time covers the period from 5/94 to 8/97. Dark square represents SO_4^{2-} concentration, solid line rainfall.

Table 6. Rainfall data and SO_4^{2-} concentration increases during the wet seasons.

Season I	Rainfall (inch)	SO_4^{2-} in Porewater (mg/l)	Season II	Rainfall (inch)	SO_4^{2-} in Porewater (mg/l)	
		OA-F			OA-F	OA-01
10-94	0.55		10-96	0.99	330	2.8
11-94	5.30	9.3	11-96	3.28		2.5
12-94	3.65	↓	12-96	12.38		
1-95	25.96	1700	1-97	12.50	↓	↓
2-95	0.40	3300	2-97	0.61	2800	70



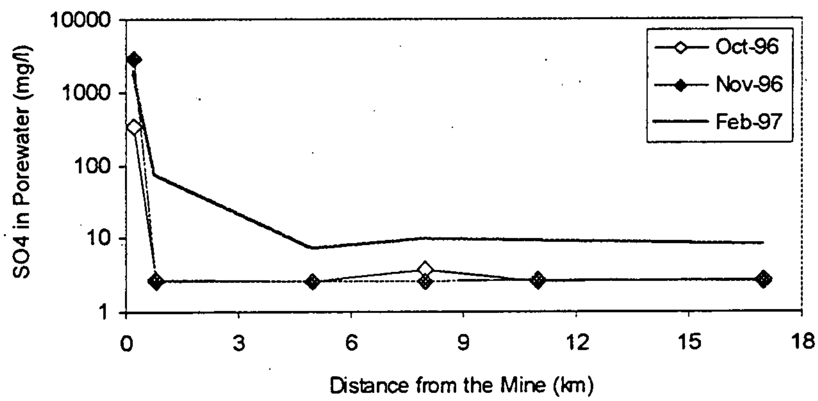


Figure 3. Increment of SO_4^{2-} from the mine site (OA-F) to the center of Upper Arm (UA-04). Increment of SO_4^{2-} in porewater can be integrated to estimate SO_4^{2-} input from pit overflow and wetland stormwater during the wet seasons.

<u>Estimate 1: SO_4^{2-} increment during 94-95 wet season</u>

$$f(x) = -0.7854\,x + 4000$$

Mass of SO_4^{2-} can be integrated along the x axis:

$$\int_0^{5000} (-0.7854x + 4000)dx$$
$$= -0.5\,(0.7854)\,x^2 + 4000\,x = -9{,}817{,}500 + 20{,}000{,}000 = 10{,}182{,}500 \text{ g } SO_4^{2-}/m^2$$

From the mine site to OA-04 within the sediment depth of 5 cm, the increment of SO_4^{2-} is:

$$10{,}182{,}500 \text{ g } SO_4^{2-}/m^2 \times 2400 \text{ m (width)} \times 0.05 \text{ m (depth)} = 1222 \text{ MT } SO_4^{2-}$$

For the other part of the lake, the average SO_4^{2-} increase in the porewater was 1 ppm. The increment of SO_4^{2-} can be estimated as:

$$1 \text{ g } SO_4^{2-}/m^3 \times 176 \times 10^6 \text{ m}^2 \times 0.05 \text{ m} = 88 \text{ MT } SO_4^{2-}$$

So, during the wet season of 94-95, total SO input from the area of the mine site was 1310 MT of SO_4^{2-}.

Estimate 2: SO_4^{2-} increment during 96-97 wet season

$$f(x) = -3.75\,x + 3000$$

Similar integrated method (see above) was used in this case. From OA-01 to OA-04 within the sediment depth of 5 cm, the increment of SO_4^{2-} is:

$$3{,}570{,}600 \text{ g } SO_4^{2-}/m^2 \times 2400 \text{ m (width)} \times 0.05 \text{ m (depth)} = 429 \text{ MT } SO_4^{2-}$$

For the other part of the lake, the average SO_4^{2-} increase in the porewater was 6 ppm. The increment of SO_4^{2-} can be estimated as:

$$6 \text{ g } SO_4^{2-}/m^3 \times 176 \times 10^6 \text{ m}^2 \times 0.05 \text{ m} = 53 \text{ MT } SO_4^{2-}$$

So, during the wet season of 96-97, total SO_4^{2-} input from the area of the mine site was 596 MT.

SO_4^{2-} input from seepage of the Herman Pit

Because of the disparity in the flow rate of throughflow from the Herman Pit to the Oaks Arm, Clear Lake, we calculated SO_4^{2-} mass input under both scenarios.

Scenario 1: Seepage = 1,635 m^3/day

The highest flow rate of seepage from the Mine, according to previous estimates, is 300 gpm (1635 m^3/day). The SO_4^{2-} concentration of the flow is 3,000 mg/L. The SO_4^{2-} input from seepage can be estimated as:

$$3{,}000 \text{ (g } SO_4^{2-}/\text{ m}^3) \times 1635 \text{ (m}^3\text{/day)} = 4.91 \times 10^6 \text{ (g } SO_4^{2-}\text{/day)} = 4.91 \text{ (MT } SO_4^{2-}\text{/day)}$$

The annual SO_4^{2-} input from the seepage of the mine is about 1,766 MT.

Scenario 2: Seepage = 4.0×10^4 m^3 day^{-1}

Schladow and Massoudi's dye measurement (1997) give us another story of SO_4^{2-} input from the mine site. 4.0~4.7 $\times 10^4$ m^3 day^{-1} of throughflow from the Herman Pit to the Oaks Arm corresponds to a mean detention time between 21 and 25 days. This means that the Herman Pit would be renewed by inflow at least 14 times per year, and annual SO_4^{2-} input along with the throughflow would be 35,000 MT or more.

The close value of SO_4^{2-} mass can be calculated by the following:

$$4.0 \times 10^4 \text{ (m}^3 \text{ day}^{-1}) \times 3{,}000 \text{ (g } SO_4^{2-}/\text{ m}^3) \times 360 \text{ (day/yr)} = 43{,}200 \text{ MT } SO_4^{2-}\text{/yr}$$

Compared with this huge SO_4^{2-} input from the seepage of the Mine Site, SO_4^{2-} increment during the wet seasons is a small part of the whole-lake S budget (< 6%). The magnitude order of this estimate coincides with that of sulfate reduction rate, which we will discuss in the next section.

SULFATE LOST

SO_4^{2-} Losses from Outlet

Clear Lake's outlet, Cache Creek, is the main route by which SO_4^{2-} is discharged from the system. It is estimated that annual release via Cache Creek is 310×10^6 m^3/yr. Table 8 shows SO_4^{2-} concentrations of Cache Creek measured in 1998 and Figure 6 gives the SO_4^{2-} concentrations of porewater and lakewater at LA-04 of Clear Lake, which is monitoring station near the outlet of Clear Lake.

Assuming that average SO_4^{2-} concentration of outlet is 6.7 mg/l, SO_4^{2-} loss from Cache Creek can be estimated:

$$SO_4^{2-} \text{ Loss} = 6.7 \text{ (g/m}^3) \times 310 \times 10^6 \text{ (m}^3\text{/yr)} \times 10^{-6} \text{ (MT/g)} = 2078 \text{ (MT/yr)}$$

Sulfate reduction

Mack and Nelson (1997) measured sulfate reduction rates in vertically stratified sediment cores of Clear Lake. They found that maximum sulfate reduction rates occur at the sediment depth between 0 and 3 cm. Their result also indicates that sediment sulfate reduction has significant effect on sulfate budget.

Whole-lake sulfate reduction measurements can be used to estimate the amount of sulfate removed by sulfate reduction. Two methods were used based on the different units of sulfate reduction rate: (1) the surface area of the lake multiplied the average of integrated sulfate reduction rate, and (2) the volume of the top 3 cm of sediment multiplied the weighted average of sulfate reduction rate. This calculated amount of sulfate removed was around 250 MT per day.

Method 1: Integrated sulfate reduction rate ($nmole/cm^2/day$)

The surface area of the lake (A) = $176.7 \times 10^6\ m^2 = 176.7 \times 10^{10}\ cm^2$

The average of integrated sulfate reduction rate (ISR) = 1500 $nmole/cm^2/day$

$= 1500 \times 10^{-9}\ mole/cm^2/day = 0.144 \times 10^{-9}\ MT\ SO_4^{2-}/cm^2/day$

Whole-lake SO_4^{2-} reduced by sulfate reduction = (A) × (ISR)

$= (176.7 \times 10^{10}\ cm^2) \times (0.144 \times 10^{-9}\ MT\ SO_4^{2-}/cm^2/day) = 254.4\ MT\ SO_4^{2-}/day$

Method 2: Sulfate reduction rate ($nmole/cm^3/day$)

The surface area of the lake (A) = $176.7 \times 10^6\ m^2 = 176.7 \times 10^{10}\ cm^2$

The depth that the majority of the sulfate reduction occurs (d) = 3 cm

The weighted average of sulfate reduction rate (SR) = 480 $nmole/cm^3/day$

$= 480 \times 10^{-9}\ mole/cm^3/day = 0.0461 \times 10^{-9}\ MT\ SO_4^{2-}/cm^3/day$

Whole-lake SO_4^{2-} reduced by sulfate reduction = (A) × (d) × (SR)

$= (176.7 \times 10^{10}\ cm^2) \times 3\ (cm) \times (0.0461 \times 10^{-9}\ MT\ SO_4^{2-}/cm^3/day) = 244.4\ MT\ SO_4^{2-}/day$

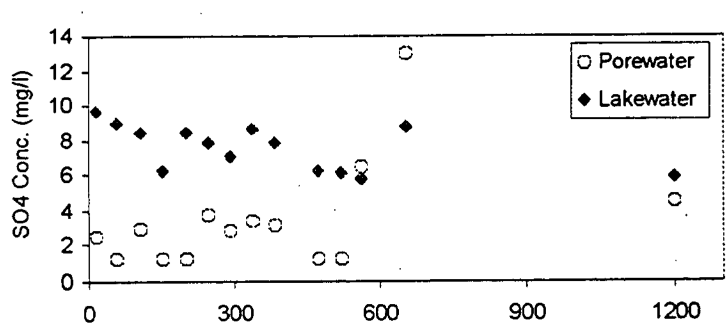
Reduced sulfides re-oxidized back to sulfate

In fact, sulfate reduction rate measurements showed more sulfate reduced than was removed from the lake, which was probably due to oxidation of a fraction of the biogenically reduced sulfides. Oxidation of reduced sulfides may be significant and is rarely measured (Rudd et al. 1986). Usually this part of sulfate gain is calculated from mass balance by difference. In coastal marine systems (Limfjorden, Denmark) it has been shown that up to 90% of the reduced sulfides will be reoxidized back to sulfate (Jorgensen, 1977). In lake systems, some of the SO_4^{2-} reduced during the summer is apparently re-oxidized during the winter (Rudd, Kelly and Furutani, 1986). In Lake Anna (Herlihy et al., 1987), the calculated amount of sulfate removed by sulfate reduction was equal to 200% of the sulfate removed by the outlet, and 30~60 % of the reduced sulfides was reoxidized back to sulfate in1983~1984. In Clear Lake, about 700 $nmole.cm^{-2}day^{-1}$ of net sulfate reduction rate and about 52 % of sulfides reoxidized back to sulfate in Scenario 2 seem to be more reasonable that in Scenario 1 (see A Whole-Lake S Budget).

Table 6. SO_4^{2-} concentrations of Cache Creek

	SO_4^{2-} Conc. (mg/l)
2/17/98	4.5 – 8.5 (Ave. 6.7)
3/11/98	9.4
5/21/98	9.0 – 9.9 (Ave. 9.45)

Figure 4. SO_4^{2-} concentrations in LA-04 of Lower Arm, Clear Lake. Time covers from May 1995 to August 1997.



A WHOLE-LAKE S BUDGET

Now we can sketch the whole-lake S budget based on two different scenarios of seepage from the mine site.

Scenario 1: Seepage = 1,635 m^3/day

	Average SO_4 Conc. (mg/L)	SO_4 Loading (MT/yr)	Percentage (%)
Influx			
Tributary Streamflow	6.0	1,700	37.7
Direct Precipitation	0.28	34	<1
Groundwater	3.0	4	<0.1
Mine Site: by seepage	3,000	1,766	39.2
by overflow		1,000	22.2
Efflux			
Releases from outlet	6.7	-2,078	
Sulfate removal by SR*	--	-2,426	
Sulfate Reduction			
Sulfate reduction rate**	1500 ($nmole.cm^{-2}day^{-1}$)		
Net sulfate reduction rate***	40 ($nmole.cm^{-2}day^{-1}$)		
Sulfate from reoxidation of S #	?		

Scenario 2: Seepage = 4.0×10^4 m^3 day^{-1}

	Average SO_4 Conc. (mg/L)	SO_4 Loading (MT/yr)	Percentage (%)
Influx			
Tributary Streamflow	6.0	1,700	3.7
Direct Precipitation	0.28	34	--
Groundwater	3.0	4	--
Mine Site: by seepage	3,000	43,200	94
by overflow		1,000	2.2
Efflux			
Releases from outlet	6.7	-2,078	
Sulfate removal by SR*	--	-43,860	
Sulfate Reduction			
Sulfate reduction rate**	1500 ($nmole.cm^{-2}day^{-1}$)		
Net sulfate reduction rate***	732 ($nmole.cm^{-2}day^{-1}$)		
Sulfate from reoxidation of S#	52 %		

* Sulfate removal was calculated from mass balance by difference.
** Integrated sulfate reduction rate provided by Mack and Nelson (1997).
*** Net integrated sulfate reduction rate was calculated by sulfate removal divided surface area of the lake.
\# Sulfate from reoxidation of S was estimated as percentage of reduced sulfides reoxidized back to sulafte.

CONCLUSIONS AND RECOMMENDATIONS

A whole-lake S budget of Clear Lake was estimated, which has emphasized inputs and outputs of SO_4^{2-}. Although the effect of seasonal change on the S budget shows its significance under the low flow rate of seepage, the high throughflow from the Herman Pit seems to be more acceptable for the whole-lake S budget. First, the amount of SO_4^{2-} input calculated from high throughflow is coincided with the orders of magnitude of sulfate reduction rates measured by Mack and Nelson (1997). Bacteria sulfate reduction appears to be major biochemical factor in changing water and sediment physical and chemical properties, generating lake alkalinity, keeping the buffering of lakewater, and minimizing the effect of acid mine drainage. Second, the result from high throughflow shows that the process of reduced sulfides reoxidized back to sulfate may has significant effects on metal speciation and mobility. Our AVS and SEM measurement confirms the some oxidation of metal sulfide phases may be taken in the Upper Arm, where mercury methylation is high. Third, the direction of throughflow is consistent with the observations of a white flocculent material in the nearshore region of the Oaks Arm. Bigger floc area and lower pH in that region outside the well MW-2 indicates there is a strong outflow that originated from the mine site. Its impact on the formation of floc and decrement of pH at this location is much more intensive than at the location of Herman Pit overflow channel.

It is the high time to refresh our knowledge of how the mine and Herman Pit cause such a dramatic change in the lake ecosystem, and how sulfate discharged from the mine effect water and sediment chemistry of the lake. It is recommended that:

(1) Historical changes of lake's SO_4^{2-} concentration may be recorded in sediments. It is necessary to raise sediment cores from both Herman Pit and Clear Lake to investigate historical changes of S, Fe, Mn, and other trace metals.

(2) The Sulphur Bank Mercury Mine is predicted as the major source of SO_4^{2-} input to Clear Lake. Dye tracer measurement in the pits and wetland of the mine site to determine or reconfirm the flow rate of seepage and possible routes will reduce the uncertainty of estimating SO_4^{2-} input.

(3) Bacteria sulfate reduction appears to be major biochemical process in the water column and sediments of the lake. The lab and field experiments of sulfate reduction and sulfides oxidation under oxic and anoxic conditions will enhance our understanding of dramatic changes in the lake.

(4) In order to further understand the mechanism of floc formation, design and conduct lab experiments to simulate the formation process will be helpful to assess roles of SO_4^{2-}, Fe, Hg, CO_3^{-2} and alkalinity on the floc formation.

(5) Mathematical modeling of S cycling and early diagenesis permits a systematic and comparative treatment data covering a wide range of sedimentary environment. Hypotheses proposed to explain field observation can be verified by the quantitative model, thereby indicating areas of further research.

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Towards a 3-D model of Clear Lake Hydrodynamics: Observations and Numerical Simulations in the Oaks Arm

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ABSTRACT

The understanding of the fate and transport of mercury in Clear Lake requires the consideration of its hydrodynamics as a first step. Previous studies of mixing and transport processes in Clear Lake, funded through the Center for Ecological Health Research (CEHR) at UC Davis, have shown the lake to be a highly complex three-dimensional system. Its complexity suggests that observations and numerical modeling should be integrated in order to gain a better understanding of the hydrodynamics. A subtle balance between STRATIFICATION, TOPOGRAPHY, AND WIND FORCING yield processes at a multitude of scales. Within each of the basins there is evidence of horizontal gyres at a range of spatial scales, related to the basin topography and the wind field over the lake. In addition, the varying vertical and horizontal density gradients produce complex patterns of baroclinic motions that contribute to both the inter-basin and intra-basin transport. This work describes our present understanding of the large-scale circulation in the Oaks arm. Data collected in the Oaks arm during the month of August in 1997, and numerical simulations obtained with the hydrodynamic 3-D finite element model RMA-10 will be presented.

Keywords: Oaks arm of Clear Lake, large scale circulation, wind forcing, stratification, observations, numerical modeling

INTRODUCTION

One of the most distinctive features of the system of currents in Clear Lake is its very complex nature. There is evidence of horizontal gyres within the basins, changing directions and magnitudes of the currents at different depths in vertical profiles and evolution of the system in time. Given that complexity, a very suitable approach to understand the hydrodynamics of Clear Lake is by combining numerical models and field measurements. The numerical model chosen for this study is the hydrodynamic 3-D finite element model RMA-10 (King, 1998). Once we understand the hydrodynamics, we would be in position of undertaking the problem of the fate and transport of mercury in the system.

With the hydrodynamic study, we would eventually be able to

- Describe how the water moves in the lake
- Explain what are the mechanisms that generate those patterns of motion

From a physical point of view, Clear Lake, and in general any lake, can be understood as a system that receives energy inputs and transform them into some other kind of energy outputs. The inputs are mechanical energy from the wind, and thermal energy (the heat) that produces stratification. Clear Lake would transform those inputs in another form of energy: water currents. The most important feature in the lake system for the generation of currents is probably the bathymetry. Hence, we can think of the currents as being driven by a subtle balance between stratification, bathymetry and wind forcings.

RESULTS

Winter conditions

During the winter months the lake is not stratified and the circulation in Clear Lake will be the result of a balance between the wind field and the bathymetry. Let's assume for a moment that the representation of the bathymetry in the model is correct (and this assumption is fairly reasonable as we used about 3000 nodes to represent the geometry of the basin). Let's further assume that the model gives the right answer to the right question (this is something we checked during the first months of 1998). If both assumptions hold, then we are left with the fact that the quality of the results given by the model will depend on how well do we represent the wind field over the lake.

Figures 1 and 2 represent the velocity field 19 hours after the start of the simulation. In the upper corner we show the measurements taken in the lake during November 1995 (data from Lynch and Schladow, 1996). In Figure 1 a constant wind field blowing from the northwest (dominant direction in the Upper arm) was applied uniformly over the lake surface. In Figure 2 the wind field was interpolated from 12 stations around the lake (data provided by LCAQMD and UCDavis IPM program). The location of the wind stations is shown in Figure 3. The circulation in the Oaks arm obtained with the interpolated wind agrees qualitatively with the observations, and shows the existence of a unique counterclockwise gyre in that basin. However, the intensity of the currents and the exact location where the currents turn to the north differ between the simulated and observed circulation. The disagreement is even more evident in the Lower arm and the region where the three basins join - close to the Narrows-.

The above experiments indicate that the wind field is NOT CORRECT and should be subject of further study. The presence of Mount Konocti may introduce significant effects of rotation and sheltering in the wind field. LCAQMD has provided recently our group with a weather station that will be soon deployed in the piling located off Buckingham point. But further measurements are desirable if the wind field is to be represented accurately.

Summer periods

During summer periods stratification cannot be neglected when explaining the patterns of motions. I shall focus on experimental results obtained in august 1997, that illustrate the role of stratification as a driving mechanism in the generation of currents. Figure 4 shows the layout of the experiment, in which

- four mooring stations equipped with acoustic doppler profilers (ADP) and temperature logger chains were deployed in the Oaks arm,
- Velocity transects were performed with boat mounted downward looking acoustic doppler current profilers (ADCP),
- Density profiles taken with Seabird CTD, and
- Surface and deep drogues (not shown in the figure) were released and tracked during 10-12 hours.

Wind and air temperature information was obtained from a weather station at the Sulphur Bank Mercury Mine.

The water temperature profiles were enough to convince us that the stratification in the Oaks arm was fairly weak. With such a weak stratification, winds with peak velocities of ~ 6 m/s, as measured in the Sulphur Bank Mercury Mine, were strong enough to overturn the initially vertical stratification and generate horizontal temperature gradients. Figure 5 shows a sketch of the sequence of events generated by a single wind event in a basin with a fairly weak stratification as in the Oaks arm. The basin is initially composed of two layers: the surface layer has water at higher temperature than the bottom layer. The surface water moves downwind in response to wind while currents in the lower layers should move upwind. Those motions will generate horizontal temperature gradients in the basin. After the wind stops the pattern of currents is reversed (upwind in the surface layer, while downwind at the lower layer) and is driven by the temperature gradients generated during the wind events.

The simple sketch of Figure 5 seems to agree fairly well with the observations. Figure 6 shows observations obtained in the field experiment. The plot at the top of the figure is a time series of wind speed in the Sulphur Bank Mercury Mine. The second plot (in the center of the figure) shows temperature records from stations 1 and 2. They were taken at 2.5 m below the surface, and illustrate the fact that the wind is able to establish horizontal temperature gradients in the Oaks arm (differences of about 4 degrees C in a distance of 4 Km). The plot at the bottom of the figure is a color plot of the east-west velocity measured at station 2. The horizontal axis represents time, the vertical depths, and the color represents the intensity of the currents (cm/s). Negative values (blue) indicate westward or upwind currents, while positive (red) indicates eastward or downwind currents. The plot shows that during wind events the water in the surface layer flow downwind and in the lower layer flows upwind. That pattern reversed during calm conditions, in agreement with the sketch shown in Figure 5.

All the above discussion has been made assuming that the Oaks arm is a closed basin. However, the Oaks arm is open at the upwind end to the Upper arm through the Narrows and to the Lower arm, and the currents in the exchange region are strong enough (~10cm/s) to influence the circulation in the Oaks arm. Velocity records from stations 3 and 4 suggest the existence of surface currents emanating from the Upper Arm and deviating south to enter in the Lower Arm. These currents could penetrate into the Oaks arm through its southernmost portion, and could be stronger in direct response to the wind. This would create a counterclockwise circulation in the Oaks arm: water flowing east at the south and to the west in the north. In transect 30R (Figure 7) we were able to capture this counterclockwise gyre.

The data gathered so far and presented here suggest a two-state pattern of motions in the surface layers during summer periods, represented in Figure 8.

CONCLUSIONS

- Currents in Clear Lake show spatial and temporal variability. It matters where and when we measure! A sampling strategy should be designed so as to provide an adequate resolution to infer lake-wide circulation patterns. An experiment, combining autonomous drifters, transects and multiple mooring stations will be performed in the near future.

- An accurate representation of the wind field over the lake is a critical input in the numerical model. Future efforts should be directed towards characterizing the wind field. A denser network of stations is desirable: a wind station off Buckingham point.

- We should perform numerical experiments in 3-D during stratified conditions, to support the hypothesis defined using field data. In order to capture the effects of stratification, while keeping the computational cost under reasonable levels, the numerical grid in the horizontal should be coarsened to increase the resolution in the vertical. As we do not want to loose accuracy in the representation of the bathymetry, the bathymetry maps should guide the generation of the computational grid.

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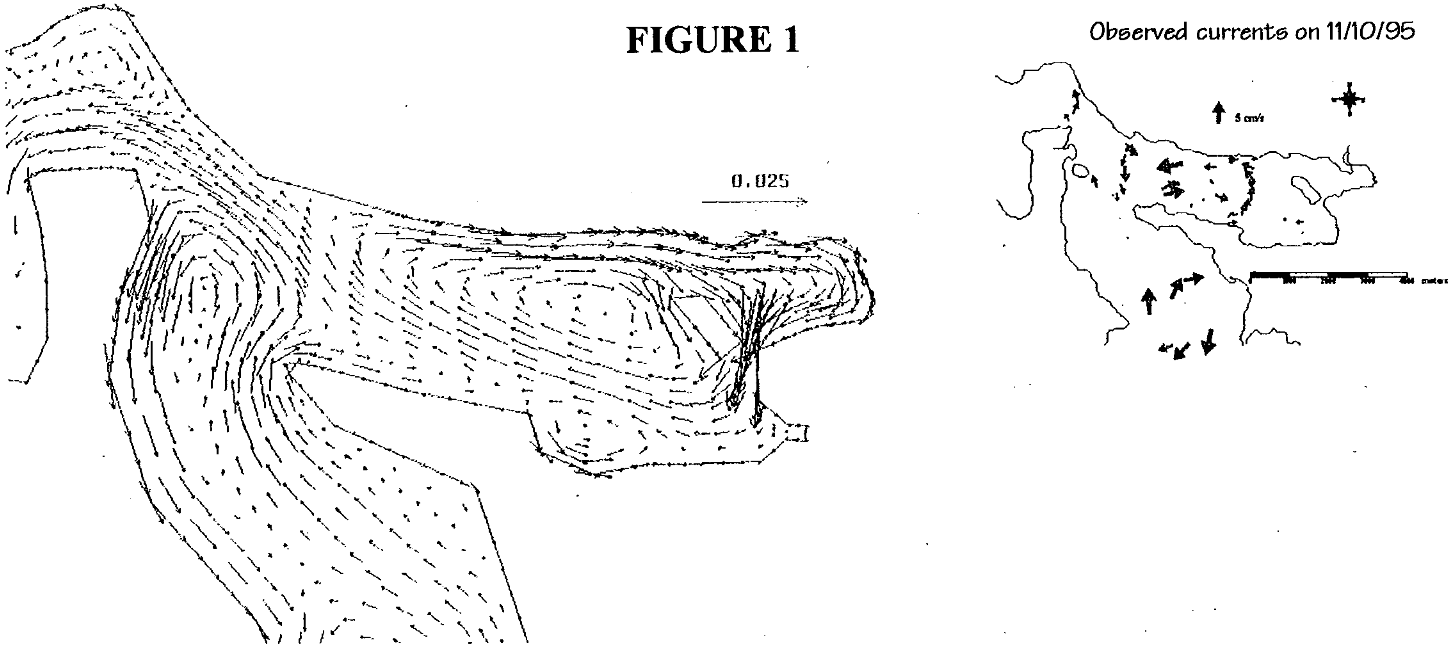


FIGURE 1

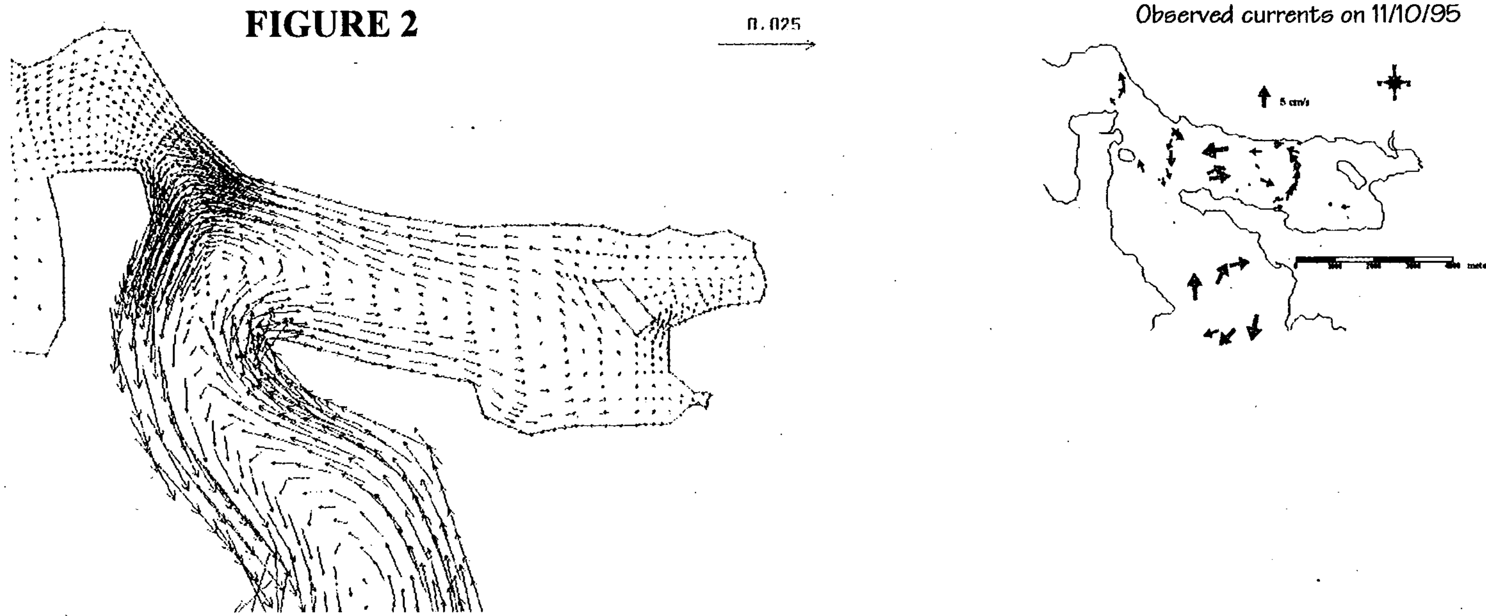


FIGURE 2

FIGURE 3

Wind stations used in interpolation

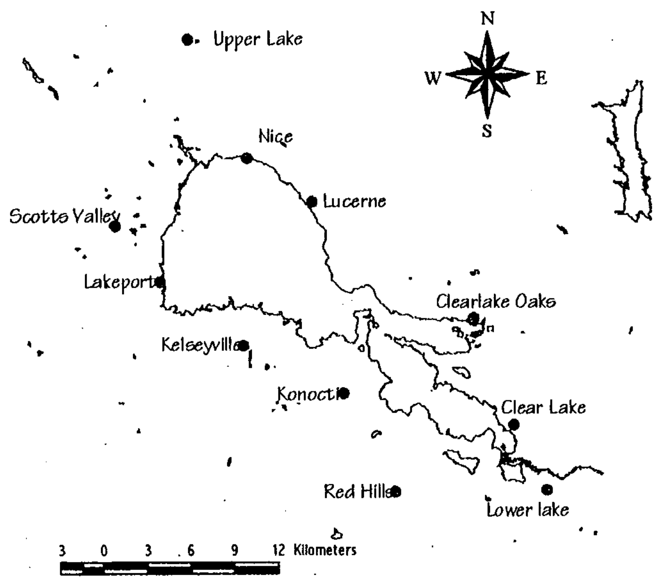
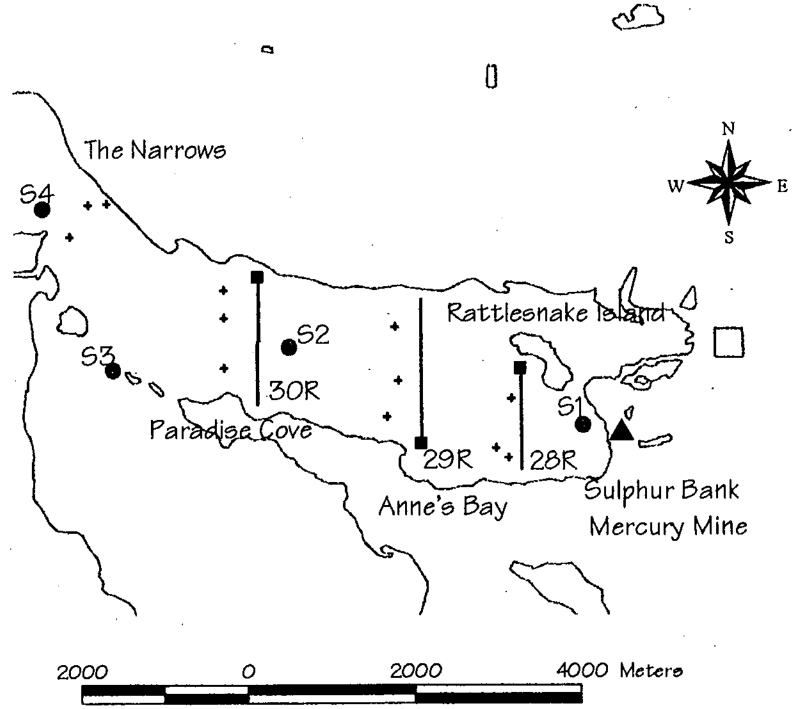


FIGURE 4

Experiment August 1997



• Mooring stations equipped with ADP and temperature loggers;
+ Temperature profiles;
■--- Velocity transects; the square indicates the starting location;
(Not shown) Surface and bottom drifters tracked with GPS
▲ Weather station;

FIGURE 5

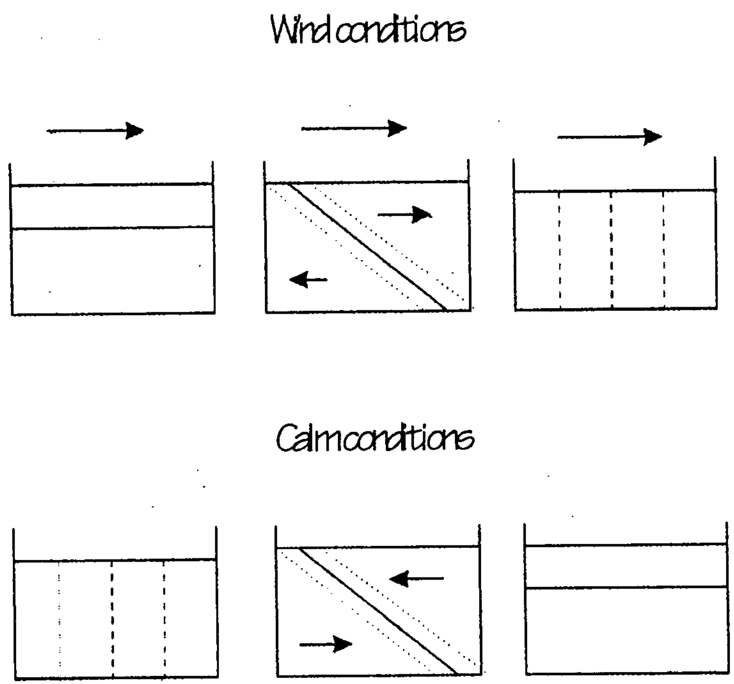


FIGURE 7

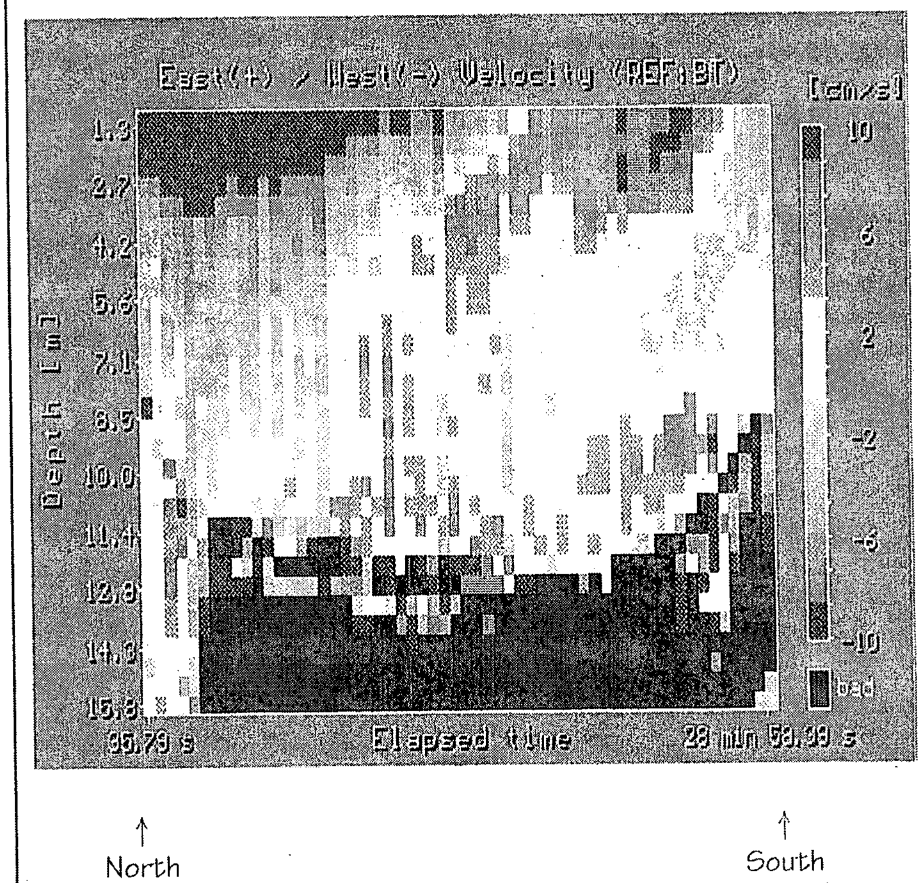


FIGURE 6

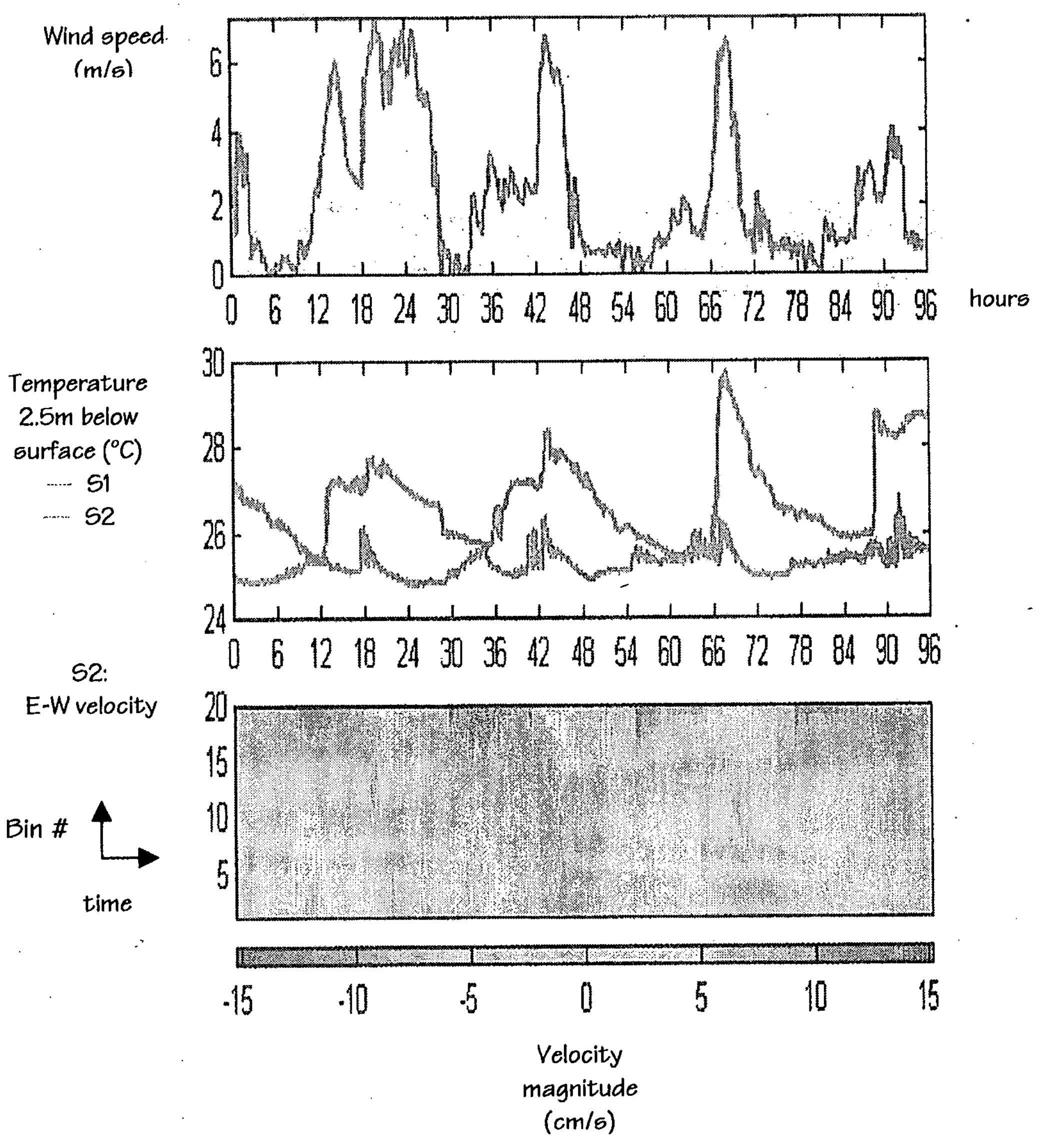
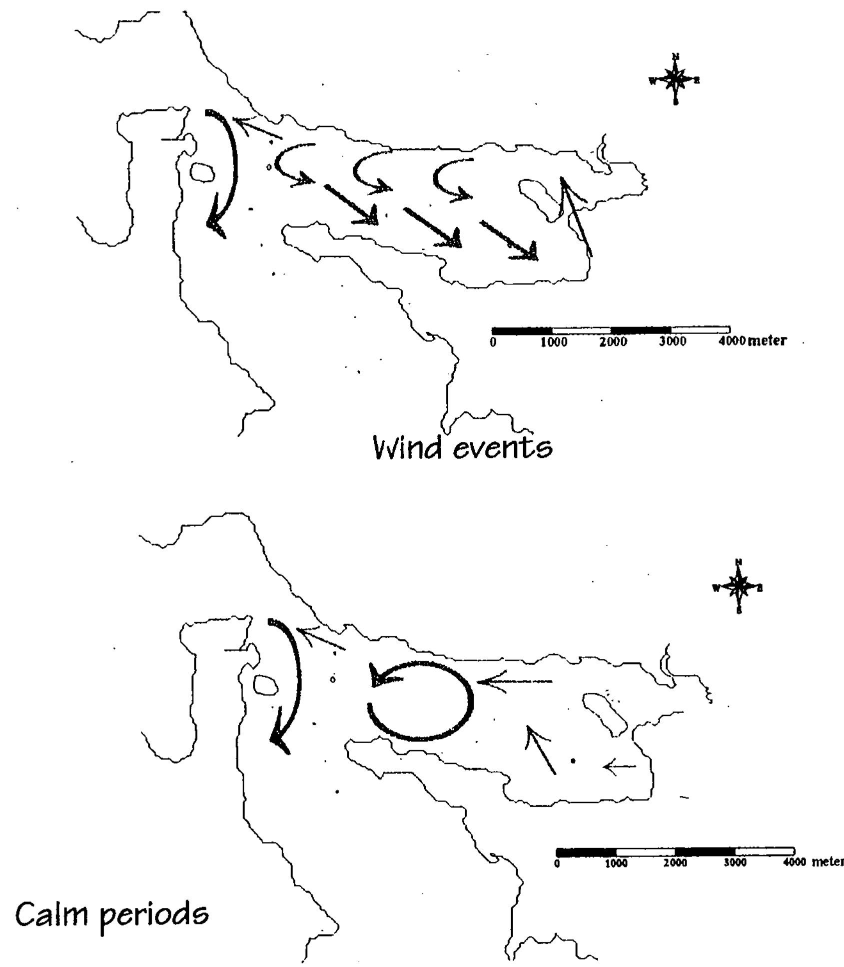


FIGURE 8



Promoting Erosion Control Conservation Practices in the Clear Lake Basin

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ABSTRACT

Soil erosion in Lake County, California, is associated with significant loss or damage in property value, land productivity, habitat, and tourist revenue. In a recent study funded by the State Water Resources Control Board, the Lake County Water Resources Division identified erosion situations of high concern in the Clear Lake Basin. Approaches for their remediation have been developed. This effort is focused on creating demonstration sites for public education, and on development of programs and policy alternatives to encourage use of Best Management Practices for erosion and sediment control. A total of five initial demonstration sites have been planned. A set of supporting recommendations have also been developed for public outreach, incentives, financial and technical assistance, and conservation policy.

Keywords: Best Management Practices, Clear Lake, conservation, erosion and sediment control, education, incentives, Northern California Coastal Range

INTRODUCTION

The 1994 Clean Lakes study (Richerson et al., 1994) identified sediment input from erosion in Clear Lake's watersheds as the principal source of phosphorous associated with increased blooms of noxious blue green algae in Clear Lake. A recent Soil Conservation Service (SCS) report (Goldstein and Tolsdorf, 1994) estimated that erosion in Clear Lake Basin watersheds was nearly double levels expected without human disturbance. The difference between these two levels was defined as "accelerated erosion" by the SCS. The study also reported an annual loss of $7 million in foregone spending by tourists because of noxious algae blooms on Clear Lake. This dollar estimate only included loss of income from tourists familiar with the area.

The full costs of erosion and sedimentation, the SCS noted, would also need to account for lower property values around the Lake, loss of income from new tourists, and the multiplier effect of lower revenues and taxes on businesses, government services, and families in the county.

Costs would also need to account for long term reduction in the productivity of Lake County's soils. The very capability of the land to support natural vegetation, crops, or gardens depends upon the fertility of the top soil. One inch of top soil generally takes several hundred years to form, and a day to a few years to erode away when not protected.

Reconstructing 19th and 20th century erosion history in Lake County is difficult. However, several erosion/sedimentation episodes in the latter half of this century are widely known, and have created public concern. Examples include creek bank and land destruction as a result of gravel mining in Scotts and Middle Creeks (Zalusky et al., 1992), creation of large gullies in Manning Flat, and recent sedimentation from land being prepared for vineyards above Lower Lake adjacent to Pt. Lakeview Road. A fuller historical presentation is provided in Thomas and Brown (in publication).

Building on the SCS and Clean Lakes reports, the Water Resources Division of the Lake County Public Works Department sought and obtained a 205j water quality planning grant from California's State Water Resources Control Board. Work on this study, now nearing completion, has included water quality sampling and the development of a geographic information system database for the Clear Lake Basin. This information was used to describe situations most vulnerable to higher rates of soil erosion and sediment yield to streams and lakes. These findings, in turn, formed a starting point for development erosion control demonstration site plans and for development of policy and program initiatives as described below.

EROSION CONTROL DEMONSTRATION SITES

The 1994 SCS study determined that several significant erosion and sediment sources exist. Table 1 provides a summary of the SCS findings. Accelerated rates of erosion were found by SCS to be occurring on all major Clear Lake Basin watersheds. Field reconnaissance in support of the current study suggests that this erosion is occurring over a wide range of land use types, properties, and owners. No one group (newcomer vs. old-timer, city vs. rural) can claim exemption.

While major erosion eyesores may catch our eye, the column on the far right of Table 1 suggests that it is, in fact, the sum total of many less obvious events that gives rise to the bulk of accelerated erosion. These include soil loss from the many smaller gullies found throughout the county and the nearly invisible, yet sizable, eroded volume leaving many acres of our Lake County upland soil every year through sheet and rill erosion.

Table 1: 1994 Soil Conservation Service Estimates for the Clear Lake Basin

Category	Total Yield of Sediment to Clear Lake (tons/year)	% of Total Sediment to Clear Lake	Total Erosion (tons/year)	% of Total Erosion
Stream Channels & Gullies	117,400	33.7	165,000	19.0
Wildfire (extrapolated fr. 300-400 ac fire near Upper Lake)	14,100	4.0	43,300	5.0
Roads	44,100	12.7	133,300	15.4
Construction Excava'n Sites	15,000	4.3	45,300	5.2
Rill & Sheet Erosion from All Other Land (excld. urban)	157,700	45.3	478,300	55.2
Totals	**348,300**	**100.0**	**866,600**	**99.8**

Consequently, erosion is a community challenge. And, because of its dispersed nature, it is also one requiring a community solution. To demonstrate "how to's" of control erosion to Lake County's communities, it was decided as part of the 205j planning grant to establish several erosion control demonstration sites. Based on the GIS analysis, and contemporary public concerns, the Division of Water Resources elected to focus demonstration sites on erosion related to roads, hillside vineyards, and horse pastures. Such sites can be used to demonstrate techniques effective in minimizing erosion and sediment yield in the categories listed in Table 1.

Approximately 20 potential demonstration sites were identified. Each of these:

- were accessible by road (most next to or easily visible from paved public roads);
- represented human-caused erosion problems common in Lake County; and
- appeared to be amenable to relatively simple, lower cost erosion control solutions.

Of these, five locations were selected for demonstration site planning. At each site the current physical situation was described, and probable causes for the existing erosion were determined. This determination was made based on evidence at the site, accounts of the land owner or tenant, and/or previous experience. A plan was developed to stabilize each area, and thereby reduce erosion and sediment yield to levels expected in a protected, undisturbed state.

The plan consisted of a site description, a listing of those who would be involved in or benefit from the project, and a description of a coordinated series of Best Management Practices to be applied to particular locations on the site. Also included in the plan were a budget and time schedule, a listing of policy and program changes that would encourage implementation of Best Management Practices (BMPs) on similar sites, and a brief description of plans for monitoring the site and for its educational use.

BMPs were designated based on professional experience and on descriptions in the literature (for example, see Marin County RCD, 1987, Mendocino County RCD 1994, Forrest and Klabunde. 1996, McCullah 1994, Soil Conservation Service and Forest Service, 1985, Council of Bay Area Resource Conservation Districts, no date, Appendix A of Thomas et al. 1997, and Association of Bay Area Governments, 1995). BMPs consist of such actions as covering the soil with protective vegetation (which by itself can reduce erosion by over 90 percent in many circumstances), minimizing surface water velocity (or erosive force) using energy dissipators, channel design, and check dams, reducing slope angles, and armoring ditch surfaces with rock, fabrics, and/or vegetation.

An appropriate and effective BMP is one meeting all three of the following criteria (McCullah 1997):

(1) it is appropriate to the situation,
(2) it is actually implemented as designed, and
(3) it is maintained through time.

Each of the 5 Demonstration Sites selected is briefly described below:

Site #1: Gully adjacent to Scotts Valley Road and Scotts Creek.

Present Situation: A gully 83 feet long, displacing 535 cubic yards of soil downstream from an 18 inch culvert crossing under Scotts Valley Road, has formed about one mile from the east end of Blue Lakes. The gully empties directly into Scotts Creek and is located at the bottom of a 3.5 acre watershed in stable condition.

Probable cause of gully: Given its physical situation, it is highly probable that the gully resulted from (1) lack of adequate dissipation of the erosive force in the water emerging from the culvert at high flow and (2) a lack of stream channel and bank protection from the erosive force of the water moving downstream in association with headcutting moving upstream from the bank of Scotts Creek. Headcutting is a process of successive, upstream movement of a steep drop (a sharp vertical displacement of the bottom of a channel) due to the erosive force of water.

Erosion mitigation measures (BMPs) to be demonstrated at this site: (1) Reduction of the erosive force of water emerging from culverts will be provided by (a) properly located energy dissipation pads (composed of sized rock), and also by a (b) in-channel willow check dam and by (c) widening of the channel water flow area. (2) Channel and bank protection will be provided through (a) reducing gully bank slope angles in combination with (b) channel bank toe stabilization using a slope toe trench filled with rock and further stabilized by (c) live willow mattresses placed on the slope underneath and behind the rock toe; by (d) establishing near complete herbaceous vegetation cover on the upper gully banks, and by additional re-enforcement using (e) live staking with cut willow branches in and around the intersection of the gully with the Scotts Creek bank area, and (f) use of a series of large wooden beams to prevent headcutting at the mouth of the gully.

Historically applicable land use management practices and regulations for this situation: In many new projects, installation of energy dissipators at culvert outflows and stabilization of banks may be required by the County as a consequence of the county's existing grading ordinance (Lake County, 1994). However, a significant number of existing culverts and related gully situations pre-date this ordinance (as in this instance), and would likely be exempted from the grading ordinance.

Site #1 also illustrates another issue: how to provide for adequate erosion control when actions are required across property boundaries involving different land owners? The approach taken in the grading ordinance (for new projects) is to mandate that effects will be mitigated on the property on which the project is located. In the case of those situations exempted by the ordinance (including Site #1), this desirable outcome is commonly not achieved. One informative and potentially useful model for addressing this issue is being demonstrated by the voluntary Upper Putah Creek Stewardship group.

Site #2: Gully adjacent to White Rock Canyon Road

Present situation A gully 147 feet long displacing 1095 cubic yards of soil has formed partially along the side of White Rock Canyon Road approximately 0.4 mile from the intersection with Elk Mountain Road. A large headcut, fed by water from inadequately drained upslope portion of White Rock Canyon Road, is moving northwest past the culvert outlet and will likely threaten both the road and adjacent property . This gully drains into Middle Creek through a short, presently stable, vegetated passage. The site drains at least part of a reasonably stable, 23.5 acre watershed.

Probable cause of gully erosion: On-site evidence suggests that the gully resulted from (1) failure of the ditch system on the north side of White Rock Canyon Road, leading to water overflow on the road directly into the gully area, (2) compounding of the ditch failure problem with plugging of the culvert transferring water from that ditch system to the gully (south) side of the road, (3) lack of an adequately engineered channel/energy dissipator leading from the culvert outlet into what is now the gully, and (4) a lack of stream bank protection from headcutting caused by water overflowing the road. Dumping or placement of junk likely magnified the gully bank instability as well.

Erosion mitigation measures (BMPs) to be demonstrated at this site: (1) Clear debris from north-side ditch (on a regular basis) to enable use of its full capacity for carrying water; (2) install a larger culvert under the road to handle large storm events; (3) remove junk from the gully and gully banks to allow stabilization of gully side slopes; (4) install an energy dissipator for the culvert as it enters the gully; (5) halt and stabilize headcut by lowering the slope angle, armoring with rock, and anchoring slope toe with rock; and 6) stabilize gully bank slopes with vegetation and rock toe support. Techniques to accomplish these best management practices will be different in some cases from Site #1, enabling a subjective comparison of relative effectiveness.

Historically applicable land use management practices and regulations for this situation: Similar to Site #1, except the road is private and is maintained by local landowners.

Site #3: Eroding Cutbanks and Ditch Along 11th Street/Scotts Valley Road

Present Situation: A 1900 foot section of Scotts Valley Road and 11th Street, located between Mountview Road and the top of a rise (known locally as Brewery Hill), has experienced significant ditch and cutbank erosion in several locations. When combined with the absence of a road shoulder, damage from this erosion endangers both the road base and pedestrian safety.

Probable cause: Cutbank erosion on this site has existed for many years according to residents of the area. The principal causes for erosion on this site appear to be (1) lack of stabilization of the base of the road cutbanks, and (2) lack of protective and stabilizing structures or cover to hold the soil on the cutbanks. Instability is further amplified by the loose conglomerate nature of the underlying parent rock material. In addition, the ditches are not armored or protected, leading to undercutting of the bank and also of the area immediately adjacent to the road.

Erosion mitigation measures (BMPs) to be demonstrated at this site: (1) widen the shoulder on the north side of this section of Scotts Valley Road/11th Street to help ensure protection of the road from ditch water undercutting, and to provide for safety of citizens transiting, especially by bike or on foot; (2) reconstruct road and driveway ditches and armor appropriately (with rock on grades in excess of 5 percent, and live herbaceous vegetation otherwise) to prevent downcutting and erosion of shoulder, road and bank; (3) stabilize road bank slopes with rock toe trenches and cement retaining walls, and protect banks surfaces by establishing a cover of herbaceous and shrub vegetation to prevent further erosion and loss of land; and (4) demonstrate county/ multiple land owner cooperation to solve erosion control problems. The latter is not a normally recognized BMP, but is included here to make the point that cooperation among watershed stakeholders is a key to erosion and sediment control success.

Historically applicable land use management practices and regulations for this situation: Eroding cutbanks are common throughout the county. In some situations, bank stabilization structures and proper drainage systems have been installed. These occur primarily in association with state highways, or where land owners, cities, or the County have had adequate resources to address the more serious situations. The grading ordinance addresses new projects of substantial character, rarely projects predating its passage. In theory, it may be possible that at least some of these situations could trigger health and safety or nuisance ordinance requirements to abate, but this has not commonly been the case as yet.

Site No. 4: Portion of Roumiguiere Red Hill Ranch Vineyard near Kits Corner

Present situation: A vineyard experiencing top soil loss due to sheet erosion, and occasional gully and rill erosion, despite good intentions of grower. Ten acre, north facing site located along a part the south edge of the main ranch access road west of the main entry gate.

Probable cause: Prior to development as a vineyard, this site had been a walnut orchard and had likely lost a significant portion of its top soil horizon. Today, erosion on this site primarily results from two common causes in vineyard settings: inadequate ground cover between rows of vines and the need for a drainage system capable of retaining as much silt and water in-place as possible, and of handling higher flow events with minimal resulting erosion. The vineyard owner has already taken important steps to control erosion and sediment yield including terracing, mulching, and installation of sediment traps.

Erosion mitigation measures (BMPs) to be demonstrated at this site: (1) establish sufficient vegetation cover to protect the soil during rainy season; (2) create periodic in-sloped alleys between vine rows to slow surface runoff water; (3) provide sufficient armoring of drainage channels and ditches to prevent their erosion; (4) create a watershed-oriented runoff removal system, including a coordinated system of drainage channels, sub-surface drains (some between vine rows in steeper terrain, others under main drainage channels), and sediment basin; and (5) transport net runoff from the entire site to an existing stream channel.

Historically applicable land use management practices and regulations for this situation: Approaches to hillside vineyard development in Lake County appear to be quite varied, depending upon the owner and development team preferences. When vineyard development involves conversion of natural vegetation, then grading ordinance and California Environmental Quality Act requirements must be met. Until June of 1998, conversion of land from an exclusively existing agricultural use to vineyard was not subject to any ordinance or industry requirements. Vineyard owners and developers making such a conversion on 5 acres or more are now required to meet with an erosion prevention education committee appointed by the Board of Supervisors.

Site No. 5: Horse Pasture and Paddock Complex Immediately East of Site No. 1 Along Scotts Creek

Present situation: A set of adjacent horse pastures covering approximately 4.5 acres on the Scotts Valley alluvial plain have been subject to surface erosion. Complete removal of herbaceous vegetation and the beginning of stream downcutting has occurred in the most upstream fenced pasture of this property. Scotts Creek bank is eroding, particularly at points where the river enters and leaves the pastures.

Probable cause: Lack of adequate bank vegetative protection along some portions of Scotts Creek, combined with long periods of grazing and stabling in the pastures. Herbaceous vegetation was lacking in the upstream-most pasture upon arrival of the current tenant. These pasture areas are particularly vulnerable to erosion and soil compaction, given the amount of bare area, close cropping of pasture, & wet season use.

Erosion mitigation measures (BMPs) to be demonstrated at this site: (1) Stabilize Scotts Creek bank slopes by planting riparian vegetation, (2) revegetate pasture areas with appropriate native grasses and legumes, (3) utilize rotational grazing and irrigation to keep full cover of herbaceous vegetation on pastures, and (4) possibly enhance stable, paddock, and composting areas to assist in implementing the rotational grazing system and for providing composted manure as pasture fertilizer.

Historically applicable land use management practices and regulations for this situation: Horse pasture management practices are highly variable in Lake County. No regulations or incentive programs directly address protection of the soil resource in these situations.

RECOMMENDATIONS TO PROMOTE ACTION TO MINIMIZE ACCELERATED EROSION

In order to spur wide-spread property owner interest in minimizing accelerated erosion and sediment from their land, the following recommendations for County of Lake action were developed in the 205j study. Details can be found in Thomas and Brown, in publication.

A. Build Public Awareness of the Benefits of Erosion and Sediment Control

(1) develop simple flyers describing what erosion and accelerated erosion are, how they occur, the benefits of minimizing accelerated erosion, and who to contact. Make these flyers widely available at stores, libraries, and offices, and via bulk mail, and public service announcements on local media.

(2) make reference materials on Best Management Practices and practical examples available to public at County and city offices and at libraries.

(3) conduct public outreach via field trips, speaking at service clubs, at County Fairgrounds events, through the media and Internet.

(4) support the development and conduct of grade school (K-12) and community college courses exploring erosion and how to minimize it.

B. Build Motivation to Act

(1) Provide individual counseling assistance to land owners (a) focused on their specific desired outcomes, (b) providing information on technical and financial assistance possibilities, (c) providing referrals to public or private service providers, and (d) providing examples of successful projects.

(2) Establish a County policy encouraging conservation and minimization of accelerated erosion & sedimentation: (a) use the wetlands policy as a starting point for the format (background, purpose, goals, tools, recommendations, definitions, summary); (b) encourage education, taking responsibility, & pro-active citizen involvement; (c) describe 9 or 10 basic elements leading to success in minimizing erosion & sediment yield; (d) describe performance standards; and (e) encourage support programs.

(3) Build on the existing grading ordinance in such a way as to: (a) motivate education, and consultation with appropriate public officials; (b) shift focus of the ordinance to conservation; (c) establish the principle that each of us is responsible and accountable for meeting erosion standards on our land regardless of whether or not a permit is required; (d) to assist in communicating the idea of responsibility, state when a permit is required rather than when it is not required (as at present); remove all language related to exemptions; (e) extend the range of situations requiring at least a basic erosion control plan; (f) provide several avenues for plan approval, including opportunities for industry or group self-governance; (g) require mitigation of areas experiencing accelerated erosion, or at risk of experiencing accelerated erosion; (h) provide greater protection of riparian and neighboring zones for protection of water quality and other purposes; (i) introduce time of year or soil moisture constraints on grading activities; (j) require conservation practices that produce a net gain in soil over time (or, at a minimum, no net loss), and which help ensure healthy soil, encouraging such activities as slope stabilization, revegetation, and management of surface water to minimize on-site and downstream erosion; (k) require follow-up monitoring and maintenance; (l) establish a permanent Erosion Prevention Education Committee to assist plan review with applicants, and to help provide incentives; (m) provide county representatives with a wider range of regulatory or legal remedies, including imposition of fines, or abatement of erosion/sediment problems and site restoration, in cases where non-compliance persists; and (n) add minor changes to bring the existing ordinance up-to-date.

C. Creation of Opportunities and Resources for Erosion and Sediment Control

(1) Solicit the following: (a) grants to help cover planning and project costs; (b) "gap financing" to cover portions of erosion project funding that the land owner is not able to provide and not otherwise covered by government programs or available from normal lending institutions; (c) funding for development of erosion and sediment control workshops; (d) soliciting conservation planning technical assistance; (e) soliciting estate planning assistance, tying conservation and erosion control planning to a long term financial plan for the property in question; and (f) purchasing or obtaining conservation easements on lands of low or erosion-degraded productivity (by the county, land trusts, or otherwise) to be managed for conservation purposes.

(2). Make these services widely known to the public

(3) Simplify the permit process: (a) eliminate the grading permit requirement on smaller projects when actions are for erosion and sediment control only, and an erosion control plan exists; (b) ensure user-friendly permit process, and latitude in use of techniques when appropriate; and (c) work toward "one-stop" permitting, to include necessary local, state, and federal permits such as being developed in other areas of California.

D. Creation of More Choices for Erosion & Sediment Control

(1) Encourage development of a wide range of Best Management Practices; remember, BMPs: appropriate, implemented, & maintained
(2) encourage do-it-yourself work where appropriate
(3) encourage community and watershed stewardship groups
(4) support scientific & land management research to answer questions about "how to", costs, and benefits
(5) support development of demonstration sites for education on benefits and "how to"
(6) encourage inter-group and inter-agency communication and coordination

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CONTAMINANTS IN CLEAR LAKE GREBES AND OSPREY: PAST AND PRESENT TRENDS OF MERCURY AND ORGANOCHLORINES

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ABSTRACT

In 1956, Hunt and Bischoff reported high levels of contamination in western grebes by p,p'-DDD, an insecticide used then to directly control the Clear Lake gnat. Their study was one of the first, anywhere, to document the phenomenon of bioaccumulation of organochlorines in wildlife; and it helped set-off a worldwide effort to document and eliminate negative ecological influences from insecticides and other potential contaminants. At the same time, mercury levels in the lake are suspected to have been elevated, as well, but no previous studies had documented this. Other researchers at Clear Lake in the 1960s documented severe population declines in Clear Lake grebes, mostly associated with the uses of DDD, but also possibly associated with a closely related insecticide used on the watersheds of Clear Lake, p,p'-DDT. Our research began in 1991 with a goal of updating the Clear Lake grebe and osprey status, to document and update contaminant residues found in present-day Clear Lake populations, and to more fully document and explain the levels and effects of mercury, both today and in the past.

Our early work with western (*Aechmophorus occidentalis*) and Clark's grebes (*A. clarkii*) (these two species were formerly considered the same) has shown that they still suffer a small degree of eggshell thinning (related to the persistence of the DDT metabolite, DDE) and their mercury residues are below (but approaching) levels where sublethal effects would be predicted from laboratory and other field studies. Yet, mercury residues today are lower (by about 1/3) of what they were in the 1960s. Residues of DDD are present today at 2-3 orders of magnitude less in both fish and grebes compared to what they were in the 1960s. We have found that occasional, severe disturbance by boats has limited reproductive output in Clear Lake grebes in several seasons (1994, 1995, and 1997) but when this has been minimized, grebe production has approached normal levels (compared to reference sites at Tulelake and Eagle Lake in northern California).

We have also found that osprey, although mostly gone from Clear Lake during the days of intensive organochlorine use, have recently increased at Clear Lake and are currently reproducing normally. Osprey still have elevated mercury levels compared to other reference areas (northern California, Idaho, and Baja California), but apparently these are not high enough at Clear Lake to affect osprey populations.

Overall, conditions at Clear Lake for osprey, *Aechmophorus* grebes, and most other species of birdlife have improved since the 1960s. Because this is a high-use/multiple-use area, careful management to avoid disturbances to sensitive wildlife at sensitive stages, continued work to understand and remediate various sources of pollution, and adequate habitat protection and enhancement should be continued.

Stable Isotope Analysis of Trophic Structure as Indicators of Mercury Pathways in the Aquatic Ecosystem of Clear Lake, CA

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ABSTRACT

Clear Lake is the site of a USEPA Superfund Site (the Sulphur Bank Mercury Mine) which was actively mined from ca. 1874-1957. Physical and biological processes have distributed approximately 100 metric tons of inorganic mercury throughout the aquatic ecosystem of Clear Lake, which is divided into three nearly discrete water bodies (arms). We will utilize stable isotopes of carbon ($\delta^{13}C$), nitrogen ($\delta^{15}N$) and sulfur ($\delta^{34}S$) to evaluate trophic pathways in parallel with analyses of mercury bioaccumulation. Preliminary data suggest that carbon flows in the lake are derived from the benthos. Since organisms fractionate ^{15}N in a predictable way as a function of trophic level, this isotope has been especially effective in elucidating pathways of mercury bioaccumulation. Largemouth bass had the highest $\delta^{15}N$ values as well as the highest levels of total mercury, consistent with studies showing that top predators experience the highest degree of contaminant biomagnification. This study suggests that $\delta^{15}N$ values can be useful in predicting bioaccumulation of mercury in the biota of aquatic ecosystems.

Key words: Clear Lake, Mercury, bioaccumulation, Stable Isotopes, Trophic Pathways

Effects of Multiple Stressors (mercury and copper) on zooplankton: Studies on Ceriodaphnia and Daphnia

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ABSTRACT

Inspired by copper additions to mercury enriched Clear Lake, CA the acute toxic nature of Cu-MeHg mixtures and the bioconcentration of MeHg in the presence of Cu was evaluated in this study using freshwater cladocerans in 48 hr laboratory exposures. Toxicity tests were conducted on *Ceriodaphnia dubia* from a standard laboratory culture and on *Daphnia pulex* cultured from an individual collected from Clear Lake. Bioconcentration trials were conducted on the Clear Lake *D. pulex*. The 48 hr Cu, MeHg, and Cu-MeHg mixture LC_{50}s were respectively 13.0, 10.2, and 18.0 µg metal/L for *C. dubia* and 13.0, 22.8, and 18.3 µg metal/L for *D. pulex*. Using Toxic Unit methodology, Cu-MeHg mixture toxicity was characterized as additive to slightly less than additive. The possible presence of a slight adaptation in Clear Lake *D. pulex* for tolerating MeHg stress is discussed in light of the LC_{50} results. Approximately 4% of MeHg in bioconcentration tests partitioned into test organisms with an average bioconcentration factor of 26,000 across MeHg treatments of 5 and 10 µg Hg/L. Two-way ANOVA analysis revealed no measurable effect of Cu on MeHg bioconcentration ($p < 0.39$).

Keywords: Methyl mercury, copper, toxicity, bioconcentration, cladocerans

INTRODUCTION

At Clear Lake, California, MeHg levels are elevated throughout the food web due to mining activity on the lakeshore (Suchanek et al., 1995, 1997). Cu-based herbicide has been applied since 1995 to control the invasive aquatic macrophyte *Hydrilla verticillata* (Trumbo, 1997). The potential importance of Cu influence on MeHg dynamics and the combined Cu-MeHg effect on zooplankton inspired this research project. The Cu applications temporarily decimate zooplankton populations in treated areas (Trumbo, 1997) which has important potential implications for zooplankton that are significant food items and agents of MeHg transfer through the Clear Lake food-web (Suchanek et al., 1995, 1997). This study addresses the toxicity to cladocerans of Cu-MeHg mixtures and cladoceran uptake of MeHg under conditions of Cu stress.

Environmental toxicants are known to frequently act in concert in exerting their effects (Borgman, 1980; Spehar and Fiandt, 1986; Enserink et al., 1991), but *a priori* predictions of toxicant mixture effects are not typically possible at present, committing us to relying upon experimental approaches to developing toxicity predictions. An extensive literature search revealed no pre-exisiting Cu-MeHg toxicity data, but did identify twelve Cu-inorganic Hg interactive toxicity studies (references in Figure 1) and one Cu-MeHg bioconcentration study (Ribeyre et al., 1995). The test results were inconsistent, ranging from less than additive to synergistic toxicity (Figure 1) across a variety of test organisms and exposure conditions, and no bioconentration effects in a multiple-metal mixture (Ribeyre et al., 1995). It is not possible to predict mixture toxicity based on understandings of Cu and MeHg toxic action and uptake mechanisms, for these mechanisms are still not well understood (Boudou et al., 1991; Cosson, 1994; Mason et al., 1995, 1996; Sunda and Huntsman, 1996). Therefore, this study utilized an experimental laboratory approach to describe toxic effects of Cu-MeHg mixtures on the cladocerans *Daphnia pulex* and *Ceriodaphnia dubia*, and is the first known research to do so. It also is the first known investigation of Cu's influence on MeHg bioconcentration in cladocerans, using *D. pulex* as the test species.

MATERIALS AND METHODS

Organisms

Ceriodaphnia dubia were maintained at 25 ± 1°C in spring water fed a *Selanastrum capriconutum* algae/blended trout chow mixture daily (3 mL food/500 mL culture) according to U.S. Environmental Protection Agency guidelines (U.S. EPA, 1993).

Daphnia pulex cultures were reared from one mature female taken from Clear Lake, CA, and maintained according to *C. dubia* culture protocol.

Treatment Solutions

All culture water was bottled High Sierra Mountain Spring Water (Sierra Spring Water Company, Sacramento, CA) reconstituted to medium hardness through the addition of appropriate salts (U.S. EPA, 1993).

Stock solutions of MeHg and Cu were mixed to 5 ppm and 50 ppm (μg metal/L; 78.7 and 787 μM) respectively, with glass distilled water and maintained at 4°C in amber glass bottles with teflon lined lids. Fresh stock was made bi-monthly. Treatment solutions were prepared 12-24 hours prior to starting tests, and maintained at 4 °C until being brought to 25 °C in a water bath 2 hours prior to use in tests.

Treatments for acute LC_{50} tests of each metal independently were 5, 10, 20, 35, and 50 μg metal/L, plus a non-spiked spring water control. Mixture treatments were prepared at total metal concentrations of 2, 5, 10, 20, 40, and 50 μg metal/L, plus a spring water control, and were comprised of equal concentrations of Cu and MeHg. A three x three factorial block design was used for the bioconcentration experiment, with control, 5, and 10 μg/L treatment levels for MeHg, and control, 1, and 4 μg/L treatment levels for Cu.

Acute Test Procedures

Acute toxicity tests were conducted in a constant temperature room at 24 ± 1°C maintained under a 16 h L: 8 h D photo period, in capped 20 mL glass scintillation vials filled with 15-18 mL treatment solution. Well fed 48-72 hr old animals were used to start each test, following a 2-4 hour isolation in clean water to eliminate residual food from their gut.

Tests were conducted using 4 replicates per treatment with 5 individuals per replicate. Tests lasted 48 hours. Mortality was recorded every 12 hours. Treatment solutions were renewed at 24 hours. To eliminate biomagnification through food and to reduce complications arising from metal complexation with organic ligands (Mason et al., 1996), animals were not fed during the test.

Single exposure and mixture exposure tests, for both species, were conducted simultaneously. Three complete trials were conducted.

Bioconcentration Test Procedures

Bioconcentration tests were conducted in a constant temperature chamber at 25 ± 1°C maintained under a 16 h L: 8 h D photo period, in capped 500 mL glass jars filled with 175 mL treatment solution. Well fed 48-72 hr *D. pulex* were used to start each test, following a 2-4 hour isolation in clean water to eliminate residual food from their gut. Animals were left in treatment solutions (80 daphnids per treatment) for 48 hours, with solution renewal at 24 hrs. To isolate the bioconcentration effect from biomagnification effects, there was no feeding throughout the test. At 48 hours, remaining live animals were transferred from treatments through a series of two control waters to a microtissue grinder, with residual water removed, for microhomogenization and analysis. Four trials were conducted, with one replicate per treatment per trial, for a total of 4 replicates per treatment. Average bioconcentration factors (BCFs) for the two mercury treatment levels were calculated, dividing tissue concentrations (in ppm) by treatment MeHg concentrations (in ppm).

Water Quality Monitoring

Temperature, pH, dissolved oxygen (D.O.), and conductivity were measured on control waters at the beginning of each test, and on all waters following their use in exposures. Alkalinity and hardness were determined on control waters for each experimental trial.

Analytical Chemistry

Total Hg and Cu concentrations in water were measured by the Veterinary Medicine Diagnostic Laboratory at the University of California Davis, using hydride generation inductively coupled plasma (ICP) for total Hg and Zeeman corrected graphite furnace atomic absorption spectrophotometry for Cu.

Metals concentrations in treatment solutions were confirmed for the first trial of the acute LC_{50} tests, and for all trials of the bioconcentration study. Ion concentrations in control waters were measured by the Division of Agriculture and Natural Resources laboratory at the University of California Davis.
Total Hg analysis for *D. pulex* tissue was performed by Frontier Geosciences (Seattle, WA) using cold vapor atomic fluorescence (CVAF).
Total Hg analysis was performed on all treatment samples for all bioconcentration trials.

Microhomogenization/Dry Weight
Daphnia pulex tissue samples for the bioconcentration study were prepared for total Hg analysis using Back et al.'s (1995) microhomogenization procedure. For each *D. pulex* replicate, 400 µL of homogenate was transferred to a 3-mL teflon vial and frozen until total Hg analysis was performed. Remaining tissue homogenate was transferred in 100 µL aliquots to pre-weighed aluminum trays and dried overnight at 60°C for subsequent dry weight determination using a Perkin-Elmer AD-4 Autobalance.

Data Analysis - LC_{50}
LC_{50} values for independent metals tests, and for the mixture tests when expressed as total metal concentration, were calculated using the data analysis program TOXIS2 (EcoAnalysis Inc., Ojai, CA) which follows the U.S. EPA's LC_{50} determination protocol for multi-concentration acute toxicity tests (U.S. EPA, 1993). LC_{50}s were determined at 48 hrs, and at 12, 24, and 36 hrs when data permitted. All calculations were based on nominal concentrations.
The toxic action of the bi-metal mixture was characterized as either additive, less than additive, or more than additive (synergistic), using the toxic unit approach (Marking, 1977). The additive response is when "the effect of a mixture remains constant when one component is replaced totally or in part by the equieffective amount of another" (Faust et al., 1994). Figure 2 graphically illustrates this point, as well as conditions of less than and more than additive toxicity.
To define the toxic response type for the mixture, the theoretical Toxic Unit (T.U.) value for an additive mixture at its LC_{50} was compared to the experimentally determined mixture LC_{50} T.U. For a two-component additive mixture at its LC_{50}, the T.U. value is defined as:

$$\text{T.U.} = 1 = \frac{[\text{A}] \text{ at mixture's } LC_{50}}{[LC_{50\text{A}}]} + \frac{[\text{B}] \text{ at mixture's } LC_{50}}{[LC_{50\text{B}}]}$$

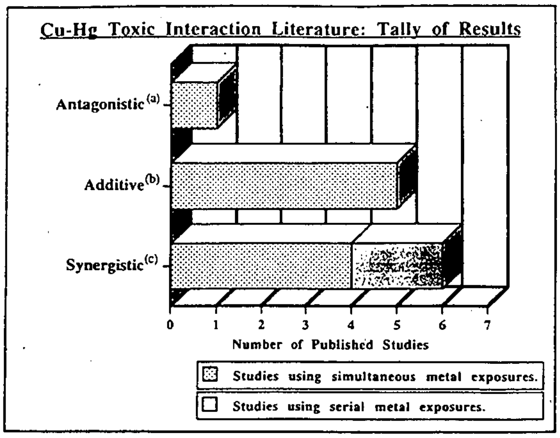
where [A] and [B] respectively represent the concentration of chemicals A and B in the mixture, and $[LC_{50\text{A}}]$ and $[LC_{50\text{B}}]$ the respective independent LC_{50}s of the chemicals. The actual T.U. for a mixture is calculated by experimentally determining the LC_{50} of a mixture of the chemicals A and B (holding the ratio [A]:[B] constant across mixture test concentrations) and substituting the resultant concentrations into the above equation, potentially redefining the T.U. as > or < 1. Actual T.U.s > 1 define less than additive mixtures, T.U.s < 1 define synergistic mixtures, and T.U.s not significantly different from 1 are considered additive.
T.U. LC_{50}s for this study were determined using the TOXIS2 program as above, using LC_{50} results from the single-metal toxicity tests to calculate T.U.s. They were determined for the 36 and 48 hr intervals for *D. pulex* and *C. dubia* as data permitted.
Each LC_{50} presented is the mean result calculated across each test's three trials, and the associated 95% confidence interval for those means.

Data Analysis - Bioconcentration
Total Hg concentrations in *D. pulex* samples were normalized to the average estimates of their respective dry weights. A 2-way ANOVA with block effects, where each trial represents a block, was performed on the resultant dry-weight total Hg concentrations, using the nominal metal concentrations as treatment categories.

RESULTS

Mortality in control treatments met U.S. EPA (1993) acceptability criteria of ^{2} 10%. Measured metal concentrations in tests averaged 22% (S.D. 5.3%) lower than nominal values for Cu and 6% (S.D. 6.2%) lower than nominal values for MeHg. Three acute-test water samples were analyzed for both MeHg and total Hg: 97.3

Figure 1. Results summary for published Cu and elemental Hg mixture toxicity experiments. Experimental methods and test organisms varied across studies.

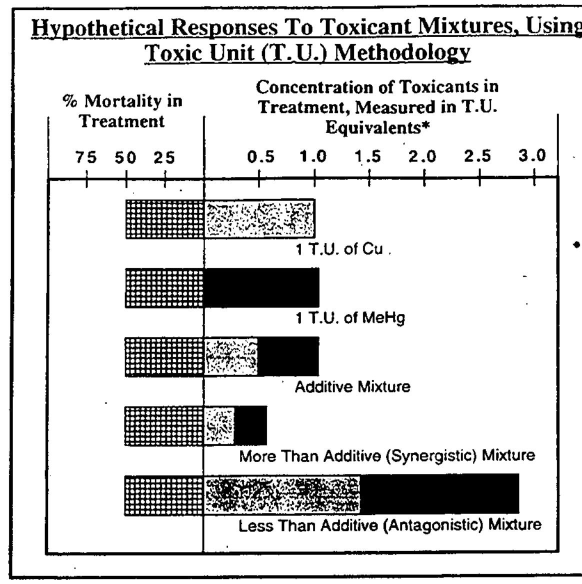


(a) Moulder, 1980.

(b) Murakami et al., 1976; Borgman, 1980; Spehar and Fiandt, 1986; Enserink et al., 1991; Jak et al., 1996.

(c) Simultaneous Exposure: Barnes and Stanbury, 1948; Pyefinch and Mott, 1948; Hunter, 1949; Reeve et al., 1977.
Serial Exposure: Corner and Sparrow, 1956; Khangarot and Ray, 1987.

Figure 2. Illustration of possible responses to toxicant mixtures, based on Toxic Unit (T.U.) methodology.



* 1 T.U. = The single-toxicant LC_{50} for a given toxicant (Marking, 1977).

Figure 3. Toxicity test results for Cu, from $CuCl_2$ LC_{50} tests on *Daphnia pulex* and *Ceriodaphnia dubia*. Mean values for each exposure duration are shown across the top of the histogram.

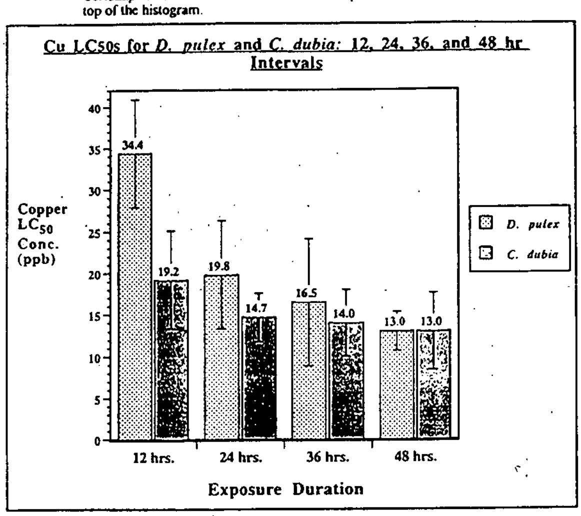
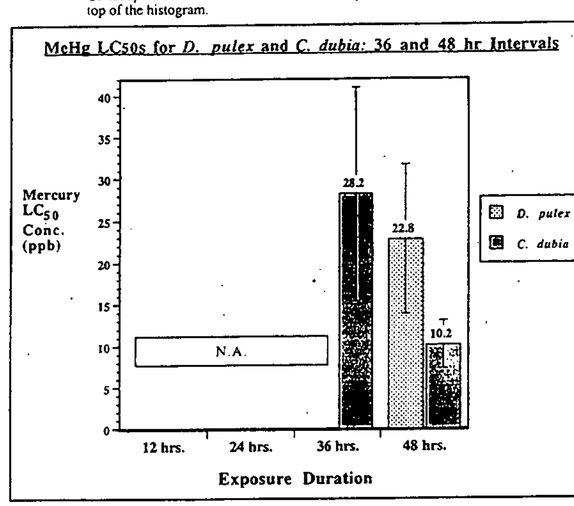
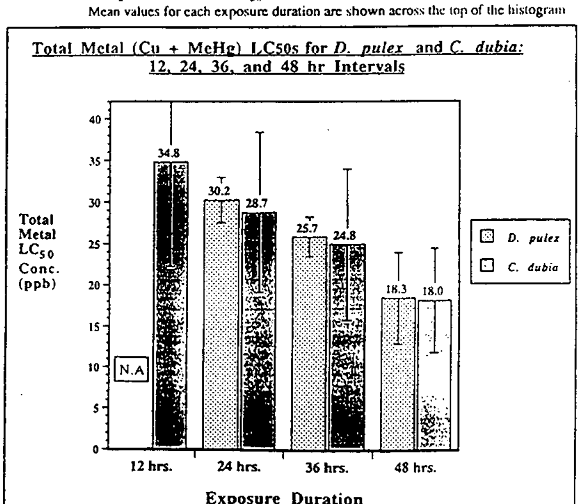


Figure 4. Toxicity test results for MeHg, from MeHgCl LC_{50} tests on *Daphnia pulex* and *Ceriodaphnia dubia*. Mean values for each exposure duration are shown across the top of the histogram.



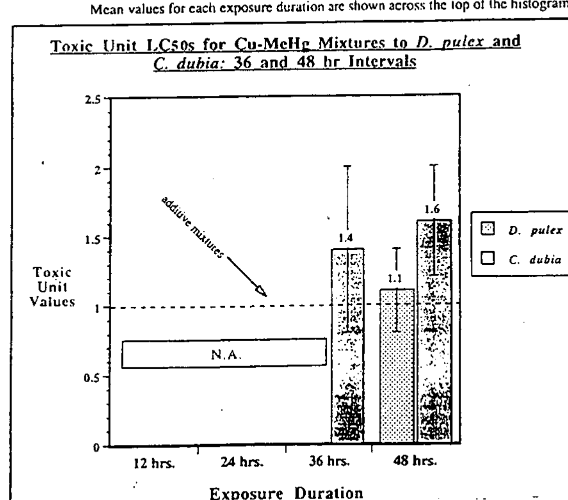
N.A. = LC_{50} not available: Maximum mortality < 50% for all treatments.

Figure 5. Toxicity test results for Cu-MeHg mixtures expressed as total metals, from $CuCl_2$-MeHgCl mixture LC_{50} tests on *Daphnia pulex* and *Ceriodaphnia dubia*. Mean values for each exposure duration are shown across the top of the histogram.



N.A. = LC_{50} not available: Maximum mortality < 50% for all treatments.

Figure 6. Toxicity test results for Cu-MeHg mixtures expressed in T.U.s, from $CuCl_2$-MeHgCl mixture LC_{50} tests on *Daphnia pulex* and *Ceriodaphnia dubia*. Mean values for each exposure duration are shown across the top of the histogram.



N.A. = LC_{50} not available: Maximum mortality < 50% for all treatments.

- 99.3 % of Hg in each sample proved to be MeHg. Water from one 10 µg Hg/L (49.9 nM) bioconcentration treatment was analyzed for Hg at the end of its 24 hr use in exposing *D. pulex*. Comparison to initial Hg concentration indicates a 30% loss of Hg from solution, presumably due to a combination of absorption by *D. pulex* (likely less 5% when compared to *D. pulex* body burdens), adsorption to container walls, and some volatilization (Beisinger et al., 1982).

Toxicity - C. dubia

Toxic response to Cu was rapid, with heavy mortality induced by hour 12. MeHg, by contrast, induced mortality slowly, with most mortality observed after 24 hrs. T.U. LC_{50}s for the mixture test were calculated for 36 and 48 hrs. Data reported are mean values from the three test trials and the 95% confidence intervals around those means. (See Figures 3-6)

The LC_{50} estimates for Cu toxicity to *C. dubia* were 19.2, 14.7, 14.0, and 13.0 µg Cu/L for the 12, 24, 36, and 48 hr endpoints respectively. Average *C. dubia* 36 and 48 hr MeHg LC_{50}s were 28.2 and 10.2 µg Hg/L respectively. Average LC_{50}s for the metal mixture, measured as total metal concentration, were 34.8, 28.7, 24.8, and 18.0 µg metal/L for the 12, 24, 36, and 48 hr endpoints respectively. T.U. LC_{50} analysis including 95% CI's classifies the metal mixture's toxic action as additive at 36 hours and slightly less than additive at 48 hours, with respective T.U. LC_{50}s of 1.4 and 1.6.

Toxicity - D. pulex

D. pulex, like *C. dubia*, more rapidly succumbed to mortality induced by Cu than by MeHg. The average *D. pulex* Cu LC_{50}s were 34.4, 19.8, 16.5, and 13.0 µg Cu/L for the 12, 24, 36, and 48 hr endpoints respectively (Figure 3). The average *D. pulex* 48 hr MeHg LC_{50} was 22.8 µg Hg/L (Figure 4). Average LC_{50}s for the metal mixture, measured as total metal concentration were 30.2, 25.7, and 18.3 µg metal/L for the 24, 36, and 48 hr intervals (Figure 5). T.U. analysis demonstrates an additive toxicity of Cu-MeHg to *D. pulex* (Figure 6) at 48 hrs, with a T.U. value of 1.1.

Both species showed an additive or slightly less than additive response to the metal mixtures, with no indication of synergism nor any indication of substantial antagonism.

Bioconcentration

Results of a 2-way block effect ANOVA on the bioconcentration data indicate Cu had no significant effect on MeHg accumulation in *D. pulex* tissue ($p < 0.39$). The average bioconcentration factor was 26,000 for MeHg spiked treatments.

The microhomogenization technique was tested for its reliability in providing dry weight values for homogenized samples, and proved to be dependable. In the actual bioconcentration test trials, the average coefficient of variation between duplicate dry weight samples was 2.3 %, with a range of 0.2-5.6%.

CONCLUSIONS

In these experiments Cu and MeHg proved to be additive to slightly less than additive in their combined acute toxic action, a finding in accord with the default assumption of additivity advocated by several researchers (Faust et al., 1994). Though additive in nature, differences in the toxic responses observed across time intervals and between species indicate there may be important interaction effects between the metals and significant species-specific differences in responses to the toxicants. In Cu and Cu-MeHg exposures, where D. *pulex* and *C. dubia* responses were nearly identical (Figures 3, 5, & 6), *D. pulex* showed a higher tolerance for Cu in 12 hr exposures, and *C. dubia* had a less than additive response to the toxic mixture after 48 hrs. In MeHg exposures *D. pulex* tolerated greater MeHg concentrations than *C. dubia* at 36 and at 48 hrs (Figure 4), though the two species tolerated identical total-metal concentrations in Cu-MeHg mixtures (Figure 5). This result suggests there may be a slight adaptation in Clear Lake *D. pulex* for greater MeHg tolerance -- relative to *C. dubia* --, a tolerance which if present is lost under Cu stress. Despite Cu's possible influence on MeHg tolerance in *D. pulex*, Cu had no significant effect on MeHg uptake by *D. pulex* in the bioconcentration trials. Additional experiments would be needed to confirm the existence of a MeHg tolerance adaptation in Clear Lake *D. pulex* and to test the applicability of these findings under other exposure regimes.

These highly controlled experiments provide results of limited applicability to the actual conditions in Clear ake, but allowed for a comparison of single and dual metal exposures which avoided confounding effects of high variability presented by more natural conditions. pH, color, organic content, and ionic concentrations vary over time in natural waters and heavily influence metal speciation, uptake, and toxicity (Welsh et al., 1993; Back and Watras, 1995; Mason et al., 1995; Allen and Hansen, 1996; Westcott and Kalff, 1996). High MeHg levels were used to elicit rapid and measurable responses, and to avoid the Hg contamination problem typical of water samples with naturally ultra-low MeHg concentrations (Gill and Bruland, 1990; Watras et al., 1991). Because organism response to toxicant mixtures is frequently dependent on exposure concentrations and the timing and pattern of exposure regimes (Pyefinch and Mott, 1948; Hunter, 1949; Borgman, 1980; Spehar and Fiandt, 1986; Ribeyre et al., 1995) these results present only the first insight into Cu-MeHg interaction dynamics, with much more experimentation needed to characterize the full extent of this important toxic combination.

Environmental management decisisions must rely on a combination of approaches to assess the impacts multiple stressors have on ecosystems, such as this case of Cu MeHg interactions and its importance for Clear Lake. Any such combination should include studies such as this, which would complement *in situ* monitoring and experimental programs.

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Dietary Uptake Rates and Physiological Effects of Methylmercury in Two Freshwater Fishes

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ABSTRACT

Several species of fish in Clear Lake, California, have high levels of methylmercury in their flesh. Mercury leaches into the lake from naturally occurring cinnabar ores in the basin drainage and from anthropogenic sources such as the Sulphur Bank Mercury Mine. These sources yield both organic and inorganic mercury, and natural biological processes can convert them to methylmercury, which is readily taken up by aquatic organisms and biomagnifies up the food chain. The roles of fishes in the biotic pathways of mercury movement from mine-contaminated sediments through the Clear Lake aquatic and surrounding terrestrial food webs are unknown and represent an evident research need. Fish are abundant (19+ species) in Clear Lake, are preyed upon by a variety of birds and terrestrial aquatic mammals (including man), and total mercury levels in edible fish tissue from Clear Lake have exceeded maximum USFDA limits. Both Sacramento blackfish (a native planktivore) and largemouth bass (an introduced piscivore) are important components of the Clear Lake ecosystem and both are consumed by humans. Blackfish are commercially fished and largemouth bass are heavily exploited by sportfishers. While it well documented that mercury accumulates in fish tissues following exposure, relatively little is known about the mechanisms involved in uptake and the physiological effects that mercury exerts on fish. Our project directly addresses these issues.

Using young-of-the-year blackfish and largemouth bass we will conduct a comparative food consumption, growth, Hg uptake rate, routine metabolism, and swimming performance study. Fish will be exposed to sub-lethal concentrations of methylmercury (control, low, medium, or high dose) in their diet for the duration of the study. Throughout the experiment, fish will be weighed and measured and subsamples will be taken to determine tissue concentrations of mercury. Results from individual experiments will be analyzed statistically (ANOVA models) and will be brought together into a bioenergetic model for the two species to quantitatively assess and compare the energy costs of dietary exposure to methylmercury. Pilot studies to establish experimental protocols have yielded "rangefinder" data for the proposed project.

In addition, in vivo stress biomarkers from blood samples will be compared among the dietary Hg-exposure treatment groups, and direct measures of MeHg on blood-oxygen equilibria will be determined. With the known importance of hemoglobin as an oxygen-transporting protein and methylmercury's affinity for proteins, ours will be the first measurements of methylmercury's effects on fish blood-oxygen equilibria. We are collaborating with a larger project developing Clear Lake mercury uptake/transport/fate models (Clear Lake Project, Center for Ecological Health Research, Environmental Protection Agency). Our bioenergetics data will assess the risks of physiological compromise (e.g., decreased growth, decreased swimming performance from decreased oxygen uptake or delivery) in fishes that are chronically exposed to sub-lethal concentrations of dietary methylmercury and determine the consequent roles of fish in the Clear Lake (and other systems') models.

KEYWORDS: water quality, mercury, biomagnification, aquatic food web, ecotoxicology

Atmospheric Mercury Concentrations in the Mayacmas and Knoxville Mining Districts of Lake County

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Abstract

Portions of the Knoxville and Mayacmas Mercury Mining Districts are located within the Lake County Air Quality Management District. Significant mercury mining and ore processing occurred between 1860 and 1960. Large-scale geothermal development has occurred in the Mayacmas mining district since 1960. The Homestake McLaughlin gold mine is located in the Knoxville mining district and began open pit mining and ore processing in 1984. Mercury vapor analysis uses a gold film electrical resistance technique, which is described. Particulate mercury is analyzed from size-selected particulate samples collected on filters which are analyzed by X-Ray Fluoresce (XRF) or Proton Induced X-Ray Emission (PIXE) for mercury and other elements. The ambient mercury vapor and particulate monitoring data was collected in the Knoxville Mercury mining district at the McLaughlin process facility and in the Mayacmas Mercury mining district by the Geysers Air Monitoring Program (GAMP) downwind of the Geysers in the residential community of Anderson Springs. Ambient concentrations from both locations have been monitored since the mid 1980's and are similar in magnitude. Particulate mercury concentrations are approximately two orders of magnitude lower than vapor concentrations. Total mercury maximum peak concentrations are below 1.0 ug/M3.[3] Average annual mercury concentrations approximate the minimum detectable levels of the analysis methods.

Keywords: Atmospheric Mercury, Mayacmas, Knoxville, Lake County

Introduction

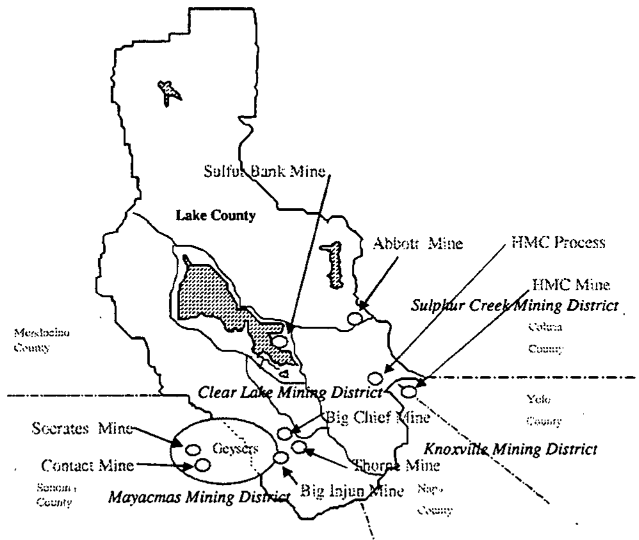
The Lake County Air Basin is coincident with the geopolitical boundaries of Lake County and is the only such air basin in the State so defined. The basin is isolated from adjacent air sheds by the northwest trending mountainous topography of the Mayacmas Mountains on the west and a series of lower elevation parallel ridges to the east. Clear Lake occupies the central portion of the air basin and the complex topography is responsible for a number of distinct micro climates and air shed sub basins within the jurisdiction of the Lake County Air Quality Management District.

Mineral resource exploration began in the early 1850's with the discovery of significant mercury deposits in the southern areas of the county. The Knoxville, Clear Lake/Sulphur Creek and Mayacmas Mining Districts were major producers of the metal, and by 1917 the mercury production of Lake County was estimated to be in excess of 19 million pounds. Neighboring mines in Colusa, Yolo, Napa and Sonoma counties were similarly estimated at that time to have produced an additional 31.6 million pounds of the metal. Sporadic, but significant mercury production continued to occur in the area until the early 1950's. Commercial production virtually ended by the time of the mid 1960's.

Mercury was first widely used in the gold recovery process until the cyanide process replaced it in the early 1900's. Mercury was also important in the manufacture of felt, munitions, electrical and scientific equipment and in chlor-alkali production. Mining activity was closely tied to the price of mercury and production was variable and intermittent with the largest production occurring in the 1870's, 1890's and during World Wars I and II.

The area's mercury deposits are associated with hydrothermal activity in the contact zone between Coastal Range metamorphic rocks and igneous serpentine rock intrusions. The eastern portion of Mayacmas Mining

District contains significant geothermal resources located deep underneath the former mine areas. Where mining activities extracted mercury from depths of several hundreds of feet, geothermal wells now produce dry, superheated steam from depths of several thousand to twelve thousand feet. The area, now known as the Geysers Known Geothermal Resource Area, is the largest developed dry steam geothermal power producing development in the world. The geothermal resources which now drive the turbines for electrical generation are likely the deposition agent for the mercury ores found in the District. Because mercury is a toxic contaminant of the geothermal steam, the LCAQMD has had an early and ongoing interest in ambient mercury concentrations for determining the potential impact of geothermal development on the District's air resources.



This same general time period was also significant in the history of the Knoxville mining district with the development of the Homestake McLaughlin gold project on a historic mercury deposit in the extreme southeastern area of Lake County. The McLaughlin project is an open pit gold mine located in Napa and Yolo counties centered on the Manhattan mercury mine, which was excavated to recover large quantities of gold from low grade ore. Active mining and ore processing started in 1987. The ore is finely ground, mixed with water and transported as a slurry to the chemical-processing mill located in the Lake County Air Quality Management District, approximately 4 miles west of the mine site. The lower grade ores are graded and stockpiled for slurry blending and subsequent processing. Overburden and waste rock is deposited in a disposal facility adjacent to the mine pit. The cyanide extraction process produces a gold/silver bullion and small quantities of elemental mercury. Because mercury is intimately involved in the gold production process, ambient mercury vapor and particulate mercury monitoring was required as a condition of the project's air pollution permit.

Monitoring

The Geysers Air Monitoring Program (GAMP) was created in 1983 as a consortium of public agencies and geothermal industry members to monitor the impacts of geothermal development. Although the program focus is primarily hydrogen sulfide, early concern for air toxics resulted in monitoring efforts directed at other steam contaminants, including mercury. An intensive toxics monitoring program was conducted in 1983-84 and again in 1986-87. The years 1983-1987 were fortuitous for monitoring purposes, as they captured the period of maximum geothermal expansion in terms of resource development and steam production. This initial monitoring effort included both mercury vapor and the elemental analysis of particulate matter <10 um (PM-10). The particulate monitoring program and elemental analysis has been continuous since 1983.

Mercury vapor and particulate mercury have been measured at the Homestake Mining Company (HMC) McLaughlin Process Facility on Morgan Valley Rd., Lower Lake, CA since July 1986. Monitoring data was initially collected near Morgan Valley Road approximately 3000 ft. northwest of the process plant through March 1992. The samplers were relocated to the area of the process facility main gate approximately 2000 ft. east of the plant and have continued ambient monitoring at that location since then.

Analytical Methods

The determination of the mercury content of the ambient samples has utilized three techniques: gold film resistance (Jerome® Instrument Company) for mercury vapor in air, and x-ray fluorescence (XRF) or proton induced x-ray emission (PIXE) for the particulate in air samples.

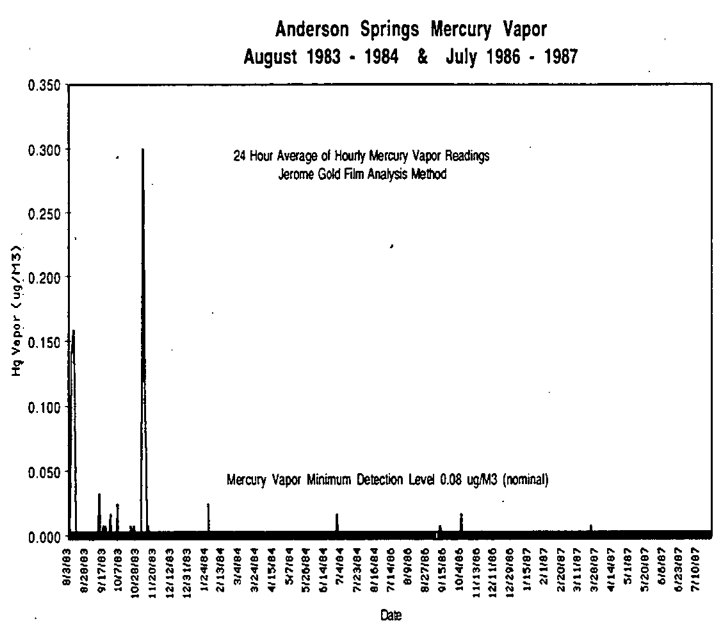
MERCURY VAPOR MONITORING

Mercury vapor analysis utilizes a Jerome® Instrument Company Model 411 analyzer for the collection of mercury vapor on a thin gold film sensor. The analytical method is a linear proportional electrical resistance increase to a balanced bridge circuit which results from the elemental mercury adsorption onto the gold film. The sensor circuit and electronics are microprocessor controlled and the sample conditioning system is equipped with internal and external filters and scrubbers to remove particulate and gas interferences. The instrument is portable, battery and/or line powered. High concentration samples may be analyzed using a 1-second or 10 second sample-scanning mode of operation. The minimum detection limit in the 1–second mode is 10 ug/M3, and 1 ug/M3 in the 10-second sample mode.

The instrument is calibrated with a known amount of mercury generated from a mercury saturated vapor at a known temperature. Calibration checks for the GAMP data were performed on a weekly basis and peak concentration values were verified by duplicating the observed peak heights with injections of known vapor concentrations. The Homestake equipment is factory maintained and certified to NBS traceable standards.

MERCURY VAPOR DOSIMETER SAMPLING

The majority of the ambient mercury vapor measurements use a time integrated mercury vapor determination method which samples a metered amount of air over a gold plated nichrome wire dosimeter. The gold coating on the dosimeter has a capacity of approximately 1800 ng Hg and a mercury vapor collection efficiency of 95 - 100% at air sample flow rates up to 50 cc/min. The dosimeters are analyzed releasing the mercury vapor by heating the gold film. The released vapor is analyzed by the Jerome® Model 411 and the atmospheric mercury vapor concentration is calculated from the instrument response and the sample volume. The majority of the ambient mercury vapor data was obtained using a 24-hour sample period operated on a once per six-day sample schedule. The nominal detection limit using dosimeters for a for a (24) hour integrated sample is 0.003 ug/M3. For one (1) hour integrated samples the minimum detection level is approximately 0.08 ug/M3.



MAYACMAS DISTRICT (GEYSERS) MERCURY VAPOR

The Geysers Air Monitoring program measured mercury vapor concentrations on an hourly and 24 hour sample basis. The Anderson Springs monitoring station is located at the Homeowners Association Recreation Center adjacent to Anderson Creek. Ambient mercury vapor concentrations were determined during the years of 1983-84 and 1986-87. The Anderson Springs watershed contains several mercury mines and significant amounts of surface disturbance occurred near the Big Injun mine with the construction of the PG&E Unit #16 power plant in 1982-83.

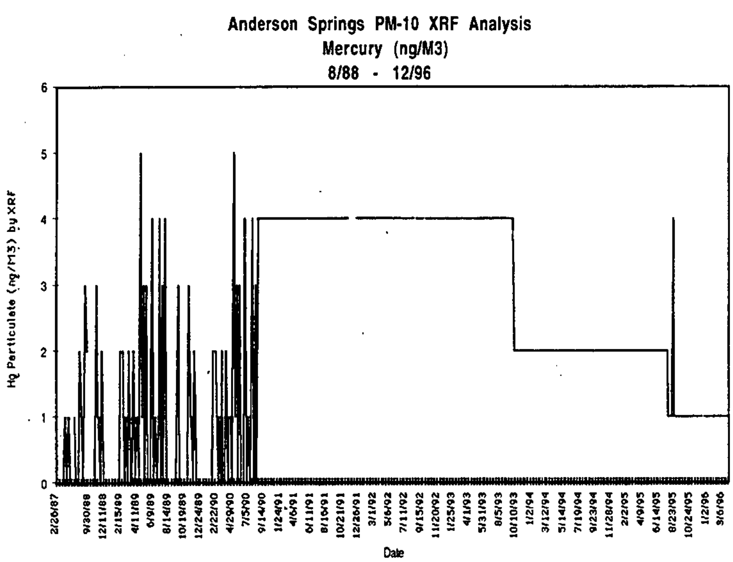
Annually averaged ambient mercury vapor concentrations at this location were 0.014 ug/M3 in 1984-84 and less than 0.003ug/M3 in 1986-87 using the 24 hour sampling one in six day data. The initial elevated peak values are believed associated with program startup difficulties, and the subsequent and only significant mercury vapor concentrations, appear in November 1983. These high values appear to be associated with the heavy rainfall, high steam flow and large sediment load transport events which were coincident with the first large winter storm activity which occurred.

Additional mercury vapor monitoring was conducted at Anderson Springs using the Jerome Gold Film dosimeter technique for hourly samples obtained during the same 83-83 and 86-87 periods. The minimum detection level for this effort is approximately 0.08 ug/M3 although data was reported as low as 0.01 ug/M3. The hourly mercury values were averaged to calculate a 24-hour average reading. The hourly averaged values yielded peak mercury vapor concentrations of approximately twice those of the single 24 integrated samples. The highest reported values were 0.30 ug/M3 and 0.15 ug/M3 and corresponded to the same dates as the highest 24 hour samples. The observed difference in agreement between the two methods is probably indicative of the accuracy of the method at this concentration level.

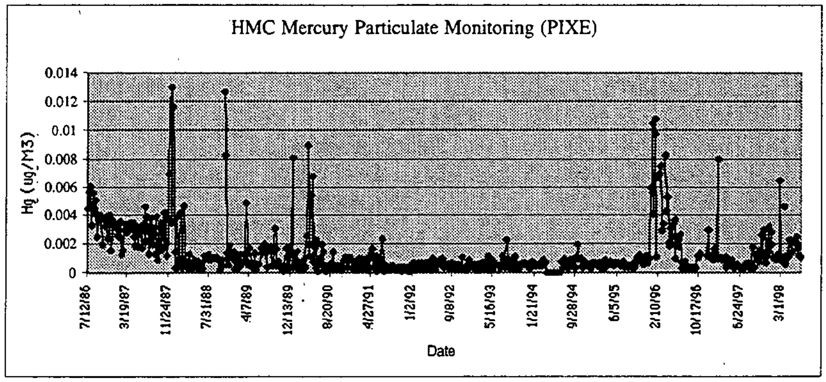
Mercury vapor concentrations have been measured in the Knoxville mining district at the Homestake McLaughlin Mill facility on Morgan Valley Rd., Lower Lake, CA since July 1986 using a 24 hour dosimeter sampler and Jerome® Model 411 instrument analysis. The Homestake mercury vapor sampling and analysis method has a minimum detection limit of approximately 0.003 ug/M3. The annual concentrations of mercury vapor at the Homestake site are demonstrating a variable upward trend from 0.01ug/M3 in 1983 and increasing to .081 ug/M3 in 1997. The thirteen-year average mercury vapor concentration is 0.032 ug/M3.

Particulate Mercury Samples and Analysis

The Mayacmas particulate samples from the Geysers Air Monitoring Program are being collected using a Sierra Instrument Company Model 245 Dichotomous sampler. The Dichotomous instrument collects particulate in two size ranges 10 – 2.5 um and <2.5 um. The sampler was modified in 1995 to increase the sensitivity of the method with the deposit of all particulate <10 microns in aerodynamic size on a single Teflon filter. The filters are weighed and a mass concentration per unit volume of air is determined. The filters are analyzed by XRF for selected elements with atomic weights between aluminum and zirconium. The minimum detection limit for mercury particulate during the program has been between 0.004 and 0.002 ug/M3. Mercury particulate concentrations have been monitored on an ongoing basis beginning in 1983 and continue to date. With few exceptions, ambient particulate mercury concentrations are at or near the current 0.002 ug/M3 detection limit with a 13-year average concentration of 0.0018 ug/M3. The stepwise nature of the above reported data beginning in September 1990 reflects a reporting convention change to the 0.004 ug/M3 detection limit. The step change noted for October 1993 is a result of a decision to report values less than the detection limit as 50% of the detection limit. The step change noted beginning in August 1995 is the improved detection limit which resulted from the sampler modification. Reported XRF mercury particulate concentrations are adjusted for blank filter background values.



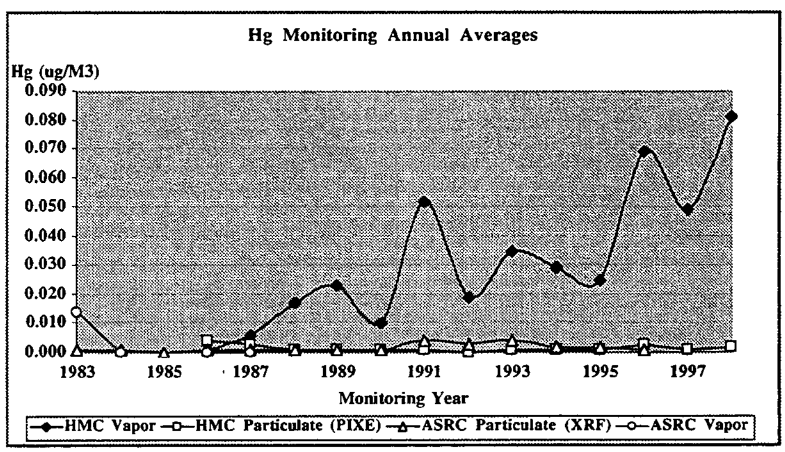
The Knoxville district particulate mercury concentrations represented by the Homestake McLaughlin data are from samples collected by a PIXE International Corp. Model I-1 cascade impactor operating at a sample flow rate of 5 lpm. The <16 micron particulate is impacted on a greased Mylar film during a nominal 24-hour sample run. The particulate is analyzed by PIXE spectroscopy. The method quantifies the mass concentrations of elements with atomic weights between fluorine and lead. The minimum detection level for the PIXE technique is dependent on the total mass, the elemental loading and the molecular complexity. The PIXE method has a reported minimum detection limit of 0.003 ug which results in a sample flow dependent detection limit of between 0.0004 ug/M3 and 0.01 ug/M3. The values reported for the Homestake PIXE mercury particulate have been adjusted for the background values of the blank impaction media. With the typical sample volumes utilized in the HMC program, the PIXE technique has a minimum detection sensitivity of approximately .0004 ug/M3 +/-100%. The annual particulate Hg concentrations have remained at or near the detection limits and have ranged between 0.000 and 0.004 ug/M3. The thirteen-year average mercury particulate concentration at Homestake site is 0.0015 ug/M3 and closely compares to the 0.0018 ug/M3 thirteen-year average concentration determined by XRF for the Geysers site.



Conclusion

Ambient mercury concentrations measured in the formerly active Mayacmas and Knoxville mercury mining districts show similar particulate mercury concentrations which are at or near the detection limits. On an annual basis, the inhalable particulate mercury concentrations range between 0.05 ng/M3 and 4.0 ng/M3. These values are considered representative of near source background values in areas with low respirable size particulate mass concentrations. Long term averaged mercury particulate concentrations in the Mayacmas and Knoxville districts are virtually identical with 0.0015 ug/M3 in Anderson Springs (Mayacmas) and 0.0018 ug/M3 at the HMC McLaughlin facility (Knoxville).

Mercury vapor concentrations at both locations are highly variable and the measurements are considered source specific event related. The small number of elevated mercury vapor concentrations at Anderson Springs appear to be related to a flood event involving sediment transport from a formerly active mining source. Mercury vapor concentrations at the Homestake process facility appear to be indicative of the near source sampling location and process activities during the monitoring period.



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A Survey of Relative Mercury Bioavailability Throughout the Upper Cache Creek Watershed, Using Benthic Invertebrates

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Native benthic macroinvertebrates were used as indicators of biologically available mercury throughout the upper Cache Creek watershed (38 sites). Biotic mercury was low in most of the Clear Lake tributaries, the headwater regions of the Bear Creek drainage, and in several additional tributaries that did not contain historic mercury mining activity. Dramatic spike concentrations of mercury were present near abandoned mercury mines, with dry weight concentrations of >1.00 µg g-1 (ppm) and maximal concentrations to over 20 µg g-1. Every significantly elevated set of samples was associated with a known mercury mine source or a stream that drained a mercury mining zone, including Sulfur Creek, Harley Gulch, Davis Creek, Schindler Creek, and Brushy Creek. Samples from the main stem of Cache Creek were elevated above background levels throughout the stretch between Clear Lake and Rumsey, but exhibited a slight decline throughout this reach. Results suggest that much of the large bulk load of mercury in Cache Creek may be relatively inert biologically. However, abandoned mercury mines are clearly sources of some mercury that is, at least initially, highly bioavailable. The fraction of mine-derived mercury which is is initially dissolved and later surface-adsorbed, coprecipitated with iron and other metal and mineral flocs, may be disproportionately important to mercury methylation and subsequent movement into the food chain--throughout the region and downstream.

Keywords: Clear Lake, Cache Creek, tributaries, mercury, benthic invertebrates, bioaccumulation

Mercury - Comments on its Dispersion, Bioaccumulation and Biocidal effects.

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Elemental mercury is a unique heavy metal which at room temperature is a liquid that readily vaporizes into the atmosphere and undergoes extensive transformations as it cycles between air, land and sea around the earth's crust. In the 1960s it was demonstrated that low toxicity mercury compounds are transformed in sediment and soil to highly poisonous methylmercury and dimethylmercury. These compounds are readily absorbed into living organisms, passing up the food chain where additional transformation, bioaccumulation and biomagnification occur. In this manner living organisms act as temporary sinks for mercury released into the environment.

Over the last 150 years there has been a substantial increase in the presence of mercury due to both natural and industrial releases into the environment. Anthropogenic emissions are due to: 1) activities in mining and extracting mercury from its ore, 2) the widespread use of mercury for amalgamating gold and silver, 3) production of chemicals for use in plastic manufacturing and other industries, 4) the burning of fossil fuels such as coal and gas, and 5) the practice of incinerating or discarding broken thermometers and other mercury products in landfills and (6) the cremation of individuals with mercury tooth amalgams. Until recently mercury compounds have been widely used as agricultural fungicides for treating seeds.

Except for its industrial benefits mercury and its compounds have no known redeeming benefits to higher organisms. The widespread pollution of terrestrial and aquatic environments will continue to increase in the future unless drastic steps are taken nationally and internationally to curtail the release into the environment and reduce the conditions necessary for its conversion to organic mercury compounds.

Environmental Mercury – An Immunotoxic Perspective

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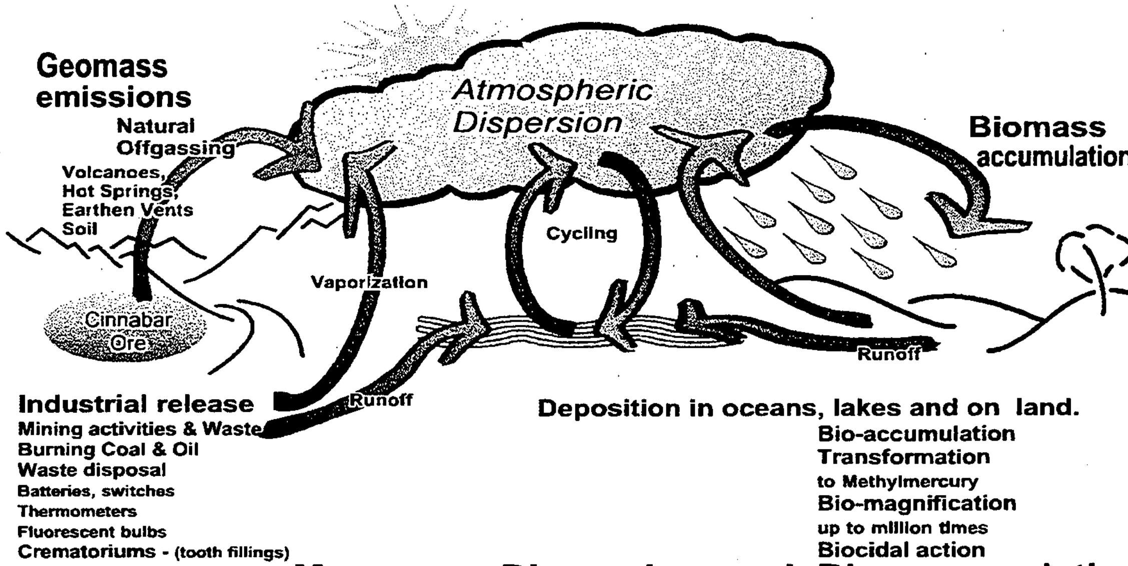
ABSTRACT

There are three stable oxidative states of mercury: elemental Hg^{o}, inorganic (mercurous Hg^{+} & mercuric Hg^{++}) and Organic in which covalent bonds are formed with carbon atoms. In its elemental form mercury is a cumulative poison that can readily enter the body when volatilized and inhaled. Inorganic compounds of mercury are also poisonous depending upon their level of solubility and their environmental situation. In sediment and soil inorganic mercury (Hg^{++}) can be methylated by bacteria to form organic mercury. Methylmercury and Dimethylmercury are highly hazardous organic forms with the capacity to penetrate cell membranes. Methylmercury reacts adversely with a multiplicity of cellular components, it accumulates within cells, and has the capacity to biomagnify as it passes up the food chain from prey to predator. Dimethylmercury is insoluble in water and readily passes from the aqueous environment where it is formed into the atmosphere where, in the presence of sunlight, it is converted to methylmercury. Since the mercury ion can readily transform from one oxidative state to another all mercury compounds are potentially toxic, the toxicity varying depending upon the solubility, mode of entry into the organism, capacity to complex with cellular components and tendency to persist within the cellular milieu

Although cell death in the brain and nervous system is its most observable action, mercury is also an extremely active agent for destruction of the microtubules responsible for mitotic spindle formation. Since this prevents cell division and blocks cell cycle progression it markedly reduces cell population. Inactivation of microtubules within the cytoplasm also affects the structure and the migratory and phagocytic activity of cells. These actions, which are not grossly observable, explain many of the mercury effects on the embryo as well as the reproductive, hematopoietic and immune systems of the adult. Mercury and its compounds have no known beneficial physiological effects within the organism.

Mercury is an element which is neither created nor destroyed so the amount present today is identical to that present at the time the earth was formed. However the location and form of mercury is changing from a relatively insoluble ore, to reactive elemental mercury and its highly toxic organic compounds spread over the earth's surface. Offgassing from volcanoes, earthen vents, and hot springs has resulted in a slow increase in atmospheric mercury. In addition, the industrial revolution has resulted in a greatly increased transfer of mercury from underground deposits of relatively inert cinnabar and other mercury ores to elemental mercury in the atmosphere, global deposition of inorganic mercury and widespread incorporation of organic mercury into plants and animals. The inadvertent exposure to elemental and organic mercury in the environment poses a serious threat to humans as well as wildlife The cumulative adverse effects of mercury make it imperative that methods be found to reduce mercury emissions into the atmosphere, prevent methylation of inorganic mercury and detoxify organic mercury. Unless substantive steps are taken all indications are that organic mercury levels in the environment will continue to increase and produce greater and greater havoc to wildlife and to mankind



Mercury - Dispersion and Bio-accumulation

Industrial activity such as mining, burning fossil fuels and land disposal of wastes has produced massive shifts, redistribution and transformation of Mercury from underground deposits of Cinnabar, Coal and Crude Oil to widely dispersed mercury vapor, inorganic and Organic Mercury

Mercury in its elemental and various inorganic and organic forms is a ubiquitous, cumulative and highly hazardous toxicant which poses a major threat to the environment and to public health (*11,21,51*). It is distributed globally and the levels of atmospheric mercury are steadily increasing as a result of industrial releases and the burning of fossil fuels as well as emissions from natural sources. Repeated endemic environmental disasters (*40*) and indiscriminate land disposal and incineration of mercury-containing products significantly impact the environmental burden (*3,14,*). In addition to atmospheric exposure, the widespread use of mercury amalgams for tooth restoration and the presence of methylmercury in fish and other foods have provided a continuous low level exposure to the toxicant throughout human lifetime.

In its elemental form, Mercury (Hg°) is a metal which, at room temperature, comprises the heaviest liquid in existence. Because of its high vapor pressure and its solubility, elemental mercury vaporizes into the atmosphere where it becomes ubiquitous, cycling in a ping pong manner back and forth between earth and atmosphere for as much as a year. As it gradually disperses over the surface of the earth it transforms into inorganic mercury (Hg^{++}) which attaches to particulate organic matter. When in the presence of a growing bacteria population, in either oceanic or fresh water environments, inorganic mercury can be converted into organic mercury CH_3Hg^+) or ($(CH_3)_2Hg$) (*41*). The ability to persist in the environment and to transform into different states with different solubility characteristics endows mercury with the capacity to penetrate lipid membranes of aquatic organisms and to accumulate in components of the food chain. The organic form of mercury, methylmercury, is relatively stable and passes from prey to predator with biomagnification occurring as it is passes up the food chain.

Mercury produces a multiplicity of adverse reactions and has no known beneficial physiological effect after its incorporation into living plants and animals. It has been widely used for a variety of purposes over the past 3,000 years (*7*). Medical uses of mercury have almost ceased since it became evident that the adverse effects were often worse than the disease it was attempting to cure. Agricultural use as a fungicide has now ceased and the use of mercury in tooth restorations is beginning to decline. Its use in amalgams has already been banned in several countries. Although somewhat restricted, industrial uses of mercury and its compounds have continued and disposal into the environment is under-regulated due to major difficulties in disposing of mercury-contaminated wastes.

Biological Responses to Toxicants

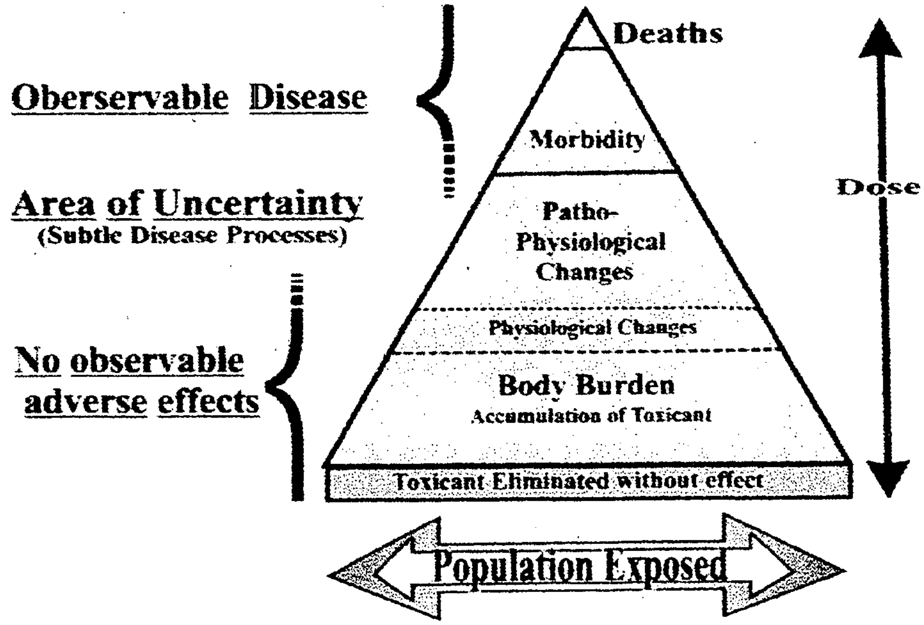


Figure 2 is a diagram representing Biological Responses to Toxicants such as Mercury. The horizontal scale represents the population exposed and the vertical line the dose of a toxicant such as methylmercury. Extremely low doses have little or no observable effect but as the dose increases the body undergoes a series of changes, eventually resulting in death if the dose is sufficiently large. It is possible to be exposed to some forms of mercury with no observable effect. We can dip our hands into liquid (elemental) mercury with little or no adverse manifestations and we can even swallow a cup of liquid mercury and live. Liquid mercury that is swallowed passes rapidly through the intestinal tract. When swallowed, it is a highly potent purgative flushing everything out of the intestine with little absorption into the blood or lymphatic vessels. In the case of skin exposure, although mercury attaches to epidermal cells, these cells are dead and keratinized and will slough off the skin, carrying the mercury with them.

On the other hand if elemental mercury is vaporized and inhaled, about 80% of it is absorbed into the respiratory tract and thence into blood vessels within the lung. Since elemental mercury is lipid soluble and, since the membranes between the alveoli of the lung and the blood vessels are extremely thin, the mercury passes through them and into the blood circulation throughout the body. Most of this mercury becomes incorporated into red blood cells (RBCs). Some of it combines with amino acids and proteins and gradually diffuses out of the blood vessels through endothelial cells and into tissues and organs. The mercury complexes within RBCs and is translocated and presumably altered when the RBC is catabolized.

A great deal of effort has been made to determine the dose levels of mercury that we can tolerate without lasting detrimental effect. The RfD is a reference standard designed to calculate the maximum amount of mercury we can tolerate per day over a lifetime without producing an observable adverse effect (NOAEL). The TLV estimates the threshold dose level for exposure during an 8-hour workday. These standards represent an educated guess of the level of toxicant below which there will be no morbidity or quantifiable adverse effect. A correction factor is usually added so that the reference dose is a factor of 2 to 10 or more below the lowest dose which produces a demonstrable adverse effect. There is a broad margin of uncertainty as to whether the effects observed are disease processes or are due to reversible physiological mechanisms. There is no established test to measure the level at which the effects become irreversible. The generally recognized adverse effect measured for mercury toxicity is based on nervous conductivity or lack of it, but there is growing awareness that this is not necessarily the most sensitive endpoint. For these reasons the generally accepted RfD and TLV values may not be adequate to protect the body from harm, especially in the case of repeated exposure. It has been the experience of toxicologists that the "no adverse response level" is continually being adjusted downward as more becomes known about each specific toxicant. Economic benefits factored in the calculations frequently play a significant role in the determination of risk and the establishment of standards.

The uptake of inhaled vaporized mercury or ingested methylmercury in fish is very efficient but its excretion in feces amount to only 1.6% of the total body load per day. In other species additional excretion can occur via hair, scales or feathers as well as urine and even sweat but the rate of excretion is generally very slow (*38, 10*). In humans it can take from 44 to 70 days to reduce by *50%* the mercury level after a single exposure (T 1/2).

Methylmercury within cells appears to act at the molecular level, compromising a large number of cellular functions. It can complex with SH groups on structural protein molecules which form the microtubular structures of cells. It can also inactivate protein enzymes such as glutathione, and combine with amino acids such as cysteine. Mercury can combine with DNA causing fragmentation, and it can inactivate RNA, thus compromising protein synthesis. Finally it can react with mitochondria, adversely affecting oxidative reactions essential to cell function (*25, 31*)

One of the most dynamic effects of mercury is on the cytoskeleton of cells. The cytoskeleton is made up, in part, of hollow tube-like structures of fibrous material composed of distinct proteinaceous globules that fit together like bricks around a chimney (*24*). Each of these tubules, which are 24 nanometers in diameter, elongates at one end as they are shortened at the opposite end, in a treadmill-like fashion. The building blocks are peptide units called tubulin which pair up and attach to the growing end of the in a spiral fashion. Each tubulin pair contains 14 sulfhydryl groups (SH) which act as points of attachment for mercury. Exposure to very low doses of mercury facilitates formation of complexes with one or more of these SH groups, which in turn results in a disassociation of the tubulin building blocks. Such disruption of the microtubules drastically affects the overall structure and activities of the cell (5,*18, 29, 25,26, 30, 32,33*). In the case of the brain, tubulin makes up the main protein component, amounting to as much as 20% of the total protein content.

Since microtubules (see Figure #3), are found in all cells of the body, all cells are therefore potential targets for methylmercury poisoning (*2,28,*). The microtubules in blood platelets and lymphocytes are especially sensitive, even to low doses of mercury and it has been suggested that they could be used as quantitative indexes of mercury toxicity (*5, 13*). Since the microtubules that make up cilia and flagella are more permanent structures, they are less sensitive to the effects of methylmercury. Mercury poisoning can have drastic effects on cells undergoing cyclic changes leading to mitosis or meiosis (*30,39*). The microtubules undergo massive changes at the beginning of cell division and then reassemble into the spindle apparatus. In laboratory experiments methylmercury treatment blocks spindle formation and thus removes the scaffolding on which chromosomes are segregated during mitosis (*23. 32,33*). In mercury-poisoned cells the separation of chromosomes into daughter cells is random with an unequal distribution of chromosomes between the two daughter cells. This process generates production of multinucleated cells and daughter cells with variable amounts of nuclear chromatin. The cells with small nuclei (micronucleated) are quite conspicuous and are especially prominent in blood smears taken from mercury-treated annuals with nucleated RBCs. Since the number of cells containing these micronuclei is proportional to the dose of methylmercury administered it can be used as an index of exposure (*4, 1*). Such impairment of cell division and failure to produce normal daughter cells is called mitotic arrest and is evident in response to extremely low doses of mercury.

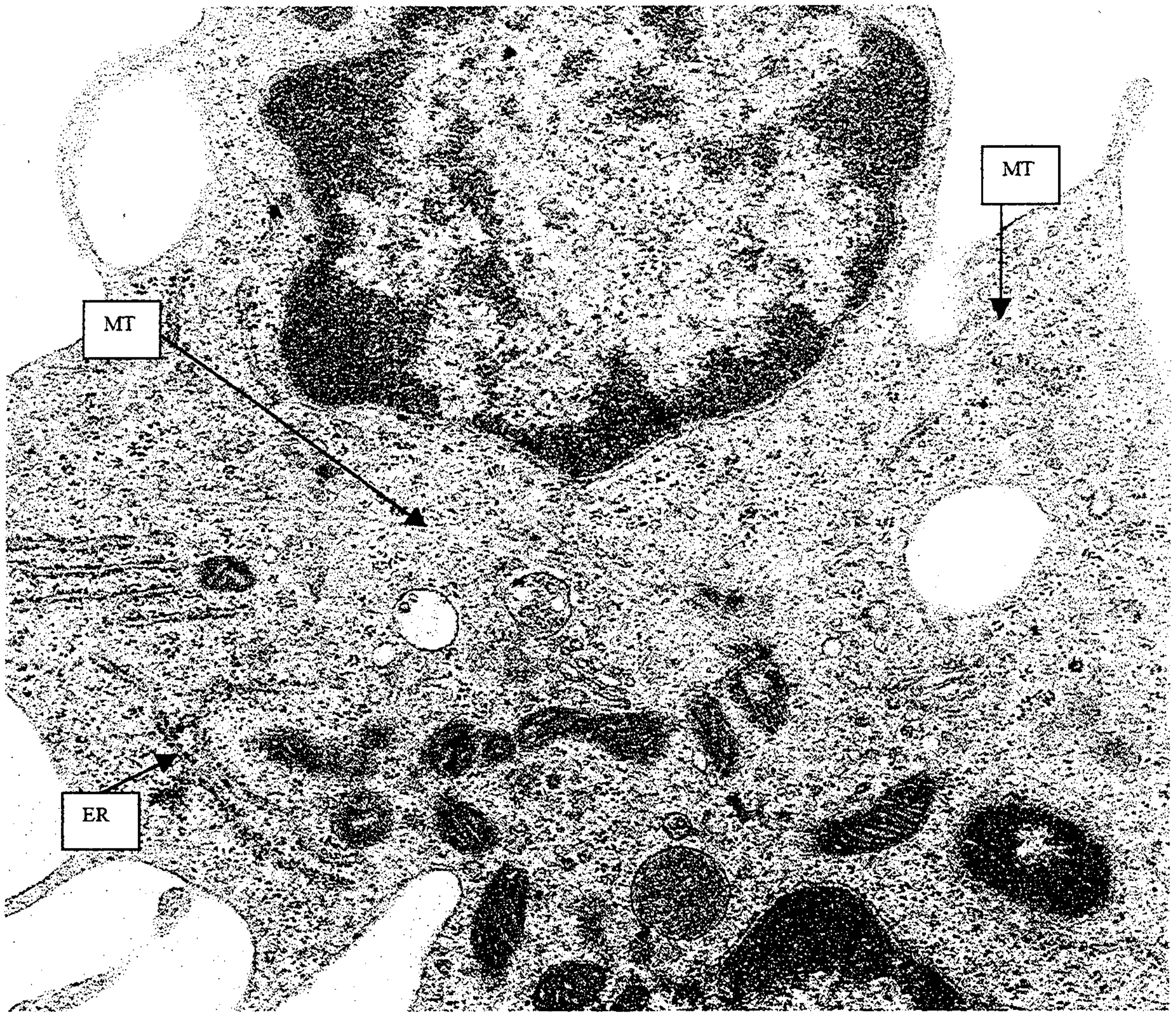


Figure 3. Electron Photomicrograph demonstrating microtubules (MT) running from centrosome area to periphery of the cell. These microtubules are essential for cellular movement, phagocytosis and cell division. The tubules (ER) with black dots (ribosomes) are endoplasmic reticulum.

Cell division as a measure of adverse mercury effects is complicated since it is not grossly observable or easily measured in human tissue except at biopsy or autopsy. Tissues vary greatly in the amount of mitotic activity necessary to maintain normal function. In the case of embryonic tissue, exposure to mercury doses with no observable effect on the mother, can result in reduction of the overall brain size of her unborn child. In higher doses this reduction can be as much as 50% (*21*). The reduction measured at birth is due in large measure to a decrease in cellularity of the brain and a reduction in the number of cell to cell axon and dendritic interconnections (*7*)

The epithelial and endothelial cells that cover our bodies and line our organs need steady replacement requiring high rates of mitosis. Such cell division is also necessary to generate the multitude of immune cells that protect our body from invasion by pathogenic microorganisms (*8*,) and for the spermatozoa involved in reproduction. The number and function of all the RBCs and WBCs present in the blood and hematopoietic tissues is maintained by mitosis. Any reduction in cell replacement thereby compromises the health and well being of humans as well as the plants and animals in our environment. Mercury is capable of such action in such an insidious manner that we are hardly aware of it until the cumulative effects are serious and often irreversible (*40*). When the doses are sufficiently high and prolonged the effects are overwhelming.

Mercury also has the capacity to affect the functioning of macrophages, which play a decisive role in maintaining health and well being. They are the garbage collectors and undertakers of the body. Their function is to remove foreign material that might make its way into the body, and also to remove the billions of worn out RBCs and other cells programmed to exist for relatively short periods (*22*). They also play a crucial role in inflammation and trauma by removing injured or killed cells and by engulfing any foreign material and/or pathogenic organisms that have penetrated into the body. Recently a histological technic (autometallographic procedure) has been developed which has the capacity to demonstrate the presence of mercury in cells in submicroscopic (nanometer-sized) quantities. When applied to tissue a crystal latticework of mercury was found in practically all mercury-exposed cells, especially in endothelial cells, macrophages, and related cells (*9*). The primary locations of mercury complexes within these cells are the phagolysosomes, the sites containing debris, and disintegrating cells in the process of digestion and breakdown. This suggests that, although mercury can penetrate and react with cytoplasmic components of all cells, its ultimate fate is to accumulate as lattice deposits in macrophages and related cells.

Worn out cells and cells injured by toxicants are engulfed by macrophages, their contents catabolized and the toxins neutralized. In the case of mercury complexes, the macrophages could act to further detoxify the mercury and initiate its excretion from the body. Recent studies have shown that macrophages produce enzymes which have the capacity to demethylate methylmercury (*36*, 8). This demethylation would transform organic mercury from a lipid-soluble highly poisonous material to inorganic mercury with different solubility and less toxicity. Inorganic mercury tends to accumulate in the kidney and is excreted from the body more rapidly than methylmercury. Thus the body does have the capacity to transform methylmercury to less-toxic forms and eventually to eliminate it

Demethylation of methylmercury does not occur in the fetus and embryo, probably explaining why these tissues are so sensitive to methylmercury. It also does not occur in adult skeletal muscle, resulting in methylmercury accumulation in muscle and its biomagnification when contaminated prey is eaten by predators. While demethylation does occur in the brain it is at a slower rate than in most other tissues of the body. Autopsy specimens have demonstrated the presence of inorganic mercury in the brain 25 years after exposure to methylmercury (*12*).

While this presentation has concentrated on the downside of exposure to mercury, some recent studies provide a little light at the end of a dark tunnel. Selenium is an essential trace element incorporated in certain polypeptide enzymes and amino acids (selenocyteines) which exist within cells as well as in blood plasma. Mercury has a very high affinity for selenium, even higher than its affinity for sulfur. (*21, 16. 42*) When autometalographic technics are applied to tissues of animals exposed to mercury, a submicroscopic crystal latticework of mercury and selenium was found (*19, 35*). This suggests that two different types of mercury complexes accumulate in cells. One type consists of the mercury combined with various protein molecules via sulfur linkages. Another type consists of mercury bonded to selenium and cellular protein. Formation of selenium-mercury complexes appears to detoxify the adverse effects of mercury (*1,19. 20*). When selenium is administered with mercury in equal molar amount, detoxification of the mercury occurs. While it is not known if this detoxification is temporary or whether it represents a more prolonged inactivation, it has been suggested that the long term persistence of inorganic mercury in the brain and other tissues may be due to the formation of insoluble complexes with selenium (*6, 17*). For example mercury uptake in certain plants such as radishes was significantly reduced by adding selenium to the soil. It is possible that selenium could be utilized in attempts to control toxicity both prior to and after plants and animals have been exposed to mercury. Zinc is another metal that could possibly be utilized to offset mercury toxicity. This area of research is relatively unexplored but it does offer the promise of offsetting mercury toxicity by developing procedures that might modify the rate of absorption and reduce toxicity

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Methyl Mercury Production at Clear Lake is Decoupled From Bulk Inorganic Mercury Loading: Biotic Contamination is Lower Than Expected

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ABSTRACT

Clear Lake is the site of the abandoned Sulphur Bank Mercury Mine, periodically mined for mercury (Hg) from 1873-1957. A variety of mining related events and processes resulted in about 100 metric tons of Hg being deposited into the Clear Lake aquatic ecosystem. Concentrations of total (primarily inorganic) Hg in Clear Lake are some of the highest reported concentrations in the world for both sediments and water, with maximum [average in brackets] Hg concentrations up to 4.4×10^5 [2.5×10^5] ng/g (ppb) in sediments and 4×10^{-1} [1×10^{-1}] µg/L (= ppb) for water in the vicinity of the mine (east end of the Oaks Arm of Clear Lake). However, the ratio of methyl Hg (meHg) to total mercury (totHg) at Clear Lake indicates that the methylation process is mostly decoupled from the bulk inorganic Hg loading, resulting in a level of Hg contamination in lower trophic level biota that is typically significantly less than anticipated based on gross bulk inorganic Hg loading at other worldwide sites. This may be due to a combination of factors, including: (1) the high pH (alkaline) nature of Clear Lake water, (2) the highly productive/eutrophic status of the lake's ecosystem, (3) the shallow depth regime of Clear Lake, preventing stratification and widespread seasonal anoxic conditions, (4) the potential inhibitory influence of sulfur binding with Hg near the mine site, and ultimately (5) the bioavailable nature of the inorganic Hg. While the bulk inorganic Hg loading to the lake may not contribute significantly to the bioaccumulation of Hg, acid mine drainage (AMD) from the Sulphur Bank Mercury Mine (which has low pH, contributes some inorganic Hg and has high concentrations of sulfate) likely promotes Hg methylation by sulfate reducing bacteria, making the AMD a vehicle for the production of highly bioavailable Hg. This is evidenced by extremely high meHg in chironomid larvae which occur in an AMD-derived floc near the mine. It is hypothesized that given different environmental conditions at Clear Lake (e.g. if the lake were deeper as a result of dredging, or less productive, or less alkaline) this system (including biota such as benthic invertebrates, fishes and birds) would likely be much more contaminated with meHg than it is in its present state.

INTRODUCTION

Clear Lake is one of the most mercury (Hg) contaminated systems in the world, with bulk sediment total Hg (totHg) concentrations sometimes exceeding 400 mg/kg (ppm) and raw water totHg concentrations sometimes exceeding 350 ng/L (pptr) in regions of Clear Lake near the abandoned Sulphur Bank Mercury Mine Superfund Site (Suchanek *et al.* 1997, 1998). Yet the lower trophic level biota of Clear Lake have considerably lower concentrations of methyl Hg (meHg) than might be expected based on the bulk Hg contamination. We analyzed totHg and meHg concentrations of sediment, water and biotic tissues and compared these values with those reported from other contaminated and non-contaminated sites worldwide which reported both totHg and meHg concentrations.

MATERIALS AND METHODS

Samples of sediment, water and biotic tissues were collected from Clear Lake during the period from 1992-1998. Sediment samples were collected using an Ekman dredge, placed in new glass jars with Teflon lined lids, and shipped on ice to the analytical laboratory. Water samples were collected using the "ultra-clean" techniques of Gill and Bruland (1990) and Watras *et al.* (1991). At each station, water samples were collected in acid-boiled, double-bagged Teflon™ bottles. Aqueous Hg analyses for each sample included totHg and meHg in both a raw (unfiltered) water sample and a filtered (< 0.45 μm) fraction. All water filtration was performed within 24-48 hrs of collection. There are undoubtedly particles and colloids (to which Hg can bind) smaller than the 0.45 μm filter size we have used, so the filtered samples do not represent a truly dissolved fraction. Biota (oligochaetes and chironomids) from Clear Lake were collected from Ekman dredge samples and sieved through a 1.0 mm screen. Zooplankton were collected using a 80 μm mesh plankton net towed at 1m depth. Biota samples were dried and stored/shipped glass jars with Teflon lined lids.

Hg analyses on 1992 samples were performed by Brooks Rand, Inc. (Seattle, WA), by Battelle Marine Sciences Laboratory (Sequim, WA) and, on occasion, by Frontier Geosciences (Seattle, WA). TotHg and meHg in water, sediments and biota were detected using cold vapor atomic fluorescence spectrometry (CVAFS). Prior to measurement, the desired species of Hg was separated from the sample matrix using a variety of techniques dependent on the type of Hg and matrix being examined. Bromine monochloride oxidation, stannous chloride reduction, and purge and trap dual amalgamation was employed for measuring totHg in water (EPA Method 1631) (Bloom and Crecelius 1983). Strong acid digestion followed by $SnCl_2$ reduction was used to separate totHg from sediment and tissues. MeHg in water and sediment samples was analyzed using aqueous phase ethylation and isothermal GC separation. For most water samples, zooplankton, and for sediment samples prior to1998, the Hg was separated from the matrix using distillation from an HCl matrix. For high Hg waters and for 1998 sediments, an acidic $HBr/CuSO_4$ extraction into methylene chloride and back-extraction into water was performed prior to analysis. It should be noted that there is still some question as to the validity of meHg concentrations reported for sediments, and to a certain extent those reported for water (when analyzed using the distillation procedure) because of a documented meHg artifact produced especially when high concentrations of totHg are present in the environment (Liang *et al.* 1996, Bloom *et al.* 1997). These potential discrepancies are currently being evaluated.

Minimum detection limits for Hg were: sediments (totHg = 0.06 ng/g; meHg = 0.012 ng/g), water (totHg = 0.1 ng/L; meHg = 0.02 ng/L), and biota (totHg = 0.001 ug/g; meHg = 0.001 ng/g).

Many of the original references for Hg concentrations at other sites from published literature were derived from Table II in Suchanek *et al.* (1998). Unfortunately, there were errors in that table which did not make comparisons with other systems accurate. Therefore, we have corrected that table and reproduced it here in Appendix I.

RESULTS

Table 1 identifies the sources of data that are used in this comparison document and abbreviations that correspond to the data figures for sediment, water and biota. The data reported are not meant to represent an exhaustive list of sites for which totHg and meHg has been reported, but more as a comparison of a variety of sites that are representative of both contaminated and non-contaminated sites. For Clear Lake we have included a number of different matrices including sediments, water and various biota.

Table 1. Data cited in this paper, with codes that correspond to the figures for sediment, water and biota.

Location	Source	Matrix	Matrix Code	Year of Data Collection	Reference
Clear Lake, CA	Hg mine	water- lakewide	CL7-r	1994-96	Suchanek *et al*. 1997
Clear Lake, CA	Hg mine	raw water- near mine	CL7-r*	1994-96	Suchanek *et al*. 1997
Clear Lake, CA	Hg mine	filtered water- lakewide	CL7-f	1994-96	Suchanek *et al*. 1997
Clear Lake, CA	Hg mine	filtered water- near mine	CL7-f*	1994-96	Suchanek *et al*. 1997
Clear Lake, CA	Hg mine	chironomids- lakewide	CL7-c	1994-96	Suchanek *et al*. 1997
Clear Lake, CA	Hg mine	chironomids- near mine	CL7-c*	1994-96	Suchanek *et al*. 1997
Clear Lake, CA	Hg mine	zooplankton- lakewide	CL7-z	1994-96	Suchanek *et al*. 1997
Clear Lake, CA	Hg mine	zooplankton- near mine	CL7-z*	1994-96	Suchanek *et al*. 1997
Clear Lake, CA	Hg mine	sediment-lakewide	CL8	1992	Suchanek *et al*. 1998
Clear Lake, CA	Hg mine	sediment-near mine	CL8*	1992	Suchanek *et al*. 1998
Clear Lake, CA	Hg mine	raw water- lakewide	CL8-r	1992	Suchanek *et al*. 1998
Clear Lake, CA	Hg mine	raw water- near mine	CL8-r*	1992	Suchanek *et al*. 1998
Clear Lake, CA	Hg mine	filtered water- lakewide	CL8-f	1992	Suchanek *et al*. 1998
Clear Lake, CA	Hg mine	filtered water- near mine	CL8-f*	1992	Suchanek *et al*. 1998
Clear Lake, CA	Hg mine	oligochaetes- lakewide	CL9-o	1992	Suchanek *et al*. 1999
Clear Lake, CA	Hg mine	oligochaetes- near mine	CL9-o*	1992	Suchanek *et al*. 1999
Clear Lake, CA	Hg mine	chironomids- lakewide	CL9-c	1992	Suchanek *et al*. 1999
Clear Lake, CA	Hg mine	chironomids- near mine	CL9-c*	1992	Suchanek *et al*. 1999
Clear Lake, CA	Hg mine	chironomids- from floc	CL-FC	1995	unpublished data
Clear Lake, CA	Hg mine	zooplankton- lakewide	CL9-z	1992	Suchanek *et al*. 1999
Clear Lake, CA	Hg mine	zooplankton- near mine	CL9-z*	1992	Suchanek *et al*. 1999
Carson River, NV	gold mining	raw water - background	CR-r*	1993	Gustin *et al.* 1994
Carson River, NV	gold mining	raw water - contaminated	CR-r*	1993	Gustin *et al.* 1994
Wisconsin	pristine seepage lakes	sediment	WI	1992-93	Gilmour & Riedel 1995
Wisconsin	pristine seepage lakes	raw water	WI-r	1988-92	Watras *et al*. 1994
Wisconsin	pristine seepage lakes	filtered water	WI-f	1988-92	Porcella 1994
Onondaga, NY	wastewater	amphipods	ON-aa	1992	Becker *et al.* 1993
Onondaga, NY	wastewater	chironomids	ON-cc	1992	Becker *et al.* 1993
Onondaga, NY	chloralkali plant + sewage plant	sediment	ON	1992	Henry *et al.* 1995
Onondaga, NY	chloralkali plant + sewage plant	amphipods	ON-a	1992	Becker & Bigham 1995
Onondaga, NY	chloralkali plant + sewage plant	chironomids	ON-c	1992	Becker & Bigham 1995
New Jersey	Hg refining plant + sewage outfall	sediment	NJ	unknown	Berman & Bartha 1986
US & Canada	varied	sediment	US/C	varied	Gilmour & Henry 1991
CANADA: Wabigoon River	chloralkali plant	sediment	Wa	1978	Jackson & Woychuck 1980
Quebec lakes	natural lakes	zooplankton	QQL-z	1992-93	Plourde *et al.* 1997
Quebec reservoirs	new reservoirs	zooplankton	QQR-z	1992-93	Plourde *et al.* 1997
Quebec: Duncan Lake	natural lake	sediment	QD	1992	Tremblay *et al.* 1996
Quebec: Duncan Lake	natural lake	insects (detritivores)	QD-id	1992	Tremblay *et al.* 1996
Quebec: Duncan Lake	natural lake	insects (predators)	QD-ip	1992	Tremblay *et al.* 1996
Quebec: LeGrand Reservoir	reservoir	sediment	QL	1992	Tremblay *et al.* 1996
Quebec: LeGrand Reservoir	reservoir	insects (detritivores)	QL-id	1992	Tremblay *et al.* 1996
Quebec: LeGrand Reservoir	reservoir	insects (predators)	QL-ip	1992	Tremblay *et al.* 1996
Quebec: LeGrand Reservoir	reservoir	chironomids	QL-c	1992	Tremblay *et al.* 1996
Finland	stratified lakes	raw water	FIN	1990-91	Verta et al. 1994
Finland	stratified lakes	sediment	FIN	1993	Verta & Matilainen 1995
Germany	chloralkali plants	sediment	GER	1989-90	Wilkin & Hintelman 1991
Nan Hu, China	sewage + industrial outfalls	raw water - background	NH-r	1991	Bloom *et al.* 1994
Nan Hu, China	sewage + industrial outfalls	raw water - background	NH-r*	1991	Bloom *et al.* 1994

Sediments:

The sediments of Clear Lake show an exceptionally high level of bulk totHg contamination, up to 440 µg/g (ppm) near the Sulphur Bank Mercury Mine, some of the highest concentrations reported in the literature. The only other sites found that are in that same order of magnitude are (1) the site in Germany that is influenced by a chloralkali plant (Wilkin and Hintelman 1991) and (2) Onondaga Lake which is influenced by both sewage plants and a chloralkali plant (Becker *et al.* 1993, Becker & Bingham 1995, Henry *et al.* 1995). Figure 1 provides a plot of totHg versus meHg in sediments for those studies that reported data on both parameters at various sites worldwide. While the plot appears to show an increasing level of meHg with increasing totHg in sediments, this is somewhat misleading because this is a log-log plot. Untransformed data show that the extremely high values for the German site (two orders of magnitude higher than any other sites) throws this linear relationship off. For Clear Lake, sediment totHg concentrations vary greatly depending on whether data are reported as lakewide averages, or averages of sediments near to the Sulphur Bank mine (those values with asterisks in Fig. 1) which are generally about an order of magnitude higher. Sediments closest to the mine (reaching as high as 25 ng/g (ppb)), also have meHg concentrations about five times higher than the lakewide average.

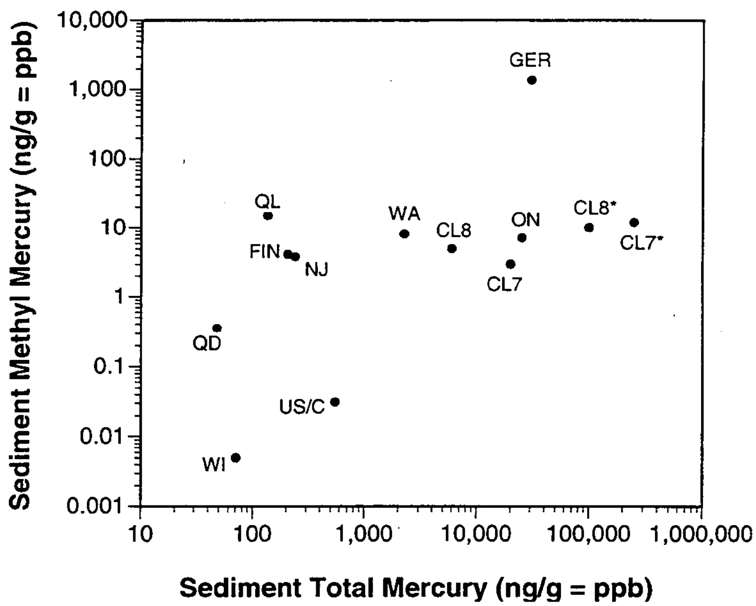

Figure 1. Concentrations of totHg and meHg (ppb) in sediments from Clear Lake and other sites reported in the literature. See Table 1 for site codes. Note both axes are log scales.

Water:

As with sediments, water in Clear Lake also has some of the highest reported totHg concentrations in comparison with other sites that have published values for both totHg and meHg (Figure 2). There is one strong outlier in this plot (for both totHg and meHg), those data from the Carson River, Nevada system (Gustin *et al.* 1994) with values one to two orders of magnitude higher than the other closest values. These data are somewhat unrepresentative of the other types of water samples in that those samples were collected directly from water, sometimes stagnant, that ran directly through tailings piles. This outlier produces a positive linear relationship between totHg and meHg for those data points reported. Other than the Carson River data (which are not from a river or lake) the Clear Lake totHg concentrations are higher than any other sites we found reported in the literature for a major water body, with values reaching as high as 0.4 µg/L (ppb) in raw (unfiltered) deep water samples closest to the Sulphur Bank Mercury Mine. While the totHg concentrations were extremely high, meHg values are moderate, typically lower than those reported from pristine Wisconsin seepage lakes and stratified lakes in Finland (see Figure 2).

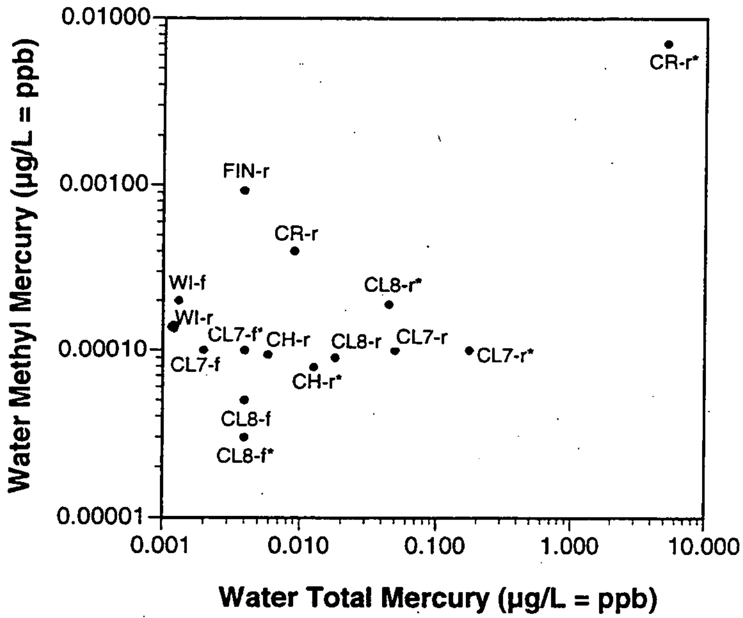

Figure 2. Concentrations of totHg and meHg (ppb) in water from Clear Lake and other sites reported in the literature. See Table 1 for site codes. Note: both axes are log scales.

Biota:

While totHg concentrations are highest in Clear Lake invertebrate biota (represented here primarily by oligochaetes and chironomid midge larvae), most meHg values are lower than those reported from other worldwide sites for which both totHg and meHg have been reported (Figure 3).

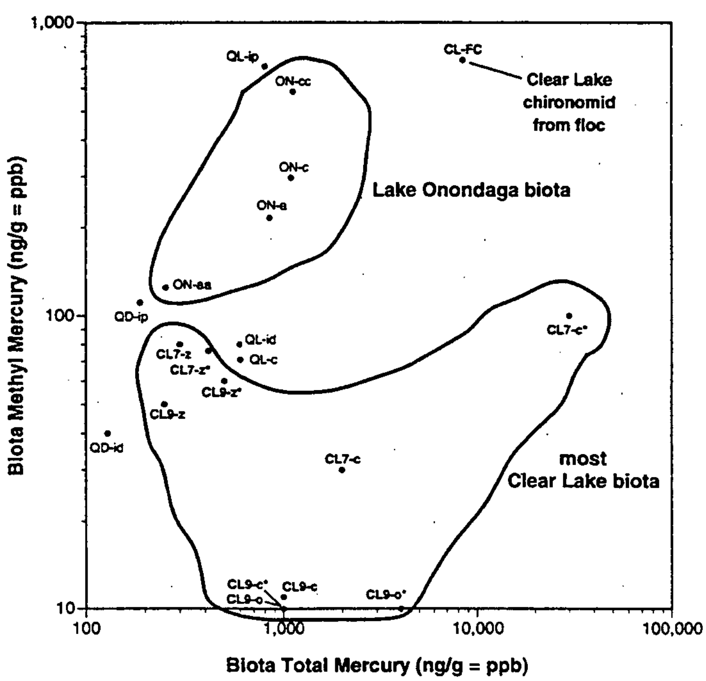

Figure 3. Concentrations of totHg and meHg (ppb) in lower trophic level biota from Clear Lake and other sites reported in the literature. See Table 1 for site codes. Note: both axes are log scales.

Both Clear Lake and Lake Onondaga in New York are categorized as highly contaminated sites. Yet biota from Lake Onondaga appear to have considerably higher proportions of meHg compared with biota from Clear Lake. The one exception to this is a sample of chironomids taken from an AMD-derived flocculent material found near the mine (Suchanek et al. in press) which has been shown to produce high levels of meHg (Mack et al. 1997).

CONCLUSIONS

The lower than expected level of Hg contamination in Clear Lake biota may reflect a difference in the bioavailability of Hg from different sources and the characteristics of the environment in which these sources are acted upon. The largest component of the bulk inorganic Hg loading in Clear Lake is believed to have been derived from the erosion of shoreline waste rock and overburden piles at the Sulphur Bank Mercury Mine, which were steeply sloped and non-stabilized before a 1992 USEPA remediation action which (1) reduced the slope angle of the piles, (2) installed rip-rap along the shoreline below the piles and (3) vegetated the slopes of the piles (although to date the revegetation has not been entirely successful). This resulted in an apparent significant reduction of totHg in surficial sediments in the Oaks Arm of Clear Lake by an estimated 50-150 ppm (Suchanek *et al.* 1997). This mostly inorganic loading (and subsequent reduction of that loading by USEPA's 1992 remediation) may have had little influence on the actual production and bioaccumulation of meHg in Clear Lake biota. Current evidence suggests that ongoing acid mine drainage (AMD) from the mine in the form of a flocculent precipitate (see Suchanek *et al.* in press) may provide Hg, sulfate and acid conditions nearshore which enhance the production of meHg by sulfate reducing bacteria (Mack *et al.* 1997, Suchanek *et al.* in press). While the bulk inorganic Hg loading (especially from previous erosion) may have not played an important role in the bioaccumulation of Hg in the Clear Lake aquatic ecosystem, it is likely that the nature of the flocculent precipitate from ongoing AMD creates conditions under which highly bioavailable Hg is produced and accumulates in the food web. This may help explain why Clear Lake ranks high as a site very contaminated with bulk inorganic Hg, yet has biota that are considerably lower in meHg than many other worldwide sites, both contaminated and non-contaminated. The significantly elevated levels of meHg in other contaminated sites like Lake Onondaga are most likely due to the nature of the Hg sources and the conditions of the environment into which those sources are deposited. In the case of Lake Onondaga Hg is derived primarily from sewage and chloralkali plants, whereas Clear Lake's primary source is from Hg mining. Based on available data, we predict that if the environmental conditions at Clear Lake were different there could be much higher concentrations of meHg in its biota. For example, if the lake were deeper (either naturally or as a result of dredging), then it would be more likely to stratify during the summer months and develop a significant thermocline and associated hypolimnion where more anoxic conditions would prevail and Hg methylation would be enhanced. When stratification broke down during winter months, this hypolimnetic water, including the meHg pool, would then mix with the rest of the Clear Lake system and likely increase meHg concentrations in biota. And, if the pH of Clear Lake were more acidic this would likely result in enhanced production of meHg as well. Considerable work remains to be done to determine which Hg source materials and which environmental conditions are more likely to produce meHg.

ACKNOWLEDGEMENTS

We thank the County of Lake for continued support and resources throughout this project, especially the Lake County Board of Supervisors, the Water Resources Division, the Air Quality Management District and the Vector Control District. Support from U.S. EPA Region IX Superfund (68-S2-9005) and the U.S. EPA Center for Ecological Health Research (R819658) made this project possible. Special thanks also go to Collin Eagles-Smith for assistance in preparing tables and figures. Although the information in this document has been funded wholly or in part by the United States Environmental Protection Agency, it may not necessarily reflect the views of the Agency and no official endorsement should be inferred.

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Appendix I. Revised and corrected table originally published in Suchanek *et al.* 1998 (Table II) comparing Hg values from different systems worldwide.

Corrected Table II (from Suchanek *et al.* 1998)

Comparison of Clear Lake data with studies of other contaminated and non-contaminated systems.
"—" indicates no data available, "nd" signifies non-detectable

MATRIX	LOCATION	SOURCE OF Hg	RANGE OF TOTAL Hg (mg/kg)	RANGE OF METHYL Hg (μg/kg)	RATIO Methyl/Total Hg	Methyl Hg as a percentage of Total Hg	REFERENCE
sediment	Clear Lake, CA, USA	Hg mine	0.5-183	0.2-15	0.0011-3.0 E-04	0.0011-3.0	this study
sediment	Wisconsin, USA	pristine seepage lakes	0.001-0.082	0.01-4.6	0.012-10 E-02	0.012-10	Gilmour and Riedel, 1995
sediment	Onondaga Lake, NY, USA	chloralkali & sewage plant	1.08-49.5	3710-10600	0.075-0.21	7.5-21	Henry *et al.* , 1995
sediment	FINLAND	stratified forest lakes	0.134-0.277	0.07-8.0	0.025-6 E-02	0.025-6.0	Verta and Matilainen, 1995
sediment	Carson River basin, USA	gold mining	0.02-1610	—	—	—	Gustin *et al.* , 1994
sediment	SWEDEN	remote lakes	0.14-0.30	—	—	—	Parkman and Meili, 1993
sediment	BRAZIL	gold mining	0.01-19.8	—	—	—	Pfeiffer *et al.* , 1993
sediment	Wisconsin, USA	seven seepage lakes	nd-0.27	—	—	—	Rada *et al.* , 1993
sediment	US & Can. lakes and res.	varied	0.007-0.12	0.15-0.64	1.3-91 E-03	0.13-9.1	Gilmour and Henry, 1991
sediment	GERMANY	chlor-alkali plant	2-59	50-2,700	0.85-46 E-03	0.085-4.6	Wilkin and Hintelman, 1991
sediment	New Jersey, USA	chlor-alkali plant	9-576	0-8	0-8.9 E-04	0-0.089	Berman and Bartha, 1986
sediment	Wabigoon River, CAN	wood treatment plant	1.3-3.0	5.5-11	1.8-8.5 E-03	0.18-0.85	Jackson and Woychuck,1980
			(ng/L)	(ng/L)			
raw water	Clear Lake, CA, USA	Hg mine	5-70	0.03-0.31	0.43-62 E-03	0.043-6.2	this study
raw water	NW Ontario, CAN	wetland: prior to flooding	—	0.05-0.28	—	—	Rudd, 1995
raw water	NW Ontario, CAN	wetland: after flooding	—	2.3	—	—	Rudd, 1995
raw water	Strait of Gibraltar	varied	—	3.9-5.0 E-05	—	—	Cossa *et al.* 1994a
raw water	Carson River basin, USA	mining	53-35,400	3.1-21	8.76 E-09 to 3.96 E-04	8.76 E-07 to 3.96 E-02	Gustin *et al.* , 1994
raw water	Wisconsin, USA	seven seepage lakes	0.7-2.1	0.05-0.33	2.4-16 E-02	2.4-16	Watras *et al.* , 1994
raw water	Clear Lake, CA, USA	mining	14-104	—	—	—	Gill and Bruland, 1990
raw water	other CA lakes, USA	varied	0.5-8.8	—	—	—	Gill and Bruland, 1990
raw water	SWEDEN	three remote lakes	—	0.09-0.32	—	—	Lee and Hultberg, 1990
raw water	Wabigoon River, CAN	wood treatment plant	3-26.5	0.18-1.4	0.68-47 E-02	0.68-47	Parks and Hamilton,1987
filtered water	Clear Lake, CA, USA	Hg mine	1-8	0.02-0.08	0.25-8 E-02	0.25-8.0	this study
filtered water	FINLAND	stratified forest lakes	—	0.058-1.20	—	—	Verta and Matilainen, 1995
filtered water	Strait of Dover	natural	0.00012-0.0013	—	—	—	Cossa *et al.* , 1994b
filtered water	Clear Lake, CA, USA	mining	3.6-12	—	—	—	Gill and Bruland, 1990
filtered water	other CA lakes, USA	varied	0.6-21.6	—	—	—	Gill and Bruland, 1990

Arsenic and mercury uptake as influenced by revegetation treatment at the Sulphur Bank Mercury Mine

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Vegetative cover of mine impacted substrates can reduce off-site transport of metal-rich colloids and sediments to surrounding watercourses. In addition, an active vegetative cover can remove water from the soil profile by evapotranspiration rather than allowing percolation through the mined materials, thus reducing contributions to acid mine drainage. Vegetative growth on these materials, however, creates the potential for bio-mobilization of metals through plant uptake and herbivory. A field survey of the Hg and As contents of native and colonizing vegetation already established at the Sulphur Bank Mercury Mine indicated that plant tissue had low (<1 ppm) levels of these two elements. Plant uptake of metals on the barren overburden was not measured because of lack of a plant cover. To estimate uptake on these materials, a greenhouse experiment was constructed using overburden samples mixed with potential revegetation amendment materials (fertilizers, lime, compost) at various rates to
simulate substrate-metal interactions following revegetation. Preliminary data indicate increased uptake of Hg (to 15 ppm) and As (to 4 ppm) in plant tissues of grasses but less than 1 ppm of either metal in oaks. Further work is in progress on the soil chemistry of the As/P/OM interaction.

Key Words: revegetation, arsenic, mercury, bioaccumulation, Clear Lake, Sulphur Bank Mine

A review on the chemical constituency, formation and potential impacts of a halloysite derivative of acid-mine drainage.

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ABSTRACT

In April 1995, a white flocculent material was observed off the shore of the abandoned Sulphur Bank Mercury Mine (SBMM), in the Oaks Arm of Clear Lake, CA, and has been found forming in a wetland north of the SBMM. The observations of this material coincided with the overflow of the Herman Pit, a 700-acre foot acidic mining pit located at the SBMM. ICP/MS and XRF analyses indicated that floc had high concentrations of aluminum (Al), silicon (Si), and iron (Fe), and that the material is a halloysite clay, dominated by SiO_2 and Al_2SiO_3. AVS/SEM analyses indicate significantly fewer sulfides in floc than neighboring lake sediments and greater metals. The mechanism of formation is metals precipitating from sulfate and mercury rich acid mine drainage (AMD). Total and methylmercury levels in floc are variable, but generally, floc has lower total mercury and higher methylmercury levels than adjacent sediments. The chemical composition of floc suggests that floc could have strong positive effects on methylating bacteria and be a source for highly bioavailable mercury. Environmental and soil conditions at the location of floc formation are favorable for high mercury solubility and extremely high rates of sulfate reduction and mercury methylation.

INTRODUCTION

The Sulphur Bank Mercury Mine was mined originally for sulfur deposits until 1871, when the focus shifted to mining cinnabar (HgS) ore deposits through shaft mining. Open-pit mining techniques were employed from 1915 until its closure in 1957, at which point the pit began to fill due to groundwater and other inputs, creating the 700 acre-foot Herman Pit (USEPA 1994). The site was declared a USEPA Superfund site in 1991. The Herman Pit has an approximate pH of 3, a total mercury concentration of 0.54 mg l^{-1} (n=7), and average sulfate concentrations of 2721 mg l^{-1} (n=14).

The overburden from the open pit was bulldozed into Clear Lake and piled between the pit and Clear Lake. The piles created a steep (60 degree), artificial shoreline which eroded at least 132 kg Hg annually into Clear Lake. However, this estimate was measured during drought years, so the actual amount could have been higher (Chamberlin et al., 1990). In 1992, the USEPA stabilized the shore by pulling the piles back from the lake and reducing the slope to 20 degrees. At that point, the mine was not considered to be a significant source of mercury to Clear Lake (USEPA, 1994).

Mercury pollution in Clear Lake has also been attributed to discharge of mercury-rich geothermal fluids (Varekamp and Waible, 1987). The presence of warm springs along fault lines in Clear Lake are identified as the probable source of Hg rich waters entering. However, the levels are not presented or compared with levels elsewhere in the lake. Also, they assume that the SBMM can only contribute mercury to lake sediments through erosion of tailings, while allowing that springs can deliver aqueous mercury to Clear Lake sediments.

In April 1995, a white material was observed off the shore of Sulphur Bank Mercury Mine (SBMM), in Clear Lake, CA. The material forms in depressions close to shore and in large, consolidated formations up to 10 cm thick and 10, 000 m^2 offshore. The observations of this material coincided with the overflow of the Herman Pit, which sits approximately 13 feet above Clear Lake.

Dye and tracer studies show that water originating from the Herman Pit and other sources at the SBMM move through the overburden piles and reach Clear Lake (Oton et al. 1998). Mercury solubility is pH dependent in the subsurface flow through the overburden piles at the SBMM, and low pH waters moving through the piles will strip mercury (Lechler). The groundwater reaching Clear Lake has a low pH, high sulfate concentrations (up to 80 mM), and high mercury concentrations (41.1 $\pm$ 10.4 ppb, n=9) from the oxidation of sulfides, and leaches out of the sediment adjacent to the face of the mine site (Nelson). The sulfate leaching into the sediment adjacent to the mine is reduced by bacteria during the oxidation of organic matter and creates highly reduced sediment conditions in the summer and fall, when water temperatures are high.

This flocculent material represents a mechanism for the introduction of mercury into Clear Lake which has high levels of methylmercury in fish tissues. The discovery of a mine-derived substance in Clear Lake fueled investigations into the chemical constituency and effects on methylation of the material to determine its impact on the mercury methylation in Clear Lake.

CHEMICAL COMPOSITION OF FLOC

Inductively Coupled Plasma / Mass Spectrometry was performed to determine the metal composition of the floc, and Scanning Electron Microscopy (SEM) to determine the structure. The floc material was relatively high in aluminum and iron, and had higher levels of silica than found in Herman Pit sediment (figure 1). The SEM (figure 2) showed the basic structure of the floc, which is composed of aluminosilica tubules embedded in a mostly aluminum matrix. Mixing lake and pit waters in a beaker yielded a material identical to that found in the lake in both appearance and composition (Reynolds, 1997). This structure and chemical composition is typical of kaolin-type materials, and the floc is considered to be either a halloysite ($Al_2SiO_5(OH)_4$)or an amorphous aluminum hydroxide($Al(OH)_3$)precipitate from acid mine drainage (AMD).

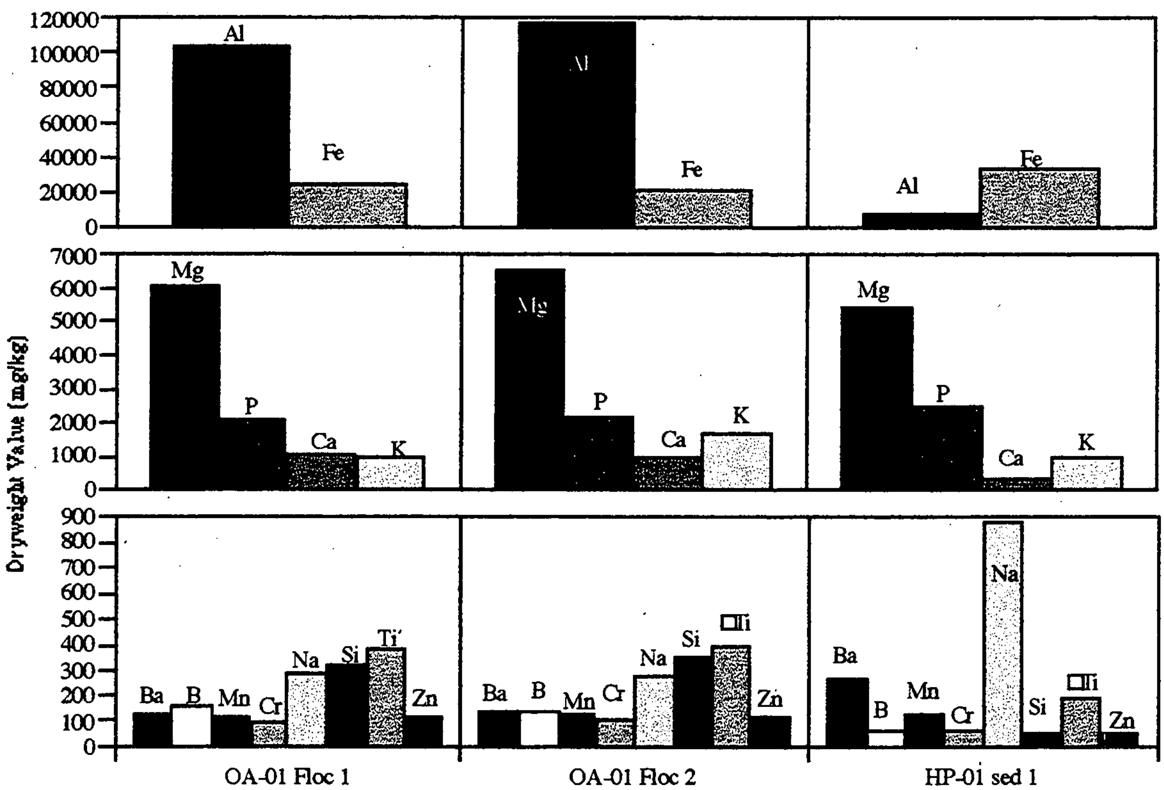

Figure 1. ICP/MS Scan of Natural, OA-01 Floc and Herman Pit Sediment. Note different scales.

Figure 2. Scanning Electron Micrograph of dried, natural floc, showing backbone and aluminosilicate tubules (approximately 40 μm long)

The metals held in solution in the AMD will precipitate when they mix with alkaline lake water, forming halloysite ($Al_2Si_2O_5(OH)_4$), water and releasing hydrogen ions (Lechler, 1998), or $Al(OH)_3$ (Zierenberg, pers. communication). Floc forms as an easily suspended material close to shore on rocky depressions, and as a thick, consolidated mass. Floc forms both as the result of seasonal overflow events and year-long, variable underground flows, and forms in larger amounts in the winter and spring, though it will form in response to light rains in the late summer/early fall. The observed seasonality is probably dependent on the hydraulic head pressure on the groundwater flow and the hydraulic conductivity of the sediment in Clear Lake. Floc has been observed forming below the sediment surface in the Oaks Arm, and on the sediment surface. Floc forming from pit overflow could be carried away from the site, whereas floc formed in or just above the sediment would not be as easily disturbed by lake currents except close to shore where scouring occurs.

When floc samples from the Oaks Arm are compared with samples from site OA-01 for total and methylmercury, floc samples generally have lower total mercury and higher methylmercury than OA-01 sediments (figure 3). The variability may derive from mixing with sediments or variations in groundwater/rock interactions in the tailings piles. Natural floc formations are generally mixed with sediments and difficult to sample purely. Considering the high total mercury of sediments adjacent to the mine site, and the high dry/wet weight ratio of floc, small amounts of sediment mixing could have disproportionately large effects on the total mercury observed in floc. Conditions in the tailings piles could be drastically altered by redox conditions in the piles, resulting in higher or lower mercury solubility during floc formation. Also, floc is possibly much lower in mercury than surrounding sediments, so mercury could be diffusing into floc, yielding concentrations that are a function of time and the concentration gradient.

In order to determine the potential toxicity of floc to biota, Acid Volatile Sulfide (AVS) and Simultaneously Extracted Metals (SEM) analysis was performed on floc and adjacent sediment. The results indicate that OA-01 sediments have an average AVS concentration of 109 mM, whereas floc has non-detectable levels of AVS. Floc also has significantly lower SEM concentrations than sediment (one-way ANOVA, $p=0.02$), but a greater ratio of SEM:AVS. Three sediment samples had SEM:AVS ratios of .007, .085., and .106, whereas two floc samples had ratios of .817 and 2.05. An SEM:AVS ratio of less than one is indicative of metal binding to sulfur, decreased solubility of metals, and lower toxicity. A ratio greater than one does not, however, indicate the opposite: metals may be bound by other ligands in sediments, and may not be dissolved. (Chapman et al., 1998; Slotton and Reuter, 1995). This evidence suggests that floc may be a source of unbound metals, but is not definitive proof.

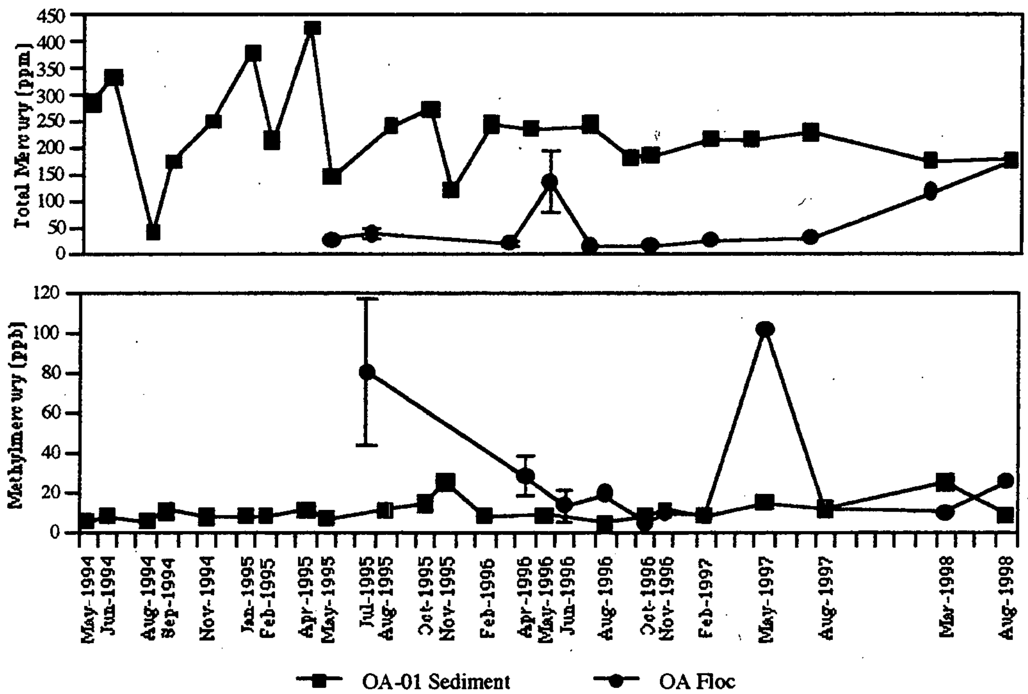

Figure 3. Total and methylmercury levels for Oaks Arm sediment and floc from May 1994 to August 1998. Floc sampling sites are not fixed, as samples are taken where floc occurs. Points with error bars are means of mulitple samples.

POTENTIAL EFFECTS OF FLOC ON MERCURY METHYLATION

The availability of metals in the sediment to biota is dependent on many factors, including the concentration and species of metal, the sediment/water partitioning, temperature, pH and redox potential

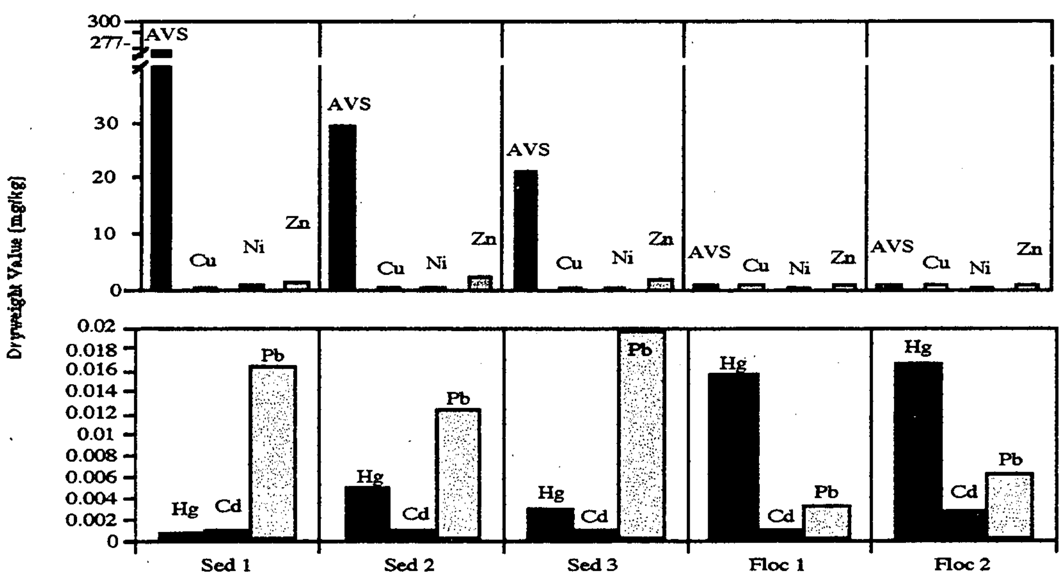

Figure 4. Acid Volatile Sulfide (AVS) / Simultaneously Extracted Metals analysis of OA-01 sediment and floc samples, July 1995. Note different scales. AVS levels are much higher than all metals in all samples except for Floc 2, which has a higher concentration of zinc (0.84 mmols/g) than AVS (0.709 mmols/g).

(Luoma 1989). Metals in the sediment are not represented by one species, but by a suite of species (with varying degrees of bioavailability) bound by soluble ligands, mostly among iron oxides, manganese oxides, various types of organic materials, and sulfides (depending on redox potential of sediments). The toxicity of metals like Pb, Cd, and Cu is more dependent on the activity of the free metal ion than mercury, which reaches toxic levels through methylation and bioaccumulation (Chapman, et al. 1998).

Generally, methylmercury production is greatest in acidic (Lechler, 1998), anaerobic environments with plentiful organic material and few sulfides (Jackson 1989). The area of the Oaks Arm where floc forms is acidic, reduced, sulfate-rich and high in organic material. The acidity results from AMD and from the precipitation of halloysite (floc), which produces hydrogen ions. The low pH would allow the dissolved mercury species entering the area with AMD to remain soluble, rather than precipitating.

The input of high levels of sulfate through AMD is important for methylmercury production in Clear Lake. Increased sulfate concentrations have been shown to stimulate methylmercury production in some situations (Gilmour 1992). Sulfate-reducing bacteria use sulfate as a terminal electron acceptor to oxidize organic material, producing sulfide and methylmercury as a by-product. Because sulfate reducing bacteria can methylate mercury in pure culture, and molybdate, a specific sulfate reduction inhibitor, completely stops methylation in certain situations, they have been widely implicated as important methylators of inorganic mercury (Mack, 1998; Compeau and Bartha, 1995; Gilmour 1992).

Clays, including aluminosilicates like halloysite, strongly influence the methylation of mercury by complexing the Hg and by affecting the biochemical functions of microorganisms. Mercury methylation rates represent the dynamic interaction between methylating and demethylating microorganisms, and very high methylation rates indicate a local environment that favors methylating bacteria over demethylators. Mercury methylation or demethylation will be enhanced by these materials depending on the nature, abundance and surface chemistry, local sediment type, and local environmental conditions (Jackson 1989). Clays, including kaolin-type clays, are largely responsible for the immobilization of mercury in natural environments, through surface complexation by the iron oxide and manganese oxide coatings on clay particles (Desauziers, et al. 1997). It is unknown how halloysite differs from other clays in its affects on mercury or microorganisms, but they are varied enough that the effects should be interpreted cautiously.

CONCLUSION

The Sulphur Bank Mercury Mine leaches some amount of AMD into Clear Lake on a continual basis, and this seepage is a source of dissolved species, sulfate, and acidity, all of which can stimulate mercury methylation. These conditions favor mercury methylation regardless of microbial substrate additions, as acidity increases mercury solubility and sulfate is a necessary electron acceptor for a group of important methylators.

Aluminosilicates similar in composition to floc have dramatic effects on methylation by altering microbial dynamics directly or other microbial functions which may indirectly affect Hg availability, and by binding mercury on their surfaces. Even without a stimulating factor, the methylation potential in the area adjacent to the mine is extremely high based on high sulfate, low acidity, and high mercury concentrations

The appearance of floc is significant at least as an indicator of AMD seepage into the Oaks Arm of Clear Lake and the wetland North of the SBMM. It could, however, be a source for weakly bound dissolved Hg species, a substrate for methylating bacteria, and a mechanism for transport over a large area from the mine, and could drastically increase methylation in very small amounts.

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Geochemical Features of Water-Rock Interactions at the Sulfur Bank Mercury Mine, Lake County, California

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ABSTRACT

The Sulfur Bank Mercury Mine on the eastern shore of Clear Lake is the source of poor quality acid mine drainage seeping into Clear Lake. Lateral and vertical geochemical trends in ground water composition point to a number of redox reactions taking place as a function of subsurface water-rock interactions. An understanding of these reactions suggests opportunities to remediate the acid mine drainage through suppression of undesirable geochemical reactions. Two geochemically based remediation steps are proposed.

INTRODUCTION

The Sulphur Bank Mercury Mine (SBMM) is located on the shore of Clear Lake in Lake County, California. The abandoned Herman Pit is now flooded and leaking through Herman Impoundment into Clear Lake. Interaction of ground and surface waters with sulfide mineral-bearing mine wastes, in and around Herman Pit, produces metal-laden acid mine drainage (AMD) and makes these leaking subsurface waters particularly deleterious for Clear Lake. Because of the argillic alteration of proximal bedrock by acidic hydrothermal waters during the formation of the original mercury deposit, there is little or no acid buffering capacity left in the surrounding rocks to neutralize AMD currently leaking from the abandoned workings before it reaches Clear Lake.

Prior to any attempt to remediate this system, a thorough understanding of the geochemistry of the water-rock interactions in and around Herman Pit is necessary. In order to characterize this geochemistry, several subsurface auger cuttings samples and water samples were collected and analyzed. Because the general hydrology of the site is known, the spatial geochemical variations from these samples can be interpreted in the context of hypothetical flow paths through the system. Geochemical trends can infer certain geochemical water-rock reactions, and an understanding of the details of these reactions can suggest how certain manipulations of the system in a remediation phase might suppress undesireable geochemical reactions, improving water quality.

MATERIALS AND METHODS

A number of surface and subsurface water samples and subsurface auger cuttings from Herman Impoundment were collected along a hypothetical flow path from east to west through the system in cooperation with ICF Kaiser. Subsurface auger cuttings samples were collected from Herman Impoundment monitoring wells (figure 1, locations 5 and 6) and water samples were obtained from the same monitoring wells, "green pond", Herman Pit, springs or seeps (figure 1, location B), and Clear Lake itself (sample locations shown on figure 1). The cuttings samples were placed in capped glass sample bottles and refrigerated until analysis at the University of Nevada (UNR). Water samples of low pH were collected without filtration or acidification and refrigerated until analysis at UNR. The Clear Lake water sample was filtered (0.45u) and acidified in the field, and refrigerated until analysis at UNR. Samples were collected during both May and August, 1998. Because sampling locations were not identical during each campaign, the data are presented separately below. Multielement analysis was conducted by ICP-AES, Hg was determined by cold vapor atomic absorption spectrometry, and anions were determined by ion chromatography.

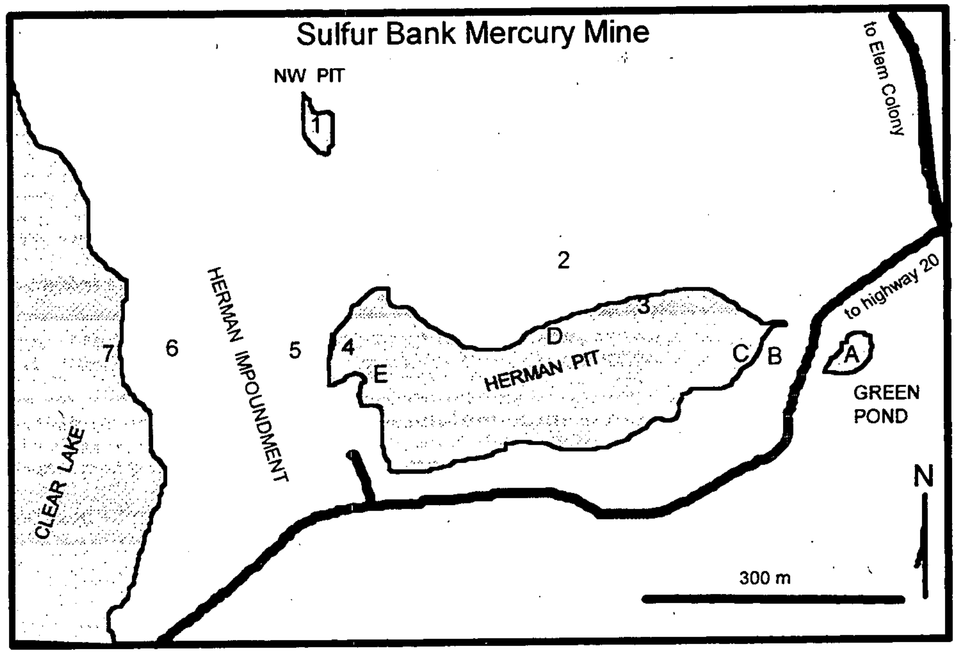

Figure 1. Generalized map of SBMM showing May (numbers) and August (letters) sampling locations.

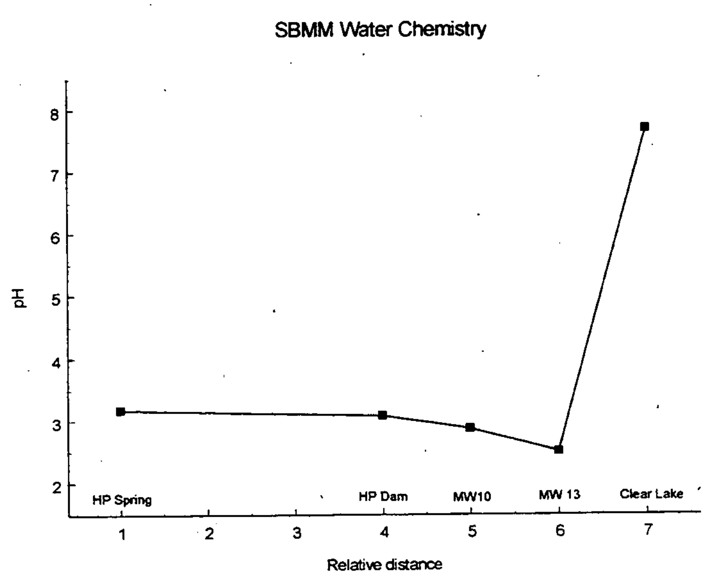

Figure 2. The pH of water samples along the hypothetical flow path from Herman Pit to Clear Lake in May, 1998 (east and west are reversed).

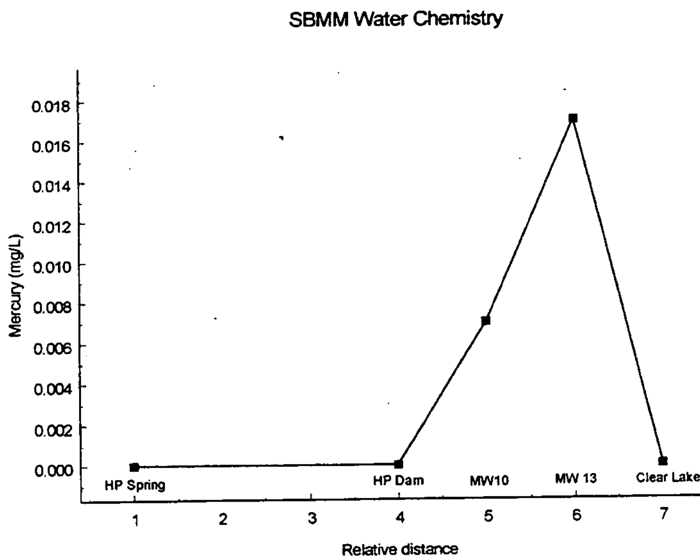

Figure 3. Dissolved mercury concentration of waters collected along hypothetical flowpath from Herman Pit to Clear Lake in May, 1998 (east and west are reversed).

RESULTS

Systematic vertical and lateral aqueous geochemical trends were observed in both May and August of 1998. Figures 2 and 3 show the pH dependence of Hg solubility and demonstrate that high Hg concentrations are produced in the subsurface where water-rock ratios are low. Figure 4 indicates that sulfate increases systematically as pH decreases, suggesting sulfide oxidation as the cause of low pH and high Hg solubility. This source of AMD is often modeled with the general reaction shown in (1) below:

$$2FeS_2 + 7O_2 + 2H_2O \rightarrow 2Fe^{2+} + 4SO_4^{2-} + 4H^+ \qquad (1)$$

Figure 5 shows that sulfate also increases in the subsurface between "green pond" and a spring near Herman Pit. Along this subsurface flowpath, pH also decreases and Hg concentration in the issuing spring water reaches 1.09 ug/L.

Silicon and Al also reach significant concentrations in the Herman Impoundment subsurface waters and then decrease to low values as the halloysite precipitate (locally called "floc") depletes the AMD in these components upon interaction with the alkaline water of Clear Lake (figure 6). Reaction (2) shows that this precipitation reaction produces hydrogen ions, resulting in a locally more-acidic environment conducive to enhanced Hg solubility and methylation.

$$2Al^{3+} + 2H_4SiO_4 + 3H_2O \rightarrow Al_2Si_2O_5(OH)_4 \cdot 2H_2O + 6H^+ \qquad (2)$$

Figure 7 shows the production of nitrate at the expense of ammonium in Herman Impoundment subsurface water during the low water-rock-ratio oxidation of ammonium. This ammonium oxidation also occurs between "green pond" and the southeast seep or spring near Herman Pit, as shown in figure 8.

Vertical geochemical trends in auger cuttings from the two sampled monitoring wells in Herman Impoundment (MW-10 and MW-13) provide additional information about systematic spatial geochemical

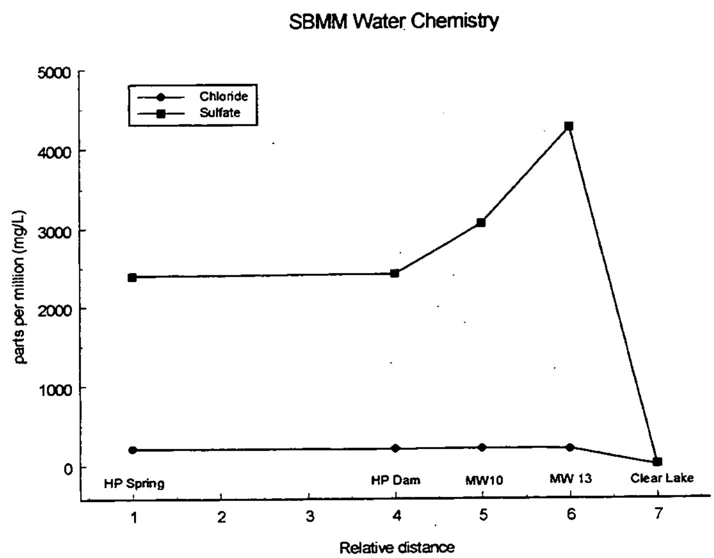

Figure 4. Sulfate and chloride contents of waters collected along hypothetical flowpath from Herman Pit to Clear Lake in May, 1998 (east and west are reversed).

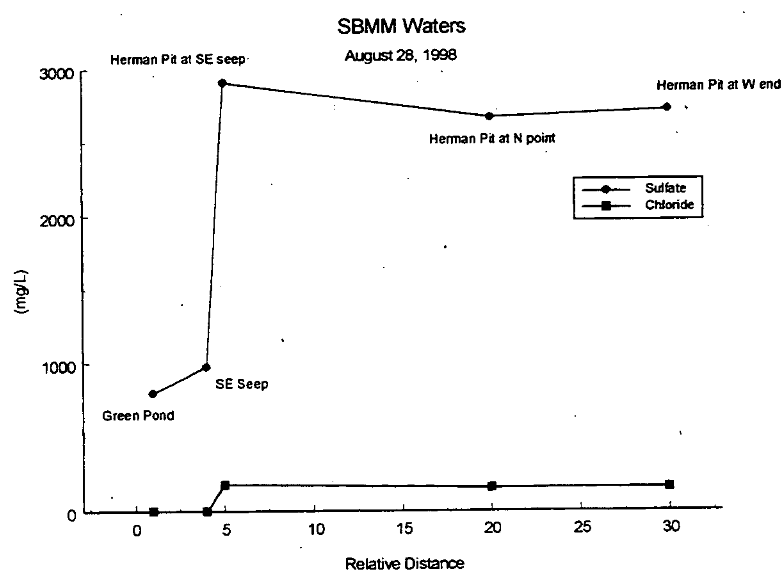

Figure 5. Sulfate and chloride contents of waters collected along flowpath from 'green pond' to Herman Pit in August, 1998 (east and west are reversed).

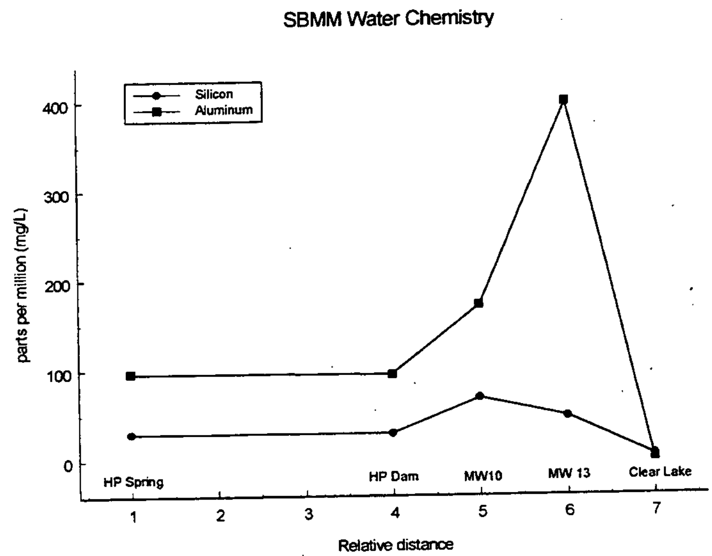

Figure 6. Silicon and aluminum content of waters collected along hypothetical flowpath from Herman Pit to Clear Lake in May, 1998 (east and west are reversed).

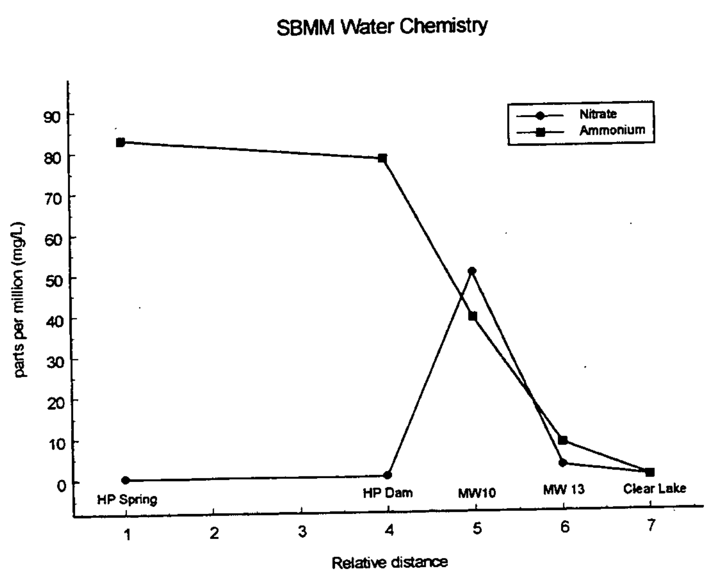

Figure 7. Nitrate and ammonium contents of waters along the hypothetical flowpath from Herman Pit to Clear Lake in May, 1998 (east and west are reversed).

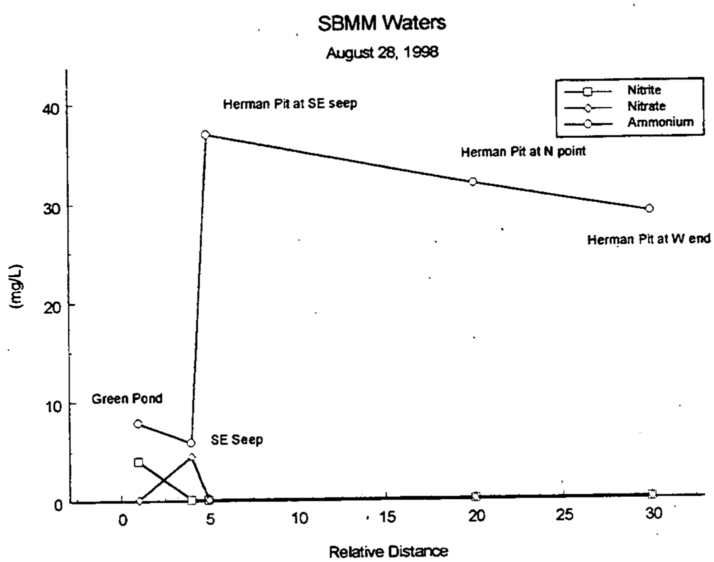

Figure 8. Nitrite, nitrate, and ammonium contents of waters along flowpath from 'green pond' to Herman Pit in August, 1998 (east and west are reversed).

trends. Nitrate contents of mine wastes are higher above the water table, and slurry pH's are lower above the water table in both wells. This is indicative of oxidizing geochemical reactions taking place within the mine wastes above the water table in Herman Impoundment, and implicates atmospheric oxygen directly as the source of at least part of the oxygen driving subsurface reactions. The presence of small amounts of nitrite and nitrate in Herman Pit waters indicates that there is at least a small amount of oxygen present in these waters as they begin to flow through the Herman Impoundment waste materials. If this were not the case, thermodynamic modeling indicates that sulfide oxidation would have consumed essentially all available oxygen and reduced nitrite and nitrate to nitrogen and ammonium as, for example, in the following reaction:

$$NO_3^-{}_{(aq)} + H_2S_{(aq)} + H_2O \rightarrow SO_4^{2-}{}_{(aq)} + NH_4^+{}_{(aq)} \qquad K = 10^{+78.4} \qquad (3)$$

The large positive value of the equilibrium constant K in reaction (3) indicates that the reaction proceeds strongly to the right.

CONCLUSIONS AND RECOMMENDATIONS

Lateral ground water sampling was conducted along a hypothetical ground water flow path to elucidate the nature of geochemical reactions that are affecting water quality at SBMM. Both lateral and vertical geochemical variations in subsurface waters at SBMM indicate that systematic and progressive water-rock interactions are taking place. These geochemical variations indicate that specific reactions are taking place, and thermodynamic modeling points to a number of redox reactions which are probably influencing subsurface water chemistry. Oxidation of sulfides in the low water-rock-ratio subsurface environment, especially beneath Herman Impoundment, drives ground water chemistry to low pH, high sulfate, high Fe, Al, Si, and Hg (and other elements) conditions.

Oxygen appears to be entering the subsurface environment both vertically through the mine wastes and laterally with Herman Pit water that is seeping through the wastes before entering Clear Lake. Addition of large amounts of scrap iron to Herman Pit may be effective at suppressing both hydrogen ion and oxygen availability (Shelp et al., 1995) before Herman Pit waters seep through the impoundment mine wastes. Reaction (4) shows how the consumption of iron by acidic Herman Pit waters would consume both oxygen and hydrogen ions:

$$Fe^{o}_{(scrap\ iron)} + {}^{1}/_{2}O_{2(aq)} + SO_4^{2-}{}_{(aq)} + 2H^{+} \rightarrow FeSO_4^{o}{}_{(aq)} + H_2O \qquad (4)$$

Covering the ground surface over Herman Impoundment and adjacent areas with charcoal or some other reactive, reduced material should effectively moderate the entry of oxygen vertically into the mine wastes, as described by reaction (5a). Reaction (5b) indicates that carbon would successfully compete for oxygen with reduced sulfur in sulfide minerals:

$$C_{(charcoal)} + O_{2(g)} \rightarrow CO_{2(g)} \qquad K = 10^{+92.5} \qquad (5a)$$

$$SO_4^{-2}{}_{(aq)} + 2C_{(charcoal)} \rightarrow 2CO_{2(g)} + S^{-2}{}_{(aq)} \qquad K = 10^{+39.5} \qquad (5b)$$

These combined remedial actions might dramatically improve the quality of the current AMD waters reaching Clear Lake.

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Overview of the Remedial Investigation / Feasibility Study process at the Sulphur Bank Mercury Mine Superfund Site.

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ABSTRACT

An overview was provided of the RI/FS (Remedial Investigation / Feasibility Study) process at the Sulphur Bank Mercury Mine Superfund Site initiated by the Environmental Protection Agency's Superfund Program from Region IX, San Francisco, California. This process will provide USEPA with remedial recommendations intended ultimately to lower mercury concentrations in fishes from Clear Lake. Scientists from U.C. Davis, University of Nevada @ Reno, U.C. Santa Barbara, Stanford University, ICF Kaiser, Battelle Northwest, US Environmental Protection Agency, US Army Corps of Engineers, State of California Division of Toxic Substances, Cal EPA, California Department of Conservation, California State Regional Water Quality Control Board and the Lake County Environmental Health Department are collaborating to complete this process for USEPA. These studies previously focused primarily on mercury contaminated lakebed sediments, but more recently are focused on the flow of acid mine drainage from the Sulphur Bank Mercury Mine into Clear Lake. Many potential alternative remedial solutions to solve this problem are being considered.

Hydrological Transport between the Sulfur Bank Mercury Mine and Clear Lake Using Gas Tracer

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ABSTRACT

Herman Pit is the larger of two pits excavated at the Sulfur Bank Mercury Mine (SBMM) located at the end of the Oaks Arm of Clear Lake, California. It is presently filled with water originating from a combination of groundwater, surface seeps, subterranean streams and surface runoff. A Sulfur Hexafluoride (SF6) gas tracer study has been conducted to verify and quantify subsurface flow between the Herman Pit and Clear Lake. A previous Rhodamine tracer study showed higher than expected flow rates. The gas tracer was distributed uniformly in the pit water and tracer concentrations in the pit, in monitoring wells and in the near lake shoreline were checked over time. SF6 gas was extracted from the water following the head space procedure described by Wanninkhof et al. 1987 and 1991, and injected through a column of $Mg(ClO_4)^2$ into a small sample loop flushing the gas into a gas chromatograph. There was an exponential decrease in the SF6 concentration over time, due to gas exchange at the air-water interface, sponging bubbles seeps and sub-surface flow between the pit and the lake. Vertical and horizontal distributions in the SF6 tracer concentration were observed during the experiment. Wells previously identified as wells of high Rhodamine concentration were observed during the experiment. Wells previously identified as wells of high Rhodamine concentration showed presence of SF6 two weeks after injection. Results support a groundwater flow between Clear Lake and Herman Pit similar to that obtained with the Rhodamine-WT study, with 6,000-8,000 gallons per minute (gpm) being the likely range.

Keywords: Sulfur Hexafluoride, gas tracer, groundwater, Clear Lake, Herman Pit.

INTRODUCTION

The Herman Pit, the largest of two mine pits excavated at the SBMM, is located adjacent to the end of the Oaks Arm of Clear Lake, California. It is currently filled with water probably from groundwater flow throughout the year, surface seeps, subterranean streams and surface runoff from rainfall. During spring of 1997 a dye tracer study was conducted to verify and quantify subsurface flows between the Herman Pit and Clear Lake (Schladow and Massoudi, 1997). Rhodamine-WT dye was injected into the pit and its concentration was monitored for over 3 months. The preliminary estimates of flow through the waste rock piles from the Herman Pit into the lake were in the range of 6,000 - 8,000 gpm. Additionally, later laboratory experiments conducted on the Rhodamine-WT dye indicated that some degradation of the compound could occur during the experiment. In order to verify the results obtained from previous studies, a new tracer study using two gas tracers (^3He and Sulfur Hexafluoride, SF_6) was recommended. By using SF_6 and He^3 the exchange rate between the pit and the shallow groundwater could be quantified. Two gases are needed since there are two unknown sinks of the gases, groundwater flow and gas exchange. Whereas the gas exchange across the air-water interface can be estimated based on many previous experiments (Clark, 1997), the presence of bubble seeps in the Herman pit represented another sink.

MATERIALS AND METHODS

1. Injection

SF6 and ^3He gas tracers were distributed uniformly in the pit water. By checking both tracer concentrations in the pit, in monitoring wells and in the near lake shoreline over time, the magnitude and potential distribution of the subsurface flow could be determined. Since SF6 is a gas and it is lost from solution via gas transfer across the air-water interface and a bubble water interface, the gas transfer flux was to be estimated by adding the second gas tracer, an isotope of helium gas ^3He. ^3He and SF6 gasses were injected into the pit using a low flow pump that mixed the SF6, ^3He and water in a cylinder and discharged the water into the pit. ^3He was also injected directly into the pit using Cu tubing. A 200-gpm pump pumped water from the bottom of the pit and discharged below the surface to facilitate mixing. Prior to the tracer injection, samples from the pit and the wells were taken to establish background levels for SF6 and ^3He.

2. Tracer measurements:

2.1. Herman Pit

Tracer measurements in the Herman Pit were taken with a 12V MasterFlex portable peristaltic sampling pump from 6 different points (fig. 1), at 3 different depths (10', 50' and 75') and collected in 300cc BOD bottles to monitor the SF6 concentration. Bottles were rinsed before taking the sample and filled to overflowing allowing with the pit water running along the inside wall of the bottle to avoid the formation of bubbles that could affect the SF6 concentration. Bottles were stored in a cooler and transferred to the University of California at Davis and inspected for formation of bubbles on the walls. The samples for ^3He analysis were collected from points 2, 4 and 6 at 10' and 50' into 3/8" refrigeration copper tubing cut to 14" lengths (fig. 2). The ends of the copper tube were crimped with clamps to prevent the escape of the sample. The 12V-pump was used from a Zodiac using a _" black Nylon hose. Before any water sample was taken the hose was purged for about 1 to 2 minutes. The 6 sampling points were set using a Global Positioning System (GPS) along the E-W pit axis. The end of the hose always remained in the water as the Zodiac moved along the axis to the next point. Additionally, the SF6 concentration of the rising bubbles was estimated. The gas was sampled using a funnel over the bubbles released. The gas was transferred to vacutainers using 50cc glass syringes (fig. 3).

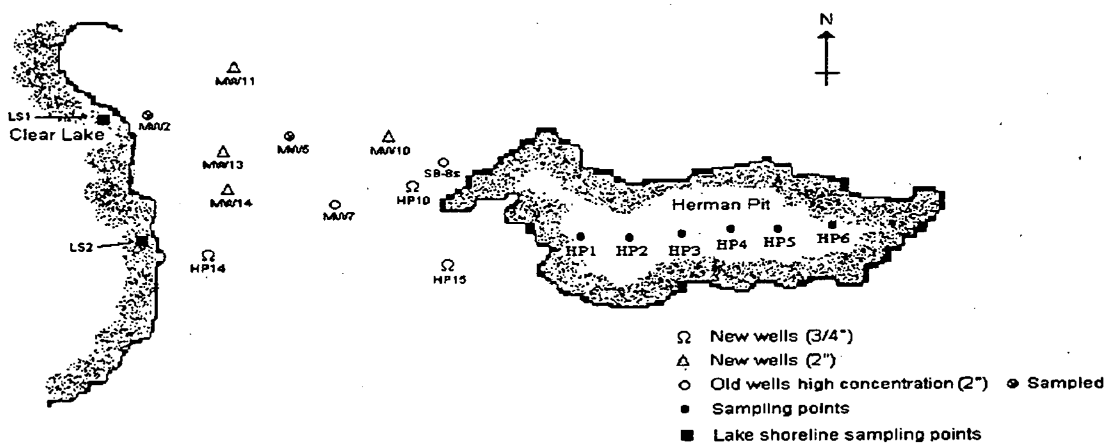


Figure 1: Sampling points at the Herman Pit and location of the monitoring wells

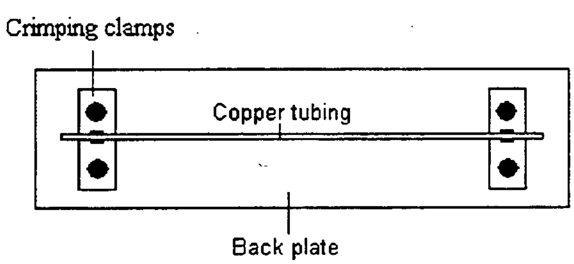


Figure 2: Unit used for ^3He samples.

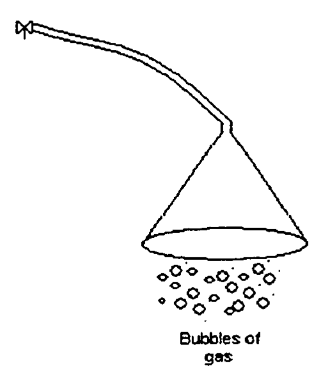


Figure 3: Funnel used to sample the gas bubbles released in the pit.

2.2. *Sampling wells*

A series of 9 wells were sampled before and after the tracer injection in the Herman Pit (fig. 1). The table 1 shows the type of pump used and the well outside diameters. Wells are divided in old wells (diameter: 2"), and new wells (diameters: _" and 2"). The new piezometers can achieve sediment-free water and increase the efficiency of the piezometers by increasing the porosity and hydraulic conductivity in the zone surrounding the piezometer. Old wells were installed for previous tests and two of them (MW2 and MW5), considered of higher concentration according to previous experiments (Schladow and Massoudi, 1997) were also analyzed this time.

Table 1: Physical characteristics of the monitoring wells

Diameter	Well name	Pump used	Hose diameter	Well total depth
New wells	HP10	Peristaltic	3/8"	10
_"	HP14	Peristaltic	3/8"	15
	HP15	Peristaltic	3/8"	8
	MW10	Submersible	_"	39.2
New wells	MW11	Submersible	3/8"	75.4
2"	MW13	Submersible	3/8"	67.8
	MW14	Submersible	3/8"	47.6
Old wells	MW2	Peristaltic	3/8"	
2"	MW5	Submersible	3/8"	

3/8" black nylon hoses, characterized by low reactivity and low adsorption, were connected to the peristaltic pump to be used in the _" diameter wells and in the old 2" MW2 well. The 2" diameter pump was lowered down each new 2" well and old MW5 well, and the water was pumped through 3/8" black nylon hose. Four ES40 submersible pumps were connected in series for use in the deep wells (MW11, MW13, MW14 and MW5). Three low flow submersible pumps in series were used in the well MW10. Hoses for each well were kept separate and were stored in plastic bags to minimize cross-contamination. Wells were pumped at a flow range of 0.3 to 1.0 gpm until a volume corresponding to 3 well casings was discharged and then samples were taken. The wells generally ran turbid at the beginning but at the time of sampling, water was clear.

2.3. *Lake shoreline*

The series assembly 3-low flow submersible pumps were used to sample the lake shoreline. Two points were sampled (fig. 1) at the lake shoreline for SF_6. Samples were collected taking into account considerations addressed above. Pumps were submerged approximately two feet from the surface and samples were taken after sufficient time for the hose to purge (1-2 minutes).

3. *Analysis in the laboratory*

3.1. *SF_6 gas tracer*

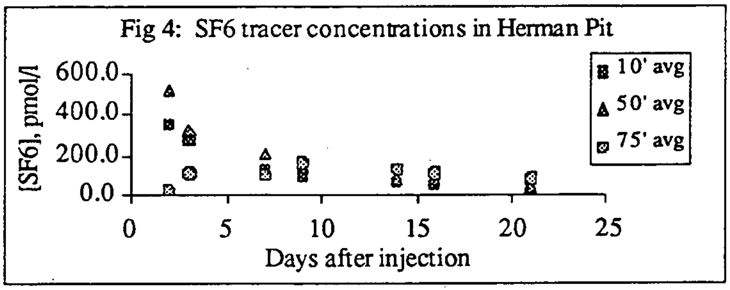
Water samples in BOD bottles and copper tubing were brought in a cooler to the Civil and Environmental Engineering laboratory at UCD. SF_6 gas was extracted from the samples by filling a 50cc glass syringe with the sample water (10 or 30 cc depending on the case). High purity nitrogen was used to fill the headspace above the sample. The syringe was shaken for 3 minutes until SF_6 equilibrated with the headspace (as SF_6 has very low solubility this method extracts nearly all SF_6 from the water sample). The head space gas was injected into vacutainers (Wanninkhof *et al.* 1987, 1991). The gas samples were sent to Dr. Jordan Clark at the University of California, Santa Barbara who conducted the pertinent analysis. Once there, they were injected through a column of $Mg(ClO_4)_2$ (to remove water vapor) into a small sample loop flushing the gas into a gas chromatograph. The SF_6 gas is separated from other gases with a molecular sieve 5A column held at room temperature (Clark *et al.*, 1997).

RESULTS

1. SF6 gas tracer

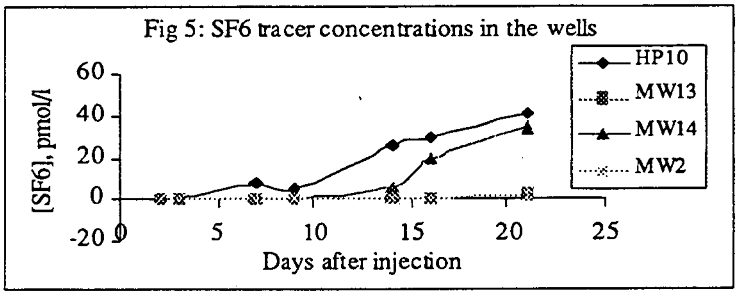
1.1. Herman Pit

Background concentrations were measured prior to injecting the gas tracers, and SF_6 concentration were found to be below the detection limit (0.01 pmol/l, 1 pmol = 10^{-12} mol). The concentration distribution of the SF_6 gas tracer is shown in figure 4. There is an exponential decrease in concentration over time. After the injection, the higher SF_6 concentrations were found at a depth of 50 feet since injection was at this depth. One week after injection the lower concentrations were at the surface due to loss across the air-water interface, diffusion through the water medium and hydraulic transport to Clear Lake. The SF_6 concentration of the rising bubbles at the northern side of the pit was analyzed and the higher SF_6 concentration results show that bubbles are stripping some of the gas tracer (~2700 ppt_v). Results obtained for ^3He are not reported since they do not show major difference between background readings and after-injection analysis.



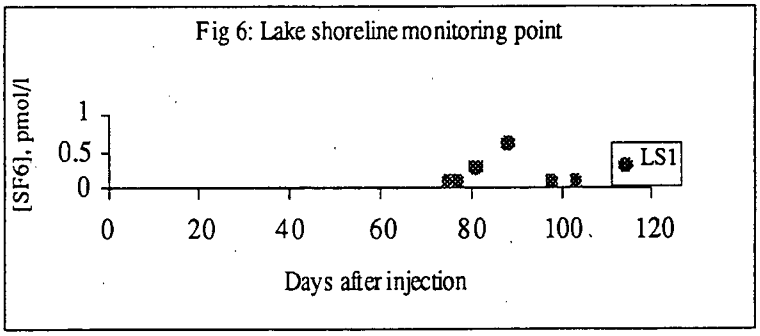
1.2. Monitoring wells

A series of 9 wells were monitored for SF6. Background measurements of SF6 concentration were below the detection limit (0.01 pmol/l). No major variations in depth to water were detected during the monitoring month. Three weeks after injection all wells showed presence of SF_6 (fig. 5) in accordance with hydraulic flow pattern from previous tests (Schladow and Massoudi, 1997).



1.3. Clear Lake

Two points at the lake shoreline (offshore of the mine face) were also sampled twice per week. SF_6 was detected in the samples and results are shown in figure 6.



1.4. About flows in Herman Pit

1.4.1. SF_6 flux across the air-water interface:

Using the data presented above, a preliminary SF_6 gas exchange flux, F SF_6, across the air-water interface has been calculated.

$$F\ SF_6 = k\ A\ (C_{sur} - C_{eq}) \qquad \text{Eq. 1}$$

Where k is the gas transfer velocity, C_{sur} and C_{eq} are the concentrations of the SF_6 gas in the air-water interface and in equilibrium with the atmosphere respectively, and A is the pit surface area.
Assuming:
$k = 3$ cm/h $= 8.3\ 10^{-4}$ cm/s
$A = 21$ acres $= 8.5\ 10^{8}$ cm^2
$C_{sur} = 350$pmol/l $= 3.5\ 10^{-13}$ mol/cm^3
$C_{eq} = 0.0005$ pmol/l ~ 0 mol/cm^3

Then, the flux of SF_6 across the air-water is: F $SF_6 = 2.4\ 10^{-7}$ mol/s

1.4.2. SF_6 flux through rising bubbles:

To estimate the amount of SF_6 that leaves the pit because of the hydrothermal bubbles, F SF_{6b}, is defined as:

$$F\ SF_{6b} = C\ SF_{6b}\ F_{air} \qquad \text{Eq. 2}$$

Where C SF_{6b} is the concentration of the SF_6 gas in the air bubbles and F_{air} the bubble flux, which is unknown. It was assumed that the average concentration of SF_6 is 2700 ppt_v as obtained from the bubbles analysis at the northern shore of the pit. Then:

$$C\ SF_{6b} = 2700\ 10^{-12}\ L_{SF6}/L_{air} \times 1\text{mol } SF_6 / 24.2\ L_{SF6} = 1.2\ 10^{-10} \text{ mol } SF_6 / L_{air}$$

Assuming a bubble flow of 10 l/s, a $F_{air} = 10$ l/s yields a value of F $SF_{6b} = 1.1\ 10^{-9}$ mol SF_6 /s. Since the F SF_{6b} is much less than F SF_6 through the air-water interface, the F SF_{6b} will be neglected.

Future experiments are planned to confirm the value of F_{air}.

1.4.3. Rate of change of SF_6 in Herman Pit:

By applying mass balance to the SF_6 gas tracer in the pit, one can write:

$$\delta C/\delta t = -(\alpha_s + \alpha_b + Q/V)\ C \qquad \text{Eq. 3}$$

where t is time, V is the pit volume, C is the mean concentration of the gas tracer, Q is water flow into and out of Herman Pit to Clear Lake, α is the first order loss term where subscript "s" means transfer across the air-water interface and "b" refers to the transfer through the rising bubbles. For purposes of this calculation it is assumed that the pit is well mixed, that the concentration in the outflow is the same as the concentration in the pit, and that the volume of inflow and outflow are equal. The solution of the equation can be written as:

$$C = C_i \exp(-\alpha_s - \alpha_b - Q/V)\ t \qquad \text{Eq. 4}$$

where C is the concentration of the gas in the pit and C_i is the initial concentration of the tracer. Since Flux of SF_6 through the bubbles has been neglected, $\alpha_b = 0$ d^{-1} and $\alpha_s = k/h = 0.024$ d^{-1}, where k is the transfer velocity of SF_6 and, h is the average depth of the pit.

Equation 4 expresses the fact that the decrease in the SF_6 concentration is due to a combination of the inflow/outflow rates of the pit, the transfer across the air-water interface and the presence of bubbles that could stripe the gas. Selecting from figure 4 the concentrations C = 55 pmol/l and C_i = 314pmol/l, changing in a time of 19 days, the calculation yields a flow value of 6.8 10^4 m^3/d (12400 gpm).

CONCLUSIONS

Preliminary calculations support the magnitude of the flow rate between the Herman Pit and Clear Lake obtained in previous experiments. A preliminary estimate suggests a value as high as 12,000 gpm. After analyzing the SF_6 concentration of the rising bubbles, it seems that the hydrothermal gas seeps are stripping the gas tracer but the net effect of this is negligibly small. Then, the SF_6 has three sinks, the hydraulic flow between the pit and the lake, the exchange across the air-water interface and gas stripping by the bubbles. SF_6 was found in all monitoring wells and in the lake shoreline close to the pit. No major information was obtained from the ^{3}He injection, in part because of the high gas background concentration probably supplied by the hydrothermal bubbles and, failure of the gas injection equipment.

A new dual gas injection will be conducted early September being Neon used instead of Helium. SF_6 will be used again.

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Possible Impacts of SO_4 and Acidity Discharges from Sulphur Bank Mine on the Clear Lake Ecosystem

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ABSTRACT

We have raised five cores raised from Clear Lake in order to test the hypothesis that increased erosion since the 1920s is responsible for nutrient increases that favored cyanobacteria (bluegreen algae). Heavy earthmoving machinery began to be employed on a large scale in the late 1920s and early 1930s, and observers of the lake noted a deterioration of water quality about the same time. As we have reported previously, several parameters in the cores are consistent with this "bulldozer" hypothesis. After about 1927, cores became markedly drier, higher in magnetic minerals, lower in organic matter, and lower in nitrogen than previously, as if increasing amounts of inorganic erosion products were diluting the organic matter deposited from the water column. However, careful calculations of dry matter deposition rates reveal a constant deposition post 1927 rate in one core and a lower deposition rate in the other four. This anomaly may result from the small sample of deposition rates or from misdating deeper horizons in the core. An alternative hypothesis is that some other change that began about 1927 is responsible for the changes in core properties. The changes in organic matter content etc. are closely correlated with a 10-fold increase in mercury in the cores. Open pit mining at Sulphur Bank began in 1927 and no doubt resulted in discharges of sulfate and acidity from the mine site, as well as mercury. The amounts are difficult to measure, but our preliminary estimates suggest that 50-90% of the current sulfate load to Clear Lake comes from Sulphur Bank Mine. Sulfate reduction is one of the most important microbial processes in Clear Lake sediments. A sulfate conveyor in which sulfide produced by sulfate reduction diffuses upward to be re-oxidized to sulfate seems necessary to account for gross rates of sulfate reduction measured by Erin Mack. A sulfate conveyor enhanced by increased sulfate loading could result in a more thorough oxidation of the sediments, and hence in the lower organic matter and nitrogen content of post 1927 sediments. Lower organic matter in turn may result in less water retention. Sulfide concentrations near the sediment surface probably regulate the iron cycle in Clear Lake. Under conditions of high sulfate, the less organic sediments will permit deeper penetration of sulfate and oxygen into the sediment surface and reduce the concentrations of sulfide that otherwise inhibit the release of soluble iron. Sediment redox and sulfur speciation likely affect the speciation and magnetic properties of iron minerals. Acidity from the mine will result in the selective dissolution of sediment minerals.

Although the mechanisms are as yet unclear, it is possible that sulfate and/or acidity loads from Sulphur Bank Mine played a major role in the deterioration of Clear Lake's water quality. The recent drought was a crude test of this hypothesis. The amount of sulfate and soluble iron decreased during the drought, bringing on the relatively clear water years beginning in 1991.

Revegetation of Sulphur Bank Mercury Mine

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ABSTRACT

Sulphur Bank Mercury Mine is an abandoned mine located on the eastern shore of the Oaks Arm of Clear Lake, Lake County, California. The California Department of Conservation (DOC) has been developing a revegetation strategy for Sulphur Bank Mine as part of the Remedial Design for this Superfund Site. Revegetation test plots were installed in two areas of the mine: on the Clear Lake shoreline tailings pile and on tailings in the northeastern portion of the mine. Phase I test plots examined soil treatments using waste processing lime and organic compost, alone and in combination, against a control; native species purchased from commercial suppliers were used to calibrate soil treatments. Phase II test plots were installed in 1997 using plants grown from seed collected on site. Phase II trials contrasted soil treatments similar to those in Phase I, with the addition of mycorrhizae, amendment depth, and weed mat treatments. Mycorrhizae were also added to treatments in Phase II. Phase I data confirm that use of waste lime + organic material results in greatest plant growth. Phase II trials showed similar results for the soil amendments and indicated that amending to depth was also beneficial. All other effects, mycorrhizae and weed mats, were not significant.

Keywords: Abandoned mine, revegetation, soil amendment, native plant species

INTRODUCTION

Sulphur Bank Mercury Mine is situated in the Oaks Arm of Clear Lake in the Clear Lake Basin, Lake County, California, in the northern Coast Range Geomorphic Province (Oakshott, 1978). The basin is dominated by Mt. Konocti, a dacite stratovolcano. The mine is at the intersection of three faults and shear zones at the center of a pipeline zone of hydrothermal alteration. Upwelling hot springs and fumaroles were observed during mining operations from 1865 to 1957 (e.g. Becker, 1888; Bradley, 1939) and can be observed at present. The mine encompasses approximately 150 acres, including about 120 acres of overburden and tailings and the 23-acre Herman Pit, known as the Herman Impoundment. The Herman Impoundment is approximately 90 feet deep, with vertical walls. Mine tailings descend from the surrounding slopes into the pit. Tailings extend into Clear Lake from the western portion of the mine. As a result of mining activities, mercury is significantly elevated in shallow sediments of the Oaks Arm of Clear Lake (Mining Waste Study Team, 1988; Columbia Geoscience, 1988). Random surface soil samples have revealed high levels of mercury and arsenic in tailings within the mine. Elevated mercury levels and the potential threat to human health have resulted in the classification of the Sulphur Bank Mine as a Superfund Site (ICF, 1992).

The U.S. Environmental Protection Agency (EPA) initiated remediation of the mine under its emergency authority in May 1992. At that time, unvegetated tailings lining the shore of Clear Lake were actively eroding into the lake. The EPA regraded the waste piles and placed a layer of imported topsoil on the tailings. The tailings were seeded with an erosion control seed mixture. The toe of the tailings pile at the lake shoreline was riprapped with large boulders to minimize wave run-up. Willow seedlings were also planted. To meet California mining waste management requirements, the EPA originally proposed to regrade mine wastes to a stable configuration; cap tailings and overburden with clean soil as necessary to support vegetation; and revegetate the site.

Working with the EPA and the California Department of Toxic Substances Control, the California Department of Conservation, Office of Mine Reclamation (OMR) developed a revegetation strategy for the mine. Principal objectives of the strategy are to conserve established vegetation, eliminate or reduce the amount of imported soil cover required, minimize potential mobilization of contaminated sediments or runoff into Clear Lake, and utilize a local source of plant propagules.

Vegetation at Sulphur Bank Mercury Mine is a complex mosaic due to human alteration of historic vegetation, site geology, and microclimatic differences resulting from slope and exposure. A diversity of native plants occurs at the mine, having established on tailings and overburden. Indicative of the revegetation potential of Sulphur Bank mine, these plants are adapted to the acidic, rocky or clay-rich soils of the site. The range of ecological tolerances observed in plants at Sulphur Bank Mercury Mine can indicate broad ecological amplitude within a given species or the evolution of specific acid-tolerant ecotypes. The vegetation can be placed generally in several series, the Interior Live Oak Series, Interior Live Oak Shrub Series, and the California Annual Grassland Series (Sawyer and Keeler-Wolf, 1995).

The principal vegetation series on undisturbed portions of the mine is the Blue Oak Series. Blue oak (*Quercus douglasii*) is co-dominant with one or more canopy associates: foothill pine (*Pinus sabiniana*), interior live oak (*Quercus wislizenii*), black oak (*Quercus kelloggii*), and a scrub oak (*Quercus* c.f. *berberidifolia*). The understory is variously dominated by shrubs or herbaceous species and grasses, including common manzanita (*Arctostaphylos manzanita*), yerba santa (*Eriodictyon californicum*), coyote brush (*Baccharis pilularis*), western redbud (*Cercis occidentalis*), toyon (*Heteromeles arbutifolia*), bush monkeyflower (*Mimulus aurantiacus*), blackberry (*Rubus discolor*), and blue wild-rye (*Elymus glaucus*). Small, open meadows of non-native annual grasses are also common.

The predominant series on mine tailings and overburden is the Interior Live Oak Shrub Series (or Scrub Oak Series), dominated by a shrub oak (*Quercus* c.f. *berberidifolia*). The Live Oak Series occurs throughout the mine and intergrades with other vegetation series. Common manzanita and toyon are often associate species. The California Annual Grassland Series is composed of native and non-native species. At the mine, this series is represented by meadows of non-native grasses interspersed in the blue oak-foothill pine association. It is the principal series on the shoreline pile. The grassland series is dominated by wild oats (*Avena* spp.), European hairgrass (*Aira caryophyllea*), zorro fescue (*Vulpia myuros*), soft chess (*Bromus hordeaceus*), mustards (*Brassica* spp.), and star thistles (*Centaurea melitensis* and others). Star thistle is not common at the mine site, even in areas of previous disturbance.

MATERIALS AND METHODS

Twenty-six soil samples from tailings and waste rock piles, as well as those from undisturbed areas (native soils) were analyzed for nutrient levels and soil pH. Soil sampling on Sulphur Bank Mine was done in October 1995, prior to Phase I test plot installation. Soils analyses identified several soil chemistry problems at the mine, including low soil pH, lack of soil organic material, and lack of plant available nutrients. Soil pH for baseline samples ranged primarily from pH 2.0 to pH 4.4, in contrast to native soils that average pH 6.3 (water pH). Three samples had pH levels of 6.4, 6.8, and 7.4. The pH of one sample, pH 5.7, approximated that of native soils. Revegetation test plots were installed in two areas of the mine: on the Clear Lake shoreline tailings pile (Shoreline) and on overburden in the northeast portion of the mine (Tailings). Selection of the two sites was based on Cluster Analysis and Principal Components Analysis (PCA). PCA grouped soils data into sites with similar soil characteristics, roughly corresponding to reference (native) soils, shoreline berm materials, overburden materials, and materials from the northern shoreline of the Herman Impoundment. "Native" and "shoreline berm" were grouped separately during all analyses. The remaining three groups, "tailings #1", "tailings #2, and "shore" were less distinct, reflecting the variable nature of most of the disturbed substrates. PCA was able to correctly identify substrates characteristic of the overburden materials from a range of locations around the mine site. About half of the samples from the shoreline berm had elevated magnesium indicating that the imported topsoil was derived from a serpenitic parent material. Importing a layer of serpentine soil (typically, high pH with unfavorable metal and calcium levels) on top of an acidic mine spoil exacerbates soil nutrient stresses for revegetation plantings.

We designed the test plots in two phases, with the objective of creating organic matter and nutrient and soil reaction levels suitable for growth of native species on the site. Test plots were installed in a completely randomized block design for statistical analysis. Because native species are growing on site, we attempted to create site conditions favorable to native plant establishment on relatively bare areas of the mine. Due to time constraints, scrub oak, foothill pine, toyon, and coyote brush used in the Phase I plots were purchased from commercial suppliers. Where possible, we obtained plants propagated from materials within the same general climatic and elevational range as the mine. Seed collected from deerweed *(Lotus scoparius)*, toyon, blue wild-rye, and scrub oak growing on the mine were also planted. Phase I test plots contrasted four different treatments: 1) waste lime only (L), 2) organic compost only (O), 3) waste lime + organic compost (L/O), and 4) no treatment (C - control). Organic amendments consisted of approximately 40,000 kg compost/ha of a composted municipal greenwaste material. The lime amendment was a waste processing lime from sugar beet production. This material was applied at the rate of 25 kg $CaCO_3$/1000 kg substrate on the shoreline berm and 6 kg $CaCO_3$/1000 kg substrate on the overburden materials, incorporated into the upper six inches of substrate. The pH of these substrates after 4 months had equilibrated at pH 5.8 on the overburden and pH 6.8 on the shoreline.

Phase II test plots, installed in March 1997, used plants grown from the site-collected seed. Phase II plants included western redbud, toyon, scrub oak, deerweed, foothill pine, bush monkeyflower, blue wild-rye, and squirreltail *(Elymus elymoides)*. Treatments examined under Phase II were 1) waste lime only (L), 2) waste lime + organic compost (L/O), and 3) waste lime + organic compost + mycorrhizae, and 4) waste lime + mycorrhizae. The test design also contrasted incorporation of lime at two depths, 30 cm and 60 cm, and use of weed mats. We amended soils to a pH of approximately 7.0. To reach this pH level, the amount of waste lime (calcium carbonate ($CaCO_3$) equivalent) required at the Shoreline was adjusted slightly from the Phase I rates to 23 kg $CACO_3$/1000 kg substrate. On the Tailings treatment area, 9 kg $CACO_3$/1000 kg substrate was incorporated into the upper six inches of substrate. Organic compost was again added at a rate of 40,000 kg/ha. Shortly after planting during Phase II, plants exhibited distinct P deficiency symptoms. We supplemented the previous amendments with a single application of ammonium phosphate ($NH_4H_2PO_4$) providing 25 kg nitrogen (N)/ha and 55 kg phosphorus (P)/ha (Claassen and Heeraman, 1998). Due to the lateness of installation, all plants were irrigated during the spring and early summer 1997. Watering ceased in June. Approximately 14 liters of water were applied to each plant every two weeks.

We collected baseline plant data (height + two perpendicular diameter measurements) when plants were first installed, followed by measurements during and after the growing season. We calculated the volume of the plants to determine plant growth: $\left[\frac{d_1 + d_2}{4}\right]^2 \cdot \pi \cdot ht.$

RESULTS

Phase I Data

Phase I data from the Tailings test plots 30 months (three growing seasons) after planting were highly significant among the soil amendments ($p<0.0001$) and among species ($p=0.0003$) (Table 1). The lime/organic treatment generated significantly greater plant growth than all other treatments ($p<0.05$) (Table 2). The lime treatment and the organic treatment were not significantly different from each other, and the control was significantly ($p<0.05$) poorer than all treatments. These results confirm results from the soil testing analyses suggesting that the substrates lack organic content and are in need of pH adjustment. Soil testing one year after installation of Tailings Phase I indicated that the pH values were slightly alkaline at the soil surface, and very acid at 30 cm depth, indicating that the incorporation of the lime amendment needs to be improved (mechanized). Among the species tested during Phase I, the increase in plant volume in *Pinus* was significantly greater than in all other species ($p<0.10$) (Table 3). *Baccharis* had the least amount of growth and the highest mortality ($p=0.10$).

On the Shoreline area, Phase I plant growth data from the test plots 30 months (three growing seasons) after planting were not significant among the soil amendments, but were significant among species ($p=0.0005$) (Table 4, Table 5). The only species that performed significantly better was *Heteromeles* (H), which had greater growth and survivorship ($p=0.10$) (Table 6). The pH on the Shoreline area was extremely variable and dramatically different with depth (i.e., alkaline at the surface and acid below). These data indicate that in order to grow any deep rooting species (possibly with the exception of *Heteromeles*), the substrate needs to be ripped and mixed (vertically) and then amended according to the ambient soil acidity.

During our September 1996 monitoring, we found that the Tailings had relatively low mortality when compared to plants installed in the shoreline plots. Many of the shoreline plants died during the summer. We feel that one possible reason for the mortality is due to the heavy infestation of weeds and depletion of available water.

Phase II Data

Phase II test plots were installed on the tailings pile and the shoreline berm, adjacent to the Phase I plots in March 1997. Amendment levels were adjusted slightly from those applied during Phase I. On the Tailings area, four different effects were tested: soil amendments (lime versus lime/organic), species, depth of amendment, and native mycorrhizal fungi. Based on the Phase II data collected 16 months (two growing seasons) after installation, the soil amendments and species effects were significant at the $p=0.0053$ and $p<0.00001$ level, respectively (Table 7). Neither the depth nor the mycorrhizae effects were significant. Lime/organic out performed the lime only amendment at $p=0.0053$. *Mimulus* outperformed all other species ($p=0.10$); and *Quercus* outperformed all others except *Mimulus* ($p=0.10$). No other differences among the seven species tested were significant.

On the Shoreline area, five different effects were tested: soil amendments (lime versus lime/organic), species, depth of amendment, native mycorrhizal fungi, and weed mats. Based on the data collected 16 months (two growing seasons) after installation, the soil amendments, species, and depth effects were significant at the $p=0.086$, $p<0.00001$, and $p=0.076$, respectively (Table 8). That is, lime/organic was significantly better than lime alone and this amendment had a significantly better effect if incorporated to depth rather than applied to the surface. Significantly greater growth was noted (at the $p=0.15$ level), with *Lotus* outperforming all other species, then *Elymus*, followed by *Mimulus*. None of the slower growing perennials were significantly different from each other. Neither the weed mats, nor the mycorrhizal fungi had a measurable effect on plant growth.

CONCLUSIONS

Based on the Phase I and Phase II data, it is feasible to revegetate Sulphur Bank Mercury Mine with indigenous species by amending soils with lime and organic matter. Use of site-indigenous plant materials is a priority since native species at the mine are adapted to substrate conditions at the mine and are performing well in field trials. The efficacy of our screening and amendment specification procedures is demonstrated in the vigorous growth of many of the selected plant materials. The improved growth on the highly acid shoreline berm demonstrates that lime incorporation deep into the soil provides growth benefits to plants, probably through greater water availability. This enhanced dewatering of the profile will also benefit control of percolation of the amendment. Future remediation should involve deeper and more homogeneous incorporation of the lime materials into the substrate, probably by specially designed equipment.

The benefits of organic and lime amendments in combination are also demonstrated in the revegetation trials. The range of plant growth responses demonstrates the ability of at least some of the tested species to establish and then colonize the amended substrates. Further monitoring will suggest species that can persist on the site even though they may be slow to establish initially. The short period of monitoring and the supplemental P amendment may have masked the potential growth benefits that are typical of mycorrhizal colonization. Moreover, mycorrhizae benefit plants growing under stress. The abundant rainfall during the winter of 1997 and spring of 1998 may have ameliorated stress on plants and masked beneficial mycorrhizal effects. Mycorrhizal fungi are sensitive to soil chemical conditions, so careful selection of appropriately adapted species is expected to be required. Plant species data need to be coupled with mycorrhizal colonization data, since plants in stressful environment are especially dependent on these symbiotic fungi.
Revegetation monitoring will continue as practicable and will include additional soils analyses, assessment of root development and penetration into the substrate, and evaluation of metal uptake into plant tissues.

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Table 1. Tailings Phase I - Analysis of Variance of Plant Volume after 30 Months

Source	Sum of Squares	Df	Mean Square	F-Ratio	P-Value
MAIN EFFECTS:					
A: Treatment	1.61242E10	3	5.37472E9	10.83	0.0000
B: Species	9.94005E9	3	3.31335E9	6.68	0.0003
RESIDUAL	6.94678E10	140	4.96199E8		
TOTAL (CORRECTED)	9.36157E10	146			

All F-ratios are based on the residual mean square error

Table 2. Tailings Phase I Multiple Range Tests of Plant Volume by Treatment after 30 Months

Method: 95.0 percent LSD

Treatment	Count	LS Mean	Homogeneous Groups
C	30	-481.848	X
L	35	12043.1	X
O	35	16868.7	X
L/O	47	28774.9	X

Contrast	Difference	+/- Limits
C–L	*-12524.9	10957.4
C–L/O	*-29256.7	10291.6
C–O	*-17350.5	10957.4
L–L/O	*-16731.8	9832.66
L–O	-4825.64	10527.6
L/O–O	*11906.1	9832.66

*denotes a statistically significant difference

Table 3. Tailings Phase I Multiple Range Tests of Plant Volume by Species after 30 Months

Method: 90.0 percent LSD

Species[1]	Count	LS Mean	Homogeneous Groups
B	29	4476.97	X
H	39	10306.8	XX
Q	42	14685.3	X
P	37	27735.7	X

Contrast	Difference	+/- Limits
B – H	-5829.83	9044.04
B – P	*-23258.7	9147.69
B – Q	*-10208.4	8905.22
H – P	*-17428.9	8464.73
H – Q	-4378.54	8202.1
P – Q	*13050.3	8316.25

*denotes a statistically significant difference

Table 4. Shoreline Phase I Analysis of Variance of Plant Volume after 30 Months

Source	Sum of Squares	Df	Mean Square	F-Ratio	P-Value
MAIN EFFECTS:					
A: Treatment	8.77249E8	3	2.92416E8	0.71	0.5517
B: Species	9.09397E9	3	3.03132E9	7.36	0.0005
RESIDUAL	1.64723E10	40	4.11807E8		
TOTAL (CORRECTED)	2.58528E10	46			

All F-ratios are based on the residual mean square error

Table 5. Shoreline Phase I Multiple Range Tests of Plant Volume by Treatment after 30 Months

Method: 95.0 percent LSD

Treatment	Count	LS Mean	Homogeneous Groups
L	2	-2644.46	X
O	10	8470.32	X
C	19	3474.7	X
L/O	16	16859.7	X

Contrast	Difference	+/- Limits
C – L	16119.2	30489.3
C – L/O	-3385.04	13916.4
C – O	5004.38	16023.3
L – L/O	-19504.2	30760.4
L – O	-11114.8	31769.2
L/O – O	8389.42	16533.2

*denotes a statistically significant difference

Table 6. Shoreline Phase I Multiple Range Tests for Plant Volume by Species after 30 Months

Method: 90.0 percent LSD

Treatment	Count	LS Mean	Homogeneous Groups
Q	11	-2454.5	X
P	11	298.873	X
B	5	8982.46	X
H	20	29333.5	X

Contrast	Difference	+/- Limits
B – H	*-20351.0	17085.3
B – P	8683.58	18430.2
B – Q	11437.0	18430.2
H – P	*29034.6	12826.9
H – Q	*31788.0	12826.9
P – Q	2753.37	14570.4

*denotes a statistically significant difference

Table 7. Tailings Phase II Analysis of Variance of Plant Volume after 16 Months

Source	Sum of Squares	Df	Mean Square	F-Ratio	P-Value
MAIN EFFECTS:					
A: Treatment	9.53081E10	1	9.53081E10	7.88	0.0053
B: Species	5.9988E12	6	9.99801E11	82.68	0.0000
C: Depth	1.92127E7	1	1.92127E7	0.00	0.9682
D: MZ	5.65082E9	1	5.65082E9	0.47	0.4947
RESIDUAL	4.23248E12	350	1.20928E10		
TOTAL (CORRECTED)	1.03358E13	359			

All F-ratios are based on the residual mean square error

Table 8. Shoreline Phase II Analysis of Variance for Plant Volume after 16 Months

Source	Sum of Squares	Df	Mean Square	F-Ratio	P-Value
MAIN EFFECTS:					
A: Treatment	3.49338E10	1	3.49338E10	2.969	0.0864
B: Species	5.61099E12	6	9.35164E11	79.13	0.0000
C: Depth	3.72974E10	1	3.72974E10	3.16	0.0765
D: Mat	1.24291E9	1	1.24291E9	0.11	0.7459
E. MZ	3.27502E9	1	3.27502E9	0.28	0.5989
RESIDUAL	4.19527E12	355	1.18177E10		
TOTAL (CORRECTED)	1.0017E13	365			

All F-ratios are based on the residual mean square error

[1]B = *Baccharis pilularis*
H = *Heteromeles arbutifolia*
P = *Pinus sabiniana*
Q = *Quercus berberidifolia*

APPENDICES

Appendix I: Symposium Agenda

CLEAR LAKE
SCIENCE AND MANAGEMENT SYMPOSIUM
SATURDAY, OCTOBER 24,1998
Lion's Community Hall
2495 Parallel Dr., Lakeport, CA

SYMPOSIUM AGENDA

7:30 SIGN-IN

8:00 WELCOME/INTRODUCTION
Dr. Thomas H. Suchanek
Director, UC Davis Clear Lake Environmental Research Center

TOPIC: CLEAR LAKE AND WATERSHED PROCESSES AND PROGRAMS

8:20 **Developing a Clear Lake Management Plan**
Morty Prisament, Lake County Department of Public Works

8:40 **The Clear Lake pollen record: A benchmark profile for California climate history.**
Dr. David Adam, California Academy of Sciences

9:00 **Ticks and Tick-Borne Diseases of the Clear Lake (California) Watershed**
Norm Anderson, Lake County Vector Control

9:20 **Habitat Partitioning Among Some Culicidae in Lake County, CA**
Dr. Art Colwell, Lake County Vector Control

9:40 **Clear Lake Hydrilla Eradication Project**
Nate Dechoretz, California Department of Food & Agriculture

10:00 - 10:30 BREAK AND POSTER SESSION

TOPIC: PHYSICAL AND CHEMICAL PROCESSES

10:30 **Micro-meteorological considerations, and a program to enhance the number and locations of meteorological monitoring stations within the Lake County air basin.**
Stephani Kling, Lake County Air Quality Management District

10:50 **Promoting Erosion Control Conservation Practices in the Clear Lake Basin**
Dr. Randall Thomas, Health and Environment

11:10 **Towards a 3-D model of Clear Lake Hydrodynamics: Observations and Numerical Simulations in the Oaks Arm**
Francisco Rueda, UC Davis Department of Civil and Environmental Engineering

11:30 **A Whole-Lake Sulfur Budget of Clear Lake, California**
Dr. Xiaoping Li, U.C. Davis Department of Environmental Science and Policy

TOPIC: CONTAMINANTS IN CLEAR LAKE AND SURROUNDING WATERSHEDS

11:50 **Mercury - An Immunotoxic Perspective**
Dr. Robert Speirs, Immunotoxicologist, Retired

12:10 **Atmospheric Mercury Concentrations in the Mayacmas and Knoxville Mining Districts of Lake County, CA**
Ross Kauper, Lake County Air Quality Managment District

12:30 - 1:30 LUNCH

1:30 **A Survey of Relative Mercury Bioavailability Throughout the Upper Cache Creek Watershed, Using Benthic Invertebrates**
Dr. Darell Slotton, UC Davis Department of Environmental Science and Policy

1:50 **Methyl mercury production at Clear Lake is decoupled from bulk inorganic mercury loading: biotic contamination is lower than expected.**
Dr. Thomas H. Suchanek, UC Davis Department of Environmental Science and Policy

2:10 **Contaminants in Clear Lake Grebes and Osprey: Past and Present Trends of Mercury and Organochlorides**
Dr. Dan Anderson, UC Davis Department of Wildlife, Fish and Conservation Biology

TOPIC: SULFUR BANK MERCURY MINE

2:30 **Overview of the Remedial Investigation/Feasibility Study Process at the Sulfur Bank Mercury Mine**
Ellen Manges, EPA, Region IX

2:50 **Hydrological Transport between the Sulfur Bank Mercury Mine and Clear Lake Using Gas Tracer**
Sonia Oton, UC Davis Department of Civil and Environmental Engineering

3:10 - 3:40 BREAK AND POSTER SESSION

3:40 **Geochemical Features of Water-Rock Interactions at the Sulfur Bank Mercury Mine, Lake County, CA**
Dr. Paul Lechler, Nevada Bureau of Mines and Geology

4:00 **Possible Impacts of Sulfate and Acidity Discharges from the Sulfur Bank Mercury Mine on the Clear Lake Ecosystem**
Dr. Pete Richerson, UC Davis Department of Environmental Science and Policy

4:20 **Revegetation of the Sulfur Bank Mercury Mine**
Mary Ann Showers, CA Department of Conservation, Office of Mine Reclamation

4:40 **Arsenic and Mercury Uptake as Influenced by Revegetation Treatment at the Sulfur Bank Mercury Mine**
Dr. Vic Claassen, UC Davis Department of Land, Air and Water Resources

5:00 WRAP UP AND QUESTIONS

POSTER SESSION

TOPIC: CLEAR LAKE WATERSHED PROCESES AND PROGRAMS

1. **The Clear Lake Events Timeline: A graphic display showing major events relevant to changes in Clear Lake quality**
   Laurent Meillier, UC Davis Clear Lake Environmental Research Center

2. **Exploratory Analysis of the Clear Lake Gnat, *Chaoborus astictopus***
   Loo Botsford, UC Davis Department of Wildlife, Fish and Conservation Biology

3. **Factors Affecting Habitat Segregation Among Treeehole Mosquitoes in Lake County, CA**
   Dave Woodward, Lake County Vector Control District

4. **Predicting Emergence of Hydrilla from Clear Lake Sediments**
   David Spencer, USDA-ARS Exotic & Invasive Weed Research Unit

TOPIC: CLEAR LAKE PHYSICAL AND CHEMICAL PROCESSES

5. **Comparison of Physico-Chemical Characteristics of Anderson Marsh and Rodman Slough Sediments, Clear Lake**
   Jae G. Kim, UC Davis Department of Environmental Science and Policy

TOPIC: CONTAMINANTS IN CLEAR LAKE AND SURROUNDING WATERSHEDS

6. **Stable Isotope Analysis of Trophic Structure as Indicators of Mercury Pathways in the Aquatic Ecosystem of Clear Lake, CA**
   Amanda Bern, UC Davis Clear Lake Environmental Research Center

7. **Effects of Multiple Stresssors (Mercury and Copper) on Zooplankton: Studies on Ceriodaphnia and Daphnia.**
   Eddie Gilmartin, UC Davis Department of Environmental Science and Policy, Coyote Creek Riparian Station

8. **Physiological Effects of Methylmercury in the Diet on Two Clear Lake Fishes**
   Ann Houk, U.C. Davis Department of Wildlife, Fish and Conservation Biology

9. **Mercury - Comments on its Dispersion, Bioaccumulation and Biocidal Effects**
   Dr. Robert Speirs, Immunotoxicologist, Retired

TOPIC: SULFUR BANK MERCURY MINE

10. **Development of and support for a theoretical model of the formation, microbial colonization, methylation dynamics and transport of flocculent material.**
    J.R. Flanders, UC Davis Clear Lake Environmental Research Center

Appendix II: Index of Authors