First Annual
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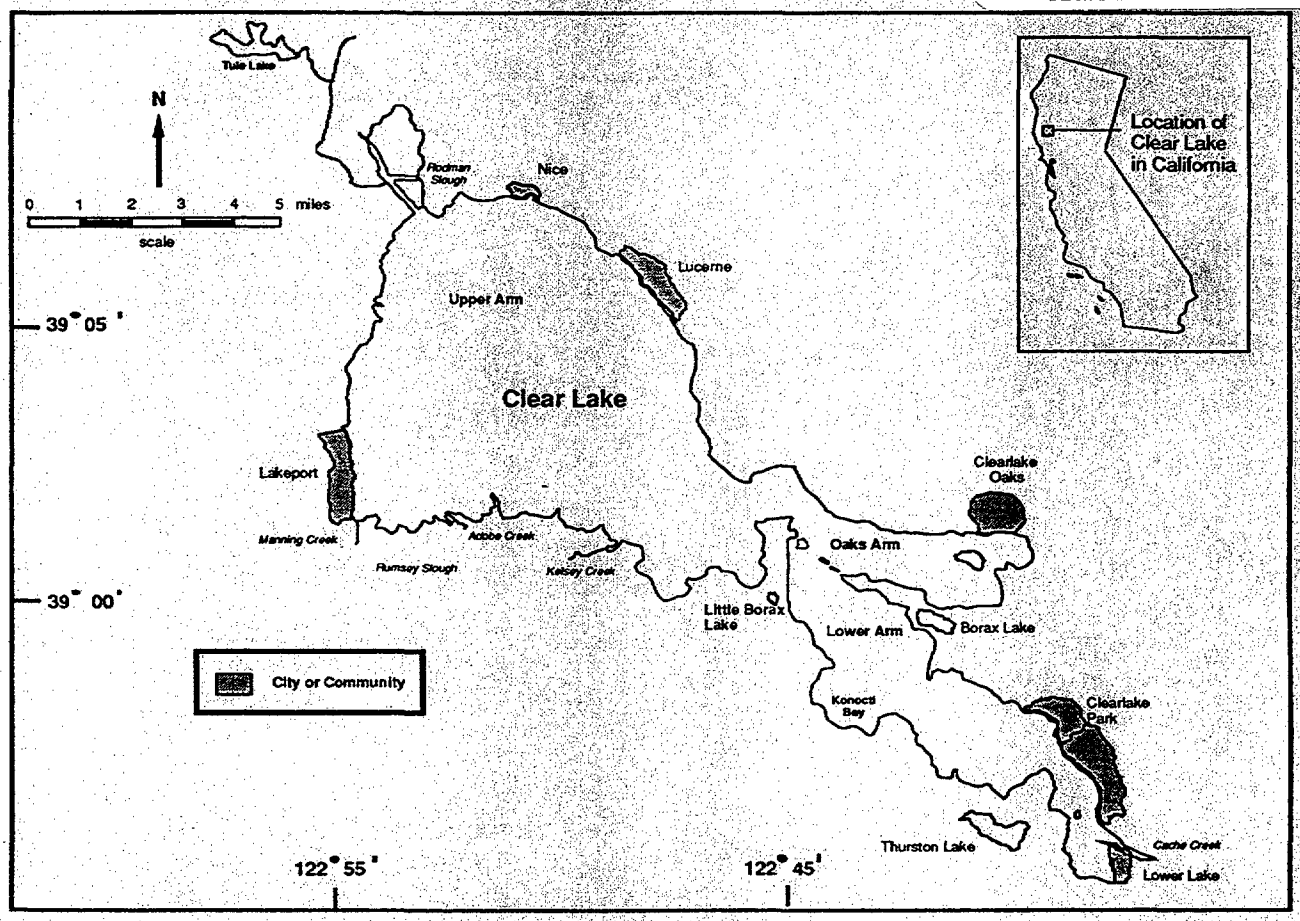

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FIRST ANNUAL
CLEAR LAKE SCIENCE AND MANAGEMENT SYMPOSIUM

September 13, 1997
Lakeport, CA

The U.C. Davis - Clear Environmental Lake Research Center hosted the First Annual Clear Lake Science and Management Symposium on Saturday, September 13, 1997. It was held at the Lions Club Community Hall, 2495 Parallel Drive, Lakeport, CA. The symposium opened at 8:00 am and concluded with a social hour at 5:45 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 pests, limnology, natural history, the Sulphur Bank Mercury Mine, and the Clear Lake watershed using posters and/or oral "platform" presentations.

This symposium and resulting proceedings were sponsored by the U.C. Davis Center for Ecological Health Research (USEPA Office of Exploratory Research Grant # R825433), the County of Lake, the Greater Lakeport Chamber of Commerce, the Clearlake Chamber of Commerce, the Lakeport Lions Club and Brunos Foods (Lakeport, CA). 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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PESTS

THE BLACK BITING GNAT (*CULICOIDES VARIIPENNIS*) AND WATER QUALITY AT BORAX LAKE , CALIFORNIA

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Abstract - Water quality at Borax Lake is characterized by high pH, alkalinity, conductivity and boron. Minimum and maximum water quality measurements coincide with periods of drought or above average rainfall, respectively. *Culicoides variipennis* immatures were found to tolerate wide ranges of water quality in the lake. Populations of many natural enemies of *C. variipennis* were reduced or eliminated during periods of poor water quality.

Key Words - Borax Lake, *Culicoides variipennis,* water quality

INTRODUCTION

The continuing public health nuisance of the Black biting gnat, *Culicoides variipennis* (Diptera: Ceratopogonidae), at Borax Lake, California has stimulated ecological research and necessitated various control strategies over the last two decades. The larval population center for this gnat is Borax Lake, an alkaline lake of about 50 ha. in area. Water quality parameters vary widely with typical late summer values: pH=10, boron=500 ppm, total alkalinity=9,000 mg/l, conductivity=20,000 micromhos/cm and total hardness=500 ppm. These factors limit the effectiveness of natural biological controls but allow for a prolific *Culicoides* population. Several unique sampling techniques have been employed to study the biology of *Culicoides* and some of its associated organisms. The macrophytes *Ruppia maritima* and *Distichlis spicata* dominate the littoral plant community with the epiphytic algae *Ctenocladus* adhering to much of the aquatic plant growth. During times of fair water quality and luxuriant plant growth a mosquito problem reciprocates with the *Culicoides* gnat nuisance. *Culicoides variipennis* is a biting pest of man and livestock and has been implicated as a vector of bluetongue viral disease in sheep and cattle, and epizootic hemorrhagic disease in deer [1]. Since traditional controls of the *Culicoides* larvae have been ineffective, alternative bio-rational strategies are being investigated. Several physical, biological and chemical management techniques have been applied at Borax Lake with varying degrees of success.

MATERIALS AND METHODS

Monitoring water quality and biota at Borax Lake is an ongoing project for the Lake County Vector Control District. Standard methods [2] were used to analyze water quality one to eight times annually from 1980 to 1997. Dissolved oxygen and water temperatures were measured at the lake with a Yellow Springs Instruments model 57 meter. Water samples were taken by dipping a glass container just beneath the surface. All samples were returned to the laboratory for analysis. The pH was determined with an Orion model 720A pH/ISE meter. Both phenolphthalein and total alkalinity were analyzed by titration with an alkalinity test kit by Ecologic Instruments. Electrical conductivity was ascertained with an Amber Science model 1061 specific conductance meter. A Turner Designs model 40 nephelometer was used to determine the turbidity of Borax Lake water. Total hardness was determined by titration using Hach reagents. Boron concentration was analyzed with a Hach DR/3000 spectrophotometer.

Various sampling devices have been used at Borax Lake to monitor the biota. Since the *Culicoides* larvae inhabit the shoreline area [3] a small core sampler [4] was employed to monitor the target species as well as other littoral invertebrates. The Ekman dredge has also been used to grab benthic samples at Borax Lake [3]. Dippers, plankton nets, and aquatic dip nets have all been utilized to sample aquatic organisms. At times when sufficient rainfall has freshened Borax Lake, mosquitofish (*Gambusia affinis*) have been planted and minnow traps have been used to sample them. Emerging adult *Culicoides* have been sampled with a sticky trap consisting of a polypropylene cup with the inner surfaces coated with Tanglefoot® [4]. Adult *Culicoides*, mosquitoes, and other insects have been monitored with New Jersey light traps, CDC light traps baited with CO_2, Blacklight traps, modified AFS sweepers and Fay traps baited with CO_2. All of the above methods were utilized to develop a species list of Borax Lake invertebrates.

RESULTS AND DISCUSSION

The results of Borax Lake water quality analyses from 1980 to 1997 are shown in Table 1. During this period minimum and maximum measurements for pH (8.87-10.05), total alkalinity (900-9,400 ppm), total hardness (100-1,600 ppm)

Table 1. Water quality sampling at Borax Lake, Lake County, California (1980-1997).

Date	pH	Phenol. alkalinity (mg/l)	Total alkalinity (mg/l)	Total hardness (ppm)	Conductivity (micromhos/cm)	Turbidity (NTU)	Disolved oxygen (ppm)	Temperature (degrees C)	Boron (ppm)
06/10/80	9.77	730	3,980	208	-	-	-	-	347
03/25/81	9.44	332	1,362	100	6,100	-	-	-	205
05/28/81	9.68	-	-	-	32,300	-	12.0	24.5	-
06/08/81	9.74	1,780	4,890	260	37,400	-	-	-	240
06/17/81	9.75	1,640	4,920	260	37,800	-	-	22.6	296
08/03/81	9.59	-	-	-	48,700	-	15.1	23.5	-
12/01/81	9.56	1,800	>3000	350	51,000	-	-	22.8	315
06/23/82	9.72	1,160	3,610	-	19,350	-	-	28.0	136
07/07/82	9.75	-	-	-	33,000	-	-	-	204
09/09/82	9.61	2,014	5,000	-	37,000	-	-	-	288
09/24/82	9.73	2,020	5,580	270	19,000	2.00	-	20.5	-
05/03/83	9.70	300	1,120	175	2,600	4.00	8.8	19.0	57
07/18/83	9.58	170	900	580	7,300	8.00	-	23.8	60
08/17/83	9.58	320	1,800	650	8,750	1.00	-	-	70
09/08/83	9.57	320	1,360	620	8,180	3.00	-	-	75
09/19/83	9.61	300	1,800	620	7,960	3.00	-	-	60
03/26/84	8.98	196	1,320	200	3,290	1.00	-	-	61
08/20/84	9.17	60	1,640	190	8,620	4.60	-	-	102
10/18/84	9.05	210	1,770	240	6,210	5.00	-	-	102
02/20/85	9.08	190	1,640	230	6,600	5.00	-	-	75
08/19/85	-	-	-	-	-	-	-	-	85
10/04/85	9.54	320	2,480	270	7,600	3.75	-	-	102
03/04/86	9.51	240	1,600	250	5,300	4.50	-	-	80
08/04/86	9.52	350	1,680	280	8,390	-	-	-	82
03/03/87	9.19	280	1,780	310	6,800	6.00	-	-	94
05/07/87	8.87	340	1,980	260	8,450	-	-	-	98
07/20/87	9.19	450	2,200	345	8,400	3.80	-	-	110
11/19/87	9.01	110	2,800	320	9,600	4.20	-	-	-
03/02/88	9.15	283	2,005	340	9,700	7.28	-	14.8	100
04/04/88	9.16	560	2,720	320	10,100	5.17	-	-	121
10/27/88	9.42	840	1,680	500	11,000	3.10	-	-	186
12/14/88	9.48	900	3,700	550	10,000	4.20	-	-	190
01/23/89	9.55	900	3,300	600	8,900	8.20	-	-	166
03/20/89	9.54	900	3,300	600	10,000	18.10	12.0	17.5	148
04/03/89	9.55	900	3,440	600	10,000	6.90	10.6	13.5	180
05/09/89	9.75	1,300	3,800	500	11,000	5.20	-	-	172
06/28/89	9.78	1,050	3,650	500	130,000	4.40	-	-	263
08/08/89	10.05	1,750	5,000	600	14,500	2.84	-	-	244
09/06/89	9.89	1,400	4,300	560	10,000	1.20	-	-	238
12/12/89	9.72	1,600	4,500	300	14,000	1.30	-	-	216
03/12/90	9.89	850	4,700	700	11,000	12.20	3.8	7.5	330
08/29/90	9.77	3,700	9,400	680	11,300	19.80	-	-	436
01/08/91	9.48	2,200	7,500	1,000	17,000	13.30	-	-	605
02/13/91	9.85	2,400	7,900	950	14,000	2.50	10.1	17.5	384
04/12/91	9.45	1,800	6,900	800	14,000	12.20	-	15.0	360
07/02/91	9.82	2,300	7,800	950	16,000	1.81	-	-	520
07/26/91	9.95	3,000	8,550	1,000	17,000	1.98	-	-	510
02/25/92	9.66	1,780	6,880	520	18,500	4.50	18.0	17.0	415
03/31/92	9.87	2,500	8,800	1,500	-	2.15	-	-	480
05/13/92	9.99	2,800	9,300	1,600	18,000	2.35	-	-	545
09/29/92	9.99	790	3,550	920	21,000	1.67	-	-	1,492
11/12/92	9.59	950	3,200	680	18,500	2.30	-	-	1,130
01/19/93	9.78	1,350	5,460	400	16,500	10.30	-	-	375
03/30/93	9.69	1,200	3,760	180	10,000	4.70	8.7	15.0	180
05/03/93	9.79	1,450	3,900	240	12,000	1.33			260
03/03/94	9.92	1,480	4,760	420	1,400	2.14	9.2	10.5	310
03/29/94	9.79	1,600	5,150	640	13,500	1.63	-	-	365
05/03/94	9.84	1,800	5,900	1,300	13,800	1.08	-	18.0	395
09/19/94	9.85	3,040	9,120	620	21,000	4.95	-	-	635
04/19/95	9.54	270	1,340	240	5,400	1.69	-	-	75
08/28/95	9.48	550	2,200	220	7,800	1.49	9.5	19.0	95
08/25/95	9.41	470	1,900	240	7,150	1.80	-	-	105
02/05/96	9.58	170	920	120	4,000	7.26	-	-	70
03/19/96	9.42	200	1,450	200	6,200	2.17	-	-	155
04/16/96	9.43	340	1,600	230	7,100	2.36	-	-	85
04/30/96	9.44	230	1,600	240	6,900	4.84	-	-	140
06/05/96	9.44	280	1,710	240	6,800	1.47	8.2	26.0	100
03/03/97	9.38	300	1,700	260	6,600	6.21	-	-	90
03/24/97	9.38	500	2,200	220	6,600	5.70	-	-	80
04/21/97	9.37	300	1,750	240	6,950	4.20	-	-	150
05/28/97	9.42	350	2,000	300	6,800	2.14	-	-	125
08/07/97	9.55	480	2,060	280	7,600	0.95	10.0	27.0	130

Table 2. Borax Lake total rainfall and water quality (April - August).

Years	Total rainfall	pH	Total alkalinity (mg/l)	Total hardness (ppm)	Conductivity (μmhos/cm)	Boron (ppm)
1979-1980	31.91	9.77	3,980	208	-	347
1980-1981	18.58	9.69	4,905	260	39,050	268
1981-1982	34.24	9.74	3,610	270	26,175	170
1982-1983	52.77	9.64	1,273	468	6,217	63
1983-1984	30.84	9.17	1,640	215	8,620	102
1984-1985	20.95	9.54	2,060	250	7,100	85
1985-1986	43.80	9.52	1,680	280	8,390	82
1986-1987	18.24	9.03	2,090	303	8,425	104
1987-1988	22.45	9.29	2,720	410	10,550	121
1988-1989	18.70	9.78	3,973	550	41,375	215
1989-1990	23.17	9.77	9,400	680	11,300	436
1990-1991	22.00	9.74	7,750	917	15,667	463
1991-1992	21.55	9.86	9,300	1,260	19,167	545
1992-1993	41.14	9.79	3,900	240	12,000	260
1993-1994	19.80	9.85	7,510	960	17,400	395
1994-1995	55.64	9.48	1,813	230	6,783	92
1995-1996	35.83	9.44	1,637	237	6,933	108
1996-1997	28.75	9.45	1,937	273	7,117	135
Averages	30.02	9.59	3,954	445	14,839	222

conductivity (1,400-130,000 micromhos/cm) and boron (57-1,492 ppm) were associated with periods of drought or above average rainfall, respectively. The Borax Lake basin holds drainage from a 6.5 km^2 watershed [5] and has no outlet. The data indicate that water quality is directly related to the amount of rain that accumulates in the basin. Annual rainfall and yearly averages for some water quality parameters are shown in Table 2. Buffering prevented extreme changes in the pH of Borax Lake water, but both total alkalinity and conductivity were negatively correlated ($P<0.05$) with rainfall (see Table 3). Since high rainfall or drought can affect lake level for more than one year, lake depth or volume would probably be more suitable measurements with which to correlate water quality parameters. Many of the water quality parameters tended to fluctuate together; for example, as alkalinity increased the concentration of boron also increased (Table 3).

Table 3. Matrix of Spearman's rank correlation coefficients (R) for annual rainfall and April through August water quality averages (1980-1997) at Borax Lake, CA.

	Rain	pH	Total alkalinity	Total hardness	Conductivity	Boron
Rain		-.132	-.550 *	-.443	-.562 *	-.428
pH			.727 **	.425	.492 *	.705 **
Total alkalinity				.517 *	.641 **	.924 **
Total hardness					.510 *	.432
Conductivity						.593 **

* $P \leq 0.05$
** $P \leq 0.01$

Table 1. Water quaility sampling at Borax Lake, Lake County, California (1980-1997).

Date	pH	Phenol. alkalinity (mg/l)	Total alkalinity (mg/l)	Total hardness (ppm)	Conductivity (micromhos/cm)	Turbidity (NTU)	Disolved oxygen (ppm)	Temperature (degrees C)	Boron (ppm)
06/10/80	9.77	730	3,980	208	-	-	-	-	347
03/25/81	9.44	332	1,362	100	6,100	-	-	-	205
05/28/81	9.68	-	-	-	32,300	-	12.0	24.5	-
06/08/81	9.74	1,780	4,890	260	37,400	-	-	-	240
06/17/81	9.75	1,640	4,920	260	37,800	-	-	22.6	296
08/03/81	9.59	-	-	-	48,700	-	15.1	23.5	-
12/01/81	9.56	1,800	>3000	350	51,000	-	-	22.8	315
06/23/82	9.72	1,160	3,610	-	19,350	-	-	28.0	136
07/07/82	9.75	-	-	-	33,000	-	-	-	204
09/09/82	9.61	2,014	5,000	-	37,000	-	-	-	288
09/24/82	9.73	2,020	5,580	270	19,000	2.00	-	20.5	-
05/03/83	9.70	300	1,120	175	2,600	4.00	8.8	19.0	57
07/18/83	9.58	170	900	580	7,300	8.00	-	23.8	60
08/17/83	9.58	320	1,800	650	8,750	1.00	-	-	70
09/08/83	9.57	320	1,360	620	8,180	3.00	-	-	75
09/19/83	9.61	300	1,800	620	7,960	3.00	-	-	60
03/26/84	8.98	196	1,320	200	3,290	1.00	-	-	61
08/20/84	9.17	60	1,640	190	8,620	4.60	-	-	102
10/18/84	9.05	210	1,770	240	6,210	5.00	-	-	102
02/20/85	9.08	190	1,640	230	6,600	5.00	-	-	75
08/19/85	-	-	-	-	-	-	-	-	85
10/04/85	9.54	320	2,480	270	7,600	3.75	-	-	102
03/04/86	9.51	240	1,600	250	5,300	4.50	-	-	80
08/04/86	9.52	350	1,680	280	8,390	-	-	-	82
03/03/87	9.19	280	1,780	310	6,800	6.00	-	-	94
05/07/87	8.87	340	1,980	260	8,450	-	-	-	98
07/20/87	9.19	450	2,200	345	8,400	3.80	-	-	110
11/19/87	9.01	110	2,800	320	9,600	4.20	-	-	-
03/02/88	9.15	283	2,005	340	9,700	7.28	-	14.8	100
04/04/88	9.16	560	2,720	320	10,100	5.17	-	-	121
10/27/88	9.42	840	1,680	500	11,000	3.10	-	-	186
12/14/88	9.48	900	3,700	550	10,000	4.20	-	-	190
01/23/89	9.55	900	3,300	600	8,900	8.20	-	-	166
03/20/89	9.54	900	3,300	600	10,000	18.10	12.0	17.5	148
04/03/89	9.55	900	3,440	600	10,000	6.90	10.6	13.5	180
05/09/89	9.75	1,300	3,800	500	11,000	5.20	-	-	172
06/28/89	9.78	1,050	3,650	500	130,000	4.40	-	-	263
08/08/89	10.05	1,750	5,000	600	14,500	2.84	-	-	244
09/06/89	9.89	1,400	4,300	560	10,000	1.20	-	-	238
12/12/89	9.72	1,600	4,500	300	14,000	1.30	-	-	216
03/12/90	9.89	850	4,700	700	11,000	12.20	3.8	7.5	330
08/29/90	9.77	3,700	9,400	680	11,300	19.80	-	-	436
01/08/91	9.48	2,200	7,500	1,000	17,000	13.30	-	-	605
02/13/91	9.85	2,400	7,900	950	14,000	2.50	10.1	17.5	384
04/12/91	9.45	1,800	6,900	800	14,000	12.20	-	15.0	360
07/02/91	9.82	2,300	7,800	950	16,000	1.81	-	-	520
07/26/91	9.95	3,000	8,550	1,000	17,000	1.98	-	-	510
02/25/92	9.66	1,780	6,880	520	18,500	4.50	18.0	17.0	415
03/31/92	9.87	2,500	8,800	1,500	-	2.15	-	-	480
05/13/92	9.99	2,800	9,300	1,600	18,000	2.35	-	-	545
09/29/92	9.99	790	3,550	920	21,000	1.67	-	-	1,492
11/12/92	9.59	950	3,200	680	18,500	2.30	-	-	1,130
01/19/93	9.78	1,350	5,460	400	16,500	10.30	-	-	375
03/30/93	9.69	1,200	3,760	180	10,000	4.70	8.7	15.0	180
05/03/93	9.79	1,450	3,900	240	12,000	1.33			260
03/03/94	9.92	1,480	4,760	420	1,400	2.14	9.2	10.5	310
03/29/94	9.79	1,600	5,150	640	13,500	1.63	-	-	365
05/03/94	9.84	1,800	5,900	1,300	13,800	1.08	-	18.0	395
09/19/94	9.85	3,040	9,120	620	21,000	4.95	-	-	635
04/19/95	9.54	270	1,340	240	5,400	1.69	-	-	75
08/28/95	9.48	550	2,200	220	7,800	1.49	9.5	19.0	95
08/25/95	9.41	470	1,900	240	7,150	1.80	-	-	105
02/05/96	9.58	170	920	120	4,000	7.26	-	-	70
03/19/96	9.42	200	1,450	200	6,200	2.17	-	-	155
04/16/96	9.43	340	1,600	230	7,100	2.36	-	-	85
04/30/96	9.44	230	1,600	240	6,900	4.84	-	-	140
06/05/96	9.44	280	1,710	240	6,800	1.47	8.2	26.0	100
03/03/97	9.38	300	1,700	260	6,600	6.21	-	-	90
03/24/97	9.38	500	2,200	220	6,600	5.70	-	-	80
04/21/97	9.37	300	1,750	240	6,950	4.20	-	-	150
05/28/97	9.42	350	2,000	300	6,800	2.14	-	-	125
08/07/97	9.55	480	2,060	280	7,600	0.95	10.0	27.0	130

More than 30 species of invertebrates have been collected at Borax Lake (Table 4). The most abundant insect in the lake is usually *C. variipennis.* Shoreline mud sampling has detected *C. variipennis* densities in excess of 1,000,000 larvae per m^2 of shoreline. *Culicoides* immatures are euryhaline and highly tolerant of poor water quality conditions [3]. During droughts water quality becomes poor, so planted fish and invertebrate predators and competitors of *Culicoides* die out,

Table 4. Invertebrates collected from Borax Lake.

Borax Lake invertebrates	Months collected
Culicoides variipennis	Jan-Dec
Bezzia glabra	Jul-Aug
Dasyhelea sp.	Jan-Dec
Culex tarsalis	Apr-May
Anopheles franciscanus	Apr-Oct
Culiseta incidens	Mar
Culiseta inornata	Mar
Cricotopus sylvestris	Apr-Sept
Tanypus sp.	Apr-Aug
Notonecta shooteri	Nov
Notonecta unifasciata	Mar-Jun
Corisella inscripta	Mar-Nov
Buenoa scimitra	Jan-Dec
Saldula pallipes	Mar-May
Gerris incognitus	Mar
Hydropyrus hians	Jan-Dec
Tabanus punctifer	Jun-Oct
Eulalia tumida	Apr-Jun
Eristalinae	Mar
Psychoda sp.	Apr-Jun
Enallagma carunculatum	Jan-Dec
Heteroceridae	May
Enochrus conjunctus	Apr-Jun
Agabus sp.	Mar
Hygrotus sp.	Mar
Ochthebius rectus	Feb-Mar
Hypogastruridae	Feb
Ostracoda	May
Moina hutchinsoni	Mar-Nov
Bosmina sp.	Mar
Cyclops sp.	Mar-Jun
Diaptomus sicilis	Mar-Nov
Hexarthra sp.	Apr-Nov
Lepadella sp.	May-Jun

allowing for a profuse Black biting gnat population. Copious rainfall dilutes Borax Lake salts and allows natural enemies of the *Culicoides* to flourish. During times of abundant rainfall the aquatic macrophytes increase in the littoral zone providing numerous insect species, including the mosquitoes *Culex tarsalis* and *Anopheles franciscanus,* with suitable larval habitat. When water quality improves, Borax Lake supports a variety of insects, but other taxa of organisms common in a fresh water lake such as Clear Lake remain scarce or absent (see Figure 1). The most effective control strategy currently available for *Culicoides* involves the use of a pyrethrin larvicide applied in a band around the perimeter of the lake [6]. A physical technique of utilizing a thermal sled to control larvae and pupae of the Black biting gnat has been investigated. Dipteran

pathogens tested have had poor efficacy against *C. variipennis* in Borax Lake water. In addition to utilizing *Gambusia affinis* when the water quality permits their survival, several other fish species have been investigated [3]. The search continues for an alkaline and boron tolerant biological control agent for use against *Culicoides variipennis* in Borax Lake.

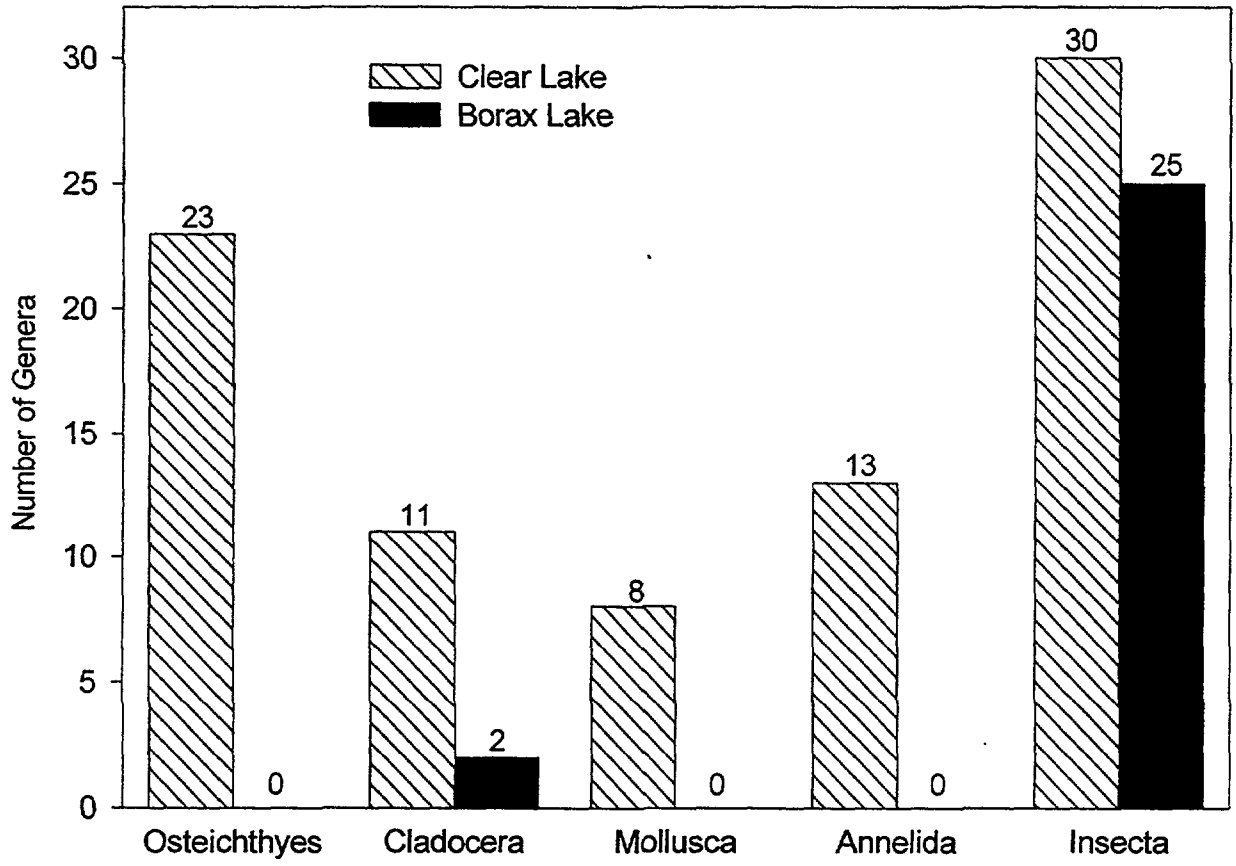

Figure 1. Comparison of genera diversity in Borax Lake and Clear Lake.

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Vector-Borne Disease Surveillance in Lake County, California. (P)

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Key Words: virus, bacteria, vector-borne disease surveillance

Some of the vector-borne diseases that are monitored in Lake County are Western Equine Encephalitis, St. Louis Encephalitis, Turlock Virus, Bluetongue, Epizootic Hemorrhagic Disease, Lyme Disease, Ehrlichiosis, Babesiosis, Malaria, Plague, Hantavirus, and schistosome dermatitis. In order to survey for WEE and SLE two flocks of sentinel chickens are maintained and mosquitoes are trapped, identified, sexed, and grouped into pools. The chickens are bled every two weeks from May to October. The blood samples and pooled mosquitoes are sent to the California Department of Health Services, Viral and Rickettsial Disease Laboratory for analysis. In 1988 a virulent outbreak of EHD in local deer populations stimulated vector surveillance studies and control of Culicoides gnats emanating from Borax Lake. To monitor tick-borne disease agents, ticks are collected and sent to various laboratories for analysis. In addition, cooperative tick and Lyme disease studies were undertaken with Dr. Robert Kimsey of the University of California, Davis. Any active cases of malaria are reported to the LCVCD so that mosquito control measures can be reviewed. Fleas are collected for plague surveillance by the Vector-Borne Disease Section (VBDS) of the California Department of Health Services. A hantavirus survey was conducted in 1996 in cooperation with the VBDS. Swimmer's itch complaints in Lake County are followed up by collection of snails from the reported areas and isolation of cercariae from the snails. In most of these endeavors the LCVCD attempts to maintain an early detection program for vector-borne diseases so that preemptive control strategies can be applied.

MONITORING OF DIPTERAN PESTS AND ASSOCIATED ORGANISMS IN CLEAR LAKE (CALIFORNIA)

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Abstract - The methods for monitoring dipteran pests and associated organisms in Clear Lake have involved the use of Ekman dredges, plankton nets, larval dip samplers, light traps, carbon dioxide baited suction traps, seines, and surface trawls. Important pest species identified include *Chaoborus astictopus, Chironomus plumosus, Culex tarsalis,* and *Aedes increpitus.* The Ekman benthic samples collected from 1954 to 1997 constitute the longest continuous monthly biological dataset from Clear Lake. New introductions of pests, diseases, fish, aquatic macrophytes or other organisms may necessitate changes or additions to Clear Lake sampling programs.

Key Words - Chaoboridae, Chironomidae, Culicidae, benthos, plankton, ichthyofauna

INTRODUCTION

Some of the most economically important nuisance pests, biting pests, and potential vectors of human disease in Lake County are dipteran insects. Twenty species of mosquitoes have been found in Lake County, but only *Culex tarsalis* (which can transmit encephalitis virus), *Culex stigmatosoma, Aedes increpitus* and *Anopheles franciscanus* have attained significant larval densities in Clear Lake.

Eighteen species of ceratopogonid midges have been collected from Lake County. *Culicoides variipennis,* the fiercely-biting vector of Bluetongue disease (in sheep) and Epizootic Hemorrhagic disease (in deer) , reaches tremendous numbers in Borax Lake [1], but less than a mile away this species is rare in Clear Lake.

Thirty three species of chironomid midges have been collected from Lake County. Some chironomids (e.g., *Tanytarsus* and *Cricotopus*) in Clear Lake are associated with aquatic macrophytes. However, the chironomids which usually cause the greatest annoyance have benthic larvae (e.g., *Chironomus plumosus, Chironomus decorus*). One of the adaptations which permits these larvae to store and retain oxygen for use during periods of anoxia is hemoglobin. Hemoglobin retained in the adult *Chironomus* is largely responsible for the allergic reactions (e.g., rhinitis, asthma, dermatitis, conjunctivitis) some persons exhibit following exposure to midges [2]. Chironomid egg masses deposited on buildings, boats or vehicles can cause defacement problems [3]. The nuisance and economic impacts of chironomid midges in North America have been reviewed by Ali [4].

Chironomid larvae can function as bioturbators when they move deep (sometimes >20 cm) lakebottom sediments to the mud-water interface while they are constructing tubes. The importance of chironomids in releasing nitrogen or phosphorus from sediments [5] or in transferring mercury to other trophic levels of Clear Lake [6] may warrant further research.

Chaoborus astictopus is a dipteran which historically has been so abundant at Clear Lake that it has the official common name of Clear Lake gnat [7]. This pest species at times has been considered a major deterrent to tourism and the overall economic development of Lake County [8, 9]. The early insecticide treatments have been extensively chronicled [e.g., 10, 11]. A biological control effort to reduce the Clear Lake gnat problem was initiated when Dr. S. F. Cook introduced a fish, the inland silverside (*Menidia beryllina*), into Clear Lake in 1967 [12]. Thereafter efforts to monitor the distribution and abundance of these (and other) fishes and to determine their feeding preferences in Clear Lake were initiated. Programs to monitor other biological, chemical and physical factors which could influence culicid, chaoborid or chironomid abundances in Clear Lake have also been established. A primary purpose for this paper is to present these data so they are available to other interested agencies or researchers.

MATERIALS AND METHODS

Adult Dipterans. Adult dipterans were collected with New Jersey light traps, CDC light traps, carbon dioxide baited Fay traps, and blacklight traps from 1976 to 1997 to compile a species list of medically-important dipterans. For routine surveillance, a standard New Jersey light trap with a 25 watt incandescent bulb and suction fan was operated at the LCVCD pier in Lakeport. All insects collected were removed once per week during 1996. Adult dipterans were identified [13, 14] and counted.

Benthos. Ekman grab samples were collected each month from the Upper Arm (n=28), the Oaks Arm (n=12), and the Lower Arm (n=18) of Clear Lake at specified stations [15]. Each 232 cm^2 sample of lakebottom sediment was sieved and processed as per Colwell and Schaefer [16]. Immature chironomids and chaoborids from these stations were monitored from 1954-1997. Chironomid subfamilies and leeches were counted each month in the Upper Arm samples from 1994-1997.

The above procedures were observed during a benthic transect across the Upper Arm of Clear Lake. Two Ekman samples were collected at a water depth of 0.5 m at Lakeport and at 1.0 m depth intervals to the center of the Upper Arm and then to the shallow water areas of Lucerne in 1977 and in 1994.

Plankton. One vertical plankton net (30 cm mouth diameter, 80 μ apertures) tow was taken at a deep water station in each of the three arms of Clear Lake. Tows were collected monthly from 1988 to 1997. Each sample was preserved in a 5% formaldehyde solution containing rose bengal dye and processed in the laboratory [17].

Water Quality. A vertical profile of the dissolved oxygen concentrations and temperatures of Clear Lake was determined (with a YSI meter) at one specified Ekman station in each of the three arms of Clear Lake. Secchi depths were measured with a standard 20 cm diameter black and white Secchi disk. Water samples were collected 0.1 m below the water surface and 1.0 m above the lakebottom with a 0.65 liter Kemmerer sampler at one station in each of the three arms of Clear Lake. These samples were collected once per month from 1988-1997. Standard methods [18, 19] were utilized to determine total hardness, pH and turbidity.

Seines. A seine (9.1 m by 1.2 m) with an ace mesh (3.2 mm apertures) was utilized to sample 70 m^2 areas of the Clear Lake shoreline. Seine hauls were collected at stations on the Upper Arm (n=18), Oaks Arm (n=6), and Lower Arm (n=9) from one to ten times per year from 1985 to 1997. All of the fish were counted in the field. A maximum of ten fish of each species per station were transported to the lab for stomach content analyses as per Vondracek [20].

Surface Trawls. A fixed-frame trawl with a 1.0 x 1.5 m mouth opening and graduated mesh sizes was towed for 300 m as per Wurtsbaugh [21] at each of 10 stations across the Upper Arm of Clear Lake. This transect was completed from one to three times per year (during summer and early fall) from 1988 to 1997.

Gut Content Analyses. All organisms in the crop of each *Chaoborus* dissected (n=86) were identified and counted as per Starrett [22]. The Tanypodinae do not have crops, so the contents of the entire digestive tract were analyzed [23]. The gut contents of the silversides (n=150) were analyzed according to Vondracek [20].

RESULTS AND DISCUSSION

Adult Dipterans. The LCVCD species list of nematocerous flies in three medically-important families appears in Table 1. Chaoborid and chironomid emergence patterns and adult abundances vary from year to year. Examples of the data collected

Table 1. Dipterans of the medically-important families Culicidae, Ceratopogonidae and Chironomidae collected from Lake County.

Culicidae	Ceratopogonidae	Chironomidae
Aedes bicristatus	*Atrichopogon fusculatus*	*Ablabesmyia* sp.
Aedes increpitus	*Bezzia setulosa*	*Apedilum subcinctum*
Aedes melanimon	*Bezzia* sp.	*Chironomus annularius*
Aedes nigromaculis	*Culicoides cavaticus*	*Chironomus decorus*
Aedes sierrensis	*Culicoides cockerellii*	*Chironomus frommeri*
Aedes vexans	*Culicoides crepuscularis*	*Chironomus plumosus*
Anopheles franciscanus	*Culicoides freeborni*	*Chironomus* sp.
Anopheles freeborni	*Culicoides neofagineus*	*Cladopelma* sp.
Anopheles punctipennis	*Culicoides posoensis*	*Cricotopus sylvestris*
Culex apicalis	*Culicoides utahensis*	*Cryptotendipes darbyi*
Culex boharti	*Culicoides variipennis occidentalis*	*Cryptochironomus fulvus*
Culex erythrothorax	*Culicoides variipennis sonorensis*	*Cryptochironomus* sp.
Culex pipiens	*Dasyhelea mutabilis*	*Dicrotendipes californicus*
Culex stigmatosoma	*Dasyhelea pritchardi*	*Dicrotendipes nervosus*
Culex tarsalis	*Dasyhelea* sp.	*Endochironomus nigricans*
Culiseta incidens	*Forcipomyia bipunctata*	*Glyptotendipes brachialis*
Culiseta inornata	*Letoconops torrens*	*Glyptotendipes lobiferus*
Culiseta particeps	*Palpomyia* sp.	*Harnischia* sp.
Coquillettidia perturbans		*Limnophyes hamiltoni*
Orthopodomyia signifera		*Nilotanypus* sp.
		Parachironomus abortivus
		Parachironomus monochromus
		Parachironomus tenuicaudatus
		Paratanytarsus sp.
		Polypedilum pedatum excelsius
		Procladius bellus
		Procladius denticulatus
		Procladius freemani
		Procladius sublettei
		Psectrocladius flavus
		Psectrocladius sp.
		Tanypus carinatus
		Tanytarsus sp.

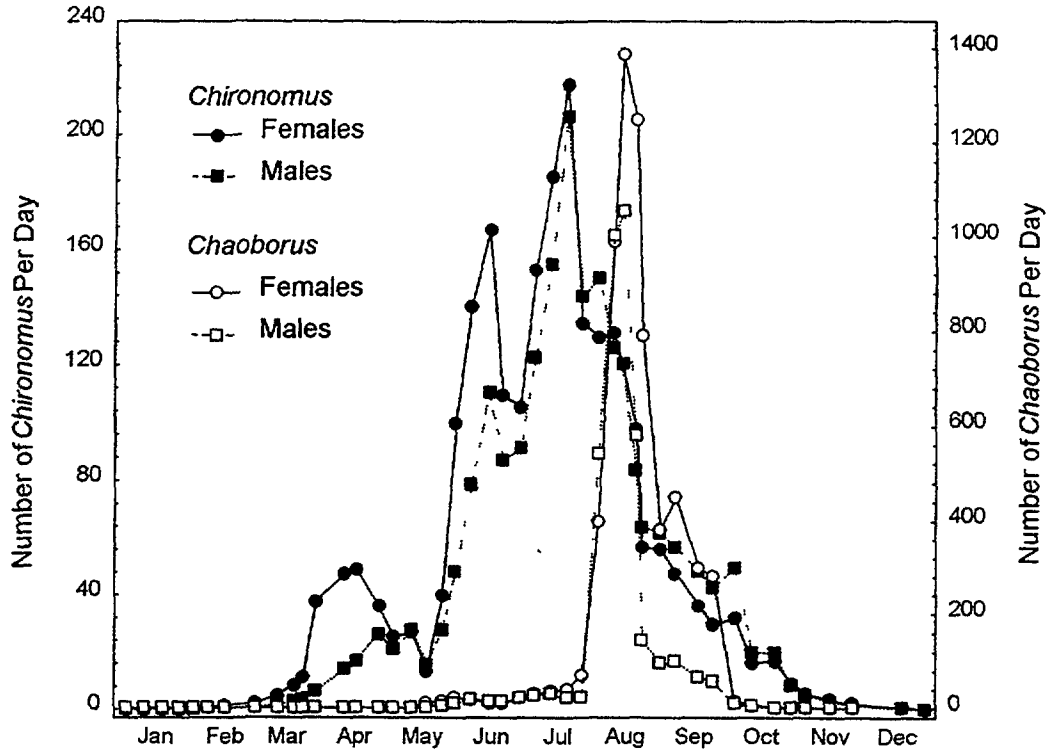

Figure 1 Mean number of adult *Chironomus* and *Chaoborus* collected per trap day by a New Jersey Light Trap located near the shoreline of Clear Lake at Lakeport in 1996.

appear in Figure 1. Larval *Chironomus* do not diapause and typically start emerging earlier in the year than the *Chaoborus* Diapause of the overwintering fourth instar *Chaoborus* larvae prevents emergence before May

Benthos Data for the immature stages of some dipteran pests appear in Figures 2-3. Some chironomids in the subfamily Tanypodinae (e.g, *Procladius*) can be predatory [24]. Laboratory tests have shown that some Clear Lake leeches (*Placobdella translucens*) can kill larvae of chironomid midges and Clear Lake gnats. Therefore monitoring of the benthic predators commenced in 1994 (Figure 4) but the data to date have not shown that the densities of Tanypodinae or Hirudinea are correlated with the densities of non-predatory pestiferous chironomids (Chironominae) or with the *Chaoborus* densities.

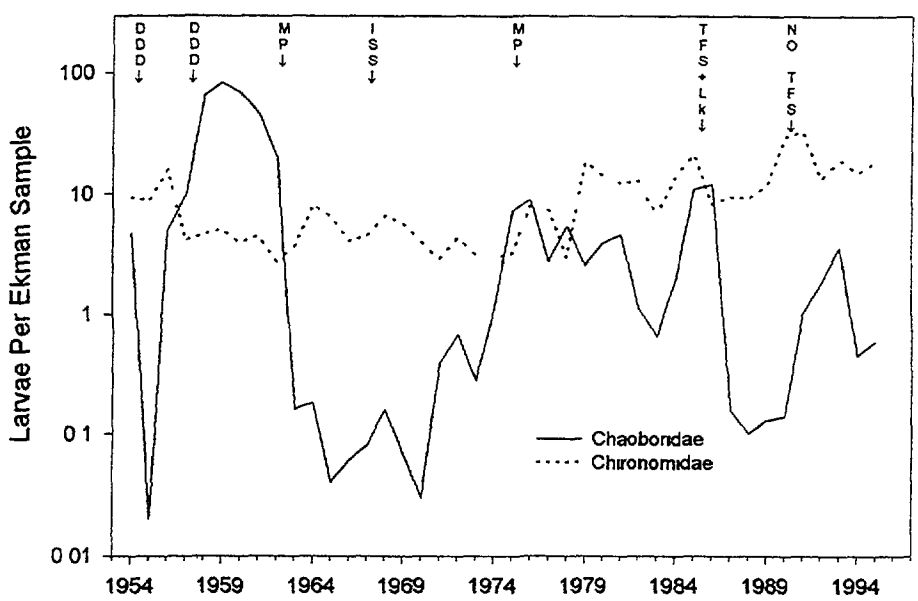

Figure 2 Mean number of immature Chironomidae and Chaoboridae collected in 696 Ekman samples per year DDD treatments occurred in 1954 and 1957. Methyl parathion treatments occurred from 1962-1975. Inland silversides were introduced in 1967. Threadfin shad and *Leptodora kindtii* first appeared in 1985 samples.

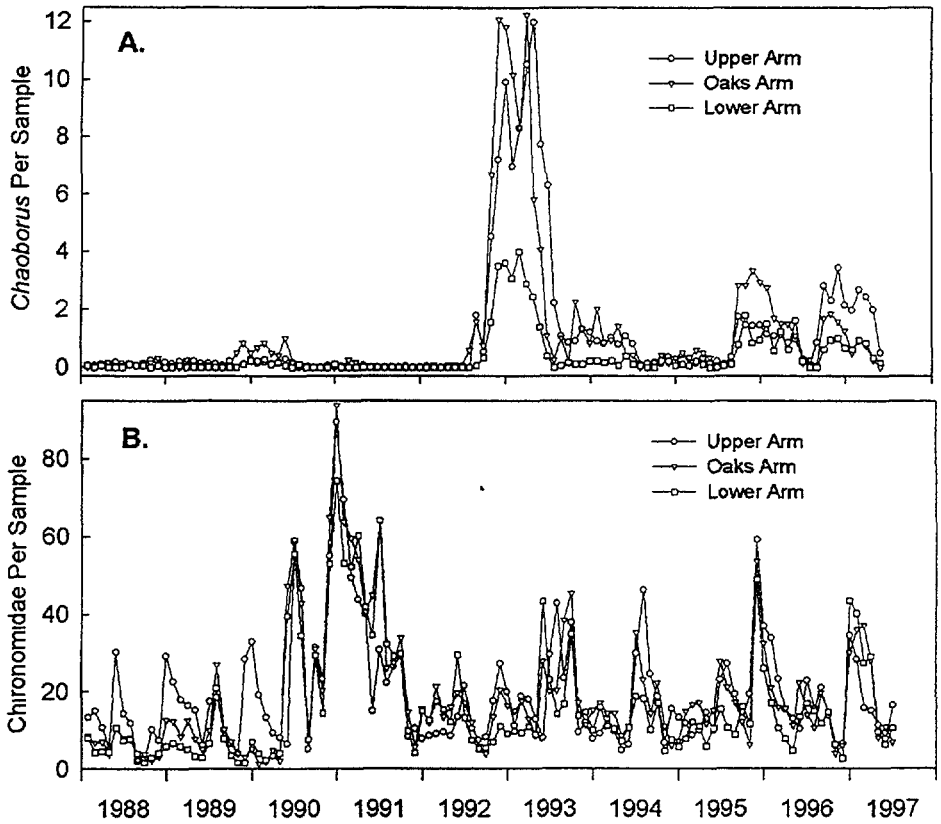

Figure 3. (A) Mean number of immature *Chaoborus* collected in 58 Ekman grab samples per month from Clear Lake (B) Mean number of immature Chironomidae per Ekman sample from Clear during the years 1988-1997

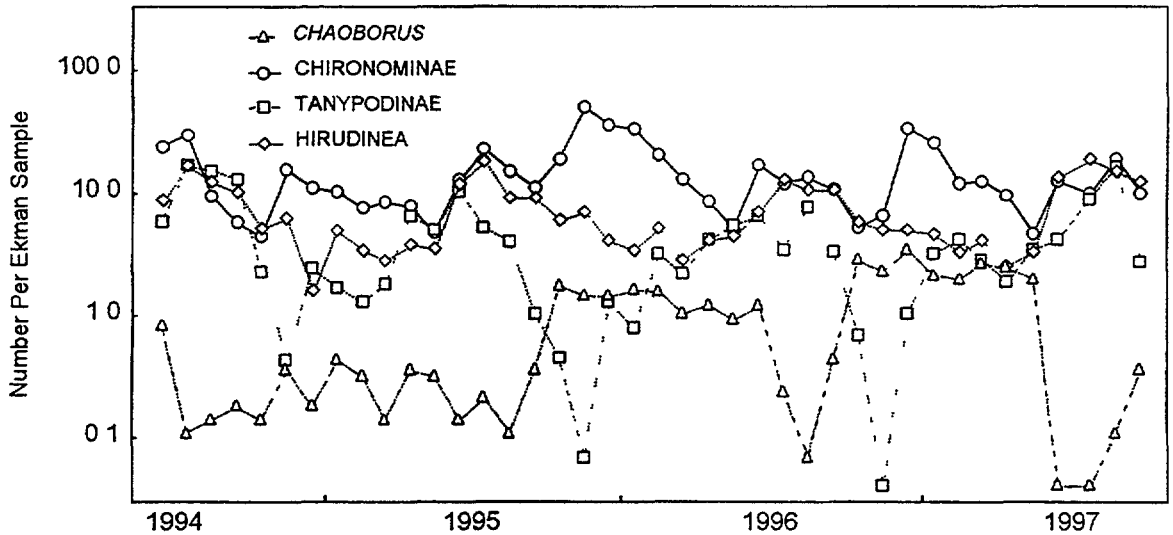

Figure 4. Mean number of Tanypodinae, Hirudinea, Chironominae and *Chaoborus* per Ekman sample collected from the Upper Arm of Clear Lake

The benthic transects indicated larvae of the pestiferous Chironominae were abundant in near shore areas The Chaoborid larvae typically occurred in offshore (deep water) areas (Table 2)

Plankton *Leptodora* was recorded for the first time in Clear Lake in 1985 [15], but has not become abundant Copepods (Figure 5) and rotifers (Figures 5 and 6) were abundant in Clear Lake in 1990 Some cladoceran (water flea) data appear in Figure 7.

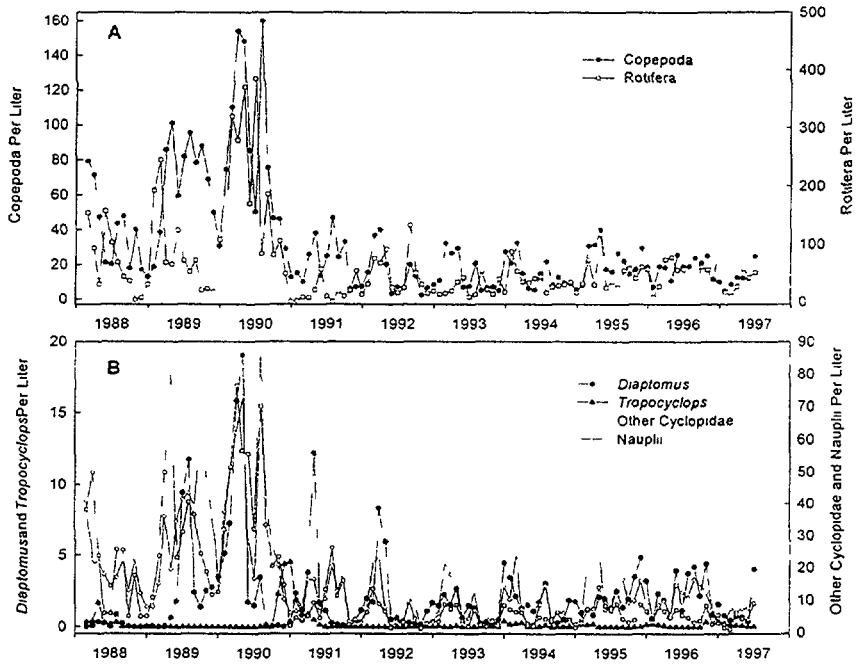

Figure 5. (A) Mean number of copepods and rotifers per liter collected in vertical plankton net tows from Clear Lake. (B) Mean number of copepods per liter. The other Cyclopidae were primarily *Acanthocyclops* and *Mesocyclops*

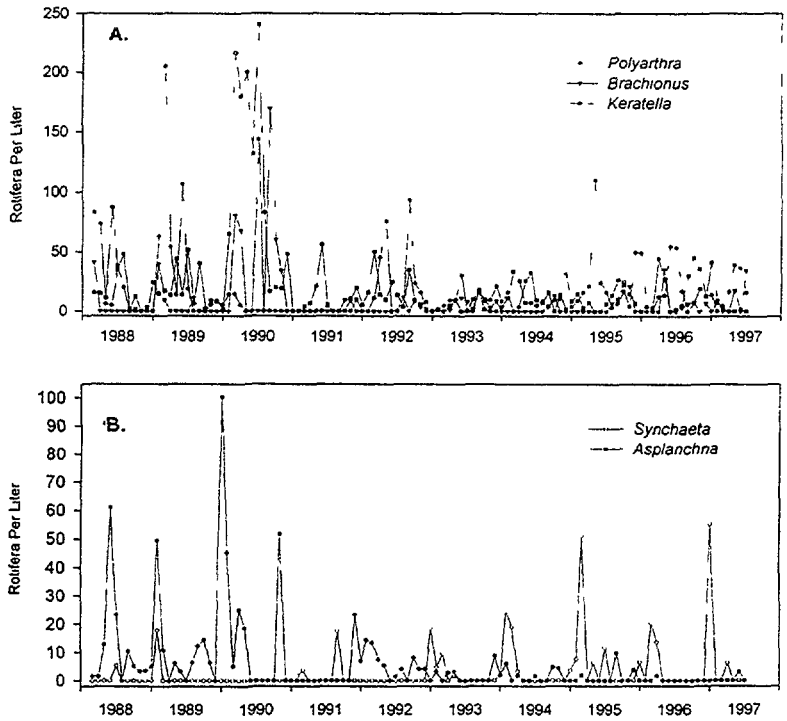

Figure 6. (A) Mean number of *Polyarthra, Brachionus* and *Keratella* per liter collected in vertical plankton tows. (B) Mean number of other rotifers per liter collected in vertical plankton tows from Clear Lake

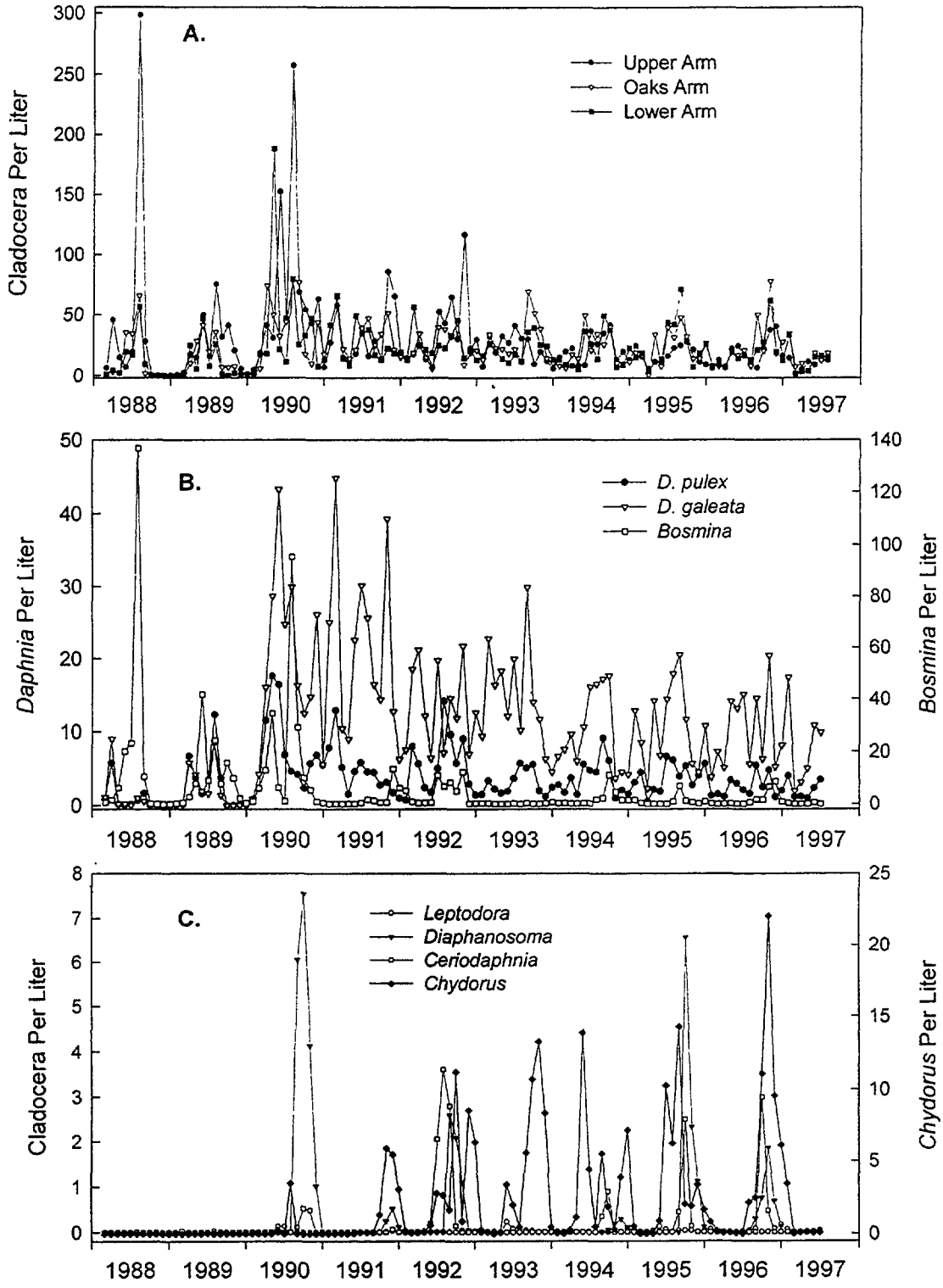

Figure 7. (A) Mean number of cladocerans per liter collected in vertical plankton net tows in each arm of Clear Lake. (B) Mean number of *Daphnia* and *Bosmina* per liter. (C) Mean number of other cladocerans per liter collected in vertical tows from Clear Lake. The *Leptodora, Diaphanosoma* and *Ceriodaphnia* densities are plotted on the left y-axis.

Table 2. Comparison of numbers of benthic organisms collected in Ekman samples during transects of Clear Lake in 1977 and 1994.

Water depth	Mean Number of Organisms per Square Meter											
	Chironominae		Tanypodinae		Orthocladinae		Chaoboridae		Oligochaeta		Hirudinea	
(meters)	1977	1994	1977	1994	1977	1994	1977	1994	1977	1994	1977	1994
0.5 (W. shore)	129	86	43	1766	0	0	0	0	0	12660	0	43
1.0	86	129	0	861	0	0	0	0	0	8612	0	0
2.0	65	10980	65	4263	0	517	0	0	129	2110	0	0
3.0	0	86	0	129	0	0	43	0	129	1163	0	0
4.0	43	258	22	172	0	0	22	43	473	3316	0	0
5.0	86	172	0	0	0	0	280	0	258	1163	0	43
6.0	0	215	22	0	0	0	215	0	516	2153	0	0
7.0	0	43	65	0	0	0	301	0	1118	4307	0	0
8.0 (Center)	0	194	65	22	0	0	345	0	559	1679	0	366
7.0	0	603	151	301	0	0	151	0	2064	2670	0	388
6.0	0	1033	65	301	0	0	43	0	258	345	0	388
5.0	43	86	280	258	0	0	0	0	86	603	0	474
4.0	108	345	108	646	0	0	0	0	43	1722	0	732
3.0	108	818	86	1421	0	0	0	0	0	818	0	129
2.0	194	1206	172	818	0	0	0	0	0	1206	0	0
1.0	0	6588	108	258	0	0	0	0	0	1421	0	0
0.5 (E. shore)	65	388	22	0	0	0	0	0	0	1679	0	43

High primary productivity in 1990 was attributable more to cyanophytes (primarily *Microcystis*) than to chrysophytes, pyrrophytes (primarily *Ceratium*), or chlorophytes (Figure 8). The cyanophyte *Gloeotrichia* was rare or absent from Clear Lake prior to 1993, but from 1993-1997 was sometimes the most abundant plankter in Clear Lake.

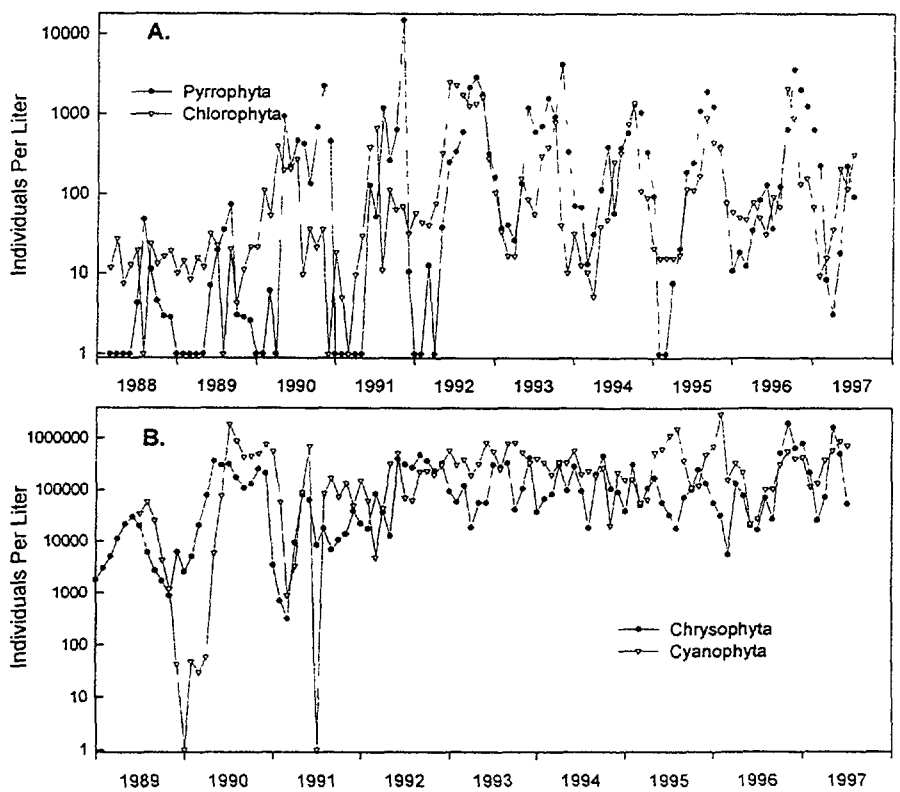

Figure 8. (A) Mean number (x + 1) of Pyrrophyta (dinoflagellates) and Chlorophyta (green algae). The Pyrrophyta were predominantly *Ceratium*. (B) Mean number (x + 1) of Chrysophyta (diatoms) and Cyanophyta (blue-green algae) per liter collected in vertical tows from Clear Lake.

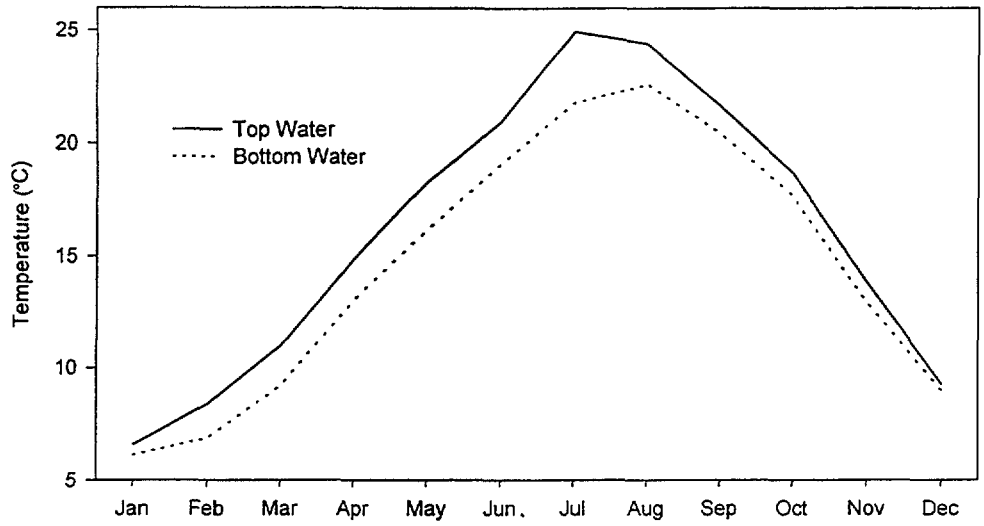

Figure 9. Mean temperature (°C) in top and bottom water samples collected from all three arms of Clear Lake during the years 1988 through 1996.

Water Quality. Differences in temperature or dissolved oxygen concentrations between the top and bottom waters of Clear Lake typically peaked during summertime periods when wind velocities were too low to achieve mixing (Figures 9-10). Large secchi depths and low turbidities are expected when the water is clear. Unclear water in Clear Lake may be associated during the winter with sediment inflow and during the summer with phytoplankton blooms [25]. The highest turbidities in Figure 11 have often occurred in the Upper Arm following winter storms. Water hardness and pH data appear in Figure 12.

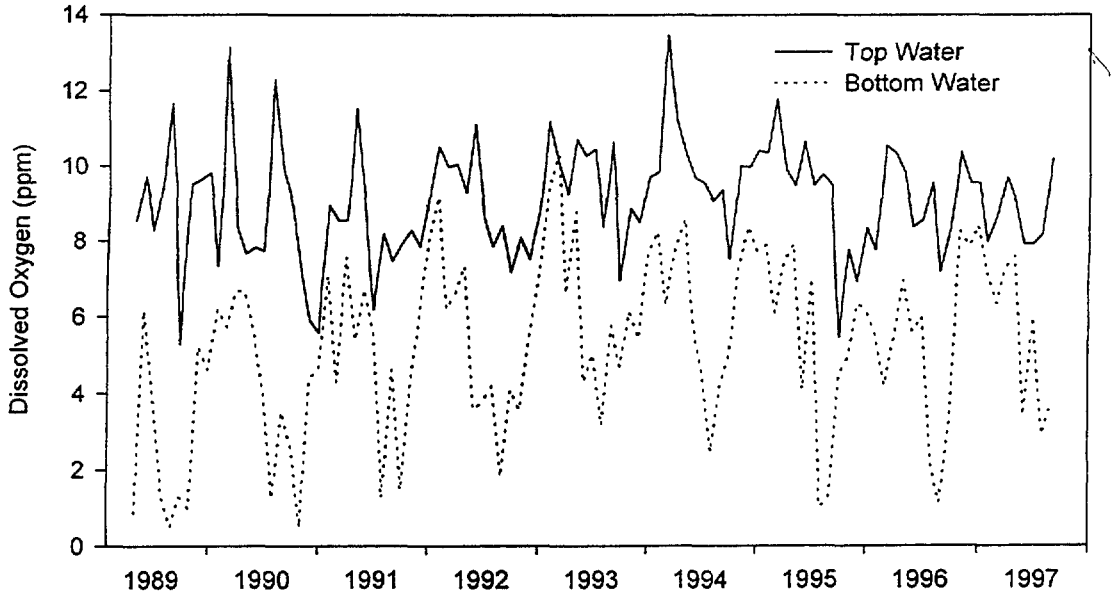

Figure 10. Mean dissolved oxygen concentration (parts per million) of top and bottom water (average for all three Arms) of Clear Lake.

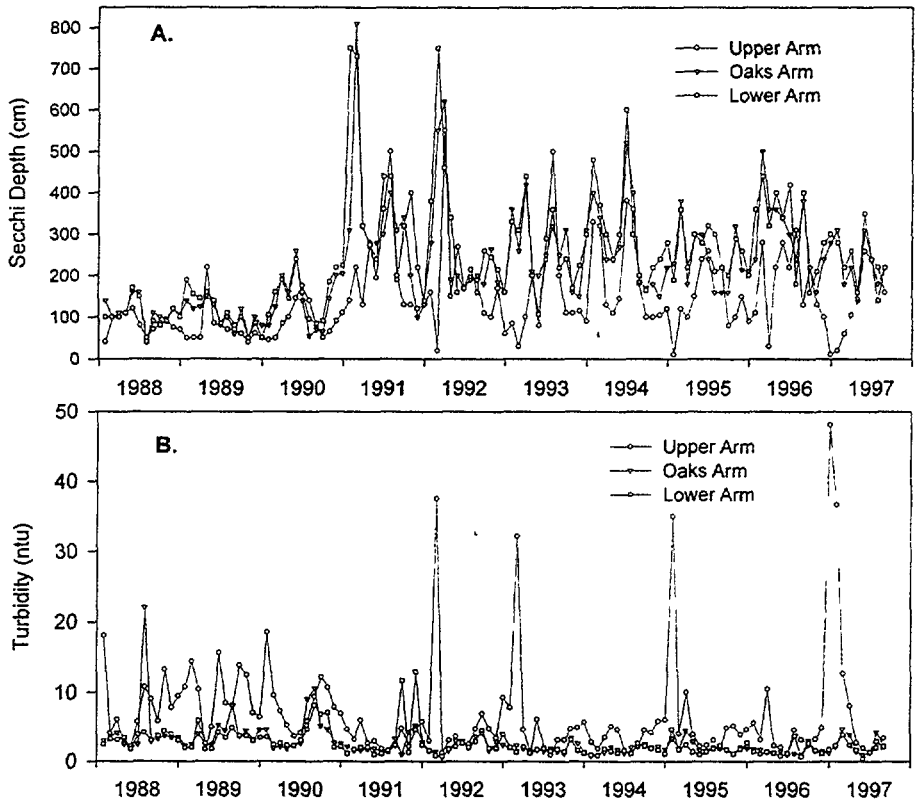

Figure 11. (A) Mean secchi depth (centimeters) recorded in each Arm of Clear Lake during monthly sampling from 1988 through 1997. (B) Mean turbidity (nephelometric turbidity units) in top and bottom water samples collected from Clear Lake.

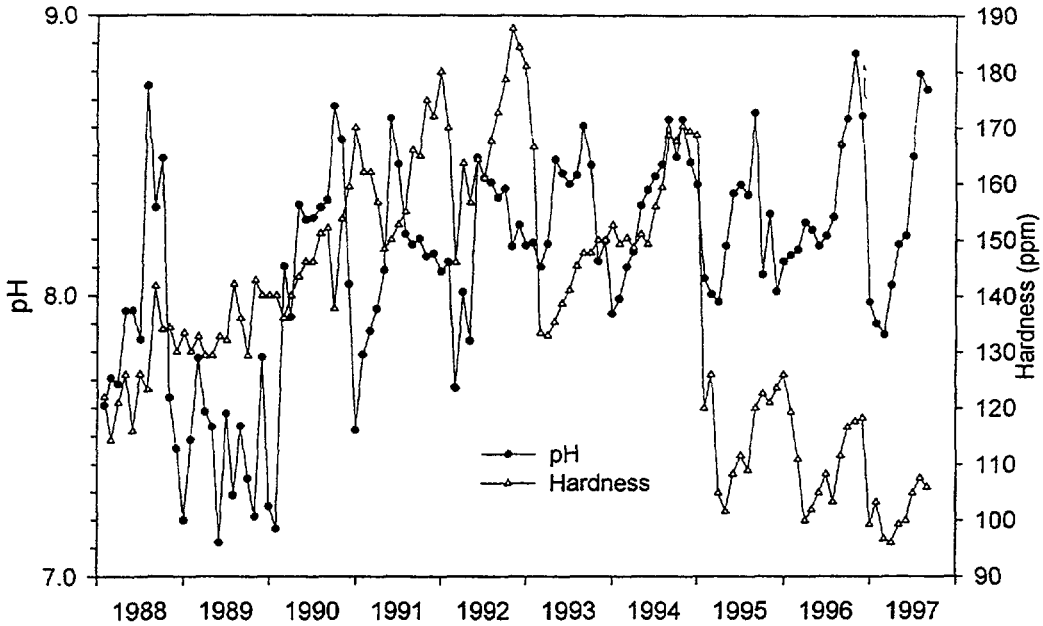

Figure 12. Mean pH and hardness in water samples collected from all three arms of Clear Lake.

Table 3. Average number of fish per hectare collected by beach seine hauls from Clear Lake. Data are averages for all three arms of Clear Lake during the period June 21 - October 21 for the years indicated.

Year	Inland silverside	Threadfin shad	Largemouth bass	Bluegill	Black crappie	Hitch	Sacramento blackfish	Carp	Brown bullhead	Prickly sculpin
1986	16,255.1	151.3	163.0	363.4	23.0	25.1	94.7	0.0	0.1	6.7
1987	7,032.0	2,067.0	78.0	151.0	12.0	0.0	0.0	0.0	0.0	3.0
1988	5,593.5	13,974.0	43.5	42.0	0.0	43.5	45.0	3.5	1.0	46.0
1989	16,655.5	4,788.5	78.5	39.0	8.0	18.0	82.5	1 5	0.0	33.0
1990	8,361.5	0.0	58.0	33.5	0.0	205.5	0.0	1.5	1.5	387.0
1991	10,604.3	0.0	1,058.6	203.1	95.7	1.9	0.0	90.3	20.2	1,892.1
1992	47,575.6	0.0	2,173.2	295.1	67.7	3.7	0.0	122.0	211.0	63.4
1993	10,683.9	0.0	551.4	296.6	3.6	199.7	0.0	17.9	3.6	598.0
1994	27,473.3	0.0	2,142.0	260.7	56.2	23.9	0.0	31.1	35.9	32.3
1995	86,778.2	0.0	1,169.7	21.5	3.6	21.5	1,815.5	6 0	4 0	0.0
1996	14,390.3	0.0	1,413.7	342.1	39.5	0.0	0.0	0.0	1.8	0.0
1997	16,303.9	3.6	825.2	747.5	33.5	352.1	0 0	0.0	4.8	473.6

Seine Hauls. Historically 12 native and 19 introduced species of fish have been identified from Clear Lake [26]. Extensive (334 seine hauls) sampling in 1986 collected 13 species of fish [20]. Data for the 10 most common species collected from 1986-1997 appear in Table 3.

Some additional seining records from previous years did not always indicate the methods utilized nor the area sampled, so it was not possible to calculate the number of fish per hectare. However the percent composition of the fish collected has been calculated for these earlier records (Table 4).

Table 4. Percent composition of beach seine hauls collected from Clear Lake.

		Percent of fish collected								
Year	Number of hauls	Inland silver-side	Bluegill	Prickly sculpin	Large-mouth bass	Black-fish	Hitch	Black crappie	Thread-fin shad	Others
1961	60	0.0	84.0	0.7	0.3	0.0	4.4	0.0	0.0	10.6
1962	84	0.0	82.4	1.0	0.2	<0.1	6.8	0.1	0.0	9.4
1964	25	0.0	90.0	2.5	0.1	0.4	1.7	0.1	0.0	5.3
1966	16	0.0	65.8	8.8	0.5	0.7	1.6	1.2	.0.0	21.4
1972	14	86.0	12.0	0.3	0.1	0.0	0.1	0.0	0.0	1.5
1974	35	79.6	16.4	0.2	0.9	<0.1	<0.1	1.0	0.0	1.9
1975	55	55.3	32.7	0.2	3.8	0.1	0.3	2.1	0.0	5.5
1976	382	89.5	8.0	1.0	<0.1	<0.1	0.4	0.4	0.0	0.6
1985	13	84.9	9.6	<0.1	4.3	<0.1	<0.1	0.2	0.0	0.8
1986	334	94.1	2.3	0.1	1.1	1.3	0.1	0.1	0.8	0.1
1987	33	75.3	1.6	<0.1	0.8	0.0	0.0	0.1	22.1	<0.1
1988	66	28.2	0.2	0.2	0.2	0.2	0.2	0.0	70.4	0.4
1989	99	78.2	0.2	0.2	0.4	0.4	<0.1	<0.1	20.6	0.0
1990	65	88.9	0.4	4.1	0.6	0.0	2.2	0.0	0.0	3.8
1991	33	56.2	0.1	14.5	4.0	0.0	23.9	0.7	0.0	0.6
1992	33	94.1	0.6	0.1	4.3	0.0	<0.1	0.1	0.0	0.0
1993	33	86.4	2.4	4.8	4.5	0.0	1.6	<0.1	0.0	0.0
1994	66	89.1	0.9	0.2	9.3	0.0	0.1	0.3	0.0	0.1
1995	65	96.3	<0.1	<0.1	1.3	2.0	<0.1	<0.1	0.0	3.0
1996	66	88.4	2.1	0.3	8.7	0.0	0.0	0.2	0.0	0.3
1997	69	86.1	4.0	2.5	4.4	0.0	1.8	0.2	<0.1	0.9

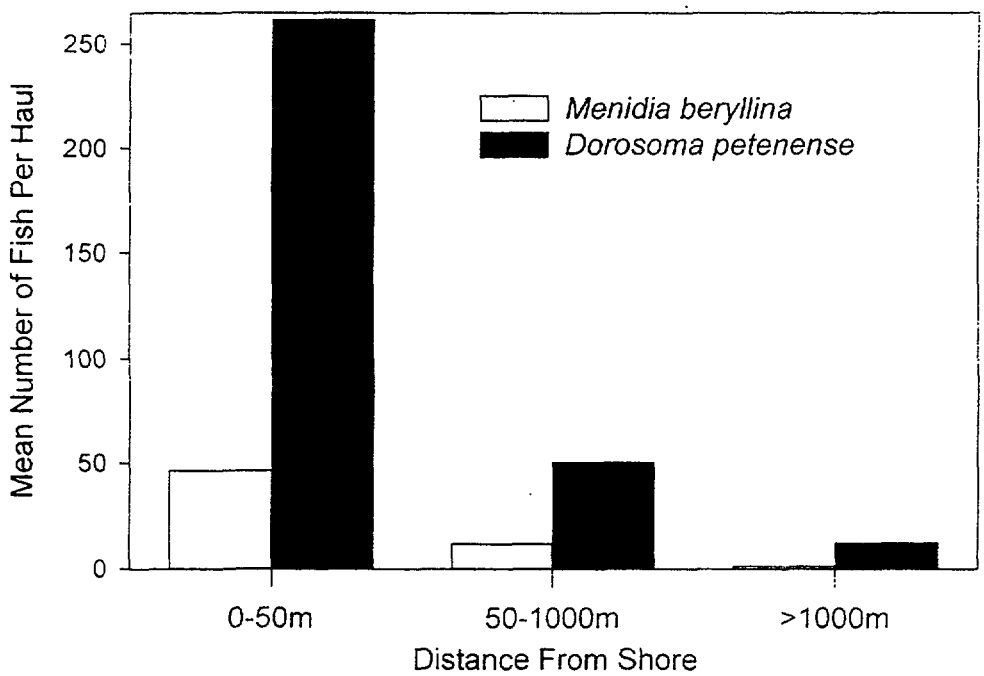

Figure 13. Mean number of inland silversides (*Menidia beryllina*) and threadfin shad (*Dorosoma petenense*) collected in surface trawl hauls from Clear Lake in 1989.

Surface Trawls. Silversides, threadfin shad (*Dorosoma petenense*), bluegill (*Lepomis macrochirus*), prickly sculpin (*Cottus asper*), black crappie (*Pomoxis nigromaculatus*), mosquitofish (*Gambusia affinis*), carp (*Cyprinus carpio*), hitch (*Lavinia exilicauda*), Sacramento blackfish (*Orthodon microlepidotus*), channel catfish (*Ictalurus punctatus*) and largemouth bass (*Micropterus salmoides)* have been collected during surface trawling. Typically silversides have been more abundant near the shore than offshore in Clear Lake (Figure 13).

Gut Content Analyses. There were differences in diet among each instar of *Chaoborus* and of Tanypodinae larvae. Also the diets of the silversides varied with the size of fish, and with the time, location, and date of collection. A summary (Table 5) of the percentage of the diet of all size classes indicated there was more dietary overlap (and possible competition for food) between the silversides and the *Chaoborus* larvae than between the Tanypodinae and the *Chaoborus*.

Table 5. Gut content analyses of some predatory organisms from Clear Lake. Data are expressed as the percent of the total number of organisms in the gut contents.

	Percent Diet of		
	Menidia	*Chaoborus*	Tanypodinae
Chlorophyta	0.0	0.1	15.9
Chrysophyta	0.0	8.2	71.1
Pyrrophyta	0.0	52.2	0.0
Rotifera	0.6	8.1	0.0
Cladocera	29.2	23.1	5.3
Copepoda	66.9	3.9	0.0
Insecta	2.6	0.0	2.0

Further Discussion. There have been several studies of possible pathogens of *Chaoborus* [27, 28, 29]. However no pathogen has been found to have caused the reduced abundance of *Chaoborus* in Clear Lake during recent years.

Statistical analyses of the data in Figures 1-12 have not revealed that any cryptic physical, water quality or biological factors were of overriding importance in controlling *Chaoborus* in Clear Lake. A significant correlation does not always indicate that a cause-effect relationship exists. For example the data from Figure 14 yield a positive ($P<0.05$) correlation between *Chaoborus* and *Leptodora* densities which may be due to similarities in the diets of these planktonic invertebrate predators. Neither predator has been sufficiently abundant in recent years to alter prey densities, but when suitable prey has become abundant, increases in the densities of both *Chaoborus* and *Leptodora* have often occurred simultaneously.

The most conspicuous change in Clear Lake from the time when *Chaoborus* were extremely abundant (e.g., 1960-

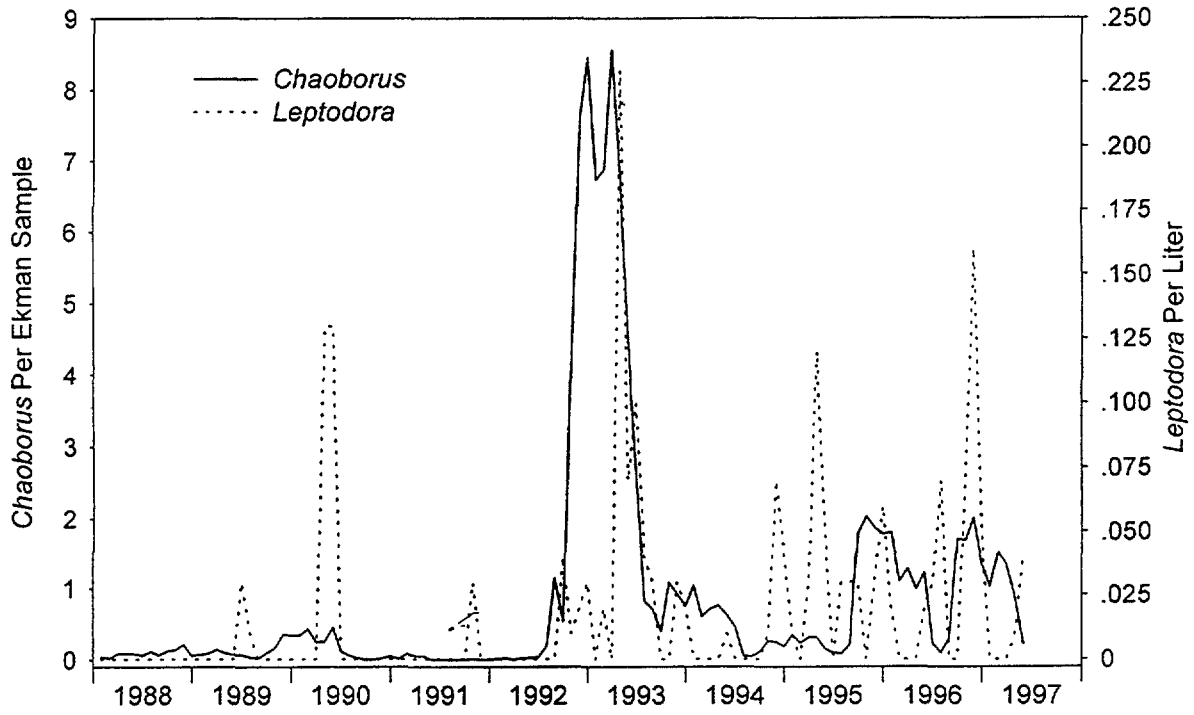

Figure 14. Mean number of *Chaoborus* larvae per Ekman sample and mean number of *Leptodora* per liter in vertical tow samples from Clear Lake.

1961) to recent (1994-1997) lower *Chaoborus* densities is that the silversides have become the most numerous fish in Clear Lake. There have been some concerns that there was little overlap in the distribution of the *Chaoborus* larvae and the silversides in Clear Lake [30]. However there is greater overlap with the locations of the silversides and of the *Chaoborus* pupae, emerging adults and ovipositing adults, and these are the stages primarily found in the stomachs of the silversides [20]. Data from 1979-1985 indicated there was a significant ($p<0.05$) negative correlation between silverside densities and gnat densities in Clear Lake [15]. However threadfin shad were found in Clear Lake for the first time in 1985, and were considered possible competitors (for planktonic food) of the silversides. By 1988 shad had become more numerous in near shore seine samples than the silversides (Table 3). Shad also became more abundant than silversides in offshore surface trawls (Figure 13). The *Chaoborus* density in Clear Lake decreased when shad were abundant (Figure 15). When the shad disappeared from Clear Lake after a cold winter in 1990, the *Chaoborus* density increased, but remained much lower than it had been prior to the introduction of the silversides.

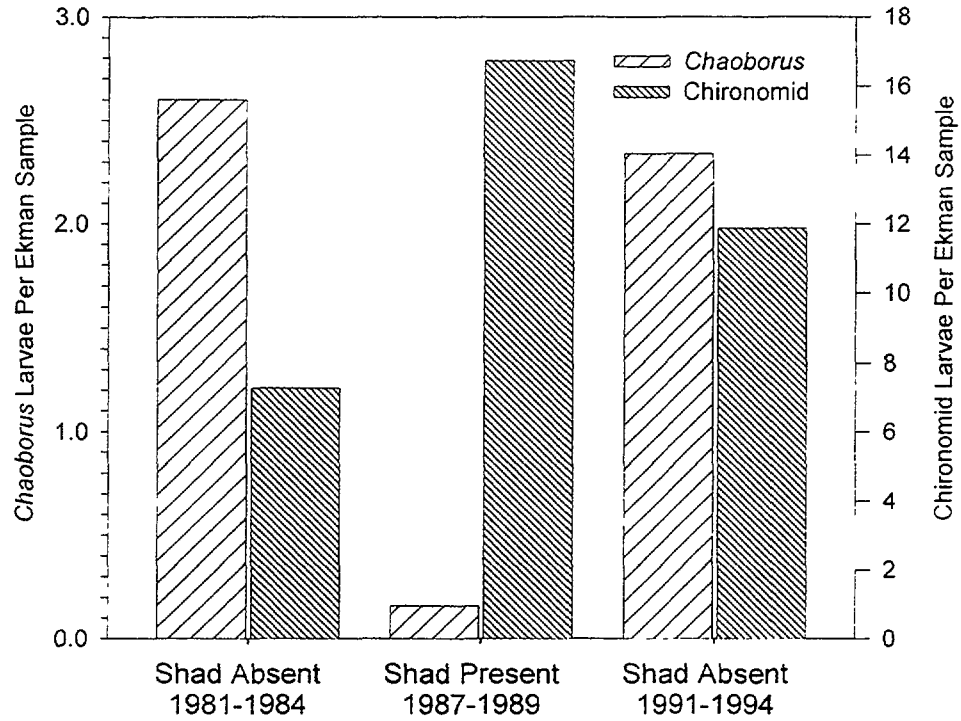

Figure 15. Mean number of larval Chironomidae and Chaoboridae per Ekman sample (in December) before shad were present (1981-1984), when shad were present (1987-1989), and after shad disappeared (1991-1994) from Clear Lake.

The threadfin shad greatly influenced Clear Lake food webs. Young-of-the-year largemouth bass (the seine hauls normally collect only these small bass) densities decreased when shad were abundant and increased when the shad population plummeted in Clear Lake. The data in Figure 16 yielded a significant ($P<0.05$) negative correlation between bass and shad densities. *Daphnia* (food for young-of-the-year largemouth bass) populations declined to record low levels when shad were present. Densities of 0.0 *Daphnia* per liter occurred only when shad were abundant in Clear Lake. When *Daphnia* densities were reduced, the densities of some smaller cladocerans (e.g., *Bosmina*) increased, especially in 1988. Statistical analyses of the data in Figure 17 indicated the densities of Western and Clark's grebes (*Aechmophorus*), Double-crested cormorants (*Phalacrocorax auritus*) and California gulls (*Larus californicus*) [31] were all positively correlated with shad abundances

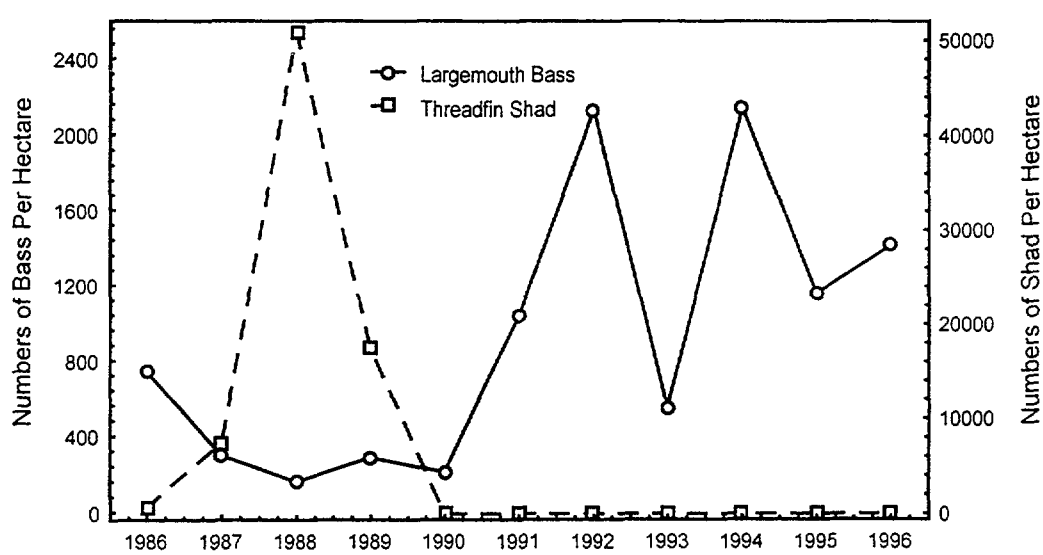

Figure 16. Mean number of largemouth bass and threadfin shad per hectare collected in shallow water seine hauls from Clear Lake.

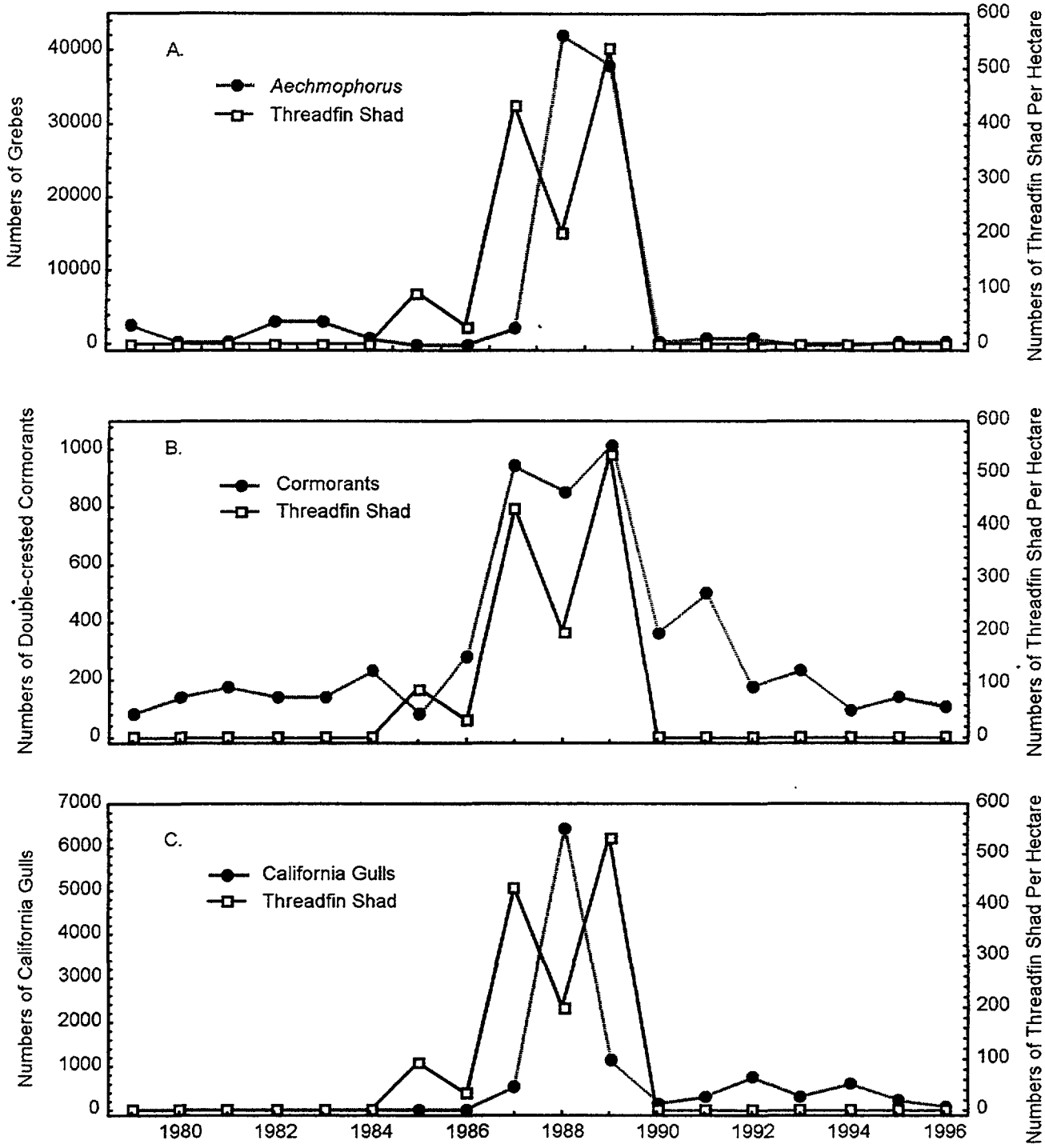

Figure 17. Mean number of threadfin shad per hectare collected in seine hauls and (A) number of *Aechmophorus* (Clark's grebes and Western grebes) counted in Audubon bird surveys; (B) total number of double crested cormorants counted and (C) and total number of California gulls counted in Redbud Audubon Society Christmas Bird Counts.

The densities of chironomid midges temporarily increased when shad were abundant in Clear Lake (Figure 15). Some species of epiphytic chironomids periodically increased when aquatic macrophytes (e.g., *Potamogeton*) were abundant in Clear Lake. Longer term changes in chironomid abundance have also occurred. The density of chironomid midges in Clear Lake was increasing from 1954-1997 (Figure 18). The biological, chemical and physical factors in Figures 1-12 have been analyzed as possible controlling factors of the chironomids. The only factors which were significantly ($P<0.05$) correlated with chironomid larval abundance (Figure 19) were pH and temperature. The LCVCD records indicate the pH of Clear Lake was relatively low during the 1960's and has been higher during the 1990's. The association of years with low winter temperatures and years with high summertime chironomid densities might be due to poor survival of natural enemies of the midges during cold winters.

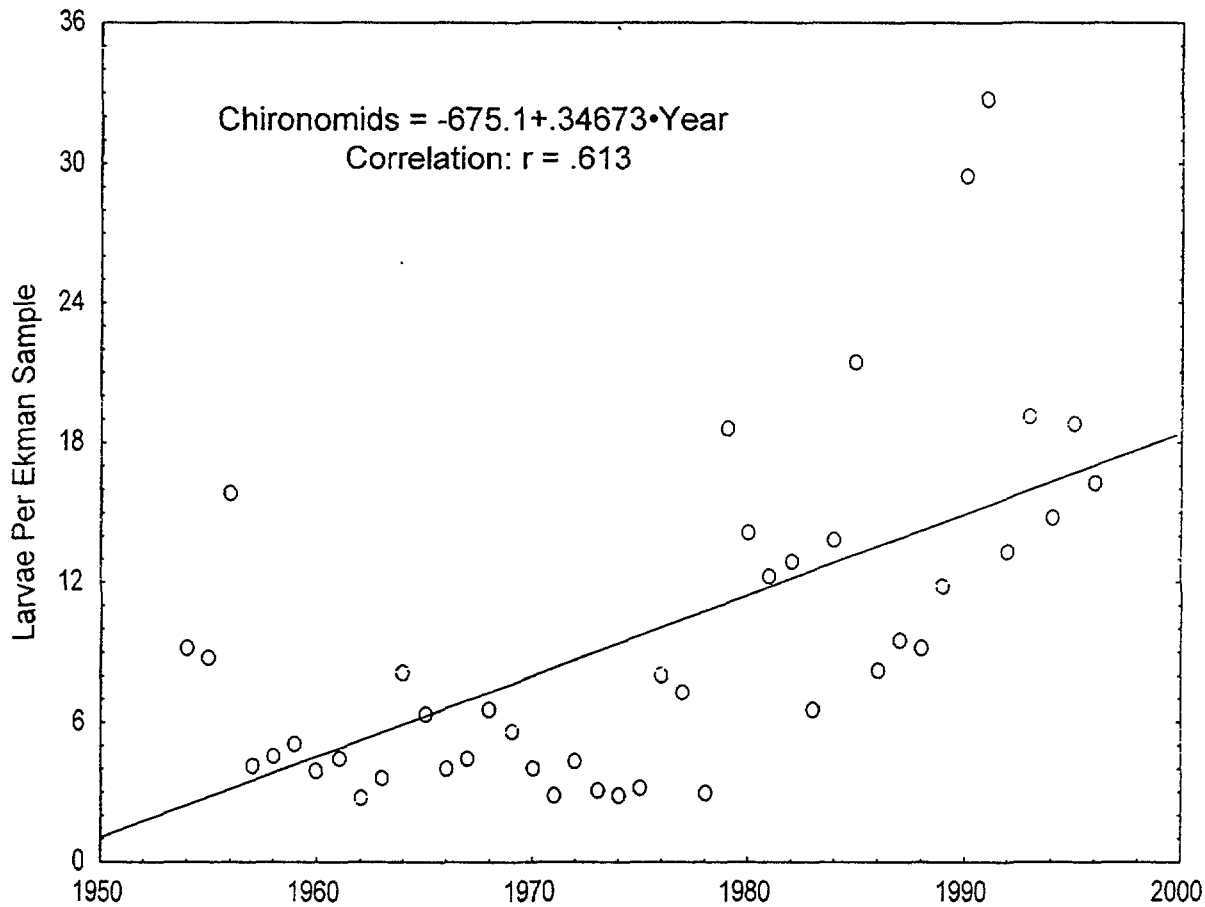

Figure 18. Mean number of chironomid midge larvae per Ekman sample collected from Clear Lake during the years 1954-1997.

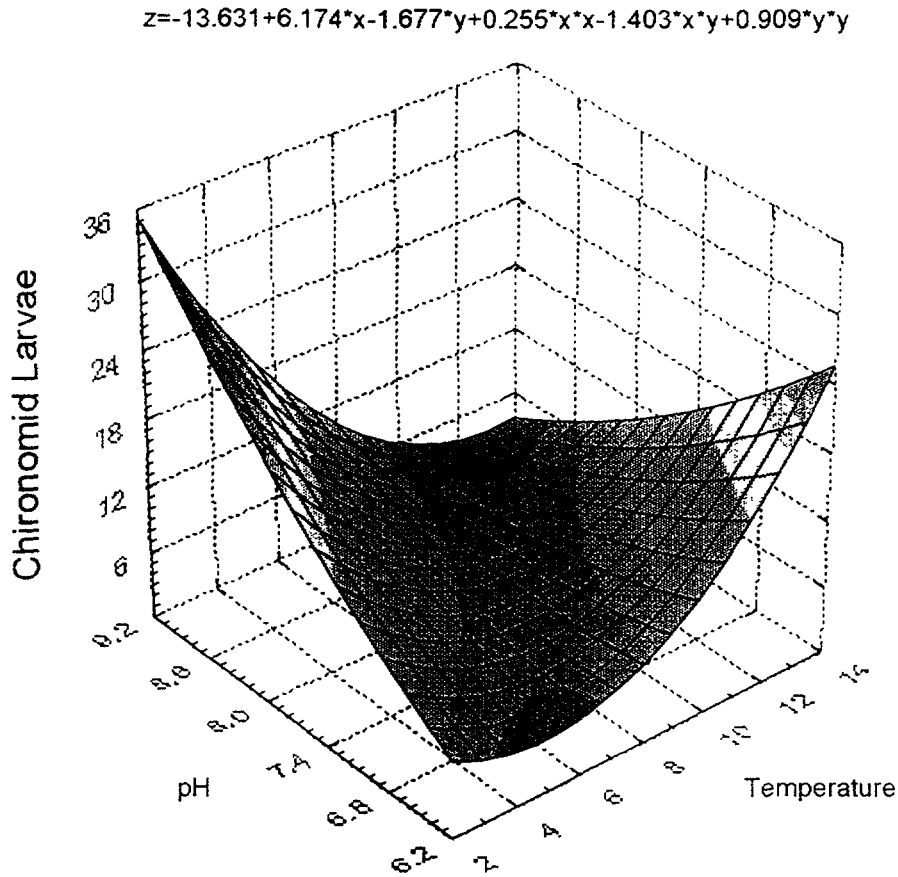

Figure 19. Number of overwintering (mean for the months November and December) chironomid midge larvae per Ekman sample, pH of top water of Clear Lake in August, and temperature (°C) of bottom water of Clear Lake during January

An objective of many monitoring programs is that of obtaining a uniformly collected data set for an extended period of time. However it is difficult to predict what standard monitoring system might be appropriate for Clear Lake throughout the 21st century. Drought, El Niño events, global warming, or introductions of new species [32] could necessitate changes in some monitoring methods. Changes in lake management strategies (to address algae or mercury problems, lake level fluctuations, etc.) might also require changes in sampling protocols. The cyanophyte *Lyngbya* can clog several types of sampling equipment and became abundant in Clear Lake for the first time in 1997. When some forms of aquatic vegetation (e.g. *Potamogeton*) become abundant, surface trawling becomes problematic. Increases in aquatic macrophytes (and associated gastropod mollusks) might prompt increased monitoring of the etiologic agents of schistosome dermatitis. If threadfin shad become the dominant fish in Clear Lake, additional midwater trawl sampling might be appropriate. However if hydrilla forms a thick mat over most of Clear Lake, it may become difficult to even access sampling stations or to obtain midwater trawl or Ekman samples. A severe hydrilla infestation would greatly increase habitat for immature mosquitoes, so additional surveillance techniques [33] may become necessary at additional locations to monitor dipteran pests in Clear Lake.

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 porgram 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 646 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 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, additional surface acres of infestation have been detected. As of January 1, 1997, approximately 646 acres of Clear Lake were infested with hydrilla. The current level of infestation includes 12 new areas, totaling 84 acres. A substantial portion of the infestation is located in the upper arm of the lake. Six 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 over the past year.

Survey Detection

Detection surveys outside the hydrilla eradication area were conducted by the Department's associate 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 noninfested 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 1996 season concentrated on the infested area. The first plant was found on May 1. The initial find of hydrilla in 1995 was on May 8. Both were found in locations 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 (shoreline) zones in the infested and noninfested areas were surveyed on a two to three week interval through December. Detection surveys in the deep sections of the lake were initiated in August and continued on a monthly basis until the end of November. No hydrilla has been detected in these areas.

During July, August, September and November, project personnel conducted detection surveys of Cache Creek, 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 Bear Creek-Cache Creek confluence. No hydrilla has been found in Cache Creek in.

Treatment

Initially in 1996, surface and subsurface applications of Komeen™ were utilized to control hydrilla. Komeen™ is a copper-based aquatic herbicide registered for use by the U.S. Environmental Protection Agency and the California Department of Pesticide Regulation. Treated water can be used for agricultural, recreational and domestic purposes.

Depending on water depth and amount of vegetation present at time of treatment, Komeen™ was applied at the rate of 12 or 15 gallons/acre. The first application was made on June 18, 1996. Infested areas were treated on a monthly basis. The last treatment was completed on November 6, 1996. All the infested areas were treated at least once with Komeen™. However, 13 areas were treated only once. Only single treatments of Komeen™ were required in these areas because no subsequent plants were found or the infested area was treated with Sonar™ and no additional Komeen™ treatments were required.

In 1996, the California Department of Pesticide Regulation issued a registration for Sonar™. Sonar™ is a systematic aquatic herbicide registered by the U.S. Environmental Protection Agency and widely utilized to control hydrilla and other submersed aquatic weeds outside of California. The slow release pelleted (Sonar™ SRP) formulation of Sonar™ was applied in all locations except one. The liquid formulation (Sonar™ A.S.) was applied in Holiday Harbor, a relatively small enclosed area with little water exchange.

Sonar™ SRP was applied with a mechanical spreader mounted on the front of the boats. The spreaders are commonly used to apply fertilizers and seeds in terrestrial situations. Applications were generally made at a rate of 10 parts per billion (ppb) twice a week for seven weeks. In some cases, application rates were increased to 20 ppb followed by subsequent treatments at 10 ppb. Sonar™ was applied at

20 to 30 ppb on a weekly basis in four locations. All the treated areas received a total maximum concentration of 140 to 150 ppb in accordance with label direction.

Complete control of all submersed aquatic weeds was obtained in all areas treated with Sonar™. In a few locations, a single application of Komeen™ was made after two or three Sonar™ treatments. These Komeen™ treatments were required to kill mature plants which show some signs of Sonar™ activity (chlorosis) but had not slumped to the bottom. Generally, once an infested area was treated with Sonar™ no further use of Komeen™ was required. The first Sonar™ application was made on July 17, 1996 and continued throughout the remainder of the growing season. The last application was made on November 21, 1996. 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 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. In contrast, 20,126 gallons of Komeen™ were applied in 1996, a 57.7% reduction in use.

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. Furthermore, the remaining 25.3 acres of Soda Bay did not require treatment and regrowth of plants resumed in the area not treated with Sonar™.

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 are 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 24,600 pamphlets were distributed to all motels, sporting good stores, gas stations and many other retail establishments. Pamphlets were also given to 1,500 lake shore homes around the lake.

Monitoring

An extensive monitoring program was conducted in 1996 by a newly established Environmental Monitoring group. This group of scientists from the California Department of Fish and Game, U.S.

Department of Agriculture - Agricultural Research Service and the University of California developed and conducted a comprehensive monitoring program to identify and evaluate the impact of eradication activities to non-target organisms. Additional monitoring activities were conducted by Big Valley Rancheria. Results of the monitoring activities will be presented in a separate report and distributed to all stakeholders and interested parties at a later date.

Progress and Plans for 1997

The hydrilla eradication program has significantly reduced the level of hydrilla in Clear Lake and prevented the spread to other bodies of water. Survey results indicate plant population is low and scattered in the infested area. Twelve new locations of hydrilla were detected in 1996. Seven of these locations contained only one or two plants. The remaining five locations contained six to twenty five plants. Spread of hydrilla to these new sites is disappointing; however, 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 1997. Treatment protocols will be developed toward optimizing the use of Sonar™ at the lowest possible rate. For example, selected areas may be treated at 7.5 ppb twice a week for 10 weeks. While this treatment schedule increases the application period from 7 to 10 weeks, it reduces the initial herbicide concentration by 25%.

STUDIES OF *AEDES SIERRENSIS* AND THE ECOLOGY OF WATER-FILLED TREEHOLES. (P)

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Abstract - Regional treeholes form rainfall-dependent lentic habitats that usually dry completely each summer. Treehole water was characterized by low levels of dissolved oxygen and highly variable, but often harsh water quality conditions. A detritus-based invertebrate community dominated by *A. sierrensis* and lacking a predatory trophic level was identified. Intraspecific competition for limited larval food resources was the most important regulatory factor for *A. sierrensis* identified in these studies. Adult activities of *A. sierrensis* primarily occurred from May until August, coinciding with the annual period when treeholes dry out.

Key Words - treehole mosquito, phytotelmata, *Aedes sierrensis*, dendrolimnetobiont

INTRODUCTION

Aedes sierrensis, the western treehole mosquito, is an economically important biting pest of humans [1] and a vector of *Dirofilaria immitis*, the canine heartworm [2]. Immature populations are largely restricted to water-filled treeholes [3].

The range of *A. sierrensis* includes most lower elevation woodlands of the Pacific rim of temperate North America [4]. In Lake County, CA., *A. sierrensis* is the predominant treehole mosquito in hardwood forests, habitats covering >28% of the county, or an area >5 times the size of Clear Lake. Vertically, water-filled treeholes occur from ground level to >20 m above ground in mature oaks. Water-filled treeholes may be the smallest, but most numerous lentic habitats in the Clear Lake watershed.

Control efforts against *A. sierrensis* have been limited by the labor-intensive, three-dimensional searches necessary to locate water-filled treeholes. Study of the ecological conditions present in water-filled treeholes could assist efforts to develop a biological control program for this mosquito. The objectives of the studies reported here were to survey the physical, chemical and biotic characteristics of water-filled treeholes in the Lake County region.

MATERIALS AND METHODS

Treehole water depths and adult activity periods of *A. sierrensis*. Water depths of eight treeholes in interior live oaks (*Quercus wislizenii*) and two treeholes in Pacific madrones (*Arbutus menziessii*) in an oak woodland near Potter Valley, Mendocino Co., CA., were measured on 44 dates during 1995. Host-seeking activities of adult *A. sierrensis* were monitored with carbon dioxide-baited Fay suction traps operated according to Garcia et al. [5]. Two traps were operated one day per week from March 24 to November 16, 1995. Oviposition was monitored with eight ovitraps operated continuously from April 11 to December 11, 1995. Each ovitrap consisted of a black plywood box with a screened vertical opening. Each box held a cup lined with rough-textured paper as an oviposition substrate and 380 ml of blue oak (*Q. douglassii*) treehole water as an ovipositional attractant. Ovitraps were constructed and serviced weekly according to Woodward et al.[6]. Concurrent daily maximum temperature and rainfall data were obtained from a U.S. Forest Service weather station near Potter Valley. The maximum depths and volumes of each of the treeholes were measured prior to the study.

Treehole water quality and adult emergence of *A. sierrensis*. During 1992, standard methods [7] were used to analyze water from 44 treeholes at sites near Lower Lake (n=12) and Kelseyville (n=9) in Lake County and at the Hopland Field Station (n=25) in Mendocino County. Water was analyzed from 30 black oaks (*Q. kelloggii*), three valley oaks (*Q. lobata*), four coastal live oaks (*Q. agrifolia*) and seven blue oaks. True color, pH, conductivity, total phenolics, total organic carbon and total Kjeldahl nitrogen were measured from samples collected on February 5 and 6. Ammonium and alkalinity were determined from samples collected April 29. Total seasonal emergence of *A. sierrensis* was determined by capturing all adults with emergence traps operated from April 8 to September 8. Adult female winglengths (n=2,717) were measured from 43 of the treeholes. Treehole volume and surface area were measured after removal of the emergence traps. Methods in this section were described in detail by Hartmann [8].

Aquatic insect communities of treeholes. Forty-four treeholes at eight sites between Lower Lake and Potter Valley were surveyed for aquatic insects from November 7, 1983 to May 3, 1985. Surveyed treeholes included 32 black oaks, five valley oaks, four interior live oaks and three Pacific madrones. Insects were collected by pipette on 3 to 24 dates per treehole. Laboratory-reared adults were identified by qualified taxonomists. Gut-content analyses were done on at least three larvae of each identified species. All adult eclosion from 15 of the treeholes was monitored with emergence traps from March 28 to September 2, 1985. Detailed methods were described by Woodward et al. [9].

Population dynamics of immature *A. sierrensis*. Larval and pupal populations of *A. sierrensis* were monitored in 10 treeholes at the Potter Valley study site. Each treehole was surveyed once per week from October 14, 1996 to July 5, 1997. Three water samples (ca. 45 ml each) were removed from each treehole with a polyethylene pipette (5 mm mouth diameter). Each sample was removed after gentle agitation of the water (three circular stirring motions with the pipette) to disperse immatures. Total sample volume was recorded, all *A. sierrensis* were identified to life stage, counted and returned to the treeholes. Treehole water depths were measured on each sampling date.

RESULTS AND DISCUSSION

Treehole water depths and adult activity periods of *A. sierrensis*. The treeholes monitored in Figure 1 had a mean maximum depth ± std. dev. of 17.1 ± 8.2 cm and mean maximum volume of 4.5 ± 4.3 l. The data indicate treehole water depths were

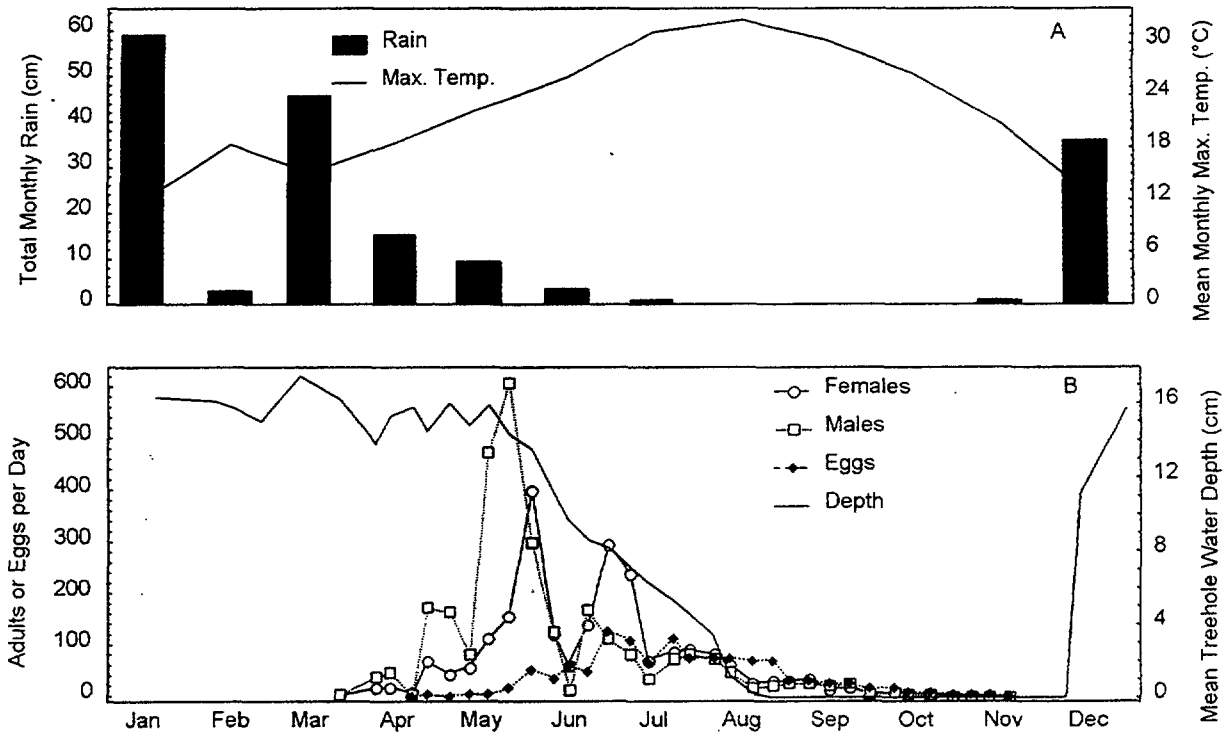

Figure 1. A) Total monthly rainfall and mean monthly maximum temperatures during 1995 at a U. S. Forest Service weather station near Potter Valley, CA. B) Mean numbers of host-seeking male and female *A. sierrensis* captured in carbon-dioxide baited Fay traps, mean numbers of *A. sierrensis* eggs oviposited into ovitraps and mean water depths in ten treeholes in an oak woodland near Potter Valley, CA.

rainfall dependent, with little influence from fluids secreted by the trees. Water depths peaked during the wet winter months and decreased during spring. All of the treeholes were dry from August 24 to December 11, 1995, when rainfall reflooded the treeholes. With Lake County's Mediterranean climate (mild, wet winters and hot, dry summers), this annual pattern of hydration is typical of most treeholes in the region.

Captures of host-seeking female and male *A. sierrensis* (males do not bite but seek the hosts of females to locate mates) peaked between April 24 and July 5. Host-seeking continued at lower levels until early November. Oviposition was detected every week from April 24 to November 9. Peak activity periods shown in Figure 1 indicate mating, host-seeking and oviposition largely occurred during spring and early summer when the treeholes in the woodland were drying. This correlation is related to characteristic traits of the eggs. Females oviposit desiccation-resistant eggs above the water in treeholes that remain quiescent until submerged by rain, usually during fall and winter [6]. Females which oviposit into treeholes partially full of water can place their eggs in a location which will be submerged by future rains. Oviposition into treeholes which have not dried completely insures the eggs are located in treeholes capable of holding water. In this study, more than 83% of the oviposition occurred between May 23 (when treehole water depths began a steady decline) and August 24 (when all of the treeholes had dried).

Seasonal ovipositional totals were remarkably high, averaging >8,100 *A. sierrensis* eggs per ovitrap. Totals in individual ovitraps (each holding 380 ml of treehole water) ranged from 4,413 to 12,580 eggs. With 100% survival and synchronous hatching, the mean ovipositional total would produce a larval density of >20,000 per liter. These data indicate *A. sierrensis* eggs were overproduced in the woodland relative to the numbers and sizes of water-filled treeholes available as oviposition sites.

Treehole water quality and adult emergence of *A. sierrensis*. Visually, treehole water color varied from transparent, light brown to opaque black. Spectrophotometrically, true color ranged from 973 to 86,050 platinum cobalt units (Table 1). Maximums for some factors were extreme when compared to most surface waters in Lake County. For example, total alkalinity in Clear Lake is usually <100 mg/l. Treehole water averaged 1,648 mg/l and reached a maximum of >11,000 mg/l of total alkalinity. Treehole water pH ranged from moderately acidic to highly basic. Measurements of most other factors were highly variable with ranges of one to two orders of magnitude. True color was positively correlated with many other water quality factors (Table 2). Darker water tended to have higher pH, conductivity, total phenolics, total organic carbon, total Kjeldahl nitrogen and ammonium. Dissolved oxygen levels in the treeholes were low, averaging <1 mg/l. Near anoxic conditions were measured in some treeholes.

Table 1. Chemical and physical measurements of 44 water-filled treeholes in Lake and Mendocino Counties, CA.

Measurement	Mean ± Std. dev.	Range
True color (Pt-Co units)	9,463 ± 17,190	973 - 86,050
pH	7.35 ± 1.02	5.31 - 9.76
Total alkalinity (mg/l)	1,648 ± 2,419	0 - 11,883
Conductivity (μmhos/cm)	1,855 ± 2,213	180 - 10,000
Total phenolics (mg/l)	113 ± 120	31 - 549
Total organic carbon (mg/l)	781 ± 1,304	82 - 6,200
Total Kjeldahl nitrogen (mg/l)	34 ± 55	3 - 302
Ammonium (μmoles/l)	334 ± 387	0 - 2,189
Dissolved oxygen (mg/l)	0.99 ± 0.65	0.05 - 2.70
Volume (liters)	5.2 ± 7.8	0.5 - 48.0
Surface area (cm^2)	347± 514	12 - 2,494

Total seasonal collections of *A. sierrensis* in emergence traps over each of the 44 treeholes averaged 275.7 adults with a range of 2 to 2,211 adults. Female winglength (a measure highly correlated with body weight) averaged 3.02 mm and ranged from means of 2.42 to 3.61 mm in individual treeholes. None of the water quality factors measured were significantly correlated with total adult emergence or female winglengths of *A. sierrensis* (Table 2). Only the physical factor of treehole volume showed a positive correlation with adult emergence. Even when corrected for volume (i.e., adults per ml of treehole water), none of the chemical factors were correlated with adult emergence of *A. sierrensis*.

Table 2. Pearson's correlation coefficients (r) for 44 water-filled treeholes comparing physical and chemical measurements with the numbers of *Aedes sierrensis* adults emerged and female winglength.

Independent variable	Total number of adults	Adults per ml of water	Female winglength	True color
True color	-0.04	0.09	-0.21	-
pH	0.03	-0.23	-0.28	0.65 *
Alkalinity	-0.14	0.01	-0.06	0.86 *
Conductivity	-0.07	0.04	-0.21	0.93 *
Total phenolics	-0.08	0.09	-0.14	0.96 *
Total Kjeldahl nitrogen	-0.14	0.04	-0.01	0.85 *
Ammonium	-0.16	-0.04	0.00	0.79 *
Total organic carbon	-0.06	0.08	-0.17	0.99 *
Dissolved oxygen	-0.01	-0.1	-0.20	-0.31
Treehole volume	0.70 *	-0.12	0.17	-0.17
Treehole surface area	0.15	-0.05	0.24	-0.07

* $p<0.05$

Some water quality factors (e.g., chloride concentration) have been negatively correlated with adult *A. sierrensis* emergence [10]. Data collected in this study, however, indicate *A. sierrensis* immatures tolerate a wide range of water quality conditions without a measurable effect on total emergence or size of the adults. The lack of a correlation between adult emergence and dissolved oxygen levels is not surprising since larvae use their siphons [3] to directly utilize atmospheric oxygen. The positive correlation between treehole volume and adult emergence indicates density-dependent factors, such as space and food limitations, may be important factors regulating *A. sierrensis* populations.

Aquatic insect communities of water-filled treeholes. "Dendrolimnetobionts" have been defined as "organisms largely restricted to water-filled treeholes" [11]. Nine species of dendrolimnetobiont insects were identified from the surveyed

treeholes. In addition to *A. sierrensis*, the fauna included a second culicid, a syrphid, two chironomids, two ceratopogonids, a psychodid and an helodid beetle (Table 3). Gut-content analyses indicated larvae of all nine species ingested particulate detritus from decaying leaf litter, as well as saprophytic protozoans and fungi. *Culicoides cavaticus* has been reported to be a facultative predator [13] but Woodward et al. [9] did not find evidence of predation on *A. sierrensis* in laboratory microcosms. Trophic relationships of the species (modified from Merritt and Cummins [12] to include laboratory and field observations) also indicate the community lacks a predatory trophic level.

Table 3. The dendrolimnetobiont insect community of water-filled treeholes in oak woodlands of Lake and Mendocino Counties, CA.

Order	Family	Genus species	Habit	Trophic relationships
Diptera	Syrphidae	*Blera humeralis* (Williston)	burrower/sprawler	filterer
	Chironomidae	*Limnophyes hamiltoni* Saether	sprawler/climber	collector/gatherer
		Polpedilum pedatum excelsius Townes	tube builder	shredder
	Ceratopogonidae	*Culicoides neofagineus* Wirth & Blanton	burrower/swimmer	collector/gatherer
		Culicoides cavaticus Wirth & Jones	burrower/swimmer	collector/gatherer
	Culicidae	*Aedes sierrensis* (Ludlow)	swimmer/planktonic	filterer/browser
		Orthopodomyia signifera (Coquillett)	swimmer/planktonic	filterer
	Psychodidae	*Telmatoscopus* sp. (undescribed)	burrower	collector/gatherer
Coleoptera	Helodidae	*Cyphon* sp. (undescribed)	climber	collector/gatherer

Results of emergence trapping indicated *A. sierrensis* was the dominant member of the community. More than 80% of the adult insects which emerged from the treeholes were *A. sierrensis* (Figure 2). Competitive advantages of *A. sierrensis* include the ability to feed by filtering and browsing, which allows the species to utilize the broadest range of particulates and microorganisms available to the community Other advantages include a desiccation-resistant egg for oversummering and the ability to utilize atmospheric oxygen.

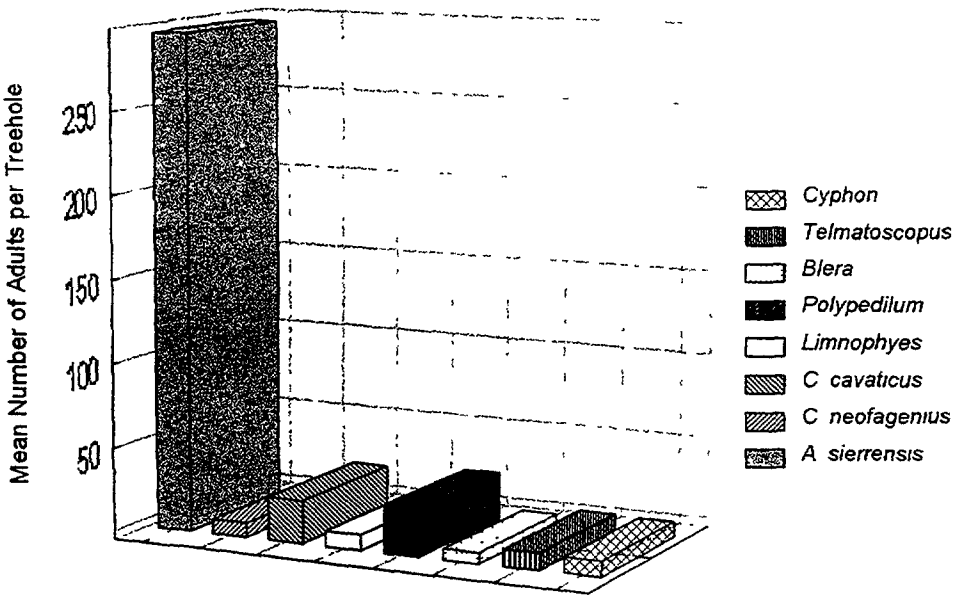

Figure 2. Mean numbers of dendrolimnetobiont insects captured in emergence traps over 15 treeholes between March 28 and September 2, 1985. The treeholes were located at seven sites between Lower Lake and Potter Valley, CA.

Emergence trap collections identified an average of 4.1 dendrolimnetobiont insect species per treehole, with a range of one to six species. Increases in diversity tended to broaden, but not heighten, the food webs in treeholes. The lack of a predatory trophic level may be related to the theory that fewer levels develop in ecosystems subjected to repeated environmental perturbations [11]. Regions where rainfall, desiccation and leaf fall are less seasonal may support more trophic levels than occur in northern California treeholes.

These results tend to support earlier studies which indicate interspecific competition [10] and predation [9] were not important factors regulating *A. sierrensis* populations.

Population dynamics of immature *A. sierrensis*. Densities of first instar larvae showed four major peaks during the fall and winter of 1996-97 (Figure 3). Each peak corresponded to a period when quiescent eggs hatched after rainfall raised water

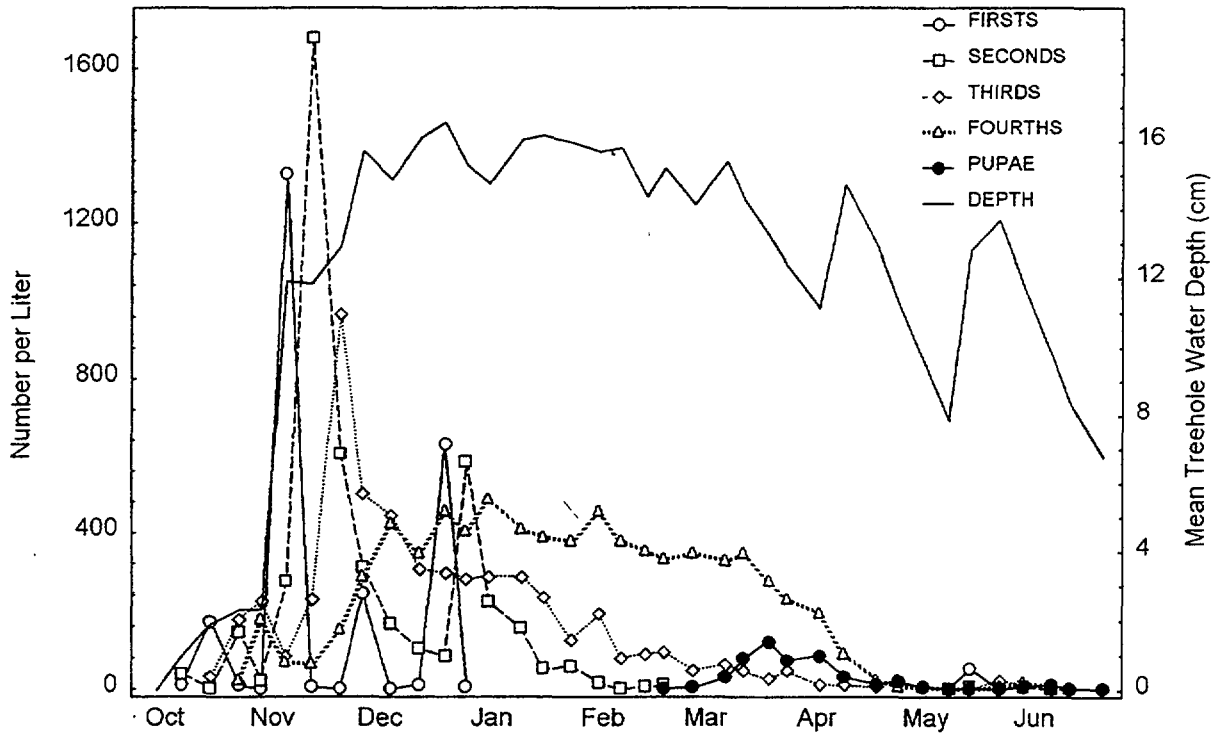

Figure 3. Mean numbers of *A. sierrensis* larvae and pupae per liter and mean water depths in 10 treeholes in an oak woodland near Potter Valley, CA. Sampling occurred weekly from October 14, 1996 to July 5, 1997.

levels in the treeholes. Rainfall between November 12 and 26 increased the mean treehole water depth from <3 to >11 cm. Egg hatches during this period produced mean larval densities as high as 1,974 per liter. Densities in two of the treeholes were >5,000 larvae per liter. First instars were last detected on January 8, six days after the treeholes reached their seasonal maximum mean water depth of 16.6 cm. Second and third instar larvae were last detected on March 4 and May 2, respectively, indicating protracted larval development periods occurred in some treeholes. Fourth instar larvae enter diapause during winter and do not pupate until increases in daylength and temperature occur near the beginning of spring [14]. In the present study, fourth instars were first observed on November 6. Even though earlier instar larvae continued to molt to the fourth instar during winter, a generally downward slope in fourth instar density occurred during January and February. Pupation of overwintering larvae was first detected on February 27, peaked during April, and was last detected on May 28. Maximum pupal density in spring was <8% of the maximum larval density during the previous fall, indicating larval mortality during the winter was high.

Figure 3 also shows a small summer generation of *A. sierrensis* occurred in some treeholes during late May and June. Eggs oviposited during early May hatched following rainfall on May 24 and 27, producing these populations of larvae.

The results indicate intraspecific competition had a major regulatory role on larval populations of *A. sierrensis*. The populations showed high initial larval densities, protracted development periods to the fourth instar and low survivorship. Hawley [15] associated similar data with low per capita resource levels. The intensity of intraspecific competition in individual treeholes may also affect the regulatory role of natural enemies of *A. sierrensis*. Mortality caused when early

instar larval densities are high can relax competition between the survivors [16], resulting in an increase in total adult emergence from some treeholes. This indicates the timing of mortality caused by potential biological control agents of *A. sierrensis* may be critical to their effectiveness. Mortality of the late-fourth instar larval or pupal stages (after attrition due to intraspecific competition has occurred) has the greatest chance for reducing the biting and vector potentials of this mosquito.

ACKNOWLEDGMENTS

Data shown in Table 1 were collected during a cooperative study with Jan Washburn and Evilene Hartmann (Univ. Calif., Berkeley). We also thank Joann Haberthur, Terry Sanderson and Lynn Woodward for their assistance.

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PHYSICAL/ CHEMICAL LIMNOLOGY

Multiple Stress in a Freshwater System: Copper's Impacts on Methyl Mercury Toxicity and Accumulation in Zooplankton (P)

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Key Words: methyl mercury, zooplankton, heavy metals, copper

This study aims to characterize how freshwater zooplankton respond to stress from multiple agents. In specific, the agents studied are the heavy metals copper and methyl mercury, two elements whose natural concentrations in Clear Lake, California are elevated through human activities in the region. The study's three objectives are: 1) to characterize the interaction of methyl mercury and copper in their toxicity to the standard aquatic toxicology test organism Ceriodaphnia dubia as antagonistic, additive, or synergistic; 2) to characterize the toxic interaction of these metals to a Clear Lake endemic zooplankton, Daphnia pulex; as antagonistic, additive, or synergistic; and 3) to determine if copper stress alters the accumulation of methyl mercury in D. pulex. Tests are conducted in a laboratory setting and are currently ongoing. Preliminary results indicate that methyl mercury and copper act conjunctively in inducing an additive toxic response in both zooplankton species, and that the response of the Clear Lake endemic D. pulex is very similar to that of the standard test organism C. dubia.

Iron Cycling and Its Effects on Algal Growth in Clear Lake

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Key Words: iron limitation, bioassay, algal growth, iron cycling, speciation, Fe/P transport

In order to understand the cycling of iron and its effect on phytoplankton production in Clear Lake, California, this study combines bioassay methods, geochemical assessment techniques and mathematical model to describe the iron behavior in the water column of the lake, and its effect on algal growth. A series of nutrient bioassays was performed to test the iron limitation hypothesis. These enrichment experiments, both in vivo and in vitro, indicated that chlorophyll a production is stimulated by chelated iron and nitrate significantly in most experiments. Nitrogen fixation of bluegreen algae is simulated (10-50% above the control) by chelated iron addition. Statistic analysis shows that the increase of chlorophyll a level is contributed by nitrate and chelated iron independently, while the increase of nitrogen fixation rate is interacted by iron and its chelator EDTA. Dynamic distribution and mobility of iron, nitrogen and phosphorus in the water column were deduced from a year-long monthly study of physical, chemical and biological parameters in the lake. The shapes of element profiles indicate that during the summer-autumn time a large amount of phosphorus released from the sediments is responsible for high concentrations of phosphorus and low N/P ratios in the water column, which favors bluegreen algae. The apparent simultaneous release of Fe and PO43- only occurs when the sediment-water interface becomes anoxic. Soluble iron does not build up concentration gradient and keep low concentrations during the summer-autumn time, which may limit algal growth and nitrogen fixation in Clear lake. Iron and phosphorus speciation were calculated by running MINTEQA2. Under the equilibrium condition, the dominant species of iron and phosphate in Clear Lake are Fe(OH)4-, Fe(OH)3(aq), and HPO-2 and Mg- and Ca-phosphate complexes. A 1-D mathematical model, including both diffusion and reaction terms, was developed to simulate the field observations of Fe/P transport at the sediment-water interface.

CURRENTS IN THE SHALLOW WATER OF THE UPPER ARM OF CLEAR LAKE

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Abstract - Currents in the near-shore shallow water of the Upper Arm of Clear Lake were studied. Currents away from shore due to differential cooling were of particular interest since very little is known about their behavior and therefore their significance in relation to other currents for transport of herbicides. The herbicides are sprayed in the shallow regions at the edges of the basin in association with a program of eradication of the aquatic weed, *Hydrilla verticillata*. High accuracy thermistor arrays were used to measure temperature stratification and to indirectly yield information about water movement. Acoustic Doppler velocity profilers measured directly the velocity profile through the water column at specified intervals. Valuable information was collected on the behavior of the water currents during different stratification scenarios. Currents due to differential cooling were found to be much more complex than was expected, possibly influenced by the shape of the shoreline just as much as the bathymetry (bottom slope) of the basin.

Key Words - temperature, velocity, differential cooling, hydrilla, transport and fate of herbicides.

INTRODUCTION

A program directed towards quantifying the currents in the shallow water of the Upper Arm of Clear Lake was commenced in summer of 1996 and continued in summer of 1997. The program is still in progress, as of September 1997, and the observations up to this point are discussed in this paper.

Different categories of currents in the basin were studied and their significance towards transport of material, in particular herbicides sprayed in the shallow regions. The spraying is associated with a program of eradication of the aquatic weed, *Hydrilla verticillata*. The study can lead to the development of a management tool to optimize timing and placement of the herbicides used for the eradication. Managing the spraying efficiently is both of economical and environmental concern. It is important to know at what rate herbicides are transported out of the treatment regions to other regions in the basin, and to know where traces of the chemicals are expected to be found. Management of other noxious plants at Clear Lake will take advantage of these studies.

Two categories of currents in particular are expected to contribute to the transport of material in the basin: wind driven currents and currents resulting from differential cooling [1]. The wind driven currents can be categorized even further in a simplified way to one-directional flows and exchange flows. The one-directional flows usually occur in the case of very weak thermal stratification in the water column or if it is fully mixed from top to bottom. Exchange flows, when the epilimnion and hypolimnion are traveling in opposite directions, are usually dominant when there is temperature stratification during a wind event (figure 1).

Of particular interest are currents away from shore due to differential cooling. Little is known about their behavior and therefore their significance in relation to other currents for material transport. Differential cooling occurs during the night, when the shallower near-shore surface water cools more rapidly than the deeper water further away from shore. Once the deepening surface layer exceeds the depth of the shallow near-shore region, continued surface cooling results in a horizontal temperature differential between the shallow and the deep regions. This can lead to an under flowing current of colder water intruding into the water column away from the shallow regions toward the deeper regions. Further water quality consequences of the away-shore transport by differential cooling can be addressed, like the ventilation of sediments due to oxygen transport.

FIELD SAMPLING

Measurements of the currents in the Upper Arm were both of a direct and indirect nature. Temperature chains were used to measure temperature stratification and to give indirectly information on currents away from shore due to differential cooling. Acoustic Doppler velocity profiler instruments were used to measure the current directly. Sampling sites were chosen from those areas that are presently being treated to control growth of *Hydrilla*. The area off Reeves Point was the area where *Hydrilla* was first discovered in Clear Lake, in summer 1994. Since then the area has been treated with Komeen, a copper based herbicide, and Sonar, which uses fluridone as its active ingredient, and is still being treated [2]. This area was studied in particular in July 1997. During that time, data on the behavior of the currents in the basin were collected. Instruments were placed at three sites off Reeves Point, 0.15, 1.5 and 3.0 km from shore (figure 2). The site closest to shore was 2 m deep, and it had a temperature chain. The sites further away from shore were 4.5 and 5.5 m deep, having both temperature chains and velocity profilers.

The temperature chains consisted of self contained temperature loggers. At sites 2 and 3, 6 loggers at 50 cm spacing

formed the chain starting 50 cm above the bottom Site 1 had 2 loggers one at the bottom and another 50 cm below the water surface The velocity profiler instruments at sites 2 and 3 measure the velocity of the current in the water column and give the velocity at evenly spaced intervals through the depth (figure 3) The profilers measure the velocity of the water using the physical principle of Doppler shift This states that if a source of sound is moving relative to the receiver the frequency of the sound at the receiver is shifted from the transmit frequency The profilers have three transducers each used as transmitter and receiver The transducer generates a short pulse of sound at a known frequency which propagates through the water As the sound travels through the water, it is reflected in all directions by particulate matter (sediment biological matter, bubbles, etc) Some portion of the reflected energy travels back along the transducer axis, where it is received by the transducer and the processing electronics measure the change in frequency

Weather data were collected from local weather stations According to these data, the weather pattern during July 1997 was very similar from day to day with a few minor exceptions The wind, being the most important meteorological parameter to determine in this study usually increased in the afternoon or early evening and died off close to midnight The diurnal cycle of solar radiation and air temperature was relatively constant

RESULTS

The sampling period in July 1997 was 18 days, from noon July 3rd to noon July 21st The water column was stratified during the day at all sites for the entire period Full mixing occured during 9 nights at site 2 and during 4 nights at site 3 Site 1 was fully mixed at some point during the night in all cases except 2 nights (figure 4) The stratification at night was often weak and confined within the 1 - 2 m bottom layer Total mixing was in general associated with strong wind events and in some cases the air temperature dropped noticeably

The wind commonly increased in late afternoon or early evening which warmed up the surface layer (epilimnion) due to increased mixing Generally the water column was either weakly or strongly stratified during early afternoon resulting in exchange flow when the wind increased The hypolimnion at sites 2 and 3 generally warmed up also often with a 5 - 7 hr lag time between the sites, indicating either flow of warmer water in the bottom layers due to the wind setup or downward mixing of surface water with bottom water The exchange flow during the wind action changed to one-directional flow when the water column became fully mixed at night At times when the water column stayed stratified during the night the exchange flow changed to multi-directional flow when the wind ceased

The transition from exchange flow to one-directional flow when the water column became mixed at site 2 was in general very clear whereas in the few instances when the water column at site 3 was close to or became mixed the transition was not as clear and the one-directional flow was interrupted by multi-directional flow On July 16 there was stratification the afternoon at sites 2 and 3, the wind picked up in late afternoon and evening, and exchange flow was experienced at both sites The water column at site 2 was fully mixed right by 9 pm and almost immediately the flow changed to be one directional (figure 5) At site 3 the water column was weakly stratified instead of being fully mixed resulting in multi directional flow

Horizontal temperature differentials were often observed in the data set both during the day and night with night generally being greater Indication of intrusion likely to be due to horizontal temperature was observed in the morning of July 20 at site 3 The wind pattern was similar to what it was on July 16 to 17 and a similar current pattern was observed in the afternoon and evening on July 19 as on July 16 But, the water column did not fully mix at site 3 during the night (it was weakly stratified) resulting in irregular current pattern and fluctuations in temperature in the lower part of the water column At that same time the water column at site 2 was fully mixed for only 4 hours and the flow was mostly one directional with some irregular fluctuations In the morning at 8 45 am there was a sudden drop in temperature at depths 3 5 m and 4 5 m Up to that time the direction of the current at these depths was mostly towards the north-east but it changed to be towards the south east with a speed on the order of 3 cm/s (figure 6) At that time the water column at site 2 south west of site 3 was warmer than site 3 which suggests that the intrusion flow came from a direction other than site 2 This indicates that the shape of the shoreline of the basin is affecting the temperature gradients, resulting in a much more complex behavior of these currents than was expected Instead of following bathymetry gradients (bottom slope) and having a relatively constant direction currents intrude somewhere above the bottom and have a complex path while the horizontal differences in temperature between different areas are leveling out

DISCUSSION

Up to this point, information on the behavior of the water currents in the Upper Arm of Clear Lake has been collected during different stratification scenarios Also, and maybe more importantly, the behavior of currents due to differential cooling have been found to be much more complex than was expected Further information is needed to estimate the significance of these currents for the transport of herbicides in the basin The question whether the shape of the shoreline affects the currents due to horizontal temperature differentials is of immediate importance Choosing sites where these effects are expected to be minimal or extending the area of interest at the current site with additional transects of instruments could

give an answer to that question.

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Figure 1: Two categories of currents in particular that can be expected to contribute to the transport of material in the basin, (a) Wind driven currents, one-directional flow (left) and exchange flow (right), (b) Currents resulting from differential cooling.

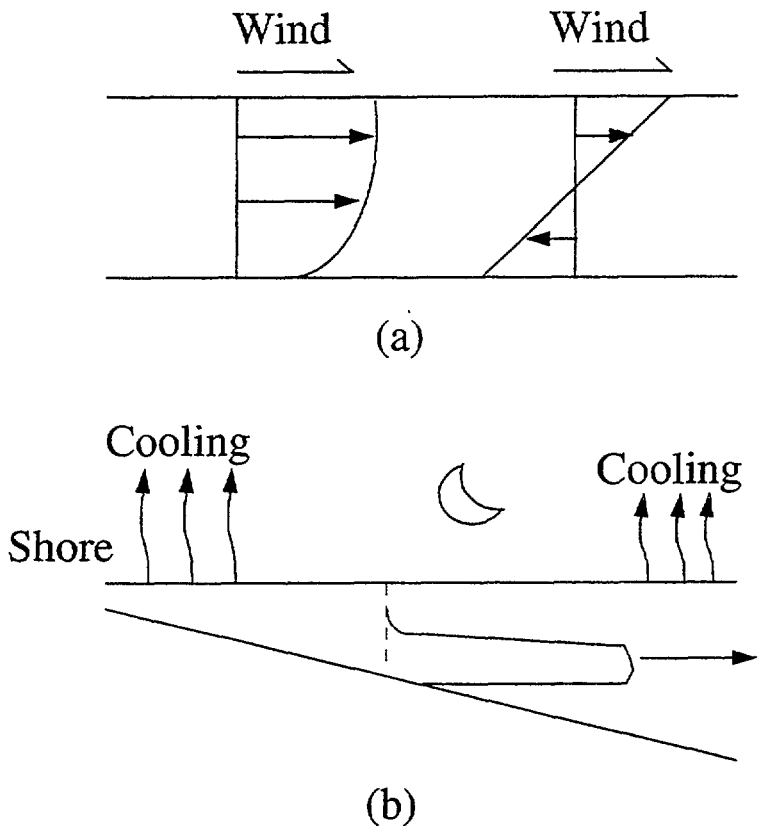
Figure 2: Sampling sites off Reeves Point in the Upper Arm of Clear Lake, in July 1997.

Figure 3: Deployment setup. Self contained temperature loggers and acoustic Doppler velocity profilers at sites 1, 2 and 3, 0.15, 1.5 and 3.0 km off shore respectively.

Figure 4: Nights of full mixing of the water column at the three sites during the sampling period in July 1997. A dot represents a night where full mixing occured.

Figure 5: Example of transition from exchange flow to one-directional flow in late evening on July 16 at site 2, resulting from total mixing of the water column right before 9 pm. The upper part of the figure shows temperature at different depths vs. time, and the lower part shows corresponding velocity measured at same site from 0.5 m to 3.5 m depth vs. time. Each arrow shows magnitude and direction of the current at corresponding depth and time, north being up. The scale of magnitude at each depth is -5 to 5 cm/s, a minus sign indicating flows towards south. The time axis runs from noon on July 16 to noon the following day.

Figure 6: Example of currents likely to be due to horizontal temperature differential in the morning of July 20 at site 3. A sudden drop in temperature at depths 3.5 m and 4.0 m, right before 9 am, with corresponding change to flows towards south-east from being towards north-east at same depths. The upper part of the figure shows temperature at different depths vs. time, and the lower part shows corresponding velocity measured at same site from 2.5 m to 4.0 m depth vs. time. Each arrow shows magnitude and direction of the current at corresponding depth and time, north being up. The scale of magnitude at each depth is -3 to 3 cm/s, a minus sign indicating flows towards south. The time axis runs from midnight to noon on July 20.



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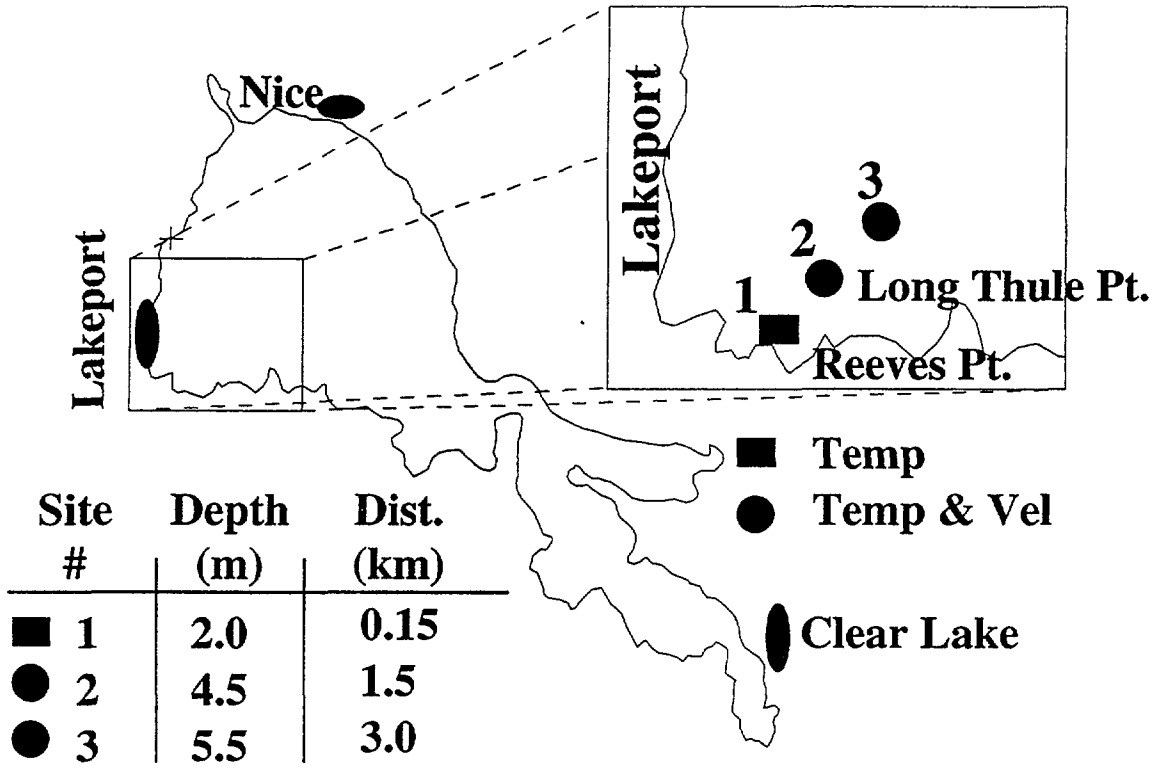
(a) Wind driven currents, one-directional flow (left) and exchange flow (right)
(b) Currents resulting from differential cooling



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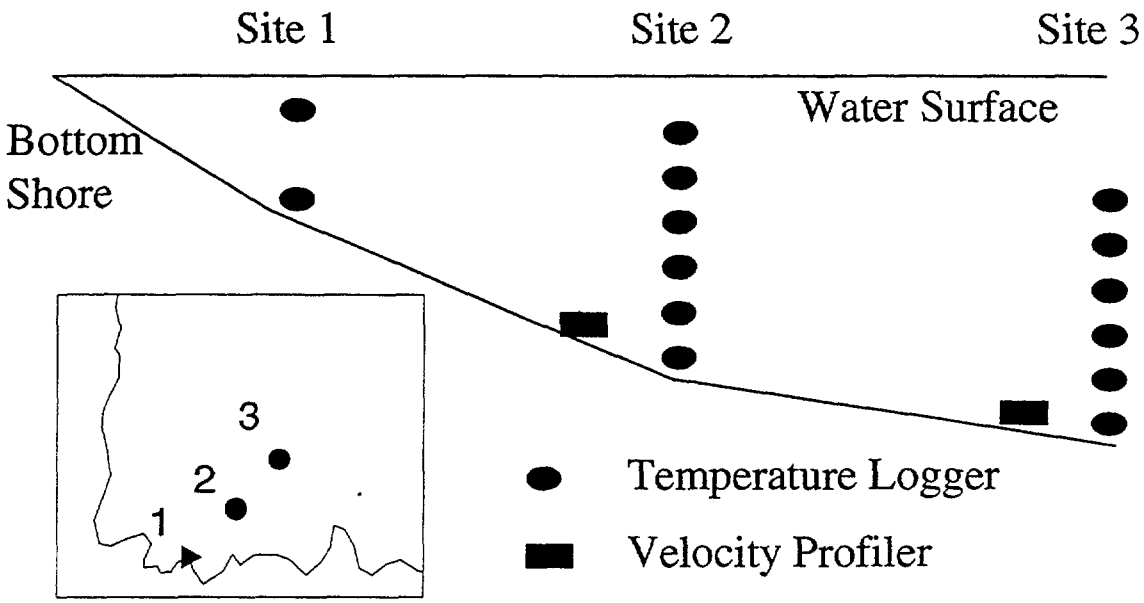

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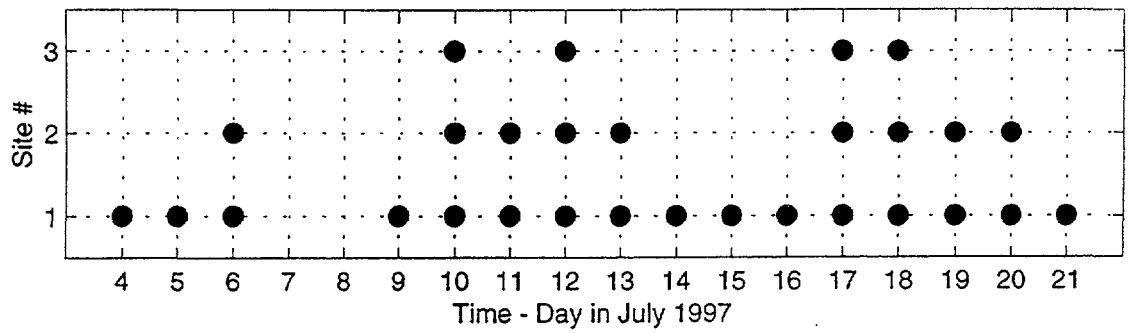
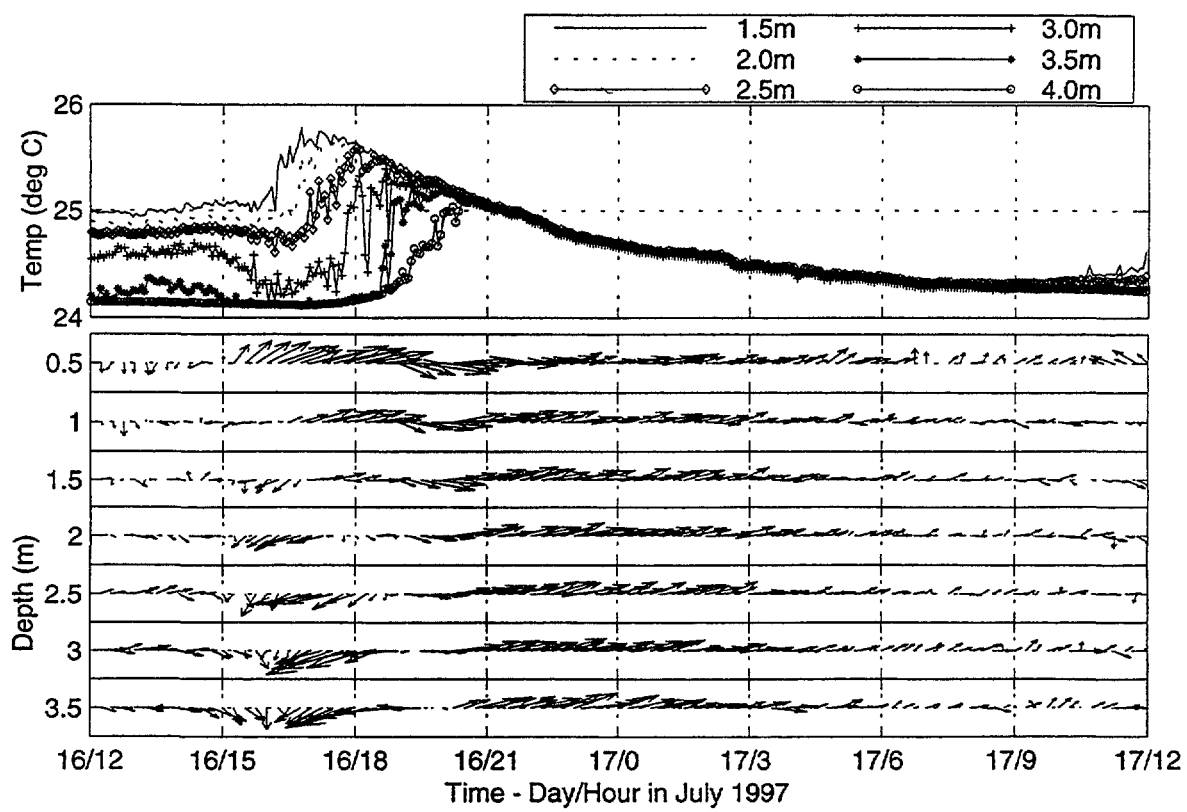


Figure 4: Nights of full mixing of the water column at the three sites during the sampling period in July 1997. A dot represents a night where full mixing occured.



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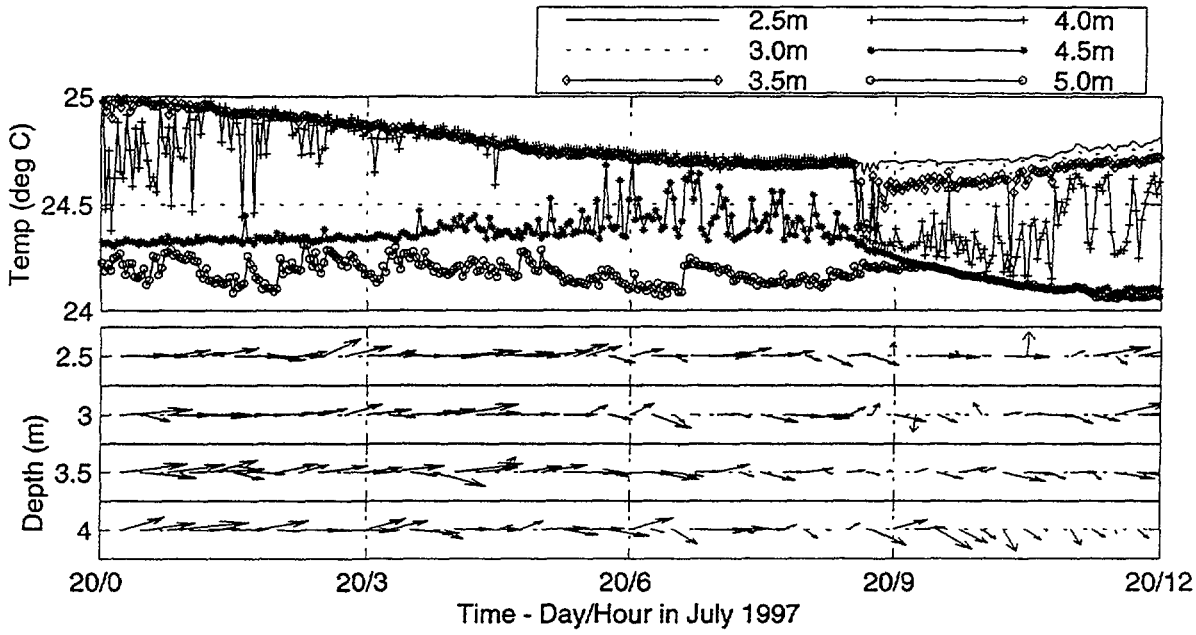

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The Phosphorus Cycle in an Iron Limited Lake

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Keywords: phosphorus, sediment cores, iron limitation, drought

Abstract-Clear Lake has been known as an iron limited phosphorus rich system since the work of Horne and colleagues in the early 1970s. The DWR water quality monitoring time series documents 3 "regimes" in the phosphorus cycle. Large amounts of P cycled in the lake on an annual basis from 1969-73. From 1974 to 1987, the amount of P in the water column in late summer-early fall was much less. Dissolved phosphate was often at or near detection limits, except in the late summer and fall peak of total P. From 1987 onwards, P levels in the lake began to increase steadily until 1991, when peak late summer values of *dissolved* P reached several hundred ppb. At peak P levels, about 80% of total phosphorus is dissolved. Algae are able to use only a small portion of the total P released from sediments. Iron limitation of nitrogen fixation keeps the lake strongly nitrogen limited, resulting in the extra-ordinarily clear-water years since 1990. We have investigated the dynamics of cycling from the sediments using grab samples and short cores. Each year, there is a large loss of iron/aluminum bound P from the sediments in the late summer and fall, corresponding with the large increase in dissolved phosphorus in the overlying water. During the winter and spring, this large mass of phosphorus is absorbed into the sediments, mostly in the upper 3-5 cm. The mass of P cycling annually has been dropping slowly since the end of the drought.

Introduction

Phosphorus is classically the limiting nutrient for plankton photosynthesis in lakes, and phytoplankton bloom situations are frequently caused by human induced increases in phosphorus loading to lakes. Treated sewage, for example, is rich in phosphorus. Many common cyanobacteria (bluegreen algae) form dense floating scums that greatly limit the attractiveness of lakes for water contact recreation and often cause taste and odor problems in water supplies. Since most scum formers fix atmospheric nitrogen, they are often able to out-compete more desirable forms of algae when phosphorus supplies are abundant but nitrogen is low. Human-caused increases in sewage or sediment loading tend to supply large amounts of phosphorus relative to nitrogen thus encouraging scum forming blue-greens.

The phosphorus cycle within lakes is normally regulated by sediment anoxia. The classic work of Mortimer (1942) showed that phosphorus is precipitated in oxygenated sediments as $FePO_4$ (ferric phosphate), whereas in reducing sediments the reduction of iron to the ferrous 2+ state solubilizes phosphate. In Clear Lake, warmth and liberal supplies of organic matter of the summer and early fall period encourage surface sediment anoxia and phosphorus release. Thus, processes of internal loading from the sediments complicate the simple picture of eutrophication caused by human activities increasing external loading.

Clear Lake is one of many lakes that are not phosphorus limited (Elser, et al., 1990). Goldman and Horne (1983) argue that steep terrain, especially in arid or semi-arid regions, will normally erode rapidly, delivering large quantities of phosphorus to lakes. Bioassay experiments conducted in the early 1970s showed that the lake is nitrogen limited and that iron limits nitrogen fixation by cyanobacteria (Wurtsbaugh and Horne, 1983). Li's (1994) experiments replicate these results. Especially during the main warm-season growth period for scum forming species, phosphorus is very abundant relative to nitrogen, favoring nitrogen fixing cyanobacteria. However, the ratio of nitrogen to phosphorus has varied substantially over the years since the Department of Water Resources has been conducting its monitoring program (1969-present). These data are analyzed in the Clean Lakes Report (Richerson et al., 1994). The years 1969-1973 where characterized by very high phosphorus in the summer growth season, with dissolved phosphorus far above the limiting concentrations of about 0.01 mg l^{-1} most of the time. The years 1974-1986 by contrast, had many months, even during the growing season, with undetectable amounts of dissolved phosphate. Peak late summer phosphorus levels did regularly exceed limiting levels, commonly leading to summer *Anabaena* blooms, although the late spring-early summer *Aphanizomenon* blooms were less intense than it earlier years. This regime, coupled with historical evidence, led the Clean Lakes Report to suggest restrictions on phosphorus loading to the lake. Control of erosion in the drainage basin should cause more severe phosphorus limitation of the cyanobacteria and reduce scum-forming populations during the summer months.

The long drought that began in 1986 had the effect of greatly increasing the internal loading of phosphorus from the

sediments. In the late summer of 1990 an unprecedented bloom of *Microcystis* occurred. From 1991 onward, the lake has had unusually clear water with small winter-spring populations of *Aphanizomenon*, and a new summer cyanobacterial dominant, *Gloeotrichia*, which has formed relatively modest scums. Levels of dissolved phosphorus in late summer reached peak concentrations 0.2-0.6 mg l^{-1}. This is the opposite trend expected from drought-caused reductions in sediment load to the lake. To investigate this puzzle, we launched a collaborative program to monitor the phosphorus cycle in Clear Lake. The results of the first part of this program were reported in the internal loading chapter of the Clean Lakes Report. The lake exhibits a version of the classic pattern of large amounts of phosphorus release from the sediments during the late summer-fall period of high microbial activity in the sediments. In the winter-early summer period, large mounts of phosphorus are re-absorbed by the sediments. Fractionation studies showed that the most dynamic pool of phosphorus is the alkaline extractable iron and aluminum bound fraction. There is good agreement between declines in sediment from the iron-aluminum bound phosphorus and increases observed in the overlying water. If phosphorus can indeed be made to limit cyanobacterial populations in Clear Lake, the lake-bottom sediments must become a significant sink rather than a source of phosphorus, and the drought induced increase in internal loading must be anomalous. This paper summarizes the results of our collaborative project for the period since early 1994 and the return to more normal rainfall.

Methods

Sediment samples were collected from the top 10 cm of Ekman dredge hauls, originally from 9 stations around the lake (Horseshoe Bend, Kelsey Creek Delta, Lower Arm East, Lower Arm West, The Narrows, Rattlesnake Island, Rodman Slough Delta, Soda Bay, and the Upper Arm). The stations were reduced in August 1994 to the 4 stations shown in Figure 1. Simultaneously, water samples were collected from the same sites. From September 1995 to present, we have collected short cores 12-28 cm long. The cores were sectioned every 2 cm to provide a fine-grained picture of internal loading from the sediments. Figure 1 shows the location of the sampling stations.

Phosphorus was analyzed using standard methods. Sediment fractions were analyzed at Hopland Research and Extension Center using the molybdate-ascorbic acid colorimetric method to determine phosphorus. Total phosphorus was estimated from perchloric acid digestions. Other fractions were determined by sequential extraction, first with an NH_4Cl solution to extract and "available" fraction, then with 0.1N NaOH to extract the "iron and aluminum bound" fraction, and finally with 0.5N HCl to extract the "calcium bound" phosphorus (Hieltes and Lijkema, 1980). The sum of the three extraction fractions was subtracted from total phosphorus to obtain a residual fraction, usually interpreted as organically bound. Water samples were analyzed using Hach kit methods for total and dissolved phosphorus based standard methods. Details of methods can be found in Richerson et al. (1994).

The data from dredge samples was reduced by taking an area-weighted average of the samples collected in each arm. The mass of phosphorus in sediments was computed assuming that 66% of the lake bottom consists of soft sediments that participate actively in phosphorus cycling.

Results

Figure 2 shows the results of the fractionation study of dredge samples. Total phosphorus and the residual show large fluctuations probably due to sediment heterogeneity, but no important long-term trends. The base extractable, Fe-Al bound fraction shows a marked seasonal cycle with drops in the last half of each year and rises in the first half. There is also an upward trend in the amount bound in this fraction over the course of the study. The acid extractable Ca bound fraction shows a weak seasonal trend in some years, especially in 1993-1994, and a considerable downward trend over the 6 year period.

Figure 3 shows the mass balance of phosphorus in the water column and in the active Fe-Al fraction in the sediments. As was evident in the earlier years analyzed in the Clean Lakes Report, there is a very good mass balance. Phosphorus release from this fraction appears in the overlying water with close agreement in timing and tonnage. It is also notable that the rising multi-year trend of phosphorus in the sediment Fe-Al fraction is mirrored by a decline in the mass cycling in the water column.

Figure 4, using DWR data for the years 1969-1992, shows the total mass of phosphorus cycling in the water column each year (late summer-fall maximum minus winter-spring minimum for each calendar year). The drought related peak of phosphorus cycling in the late 1980s and early 1990s is the most striking feature of the record. The trend toward a return to the conditions characteristic of 1974-86 is evident during the wetter years of the 1990s.

Figure 5 shows the behavior of phosphorus fractions in the cores. It is evident that the top 4-5 cm of the sediments account for most of the activity on the annual cycle. Even in the case of the dynamic Fe-Al bound fraction, only the top few cm participate in the cycle. This is consistent with work by Mack and Nelson (1997) showing that microbial activity is much more intense in the upper few cm of the sediments. O_2 and SO_4 diffusing downward from the overlying water provide oxidizing power for microbial metabolism near the surface. Fermentative metabolism deeper in the sediments is much less vigorous. It is especially notable that there is no evidence of a phosphorus-depleted zone below the active upper layers where phosphorus is deposited in the winter.

Discussion and Conclusions

The Clean Lakes Report interpreted the high phosphorus cycling in the water column as due to recent watershed disturbances, especially gravel mining in creek channels. Unfortunately, there is no contemporary data on sedimentation rates or sediment processes. Long core phosphorus levels are relatively uniform, and we have not discerned any trends to critically examine this interpretation (Becker et al., 1997). The sharp rise in cycling phosphorus during the drought years must have been dominated by internal loading processes as for water years 1987-1992 there was essentially no external load. The lack of any information on sediment processes until the drought maximum years make interpretation of events tenuous. The lack of a marked phosphorus depleted zone in the sediments below the active surficial layer speaks against the upward mobilization of phosphorus from deeper in the sediments. The end of the drought has caused a substantial drop in the mass of cycling phosphorus. There has been a rising trend of the Fe-Al bound fraction. Substantial amounts of the phosphorus have migrated from the Ca bound to the Fe-Al bound pool, all suggesting that conditions have favored a tendency of the Fe-Al binding process to become more intense since 1991. According to the classic model, this should result from the sediments becoming more oxidizing. Three classes of hypotheses suggest themselves. (1) Smaller supplies of organic matter due to the relatively clear water since 1991 may have resulted in less intensely reducing sediments. (2) The supply of electron acceptors to the sediments may have increased. Oxygen changes are not likely to be significant, but sulfate is an important source of oxidant to the sediments (Mack and Nelson, 1997), and sulfate loading to the lake appears to respond to rainfall. Low sulfate supply during the drought may have increased sediment anoxia and hence phosphate recycling. (3) Highly alkaline lakes are commonly rich in phosphorus and poor in iron (Evans and Prepas, 1997). Perhaps alkalinity increases during the drought drove the increase in P and strong iron limitation of nitrogen fixation and photosynthesis observed during this regime. Clearly, a better understanding of sediment processes tending to mobilize phosphorus and iron from Clear Lake sediments is required to understand nutrient dynamics and the regulation of phytoplankton biomass in this system.

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Figure 1. Map of Clear Lake showing locations of sediment sampling

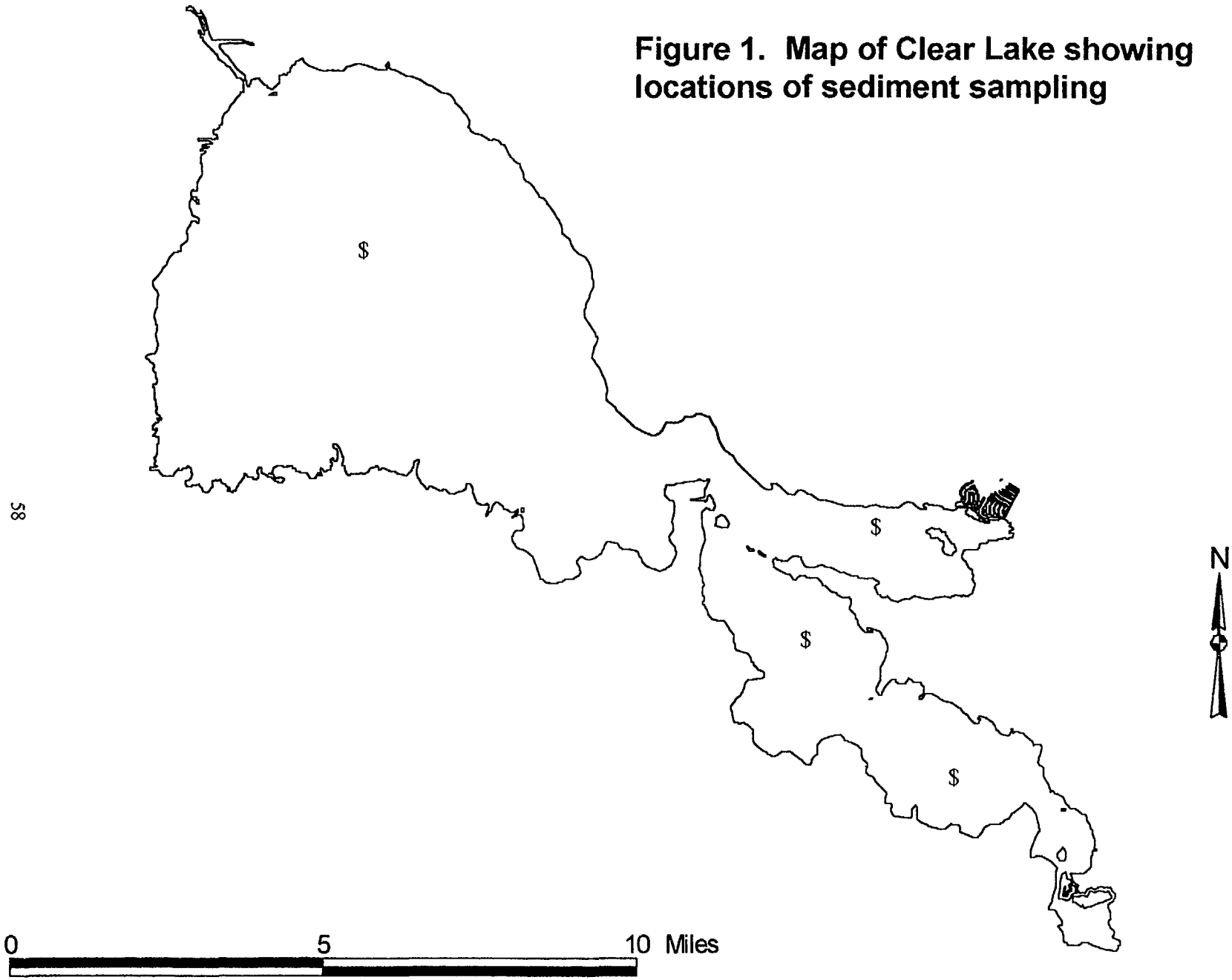


Figure 2a: Phosphorus concentrations in the iron/aluminum and calcium bound fractions in Clear Lake sediments, 1991-1997

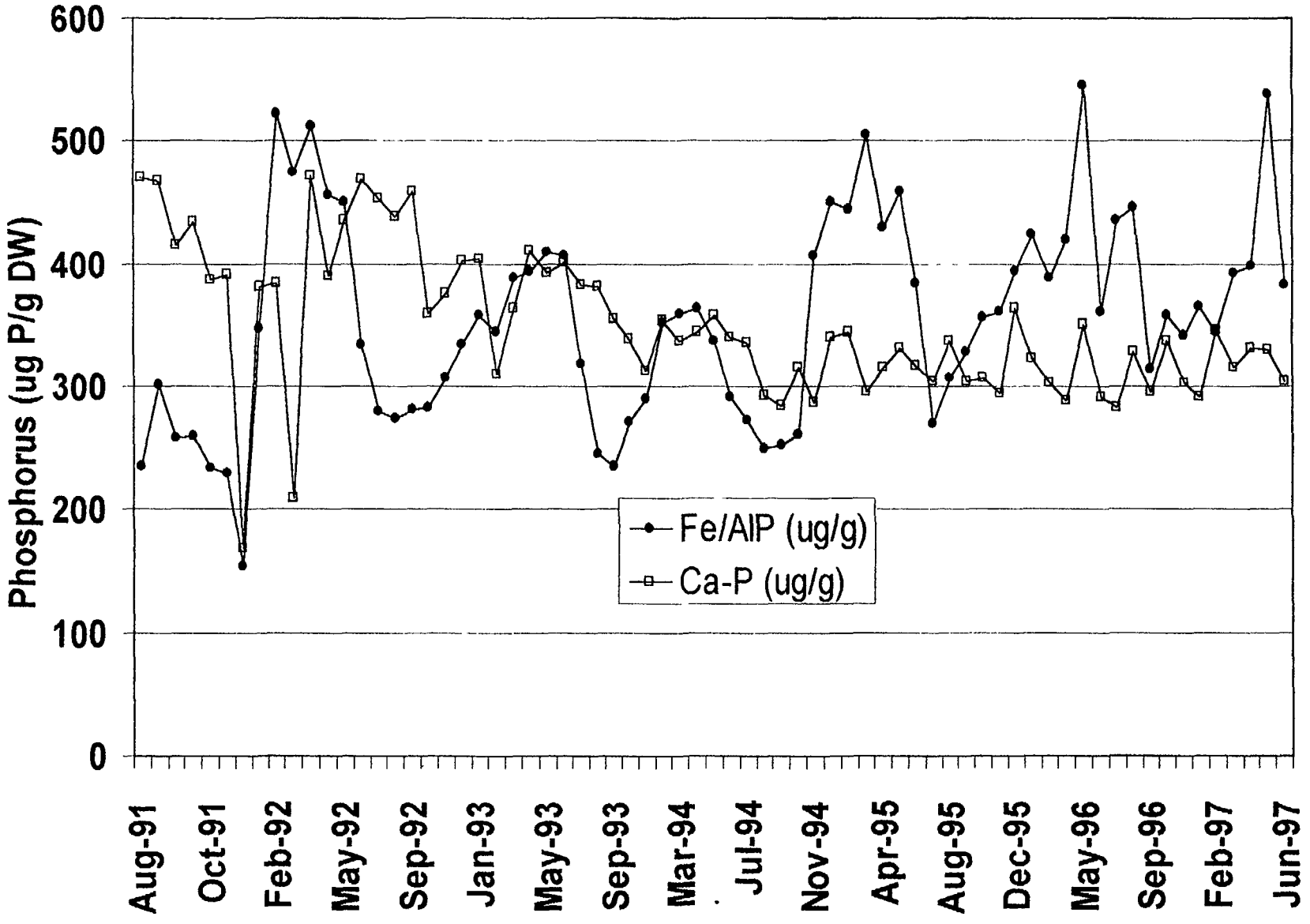


Figure 2b. Phosphorus Concentrations residual (organic) and total fractions in Clear Lake sediments, 1991-1997

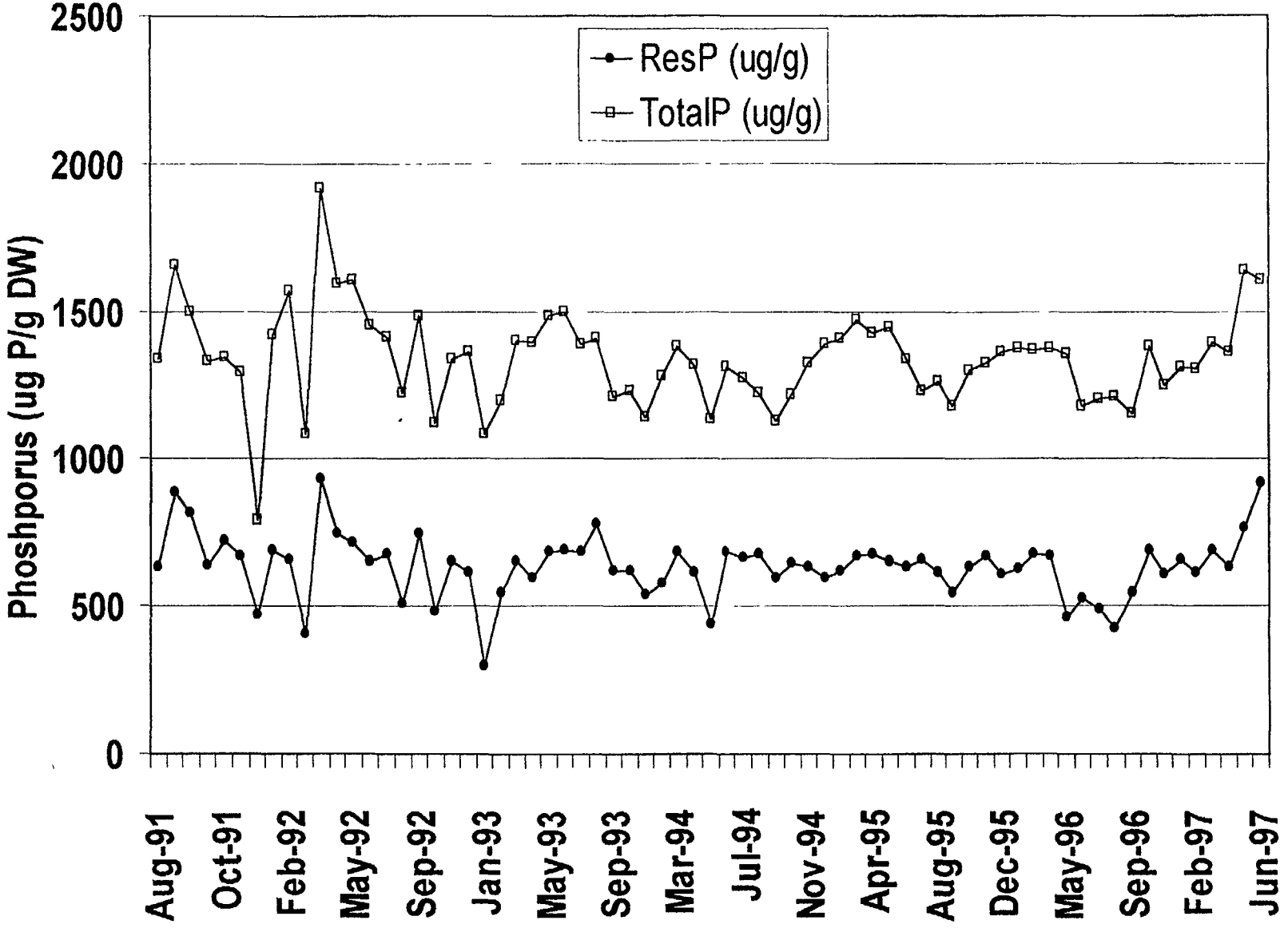


Figure 3. Mass of Total Phoshprus in Water Column and Fe-Al Bound Fraction of Sediments

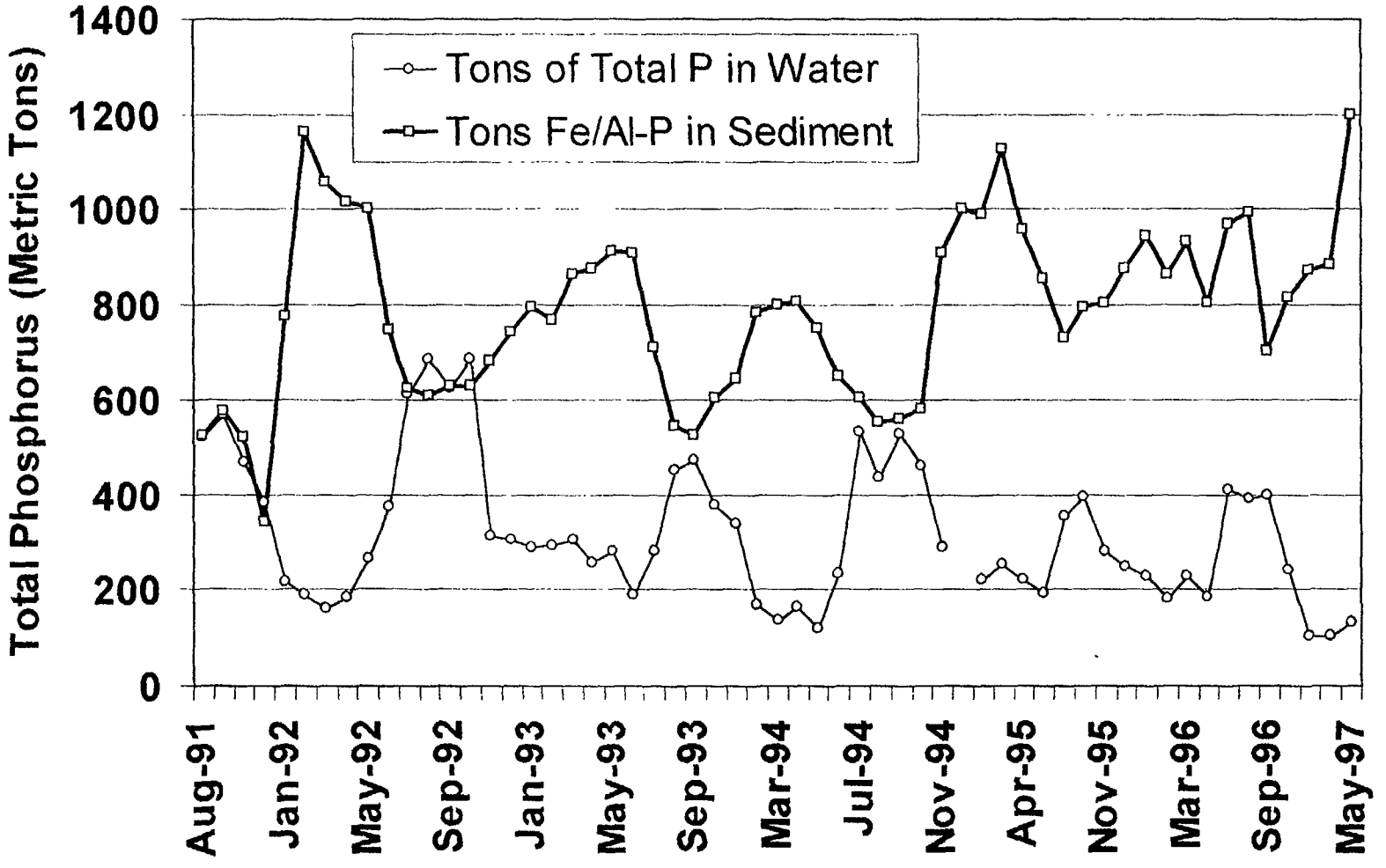


Figure 4. Total Phosphorus Flux From the Sediments to the Water Column Per Annum in Metric Tons

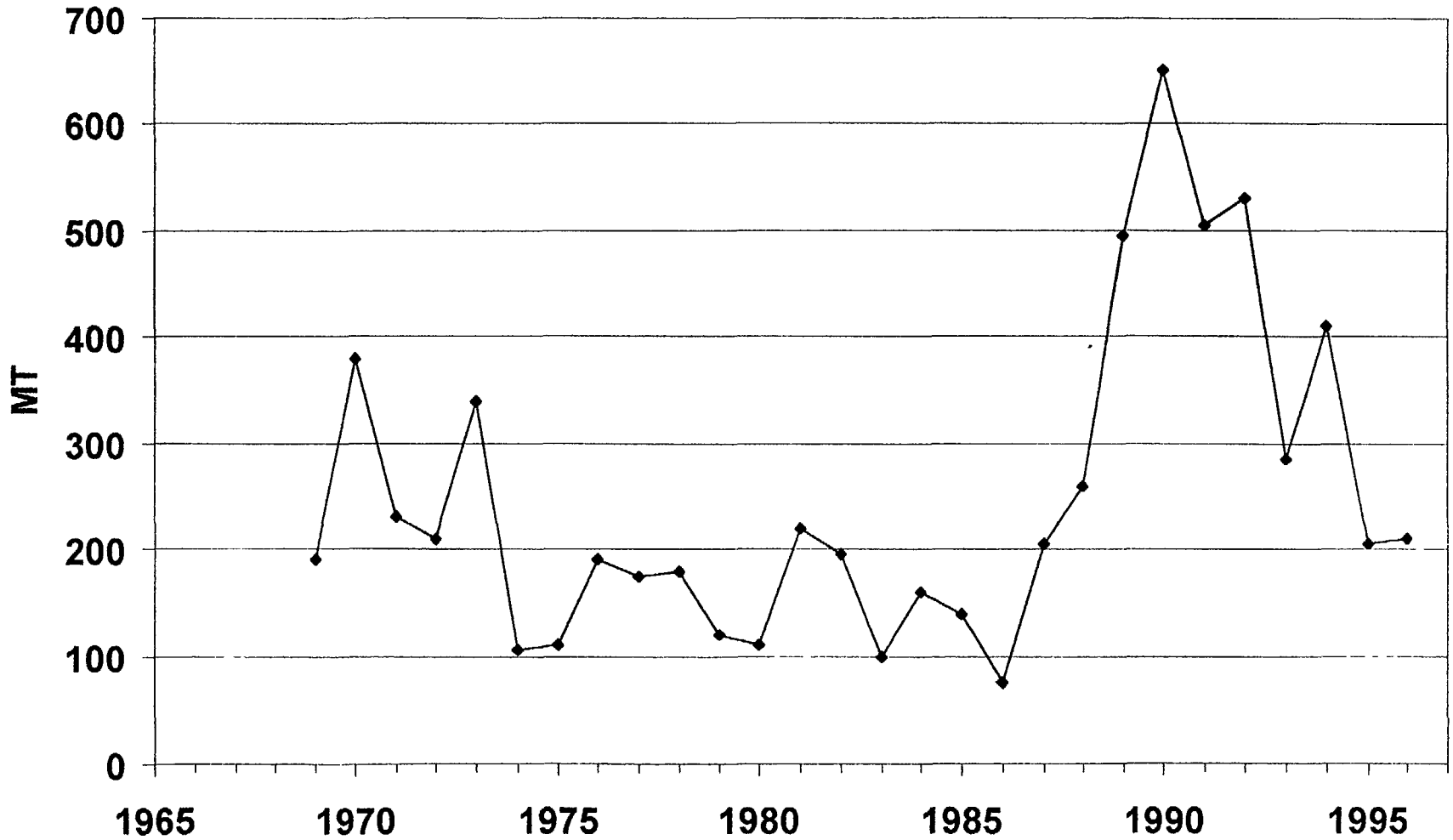
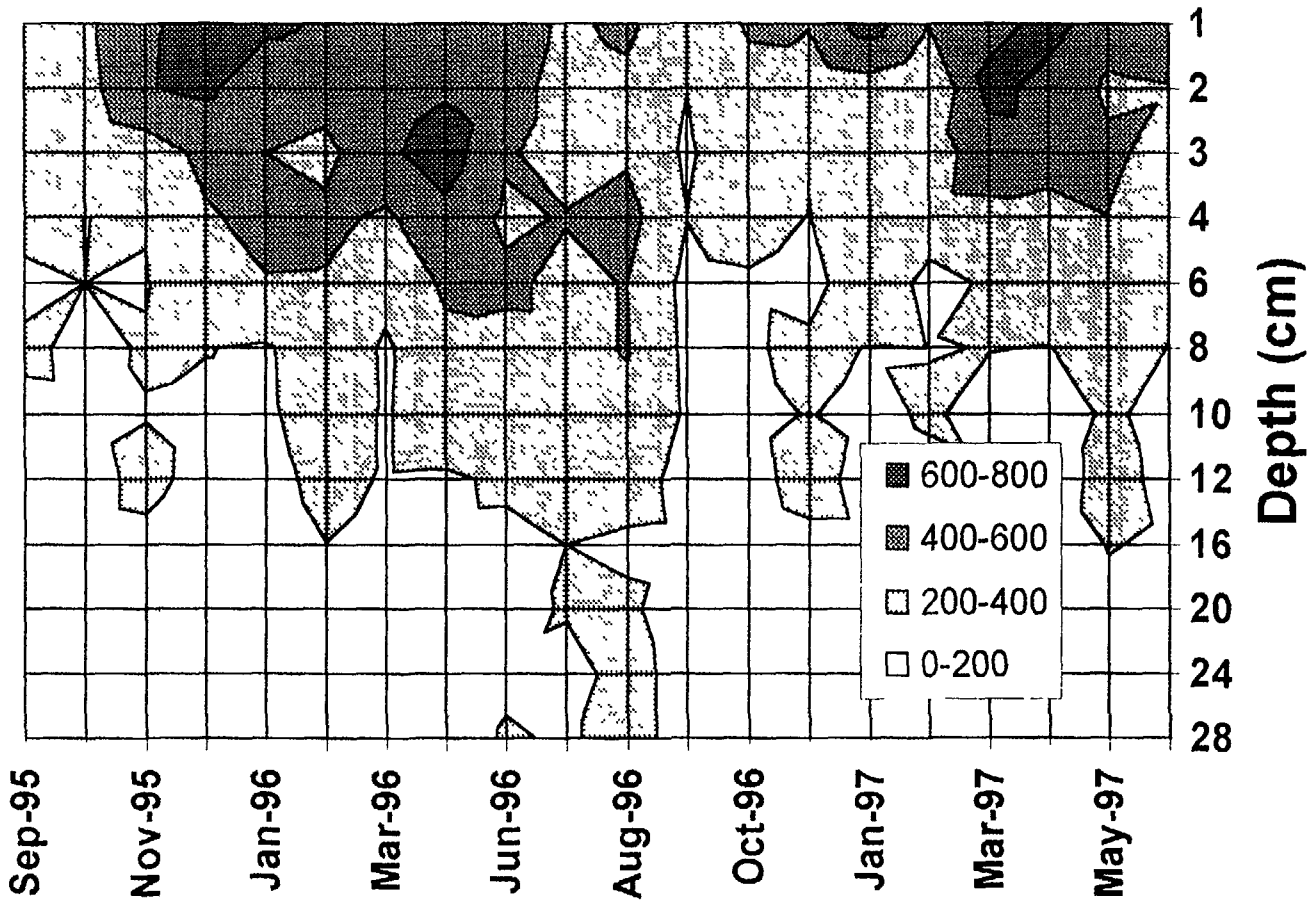


Figure 5. Contour plot showing the distribution of the iron/aluminum bound phosphorus fraction as a function of depth in sediments, ug/g dry weight



3

SULPHUR BANK MERCURY MINE AND RELATED PROCESSES

The Use of X-ray Techniques to Measure Elemental Concentrations in Feathers. (P)

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Key Words: X-ray Fluorescence (XRF), Particle Induced X-ray Emission (PIXE), bird feathers, elemental analysis, metals

The analysis techniques of X-ray Fluorescence (XRF) and Particle Induced X-ray Emission (PIXE) provide an efficient alternative to atomic absorption for conducting multi-elemental analyses. Feathers have not been extensively analyzed using either of these methods although the thickness of the feathers is almost ideal for these systems. This project will test the effectiveness of these techniques in conducting elemental analyses on feathers and then apply the techniques to the Clear Lake Ecosystem Project.

Relationships Between Mercury and Yearly Trends in Osprey Production and Reproductive Status at Clear Lake. (P)

D.W. Anderson, T. M. Cahill, Jr., T. H. Suchanek, and R. A. Elbert

Abstract

Observations of osprey productivity (as measured by fledgling success on a per-nest and/or sub-population basis) were collected during the summers from 1992-1996. During the summers of 1992-1993, osprey productivity exhibited a significantly positive relationship as a function of distance from the Sulphur Bank Mercury Mine, suggesting some negative relationship to environmental mercury concentrations. During 1994 all osprey nests and sub-populations improved their productivity dramatically, yet there was still evidence of a relationship between fledgling success and distance from the mine, although this relationship was not statistically significant. Osprey productivity during the summers of 1995-96 exhibited a similar trend. Mercury concentrations in osprey feathers collected during the 1995-96 breeding years exhibited no relationship to distance from the mine. Monitoring normal year-to-year variability (baseline data) in osprey productivity levels and mercury residues will be important to determine how permanent these changes are.

Introduction

Osprey (*Pandion haliaetus*) are likely, along with western and Clark's grebes (*Aechmophorus occidentalis and clarkii*), to be among the top avian piscivores in the Clear Lake ecosystem. These three species are known to bioaccumulate residues of mercury (Hg) and to be among the most highly contaminated birds at Clear Lake (Elbert and Anderson in preparation; Cahill et al. in preparation). One commonly observed effect in birds affected by various forms of Hg is impaired reproduction. The measurement of reproductive rates is also useful in that it measures a parameter of interest in evaluating population-level effects of birds in contaminated systems. Reproductive rates and population status are difficult to interpret because they are subject to so many normal environmental variables, such as weather, food supply, and disturbances (Poole 1989). When compared to known critical Hg levels from other field and laboratory studies it was previously shown (Elbert 1996) that grebes from Clear Lake collected in 1992 carried total Hg levels that were near critical-effect levels in some instances, and there was no reason to assume otherwise for osprey, which were similarly or even more contaminated than grebes (Cahill et al. in preparation). Earlier studies of sediments, fish and invertebrates also showed a general decreasing trend in Hg levels as a function of distance from the Sulphur Bank Mercury Mine (the mine) (Suchanek et al. 1993, 1995, 1997). Osprey are known to be rather sedentary and to utilize feeding areas relatively near their nest sites (Poole 1989). It follows that piscivores such as osprey, utilizing various parts of the lake, should also then show indications of residues or potential residue effects away from the mine site.

Methods

Osprey nests around the perimeter of Clear Lake were sampled for productivity using the number of successful fledglings as a proxy for productivity. Figure 10B-1 indicates the location of active nest sites used in the osprey productivity study during 1995/96. Data on fledgling success were collected during the summers of 1992, 1993, 1994, 1995, and 1996.

Results of Clear Lake Osprey Productivity Studies

At the time our study was initiated, our working hypothesis was that osprey should exhibit residue patterns or effects as a function of distance from the mine, using their reproductive rates as a measurement criterion for effects. Although osprey were not sampled for Hg from 1992-1994 (in order to avoid potential disturbance-causing effects on reproductive rates in this state-designated sensitive species), their subpopulation and individual nest productivity rates in 1992-1993 varied significantly with distance from the mine (Elbert 1996) (Fig-

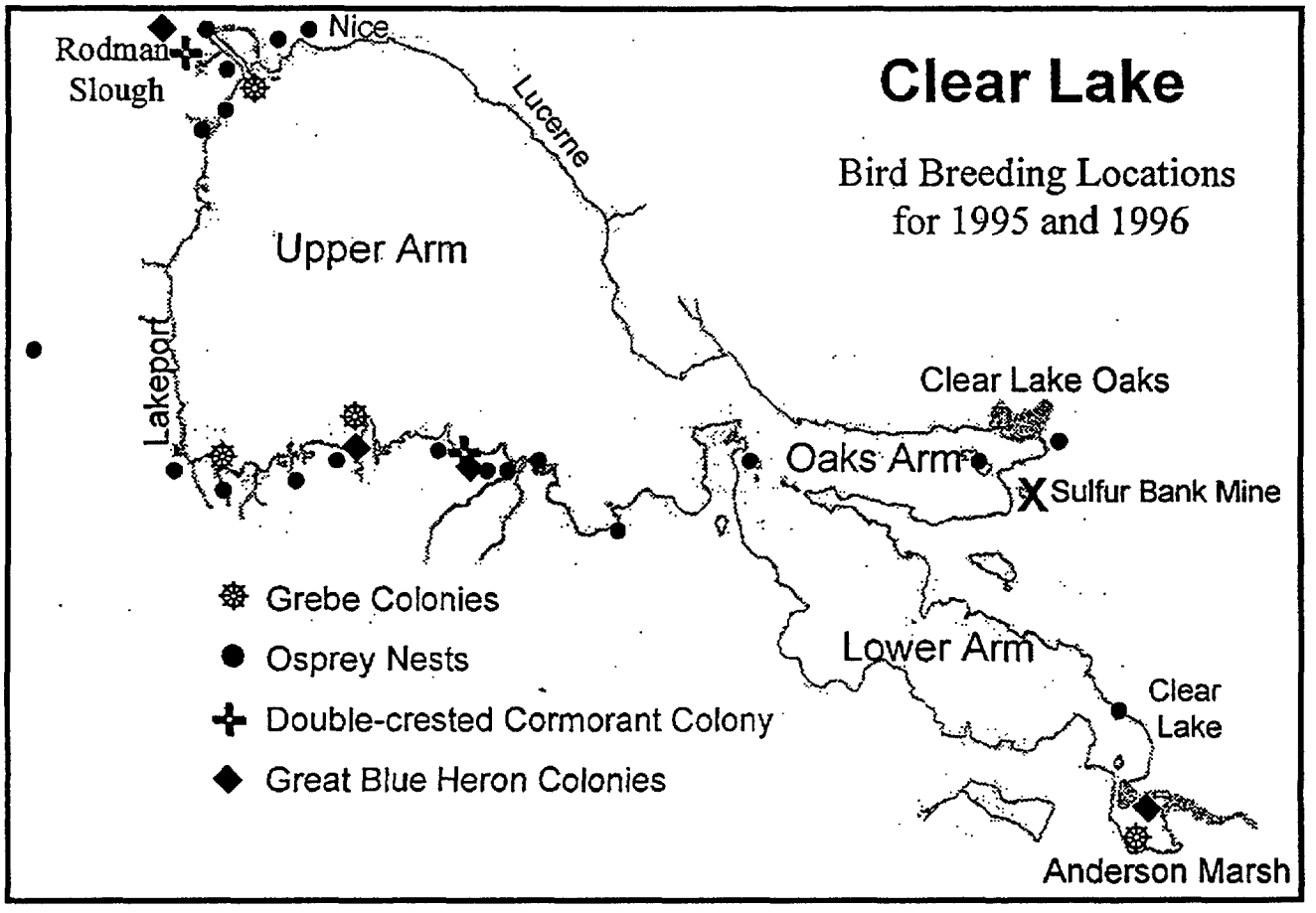

Figure 1. Active nest locations for osprey, western grebes, great blue herons and double-crested cormorants at Clear Lake in 1995 and 1996.
ure 2, $p < 0.02$).

Several significant events occurred in the Clear Lake system in 1992-1993 and in the winter of 1994/1995 to potentially alter these relationships. First, the USEPA remediated (reduced the slope of) waste rock piles at the mine along with revegetation on the slopes to reduce Hg-laden sediment input into the lake. Second, the lake experienced a period of unusually high water levels, potentially affecting turnover rates, movements, and environmentally induced change rates in various pollutants in the lake, including Hg compounds. The distribution of Hg residues in fish were generally more uniform after that period. A second working hypothesis was therefore developed that predicts a breakdown in the relationship between residues and effects in relation to distance from the point source at the mine.

In 1994, the relationship between osprey productivity and distance to the Sulphur Bank mine was not significant (Figure 3, $p > 0.10$), suggesting a changing trend between these two variables. Figure 4 summarizes the productivity data for sub-populations (as opposed to individual nests) of osprey at Clear Lake as a function of distance from the mine. It shows that all of the 1994 sub-populations had higher productivity than the previous two years, and that there was a flattening of the relationship between productivity and distance from the mine.

In the years 1995-96, we sampled feathers from accessible pre-fledging osprey at Clear Lake and continued to examine the previously shown relationship between productivity and distance to the mine (Figure 5, $p > 0.10$) which showed a very similar pattern to that observed in 1994. In addition to minimal population effects being documented during these two years, it was also observed that Hg residues exhibited no relationship to distance from the mine (Figure 6, $p > 0.10$).

We will attempt to obtain more feather samples from both osprey and grebes in 1997 to determine if these relationships continue. Largemouth bass (*Micropterus salmoides*), which are consumed by osprey, also show relatively uniform Hg residues throughout Clear Lake.

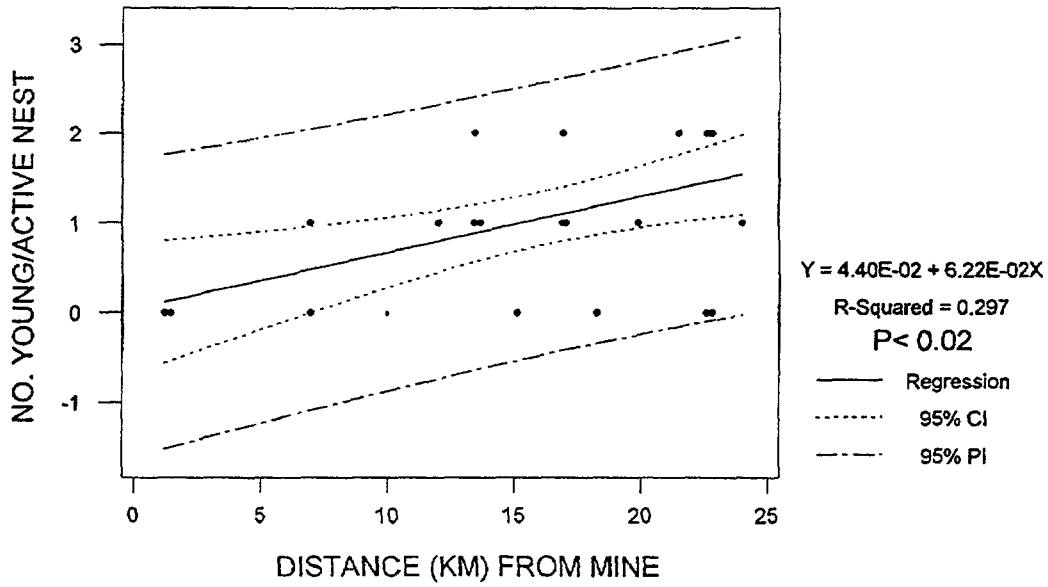

Figure 2. Productivity (osprey young fledged at Clear Lake as a function of distance from the Sulphur Bank Mercury Mine, 1992-1993).

Discussion

To date, the precise reasons for these observed changes are only speculative.

(1) It is possible that the USEPA 1992 remediation of the shoreline waste rock piles was indeed effective and since this population was at or near an effect threshold, it showed a rapid response to remediation.

(2) It is possible that presumed residue change patterns were related to changes in the system due to the flooding conditions during the winter of 1994/1995.

(3) It is also possible that none of the observed changes had anything to do with Hg effects, but rather some

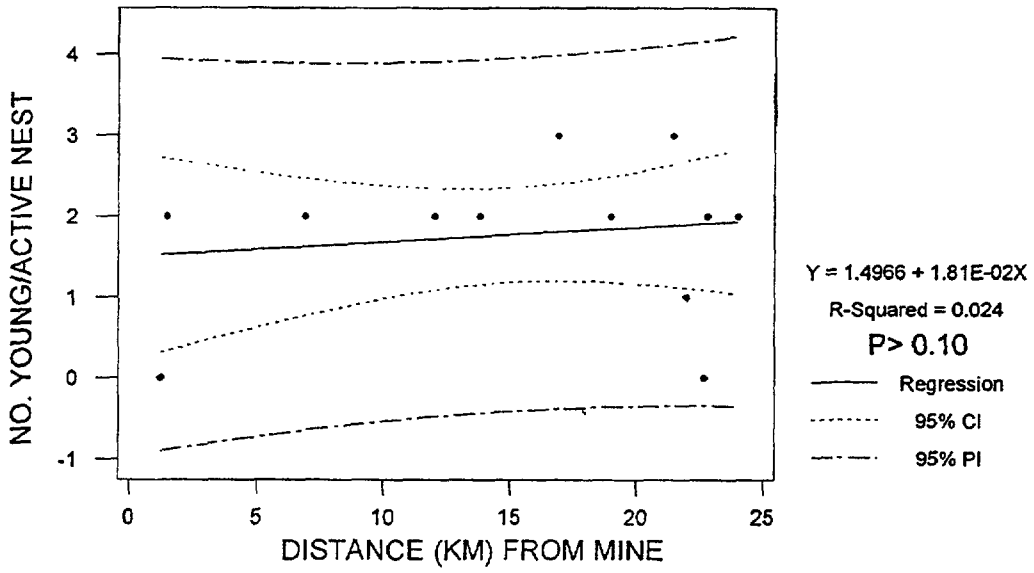

Figure 3. Productivity (osprey young fledged at Clear Lake as a function of distance from the Sulphur Bank Mercury Mine, 1994).

OSPREY FLEDGELING SUCCESS RATE AT CLEAR LAKE

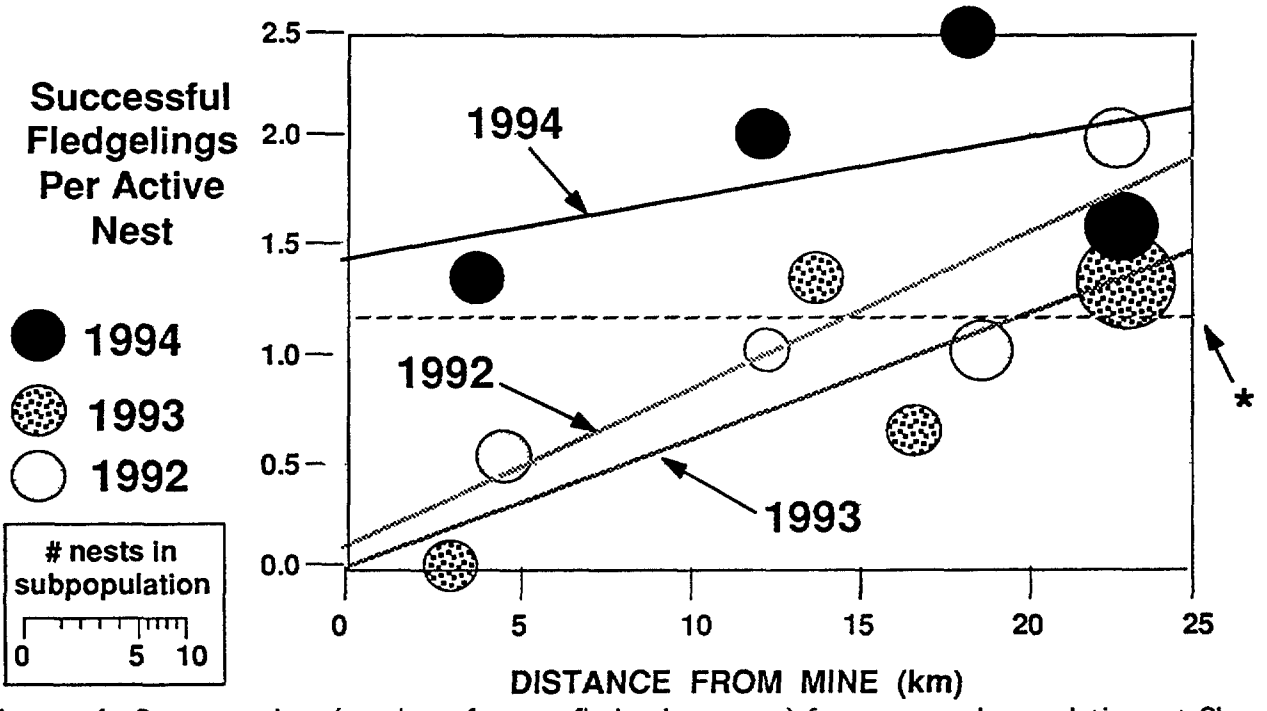

Figure 4. Summary data (number of young fledged per nest) for osprey sub-populations at Clear Lake as a function of distance from the Sulphur Bank Mercury Mine, 1992-94. The size of each bubble indidcates the number of nests in that subpopulation (see scale at left). Asterisk indicates long term replacement value of 1.2 young/nest needed to maintain population viability.

other ecological change induced by the flooding (such as changes in the distribution and abundance of large prey fish).

Data from 1997 will be important to indicate if these changes might be more permanent. Population related changes, even in normal populations, are often difficult to interpret due to multiple-factor environmental varia-

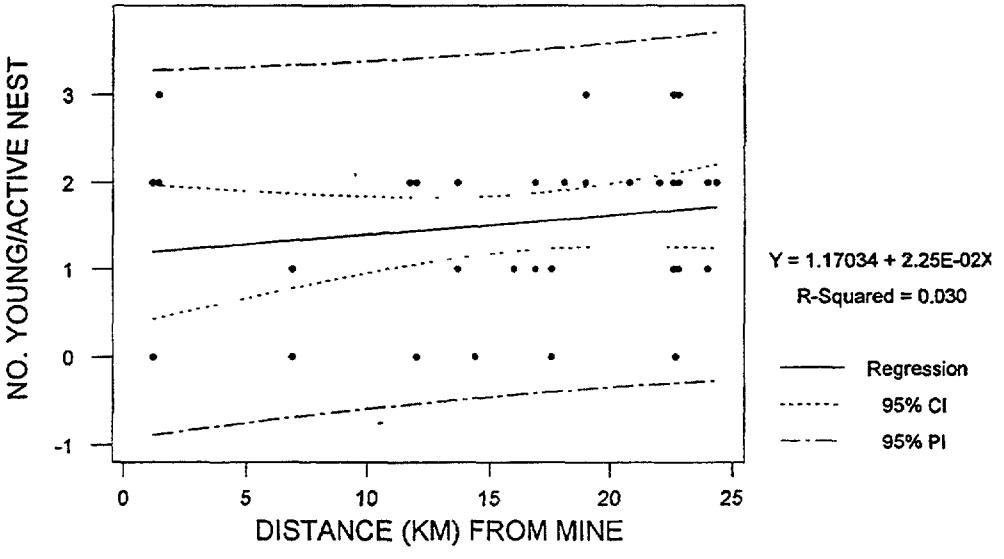

Figure 5. Osprey productivity (number of young fledged per nest) at Clear Lake as a function of distance from the Sulphur Bank Mercury Mine, 1995-96.

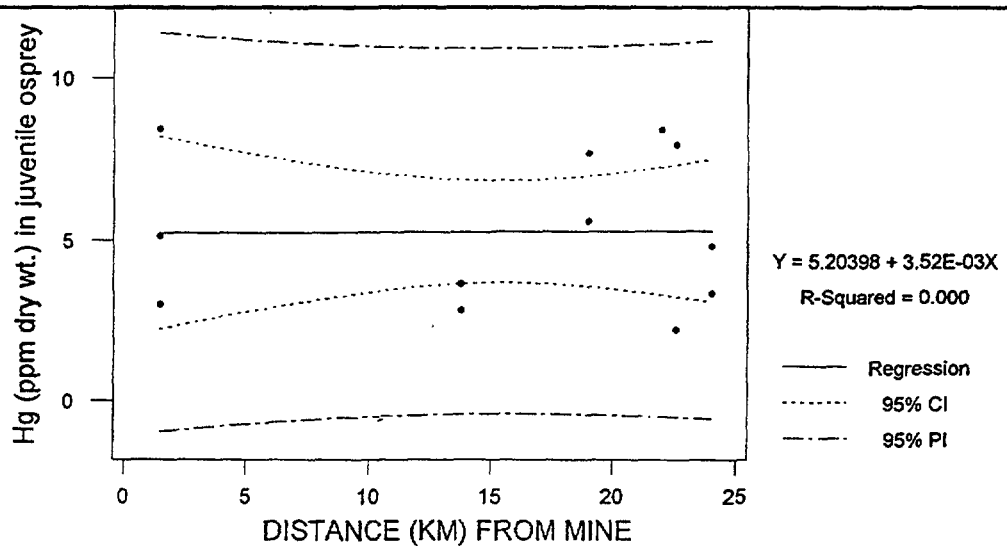

Figure 6. Mercury (ppm) in osprey feathers as a function of distance from the Sulphur Bank Mercury Mine, 1995-96.

tion. Overall, the nesting population of ospreys at Clear Lake has generally increased from 10 active pairs in 1992, 17 pairs in 1993, 14 pairs in 1994, 17 pairs in 1995, to 20 pairs in 1996. Most importantly, these results stress that there is a need for long-term population monitoring and that normal year-to-year variability (baseline data) needs to be understood.

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Concentrations of Mercury and Other Elements in Five Species of Bird at Clear Lake

T.M. Cahill, Jr., D.W. Anderson, B. Perley, and T. H. Suchanek

Abstract

Feather samples from five bird species were collected at Clear Lake, CA in order to determine the extent of mercury contamination. The results clearly showed that the osprey at Clear Lake have elevated concentrations of mercury when compared to control sites, both within California and in other states. Mercury contamination was greatest in bird species from higher trophic levels. Adult osprey had about 3.8 times higher mercury concentrations than juvenile osprey, thus suggesting strongly that the adult birds accumulate mercury over their lifetime.

Introduction

Clear Lake, CA, has long been known for its mercury (Hg) contamination (Chamberlin et al. 1990, Suchanek et al. 1993); but does Clear Lake wildlife show elevated Hg levels and, if so, does this Hg contamination significantly impact the wildlife in Clear Lake? We collected feather samples from five bird species at the lake and analyzed them for Hg and other elements using an X-Ray Fluorescence system (Cahill et al. in preparation). In particular, we emphasized high trophic birds, such as osprey (*Pandion haliaetus*) and western grebes (*Aechmophorus occidentalis*), that would likely bioaccumulate the highest concentrations of Hg (Jernelov and Lann 1971, Hamelink et al. 1977). We also surveyed the reproductive success of osprey and western grebes at Clear Lake and at control sites to determine if the reproductive rates differ between Clear Lake and the control sites.

Methods

Sample Collection

Elemental concentrations in feathers have been used as an index of the exposure or body burden for birds at the time of feather growth (Browerman et al. 1994, Burger and Gochfeld 1995). Since birds actively excrete metals and other elements into growing feathers (Goede and de Bruin 1984), the elemental concentration in the feather would be higher, and hence easier to detect, than the metals present in blood or other tissue samples (Honda et al. 1986). Feather sampling also provides an easy non-invasive method for quantifying the exposure of a bird to elemental contamination (Cahill et al. in preparation).

Flight feathers were used as the principal samples during the study. The tip of a distal flight feather, about 5 to 7 cm long, was utilized for the sample. If the sample was removed from a dead bird, then the entire feather was collected, but only the tip of the distal flight feather was used for analysis.

Feather samples from osprey were collected in order to determine the extent of Hg contamination at Clear Lake, CA (Figure 1). One flight feather was clipped from 12 prefledgling osprey from around the lake. An additional 12 flight feathers, molted from adult osprey, were

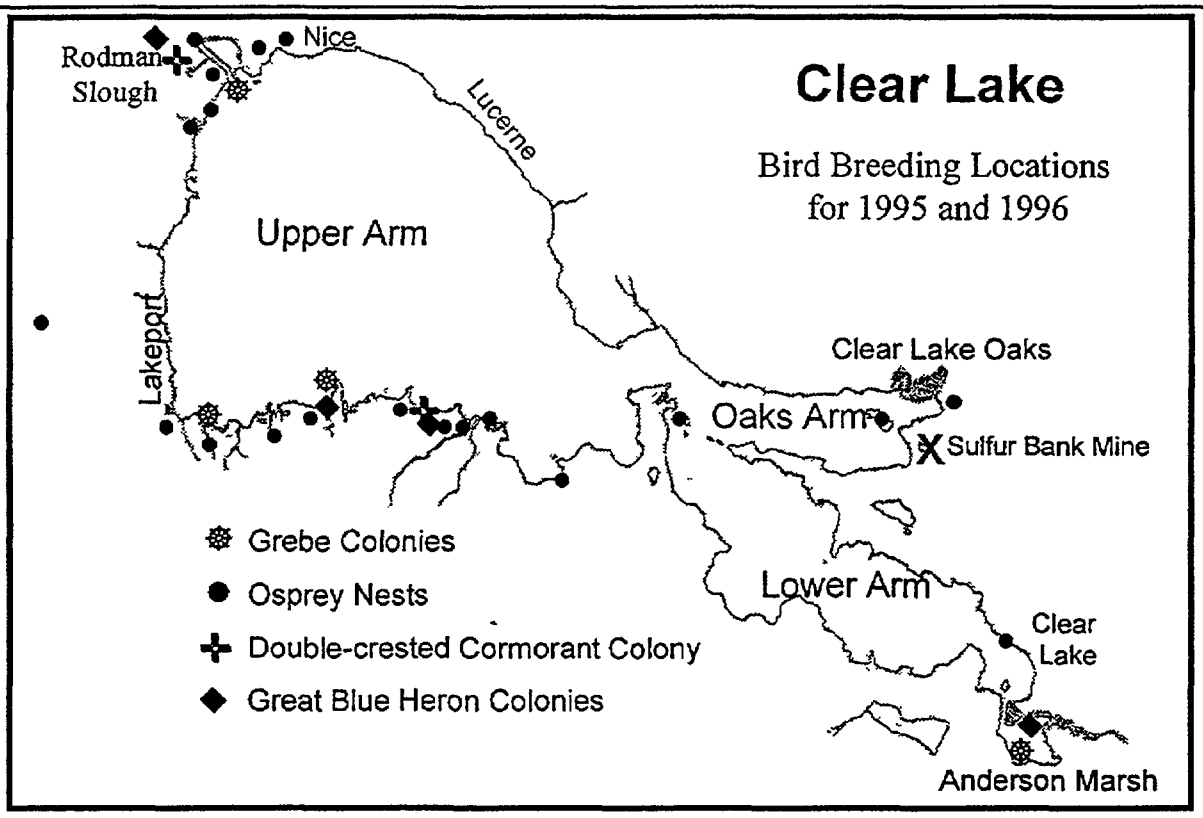

Figure 1. Breeding locations of osprey, grebe, double-crested cormorant, and great blue heron at Clear Lake, 1995 and 1996.

collected from under nest sites. Although we cannot be certain where an adult osprey was when it grew the feather, we are assuming that it was in the same nesting territory, hence the osprey was exposed to the same environment the previous year. Osprey have high nest-site fidelity; approximately 97% of the birds return to their nests the following year (Poole 1989), so this is a reasonable assumption. The osprey generally undergo wing molt while on or near the nest site especially, in non-migratory or weakly migratory populations such as Clear Lake (unpublished data). Radio-telemetry data (unpublished) from 4 fledged osprey indicated that some of the Clear Lake fledglings may winter within California.

Osprey feathers were also collected from three control sites. These sites were: 1) St. Joe, Idaho (n=18 juveniles), 2) Coeur d'Alene, Idaho (n=18 juveniles), and Bahía de Los Angeles, Baja California (n=5 juveniles and 6 adults). Once again, a single flight feather was collected from the juvenile osprey and used for the sample. Samples from adult osprey were collected as molted flight feathers.

In order to examine potential trophic differences in Hg concentrations, we collected samples from four other species at Clear Lake, CA. Feather samples were collected from adult western grebes (n=12), adult great blue herons (*Ardea herodias*, n=19), juvenile double-crested cormorants (*Phalacrocorax auritus*, n=9), and adult turkey vultures (*Cathartes aura*, n=13). The western grebe feathers were removed from carcasses of individuals collected in 1992 (Elbert 1996). Many of these adult grebes were in the process of molting at the time of collection. The feather samples from the great blue herons were collected as molted flight feathers under the heron colony in Anderson Marsh (Figure 1). An additional 16 flight feathers were collected from under a heron breeding colony in Rodman Slough in 1996, but these samples have not

yet been analyzed. The double-crested cormorant samples were opportunistically collected from dead juvenile carcasses found under the main breeding colony on the south side of the Upper Arm of Clear Lake. Turkey vulture feathers were collected under roosting sites in Anderson Marsh.

Sample Preparation and Analysis

After collection, the samples were washed for 30 seconds in three alternating baths of distilled water and acetone, starting with the distilled water. After washing the feather, a section of the vein near the tip of the feather was cut away from the shaft. In order to mount the sample in a plastic 35mm slide frame, a 20 X 25 mm section of the vein was removed from the feather intact.

The samples were analyzed using an X-Ray Fluorescence system at Crocker Nuclear Laboratory at the University of California at Davis (Cahill et al. in preparation). X-Ray Fluorescence operates by removing the low energy electrons from the atoms in a sample with an X-ray beam. Once a hole in the low energy shell has been created, then another electron falls from a higher energy state into the lower energy state, thus losing energy. The excess energy is emitted in the form of X-rays whose profile is unique for each element (Johansson et al. 1976). By using a silicon-lithium detector, the energy of the X-rays radiated from the sample are determined. The X-rays emitted from the sample generate a spectrum with the peaks corresponding to different elements.

Crocker Nuclear Laboratory's X-Ray Fluorescence analysis system used an 17.5 KeV X-ray beam produced from a molybdenum source. The samples were analyzed for 10 minutes with a beam current of $12X10^{-6}$ amperes. The X-Ray Fluorescence system sampled a 1 cm by 1.8 cm elliptical section of the feather. A section of "Whatman 41", a clear cellulose filter material with a density of about 5 mg/cm^2, was used as a background subtraction. The Whatman 41 sample was able to determine the extent of background noise resulting from the analysis system when measuring a pure CHO target that is a similar thickness as the feathers.

After the X-Ray Fluorescence analyses were conducted, the four osprey feathers with the highest Hg concentrations were sent to Battelle Northwest Laboratories, Sequim, WA, for further analysis. Battelle used the methods of Roesiyadi (1982) to determine the total Hg present in the feathers. The methods of Bloom (1989) were used to determine the concentration of methyl Hg in the feathers.

Results

Determination of elemental contamination

We were able to detect 15 different elements in the 24 osprey flight feathers collected at Clear Lake, including Hg which was the focus of the study (Table 1). The Hg levels in osprey feathers, adult and juvenile combined, showed a mean value of 8.92 ppm (SD = 2.25) although the values ranged from 2.83 ppm to 46.5 ppm (Figure 2). As shown in Table 2, these Hg values are statistically higher than the three control sites at the $p<0.0001$ level (pairwise

Table 1. Mean elemental concentrations (in ppm) and their standard deviations/ranges (SD/R) detected in five species at Clear Lake, CA. Chlorine, manganese, copper, and potassium are not reported due to interference with other peaks.

	Osprey (n=12 adult &12 juv.)		Western Grebe (n=12 adults)		Great Blue Heron (n=19 adults)		Turkey Vulture (n=13 adults)		Juv. Double-crested Cormorant (n=9)	
Element	Mean[1]	SD/R[1]	Mean[1]	SD/R[1]	Mean[1]	SD/R[1]	Mean[1]	SD/R[1]	Mean[1]	SD/R[1]
Sulfur	33245	2743	33449	2642	32606	2266	29048	2900	32653	2548
Calcium	1793	615-5905	757	373-2201	2685	1397	1199	407-6265	1343	761
Titanium	15.4	mdl-118	12.3	mdl-55.8	12.2	mdl-176	40	14.7-169	8.34	mdl-89.0
Vanadium	trace[b]	mdl-4.95	ND	—	ND	—	ND	—	ND	—
Chromium	trace[b]	mdl-3.80	ND	—	ND	—	ND	—	ND	—
Iron	278	203	66.4	24.5-511	224	43.0-1904	525	110-2227	366.8	308
Nickel	0.82	mdl-2.65	trace[b]	mdl-0.96	trace[b]	mdl-3.57	trace[b]	mdl-4.95	trace[b]	mdl-0.29
Zinc	163	27.4	164	41.6	263	68.0	186	46.3	164	26.3
Arsenic	trace[b]	mdl-1.22	1.02	mdl-3.55	trace[b]	mdl-0.87	traceb	mdl-1.62	trace[b]	mdl-0.58
Selenium	3.07	mdl-22.3	1.38	mdl-2.13	2.49	1.03	1.1	0.14	4.69	0.91-35.7
Bromine	19.8	mdl-142	42.6	36	10.6	6.16	5.32	3.45	34.2	29.7
Rubidium	trace[b]	mdl-3.0	trace[b]	mdl-1.75	0.83	mdl-5.26	trace[b]	mdl-4.98	trace[b]	mdl-1.99
Strontium	5.74	mdl-50.1	4.5	mdl-16.2	16.5	10.4	7.74	2.94-28.3	8.03	7.53
Mercury	8.92	2.25-46.5	9.75	4.99-20.7	7.4	3.43-30.7	1.49	mdl-20.3	6.48	2.81
Lead	trace[b]	mdl-2.56	trace[b]	mdl-1.53	1.5	mdl-3.91	2.56	2.0	trace[b]	mdl-0.93

[1] Arithmetic means (+/- SD) are given for normally-distributed data sets (Anderson-Darling normality test $p > 0.01$); if the data is not normal, then the geometric means and ranges are given.

* The element was detected in less than half of the samples from that species, hence the mean value incorporates a large number of nondetections where half the mdl value was used for a nondetection value.

ND = Not detected in any samples from that data set.

Mann-Whitney tests). Osprey samples from the control sites in Coeur d'Alene and St. Joe river drainages in Idaho showed consistently low levels of Hg in the samples with average values of 2.17 (SE = 0.23) and 2.25 (SE = 0.18) respectively. The osprey samples from Baja California also showed sporadically low Hg levels, typically below the detection limit. These samples were statistically different from the other groups at the $p<0.02$ level (paired Mann-Whitney tests).

The results from the Atomic Absorption analysis were in good agreement with the results from the X-Ray Fluorescence analysis (Figure 3), thus proving that the high Hg concentrations detected were not the result of an analytical error. Battelle also conducted Hg analyses on these feathers using atomic absorption spectroscopy in order to determine the concentration of total and methyl Hg. On average, 90% of the total Hg was in the form of methyl Hg. Two of the samples from Battelle showed slightly higher concentrations of methyl Hg than total Hg. This is probably the result of minor variations in the techniques and precision of the results.

Variation in elemental concentrations based on trophic status

Osprey, western grebes, great blue herons, and turkey vultures showed differences in their Hg concentrations that reflected their trophic status (Figure 4). Adult osprey showed the highest Hg values with a mean of 20.0 ppm (SE = 3.3). Of all the piscivorous birds at Clear Lake, osprey eat the largest (=oldest) and most contaminated fish (Suchanek et al. 1993), so it is not surprising that they contained the highest Hg values. Western grebes and great blue herons followed the osprey with mean Hg values of 9.74 ppm and 7.4 ppm respectively.

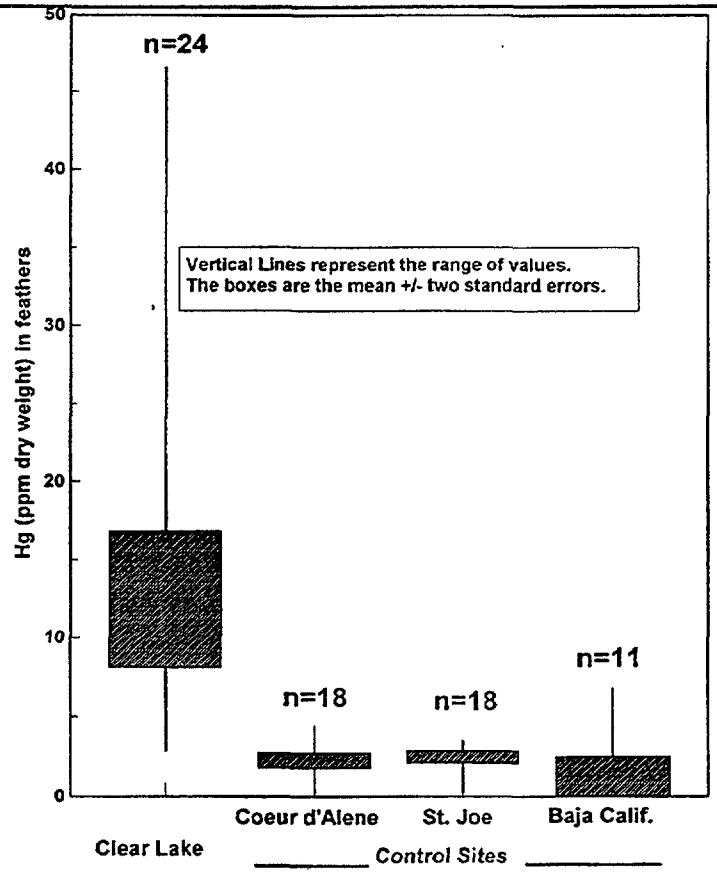

Figure 2. Comparison of Hg concentrations in osprey feathers between Clear Lake, Ca and the control sites of Coeur d'Alene, ID, St. Joe, ID and Bahia de Los Angeles, Baja California. The Clear Lake site had statistically higher Hg concentrations than the other three sites ($p < 0.0001$, pairwise Mann-Whitney tests).

Table 2. Elemental concentrations (µg/g dry weight) in feathers of juvenile osprey from Clear Lake, CA, and three comparison sites.

	Clear Lake, CA (n=12)		Coeur d'Alene, ID (n=18)		St. Joe, ID (n=18)		Bahia de los Angeles, Baja California (n=5)		Statistical comparisons	
Element	Mean[a]	SD/R[a]	Mean[a]	SD/R[a]	Mean[a]	SD/R[a]	Mean[a]	SD/R[a]	Test[a]	P-value
Sulfur	34099	2692	34901	1710	34255	1291	34505	3374	ANOVA	> 0.1
Calcium	1030	272	881	175	989	139	1219	809	ANOVA	> 0.1
Titanium	6.48	mdl-38.1	2.06	mdl-6.67	15.4	mdl-54.5	2.88	1.81	K-W	0.009
Vanadium	ND	—	ND	—	trace[b]	mdl-1.93	trace[b]	mdl-1.81	—	—
Chromium	trace[b]	mdl-3.8	ND	—	ND	—	ND	—	—	—
Iron	96.6	42.3-305	87.9	45.9-349	56.3	28.5	26.0	7.79	K-W	< 0.001
Nickel	1.01	mdl-22.3	ND	—	trace[b]	mdl-2.83	ND	—	—	—
Zinc	151	30.6	140	23.3	133.6	10.5	148	44.9	ANOVA	> 0.1
Arsenic	ND	—	ND	—	trace[b]	mdl-0.32	ND	—	—	—
Selenium	3.38	mdl-22.3	3.77	2.43	2.61	1	5.21	1.02	K-W	0.024
Bromine	76.5	30.6	30.5	17.4	15.7	4.73-59.7	32.5	2.47	K-W	< 0.001
Rubidium	ND	—	trace[b]	mdl-0.98	0.43	mdl-1.15	ND	—	—	—
Strontium	1.26	mdl-3.71	0.81	0.53	0.82	mdl-2.18	2.44	0.74	K-W	0.008
Mercury	5.25	2.22	2.17	1	2.25	mdl-3.50	ND	—	K-W	< 0.001
Lead	trace[b]	mdl-2.56	7.78	2.99-31.0	trace[b]	mdl-2.56	1.05	mdl-2.47	K-W	< 0.001

a Arithmetic means and standard deviations are given for normally-distributed data sets; if the data are not normal, then geometric means and ranges are given. When "mdl" is included in the range, it means that the element was below the minimum detectable limit in one or more of the samples from that site. Comparisons between normally distributed data were conducted by analysis of variance

b "Trace" indicates that the element was detected in less than half of the samples from that site, hence a mean value calculation would have been biased. "ND" denotes that the element was not detected in any samples from that site.

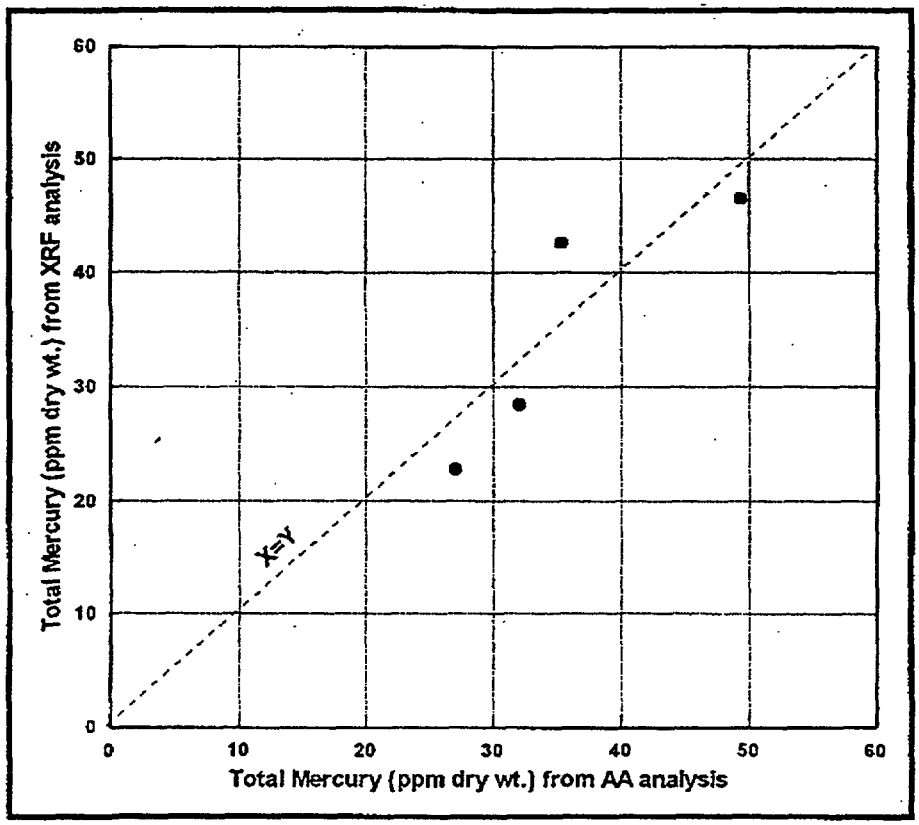

Table of values:

Sample Number	Total Hg (AA analysis)	Total Hg (XRF analysis)	Methymercury (??? analsysis)	% MeHg
1	49.4	46.5	50.6	102.4
2	32.1	28.4	25.3	78.8
3	27.0	22.8	18.3	67.8
4	35.4	42.6	39.3	111.0

Figure 3. Comparison of Hg concentrations obtained by atomic absorption and by X-ray Fluoresence for the same samples. The regression values are: slope = 1.03 and R^2 = 0.77.

Western grebes eat almost exclusively small fish while great blue herons also consume fish as well as aquatic invertebrates. Turkey vultures, which rarely interact with the contaminated aquatic system, showed the lowest Hg values at a geometric mean of 1.35 ppm. Some turkey vultures, however, have been observed eating dead fish that wash up onto the shore. Unfortunately, turkey vultures can be migratory, so we do not know where these particular turkey vultures were when they grew their feathers.

While the concentration of Hg in the feathers was highly dependent on the bird's trophic status, other elements were more consistent between different species. Zinc concentrations were fairly constant between different species, with the exception of great blue herons (Figure 5). The reason for the elevated zinc values in great blue herons is unknown.

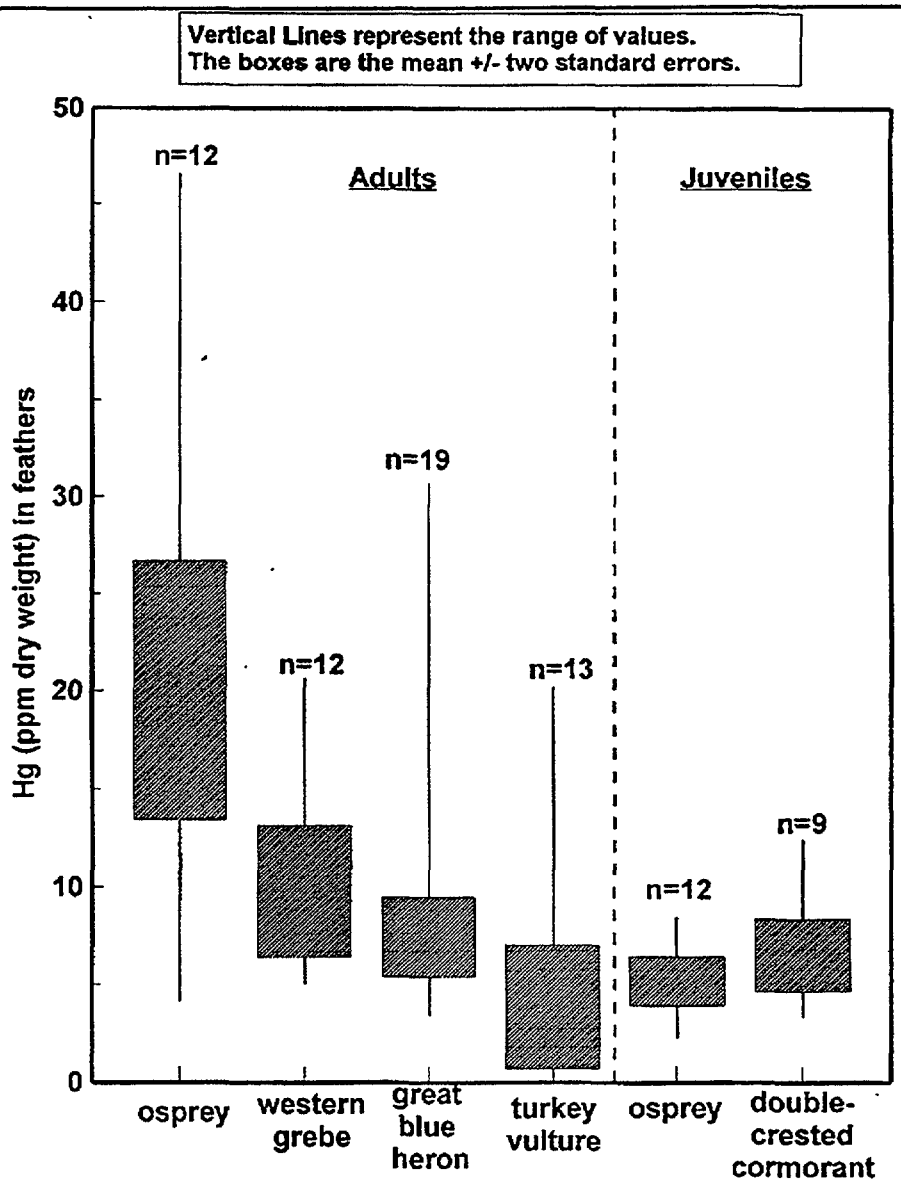

Figure 4. Trophic differences, with respect to Hg, between adults of four species and the juveniles of two species at Clear Lake, CA. Adult osprey showed higher Hg concentrations than the other three species ($p < 0.026$), paired Mann-Whitney tests) and the turkey vultures showed the lowest Hg concentrations of the adult samples ($p < 0.0043$, paired Mann-Whitney tests). Adult osprey exhibit significantly higher Hg concentrations than juvenile osprey ($p < 0.0024$, Mann-Whitney test).

Age Variation in Mercury Concentration

Several studies have reported that age influences the elemental profiles found in feathers (Honda et al. 1986, Denneman and Douben 1993). Since we have collected osprey flight feathers from both adults and prefledgling birds, we were able to compare the Hg values between these two age classes (Figure 4). The two age classes of osprey clearly show different Hg concentrations ($p<0.0024$, Mann-Whitney test). Adult osprey Hg concentrations were 3.87 times higher than the juvenile Hg concentrations. This suggests that Hg is accumulated

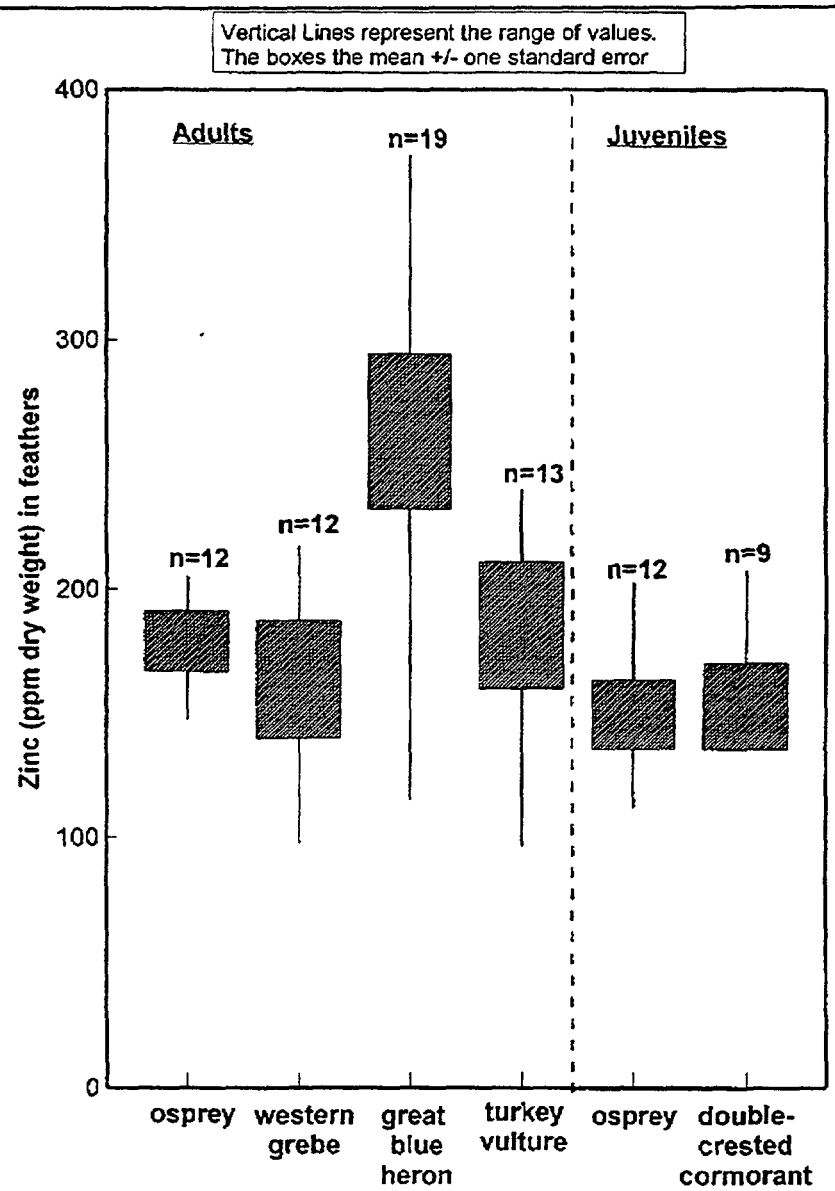

Figure 5. Zinc concentrations in the five species sampled at Clear Lake. Adult osprey exhibited a higher zinc concentration than juvenile osprey ($p < 0.009$, t-test).

throughout the life of fish eating birds like osprey that live in a high Hg environment such as Clear Lake. This result mirrors the results from the fish Hg levels where older fish have higher Hg concentrations (Suchanek et al. 1993).

In addition, as a comparative tool, we analyzed the adult osprey feathers for zinc. Like Hg, zinc was higher in feathers of adult osprey when compared with juveniles (T-test, $p< 0.009$), but the difference in zinc concentration between the two age classes is significantly smaller than it is for Hg (Figure 5). The zinc concentration in the adult osprey was about 1.2 times higher than the juvenile concentration, whereas the Hg concentration was approximately 3.87 times higher than the juveniles.

Juvenile double-crested cormorants were also sampled at Clear Lake (Table 1). Since age is known to affect the body burden of an organism (Honda et al. 1986, Denneman and Douben 1993), we could not directly compare samples from juvenile cormorants to the other four

species where adults were sampled. The juvenile cormorants showed slightly higher Hg concentrations (mean = 6.48, SE = 0.93) than juvenile osprey (mean = 5.17, SE = 0.63). If adult cormorants have a similar young:adult Hg ratio as osprey, where adults have 3.87 times higher Hg levels, then we would expect to see adult cormorant Hg levels near 25 ppm.

Discussion

The X-Ray Fluorescence analysis technique was promising in conducting elemental analyses on feathers. This technique provided considerably more information about the elemental constituents in feathers than a single-element atomic absorption analysis would have, and many other elements were also analyzed. Fifteen elements were detected in the five species from Clear Lake, including the important potential contaminants mercury, lead, selenium, arsenic, nickel, chromium and vanadium. The minimum detectable limits for these elements were generally below one part per million in the sample matrix (Cahill et al. in preparation). Since the levels of other contaminants, like lead, selenium, and arsenic, were low, other elements were probably not compounding the effects of the Hg contamination.

The levels of Hg detected in the birds from Clear Lake are clearly elevated above the background concentrations of our control sites (Table 2). The next question is whether the birds are impacted by the higher Hg levels when compared to birds in control sites. We monitored the reproductive success of osprey and western grebes to determine if their reproductive success was different at Clear Lake in comparison with control sites. The reproductive success of the osprey at Clear Lake in 1995 was similar to our control site at Lake Almanor (Table 3). In 1996, the osprey improved their reproductive success rate at Clear Lake and the breeding population had grown by 4 nests over the previous year. Also, the two osprey nests closest to the mine site both produced 2 fledglings each in 1996, which was above average. The average reproductive success for osprey reported in the literature was approximately 1.35 fledglings per nesting attempt (Poole 1989). From this small data set, it would seem that for 1995 and 1996, Hg is not having a noticeable impact on the breeding success of the osprey in the area.

Table 3. Reproductive success of osprey and western grebes at Clear Lake and control sites (Lake Almanor and Eagle Lake).

Species and Year	Population	Reproductive Success
Osprey 1995 -- Clear Lake	16 active nests	1.2 young per nest attempt
Osprey 1996 -- Clear Lake	20 active nests	1.75 young per nest attempt
Osprey 1995 -- Lake Almanor[1]	47 active nests	1.25 young per nest attempt
Osprey-- Average literature value	---	1.35 young per nest attempt[3]
Western & Clark's grebes 1995	Est. 2000-2500 adults	1 young per 5.5 adults
Western & Clark's grebes 1996	Est. 2500 adults	1 young per 2.5 adults
Western & Clark's grebes 1996 (Eagle Lake control site)	Est. 4400 adults[2]	1 young per 1.5 adults

[1] Data from Scott Armentrout, US Forest Service, Chester Ranger Station, CA

[2] Data from Dan Shaw, Chico State, CA.

[3] Average reproductive success from 9 different geographic locations reported (Poole 1989).

Unlike the osprey, the western and Clark's grebes showed large differences in reproductive success between Clear Lake and our control site at Eagle Lake (Table 3). Since the reproductive rates at Clear Lake are clearly lower in both years than our control site, some factor seems to be impacting the grebe breeding success at Clear Lake. However, the grebes showed lower levels of Hg than the osprey, which are apparently unaffected, so it seems that Hg may not the primary factory impacting the grebe reproductive success. There was also a very large difference in reproductive success between 1995 and 1996 at Clear Lake. This high variability makes it difficult to determine what factors are influencing grebe reproduction.

The differences in Hg concentrations between the adult osprey and the juvenile osprey pose questions about what should be sampled. Pre-fledgling birds are sampled more often because they represent the local conditions and the samples are usually easier to collect. However, the adult osprey have higher Hg concentrations, so they are more likely to be impacted by the Hg. Unfortunately, the adult osprey also migrate or wander away from Clear Lake, so we cannot be certain where the birds spent their winter. Of four 1995 radio-tagged fledgling osprey that our group had monitored , two were detected in southern California while the other two were not detected during the winter. This problem could be answered through satellite telemetry where we could track the birds on their winter migration. It is possible, although unlikely, that the osprey spent the winter in a contaminated area, so not all of the mercury present in the bird is from Clear Lake. It is probably best to continue to sample both adult and juvenile osprey in order to get a complete understanding of the Hg body burdens of the osprey.

Acknowledgments

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Effects of Dissolved Organic Carbon (DOC) on Methyl Mercury Uptake by Sacramento Blackfish (Orthodon microlepidotus). (P)

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Key Words: Methyl Mercury, Sacramento blackfish, Orthodon microlepidotus

The effect of DOC on methyl mercury (meHg) uptake across the gills of Sacramento blackfish (Orthodon microlepidotus) was investigated using Hg-203 radioisotope (half life = 46.9 days). The efficiency of fish gills in extracting meHg (1.4 ng/L) from water was measured using a McKim-type fish respirometer that separated exposure (inspired) water from expired water. Blackfish gill ventilation and oxygen consumption rates remained constant, while me203Hg uptake was decreased significantly ($p < 0.05$) with the presence of DOC (2 and 5 mg C/L). Since respiratory parameters remained unchanged, reductions in me203Hg uptake indicate strong interactions between DOC and me203Hg. Methyl203Hg levels in fish gills, kidney, and spleen from 2 and 5 mg C/L were significantly lower ($p < 0.05$).

SULPHUR BANK HYDROTHERMAL FLOWS, GEOTHERMAL ENERGY POTENTIAL AND ENVIRONMENTAL REMEDIATION

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Abstract - Mining activities are described briefly, indicating the presence of over 1,540 ft. (468 m) of mine shafts and over 1,250 ft. (387 m) of tunnels, both potential conduits for geothermal toxic fluid flows into Herman Pit and/or Clear Lake. Historical hot spring and meteoric flow rates are discussed, ranging from 300 gpm (1.1 m3/min) suggested by Veatch to less than 5,000 gpm (18.9 m3/min) suggested by White and 40 gpm (0.15 m3/min) to 290 gpm (1.1 m3/min) suggested by Humboldt State. The Herman Pit required 6.8 years to fill in 1954 which equates to a combined meteoric flow plus hot spring flow of 95 gpm (0.36 m3/min). Using a chloride and boron chemical balance, hot spring flows are estimated at 30 gpm (0.11 m3/min) and 51 gpm (0.19 m3/min), respectively, which suggests that the meteoric inflow to the Pit is between 65 gpm (0.25 m3/min) and 44 gpm (0.17 m3/min]. Thus flow values obtained from observation, chemical balances and a heat budget support a hot spring inflow rate in the 40 gpm range with a meteoric annual average inflow in the range of 55 gpm, mean annual averages. The estimated yearly potential hot spring transport from the Herman Pit is estimated, based on a flow rate of 40 gpm (0.36 m3/min) and a concentration of 0.55 ppm for mercury and 0.3 ppm for arsenic, to be 44,000 g-Hg/yr (44 kg-Hg/yr) mercury and 24,000 g-As/yr (24 kg-As/yr) arsenic. Environmental remediations recommended to prevent these hot spring flows from continuing to contaminate Clear Lake include:

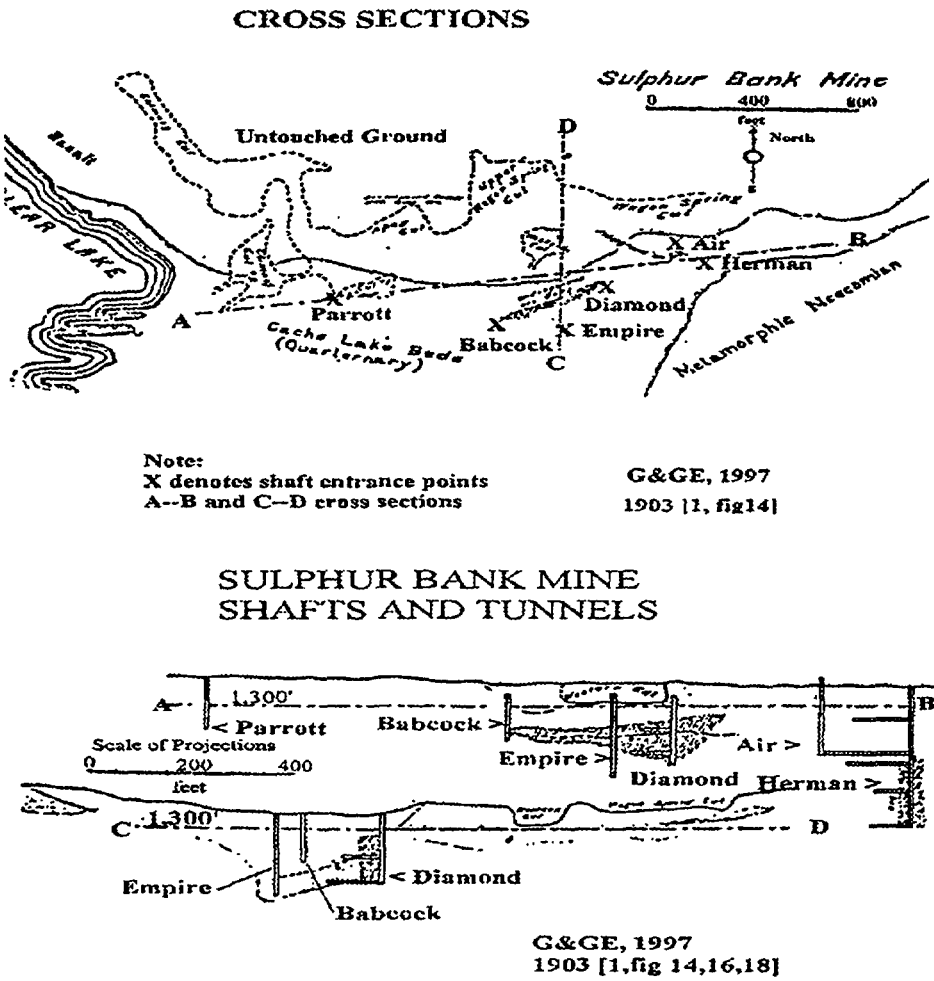
- drilling a geothermal production well at Bradley 1;
- treatment of the toxic geothermal fluids produced through the Bradley 1 well;
- piping the 4.9 mgd (18,000 m3/dy) treated fluids into The Geysers pipeline;
- producing the available 2.9 MWe electrical power;
- draining and treatment of the fluids from Herman Pit and sending the effluent to The Geysers pipeline;
- filling Herman Pit with the mercury laden tailings adjacent to the Elem Rancheria; and
- making waste heat and electricity available to the Elem Rancheria as a mitigation measure.

Key Words - Sulphur Bank, Herman Pit, shafts, tunnels, fluid flows

INTRODUCTION

History - Hydrothermal sulfur deposits were mined at Sulfur Bank starting in 1865 and running through 1868 with some 2,000 tons produced. The first mercury mining began in 1875 and lasted until 1883 during which period the initial excavation of Herman Pit occurred. Mercury mining resumed in 1887 to 1897 during which period the Diamond and Babcock shafts were sunk. Intensive periods of tunnel mining continued during 1899 to 1902 and 1915 to 1918. Open pit mercury mining began in a small way during 1899-1902 and 1915-1918 then on a large scale from 1927 through 1947 and again in 1955 through 1957 [1,2]. While it was necessary during tunneling to pump fluids out of the mine, during periods of open pit mining much larger quantities of fluid and silt were pumped into Clear Lake. The episodes of open pit pumping are evident in lake bottom cores where distinct layers of mercury contamination are present. By 1903 some 92,000 flasks had been produced with a final estimate of 129,418 flasks (9,835,768 lb) of mercury produced and retorted at the mine [1,2]. The mercury production at Sulphur Bank is considered to be the largest in the world from this type of hydrothermal hot spring deposit. It was estimated by White and Roberson that half of the deposited mercury, some 9,800,000+ lb, still remain [2].

SULPHUR BANK SHAFTS, TUNNELS AND WELLS



AS CONDUITS FOR

Figure 1: Composite of Sulphur Bank Shafts and Tunnels

TOXIC FLUIDS

Sulphur Bank mine shafts and tunnels are described in the 1903 California State Mining Bureau Sulphur Bank Section and by White and Roberson, see Figure 1, [1,2 Figs. 2&3]. The shafts in some instances are described as passing through some unconsolidated loose materials with tunnels primarily passing through hard rock. Based on scaled estimates from the figures in the Sulphur Bank report, the six vertical shafts measured, respectively, 170 feet below ground level (ftbgl) [52 meters below ground level (mbgl)], 150 ftbgl (45.7 mbgl), 300 ftbgl (91.4 mbgl), 260 ftbgl (79.2 mbgl), 240 ftbgl

(73.2 mbgl), and 417 ftbgl (127 mbgl) for a total vertical shaft length of 1,540 ft. (468 m) [1, Fig. 14]. Again, based on figures in this report, the horizontal tunneling totals 950 ft. (290 m) for the Herman Shaft and 296 ft. (90.2 m) for the Diamond shaft, with a total length of 1,250 ft. (387 m). No tunnel data was found on the other four shafts. Below ground excavated ore chambers must be added to the tunnel information [1, Fig.s 14, 16, 18; 2, Fig. 2]. To these major shafts must be added the many documented shallow shafts and short tunnels which were excavated in the many cuts during mining of the areas [1]. The Herman shaft, constructed in 1875 to 1883, has seven horizontal levels with an air shaft at level three 220 ftbgl (67 mbgl) running some 300 ft (91 m) horizontally then returning to the surface. Horizontal tunnels were excavated extending from the Herman shaft, level one at 104 ftbgl (31.7 mbgl), level three at 210 ftbgl (64.0 mbgl), level four at 262 ftbgl (79.9 mbgl), level five at 310 ftbgl (94.5 mbgl), and the last level seven at 417 ft (127 mbgl). During mining operations from 1887 to 1897 the 90 ftbgl deep Diamond shaft, which had at least two levels 110 ft. (34 m) and 186 ft (57 m) long, and the 100 ft (30 m) deep Babcock shaft were excavated [1]. Based on scaled measurements from the 1903 report, the distance from shaft entrance points to the 1903 Clear Lake shore range from 650 ft (198 m), a 170 ftbgl (52 mbgl) shaft, to 2,050 ft (625 m) [1, Fig. 14]. Geothermal exploration beginning in 1964 drilled the Bradley 1 geothermal well, 1,631 ft (497 m) to 2,089 ft (637 m) in depth and Bradley 2 geothermal well 2,497 ft (761 m) deep just to the north and south of Herman Pit respectively. The Bradley 1 well flowed up to 3,500 gpm (13 m3/min) of fluids at a temperature of 310 degF (154 degC) and from 180 psi to 350 psi well head pressure [3]. These 30+ year old wells and concrete plugs have been subjected to high temperatures, high pressure, corrosive hydrogen sulfide gases and acidic fluids. It is likely that these wells provide another vertical conduit for the toxic fluids.

Each of the vertical and horizontal mine shafts, tunnels, excavated galleries, and the corroded 1964 Bradley 1 and Bradley 2 geothermal wells, has a major potential to conduct toxic-rich fluids with suspended particulate mercurial sulfide (floc) and other toxics, allowing the toxic fluids to move horizontally and vertically at unknown rates into Clear Lake. Many of the present day subsurface flow calculations have been based upon monitoring wells ranging in depth from 21 ftbgl (6.4 mbgl) to 100 ftbgl (30.5 mbgl) [3, Appdx B, Table 3.2] . Reliance on such monitoring well data is suspect in light of the above description of shafts, tunnels, galleries and wells underlying the mining area. Each shaft and tunnel is a potential pathway of meteoric, hydrothermal and/or geothermal fluid flows which may be conducted into Clear Lake.

SULPHUR BANK FLUID FLOWS

Historical estimates of the Sulphur Bank hot spring system flow rates include Veatch's 1883 estimate of 300 gpm (1.1 m3/min) which White suggested included surface runoff. White suggests that historical flows ranged from 500 gpm (1.89 m3/min) to less than 5,000 gpm (18.9 m3/min) [2]. The review during the Humboldt State Study suggested a range of 40 gpm (0.15 m3/min) to 290 gpm (1.1 m3/min) [4].

White observed that the Herman Pit required 6.8 years to fill starting from May 1947 to full in March 1954 when he estimated the volume was 340 million gallons (1.29 E6 m3). During this period, the combined meteoric plus hot spring flow rate into the Pit is calculated to be 95 gpm (0.36 m3/min). In 1987 Columbia Scientific estimated the Pit volume as 31,790,599.06 ft3 (31.8 million ft3) (237 E6 gal) (0.9 E6 m3) with up to 27 ft (8.2 m) depth of sediments [5]. The 30% decrease in the estimated Pit volume over the 30+ year period appears plausible considering runoff erosion and the transport of particulates from the chemical laden hot springs. It was observed by Trumbull, CRWQCB during a visit on July 10, 1954 that the water inflow to the Pit required a 1,000 gpm pump running three days every three weeks to drain it, which equates to a combined meteoric and hot spring inflow of 143 gpm (0.54 m3/min) [5].

Chloride and boron chemical balances were used by White to estimate the hot spring flow [2]. Using the 95 gpm flow for combined meteoric and hot spring flows, estimating the Pit's chloride level at 370 ppm and the hot spring's at 700 ppm, the hot spring flow was estimated at 50 gpm (0.19 m3/min) [95*370/700]. Using the 95 gpm flow for combined meteoric and hot spring flows and estimating the Pit's boron level at 280 ppm and the hot spring's at 670 ppm, the hot spring flow was estimated at 40 gpm (0.15 m3/min) [95*280/670]. This suggests that the meteoric inflow to the Pit was between 45 gpm (0.17 m3/min) [95-40] and 55 gpm (0.21 m3/min) [95-40].

These estimates of meteoric and hot spring flows were updated using more recent data, where the Pit's chloride level has been measured at an average of 310 ppm and the boron level at an average of 269 ppm, and then using the Bradley 1 well measured chloride level of 990 ppm and the boron level of 500 [5,3]. Using again the 95 gpm flow for combined meteoric and hot spring flows which still seems the best long term annual average leads to a hot spring flow estimate of 30 gpm (0.11 m3/min) [95*310/990] based on the chloride balance and 51 gpm (0.19 m3/min) based on the boron balance [95*269/500]. This suggests that the meteoric inflow to the Pit is between 65 gpm (0.25 m3/min) [95-30] and 44 gpm (0.17

m3/min) [95-51].

A flow of 40 gpm (0.15 m3/min) at 160 degF (73 degC) to the Pit from the hot spring would supply enough heat to increase the mean annual temperature of the Pit's volume by 14 deg F (8 degC) if no heat were lost to the environment. Heat is lost to the environment from the pit by conduction, evaporation, radiation and through inflow and outflow. Review of the sparse data comparing the temperature of the Pit water and the water of Clear Lake in the Oaks Arm suggests that the Pit is estimated to be no more than 3 degF (2 degC) warmer. The small difference in temperature between the waters in the Lake Oaks Arm and the Pit does not suggest that the hot spring flow is ten or 100 times greater than the estimated 40 gpm. Such large flows suggested by recent tracer studies would result in a much greater temperature differential than that observed.

The time that water on the average is retained in Herman Pit (the residence time) is estimated to be 1,740 days (4.8 years) for a hot spring flow of 40 gpm (0.15 m3/min) and a meteoric annual average inflow of 55 gpm (0.21 m3/min) totaling 95 gpm (0.36 m3/min). If the Bradley 1 well was full flowing at 3,500 gpm (13 m3/min) into the Pit, the residence time would be 47 days and this flow would raise the Pit temperature significantly above the observed lake temperature.

White's observed 1946 to 1954 Pit fill time, Trumbull's observed 1954 pit mining pumping rate, the chloride and boron chemical balance and the elemental heat budget all support a hot spring inflow rate in the 40 gpm range with a meteoric annual average inflow in the range of 55 gpm. These observations do not preclude the possibility of high inflows originating from the Sulphur Bank deep geothermal resource entering Clear Lake by some route other than a route through Herman Pit.

The estimated yearly potential hot spring transport through the Herman Pit to Clear Lake would be 44,000 g-Hg/yr (44 kg-Hg/yr) mercury and 24,000 g-As/yr (24 kg-As/yr) arsenic based on a flow rate of 40 gpm (0.36 m3/min) and concentrations of 0.55 ppm mercury and 0.3 ppm arsenic, respectively.

SULPHUR BANK GEOTHERMAL ELECTRICAL POWER FOR THE GEYSERS PIPELINE

In 1991 G&GE performed a feasibility study of deep injection disposal of excess wastewater originating from the Lake County Southeast Regional Wastewater Treatment Facility. The study included the evaluation of the potential of producing geothermal electrical power using the known Bradley 1 geothermal resource at Sulphur Bank, treating these fluids and using the fluids to augment wastewater sewer plant flows [6]. An analysis of regulatory opportunities and restraints of deep injection of wastewater for geothermal resource replenishment supported the concept [7]. The position and depth of the Bradley 1 well geothermal resource which supplies the hot spring fluids to Sulphur Bank are known. Evaluation of the Bradley 1 geothermal flow test data indicated that the 70,000 lb-steam/hr and 1,700,000 lb-brine/hr would equate to an estimated 2.9 MWe electrical production potential and make 4.9 mgd (18,000 E3 m3/dy) of fluids available to The Geysers pipeline. It was estimated that the electrical power produced would pay for, or provide power for, pumping the sewer wastewater and Bradley 1 brine to The Geysers geothermal area and allow operating the pipeline without buying and adding Clear Lake water.

SULPHUR BANK ENVIRONMENTAL REMEDIATION

Plans to evaluate the technical and institutional feasibility of harnessing the present hydrothermal and geothermal resources at Sulfur Bank Mine were developed by G&GE and proposed to the California Department of Health Services Hazardous Waste Reduction Grant on December 1, 1988 and a request for support for the project was presented to the Lake County Board of Supervisors on February 7, 1989 [7]. With the private sector interested in developing geothermal energy at Sulphur Bank, the production of geothermal power using the Bradley 1 resource was shown to include the following benefits:

production of an estimated 2.9 MWe of geothermal electrical power at Sulphur Bank;

off-setting Geysers Pipeline pumping electrical costs through power contract sales;

generation of waste heat and electrical power for the Elem Rancheria as a mitigation measure;

use of electrical revenue sales for further Sulphur Bank Remediation;

stop toxic flows, drain Herman Pit and rerouting of meteoric flows; and

filling in Herman Pit using high mercury laden tailing adjacent to the Elem Rancheria.

The following environmental remediations are recommended to prevent these hot spring flows from continuing to contaminate Clear Lake:

drilling a geothermal production well at Bradley 1;

treatment of the toxic geothermal fluids produced through the Bradley 1 well;

piping the 4.9 mgd (18,000 m3/dy) treated fluids into The Geysers pipeline;

producing the available 2.9 MWe electrical power;

draining and treatment of the fluids from Herman Pit and send to The Geysers pipeline;

filling Herman Pit with the mercury laden tailings adjacent to the Elem Rancheria; and

making waste heat and electricity available to the Elem Rancheria as a mitigation measure.

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RESULTS OF STAGE 1 GEOPHYSICAL INVESTIGATION SULPHUR BANK MERCURY MINE CLEAR LAKE, CALIFORNIA

Prepared for ICF Kaiser Engineers, Inc. by Martin Miele, R.G. R.Gp.

1.0 INTRODUCTION

This report presents the results of the Stage 1 geophysical investigation conducted at the Sulphur Bank Mercury Mine (SBMM) Superfund Site near Clearlake Oaks, California. The purpose of the geophysical investigation was to identify potential subsurface conduits leading from a mine-related surface water impoundment to Clear Lake. The investigation was performed in April 1997 by ICF Kaiser Engineers, Inc. (ICF Kaiser). The geophysical surveys were conducted on the western waste rock pile (the shoreline pile) and areas southwest of Herman Impoundment. As discussed in this report, several geophysical anomalies that may be indicative of subsurface conduits were identified during this Stage 1 investigation. The geophysical investigation was conducted in accordance with EPA-approved plan entitled Field Sampling Plan for Geophysical Activities, Sulphur Bank Mercury Mine, Clear Lake, California (ICF Kaiser, October 1996).

1.1 Site Description

The SBMM Superfund Site is located at the eastern edge of the Oaks Arm of Clear Lake. In 1865, elemental sulfur was first mined at the site from above the groundwater table. As mining proceeded, cinnabar, a mercury ore, was discovered below the water table. When near-surface sulfur deposits were exhausted in 1873, andesite (an extrusive volcanic rock) was then mined for the cinnabar (mercury) that it contained. Andesite mining continued intermittently until 1957.

Herman Impoundment was created from a 21-acre, 100-foot deep mining pit that subsequently filled with water. The tailings and waste rock generated from mining activities were distributed over 107 acres surrounding Herman Impoundment. West of Herman Impoundment, the western waste rock pile was constructed to separate the impoundment from the Clear Lake shore.

In 1995, due to higher-than-normal rainfall, Herman Impoundment overflowed its western bank. The water flowed through the constructed spillway channel within the western waste rock pile into Clear Lake. Later that year, a white flocculent material was discovered near the Clear Lake shore extending out from the mine site. The material was discovered at the northern end of the shoreline waste rock pile and at the southern end along the beach area. The material was identified by UC Davis as halloysite, a clay mineral known to precipitate from acid mine drainage, and unusually high methyl mercury levels were found associated with the flocculent material.

The mine drainage and Herman Impoundment water, which contain mostly inorganic mercury, also contain high levels of sulfate. The sulfate, coupled with the surface area provided by the clay precipitate, provide a fuel source and substrate for the colonization of sulfur-reducing bacteria. These bacteria are known to produce methyl mercury in lakes.

According to mass balance calculations it was concluded that the amount of mercury in Herman Impoundment overflow water was not sufficient to account for the amount of mercury found in the flocculent adjacent to the mine site. Therefore, another source of mercury must exist.

Several sources are possible which can range from permeable zones or groundwater conduits within the western waste rock piles (between Herman Impoundment and Clear Lake) to deeper geologic features such as faults/geothermal vents. In addition, it is reported that some tunnels (mine adits and shafts) are still present on the property. One is reported to exist in the northern section of the waste rock piles which could connect Herman Impoundment with Clear Lake.

The main focus of this investigation was the western waste rock piles, along the Clear Lake shore, and selected other locations in that vicinity. The approximately 13-foot hydraulic head difference between Herman Impoundment and Clear Lake indicates that flow occurs through the waste rock piles to the lake, and the flow may occur in highly permeable zones or conduits.

1.2 Objective

The primary objective of this Stage 1 reconnaissance geophysical investigation was to identify groundwater transport zones

or conduits within the western waste rock pile adjacent to the Clear Lake shore. This is one possible source of the elevated methyl mercury concentrations in flocculent offshore. A secondary objective was to assess possible transport zones or conduits in the vadose zone on the western perimeter of Herman Impoundment and along the toe of the hills south of Herman Impoundment. Resistivity and spontaneous-potential (SP) geophysical survey methods were used for the investigation.

To maximize the potential for detecting groundwater conduits, the geophysical survey was conducted after the winter wet season in March-April 1997. The target of the investigation was geophysical anomalies that typically represent subsurface saturated zones. These are generally characterized by zones of relatively low resistivity and anomalous self-potential values. Some of the buried waste rock materials generate localized anomalies that are limited in lateral extent. However, the targeted anomalous zones in this investigation should extend from Clear Lake to Herman Impoundment if they represent potential zones of groundwater transport.

Some of the geophysical profile locations were adjusted during the field investigation. The profile locations were partly controlled by access within the thick brush and local topographic effects, as well as the survey objectives.

2.0 METHODOLOGY AND FIELD SURVEY

In all, five profiles of resistivity and self-potential (SP) data were obtained in the spring of 1997. Two profiles, A and B, are located on the western waste rock pile (Figure 1). Profile A and B are both oriented approximately north-south with Profile A located on the western side of the waste rock pile. Profiles A and B are 800 and 720 lineal feet, respectively, and have an overall elevation change of approximately 20 feet. The purpose of these profiles was to assess conditions within the piles between Herman Impoundment and Clear Lake shore. The target of the investigation was geophysical anomalies indicative of subsurface conduits that extend between Herman Impoundment and Clear Lake.

Profile C was located along the foot of the hillside and tailing piles south of Herman Impoundment. Profile E was located along the east-west access road south of Herman Impoundment. The purpose of profiles C and E was to assess subsurface conditions (geophysical anomalies) that may be related to conduits leading from south of the waste rock pile to the beach area where flocculent was observed. Profiles C and E are 520 feet and 640 feet in length, respectively. Profile D was located along the western periphery of Herman Impoundment to assess conditions or anomalies in the immediate vicinity west of the impoundment. Profile D was 780 feet in length. Profiles C, D, and E are generally flat.

The resistivity surveys were designed to investigate materials (soil/rock) between the ground surface and water table (vadose zone). Therefore, the current electrode spacing (investigation depth) was adjusted according to the thickness of the vadose zone/waste rock pile along each profile. Two investigation depths were applied along profiles A and B (approximately 65 feet and 35 feet). One investigation depth was applied along Profiles C, D, and E, which was adjusted to the elevation of the assumed water table depth along each profile.
Survey grid markers were located at specific intervals along the profiles (30 to 80 feet depending upon profile). The intervals varied according to the geometry of the resistivity arrays used to construct each specific profile. The grid markers consisted of wooden lath hammered into the ground. Each grid marker was annotated with the profile name and distance (in feet) along the profile. Survey ribbon was tied to the top of each marker.
The gradient array (modified Schlumberger array) was used in the electrical resistivity surveys for this investigation. Electrical resistivity data were obtained at 10 foot intervals along each profile. A Sting R1 Resistivity Meter and non-polarizing electrodes were used to obtain the data. Several readings were repeated at numerous stations along each profile to assure data reliability (minimum of 10 percent duplicate measurement stations). Data repeatability was very good: all duplicate readings were within 99 percent of each other. Resistance data were obtained in the field and later converted to gradient array resistivity data using the pertinent equations presented below.

(a = L2/a (K) (V/I)

and

K = 2([(1-X/(Y2+(1-X)2)3/2) + (1+X/(Y2+(1+X)2)3/2)]-1

where

(a = apparent resistivity (Ohm-feet)

V/I = measured resistance (m()
K = geometric constant for each measurement
L = Ω current electrode spacing (AB/2) in feet
a = potential electrode spacing in feet (MN)
X = distance parallel to array from center of array (feet)
Y = distance offset from array from center of array

In the case of this investigation, Y = 0, because reconnaissance profiles were conducted. Therefore the equation for the geometric constant becomes:

K = 2([(1-X/((1-X)2)3/2)+ (1+X/((1+X)2)3/2)]-1

Calculations were made and the resulting resistivity data were plotted versus distance along each profile for all profiles used in the investigation.

The target of the resistivity investigation was low resistivity anomalies. Electrical resistivity is the reciprocal of electrical conductivity. The resistivity of earth materials is dependent upon the several factors such as grain size, composition, the amount of mineralization, moisture content, and other physical conditions of the rocks. In general, coarse-grained materials (sand, gravel, etc.) have a higher resistivity than fine-grained materials. Hard, relatively impermeable rock generally has a higher resistivity than sediments. However, the presence of water (especially saturated conditions) greatly lowers the resistivity of all earth materials. The low resistivity anomalies caused by the presence of water are good targets for resistivity surveys. The electrical resistivity method has been employed for several years in groundwater exploration.

Self-potential data were also obtained at 10-foot intervals along each profile using the fixed-base method to obtain the data. The potential difference between the base electrode and the measuring electrode was recorded for each measurement location. Data were obtained from the same stations used in the resistivity survey. Non-polarizing electrodes were used to collect the data. One base station was used for Profiles A and B. Profiles C, D, and E each used a separate base station due to the local field conditions.

3.0 RESULTS

The results of the geophysical investigation are presented in Figures 1 through 6. Figure 1 is the geophysical location map showing the location of the geophysical profiles, anomalies detected, and general results of the investigation. Figures 2 through 6 are graphs of the data where the horizontal axis is distance along each profile and the vertical axis represents the measured resistivity and SP data.

Several anomalies were resolved during the investigation. Some anomalies are consistent with the types of anomalies typically generated by groundwater or groundwater pathways (low resistivity and SP anomalies). In general, the resistivity and self-potential data show a good correlation. A general description of the anomalies is presented in the following sections.

3.1 Profiles A and B

Four low resistivity anomalies were resolved along Profiles A and B. Three of the anomalies occur in the northern section of the western waste rock pile. Each anomaly is a low resistivity zone, which can indicate the presence of water. In general, measured resistivity values within the anomalous zones are less than one half the resistivity values in non-anomalous zones. The anomalies occur in both the shallow (approximately 30 feet deep) and the deeper (approximately 60 feet deep) resistivity data.

The three anomalous zones correlate between Profiles A and B. They have a general east-west orientation. They extend from the waste rock pile toward the northern section of Herman Impoundment. The zone containing the resistivity anomalies also correlates with an area consisting of low, negative SP values. The combined anomalies (resistivity and SP) could be generated by the presence of groundwater or groundwater conduits. The anomalies are apparently continuous across profiles as would be expected for transport zones.

The fourth low resistivity anomaly extends between Profile A and B in the southern section of the waste rock pile. It

correlates with the location of a former channel that extended from Herman Impoundment to Clear Lake and was backfilled when the new spillway channel was constructed. The drainage may still transmit water or may have been filled in with clay or some other similar low-resistivity material.

3.2 Profile C

One low resistivity anomaly occurs along the southwestern end of Profile C, at the foot of the tailings pile near the southern hillside. The anomaly occurs at the end of the profile. The profile could not be extended due to the private property adjacent to the profile. The anomaly on Profile C is located in a localized surface low area where surface water may have accumulated and infiltrated into the soils. The low resistivity values may represent a section of a localized groundwater pathway or a zone of increased saturation due to the localized low area. The anomaly is not apparent along Profile E to the north.

One other low resistivity anomaly occurs in the northern section of the profile. However, the anomaly occurs in an area adjacent to saturated surface soils and standing water. The standing water likely caused the anomaly. Therefore, it is not indicated on the map.

3.3 Profile D

One limited low resistivity anomaly occurs along the southern section of Profile D at the western periphery of Herman Impoundment. It is located near the drainage that extends from the Herman Impoundment to Clear Lake. The anomaly may represent a localized saturated zone resulting from surface water infiltration along the drainage channel. However, it may also represent a groundwater conduit along the channel. It is reported that rocks of the Franciscan formation were used for fill material in the area. This may be the cause of the change in resistivity values.

Another major low resistivity anomaly is located at the northern extent of Profile D. The anomaly is located within an area characterized by negative SP values. The data in this area have the same characteristics as those detected along Profiles A and B. Therefore, the anomalies are correlated across all three profiles in that area. They extend to Herman Impoundment.

3.4 Profile E

The southwestern end of Profile E, along the access road, exhibits anomalous readings in both resistivity and SP. Resistivity values show an increase of at least an order of magnitude in that area and SP data show high magnitude values with large positive and negative variations. Those types of values are typical of a rock contact. The values may represent a contact between two different rock types or between fill material/sediment and rock. The high resistivity values suggest massive rock. The SP values and large variations suggest differentially banded or heterogeneous mineralized rock.

4.0 DISCUSSION

The low resistivity zones delineated in the resistivity survey can be caused by two mechanisms. Saturated rocks/soils have lower resistivities than unsaturated and dry materials. This is due to electrolytic conduction where aqueous ions migrate through an interconnected matrix. Low resistivity anomalies may also be caused by the presence of clay or highly mineralized materials. This is due to surface conduction (clay) or the presence of electrically conductive minerals.

Since the anomalous zones in the northern section of the waste rock piles extend between profiles, it is assumed that they are caused by the presence of a zone of increased moisture. Due to the assumed construction of the waste rock piles it is unlikely that linear zones of clay or mineralized rock were installed across the piles. It is assumed that those materials will have a more dispersed distribution. The locations of the linear anomalies are consistent with the observed location of flocculent material in Clear Lake adjacent to the northern section of the waste rock piles. If the northern section of the waste rock pile does represent a zone of groundwater transport, the SP values within that zone are consistent with a groundwater flow direction towards the west and Clear Lake.

The linear anomalies that extend across the northern section of the waste rock pile may be indicative of the suspected subsurface tunnel. It should be detected as a low resistivity anomaly if the tunnel is saturated with water. However, if the tunnel is completely dry it should be detected as a high resistivity anomaly. In that case, it would be located in the high resistivity zone between the low resistivity anomalies.

The interpreted contact between two rock types or between fill material/sediment and rock delineated along Profile E may act as a groundwater conduit. Contacts can act as linear transport zones for groundwater if conditions are favorable. The interpreted contactís general orientation is consistent with flocculent observed in the southern beach area.

The anomalies detected during this investigation are likely to be conduits for groundwater migration; however, no definitive conclusions can be made. A Stage 2 geophysical survey will be performed at the Sulphur Bank Mercury Mine, extending the previously surveyed profiles northward to include the area between the Northwest Pit and Clear Lake. The survey will use a dipole-dipole array to better delineate subsurface features. Additionally, a seismic survey will be performed to examine deeper features that may control groundwater flow in the shoreline pile.

Methyl Mercury Production From Unamended Sediment Cores (Core Tube Microcosms). (P)

E.E. Mack, D.C. Nelson, L.L. Brister, T. H. Suchanek

Abstract

Methyl mercury evolution was evaluated from sediment cores from various sites on Clear Lake. *In situ* total mercury concentrations were used under several conditions. Two experiments were conducted. In the first, the effects of anoxia were examined by sparging with nitrogen gas in the overlying filtered site water. Overlying water was removed and replaced at 0.5, 2, 4, 8, and 16 days of incubation. Methyl mercury was found to evolve from these sediments in a relatively short period of time, with peaks usually within the first four days. The effects of anoxia varied from site to site. In the second experiment, the effects of anoxia, chironomids, molybdate, and flocculent material collected offshore of the mine (floc) were analyzed as well as the microstratigraphy of sediment methyl and total mercury in sediment from the examined sites. All cores were incubated for four days. The presence of floc greatly increased the amount of methyl mercury present in overlying cores. The effects of the other treatments varied from site to site. In the sediment microstratigraphy analyses, methyl and total mercury decreased with depth at site OA-01 and remained constant at site UA-03. Although floc was found to have initial mercury concentrations similar to other sites in sediment and the same as OA-01 in water, it was able to evolve an average of approximately 4 to 12 times more methyl mercury to the water than other sites.

Introduction

Methyl mercury evolution from vertically stratified sediment cores at *in situ* total mercury (Hg) concentrations was measured under several conditions at two different times. It was felt that large (7 cm diameter) stratified sediment cores would be more comparable to *in situ* lake conditions than small sediment slurries. It should be kept in mind that these experiments measured the evolution of methyl Hg into overlying water. This does not necessarily imply the production of methyl Hg, although it probably does. Indeed, the amount of methyl Hg evolved into the water during the experiment (as calculated in the second experiment) is trivial when compared to the standing concentrations of methyl Hg within the sediment at the start of the experiment. Additionally, the effects of demethylation in these experiments are unknown. Methyl Hg measurements must be considered net measurements, representing the result of the competing processes of Hg methylation and methyl Hg demethylation.

Methods of Experiment #1

The first experiment was carried out in October 1995 and examined methyl Hg evolution into water over time from sediment cores collected at sites OA-04, LA-04 and UA-03. The effects of anoxia were also examined by sparging some cores with N_2 gas and some with compressed air. The water overlying the cores was removed and analyzed for methyl Hg and was replaced with fresh, filtered site water at 0.5, 2, 4, 8, and 16 days of incubation. The results from this experiment were never fully analyzed due to methodological problems that became clear after reviewing the first data set. Additionally, the variability among

replicate samples in this experiment was very high. However, the results in hand are intriguing and led directly to the procedural changes in the second experiment. The following discussion of these results and the results from the second experiment are based upon averages of the replicates in each condition. Statistical analysis of the significance of the variability in these experiments will be done in the future and may change the interpretation of these results.

Results of Experiment #1

Perhaps most interesting is that methyl Hg evolved from these unamended sediments into the overlying water within in a relatively short period of time (Figure 1). Methyl Hg evolution appears to peak early in incubation (usually within the first 4 days of incubation) and drops off thereafter. At site OA-04 the air sparged (oxic) cores evolved methyl Hg concentrations indistinguishable from the water only controls. However, the N_2 sparged (anoxic) cores evolved 2-4 times more methyl Hg than the controls at site OA-04. At site UA-03 both oxic and anoxic cores evolved more methyl Hg than the water only controls and one of the oxic cores evolved methyl Hg at concentrations comparable to the anoxic cores. This discrepancy

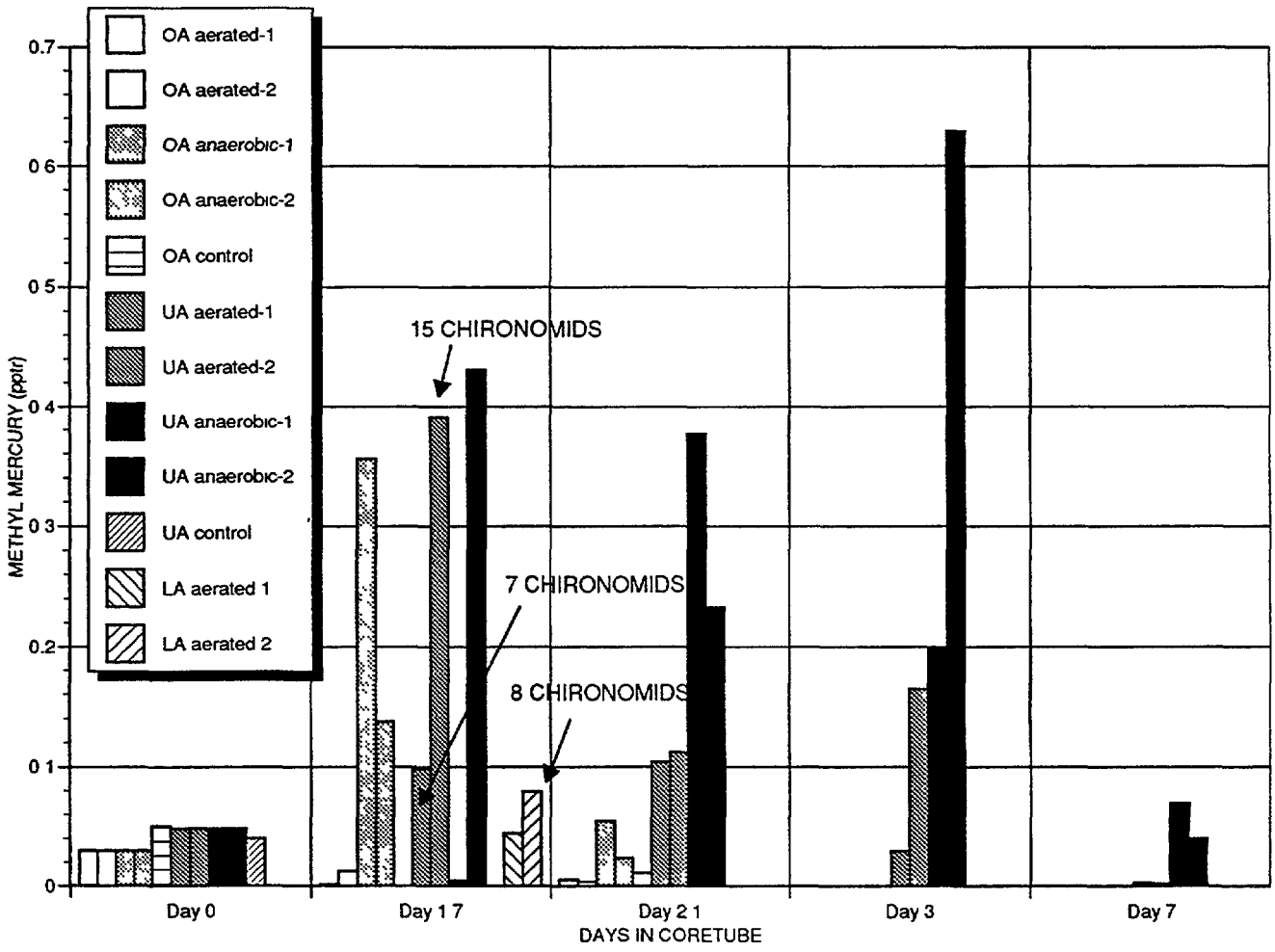

Figure 1. Core tube experiment #1, net methyl mercury production per day.

might have been due to the presence of chironomids in the UA-03 cores. Based on the limited data set there appeared to be a positive correlation between the number of chironomid burrows present and methyl Hg evolution rate. It is possible that the chironomid effect is due to increased sediment anoxia even in the presence of oxygen because of an increased sediment biological oxygen demand. Alternatively, these organisms may affect sediment - water partitioning of methyl Hg. Methyl Hg was not produced in water only controls. However, these controls clearly showed the loss of total Hg from the water, possibly adhering to the plexiglass wall of the corer.

Methods of Experiment #2

The second core microcosm study (October 1996) was carried out on sites OA-04, UA-03 and OA-01 +/- floc. The reactivity of the core tube walls was addressed by constructing the core tubes out of Teflon. Other changes made to the experimental methods to address uncertainties from the first experiment included a reduced incubation period (4 days) with a single measurement of methyl Hg in the water at the end of this time, and continuous sparging for the oxic conditions to avoid transient anoxia. Each condition was represented by triplicate or quadruplicate cores. The relationship between methyl Hg evolved and chironomids present was examined by rendering all of the cores anoxic over night in an attempt to kill any chironomids present at sampling. Fresh, live chironomids were added back to the overlying water of some of the oxic cores. At sites OA-04 and UA-03 0.6 mM molybdate was added to the water over some of the anoxic cores in order to see if inhibition of Hg methylation in sediments with *in situ* Hg levels was similar to the experiments carried out in Hg amended slurries. The molybdate experiments were not carried out on cores collected from site OA-01 because the effects of the presence or absence of the mine site floc was deemed of greater interest. This was, in retrospect, a missed opportunity.

In addition to measuring the evolution of methyl Hg, the *in situ* gradients of sediment total and methyl Hg were measured in the second microcosm experiment. Cores from sites OA-01 and UA-03 were sectioned at 0.5 or 1 cm intervals and each section was analyzed for total and methyl Hg. The top three centimeters of sediment at sites OA-01+floc and OA-04 were also analyzed for total and methyl Hg. These measurements were taken for two reasons; 1) to understand the gradients of total and methyl Hg in surficial sediments of Clear Lake and 2) to help estimate the mass balances for Hg in the microcosm systems.

Results of Experiment #2

Results from the fine scale analyses of the surficial sediments of sites OA-01 (Oaks Arm) and UA-03 (Upper Arm) are presented in Figure 2. At site OA-01, total Hg decreased dramatically (from roughly 200 μg/g to approx. 125 μg/g) within the top centimeter of sediment. Site UA-03 showed no decrease in total Hg concentrations with depth, maintaining an average of 2.25 μg/g over the top three centimeters. Replicate measurements at both sites agreed well. The dramatic decrease in total Hg in the top centimeter of sediment at site OA-01 is intriguing and one of several hypotheses may explain this result : 1) Lowered erosion of high Hg sediments into the lake as a result of the EPA directed mine site

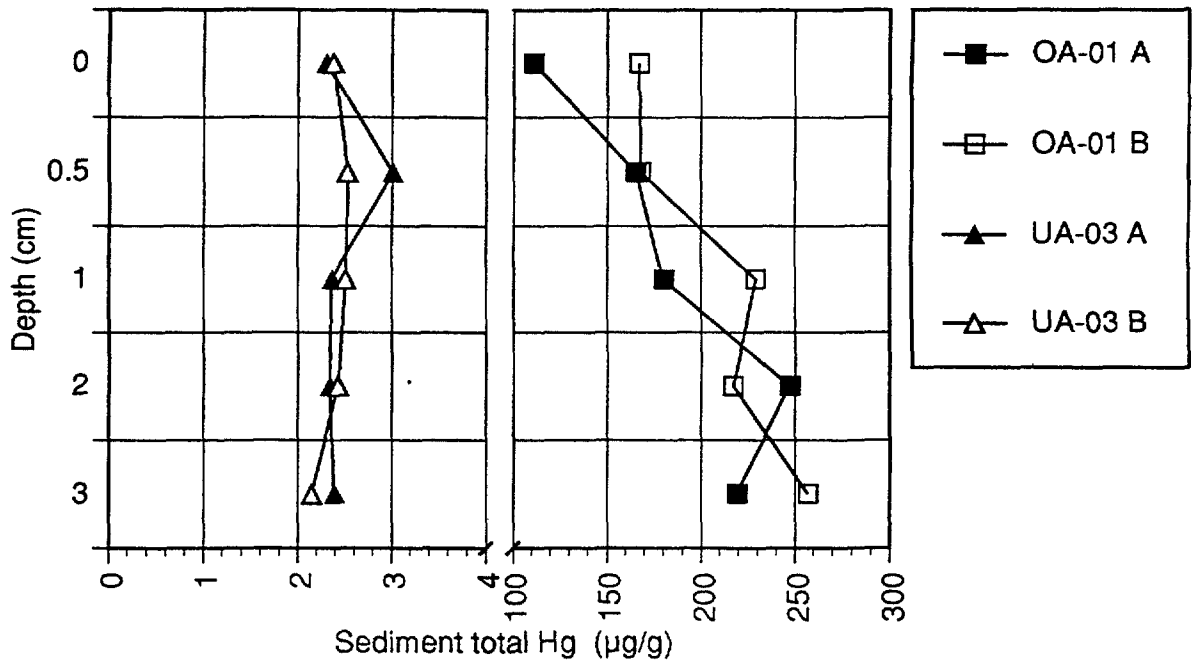

Figure 2. Total mercury microstratigraphy.

remediation in 1992, 2) Recent coverage of sediments by lower Hg content material (e.g. as a result of heavy rains and floods), 3) High activity of Hg methylation and / or volatilization in surficial sediments (unlikely given the low amount of methyl Hg evolved in these experiments). However, the sediments at this site are subject to scouring and replacement due to wave action and therefore may not be representative of true Hg gradients in the Oaks Arm. On the other hand, the deeper (below 1 cm) sediments from OA-01 did show total Hg values of 200 µg/g, a normal level for sediments at this site. Possible future experiments to discriminate among these hypotheses include analysis of total Hg gradients in deeper sediments at a number of sites in the Oaks Arm at different times of the year.

Fine scale analysis of sediment methyl Hg at sites OA-01 and UA-03 yielded trends similar to the total Hg described above (Figure 3). At site OA-01 methyl Hg decreases by 2-3 ppb in the top 3 centimeters of sediment. At site UA-03 methyl Hg remains around 2 ppb throughout the top three centimeters of sediment. Again the replicates agreed satisfactorily. If the methyl Hg at site OA-01 were soluble in pore water and had an approximate diffusion coefficient of 5×10^{-6} cm^2/second (as calculated for a Hg contaminated estuarine site; Bothner, et al, 1980), the predicted 4-day flux into the overlying water should have been more than 1,000-fold greater than the observed net flux out of the sediment. From this discrepancy it can be assumed that most of the measurable sediment methyl Hg is not available for diffusion out of the sediment.

Although not yielding information about the gradients of total and methyl Hg within the sediments, the analysis of bulk surficial sediments at sites OA-01+floc and site UA-03 were nonetheless interesting. Sites OA-01+floc and OA-04 yielded similar total Hg values (about 30 ppm) (Figure 4A). It should be remembered that the nearby site OA-01 had total Hg values of 100-200 ppm. It is possible that the low total Hg concentrations at the OA-01+floc site is due to mixing of the high total Hg surficial sediments of OA-01 with low total Hg floc sub-

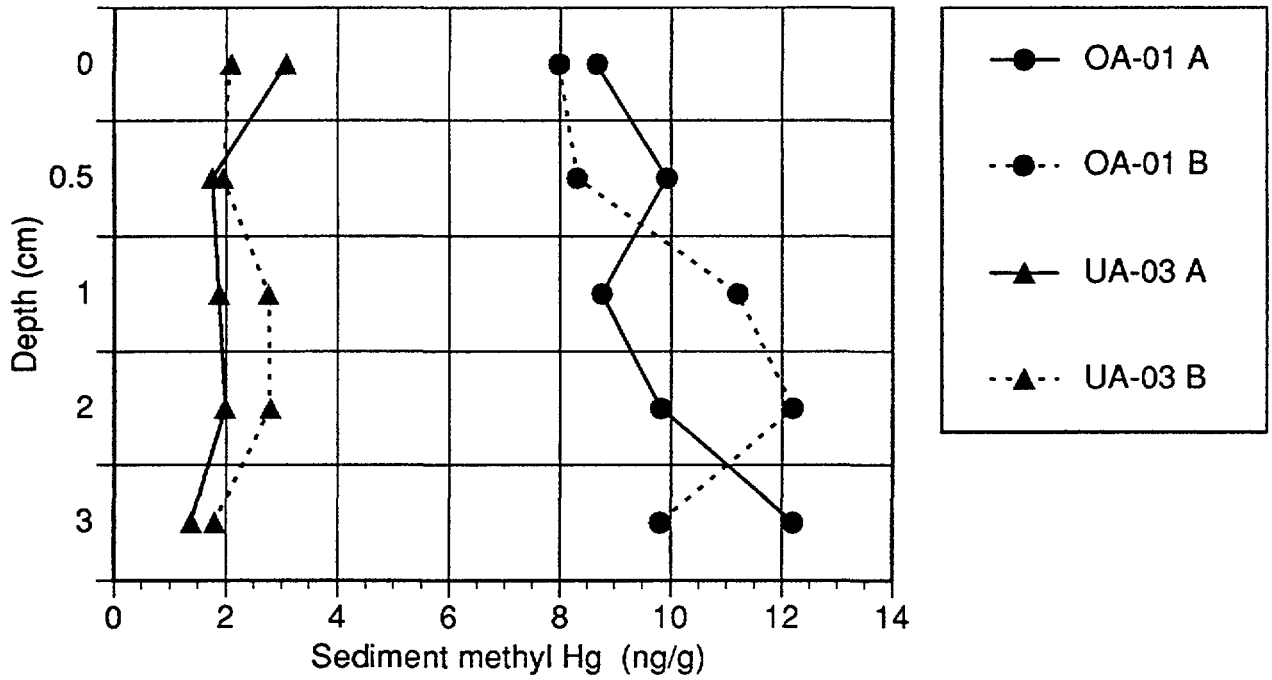

Figure 3. Methyl mercury microstratigraphy.

stance (about 30 ppm, Suchanek, et al, special interim report 9/95). However, it should be remembered that at least, in a representative core of the OA-01+floc site the white floc substance was buried 11 centimeters below a black and extremely sulfidic sediment. Again, because of the wave action induced scouring at this site, more measurements would have to be carried out before any conclusions about sediment mixing could be made.

Methyl Hg levels in the top three centimeters of sites OA-04 and OA-01+floc were similar (although site OA-04 is slightly lower) and comparable to site OA-01 (Figure 4). It is interesting that sites OA-04, and OA-01+floc, sites that contain roughly seven times less total sediment Hg, have methyl Hg levels comparable to site OA-01. This increase in ratio of methyl Hg to total Hg is much larger at site UA-03 which has 15-100 times less total Hg than the Oaks arms sites, but only 4-5 times less methyl Hg (Figure 5). The reason for

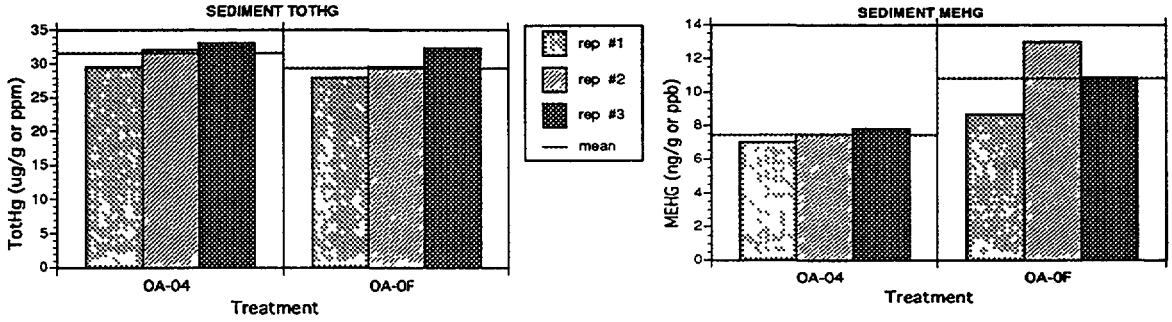

Figure 4A and 4B. Total and methyl mercury in bulk sediments.

this trend in the ratio of methyl Hg to total Hg is not clear, but these observations are consistent with previous reports on Clear Lake.

Analysis of the effects of anoxia, molybdate and chironomids upon methyl Hg evolution from sediments was difficult due to the high variability among the replicates. This high variability among replicates was similar to that observed in the earlier microcosm experiment. However, it can be clearly seen that the presence of floc at site OA-01+floc greatly increases the amount of methyl Hg evolved into water overlying the microcosm (Figure 6). In fact, the presence or absence of floc largely swamped the effects of any of the other environmental conditions tested. In light of what is already known about the floc substance (freshly precipitated Fe/Mn/Al oxides that have high sulfate and methyl Hg concentrations but relatively low total Hg) there are several hypotheses to explain its stimulation of methyl Hg evolution : 1) Hg bound to floc is more available for methylation than ambient Hg found in the sediments, 2) Floc provides a high surface area for colonization by Hg methylating bacteria, 3) Floc has a high sulfate concentration, allowing the stimulation of sulfate reducing bacteria which may, under these conditions become the dominant Hg methylating organisms, 4) Floc may become heavily loaded with organics thus making it a good environment for Hg methylating organisms, 5) Floc may be capable of methylating Hg via an abiotic mechanism. These are not mutually exclusive hypotheses. Although none of these hypotheses can be eliminated with data currently in hand, the question of the bioavailability of Hg bound to floc is very important. Not only would highly available Hg make sites with floc extremely high in methyl Hg, the transportability of floc by currents suggests a method of moving bioavailable Hg throughout the entire lake system. Sediment slurry experiments described previously (in chapter 5A of this report) suggest that Hg in the lake is largely unavailable for Hg methylation but that any added Hg^{++} is readily methylated. In light of these results it is important that the possibility of floc being an easily transported form of bioavailable Hg be thoroughly examined.

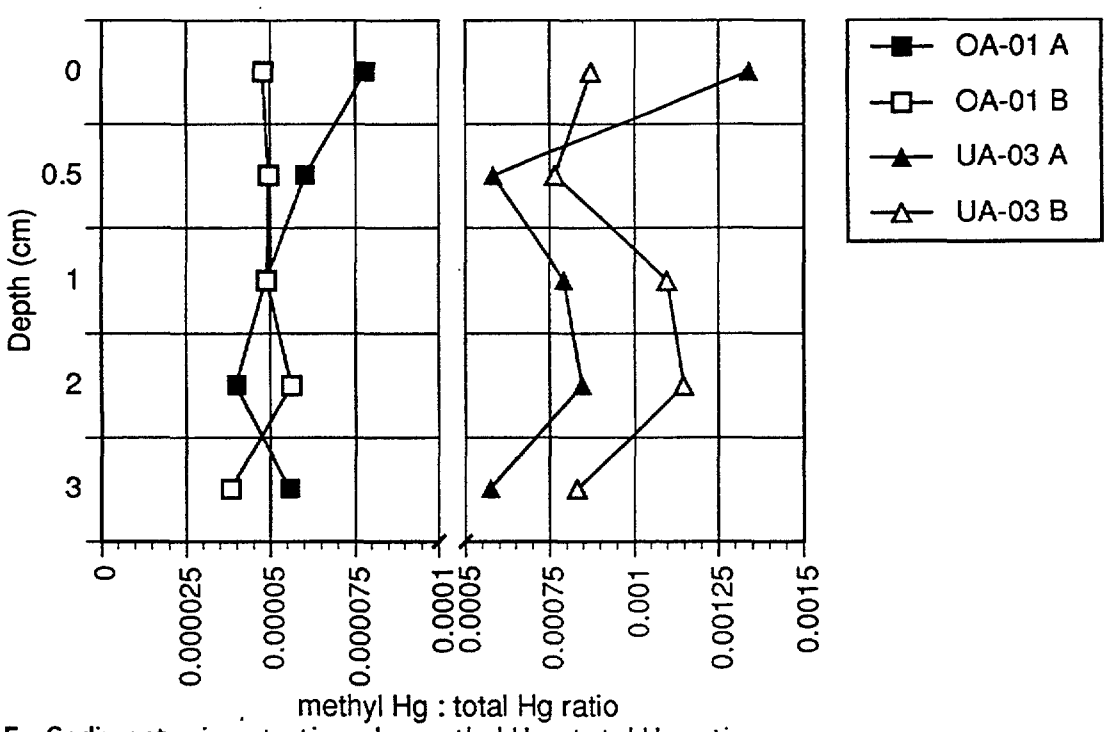

Figure 5. Sediment microstratigraphy methyl Hg : total Hg ratio.

When examined separately, the effects of anoxia, molybdate, and chironomids varied with each site. Cores incubated under anoxic conditions always evolved more methyl Hg than the water only controls. Anoxia, as compared to oxic conditions appeared to stimulate evolution of methyl Hg into overlying water at sites OA-01+floc, OA-04, and UA-03 (Figures 7A,B,C,D). Anoxia had a proportionately greater effect upon methyl Hg evolution at site UA-03 than the other two sites. However, anoxia, as compared to air sparging, had little or no effect at site OA-01 (Figure 7B). Cores under oxic conditions at sites OA-04 and UA-03 did not evolve methyl Hg to a greater extent than water only controls. It should be remembered that the effects of anoxic vs. oxic conditions may only apply to the water column and the surficial 0.5 cm of sediment (Sweerts, et al, 1986). High ambient sulfate sites such as OA-01+/- floc may have been largely anoxic due to more sulfide produced by sulfate reducing bacteria and therefore the presence or absence of air sparging may have had little effect. Conditions at all four sites at the time of collection were oxic at 0.5 m above the sediment water interface. In the first microcosm experiment the effects of anoxia were comparable in that anoxia, as compared to oxic conditions, increased methyl Hg evolution. These data suggest that transient anoxic conditions at the sediment water interface observed in late summer stimulate Hg methylation. This interpretation correlates with increases in Hg methylation potential, and methyl Hg concentrations measured in the biota and the water in the late summer.

Molybdate was added to anoxic cores from sites OA-04 and UA-03 in order to test the effects of inhibition of sulfate reducing bacteria. Like the results of anoxia vs. oxic conditions, the effects of molybdate were variable. At site OA-04 molybdate treated cores showed only a slight decrease in the amount of methyl Hg evolved as compared to the anoxic cores (Figure 7C). However, at site UA-03 molybdate treated cores evolved almost 75% less methyl Hg than the anoxic cores and, like the oxic cores at this site, were indistinguishable from the water only controls (Figure 7D). These data are puzzling in

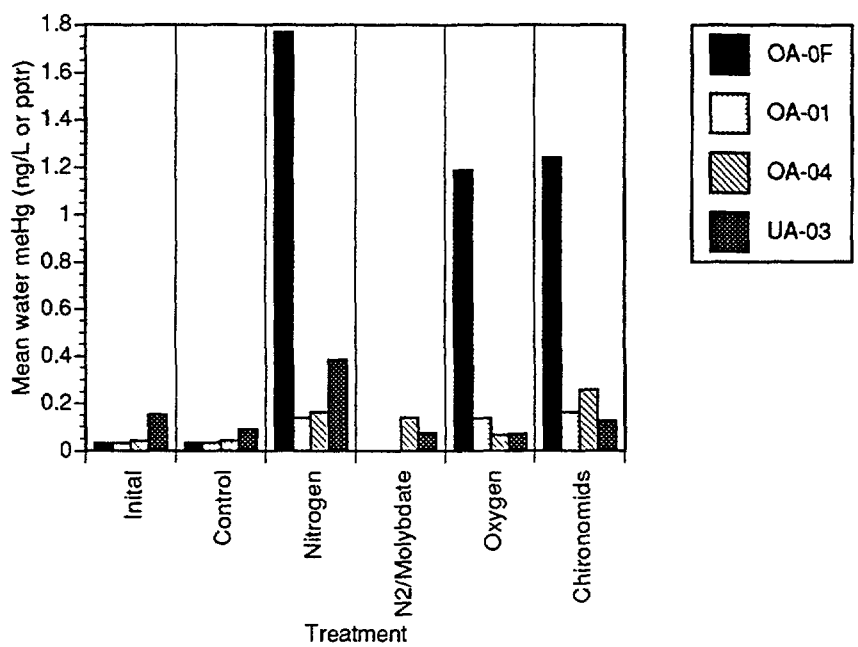

Figure 6. Average methyl mercury evolution from all cores.

light of the data from the sediment slurry experiments in which the concentration of molybdate that effectively inhibited sulfate reduction had little effect on Hg methylation. It is possible that conditions in anoxic sediment cores are very different from those in Hg amended anoxic sediment slurries. Alternatively, these results may indicate that factors / microorganisms controlling Hg methylation may differ with lake site and season. It is possible that sulfate reducing bacteria were responsible for more Hg methylation at site UA-03 than at site OA-04 at the time of this experiment. In retrospect it would have been useful to see the effects of molybdate upon OA-01 samples with and without floc because such an experiment might have yielded some information as to the nature of the floc's stimulation of methyl Hg evolution. The effects of molybdate upon Hg methylation in freshwater systems at ambient Hg concentrations have been controversial in the literature. In the only other experiment at ambient Hg concentrations, Gilmour and Riedel (1995) saw extremely variable effects of molybdate upon methyl Hg production in sediment cores from a single lake site. Molybdate treated cores resulted, on average, in a larger net decrease in methyl Hg than unamended cores, indicating net demethylation. However, the variability in this experiment was such that it is difficult to confidently assign a clear effect of molybdate. Gilmour and Riedel attribute the variability in their system to variable ambient methyl Hg levels, clearly not the case in the experiments described above (Figure 7 A,B,C,D).

The addition of live chironomids to the microcosms also yielded variable results. At sites OA-01 with and without floc the addition of chironomids had no effect upon methyl Hg evolution when compared with the air-sparged (oxic) microcosms, although the microcosms with

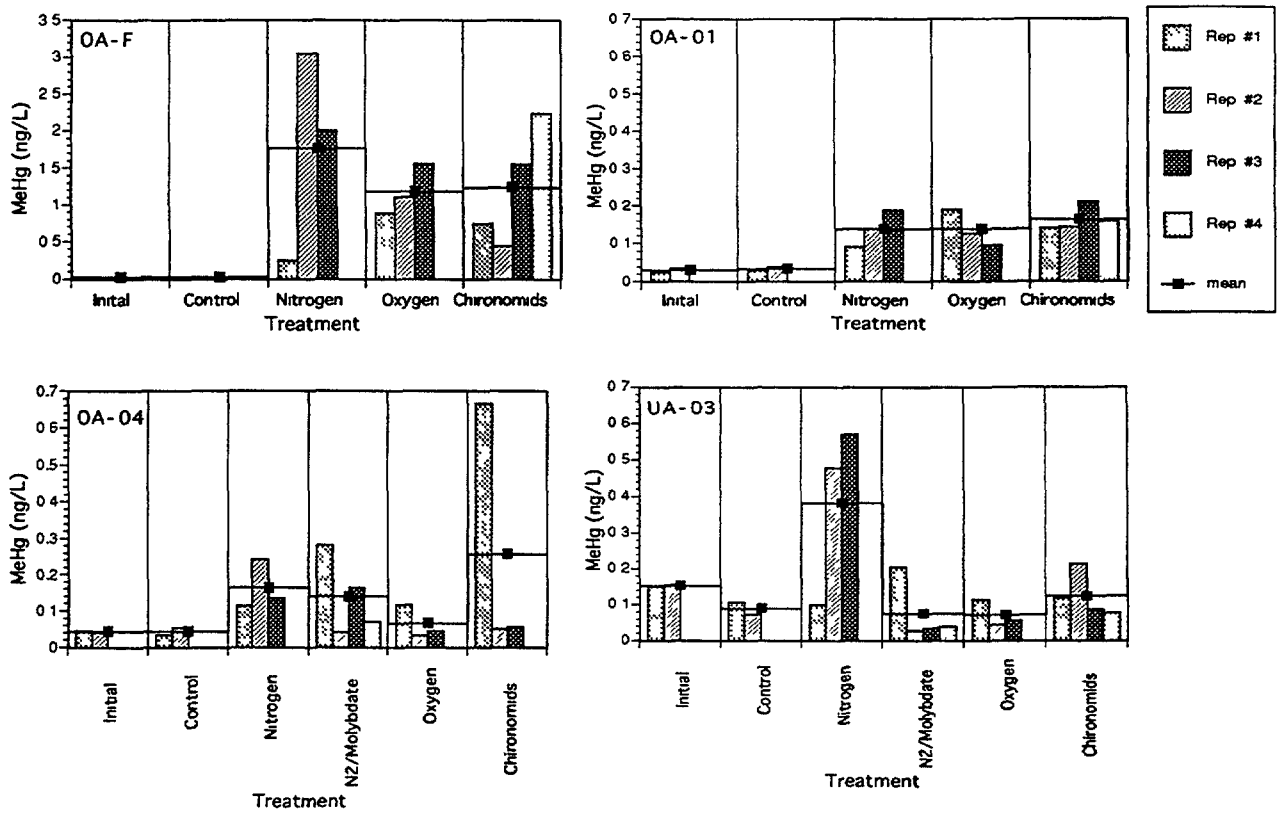

Figure 7A,B,C,D. Methyl mercury evolution from all cores, sites, and replicates. (Note: Different scale for OA-F).

floc evolved a great deal more methyl Hg (Figure 6). At sites OA-04 and UA-03 the addition of chironomids resulted in a slight increase in methyl Hg evolution as compared to the air sparged microcosms (Figures 7C&D). In the case of OA-04, the chironomid amended microcosms evolved more methyl Hg than those under anoxic conditions, although this is due to a single, extremely productive replicate. Considering these results the effects of chironomids on methyl Hg evolution from sediment are still largely unknown. The variability in results could be due to differing amounts of surviving chironomids within the sediments, particularly in those cores from the high sulfide sites, OA-01 +/- floc.

Evolution of total Hg to the water overlying the sediment cores was measured for the oxic and anoxic conditions of sites OA-01 and UA-03 (Figures 8A,B). Again the results were variable. Interestingly, considering the vast difference in ambient sediment total Hg concentrations, sites OA-01 and UA-03 had comparable water total Hg values, even after the water (collected at the respective sites) had been in contact with the respective sediments for 4 days. Evolution of total Hg was similar for both sites. Under anoxic conditions total Hg did not change appreciably from initial concentrations. However, under oxic conditions total Hg increased in the water column to a concentration 3-4 times that of the initial concentrations. These results are interesting when compared to the results for methyl Hg. At OA-01 there was an increase in methyl Hg in both the oxic and anoxic conditions, whereas at site UA-03 there was an increase in methyl Hg only under the anoxic conditions.

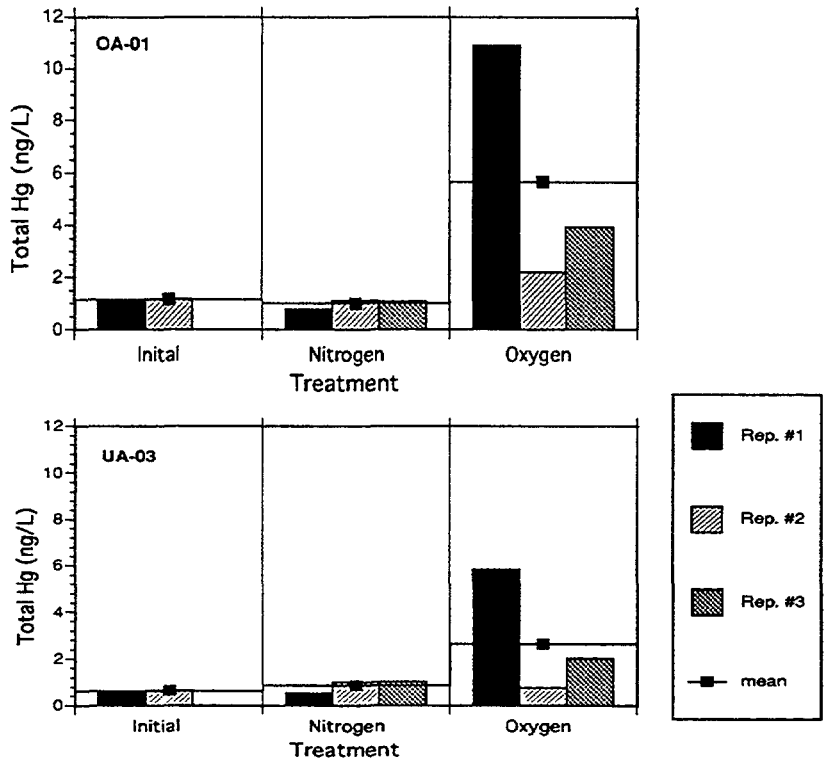

Figure 8A, B. Evolution of total mercury into the water column for sites OA-1 and UA-3.

Granted, these increases in methyl Hg are not significant when compared to total Hg in the water. However, the decreased proportion of methyl Hg under oxic conditions is suggestive of predominately aerobic demethylation reactions not active under the anoxic conditions. Demethylation is generally considered to be an aerobic process although anaerobic demethylation has been shown to occur in anoxic sediments (Oremland, et al, 1995). Alternatively, the greater evolution of total Hg into the water column under oxic conditions may be the result of continual resuspension of sediments during the constant sparging of the oxic conditions. This resuspension may have caused a greater net dissolution of sediment total Hg to the water column. It would have been interesting to see if the presence of floc had a similar effect upon the evolution of total Hg.

Although the microcosm experiments yielded highly variable results they are nonetheless valuable. First, these experiments have indicated that mine site floc likely plays a large role in the production of methyl Hg in Clear Lake. Secondly, they have supported the concept that different sites in Clear Lake may behave very differently with respect to Hg and methyl Hg dynamics in sediments and the water column. The only comparable experiments in the literature, that specifically measure methyl Hg evolution from vertically stratified cores under ambient Hg concentrations, are those of Henry et al (1995) in Lake Onondaga. The results from that study are comparable to the results from the current Clear Lake study in that anoxia leads to increased methyl Hg evolution and that the results were highly variable. Arguably, the results from Onondaga show more variability than those from Clear Lake. Additionally, it can be seen that Clear Lake is a more productive lake in terms of methyl Hg evolution (Table 1). However, conclusions based upon extrapolating methyl Hg evolution from a 38.5 cm^2 sediment core to hundreds of square meters of lake bottom should be considered extremely tentative. The source of the variability in these experiments is puzzling. Henry, et al do not

Table 1. Comparison of methyl Hg flux from sediments in Clear Lake and Lake Onondaga[a].

Methylmercury Evolution into Water Overlying Sediment Cores
(ng Methylmercury/m^2/day)

Site	Anoxic	Anoxic + MoO4	Oxic	Oxic + Chironomids
OA-01 CL	2.9 [b] (n=3)	ND	2.9 [b] (n=3)	3.9 [b] (n=4)
OA-01 + floc CL	34.1 (n=3)	ND	23.4 (n=3)	23.4 (n=4)
OA-04 CL	3.1 (n=3)	2.7 (n=4)	1.2 [c] (n=3)	4.9 (n=3)
UA-03 CL	7.6 (n=3)	1.5 [c] (n=4)	1.36 [c] (n=3)	2.7 (n=4)
S4A Onondaga [d]	ND	ND	(-0.30) +/- 3.4 (n=2)	ND
S73A Onondaga [d]	ND	ND	0.90 +/- 0.45 (n=3)	ND
S90A Onondaga [d]	38 +/- 10 (n=3)	ND	ND	ND

a Henry, et al (1995)
b Only slightly greater than water only controls.
c Less than water only controls.
d Oxic or anoxic refers to in situ conditions that were then reproduced in the laboratory experiment, not to conditions experimentally imposed upon samples.
CL = Clear Lake
ND = Not Done

describe their sampling methods in detail but every effort was made in the current study to reduce variability. Cores from the same Ekman grab were marked and dispersed as evenly as possible among the treatments and sediments were adjusted as necessary so that each core had the same amount of sediment. Incubation temperatures stayed constant +/- 2°C and every effort was made to ensure even sparging of the cores with N_2 or air. During collection of water from the core surfaces fresh tubing was used for each treatment and each sample was filtered in a seperate acid cleaned filter apparatus. Water was kept in the plastic filter apparatus for about the same amount of time in all cases. Gilmour and Riedel (1995) attributed variability in their study of methyl Hg production in lake sediments at ambient Hg to differences in ambient methyl Hg concentrations. As noted previously in this discussion, results from these experiments indicate that methyl Hg is unlikely to be a source of in-site variability in Clear Lake. More likely, variability in Clear Lake arises form "patchiness" in the sediments. Porewater sulfate concentrations vary among cores collected from a single Eckman grab (Mack, data not shown) so it is reasonable to assume that organics, drivers of Hg methylation, might be similarly variable.

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Evidence of a Hydraulic Connection Between the Herman Pit and the Oaks Arm of Clear Lake (California)

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Key words: Herman Pit, acid-mine drainage, Rhodamine, mercury transport

The observation of large areas of flocculent material immediately offshore from the Herman Pit in the Oaks Arm during the spring and summer of 1995, suggested the possibility of direct hydraulic connection between these two water bodies. To test this idea, a fluorometric dye tracer study was commenced in April, 1997. Rhodamine-WT was uniformly distributed into the Herman pit at an initial concentration of approximately 3 ppb. Over the next 2 months, dye concentrations were monitored in the Herman Pit, at observation wells between the mine pit and the Oaks Arm, and within the Oaks Arm. Within a week of the dye release, significant concentrations of Rhodamine were being detected at specific wells. Through the subsequent weeks, many of the wells indicated the presence of Rhodamine, albeit at different concentrations and at different times. Based on the measured decay rate of Rhodamine in light, the observed concentrations in the Herman Pit, and measured evaporation rates, an estimate of the flow of water from the Herman Pit and into the Oaks Arm can be determined.

In Vitro Production of a White Coagulant Material from the Mixing of Herman Pit and Clear Lake Waters Similar to that Observed in the Field, and Remedial Suggestion.

Authors: Bob Reynolds[1], Ross Kauper and Hilary Keller; LCAQMD, 883 Lakeport Blvd., Lakeport, California.

Key words: Sulfur Bank, floc, coagulant, acid, float, overflow

Abstract

To investigate the origin and nature of a white material that lined the shoreline, beaches and vegetation in the Sulfur Bank Mine area subsequent to the 1994/95 flooding. Observations and in vitro experiments were conducted. A several months long unreported overflow of Herman Pit was brought to the attention of the RMC[2] in May 1995. Substantial erosion and deposits of a white material onto the shoreline beach, rocks and vegetation was observed. Material analyzed using X-ray Fluorescence[3] had high levels of aluminum and sulfur, as well as traces of mercury and arsenic. Herman Pit again overflowed in November 1995. During the overflow, waters collected from Herman Pit and Clear Lake were immediately transported to the AQMD laboratory, and mixed at ratios from 10:1 to 1:10 to simulate the actual mixing of waters at the site. The pit water was pH 3 and lake water pH 8. Pit to lake water ratios of 1:1 through 1:10 readily formed a white coagulant similar to that appearing at and surrounding the Sulfur Bank mine area. At lower ratios (1:5) solids were not easily observed. Addition of 10N NaOH to pit water caused a reddish-white precipitant as pH shifted to less acid, with substantial solids at a pH of 8. Material of varying density from a float to a floc have been observed. The experiment was repeated in substantially more detail and variation by UCD researchers[4]. Video surveys of the site and experiments were made, and remedial suggestions are offered.

Introduction

The Lake County Air Quality Management District (AQMD) is charged with management of all stationary sources of air pollution required under federal, state and local laws within our air basin. Superfund sites however are given an exception in law to this regulatory authority, though the AQMD has a continuing obligation to participate and comment on proposed actions.

The AQMD reviewed and commented on the initially proposed remedy that was based upon insignificant water inflow or outflow at the site, that only 50 grams per year of Hg was entering the lake from the site, and for which EPA ignored the possible complications of on site water and gas escaping from natural springs and, or, improperly abandoned geothermal wells. The AQMD staff became more interested in both EPA's reasoning in selecting the proposed remedy, and the curious lack of information about the behavior of sources located at the site. This interest was increased when during the 1994/95 period Herman Pit overflowed for several months and went unreported, until reported by a public member at the Resources Management Committee meeting. An inspection of the site indicated substantial overflow and likely deposition of a white material containing many components similar to those of Herman Pit. At this time UCD was doing no mine site work, but expressed concern with the white material deposited in the lake adjacent to the mine site and the elevated Hg present in it.

AQMD staff agreed with the encouragement of the local elected officials to monitor more carefully for overflows, and seek to encourage EPA to identify the source of the white material. This was done by staff randomly inspecting the site and making observation when in the area for other reasons. The pit overflowed again in November 1995 and that event enabled the in vitro experiment.

This early work was repeated twice by AQMD staff and reported to colleagues at UCD (footnote) whom repeated the experiment and more extensively characterized the floc formed. It has assisted in a bringing about a new understanding of the interactions likely between the mine site and the lake, and further work that the interested reader is referred to.

Methods

XRF analysis of bark from a tree subjected to the water from Herman Pit during an overflow event was performed courtesy of Bill Davis of the California Air Resources Board (ARB), at the ARB laboratory, located in Sacramento, California.

Grab samples were obtained from Herman Pit (Sulfur Bank Mercury Mine) and from Clear Lake. Each sample was initially assayed to detect the presence of sulfide using the La Motte™ sulfide test. The assay was performed according to the manufacturer's instructions.

National Testing Laboratories ("Watercheck", Claveland Ohio) was utilized to perform a Title 22 water quality analysis on the same waters collected at the pit and at the lake.

All pH measurements were performed using a Cole Parmer pH electrode (model # 5941-00) which was calibrated using a three point calibration at pH 4.01, 7.00, and 10.00.

10 ml aliquots of Pit water were diluted with varying proportions of Lake water, and mixed thoroughly. The pH of each sample was read and recorded at the time of dilution, and the samples were visually inspected for precipitate formation. The samples were then left covered, at room temperature, overnight. After standing at room temperature overnight, the samples were visually inspected again, and the pH of each was determined.

Portions of pit water were also titrated with 10 N NaOH to pH end points of 5 and 8 and visual observations made on the coagulant that formed.

The dilution experiment described above was subsequently repeated 3 days later using the same samples, which had been stored at 4°C by separate AQMD staff.

Results

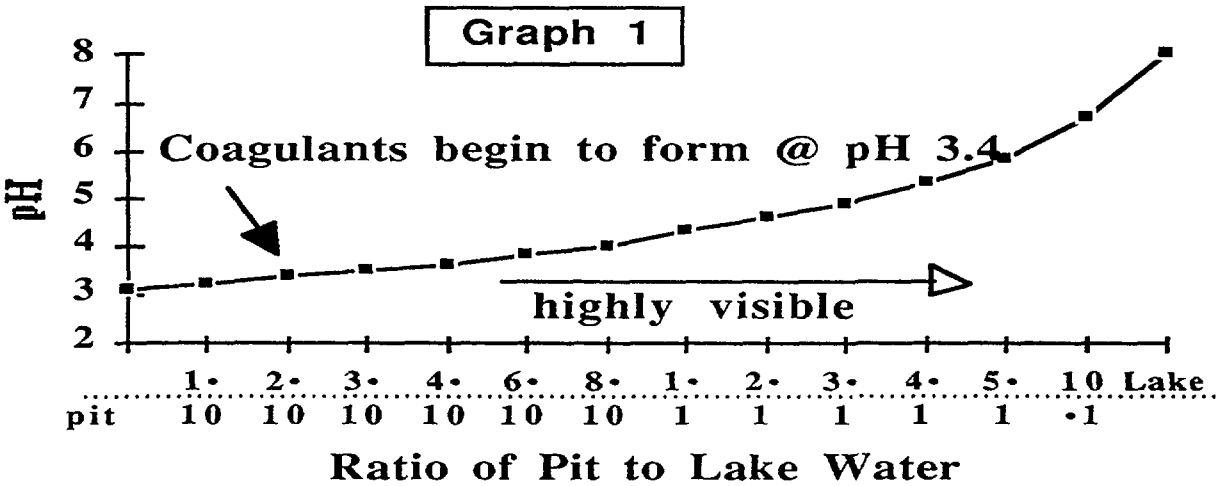
Material gathered from a tree trunk covered with the unidentified white materials was analyzed using XRF. The spectrum is presented in Figure 1 A & B. Aluminum and sulfate were found in high concentrations while many trace components such as mercury and arsenic were also detectable.

The National Testing Laboratories and the XRF scans indicated material of similar origin, and chemical constituents.

The results of titrating the Herman Pit water (acid) with Clear Lake Water (base) was carried out and are shown in Graph 1 and Table 1.

Table 1 - pH Resulting from Pit and Lake Water Mixtures

ratio	**Pit**	1:10	1:5	3:10	2:5	3:5	4:5	1:1	2:1	3:1	4:1	5:1	10:1	**Lake**
ph	3.1	3.2	3.4	3.5	3.6	4.8	4.0	4.3	4.6	4.9	5.3	5.8	6.7	7.8



Coagulant material formed at ratios of pit:lake water as low as 1:5 and very visible with the unaided eye at 1:2 and a mixture pH of 3.4, and more extensively as lake water content was increased. The material was white in color and similar in appearance to that observed in the field. The coagulant formed when pit water was adjusted to pH 5 was white and similar to the in vitro and field coagulants. The coagulant formed when pit water was adjusted to pH 8 was red brown and similar to coagulants noticed on occasion in the pit. Videos were made of the coagulants. The coagulant would float or sink depending on agitation and the method of mixing.

During the fall of 1995 through the spring of 1996 additional observations were made and visual evidence (video) taken at the mine site. In general the coagulant white material could be observed wherever mine site water interfaced with offsite water.

Discussion

Initially other scientists investigating the phenomena had stated that the in vitro mixing of Herman Pit and Clear Lake waters did not form a white floc material similar to that deposited in the lake. Since the opinion appeared to be of questionable validity judged from AQMD staff's field observations, the AQMD took on the task of demonstrating an in vitro formation. We were surprised to observe both the extent and similarity of material concurrently being formed artificially in the lab and in the lake during November of 1995. The adjustment of pit water pH to near 5 using NaOH also showed a material similar to that of the in vitro experiments. When pH was adjusted to 8 using NaOH a material reddish-brown in color was produced, and this material was very similar to that noted in the vicinity of old geothermal wells. The pH of the water adjacent to an abandoned well has been measured to be markedly more alkaline than the general pit. Water analysis performed by National Testing Services, Inc. showed that both the pit water and Lake water sampled at the point of overflow contain relatively high concentrations of Al, Fe, Mn, SO4 and total dissolved solids. Considerable Hg was picked up as the overflow traveled to the lake. Additional observations and inspections around the drainage portion of the mine site perimeter showed material similar to the in vitro experiment formed whenever the mine site waters mixed with the offsite waters. Though the experiment does not establish with certainty the mine site as the source for the lake floc, these experiments served to generate additional interest and consideration to both repeat the initial experiments and enlarge markedly upon their meaning. In this regard the authors note that UCD's Clear Lake Research Center Research Group[5] have always given us more than adequate credit for our effort and have much expanded and improved upon this work.

Given the continued observations on site by AQMD staff, new information made available by EPA funded research, and the debate that has occurred as to the proper direction for EPA to pursue in its continued effort, we offer the following suggestions. We realize that all have probably been advocated by others, and that while they are not based upon hard science they are based upon reasonable field observations and concerns that have been expressed by participants including ourselves.

Suggestions for further Research:

1) investigate and quantify, by physical measurement, the historic springs and abandoned geothermal well water flows, instead of accepting data that is more than 30+ years old;
2) characterize the pit geochemistry, to include spatially differentiating the water around the old geothermal wells;
3) determine whether oxidation of hydrogen sulfide within the pit and other waters prevents significant release of H2S, and if this is the source of the sulfate;
4) determine the treatability and estimate the cost of treating pit water to an acceptable level for reclaimed use (Geysers pipeline project, though tardy, should receive an initial emphasis); and
5) identify alternative pit water uses possible on site and their cost;
6) inspect the bottom and banks of the pit, especially in the vicinity of geothermal wells and gas expulsions, to determine any manifestations of water or gas flows;
7) perform a second tracer test, *using an alternative tracer*, to determine pit water inflows and outflows, and attempt to discern entry into the lake; and
8) characterize the entire site as to levels of toxic components easily mobilized with an emphasis on the old retort area;

Suggestions for interim mitigation steps :

1) prevent as much surface and ground water from entering the site as is practical and contain any passage of water through the site in a manner which prevents erosion, and/or dissolving of components while passing through the mine site;
2) determine the treatability of the water in Herman Pit for possible reuse;
3) give serious consideration to the Oaks Sewer District proposal (Attachment I, a draft proposal was prepared for discussion by Special District and AQMD staff which provides more detail);
4) maintain Clear Lake at as high an elevation as is workable to avoid the detrimental effects of any hydraulic head;
5) Inspect and evaluate all four abandoned geothermal wells integrity and record of abandonment to ensure they are not an indirect or actual source of inflow and properly abandon if needed; and
6) provide, as much as possible, site contract work to local contractors and the UC, who have a presence and interest in the Clear Lake community.

Conclusions

That the white material (commonly referred to as floc) present in the lake, along the beaches and wetlands adjacent to Sulfur Bank Mine, can be demonstrated to form in vitro by mixing Herman Pit and Clear Lake water. This is done over a wide range of mixing ratios that occur in situ.

This was clearly demonstrable using waters collected from Herman Pit on November 14, 1995 during an overflow into Clear Lake and the adjacent lake waters.

Further, from observations at and adjacent to the site the material is present in all areas where water exits the Sulfur Bank Mine Site and mixes with uncontaminated waters.

From observation the material behaves as a coagulant that floats or sinks, a floc that takes water column suspended material down with it, and a material that forms crystalline rod-like material when dried or when attempts to redissolve with acid are made.

ATTACHMENT #1

SULFUR BANK

Objective: To minimize or prevent Mercury (Hg) entering Clear Lake from the Sulfur Bank Mine (SBM) tailings leach and Herman Pit. To do such after gaining an adequate scientific understanding of the sources of Hg and the interactions within the pit, leachate flows into the lake and the importance to the ecology of the lake.

Assumptions: 1) That pit water enters the lake through both surface and subsurface flows; 2) That ground water, spring and surface waters flowing through tailings, abandoned shafts and/or other deposits enter the lake and pit with appreciable leached and eroded components; 3) That it is necessary to quantify, by physical measurement, the historic springs and abandoned geothermal well water contributions to those of meteoric water in the pit, instead of accepting data that is more than 30+ years old; 4) that the pit geochemistry itself is poorly understood and characterized, and chemical interactions within the pit prevents significant release of H2S and assists in immobilizing Hg; and 5) that EPA will give additional thought to the remedy presently proposed prior to implementing.

Critical Issues/Needs: 1) Determine a reasonable method and the cost of treating pit water to an acceptable level for reclaimed use in the Geysers pipeline project. The issues to be resolved are the solids/floc formation either in the pipe or re-injection formation, corrosivness and long term assurance/determination that such reclaimed use is not detrimental to the use; 2) Determine the volume necessary to be treated to avoid offsite flows and perhaps lower the pit water level to enable flow of on site waters into the pit, or pumping captured flows in collection basins; 3) Determine alternative pit water uses possible on site and their cost; 4) Investigate the physics and geochemistry of the pit. Does it significantly precipitate mercury sulfides and oxidize hydrogen sulfide? Does the pit communicate with the lake? Will the behavior of the springs or the abandoned geothermal wells change if the water head is removed?; 5) Determine if the EPA and/or state can support a long term management approach; and 6) Develop a means to reach a reasonable formal/legal agreement between potential participants on each parties mutual long term roles and obligations, if such a mangagement approach proves more feasible.

Proposal : 1) Determine the economic feasibility of treating pit waters and possibly combining with the Clearlake Oaks CWD and Geyser's pipeline projects to dispose of treated pit effluent and maintain the pit level at an elevation level near or below that of the lake; 2) provide for re-direction of off site waters now entering the site around the site, or via contained flow to the lake or to the pipeline; and 3) direct all on site rain water to flow to the pit for subsequent disposal of excesses to prevent overflow into the lake.

The pit pumping and treatment would have to be done with caution, but dropping the pit level below lake level would offer the option of evaluating the abandon geothermal well(s) contribution, spring flows of gas/waters, evaluation of bottom deposits in the pit and inflow of lake waters (correlated with the ability to outflow). It could lead to a more rational, careful evaluation of a final decision to backfill the pit and/or attempts to manage the site in alternative ways.
—3/17/96————Bob R & Mark D - - For discussion only! — Minor Changes 4/22/96

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[2] Resources Management Committee, Dennis Pluth

[3] Dr. William Davis AMD/ARB

[4] 4 Tom Suchanek, Peter Richerson, Laurie Mullen, Linnie Brister, Jesse Becker, Cat Woodmansee, et. al.

[5] 5 Tom Suchanek, Peter Richerson, Laurie Mullen, Linnie Brister, Jesse Becker, Cat Woodmansee, et. al. 1997. Discovery of a Flocculent Material Associated with the Sulphur Bank Mercury Mine. Report prepared for EPA Region IX. Interim Final Report: The Role of the Sulphur Bank Mercury Mine Site (And Associated Hydrogeological Processes) In the Dynamics of Mercury Transport and Bioaccumulation Within the Clear Lake Aquatic Ecosystem

Figure 1A

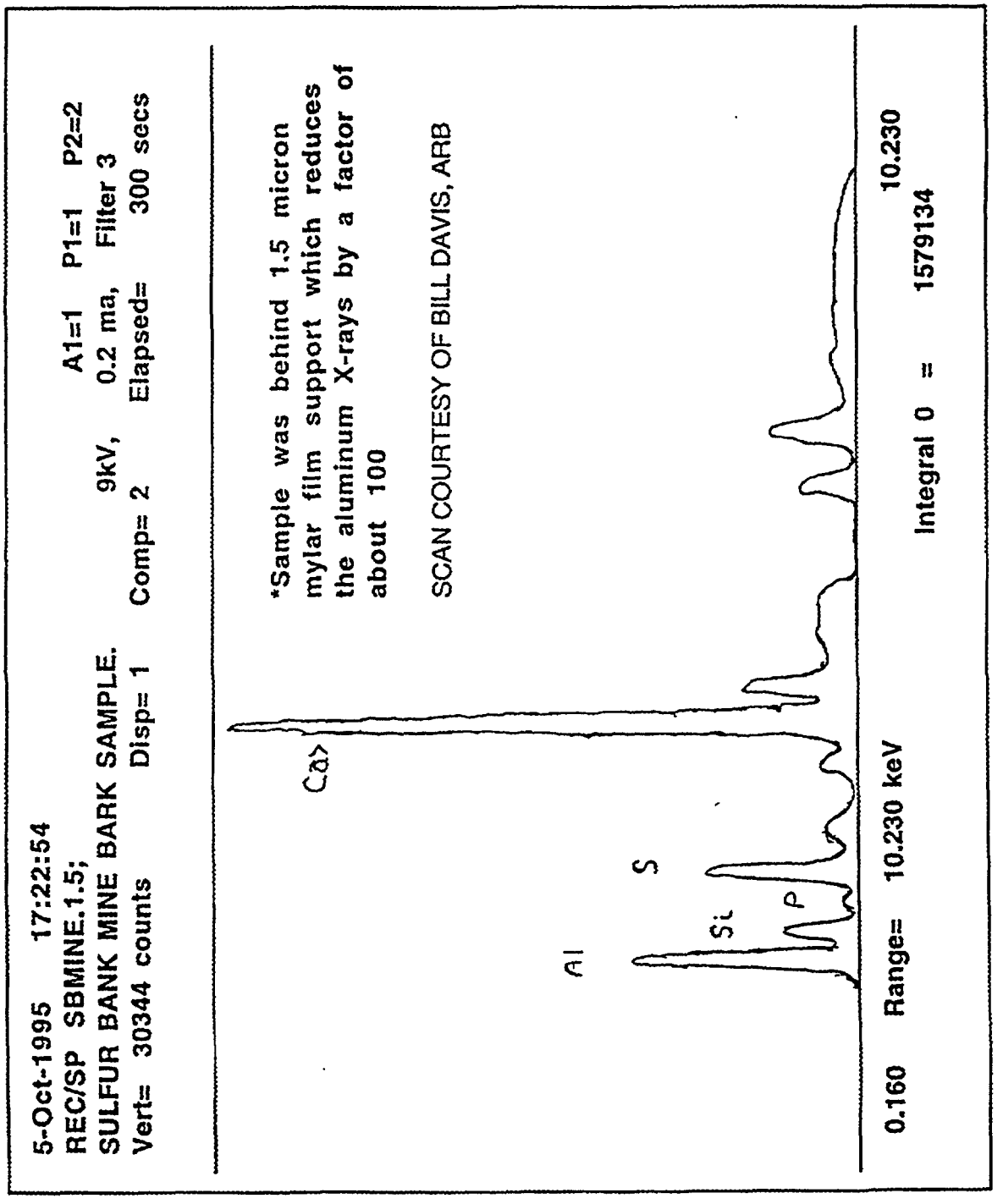
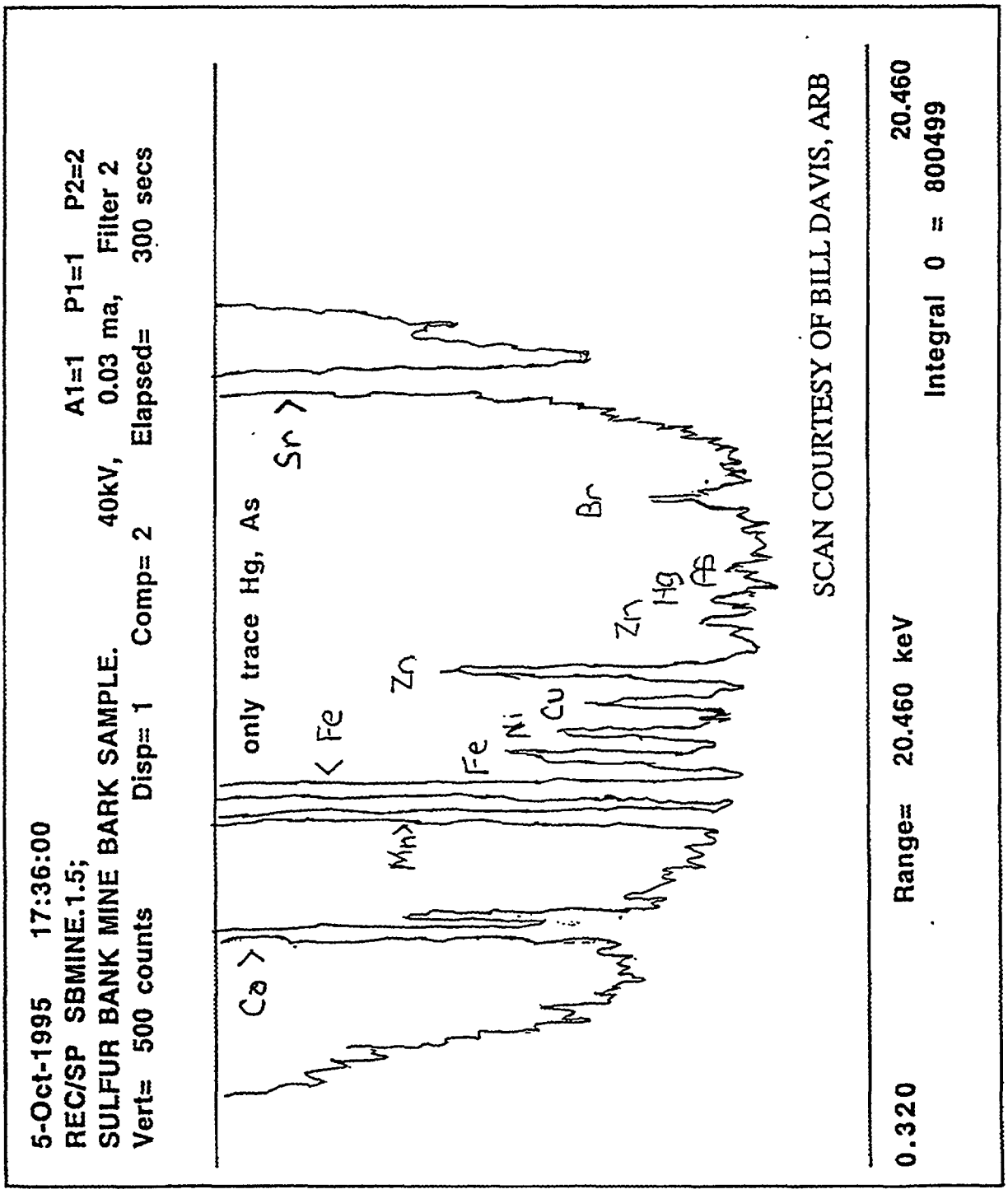

5-Oct-1995 17:22:54
REC/SP SBMINE.1.5;
SULFUR BANK MINE BARK SAMPLE.
Vert= 30344 counts Disp= 1 Comp= 2 9kV, 0.2 ma, Elapsed= A1=1 P1=1 P2=2 Filter 3 300 secs
*Sample was behind 1.5 micron mylar film support which reduces the aluminum X-rays by a factor of about 100
SCAN COURTESY OF BILL DAVIS, ARB
Ca>
S
Si
P
Al
0.160 Range= 10.230 keV
Integral 0 = 1579134
10.230

Figure 1B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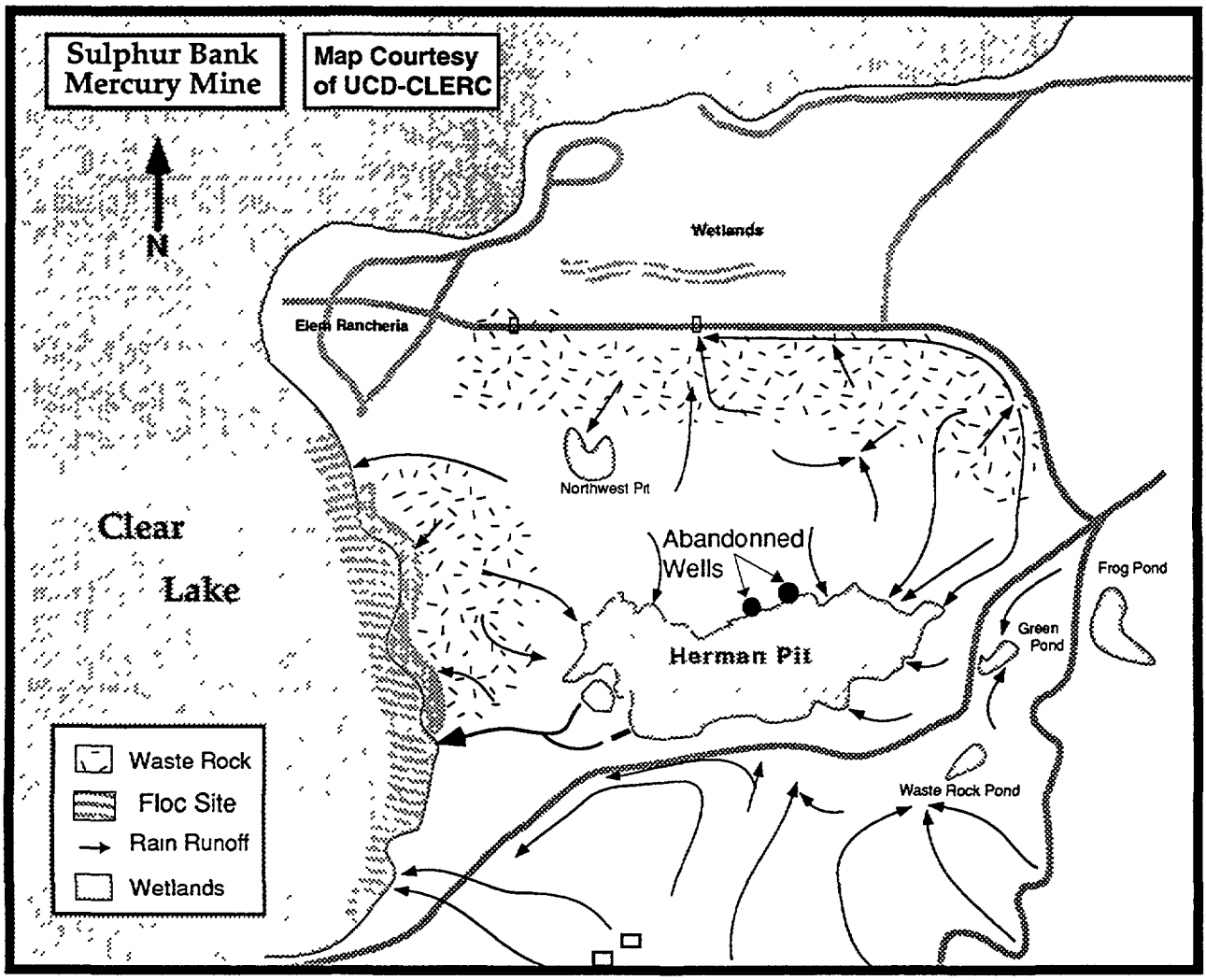
Disp= 1
Comp= 2
40kV,
A1=1 P1=1 P2=2
0.03 ma, Filter 2
Elapsed= 300 secs
only trace Hg, As
Ca >
< Fe
Mn>
Zn
Fe
Ni
Cu
Sr >
Br
Zn
Hg
Pb
SCAN COURTESY OF BILL DAVIS, ARB
0.320
Range= 20.460 keV
Integral 0 = 800499
20.460


Sulphur Bank Mercury Mine
Map Courtesy of UCD-CLERC
N
Wetlands
Elem Rancheria
Northwest Pit
Clear Lake
Abandonned Wells
Herman Pit
Frog Pond
Green Pond
Waste Rock Pond
Waste Rock
Floc Site
Rain Runoff
Wetlands

Revegetation of Sulphur Bank Mercury Mine.

M.A. Showers. California Department of Conservation, Office of Mine Reclamation, 801 K Street, MS 09-06, Sacramento, CA 95814-3529.

Key words: Sulphur Bank Mercury Mine, revegetation, native plants

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) is developing a revegetation strategy for Sulphur Bank Mine as part of the Remedial Design for this Superfund Site. Conservation of existing native vegetation, elimination or reduction of the amount of imported soil cover required, mitigation of transport of contaminated sediments or runoff into Clear Lake, and reestablishment of native vegetation are the principal objectives of the revegetation project. Revegetation test plots have been installed in two areas of the mine: on the Clear Lake shoreline tailings pile and on tailings in the northeast portion of the mine. Based on cluster analysis of soils data, the two sites are considered representative of substrate conditions over large portions of the mine. Initial test plots contrasted soil treatments using waste lime and organic compost against a control. For the initial test trials, native species purchased from commercial suppliers were used to calibrate soil treatments. Additional test plots were installed in 1997 using plants grown from seed collected on site. Preliminary data indicate that use of waste lime + organic material result in greatest plant growth.

Mercury Partitioning Trends in Fish From Clear Lake, a Mine-Impacted EPA Superfund Site in California

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(excerpts from a recent article submitted for publication)

ABSTRACT

We present data for methyl and total Hg in adult fish muscle and major organs/tissues, and spatial trends of fish Hg accumulation within Clear Lake. Muscle Hg varied with trophic level, with piscivorous adult largemouth bass exhibiting concentrations to over 1.00 $\mu g\ g^{-1}$ (mean = 0.71). Carp to ages >10 years had consistently low Hg ($\leq$0.27 $\mu g\ g^{-1}$, mean = 0.15 $\mu g\ g^{-1}$), despite residing and feeding among heavily contaminated bottom sediments (to ca. 300 $\mu g\ g^{-1}$ Hg). Diet is implicated as the predominant pathway of Hg accumulation in this system. Overall, Clear Lake fish Hg, while elevated, is far lower than sediment loads might suggest, and is generally comparable to many other lakes worldwide without significant Hg contamination. Inland silversides of ages $\leq$1 year exhibited a striking spatial variation in Hg and a weak relationship between fish Hg accumulation and underlying sediment loads. For largemouth bass, carp, and channel catfish, muscle tissue contained $\geq$87% of the total Hg and $\geq$93% of the methyl mercury. Liver exhibited high concentrations but did not contribute greatly to overall Hg body burdens. Intermediate concentrations were found in kidney, spleen, intestine, testes, and brain. Negligible Hg was found in bone, skin, scales, fat, and eggs.

Keywords: mercury, methyl mercury, fish, partitioning, mining

INTRODUCTION

Elevated Hg levels in Clear Lake were first detected in 1970 by the California Department of Health Services [1]. Since that time, hundreds of water, sediment, fish, and waterfowl samples have been collected near the mine site and elsewhere in the lake. Hg concentrations in Clear Lake fish have frequently exceeded the State of California and National Academy of Sciences 0.50 $\mu g\ g^{-1}$ guideline for human consumption and have occasionally exceeded the U.S. Food and Drug Administration 1.00 $\mu g\ g^{-1}$ guideline [2]. Historically, the highest tissue Hg levels tended to be found in fish taken from the Oaks Arm of the lake. The mine site has been clearly identified as the most significant source of Hg contamination entering the lake, both historically and ongoing [3, 4, 5]. The Sulphur Bank Mercury Mine has been designated as a U.S. EPA Superfund Site, primarily as a result of Hg contamination in Clear Lake fishes. As part of a Remedial Investigation / Feasibility Study of the site and adjoining lake, a preliminary baseline study was conducted in the summer of 1992, resulting in a series of publications [4, 6-8] including this one. This paper provides historical Hg concentrations in fishes and data from the 1992 study.

METHODS

We chose four species representing widely varying trophic status: (1) inland silversides (*Menidia beryllina*) are planktivores, (2) carp (*Cyprinus carpio*) are detritivores, (3) channel catfish (*Ictalurus punctatus*) are omnivores, and (4) largemouth bass (*Micropterus salmoides*) are top predators. Fishes were sampled toward the back region of each of the three arms of the lake, as well as at the Narrows, located at the intersection of the arms. Adult carp, channel catfish, and largemouth bass were collected by commercial otter trawl. Inland silversides were collected by seine in shallow water (1-2 meter depth) along the shorelines. Silversides were separated into several consistent size classes and replicate whole-fish composites of 6-100+ individuals of each size class were prepared from each site. Axial muscle samples of individual largemouth bass, channel catfish, and carp were taken from the dorso-lateral ("shoulder") region. Equal sized muscle portions were composited from 1-5 similar-sized individuals for each size class. For multi-organ comparative samples of largemouth bass, channel catfish and carp, organs were collected from 1-3 similar sized individuals taken from the highest Hg region of the lake, near the mine.

Total Hg was determined in the majority of tissue samples with standard cold vapor atomic absorption (CVAA) methodology by a governmental contract analytical laboratory. The low-Hg series of inland silversides whole fish composites were analyzed by the first author, using a modified CVAA micro-technique [9]. All methyl Hg analyses were performed by Brooks Rand, Inc., Seattle, WA, utilizing state of the art techniques, as developed by Bloom [10].

Equal sized portions of muscle tissue were used from each individual within a given composite. Hg whole body burdens were calculated using the measured weight percentages of each of the various tissues/organs and the analyzed concentrations.

RESULTS

Differential Hg in fish organs

Liver tissue contained the highest concentrations of total Hg in both largemouth bass and channel catfish, at levels 60-100% greater than corresponding muscle. Carp, however, had liver total Hg concentrations <50% of corresponding muscle levels. Muscle tissue contained the next greatest concentrations, with kidney, spleen, intestine, testes and brain exhibiting intermediate levels, generally ≤40% of corresponding muscle concentrations. Hg in bone, skin, scales, fat, and eggs of all three species was below the 0.13 ug g^{-1} operational level of detection.

Methyl Hg in different body constituents generally conformed to the trends noted for total Hg. Bone, skin, scales, fat, and eggs contained the lowest methyl Hg concentrations. Kidney, spleen, intestine, testes, and brain exhibited intermediate levels, at 10-20% of corresponding muscle tissue concentrations. In contrast with the total Hg data, methyl Hg in liver tissue of all three species was significantly lower than corresponding methyl Hg concentrations in muscle. This was a function of low methyl Hg percentage in liver tissue, at 11-36% of total Hg, relative to a very high methyl Hg fraction in muscle. Hg in muscle tissue was almost entirely methyl Hg (93-97%). The methyl Hg proportion in other tissues, where it could be calculated, ranged from 32-77%, with most values ≤50%.

When weighted for mass contribution to overall body burdens, a dramatic Hg distribution pattern becomes apparent. The differential Hg body burden data are presented in Figs. 1a (total Hg) and 1b (methyl Hg). Muscle tissue was found to contain the overwhelming majority of the Hg body burden in all three species, with 87-90% of the body burdens of total Hg and 93-95% of the methyl Hg.

Fish muscle Hg

Among the larger fishes sampled (carp, channel catfish, and largemouth bass), carp were consistently lowest in Hg (Table 1). With the exception of two of the composite collections taken directly off the mine site (0.27 and 0.21 ug g^{-1}) and one from the Upper Arm (0.17 ug g^{-1}), muscle Hg concentrations in all other samples were ≤0.13 ug g^{-1}. The mean concentration was 0.15 ug g^{-1} (n=26). Individuals <4,000 g had a mean of 0.12 ug g^{-1}, while those >4,000 g averaged 0.19 ug g^{-1}. Scale analyses of a subset of the carp indicated ages of 5-15 years. This benthic insectivore/detritivore species resides on and feeds directly in the contaminated bottom sediments of the lake [11].

Channel catfish muscle tissue exhibited greater levels of total Hg than carp, with a mean individual concentration of 0.27 ug g^{-1} (n=27). Channel catfish <2,500 g averaged 0.18 ug g^{-1}, while individuals >2,500 g averaged 0.36 ug g^{-1}. A positive relationship between fish size and Hg concentration was apparent at three of the four sampling locations. It was only possible to sample a single size class in the Upper Arm, despite extensive collection efforts. The channel catfish is a benthic species, residing and feeding in a habitat similar to carp. However, this species feeds on higher trophic level food, including fish and crayfish [11].

Largemouth bass contained the highest levels of total Hg found in all four fish species analyzed, with a mean individual muscle Hg concentration of 0.71 ug g^{-1} (n=38). Largemouth bass <1,000 g had a mean of 0.48 ug g^{-1}, whereas individuals >1,000 g averaged 0.85 ug g^{-1}. A general trend of increasing Hg with increasing size was apparent in the data set as a whole and most consistently in the Lower Arm and the Upper Arm. Largemouth bass are midwater top predators, feeding primarily on other fish.

Inland silversides are a small, annual species (generally <5 g) that feed on water column plankton. This species contained the lowest levels of total Hg of the four fishes analyzed (as measured on a whole fish basis), most of these concentrations being below the operational detection level of 0.13 ug g^{-1} available from the governmental contract analytical laboratory required for the large fish muscle work. To obtain useful data for inland silversides, we analyzed these samples at our U.C. Davis laboratory, with a detection limit of 0.01 ug g^{-1} (Fig. 2). Mean whole body total Hg concentration for all size classes of silversides was 0.105 ug g^{-1}. Fish <50 mm had a mean concentration of 0.065 ug g^{-1}. Individuals between 50 and 85 mm averaged 0.145 ug g^{-1}. Within the low range of 0.03-0.19 ug g^{-1} found in this species, the data exhibit a consistent trend of increasing Hg concentrations with increasing size classes for all four study regions.

Spatial variation in fish Hg

The data in Table 1 are arranged to facilitate inter-site comparisons with individuals of similar size. Hg in carp of all size classes was low, with the only composites greater than 0.13 ug g^{-1} originating from the near-mine (Oaks Arm) site and the Upper Arm. Channel catfish less than 3,000 g exhibited no clear spatial trends. Near-mine samples were not notably higher than similar samples from other regions of the lake and were in some cases lower. Oaks Arm and Narrows samples greater than 3,000 g did not generally have corresponding size classes to compare from other arms, though a single comparison between Narrows and Lower Arm catfish in the 3,400-4,000 g range demonstrated more than double the Hg concentration (0.58 ug g^{-1}) at the Narrows site, closer to the mine, as compared to the sample from the Lower Arm (0.26 ug g^{-1}). The single channel catfish taken from the Upper Arm had a muscle Hg concentration (0.40 ug g^{-1}) more than double that found in fish of similar size from the other locations.

Largemouth bass were a particular focus of this study and previous collections because of their relative site fidelity, as compared to other large species present in the lake [11], and because of their popularity as a sport fish. Our data suggest a large region of relatively elevated Hg in largemouth bass, extending throughout the Oaks Arm, Narrows, and Upper Arms, with significantly lower levels of Hg in the Lower Arm. Largemouth bass from the near-mine site exhibited a range of elevated muscle Hg concentrations (0.63-1.15 ug g^{-1}). Concentrations above 0.50 ug g^{-1} were found in the majority of samples from the Narrows and Upper Arm sites. Lower Arm largemouth bass were consistently lower in Hg. Mean individual largemouth bass muscle concentration from the Lower Arm was 0.40 ug g^{-1} (n=7), as compared to means of 0.85 from the Oaks Arm (n=10), 0.69 from the Narrows (n=9), and 0.78 from the Upper Arm site (n=12). Excluding the composite of very large Upper Arm individuals >4,000 g, for which there were no comparable fish at other sites, the Upper Arm mean was 0.67 (n=10).

Carp, channel catfish, and inland silversides exhibited spatial patterns similar to largemouth bass, although inland silversides had significantly lower Hg concentrations (Fig. 2). For inland silversides, within each size class an identical spatial pattern was observed, with a slight decline in concentrations moving from the Oaks Arm (mine site) to the Narrows, where same-size individuals averaged 85% of the Hg concentration in corresponding near-mine samples. A larger decline was observed in the Lower Arm, with same-size individuals averaging 50% of the levels in corresponding near-mine samples. The Upper Arm inland silversides demonstrated the reverse trend, with *higher* Hg concentrations relative to the intermediate site at the Narrows. Upper Arm samples had higher Hg levels than even the near-mine samples, averaging 124% of near-mine levels among all size classes at this time.

In contrast with the relatively spotty, variable data available for large fish, composite samples of small, short-lived inland silversides provided a more consistent measure of relative localized Hg accumulation, though within a much lower range of concentrations. As a result of their short lifespans and relatively localized ranges, samples taken from distant regions of this large lake will generally reflect conditions of Hg bioavailability at or near the site of capture. Localized Hg bioavailability will tend to determine a significant proportion of the lifetime accumulation, as compared to larger species, which integrate Hg accumulated in potentially diverse locations inhabited over a period of several years before reaching a size commonly sampled. Also in contrast with the larger fish species, the schooling nature of the small, numerous inland silversides facilitated the collection of virtually identical composite samples of many individuals at each of the sites and for each of the size classes, including identical replicate samples. These data (Fig. 2) demonstrated a very consistent pattern within each size class, similar to the spatial trends suggested by the large fish data. Steady declines in whole body Hg concentration occurred along a transect between the mine site and the Lower Arm, but not in the direction of the Upper Arm. Some of the highest concentrations were found in the Upper Arm, at sites with very low sediment Hg. Similar anomalous Upper Arm findings have been recorded for total and methyl Hg in water [4], zooplankton [7], and benthic invertebrates [6, 7].

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FIGURE CAPTION

1.a. Total mercury body burden percentages in major fish body constituents.

1.b. Methyl mercury body burden percentages in major fish body constituents.

2. Total mercury from inland silversides whole body composites (wet weight $\mu g\ g^{-1}$); mean values of 2-3 replicate composites per size class.

Table 1 Clear Lake fish muscle mercury concentrations in 1992 baseline study.
Composite data arranged by increasing weight categories; mean wet weight (ww) in grams; (n) = number of individuals in composite; "Hg" = wet weight µg/g (ppm); "———" indicates no data available for those weight categories.

MINE SITE		NARROWS		LOWER ARM		UPPER ARM	
Mean ww (n)	Hg	Mean ww (n)	Hg	Mean ww (n)	Hg	Mean ww (n)	Hg
CARP							
1,814 (1)	< 0.13	1,645 (1)	< 0.13	———		———	
———		———		3,272 (3)	< 0.13	———	
3,642 (4)	0.13	———		3,791 (4)	0.13	3,787 (1)	0.17
4,635 (4)	0.27	4,374 (1)	0.17	5,021 (3)	< 0.13	4,042 (1)	0.13
5,625 (3)	0.21	———		———		———	
———		———		7,567 (1)	< 0.13	———	
CHANNEL CATFISH							
———		819 (3)	< 0.13	606 (1)	0.19	———	
1,545 (1)	0.13	1,618 (3)	0.17	———		1,114 (1)	0.3
2,115 (1)	0.13	———		2,274 (3)	0.23	———	
2,756 (1)	0.35	2,631 (3)	0.21	2,823 (3)	0.31	———	
———		3,979 (1)	0.58	3,452 (3)	0.26	———	
7,200 (1)	0.8:	5,196 (2)	0.41	———		———	
LARGEMOUTH BASS							
691 (1)	0.79	342 (2)	0.36	398 (1)	0.16	473 (3)	0.35
966 (3)	0.63	839 (2)	0.97	832 (2)	< 0.13	862 (1)	0.47
———		1,078 (4)	0.76	———		———	
1,362 (3)	0.8:	1,358 (1)	0.55	1,752 (2)	0.48	1,871 (5)	0.84
2,274 (2)	1.15	———		2,106 (2)	0.73	———	
2,913 (1)	0.83	———		———		2,709 (1)	0.97
———		———		———		4,443 (2)	1.33

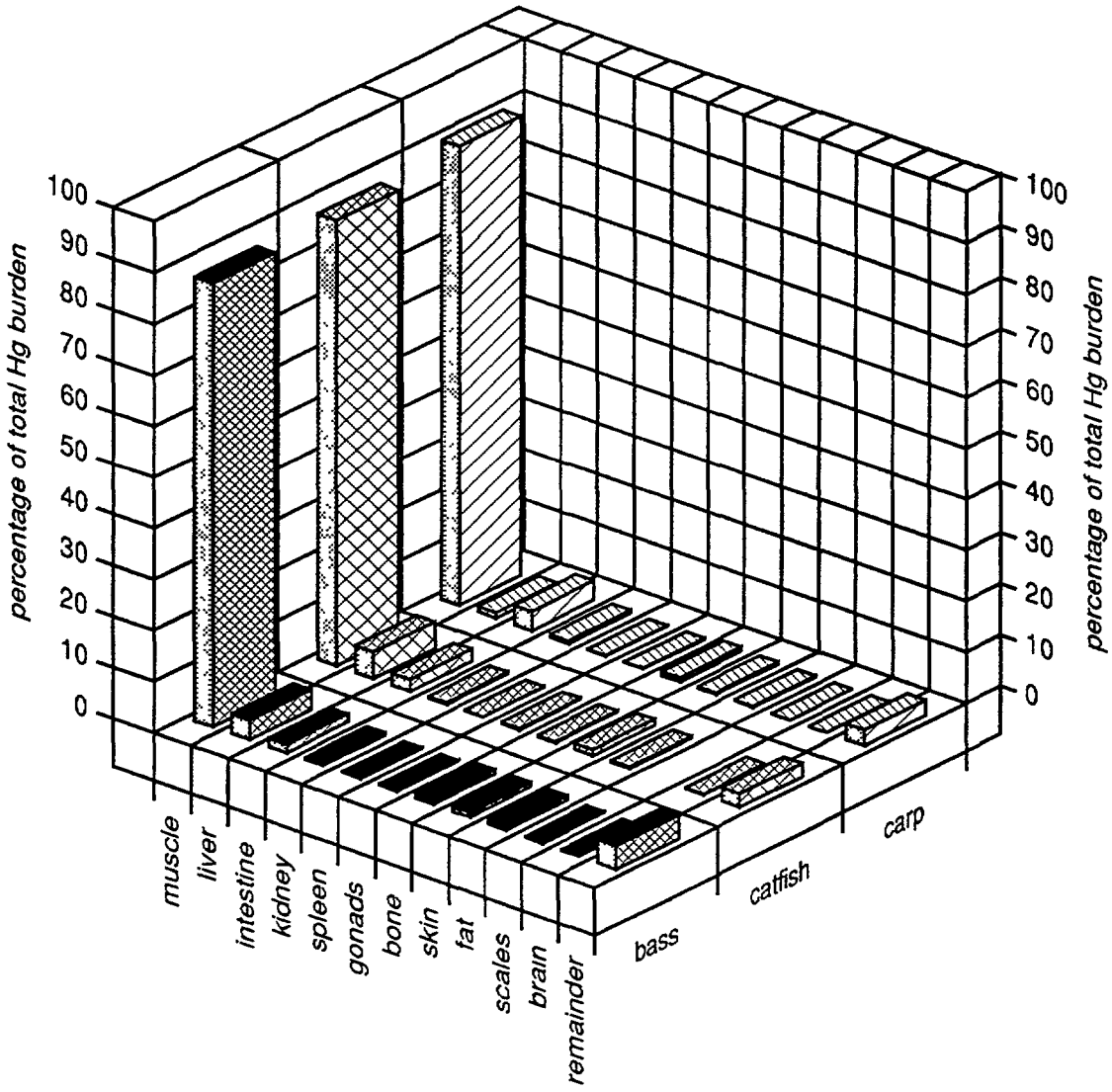

Fig. 1(a). Whole body burden percentages: total mercury

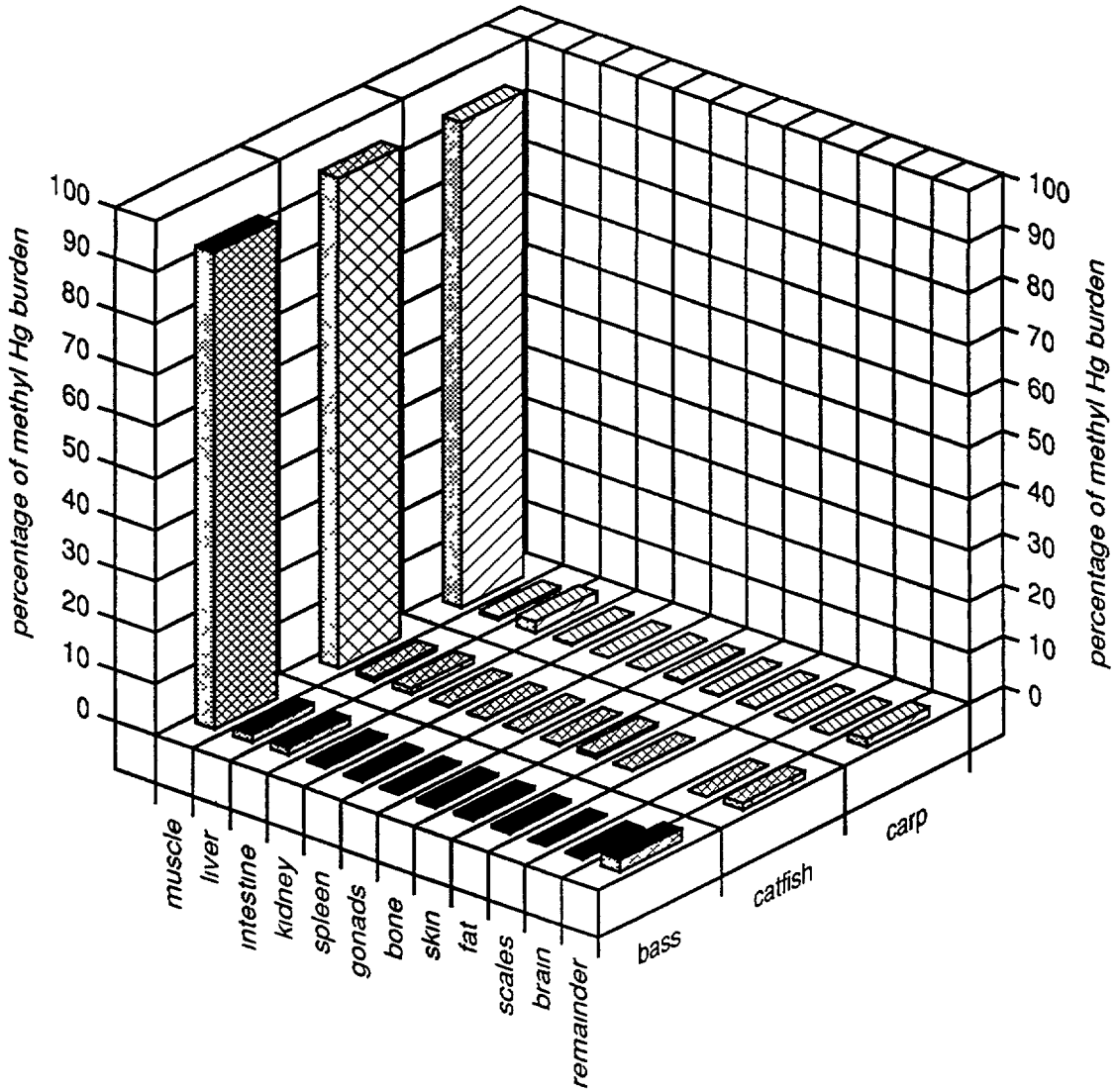

Fig. 1(b). Whole body burden percentages: methyl mercury

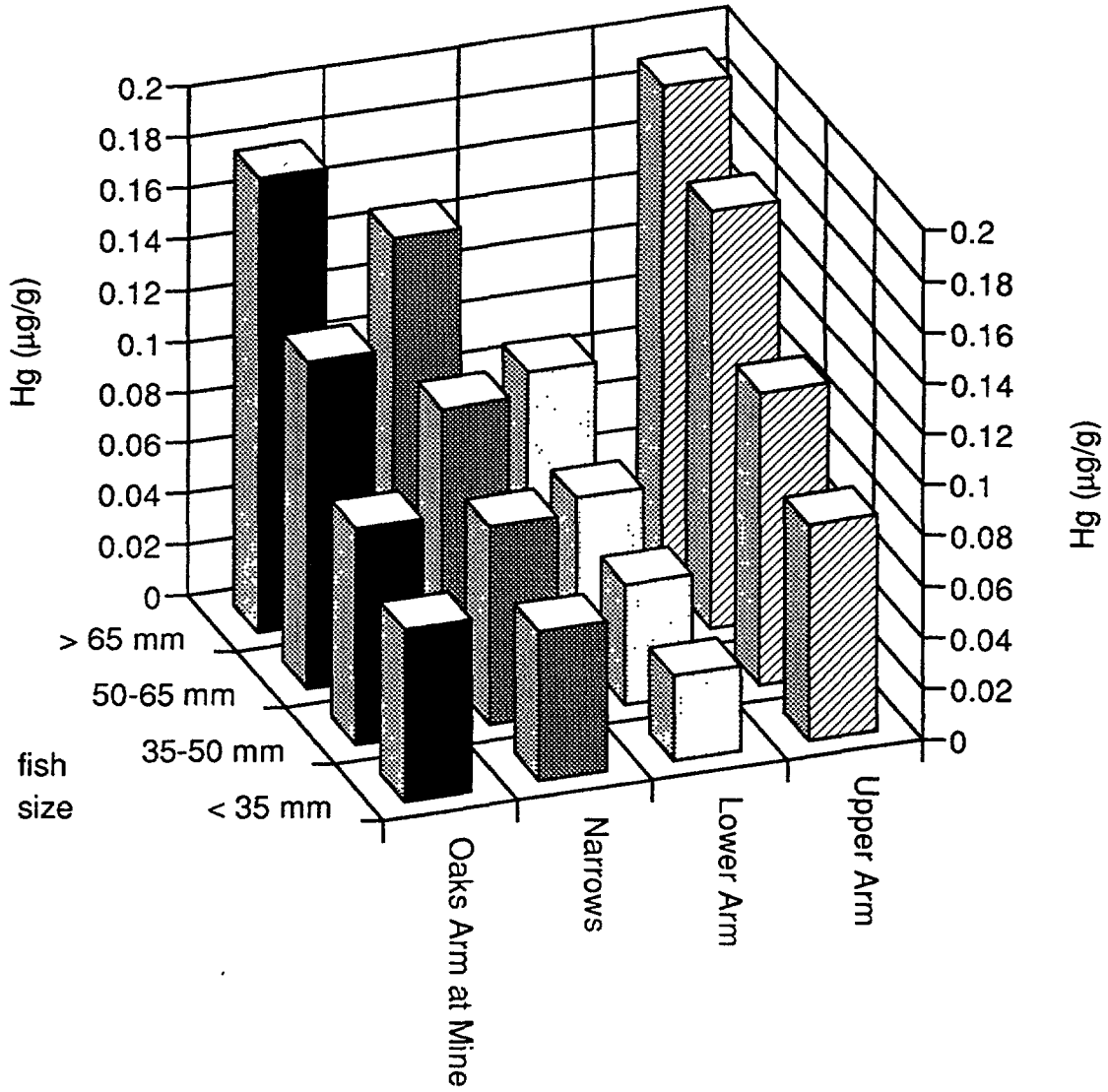

Fig. 2. Whole body silversides mercury

Anomalous Mercury Concentrations in the Lower Trophic Levels of the Clear Lake Aquatic Ecosystem, California

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Key Words: mercury, methyl mercury, Sulphur Bank Mercury Mine, biota, plankton, benthic

Total and methyl mercury were analyzed in plankton and benthic invertebrates from Clear Lake, CA, an aquatic ecosystem contaminated from over 84 years of mining from the Sulphur Bank Mercury Mine. Total (primarily inorganic) mercury in Clear Lake biota (up to 855 ng/g in plankton, 41,671 ng/g in oligochaetes, and 27,686 ng/g in chironomids) was found to reflect the concentration of mercury in the organisms' surroundings (water or sediment). Methyl mercury, however, (up to 67 ng/g in plankton, 19.9 ng/g in oligochaetes, and 61.9 ng/g in chironomids) was typically not correlated with inorganic mercury concentrations in water and sediment. Anomalously high concentrations of methyl mercury in biota at great distances from the point source of inorganic mercury suggests either 1) methyl mercury is produced in situ at sites with low inorganic mercury, or 2) methyl mercury is produced in regions with high inorganic mercury and transported to other regions of Clear Lake. An increasing ratio of methyl/total mercury with increasing trophic level supports a bioaccumulation model for methyl mercury dynamics in Clear Lake biota, and suggests that bioavailable mercury increases as a function of distance from the mine. Compared with other contaminated sites worldwide, Clear Lake's plankton are relatively low in both total mercury and methyl mercury. Benthic invertebrates exhibit the highest total mercury values, yet have the lowest methyl mercury concentrations of any known contaminated sites.

THE BIOGEOCHEMISTRY OF MERCURY CONTAMINATION FROM THE SULPHUR BANK MERCURY MINE AT CLEAR LAKE, CALIFORNIA: A MINING LEGACY

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Abstract: Mining operations conducted at the Sulphur Bank Mercury Mine from 1872-1957, together with acid mine drainage since abandonment, deposited *ca.* 100 metric tons of mercury (Hg) in the sediments of Clear Lake. Natural lakebed springs are not a significant source of Hg to Clear Lake. Hg in surficial sediments (up to *ca.* 450 mg kg^{-1} [=ppm] total Hg and *ca.* 25 μg kg^{-1} [=ppb] methyl Hg) exhibits a classic point source distribution with maximum concentrations adjacent to the mine. However, the ratio of methyl:total Hg in sediments increases with distance from the mine, suggesting either differential transport of methyl Hg or a non-linear relationship between sediment inorganic Hg concentrations and methylation. Water exhibits an even more gradual decline in total Hg concentrations with distance from the mine, in both unfiltered (max. *ca.* 400 ng L^{-1} =pptr) and filtered (max. *ca.* 12 ng L^{-1} =pptr) bottom water. In comparison with studies of other sites, Clear Lake exhibits high inorganic Hg in sediments and water, yet relatively low methyl Hg. Inorganic Hg in biota declines exponentially away from the mine, yet methyl Hg is often highly elevated at sites distant from the mine. Floc, precipitated from acid mine drainage during winter rain/ flooding periods, appears to become highly charged with methyl Hg and likely transported to lake sites distant from the mine. Further studies are needed to understand the origin of the floc, how it participates in Hg methylation and its impact on biota.

Key Words: mercury, bioaccumulation, acid mine drainage, Sulphur Bank Mercury Mine

INTRODUCTION

Periodic mercury (Hg) mining was conducted at the Sulphur Bank Mercury Mine at the east end of the Oaks Arm of Clear Lake, CA from 1872-1957 [1,2,3]. Earlier estimates [2] indicate that *ca.* 100 metric tons of Hg contaminate the aquatic ecosystem of Clear Lake as a result of this mining. The Sulphur Bank Mercury Mine became an Environmental Protection Agency (EPA) Superfund site in 1991 and U.C. Davis was awarded a contract to conduct an ecological assessment of Hg contamination within the Clear Lake aquatic ecosystem in 1992. Our preliminary baseline survey of Hg concentrations in the physical and biological components of Clear Lake took place in the fall of 1992. This study verified that the proximate source of the Hg pollution originated from the east end of the Oaks Arm, and not from springs in the lake bed, but we could not precisely identify the ultimate source.

We began to monitor seasonal variations in both total (primarily inorganic) and organic (primarily methyl) Hg in these components in May 1994. In addition to our in-lake seasonal monitoring program, we have: (1) conducted numerous experiments to understand the transformation of inorganic mercury to the more toxic organic form, methyl Hg, (2) evaluated the potential for the Sulphur Bank Mercury Mine to act as an ongoing point source of contamination, (3) investigated Hg concentrations in the wetland just north of the mine, (4) collected and analyzed sediment cores to evaluate the historical deposition of Hg within Clear Lake over the past 200-250 yrs, (5) evaluated the concentration of Hg in birds within the Clear Lake ecosystem and Hg's potential influence on the reproduction of ospreys, (6) developed a computer model (viewable in video format) to predict the uptake of Hg through the Clear Lake food chain, (7) developed a computer-based particle tracking model to help predict the movement of Hg-laden particles through water from near the mine site to other regions within Clear Lake, (8) conducted two current studies to evaluate current speeds/directions that would be capable of moving Hg-laden particles around Clear Lake, (9) conducted a dye tracer study in the Herman Pit (on the mine site) to evaluate the potential route of acid mine drainage from the mine to Clear Lake, and (10) initiated bioturbation experiments to estimate the depth of sediment mixing possible from benthic invertebrates.

This paper is intended to provide a broad overview of our Hg studies on the Sulphur Bank Mercury Mine to date. Although many components of our studies have been completed, many others are still in progress and further data and/or analysis may alter the final interpretations. Field studies on this project are anticipated to be completed by fall of 1998.

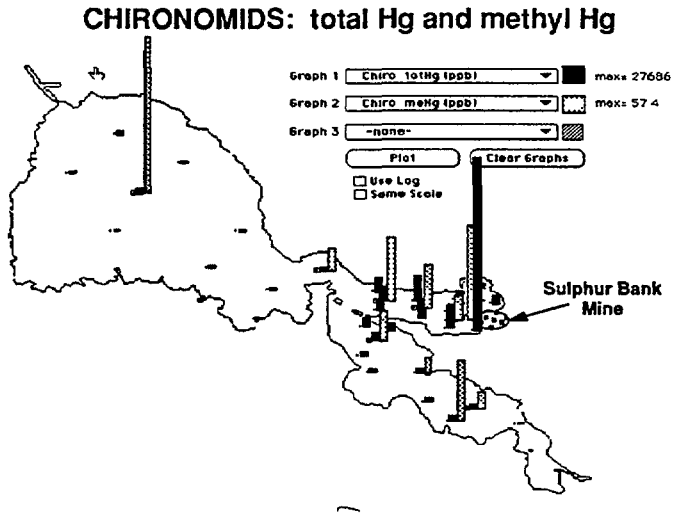
RESULTS

Phase 1: Baseline Survey of Hg in the Clear Lake Aquatic Ecosystem

Our baseline study in the fall of 1992 provided us with confirmation that the Sulphur Bank Mercury Mine represents a clear point source of total (mostly inorganic) Hg contamination in Clear Lake. This was evidenced in water, sediments, periphyton, plankton, and benthic invertebrates. These observations were mostly consistent with a few earlier studies that measured total Hg in sediments, water, and biota from Clear Lake.

Fishes, however, which contain an estimated 95-99% of total Hg as methyl Hg, had variable distributions of Hg, with individuals collected from the Upper Arm exhibiting relatively high (or higher) levels of total Hg (mostly methyl Hg) in comparison with those from the Oaks Arm near the mine. Hg in fishes was mostly concentrated in liver and muscle tissues. The species of fishes

1A



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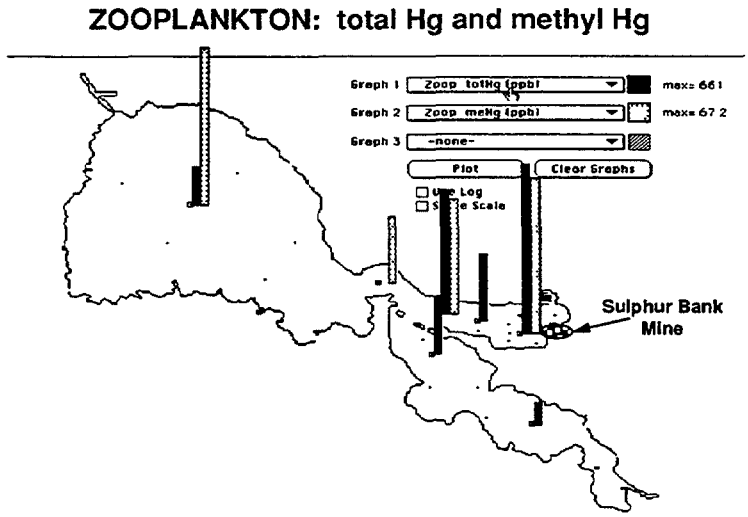

Figure 1: Total and methyl Hg concentrations in (A) chironomids and (B) zooplankton. Note elevated levels of methyl Hg in the Upper Arm.

that accumulated Hg most effectively were largemouth bass, catfish and silversides. Carp exhibited very low concentrations of Hg in all individuals from all sites.

Methyl Hg distributions in Clear Lake sediments, water and biota do not correlate well with the distributions of total Hg. Significantly elevated levels of methyl Hg in biota (e.g., chironomids, oligochaetes and zooplankton) were observed at sites in the Upper Arm, far distant from the point source of inorganic Hg at the mine. **Fig. 1** demonstrates the unusual pattern of methyl Hg observed during the Baseline Survey in 1992.

There are some indications that methyl Hg in Clear Lake biota bioaccumulates much more efficiently through benthic trophic pathways as compared with planktonic pathways [2,3]. Our observations of surprisingly elevated levels of methyl Hg in biota from Upper Arm sites is consistent with the hypothesis that methyl Hg in floc particles is transported differentially (more easily compared to inorganic Hg) to the Upper Arm via wind-driven currents and bioaccumulated through the benthic trophic pathway. In addition, the inorganic Hg in floc particles may be subject to further methylation after they are incorporated in sediments distant from the mine. Further studies are planned to better define the bioaccumulation pathway to higher trophic levels.

In general, the ratio of methyl Hg to total Hg in abiotic and biotic matrices in Clear Lake increases as a function of distance from the mine. This pattern could be: (A) an artifact caused by low sample numbers in some of the sampling regions rather distant from the mine; (B) caused by unique internal circulation dynamics of Clear Lake which may remobilize Hg-laden particles from near the mine site and deposit them in a region of the Upper Arm; or (C) caused by two independent (and counteracting) processes being responsible for the production of methyl Hg, one decreasing away from the mine (total Hg concentration) and one increasing away from the mine (bioavailability of Hg for methylation). An understanding of the dynamics of methyl Hg production, mobilization and bioaccumulation was not possible based on this baseline study.

Phases 2, 3, and 4: Seasonal Monitoring Program & Experiments and Other Surveys

Begun in May 1994, our Seasonal Monitoring Program (Phases 2, 3, and 4) was designed to evaluate any potential differences among sites relative to the influence of intra- and inter-annual variability on the production and ultimate bioaccumulation of methyl Hg within the Clear Lake aquatic ecosystem. For surficial sediments, water, and biota, the ratio of methyl:total Hg increased as a function of distance from the mine, indicating that more "bioavailable" Hg is present at sites distant from the mine. These results also indicate that a majority of the total and methyl Hg is bound to particulate material in the water column.

Summer/fall (August-November) is a time when methyl Hg is found in highest concentrations, especially during the summer of 1995, following the development of large volumes of floc in the Oaks Arm near the mine (see below). The greatest concentrations of methyl Hg were found in the deepest regions of Clear Lake, which also exhibited the lowest concentrations of dissolved oxygen, which is also condusive to the survival of sulfate reducing bacteria.

Water Current Studies

Current studies, using dye tracers and drogues, indicate that mixing processes differ significantly between seasons. Whereas mixing and inter-basin exchange in August appeared to be primarily wind-driven, mixing in November varied as a function of both wind and temperature. Mean velocity of currents in August ranged from *ca.* 3.7-6.0 cm/sec (maxima = 11.3-18.8 cm/sec), whereas the velocity of currents in November were significantly lower, with a mean of 2.9-4.5 cm/sec (maxima = 7.4-11.9 cm/sec).

Discovery of Floc and Associated Changes in Hg Contamination

The discovery of a white flocculent material (and lowered pH) in the nearshore region of the Sulphur Bank Mercury Mine in the spring of 1995, following heavy rain and flooding, significantly altered our perceptions of how methyl Hg could be entering the Clear Lake aquatic ecosystem. Previous to this discovery we had assumed that the primary proximate source of Hg bioaccumulating in Clear Lake biota was derived from the lake sediments in the Oaks Arm in the immediate vicinity of the mine. The presence of the floc, with relatively lower levels of total Hg and considerably higher levels of methyl Hg than native sediments within the Oaks Arm, provided us with a more probable source for the elevated levels of methyl Hg found in Clear Lake biota, especially at sites in the Upper Arm, far distant from the mine. There are at least several potential sources for the lowered pH water and associated floc: (A) rain water flowing over and through waste rock piles, (B) overflow from the Herman Pit, (C) percolation of Herman Pit water through the waste rock piles, (D) some common source of low pH water that feeds both the Herman Pit and the lowered pH water fields in the nearshore region of Clear Lake next to the mine.

The floc is believed to consist of an alumino-silicate kaolinite clay-based material (resembling halloysite), likely derived from

acid mine drainage from the mine during periods of extensive rains or flooding. It was found to be associated with low pH waters emanating from sub-aquatic sources near the mine face. From there, depending on weather conditions, which can suspend the floc into the water column, it may become dispersed to other regions of Clear Lake by wind-driven currents.

In 1995 the floc covered about 1,000 X 1,000 m of lakebed in the immediate vicinity of the mine face. Based on the concentration of total and methyl Hg observed in the floc, and known concentrations of Hg in Herman Pit water, we calculated that ongoing Hg loading from the mine is likely to be on the order of a few hundred kg/yr. The precise source of that input remains to be documented.

The mine site flooding/overflow event that occurred during the winter of 1994/95 produced dramatic changes in the nearshore environment next to the mine over the next several months. In sediments near the mine, Acid Volatile Sulfides increased immediately and dramatically, as did porewater sulfate. By November 1995, methyl Hg in sediments near the mine was elevated to its highest level yet recorded. In water, total Hg was elevated in filtered water fractions and by the summer of 1995 methyl Hg increased 2-3 fold. Sulfate in water increased dramatically at site OA-01, providing an excellent environment for potential methylation by sulfate reducing bacteria. During the summer of 1995 zooplankton and chironomids also exhibited their highest concentrations of methyl Hg. The ratio of methyl:total Hg also typically increased during summer periods, especially at sites far distant from the mine.

Possible sources of acid mine drainage, which we believe contribute to the production of floc, could be: (A) rain water passing over/through waste rock pile soils, (B) overflow of Herman Pit water, passing over Hg-rich soils before entering Clear Lake, (C) percolation of Herman Pit and/or Northwest Pit water through waste rock piles before entering Clear Lake, (D) some as yet unidentified common source for acid fluids that feeds both the Herman Pit and subaquatic nearshore areas in front of the mine.

Bacteria/Methylation Studies

A very striking result of these studies indicates that the presence of floc greatly enhances the production of methyl Hg. The most important factor (by an order of magnitude) in the production of methyl Hg appears to be the presence of floc. The ultimate reason why floc is so important in the production of methyl Hg is still not understood, but may be related to one or more of the following processes: (A) floc provides high levels of biologically available Hg for methylation (in contrast to other native sediments near the mine), (B) floc provides high surface area for colonization by sulfate reducers, (C) floc near the mine is likely very high in sulfates, enhancing sulfate reducers and their subsequent production of methyl Hg, (D) floc may become "inoculated/seeded" with heavy organic loading from summer blue-green algae (cyanobacteria), providing a food source for sulfate reducers. These hypotheses are being investigated further.

Investigations at the Sulphur Bank Mercury Mine and Nearby Wetlands

Preliminary data indicate that acidic water flowing over or through mine soils likely strips Hg from these sources, eventually depositing them in Clear Lake. Although water within the Herman Pit is relatively high in total Hg (about 2-10 times higher than typical lake water in the Oaks Arm), it is much lower in methyl Hg compared with bottom water from site OA-1 in the Oaks Arm.

Water level declines in the Herman Pit and the Northwest Pit over time cannot be accounted for by evaporation alone. Thus, there is likely to be significant subsurface drainage of both the Herman Pit and the Northwest Pit; there may also be identifiable conduits that allow large volumes of acid mine drainage to reach Clear Lake.

Floc was also observed in the wetland immediately to the north of the mine. Sediments, water, and biota were sampled and results indicate that (1) Hg in biota increased over time from April to June 1996, and (2) large crayfish exhibited some of the highest concentrations of Hg in any biota studied to date at Clear Lake. This wetland should be included in further studies associated with the mine remediation.

Sediment Coring Studies

To evaluate Hg deposition (and other anthropogenic changes) within the Clear Lake aquatic ecosystem, we collected sediment cores (to *ca.* 250 cm depth) from five sites around Clear Lake: two sites in the Upper Arm (UA-03 and UA-02), two in the Lower

Arm (LA-05 and LA-03) and one in the Oaks Arm (OA-03). Dating of the cores was accomplished using 210Pb dating techniques, and the presence of specific pollen from introduced species as date proxies. These cores likely span dates 200-300 ybp. Both total and methyl Hg were analyzed at 5 cm intervals. Sedimentation rates appear to vary among cores so that the apparent horizons for various temporal events occur at depths that vary as much as 40 cm between the different cores. 210Pb dating techniques provide an estimate for a sedimentation rate (constant, or averaged, rate for the entire core) of *ca.* 1.33 cm/yr. Site UA-02 appears to have experienced a dramatic increase in sedimentation rate since approximately 1930.

There appears to be a clear signal for total (primarily inorganic) Hg at both sites in the Upper Arm, at the Lower Arm site (LA-03) closest to the mine (but not at the furthest site), and a likely signal in the Oaks Arm reflecting the onset of Hg mining in *ca.* 1875. While this event appeared to double very small concentrations in the Upper Arm, interpretation in the Oaks Arm is more problematic. However, at depths which correspond to *ca.* 1927, a 5 to 10-fold rise in total Hg was observed at all sites in Clear Lake. This time period also corresponds to (1) the advent of powered earth moving equipment and (2) the second largest period of Hg mining lasting from *ca.* 1927-1944 during which *ca.* 1,100 metric tons of Hg were extracted from the mine. The next major horizon for total Hg in these cores occurs at 15-35 cm depth, possibly relating to the 1944 or 1957 cessation of large-scale mining operations (our best estimate for this horizon is 1961±7 yrs). Shallower sections of the cores indicate a slow but significant decline of total Hg from 1961 to the present. Other data indicate a drop of *ca.* 150 ppm total Hg (within the top 3 cm of lake sediments collected near the mine), which may imply that 1992 USEPA remediation efforts at the mine are having a positive effect on the input of total (but not necessarily methyl) Hg to Clear Lake. Also evident in the Oaks Arm core and the Lower Arm core closest to the mine is a slow decline in total Hg from the earliest (deepest) portions of the core to about the depth we believe is the onset of Hg mining. The cause behind this steady decline is unknown at this time. Methyl Hg trends within these lake sediments follow closely the results of total Hg, with the greatest pools of methyl Hg sequestered within sediments at depths of 10-40 cm. Using ant-farm type aquarium microcosms, effective bioturbation depth by chironomid insect larvae appears to reach sediment depths of *ca.* 5-20 cm, with mixing upwards likely being more important than mixing downwards.

CONCLUSIONS

Regardless of the routes by which acid mine drainage reaches Clear Lake, the most important aspect relevant to our assessment of the aquatic ecosystem of Clear Lake is how methyl Hg (with floc derived from acid mine drainage being a significant source) ultimately interacts with biological/ecological endpoints within Clear Lake. **Figure 2** represents a conceptual diagram which summarizes our ideas of how floc might be formed, become methylated, transported and bioaccumulated. According to our conceptualization, floc could form from fluid drainage from the mine site in any one or a combination of pathways identified above. The floc, once formed as a very light and unconsolidated precipitate with 1st order (non-coherent) particles is high in total Hg, yet low in methyl Hg, with high surface area and high sulfate, can be transported to other regions of Clear Lake, or remain in the vicinity of the mine or in that region of Clear Lake. Second order particles (slightly

REFERENCES

more consolidated) settle to the lakebed. When large cyanobacterial blooms of blue-green "algae" accumulate during the summer and eventually die [4] they settle to the lake bottom and can provide rich organic materials to the floc to help drive the methylation process by micro flora (as yet not fully characterized). Once this high methyl Hg floc is formed it is still light enough to be carried by currents to distant parts of Clear Lake, where uptake by plankton and benthic invertebrates can occur. These lower trophic level components can then be taken up by predators and bioaccumulation of methyl Hg is well underway.

Under this scenario, our previous dilemma (i.e. difficulty in understanding how methyl Hg could be found in such elevated concentrations at far distant sites from the mine) is resolved. The light and largely unconsolidated nature of the floc material (as opposed to consolidated lakebed sediments containing primarily tightly bound inorganic Hg) can easily be transported by the currents we have measured in Clear Lake and thus floc could accumulate in the Upper Arm to produce the patterns of Hg distributions which we have observed.

The next phase of our work involves ongoing studies at the mine to determine the pathways by which acid mine drainage is transported from the mine site to Clear Lake. These studies will continue through the winter of 1997/98, likely with more dye tracer studies in Herman Pit, Northwest Pit and more geophysical studies at the mine site. Our Final Report, with a set of remedial recommendations, is expected to be completed in December 1998.

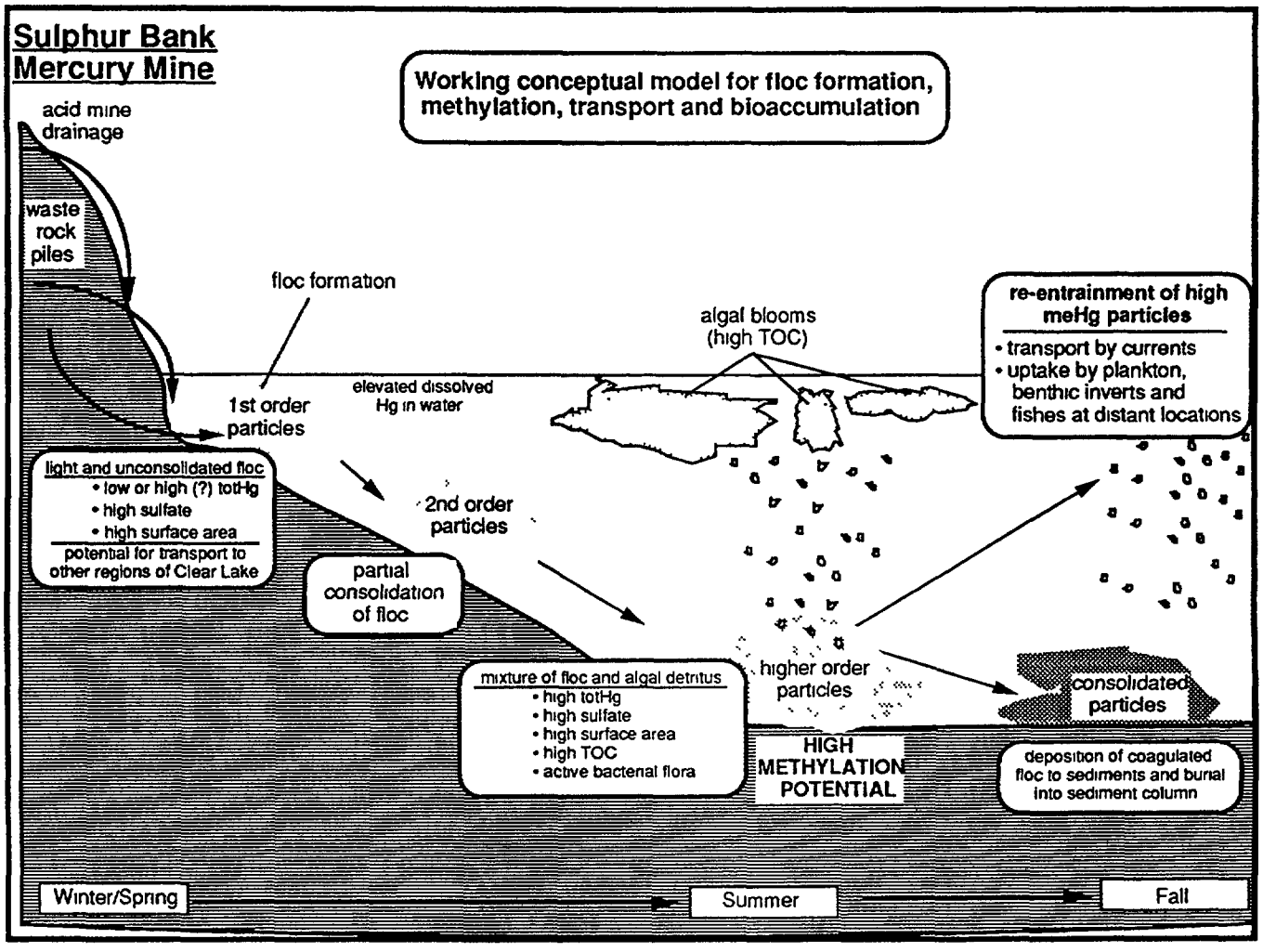

Figure 2. Conceptual diagram of floc origin, formation dynamics, methylation and transport.

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Effects of Waterborne Mercury on Terrestrial Wildlife at Clear Lake: Evaluation and Field-Testing of a Predictive Model

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Key Words: mercury, bioaccumulation, ecological modeling, Great Lakes Water Quality Initiative

Birds and mammals exposed to waterborne mercury (Hg) and methyl mercury (meHg) were collected and/or sampled at Clear Lake, California. This study reports the sampling of fish-eating birds and mammals and insectivorous birds from Clear Lake, CA, which is presently contaminated with Hg, and was historically contaminated with organochlorines (OCs), in order to field test the predictive Wildlife Criteria model developed for the Great Lakes Water Quality Initiative (GLWQI). Tissue samples collected from sampled animals were analyzed for mercury and OC residues, and for selected physiological parameters known to be affected by Hg. All mammalian organ tissues analyzed contained less than 12 ppm wet weight. All avian tissue samples analyzed contained less than 3 ppm Hg wet weight. Animal food samples were also analyzed for Hg content, and with water, sediment and invertebrate Hg concentration data from an accompanying study, used to characterize bioaccumulation of mercury in the Clear Lake food web. Our results support the final GLWQI wildlife criterion (1,300 pg/L) and suggest that, with site-specific modifications, the model will produce a reliable wildlife value for Hg-bearing aquatic systems.

4

WATERSHED AND RESOURCE MANAGEMENT ISSUES

Organochlorine Pesticide Residues in Clear Lake Wildlife Three Decades After Cessation of Pesticide Use. (P)

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Key Words: pesticide, insecticide, organochlorine (OC) contaminants, DDT, DDD, DDE

We investigated the hypothesis that after 35 years since the last use of the insecticide Rhothane (p,p'-TDE) at Clear Lake, residues in fish and wildlife have declined greatly or have disappeared. We found that residues of p,p'-DDD have decreased by 1-3 orders of magnitude in both fish and western grebes (Aechmophorus occidentalis), but both p,p'-DDE and p,p'-DDD were still the dominant two organochlorine (OC) contaminants in these trophic elements of the Clear Lake ecosystem in 1994. The metabolite, p,p'-DDE, reflecting past uses of another closely related insecticide, p,p'-DDT, is now the dominant residue, as with most other areas in North America. What remains unique about Clear Lake is the high DDD/DDE ratio compared to other areas. This high relative presence of DDD is likely due to past DDT-use in the watershed of Clear Lake. Interestingly, DDE residues are still significantly correlated ($P<0.02$) with low levels of eggshell thinning in western grebes. Nonetheless, none of the OC residues we detected are believed, by themselves, to be dominant contaminants significantly affecting birds or fish at Clear Lake as they were in the past.

Watershed Rehabilitation and Restoration in the East Fork Middle Creek Watershed

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Key Words: soil erosion, habitat restoration, watershed

Emergency Rehabilitation Measures to control soil erosion were implemented during the fall of 1996 following the 83,000 acre Fork Fire. The effectiveness of these treatments one year after the fire will be discussed. The Mendocino National Forest will be entering into an agreement with Lake County for preparation of a watershed analysis for the Middle Creek watershed to analyze ecological processes and interactions within the watershed and identify future management opportunities. The proposed East Fork fire salvage and restoration project within the Middle Creek watershed will also be discussed.

HUMAN DISTURBANCE IN THE CLEAR LAKE WATERSHED (LAKE COUNTY, CALIFORNIA) SINCE 1800. (P)

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Abstract – To investigate multiple stresses on Clear Lake, CA over the past *ca.* 250 yr, we raised sediment cores from the three basins of the lake. These cores are *ca.* 2.5 m long and span 200-300 years of the lake's history. We present the results for our, as yet, most thoroughly analyzed core. ^{210}Pb dating yielded a 1.33 cm/yr average sedimentation rate for this core. Total (primarily inorganic) mercury, methyl mercury, organic matter content, carbon, nitrogen, phosphorous, and percent water content were measured at 5cm intervals down the core. Nearly all parameters show major changes at depths of 75-80 cm, corresponding to an estimated date of 1927. Organic matter, total carbon, water content and total nitrogen all show significant decreases above this depth. A peak in bulk density and minimum values for percent water at depths corresponding to about 1971 suggests a period of heavy inorganic erosional input into Clear Lake. Both total and methyl mercury concentrations show major increases in concentration (roughly 10 fold) above the 1927 horizon. There is also a smaller rise in total and methyl mercury at 145-150 cm deep in the core. This horizon is beyond ^{210}Pb dating capabilities, but most likely represents the early episodes of mercury mining, which started in 1873 at the Sulphur Bank Mercury Mine, located on the lake's shore. Peak total mercury levels occur at an estimated date of 1961 and a modest decline has occurred since. Methyl mercury profiles are more complex. Interestingly, the first 75 years of European settlement in the Clear Lake basin (including the most productive years of Sulphur Bank Mercury Mine) appeared to have barely detectable effects on methyl mercury deposition despite considerable presence after the 1870s. Impacts since 1925 are much more dramatic. We hypothesize that around 1925 powered earth moving equipment became much more available, and the cost per unit of earth moved dropped drastically, leading to dramatically increased soil erosion. Road building and similar activities in the basin increased at about this time, and the Sulphur Bank Mercury Mine began to be operated using large scale, open pit techniques.

Key Words: human impacts, mercury, lakes, watersheds, sediment cores

INTRODUCTION

Clear Lake is a large (177 km^2), shallow, eutrophic, polymictic lake in the Central Coast Ranges of Northern California. It sits at about 404 m elevation and has a typical Mediterranean climate with cool rainy winters and hot, dry summers. The topography of the drainage basin is mostly steep, with typical elevations along the rim around 1,000 m. About 195 km^2 of the total drainage basin (1,219 km^2) are level enough for agriculture and urban development. Mean annual high temperature is about 22°C and rainfall averages about 690 mm yr^{-1}. The lake lies in a tectonic basin blocked by a landslide at one end and a volcanic dam at the other. Since 1914, the top 3m of water depth has been controlled by a dam. Clear Lake is thought to be the oldest lake in North America, with continuously recorded lake sediment cores dating to *ca.* 500,000 ybp [1]. The lake is divided into three main basins, or arms (see **Figure 1**), the largest of which is the Upper Arm. Nearly all of the seasonal inflow of water into Clear Lake is into the Upper Arm (approximately 90%, using the estimates in Richerson *et al.* [2]). Thirty percent of the total inflow is from the Scotts Creek and Middle Creek watersheds [2], which enter the lake through Rodman Slough. There are some very small, seasonal inflows into the two other arms. The outflow is through Cache Creek at the southern end of the Lower Arm. Aside from the natural stresses of fire, floods, drought, etc., there are many anthropogenic stresses to the system. The most important management problems at the lake are the frequent occurrence of blooms of scum forming cyanobacteria (bluegreen algae), and contamination of the aquatic ecosystem by mercury and, in the 1950's, by the pesticide DDD.

The history of Native American settlement in the Clear Lake basin is typical of rural California [3,4]. Settlement of the basin reaches back to the beginning of the Pleistocene, and although the basin supported relatively dense populations of Native Americans, we are unlikely to see any significant effect in our cores [5]. The history of European settlement in the basin is described by Simoons [6]. Mexican land grants in the 1840s initiated European settlement, but only a handful of

ranchers resided in the Clear Lake watershed. Agricultural settlement began in 1854, and the rate of settlement increased in 1866. Early censuses showed nearly 1,000 people in 1860, nearly 3,000 in 1870, and just over 7,000 in 1890 in Lake County, most residing in the Clear Lake watershed. Farming, ranching, and farm services were the most important occupations, followed by mining for sulfur, borax, cinnabar, and a small lumber industry. Tourism, centered on hot spring spas, began in 1852 and was well developed in the late 19th century, with several resorts having capacities for hundreds of guests. Early farming focused on wheat, barley, beef, and wool. Sheep numbers peaked sharply in the late 1870s at around 50,000 head, and declined thereafter due to deteriorating pastures. Cattle numbers were much more stable, with around 10,000 head from 1861 through the 1980s. Several activities are likely to have contributed to increased erosion in the 1900s. Until 1987, gravel mining in streambeds was common ([7] and personal communication). Mining for sulfur began at the Sulphur Bank Mine at the east end of the Oaks Arm in 1865, and cinnabar was soon discovered at the same site. Large amounts of mercury (Hg) were mined from 1873 to 1957, with the largest episodes of mining from 1873-1883 (3,000 MT) and 1927-1944 (1,100 MT), with a total of *ca.* 4,700 MT of Hg removed [8, 9]. The Sulphur Bank Mercury Mine was operated using open pit methods after 1927, and *ca.* $1X10^6$ MT of overburden and waste rock were moved. Some excavated material was pushed directly into the lake, and eventually 400 m of lake shore were covered with waste rock and tailings piles, about 10m high at the angle of repose; subsequent erosion from these piles into the lake was significant [10, 11]. Road building and lake filling activities also increased after 1920. Estimates by Goldstein and Tolsdorf [12] suggest that current erosion rates are roughly twice pre-European rates, with stream channel disturbance and road cuts/fills being the most important causes. Large areas of wetlands, which may have trapped significant quantities of sediment and nutrients, were lost to a 2,000 acre agricultural reclamation project on the north end of the lake, completed by the U.S. Army Corps of Engineers in 1928 [2]. Richerson *et al.*'s [2] review of the historical data also suggests that water quality declined after 1925 due to an increased flux of phosphorus and iron to the lake, and that the modern frequency and intensity of cyanobacterial blooms is a relatively recent phenomenon.

METHODS

The cores were taken from a 22' research vessel, using a handmade, drive-rod operated piston corer. The core sleeve was constructed from 2" x 10' Schedule 80 PVC conduit, and the piston was a modified 2" pipe test plug, with a nylon rope attached to the boat to keep the piston at the sediment-water interface during the driving of the core. The drive rod was 1.5" galvanized steel pipe in 11' sections. The corer was lowered to just above the sediment water interface, and the piston rope tied off. The core was driven in one steady push, as smoothly as possible, and retrieved using a winch on the boat. Just before the core was removed from the water, the bottom was capped to prevent any loss of material. Once on board, the core was split into three sections, approximately 1m long, by sawing around the core sleeve, and then cutting through the core with a knife. The sections were then capped top and bottom and sealed using silicone sealant to prevent moisture loss during the short storage period. Core sections were transported and stored vertically. Any storage was done at approximately 4°C in the dark. The core was extruded in 5cm sections immediately after X-ray photographs were taken. For approximately the top 30cm this had to be done vertically, as the sediments were very soft and easily disturbed. Below this level, cores were extruded horizontally onto an aluminum foil sheet, marked in 5cm intervals and sectioned. All sections were placed in preweighed 4oz specimen containers, weighed, homogenized, and then stored at 4°C under dark conditions.

Once all sections had been weighed and homogenized, subsamples were taken for total mercury (Hg) and methyl Hg, carbon (C), nitrogen (N), and phosphorous (P) analyses. Enough sample was taken for C/N, so that the remainder could be used for ^{210}Pb dating. Subsamples for the C/N and ^{210}Pb dating the sediment sections were dried at 60°C, and crushed by mortar and pestle to pass through a #60 mesh (250 micron) sieve. The remaining sample was stored at 4°C without light for use in pollen analysis. Percent loss-on-ignition (equivalent to organic matter), percent water, and bulk density were determined at the UC Davis - Clear Lake Environmental Research Center. Loss-on-ignition subsamples were dried at 105°C and then combusted at 500-550°C for two hours. The percent water presented is the average between the percent water values for the C/N and loss-on-ignition subsamples. Bulk density was determined by drying a known volume of sediment at 60°C, and calculated as the dry weight per unit volume.

Methyl Hg concentrations were determined at Battelle Marine Sciences (Sequim, WA) using the method of Bloom and Crecelius [13]. Total Hg (essentially inorganic mercury,) was determined at the UC Davis Environmental Mercury Laboratory on dry, powdered subsamples using a modified cold vapor atomic absorption method [14]. Carbon and nitrogen (C/N) concentrations were determined at the DANR Analytical Lab at UC Davis, using a Carlo-Erba Nitrogen Carbon Analyzer. The concentration of ^{210}Pb was determined in the laboratory of D.N. Edgington (Center for Great Lakes Studies, University of Wisconsin, Milwaukee) by the methods specified in Robbins and Edgington [15]. Deposition dates were computed from 210Pb concentration using the constant flux, constant sedimentation rate model of Robins [16]. Total P concentrations were determined at the University of California Hopland Research and Extension Center, using the methods of Sherman

[17]. Pollen counts were done by modifying the method of Fagri and Iverson [18]. All pollen data analyses were based on pollen percentage.

RESULTS

Dating

The ^{210}Pb data for the Upper Arm 2a core is presented in **Figure 2**. The best nonlinear fit to this data was derived with simplex and quasi-Newton minimization algorithms. The data are described by an exponential decay equation. **Table 1** presents the deposition dates corresponding to midpoints of each sediment section. Any error in the estimated sedimentation rate would also introduce an error in date assignments that increases cumulatively with depth. This is evident in the upper and lower 95% confidence limits. Although the dates are extended to the bottom of the core, it should be noted that these are extrapolations beyond the information actually contained in the ^{210}Pb signal, which typically decays below the limits of detection within approximately five half-lives (112 years). The calculated linear sedimentation rate is 1.33 cm y^{-1}, averaged over the length of the core from the sediment surface to 250 cm. Since the sedimentation rates in the lower part of this core may be overestimated, nominal dates earlier than about 1927 would be increasingly too young.

Physical and chemical changes downcore

The most striking pattern in the record are the recent coincident changes of many parameters in the upper 80 cm of the core (see **Figure 4** for physical data and **Figure 5** for chemical data). Above this horizon, sediments become markedly drier, have lower organic and nitrogen content, and contain significantly more inorganic and methyl Hg. The ^{210}Pb date estimate for this horizon is 1927. There is a smaller but statistically significant increase in Hg concentrations beginning at 145-150cm in the Upper Arm 2a core, with an estimated date of 1886. The spike probably represents the opening of Sulphur Bank Mine (first Hg production in 1873).

If our hypothesis that sedimentation rates increased significantly above 75-80 cm is correct, then the age of the 145-150 cm section would be an underestimate. In support of the interpretation that the sedimentation rate changed after *ca.* 1927, water content in both cores taken at the this site shows a dramatic decrease above 80 cm. This is atypical of steady state deposition, where water content would increase monotonically toward the sediment surface, as it does below the putative 1927 layer. Observations of sediment consistency, recorded during the core sectioning process, noted a dramatic change at this horizon as well. Furthermore, undated cores from other locations have thinner sections of drier, Hg-rich sediments than the Upper Arm 2 core, presumably reflecting lower sedimentation rates (authors, unpublished data).

For several parameters, the upper few centimeters of sediment show a partial return to pre-1927 values. This return is considerable for percent water, which rises to near the values one would expect by extrapolating the slowly rising pre-1927 values to the surface. Loss-on-ignition (organic matter) shows the smallest return to earlier concentrations. The decline in total (i.e. inorganic) Hg is considerable. Methyl Hg exhibits little consistent sign of return to previous levels, though there are distinct subsurface peaks. The secondary peaks in the methyl Hg profile are only loosely related to the total Hg content of the sediments, as substantial fluctuations in the methyl:total ratio show.

These changes must be interpreted in terms of the 1987-92 drought, during which time there was very little sediment input to the lake and relatively high rates of organic matter production. Bioturbation undoubtedly integrated the sediments deposited during these years with the four more normal years since the end of the drought. Nevertheless, recent changes appear to have started well before the drought. One exception is the surface peak of total P. We know from Richerson *et al.* [2] that increasingly large amounts of base extractable (iron and aluminum bound) P cycled from the water column (late summer) to sediments (winter). The large mass of P that appeared during the drought has resisted burial in the subsequent years, presumably due to its mobility during summer sediment anoxia.

The deeper sections of the core show similar features for most parameters. These data suggest a large increase in inorganic sediment load starting in about 1927 at the Upper Arm 2 site, which is somewhat close to the mouth of Kelsey Creek, a major tributary. As with water content, nearly all of the analyzed constituents show a major change at the 1927 horizon. Total C and N, and loss-on-ignition percentages show a large and rapid decrease above this depth, while dry bulk density shows concomitant increases. Total P declines slightly but rises again at the surface. The total P content of creek sediments is about 1,000 ppm [2], similar to values in the cores. Below the 1927 layer there is a coherent series of cycles involving percent water, organic matter and N.

Pollen Profiles

Pollen diagrams (**Figure 6**) show a very different pattern from the physical and chemical constituents discussed above.

Major taxa in the pollen record show little or no evidence of changes over the whole period represented in the core. The dominant pine and oak pollens show no detectable pattern of change at all. TCT (Taxodiaceae, Cupressaceae, Taxaceae) group pollen shows a small decrease over the past 100 years as does Compositae. *Salix*, *Acer*, *Artemisia*, and *Chrysolepis* increase somewhat in recent times. Only *Juglans* (walnut) records the agricultural activities of the European settlers. There is a hint that *Potamogeton* was absent after around 1925, and reappeared recently.

DISCUSSION

The early, relatively small increases in total and methyl Hg (e.g. at 145-150cm) are almost certainly a result of the early mining at the Sulphur Bank Mercury Mine, which started as a sulphur mine in the 1860's, and began extracting Hg in 1873. The much larger amounts of Hg in the sediments after 1927 are almost certainly due to the use of open pit mining methods. The disturbance of large volumes of overburden and rock, and the production of large volumes of tailings likely increased the loading of Hg to the lake ecosystem by three routes. First, mine waste rock was pushed directly into the lake. Second, sheetwash erosion from and wave undercutting of the waste piles transported Hg directly into the lake until the US EPA reduced the slope and rip-rapped the toe of the piles in 1992. Third, the mine currently discharges significant but poorly estimated volumes of acid mine drainage into the lake (Suchanek *et al.*, unpublished data). This Hg-rich discharge could be from surface or subsurface flows from the waste piles or fractures in the bedrock. The declines in total Hg from post 1927 peaks (estimated age 1961) appear to have begun too long ago to be explained by the recent drought or the US EPA erosion remediation work at the mine site in 1992. If the erosion remediation is having a material impact on Hg loading to Clear Lake, the sediments should provide a clear indication in the next decade.

The other coincident changes in physical and chemical characteristics of the core above the 75-80 cm level are probably not directly related to mining activity. Inorganic Hg in surficial sediments declines exponentially with distance from the mine [19]. Oaks Arm surficial sediments near the mine have total Hg concentrations of *ca.* 100-300 ppm in contrast to 3 ppm in the vicinity of Upper Arm 2, indicating that the direct effects of mining activity are far more important close to the mine than at distant sites. We believe the most likely cause of dramatic changes in the core profiles is the widespread application of powered earthmoving technology to streambed gravel mining, road construction, wetland reclamation projects, and Hg mining. This type of machinery was developed just before and after WWI, and its use became widespread in the late 1920s [20]. By the early 1930s, Caterpillar™ had a line of earthmoving machinery much like contemporary types. The core data are thus strikingly consistent with the hypothesis of increased sediment load and altered nutrient conditions. This hypothesis was previously proposed, based on historical records [2] and limited erosion surveying [12]. The end of active Hg mining in 1957 and the end of gravel mining in most streams by 1987 [7] may have resulted in a reduction in sediment loading, which is consistent with a trend toward moister, more organic sediments above 30-35 cm (1971) in the Upper Arm 2 core. Beginning in the winter of 1971-72 and persisting until the present, water quality monitoring data [2] show less winter turbidity, which is normally due to inorganic material derived from streams. Unfortunately, water quality data are only available back to 1968.

The decline in the N:P ratio to one third of pre-disturbance values is consistent with a shift to N fixing, cyanobacteria after 1925 (**Figure 7**). Historical records also suggest that rooted aquatic vegetation was abundant before 1925. Turbid water probably inhibited the growth of *Potamogeton* for several decades. In the last few years, severe iron limitation (Richerson, *et al.*, unpublished data), the causes of which are unknown, have produced clear water conditions again, and *Potamogenton* species are thriving once again, consistent with the record of *Potamogeton* pollen. The increases of *Salix* and *Acer* pollen may represent the colonization of disturbed stream channels by willow and box elder, whereas intact riparian vegetation tends to be dominated by oaks (*Quercus lobata*).

The absence of major impacts from the first half of the period of European settlement are striking. The transformation of the grasslands to grain fields and the replacement of native pasture grasses by Mediterranean weeds was a major impact that is unfortunately difficult to detect in the pollen record [21]. Nevertheless the overall stability of the vegetation and the lack of a conspicuous increase in sediment yield is remarkable. Judging from impacts recorded in cores, grazing, wood cutting and lumbering, agricultural clearing, and the development of small towns and recreation facilities, as conducted from 1854-1927, were relatively low-impact activities from a watershed mass-balance perspective. Simoons [6] also notes that no major shifts in the distribution of woody plant communities could be detected, apart from the conversion of valley floor oak woodlands and certain favorable hill slopes to agriculture (and urban uses).

CONCLUSIONS

The most dramatic impact of European settlement on the hilly and rural environment of the Clear Lake watershed occurred midway through its settlement history with the advent of powered earth moving equipment. It is especially interesting that the mining of relatively low grade ores by open pit techniques using heavy machinery caused a 10-fold greater loading of Hg to the lake than the labor intensive practices of the 19th Century, which had extracted 72% of the Hg produced by the Sulphur Bank Mercury Mine. The power of such equipment to expose soils to erosion and consequently to alter the lake environment appears to be considerably greater than overgrazing, clearance of lowlands for agriculture, limited mining and urbanization, and other land use changes which were quite conspicuous by 1927. Some evidence suggests that more stringent regulations (especially restrictions on gravel mining from stream channels) and mine remediation have begun to reduce the impacts of earthmoving activities on the watershed. Although Hg loading was somewhat higher 30 years ago, there is no evidence that the Sulphur Bank Mercury Mine will cease contaminating Clear Lake in the absence of further remediation.

Acknowledgements – We appreciate the advice and assistance of Lauri Mullen, Linnie Brister, Bradley Lamphere, and Cat Woodmansee. Grace Brush was a source of encouragement and much sound advice, as was Reds Wolman. Steve Zalusky's knowledge of the relevant aspects of Lake County mining and earthmoving activities was invaluable. Steve Hill, Steve Why, Tom Smythe, and Sue Arterburn smoothed collaboration with Lake County on the field work for this project. Bob Bettinger gave us sound advice on issues of prehistory. Many thanks to David Edgington for help with the ^{210}Pb analysis, and Shaun Ayers for Hg analyses. Support from the EPA Center for Ecological Health Research (R819658) and EPA Region 9 Superfund (68-S2-9005) made this project possible. Thanks to Carolyn d'Almeida and Jeri Simmons of EPA-Superfund and to Karen Moorehouse of EPA-ORD. Dennis Rolston and Cheryl Smith made many things easy. 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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Table Caption:
Table 1. Midsection depths and their corresponding deposition dates, with upper and lower 95% confidence intervals, for core Upper Arm 2a

Figure Captions:
Figure 1. Map of the Clear Lake area, showing core site Upper Arm 2
Figure 2. 210Pb data for core Upper Arm 2a
Figure 3. Physical data for cores Upper Arm 2a and b
Figure 4. Chemical data for core Upper Arm 2a
Figure 5a. Arboreal pollen data for core Upper Arm 2a
Figure 5b. Aquatic and Herbaceous pollen data for core Upper Arm 2a
Figure 6. N:P ratio for core Upper Arm 2a

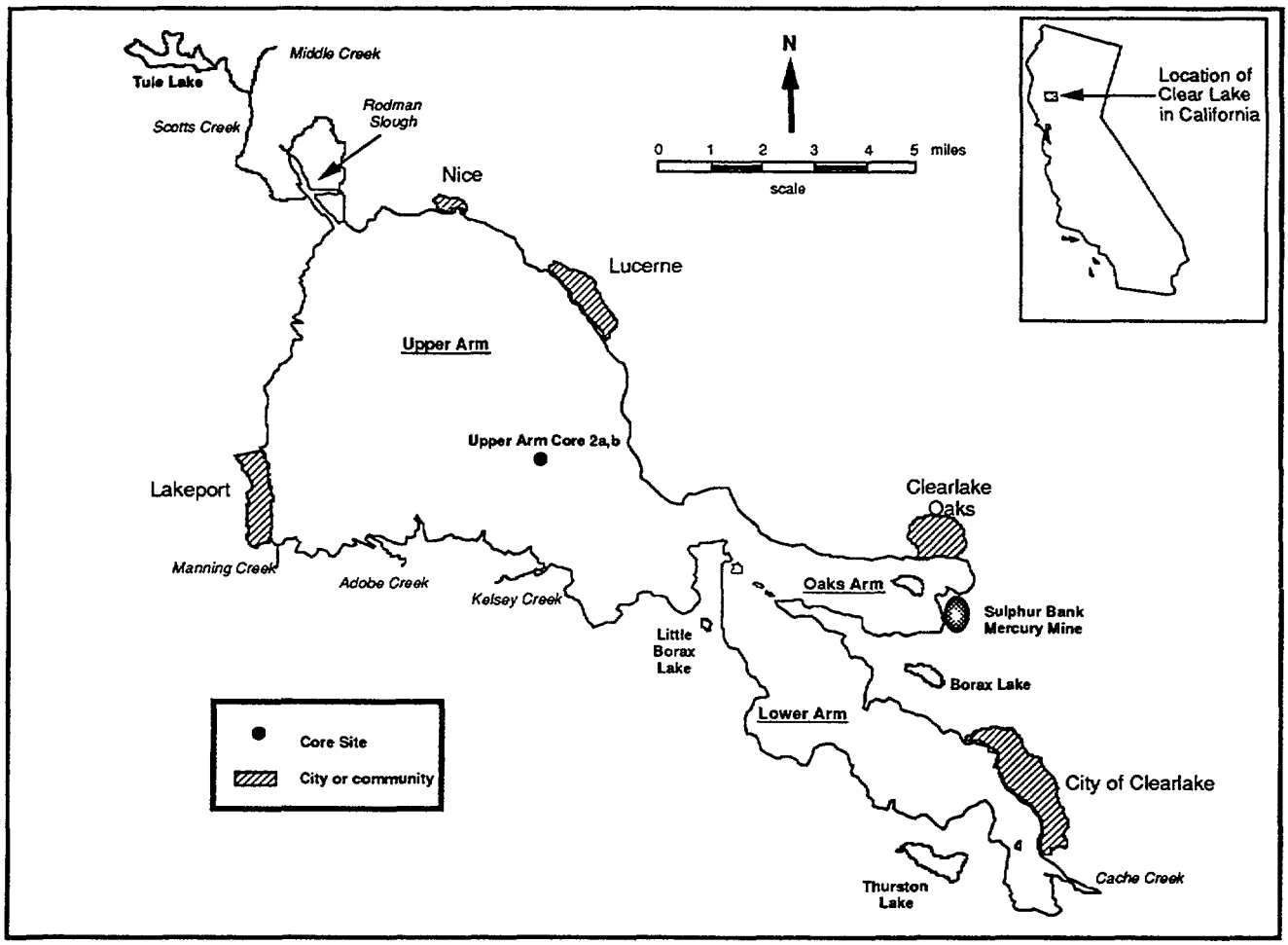


Figure 1. Map of the Clear Lake area, showing core site Upper Arm 2

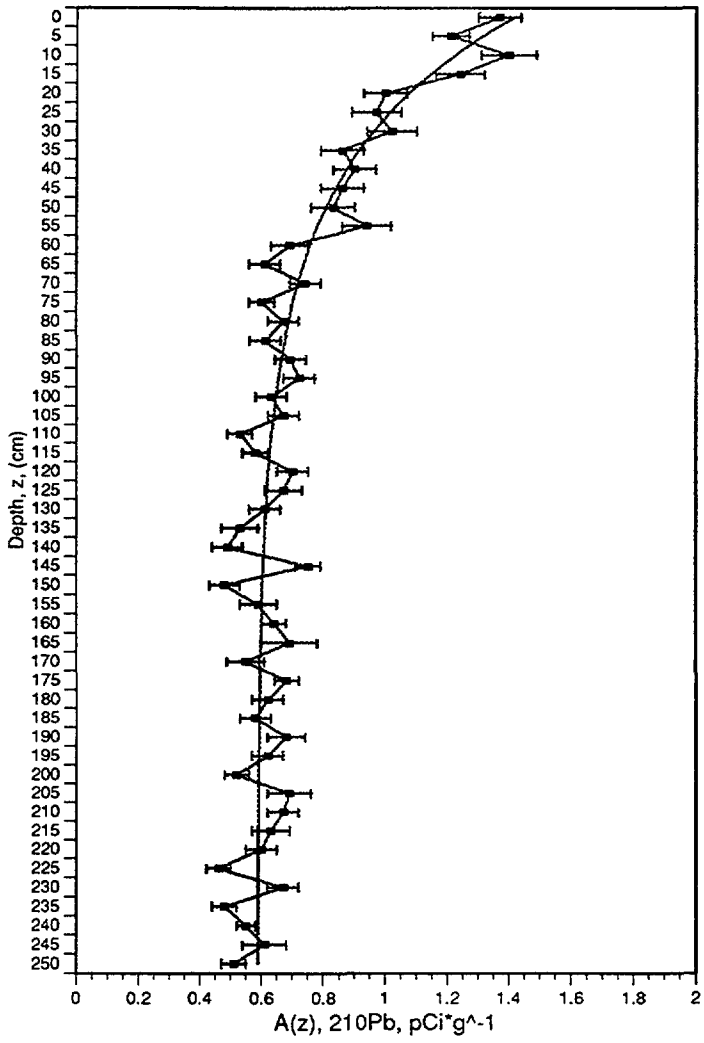


Figure 2. ^{210}Pb data for core Upper Arm 2a

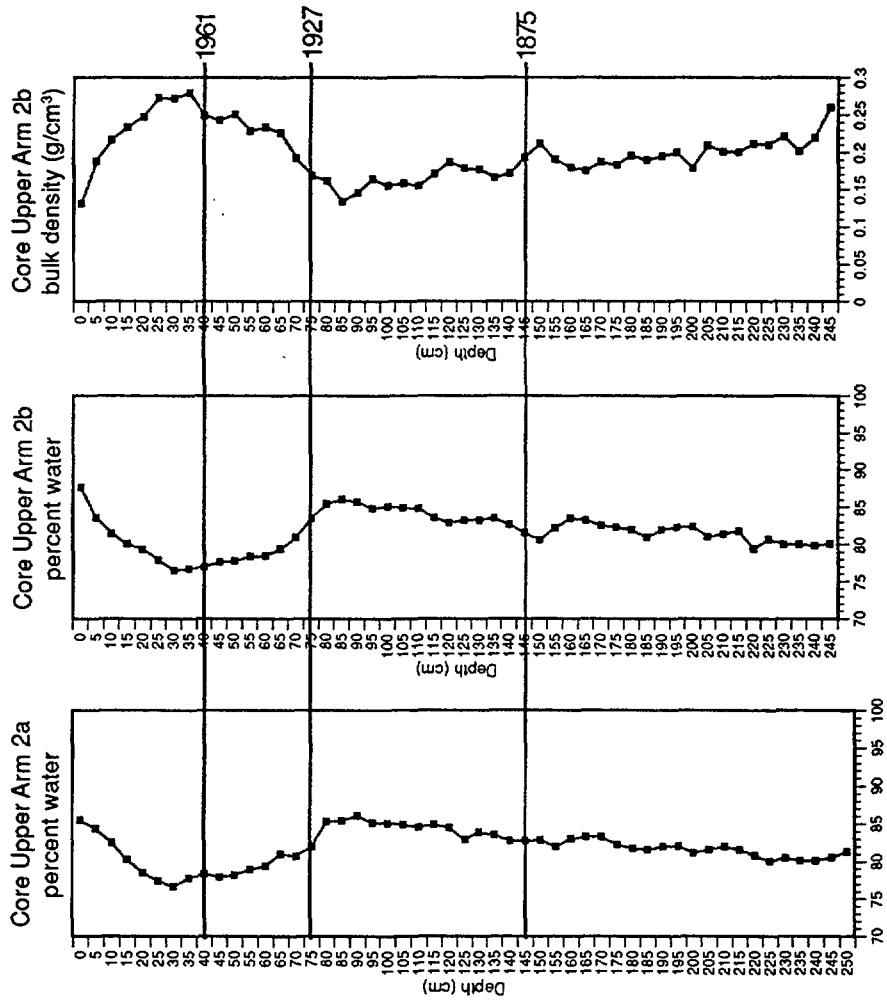


Figure 3. Physical data for cores Upper Arm 2a and b

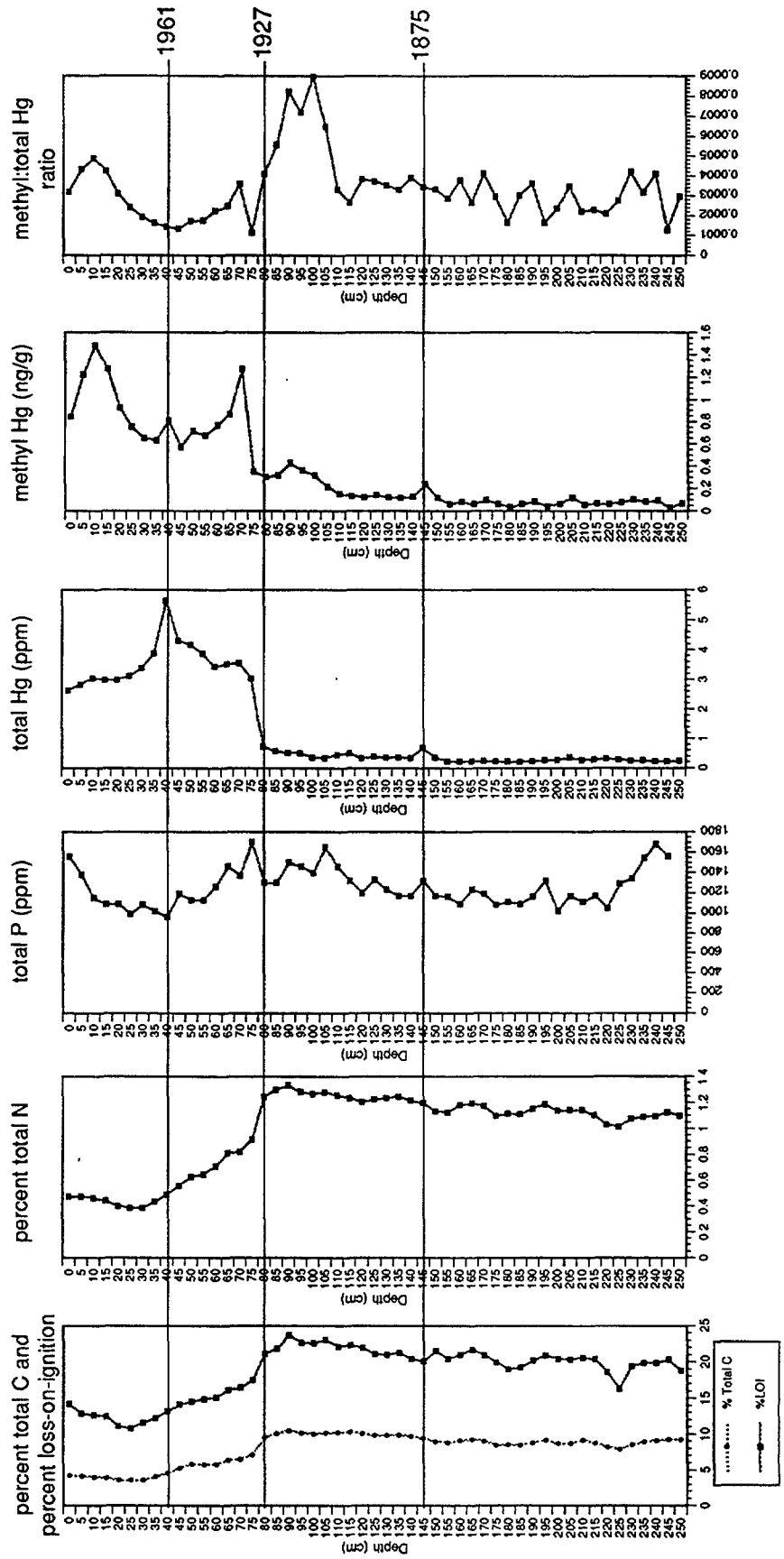


Figure 4. Chemical data for core Upper Arm 2a

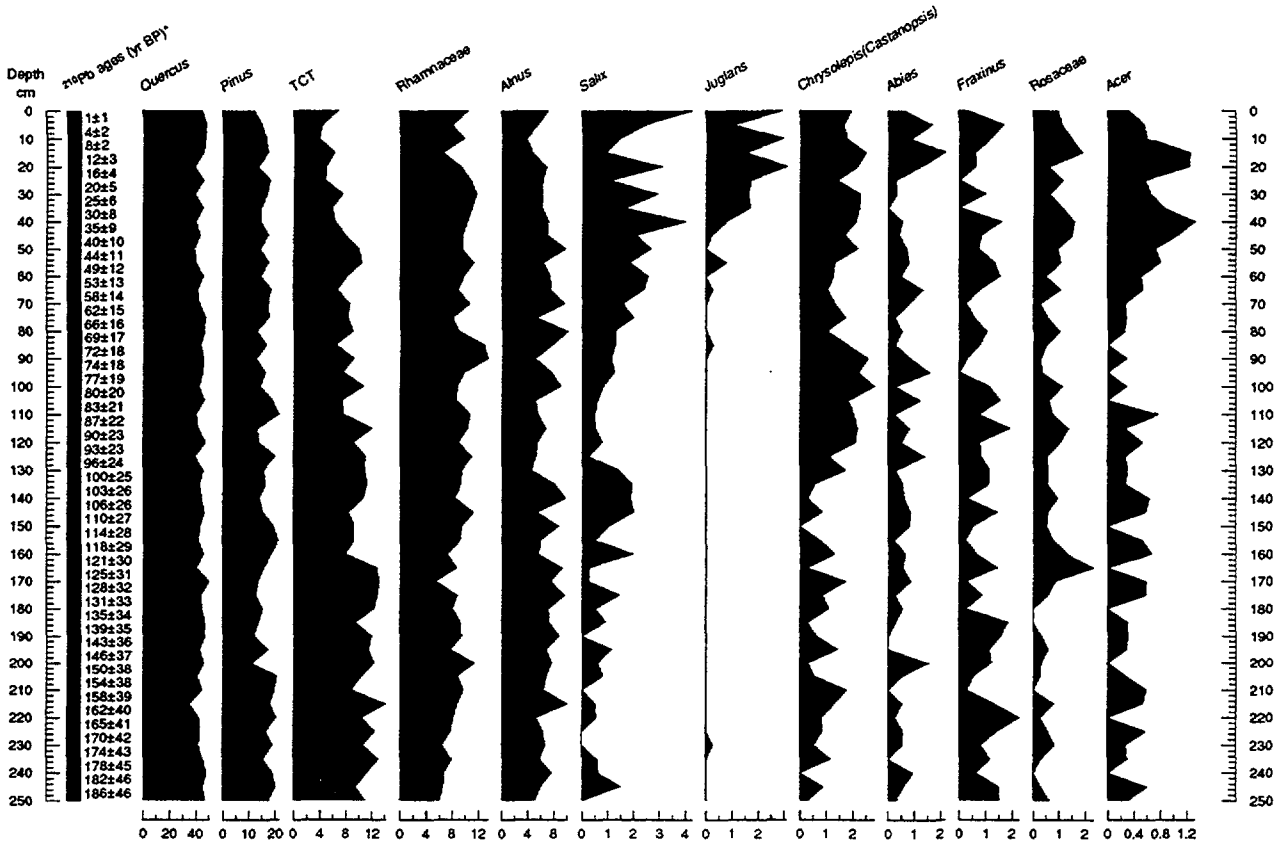


Figure 5a. Arboreal pollen data for core Upper Arm 2a

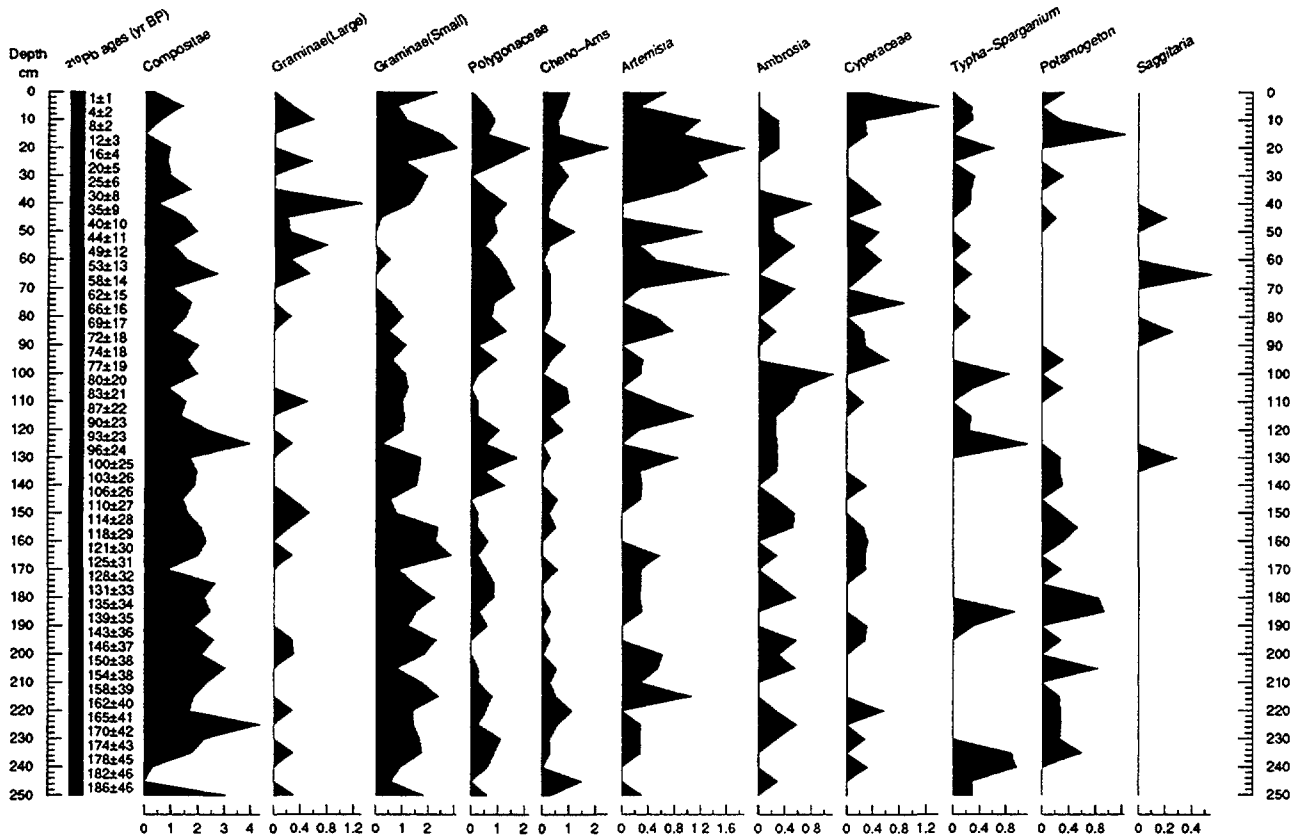


Figure 5b. Herbaceous and aquatic pollen data for core Upper Arm 2a.

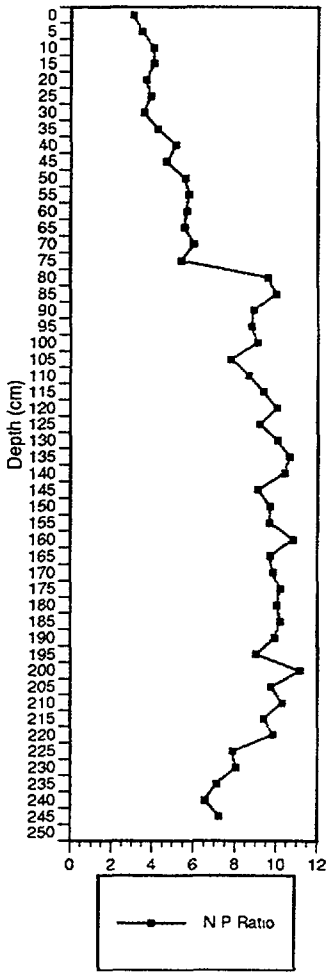


Figure 6. N:P ratio for core Upper Arm 2a

THE CLEAR LAKE EVENTS TIMELINE
A graphic display showing major events relevant to changes in Clear Lake Water Quality. (P)

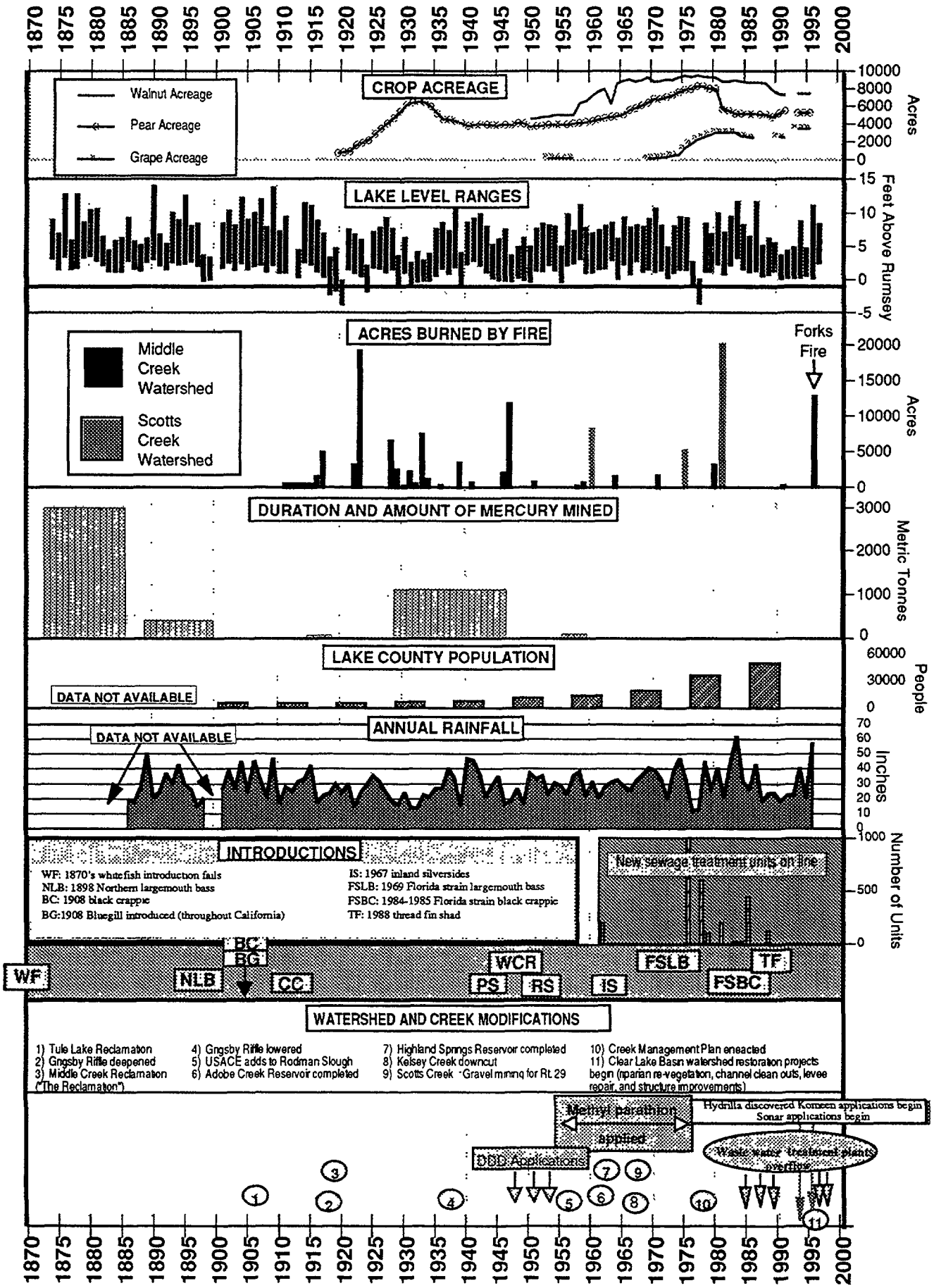
L.L. Brister[1]

[1]U.C. Davis - Clear Lake Environmental Research Center, 200
Park Street, Lakeport, CA

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 (UCD-CLERC) by several sources and agencies including : Lake County Flood Control and Lakebed Management, Lake County Environmental Health, Lake County Department of Agriculture, Lake Counnty Vector Control District, California Department of Fish and Game, Community Development Services, and the Mauldin and Goebel archives at the Lake County Museum historical library.


CROP ACREAGE
Walnut Acreage
Pear Acreage
Grape Acreage
Acres
LAKE LEVEL RANGES
Feet Above Rumsey
ACRES BURNED BY FIRE
Middle Creek Watershed
Scotts Creek Watershed
Forks Fire
Acres
DURATION AND AMOUNT OF MERCURY MINED
Metric Tonnes
LAKE COUNTY POPULATION
DATA NOT AVAILABLE
People
ANNUAL RAINFALL
DATA NOT AVAILABLE
Inches
INTRODUCTIONS
WF: 1870's whitefish introduction fails
NLB: 1898 Northern largemouth bass
BC: 1908 black crappie
BG:1908 Bluegill introduced (throughout California)
IS: 1967 inland silversides
FSLB: 1969 Florida strain largemouth bass
FSBC: 1984-1985 Florida strain black crappie
TF: 1988 thread fin shad
New sewage treatment units on line
Number of Units
WF
NLB
BC
BG
CC
WCR
PS
RS
IS
FSLB
FSBC
TF
WATERSHED AND CREEK MODIFICATIONS
1) Tule Lake Reclamation
2) Gingsby Riffle deepened
3) Middle Creek Reclamation ("The Reclamation")
4) Gingsby Riffle lowered
5) USACE adds to Rodman Slough
6) Adobe Creek Reservoir completed
7) Highland Springs Reservoir completed
8) Kelsey Creek downcut
9) Scotts Creek - Gravel mining for Rt 29
10) Creek Management Plan enacted
11) Clear Lake Basin watershed restoration projects begin (riparian re-vegetation, channel clean outs, levee repair, and structure improvements)
Methyl parathion applied
Hydrilla discovered Komeen applications begin Sonar applications begin
DDD Applications
Waste water treatment plants overflow

Lake County Career Center's Ecosystem Programs Update

Earl Brown
Ecosystem Management Project Coordinator
Lake County Career Center

In September 1995 the first ecosystem crew from Lake County Career Center began working in Lake County streams. It consisted of 11 members, funded by the Department of Labor to clean-up and repair flood damage sustained in the winter 94-95 storms. This was a temporary job creation program and work could be done on public land and private land for reasons of public health and safety. This crew focused on flood prevention, debris clean-up and erosion control using bio-engineering techniques. Bio-engineering is the use of vegetation to stabilize streambanks and filter the winter storm runoff helping to keep the soil in the upper watershed. In six months we removed tires, furniture, boat parts, petroleum product containers, appliances, steel, household garbage and more from miles of stream channel. The crew also planted thousands of willow cuttings along eroding streambanks in the Scotts Creek, Kelsey Creek, Adobe Creek and Middle creek watersheds. Work was also done with the Bureau of Land Management, South Cow Mountain Recreational Area on erosion control, gully stabilization and stream crossing repair.

Lake County Career Center used the success and experience gained with this first program to plan and implement the Ecosystem Management Technician Training Program. This project called for 200 hours of college level instruction and 1040 hours of on-the-job training. The class was held at Mendocino College's Lakeport Center and jointly taught by Bob Wallen of Mendocino College and Earl Brown from the Career Center. In this course students learned the basic principles and practices of ecosystem management. The focus was on assessment techniques for watershed management and restoration with emphasis on regional characteristics and interrelationships among biological, cultural, and social aspects of the watershed. Classroom material was reinforced by fieldtrips to key locations where students applied their skills to gather information and perform specific tasks concerning assessment and restoration of the watershed.

Lake County's Business Outreach and Response Team was the ecosystem's project employer of record and handled the payroll and contract negotiations. Contracts were written with the Mendocino National Forest, Lake County Flood Control and Water Management Division, Homestake Mine, New Growth Forestry and West Lake R.C.D. for projects including tree planting, exclusionary fencing for sensitive riparian areas, gully repair, and streambank erosion control. Rehabilitation work in Mendocino National Forest after the Forks Fire provided an opportunity for contracts in hillside erosion control, stream channel stabilization, re-vegetation and grade stabilization structures to retain sediment. A total of 5,895 hours of field work was completed during the Ecosystem Management Technician Training Program.

Lake County Career center is currently involved in a second TJC flood clean-up program and have had as many as 90 people on 15 separate crews placed around Lake County cleaning debris from the streams and other flood related jobs. The projects remain the same as the previous program revolving around clearing of stream channels to reduce the chance of localized flooding and the control of accelerated erosion. As last time tons of debris of all kinds are being taken from local streams. We have removed nearly 1000 tires, appliances by the dozens, refrigerators and air conditioners with the freon still in them, steel and truck loads of household garbage. Through this program and with the aid of the Office of Emergency Services we are collecting and disposing of this debris. From March 1 to October 1, 1997 over $780,000.00 had been spent on payroll alone for this current clean-up program. The program will continue until June 30, 1998.

Through these programs we have shown how people and agencies can form partnerships and alliances that can solve problems and work together through difficult times. In fact the partnerships that were created is the real success of the ecosystem programs. Through our work we helped some land that had a small yet positive effect on the streams that feed Clear Lake. People earned a living wage, gained some job and life skills and did a lot of good hard work for the local environment. The ecosystem programs from Lake County Career Center has enjoyed a good deal of success and through its development has introduced new and valuable information to the community. Since September 1995 we have directed over 2 million dollars into the local economy providing ecosystem management training to unemployed citizens. This includes payroll, support services to crew members and products rented or purchased from local merchants.

In closing I would like to thank U.C. Davis for having this symposium and giving us a place to exchange information and make it available to the public. I hope this is the first of many such gatherings here in Lake County. People are as diverse as our watersheds and are an integral component of any land management strategy. Partnerships based upon mutual trust and respect are necessary in order to build the common values and goals we need to develop and implement large scale landscape management. I believe we are well on our way with this process here in Lake County and I am glad to be a part of it.

Osprey Nest Behavior at Clear Lake, CA. (P)

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Key Words: osprey, nest behavior, Sulphur Bank Mercury Mine, Clear Lake, CA

In 1994, seven osprey nests at Clear Lake, Lake County, CA, were observed to determine nest behavior. Nests were observed from incubation to when the chicks fledged (started flying). Observation periods averaged just under nine hours each. Approximately 400 hours were spent observing nests. Data were coded to determine percent time each bird spent at each activity. During any time, each bird is doing one of four primary behaviors: flying (FLYI), standing in the nest (STAN), perched outside of nest (PERC), or out of sight (OOFS). An attempt was made to relate nest behavior to distance from the Sulphur Bank Mercury Mine.

The Lake County Resource Management Committee - An Interdisciplinary Approach to Achieve Resource Management Consensus

G.A. Giusti. Integrated Hardwood Range Management Program, University of California Cooperative Extension, Ag. Ctr./Courthouse, Ukiah, CA 95482.

Key Words: watershed, management, policy, conflict resolution

Lake County's watershed resources, and consequently its biological diversity, is managed through a fragmented system of private, city, county, state and federal control. Planners, consultants, agriculturists, commodity organizations, business and recreational promoters, special districts managers, engineers, biologists, resource agency personnel, area managers, regional managers, and statewide directors all have some level of impact on the County's biological diversity. Because of personal experiences, academic training, geographic limitations, and agency dogma this conglomerate of technical expertise often has a great deal of difficulty communicating each others point of view. Instituting attitudinal changes, increasing subject awareness and promoting inter-disciplinary communications and partnerships between all levels of decision makers is essential to the long-term stability of Lake County's biodiversity. This paper discusses a conceptual model developed to demonstrate the effectiveness of the Lake County Resource Management Committee, Lake County, California. The model demonstrates the importance of recognizing the inter-disciplinary aspects of human resource management, various communication techniques necessary to promote resource awareness, and the importance of building inter-disciplinary partnerships, and long-term networking relationships. The model is useful in demonstrating the importance of adopting an inclusive approach to resource management that could be adopted for other projects addressing multiple resource issues.

RECENT ENVIRONMENTAL CHANGE OF CLEAR LAKE BASIN SHOWN BY POLLEN ANALYSIS. (P)

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Abstract- Since European settlement in the basin, the Clear Lake watershed has changed due to human activities. This study was conducted to estimate changes the vegetation of the Clear Lake basin using pollen analysis of three cores, CL-UA-02, CL-UA-03, and CL-OA-03.

Fluctuations of the pollens of major taxa are coincident among sites and there was no detectable change in the rain of pollen from the dominant oaks and pines. Taxodiaceae, Cupressaceae, and Taxaceae (TCT) pollens have continually decreased but walnut, *Salix*, *Acer*, and large grass pollens have increased recently. *Potamogeton* pollen was not observed between 55 and 95 cm intervals of UA-02 core. The rain of pollen from aquatic species tends to decline in droughts.

We can conclude that climate has been stable during covered period. Human activities have resulted in small-scale disturbance, especially in wetlands. Water quality has been increased since 55 cm by unknown factor.

Key words- pollen analysis, Clear Lake, human activity, environmental change

INTRODUCTION

The Clear Lake basin is one of the first sites of human occupation in the North America, probably because of abundant acorns, fish and other resources (1). However, significant disturbance prior to European settlement by native Americans might not have occurred (2). Beginning in the 1840s, European settlers started farming and ranching (3). Mining for sulfur and mercury began in the late-19th century and continued until the mid-20th century (4). In the 20th century, agriculture, cattle farming and tourism have been the most important industries in the basin (5). In the process of these human activities, Clear Lake watershed has been changed in the small and large scale. Especially, agricultural activities in the 20th century changed the landscape of the north-eastern part of the Clear Lake watershed. Vegetation mapping has been insufficient to directly document changes.

Pollen is well preserved in anaerobic conditions because it is made of decay-resistant cellulose-protein compounds (6). Thus the record of pollen buried in lake sediments can show the historical change of plant community. Adam et al. (7) described the vegetation change in the upper Quaternary cores from Clear Lake. They focused on long term climatic change through pollen composition change, not on recent history. This study was done to reveal the environmental change of the Clear Lake basin during the last ~200 years.

METHODS

Samples for pollen analysis were obtained from UC Davis Clear Lake Research Center. For sampling procedures, Becker et al.(5) and for sampling sites, Suchanek et al.(8). Pollen analysis was done by the modification of the standard method of Fagri and Iversen (6). Subsamples were boiled 10 min. in ten percent KOH in water bath to remove organic acids and washed several times with distilled-deionized water until upper liquid became clear. Ten percent HCl and ninety five percent ethanol were added and the samples boiled again to remove calcium carbonate. HF was added and samples were boiled in water bath in fume hood for 5 minutes to remove siliceous matter. The ethanol/HCl treatment was repeated. To remove coarse cellulose on the pollen surface, acetolysis was done for 3 minute in a boiling water bath. Samples were rinsed with glacial acetic acid to remove remaining water before and after acetolysis. After three washing with distilled-deionized water, glycerol was added. Minimum 300 pollen grains were counted under100 or 400 magnification microscope. The remaining solution was stored in a small glass vial. All data analyses were based on pollen percentage of the taxa enumerated.

The CL-UA-02 core was dated by lead 210 dating method by Becker et al. (5). Lead 210 dates over 110 ybp (year before present) are linear extrapolations.

RESULTS AND DISCUSSION

50 pollen groups including *Quercus*, *Pinus*, TCT (Taxaceae, Cupressaceae, Taxodiaceae), Rhamnaceae *Alnus*, *Salix*, *Juglans*, *Chrysolepis*, *Abies*, *Fraxinus*, Rosaceae, *Acer*, Compositae, Graminae, Polygonaceae, *Chenopodium-Amarantus*, *Artemisia*, *Ambrosia*, Cyperaceae, *Typha-Sparganium*, *Potamogeton*, and *Saggitaria* were observed in the Clear

Lake cores. *Quercus*, *Pinus*, TCT, and Rhamnaceae pollens were the dominant taxa. In the pollen profile of CL-UA-02 core (Figure A) which was analyzed based on 5 cm interval, *Quercus* and *Pinus* pollen frequencies are constant with depth and indicating that climate has been constant over covered period. TCT pollen frequency has been decreased. TCT group represents primarily genera Calocedrus, Juniperus, Cupressus, and Sequoia of which *Calocedrus* and *Cupressus* are represented in the Clear Lake basin. *Salix* pollen frequency has two peaks: one is at the depth of 135-145 cm and the other is at the depth of 35 cm. The increase of *Salix* pollen might be caused by decrease of water level of wetland because the northeastern wetland were drained and diked for agriculture. Even though *Juglans* (walnut) pollen appeared at the depth of 230 cm, the main increase of this pollen started at 55 cm depth. Lead 210 dating estimates this depth as 1942 and this is consistent with the post W.W.II development of the walnut industry(9). This fact confirms the accuracy of lead 210 dating. *Acer* pollen (box elder and big-leaf maple are present in the basin) frequency has increased since the depth of 80 cm even though relative frequency is low. Compositae pollen including high spine group and low spine group has decreased but *Artemisia* pollen has increased. Graminae pollen associated with ranching decreased until 40 cm depth and increased again. *Chrysolepis* pollen frequency has increased since the depth of 130 cm. *Potamogeton* pollen was not observed between 55 and 95 cm intervals and this result is consistent with historical record (5) which suggests the sequential change of water quality: clean water (abundant aquatic vegetation until 1925), turbid water, and clean water in recent years, with *Potamogeton* becoming abundant again in the 1990s. Even though there has been the development of many small towns and recreational facilities, the frequency of *Chenopodium* and *Amarantus* pollens which are indicators of human activities (10) have not increased significantly. Thus, human activities can be judged as non-destructive activities from this record.

Pollen taxa in CL-UA-03 core and CL-OA-03 core (Figure B, C) which were analyzed based on 10 cm intervals, were the same as in CL-UA-02 core. Frequencies of some pollen taxa including *Salix* and *Potamogeton* in CL-UA-03 core and CL-OA-03 core were lower than those in CL-UA-02 core. This might be caused by the differences in regional topography and vegetation. The big main tributaries and broad wetland are in the north-west side of Clear lake and small wetlands are present along toe of Big Valley on the south shore of the Upper Arm. Oaks and Lower Arms are surrounded by steep slope and small wetlands. Even though the changing patterns of TCT, *Salix*, and *Juglans* in CL-UA-03 core and CL-OA-03 core were similar to those in CL-UA-02, increasing or decreasing depths were different. The change of TCT, *Salix*, and *Juglans* at 60 cm depth in CL-UA-02 corresponded to the change of those pollen at 30 cm depth in CL-UA-03 core and CL-OA-03 core. This indicates that the sedimentation rate in CL-UA-02 is almost two times of that in CL-UA-03 core and CL-OA-03 core. This difference is also reflected in the methyl mercury profiles and nutrient profiles of the cores(5).

The proportion of major components in three cores was almost the same (Figure D, E, F). Average pollen sums of arboreal, herb, and aquatic taxa are 93.5 percent, 5.6 percent and 0.9 percent respectively. Because aquatic taxa is very sensitive to water depth (11), this group can be used to indicate the water level of Clear Lake. To show the water level change with time, annual precipitation data in Clear Lake region are given (Figure G). There is a common changing pattern of aquatic taxa among three cores (Figure D, E, F) and this pattern can be compared with precipitation pattern (Figure G). The peak of aquatic taxa at the depths of 70 cm in CL-UA-02 core and 30 cm in CL-UA-03 core and CL-OA-03 core correspond to precipitation peak in 1940. The peak of aquatic taxa at 95 cm in CL-UA-02, 60 cm in CL-UA-03 core and 50 cm in CL-OA-03 core correspond to precipitation peak in 1916. The intervening minimum of aquatic taxa appears to reflect the prolonged drought of the 1930s. As indicated in the TCT, *Salix* and *Juglans*, this result shows that the sedimentation rate in the upper part of CL-UA-02 core is almost two times of that in CL-UA-03 and CL-OA-03 cores. However, the sedimentation rates in the lower parts of three cores might be almost the same. This result suggests that recent agricultural activity in the north-west of Clear Lake has increased the erosion and sediment loading.

In sum, pollen records suggest changes of wetland and aquatic systems, not of grassland, chapparal and forest communtiies. The impact of agricultural activities in wetland resulted in the increase of willow (*Salix*) and box alder (*Acer*). Disturbance of the riparian areas of streams, probably historically dominated by *Quercus lobata*, most likely favored the weedier *Salix* and *Acer* genera after European settlement. Beside agricultural activities in wetlands and riparian corridor encroachment, much level grassland area was converted first to grain fields and later to orchards and vineyards. On steeper slopes native grasses have been replaced by Mediterranean weeds. This change is difficult to detect in the pollen record (12) because Graminae pollens are very similar and it is not practical to identify them in genus or species level. However, we might detect this change because agricultural grain pollens are generally larger than grass pollens. The low frequency and small change of large Graminae pollen suggest that we have failed to resolve these changes. The pollen record based on 5 cm interval (Figure A) has much better resolution to show the short-term scale change than the pollen records based on 10 cm interval (Figure B, C). This suggests that pollen analysis based on short interval can show all the changes of plant community better than pollen analysis based on long interval.

CONCLUSION

Climate has not changed during the period covered by this study but the recent increase of small scale disturbance in wetland and shallow aquatic system by human activities was detected. Willow and box elder have colonized disturbed stream channels in recent year. The frequency of Oak (dominated by upland species) and pine pollen has not changed. Human activities have had modest impacts on the upland communities. Water quality was changed, probably mainly by streambed disturbance, road building, and similar activities. The disappearence of *Potamogeton* in the Upper Arm core from approximately 1925 until the top of the core may signal more turbid water resulting from the use of powered earthmoving machinery in the aggregated extraction and road building industries.

The comparison of the frequency change of TCT, *Salix*, *Juglans* among three cores, and of the aquatic pollen sum of three cores and precipitation, showed that the sedimentation rate of the upper part of CL-UA-02 was almost double of that of CL-UA-03 and CL-OA-03 cores. This increased sedimentation rate since 1940 implies the increase of human impacts on sediment loading through agricultural activities in north east of Clear Lake.

Even though pollen analysis is useful to track the climatic change and the land use change, it has limited value to track some important changes known to have occurred, such as the change of grassland to grain field and the replacement of native grasses by Mediterranean weeds.

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Late Holocene Diatom Biostratigraphy of Three Short Cores Retrieved from Clear Lake, Lake County, California. (P)

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Abstract - Three cores (ca 2.5 m) spanning the last 250 - 350 years were retrieved from the Upper, Oaks and Lower Arms from Clear Lake, CA. Diatom biostratigraphic assemblages were described from these cores. A total of 115 species from 20 genera were identified, some of which are as yet undescribed. Species assemblages are dominated by centric diatoms such as the genera *Stephanodiscus* spp. and *Aulacoseira* spp. (characteristic of eutrophic, high conductivity conditions). The average Shanon-Wiener Diversity Index was 2.13 (ranging from 2.01 for UA-02 to 2.21 for LA-05). Average diatom concentration was calculated at 9.08×10^6, 1.01×10^7 and 6.89×10^6 diatom valves/g dried sediments/year in the Upper, Oaks, and Lower Arms, respectively. A decrease in diatom productivity for all cores was noted in the recent sections. This decrease may be concurrent with the onset of watershed changes such as the increased erosion from road building, mining activities, wetland alteration and the application of pesticides in the latter part of this century.

Keywords - Clear Lake, Northern California Coastal Range, centric diatoms, paleolimnology, eutrophy, late Holocene

Introduction

Clear Lake is a large (17.6 kHa) eutrophic natural lake located at an elevation of 404 m in the Northern California coastal range (39°00'N, 122°45'W). It is the largest natural lake within the borders of California. Clear Lake is believed to have existed and been shaped by faulting, volcanism and erosion over the last 500,000 years and possibly longer. Clear Lake, is a warm, shallow (7.1- 11.1 m), and polymictic system with three distinct sub-basins or arms. It occupies a shallow basin in the Central Coast Range about 80 miles north of San Francisco (Figure 1). Clear Lake, better described by its Pomo Native North American name Lupiyoma (large water), is a warm three basin (Upper; Oaks and Lower Arms) foothill lake with a Mediterranean climate and a large drainage basin to lake area ratio (about 12:1) [1]. Clear Lake is a moderately hard water system with an average pH of 8.05 and an average conductivity of 265 µmhos/cm. Clear Lake has long been culturally and economically significant to Native Americans tribes that have settled their shores. An "Early Man" site has been documented [2] at Borax Lake, a small closed basin lake on the peninsula between the Oaks and Lower Arms of Clear Lake. Most of the inflows occur via Middle, Scotts, Adobe and Kelsey Creeks which meet the lake in the Upper Arm. The Clear Lake basin was shaped by a variety of processes (faulting, volcanism) over the last 1 to 2 millions years (Richerson, 1994). The lake is located over multiple faults to the NE and SW. The shifting of these faults, together with the occurrence of explosion craters (eg. Little Borax Lake) which hurled debris into the lake, explain the irregular outline of the lake's shore. Downward vertical movement within the basin created by the active faulting is at a rate approximately equal to the average sedimentation rate of 0.9mm/ yr [4]. A nearly continuous sequence of lake sediments reaching back 480,000 years has been recovered by the US Geological Survey [4] and other lake sediments in the region date back to the early Pleistocene 1.8 - 1.6 million years ago [5]. The diatom sequence in the cores indicate that Clear Lake has been a shallow, productive system, floristically similar to the modern lake, since the end of the Pleistocene [6]. Clear Lake is usually well mixed across its shallow depths. It usually stratifies during summer days due to solar heating of the top few meters of water, and usually circulates each night. Numerous gas vents and water springs provide an additional source of vertical mixing. Lake temperatures vary between ca 7°C in the winter and ca 27°C in the summer.

Description of Sites and Cores Studied

One site in each arm (Figure 1) was chosen for coring. Cores ca 250 cm long were retrieved vertically from the surface of the lake's sediments [7]. An array of various biostratigraphic profiles (pollen, sedimentology) were described in [7]. The cores span ca 250- 350 years of the late Holocene sedimentation record. Not all parameters have been

measured on all cores, and dating is available only from UA-02 at the time. The ^{210}Pb data for the UA-02 core is presented along with the diatom assemblage of that core (Figure 2). The calculated linear sedimentation rate is 1.33 cm y^{-1} averaged over the length of the core from the sediment surface to 250 cm. The most striking pattern in the record are the recent coincident changes of many parameters in the upper 30-80 cm of the cores. Above this horizon, sediments become markedly drier, have lower organic and nitrogen content, and contain significantly more total and methyl Hg [7].

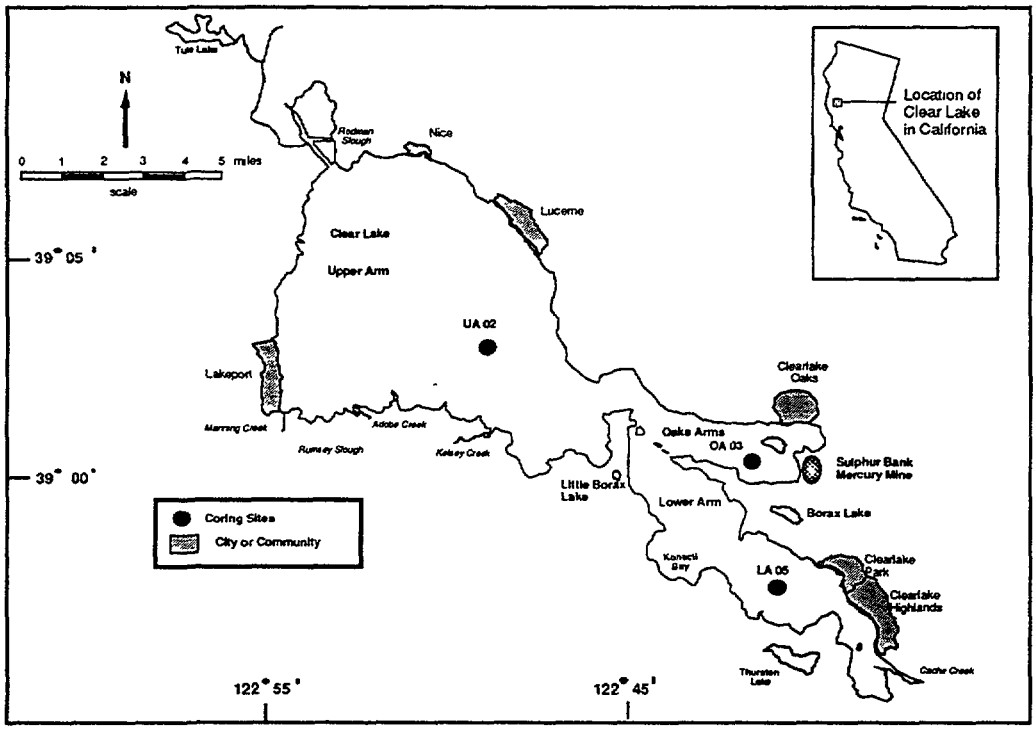

Figure 1: Clear Lake Map with Coring Sites and Location in California

Materials and Methods

To describe diatom assemblages the cores were subsampled at 5 - 10 cm intervals; sediments samples were placed in a drying oven at 105°C overnight. About 0.10 to 0.15 g of dried sediments were oxidized with 30% hydrogen peroxide and washed with deionized water. 2mL aliquot of homogenized sediments were placed in a Battarbee plate (evaporation plate fitted with 4 cover slip depressions) [8]. Coverslips were mounted onto slides with Cargile mounting media. For each slide 300 diatom valves were counted using a Zeiss phase microscope at 1000X power. Diatom taxonomy was based primarily on [9,10,11,12,13,14,15,16,17,18,19]. A photographic and taxonomic diatom database was created using a SLR camera mounted onto the microscope. On an area of $1mm^2$ total diatom valves were enumerated, and diatom productivity was calculated using algebraic functions. Statistical indices (Shanon-Wiener Diversity Index and Pielou's Equitability Index) were determined using the following statistical formulas:

Shanon-Wiener Diversity Index (H)

$H = -\Sigma P_i \times \log_2 (P_i)$

where P_i equal the proportion of each species, varieties.

Pielou's Equitability Index (E)

$E = H / H_{max}$ where $H_{max} = \log_2 S$

where S equal total number of species present in a community.

Results

The dominant diatom taxa encountered in the Clear Lake cores studied were: *Aulacoseira ambigua* (Grunow) Simonsen; *Aulacoseira distans* (Ehrenberg) Simonsen; *Aulacoseira granulata* (Ehrenberg) Simonsen; a

yet undocumented unconfirmed species temporarily named *Aulacoseira konoctinensis* Meillier; *Cyclostephanos dubius* (Fricke) Round; *Cyclostephanos invisitatus* (Hohn & Hellerman) Theriot, *Cyclotella krammeri* var *planetophora* (Cleve) Hakånsson; *Fragilaria crotonensis* var. *constricta* Ehrenberg; *Stephanodiscus niagarae* Ehrenberg; *Stephanodiscus subtransylvanicus* var. *minutula* Gasse/; *Stephanodiscus minutulus* (Kützing) Cleve & Möller the *Stephanodiscus* complex was not optically discernible with the Zeiss microscope.

The diatom biostratigraphy of the three cores studied (Figures 2, 3 and 4) was dominated by centric diatom genera such as *Stephanodiscus* and *Aulacoseira*. These two genera represent more than 90% of the total diatom abundance in all cores. Nineteen diatom genera were identified from which 115 species were documented. Average diatom concentration across all arms (Figures 5, 6 and 7) was at 8.69×10^7 diatom valves/g/yr of dried sediments. The average diatom concentration was found to vary only slightly between cores. All cores show a decrease in diatom concentration in the recent sections starting in the 1940's and UA-02 suggests a long term decline througout the entire core. Diatom abundance and preservation in lake sediments is a function of production in the overlying water column [19] and loss processes, of which dissolution and outflow losses are the most important. Unfortunately, these processes are complicated further by the spatial heterogeneity of sedimentation and productivity. Hence, the diatom concentration data should be taken with caution. With the decrease in frequency, a shift in diatom composition was also observed. The dominance of the *Stephanodiscus subtransylvanicus* complex and *Aulacoseira granulata* is weakened by the increase in abundance of *Aulacoseira ambigua* and *Aulacoseira distans* in more recent sections.

The Shanon-Wiener Diversity Index (H) exhibited a high degree of constancy (ca. 2.0) throughout the older sections of all cores up to at least 75 ybp. From that time period, H exhibited a consistent increase in the upper sections of all three cores, corresponding to approximately the last 75 yrs in the Upper Arm (UA-02, Fig. 5), the last 25 years in the Oaks Arm (OA-03, Fig. 6) and Lower Arm (LA-05, Fig. 7). Pielou's Equitability Index (E) closely mirrored the trends exhibited by the diversity index throughout all cores and was found at an average of 0.55.

Discussion and Conclusions

The modern phytoplankton community at Clear Lake is represented by a predominance of Cyanophyceae (blue greens) over Bacillariophyceae (diatoms).

Clear Lake is a classic eutrophic lake in which massive algal blooms deplete the apparently inexhaustible combination of winter nutrient inflow and summer recycling [3]. Horne's work [1] has shown that Clear Lake is an iron limited phosphorus rich system in the 1970's, a finding confirmed by Li, et al. (this volume). Iron is involved in the enzymatic and respiratory pathways of algae. Iron limitation consequently reduces cyanobacterial generated nitrogen.

Clear Lake diatom flora is dominated by centric morphotypes, particularly two genera: *Stephanodiscus* and *Aulacoseira* in the Late Holocene period. These two genera represent more than 90% of the diatom biostratigraphic profiles for all cores across all depths studied. These planktonic genera are indicative of a nutrient rich limnic system [20], Clear Lake being a hard water (200- 300 microhms/cm) alkaline system. *Aulacoseira granulata* is a summer to late summer taxa, and tends to dominate in shallow, turbid lakes with abundant nutrients, especially silica [21]. *Stephanodiscus* competes successfully for silica when phosphorus levels are high and blue-green algae diminished. Small *Stephanodiscus* species bloom in low light conditions during the winter or early spring when nutrients, especially phosphorus, become abundant and winds are sufficient to transport and suspend the cells in the photic zone [22,23].

The last 70-75 years indicate a change in limnological conditions with the shifting of diatom taxa to more stress tolerant species such as: *Aulacoseira ambigua, Aulacoseira distans* and possibly *Aulacoseira konoctinensis* (newly observed species) and likely represent a shift to a more blue-green dominated system, although this cannot be confirmed microscopically because blue-greens leave no valve remains. The decrease in diatom concentration is concurrent with the onset of these taxonomic distribution shifts. These shifts in diatom distribution and frequency may be concurrent with alterations in the uses of the Clear Lake watershed such as destruction of wetlands, increased erosion from roadbuilding and associated gravel mining from creek beds, stream flow alterations, pesticide applications and open pit mining practices at the Sulphur Bank Mercury Mine. Furthermore, nutrient inputs usually increases with human population growth and catchement size [24]. Clear Lake has a large catchment area to lake ratio, hence it is sensitive to watershed disturbances. Though, the most probable direct impacts on the diatom flora are increases in winter and spring turbidity, which reduces the light available below growth conditions during the usual season of dominance by diatoms.

There is a great need to investigate the nutrient and pollutant thresholds for common stress tolerant

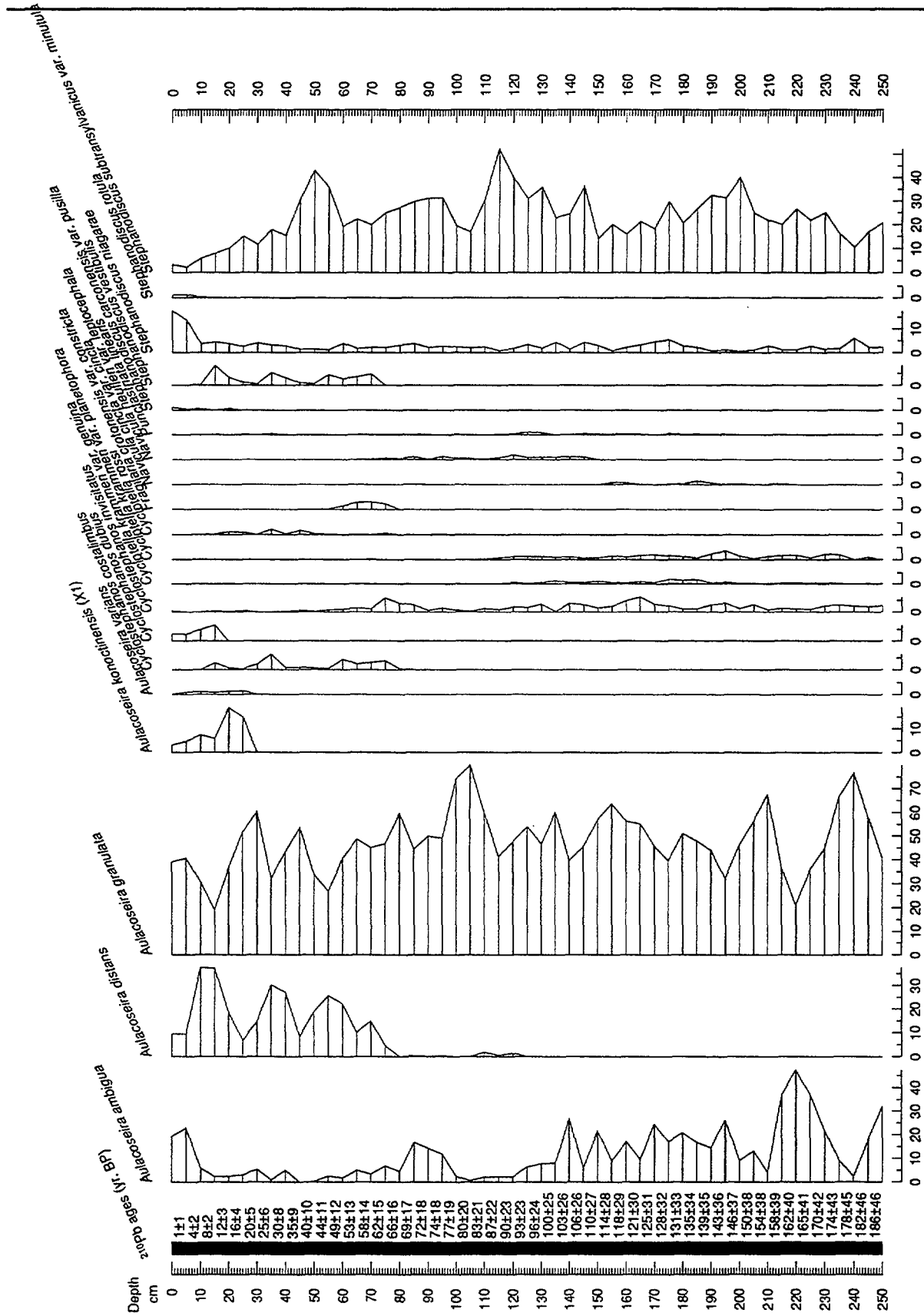

Figure 2: Diatom Biostratigraphy of UA- 02

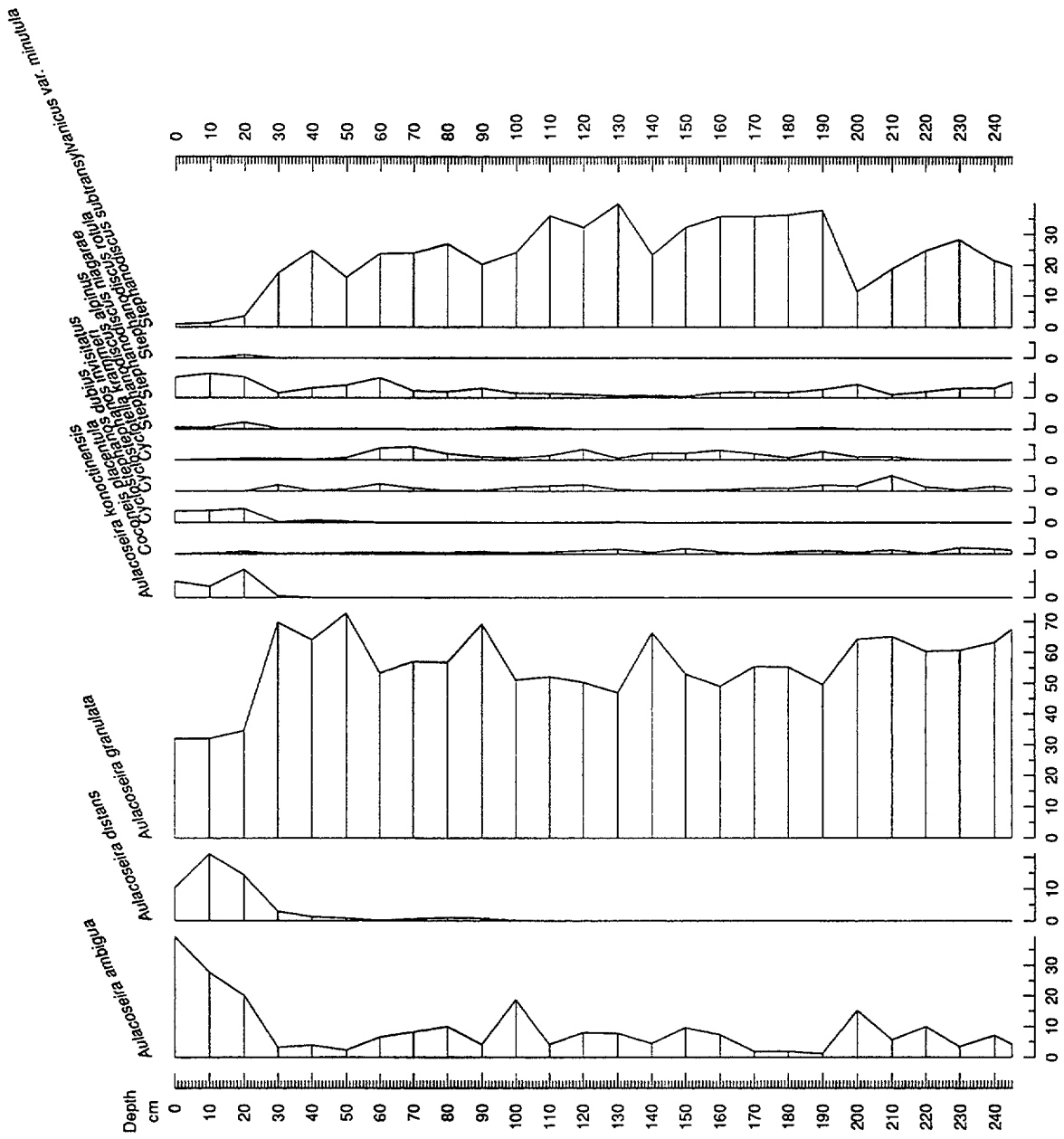

Figure 3: Diatom Biostratigraphy of OA-03

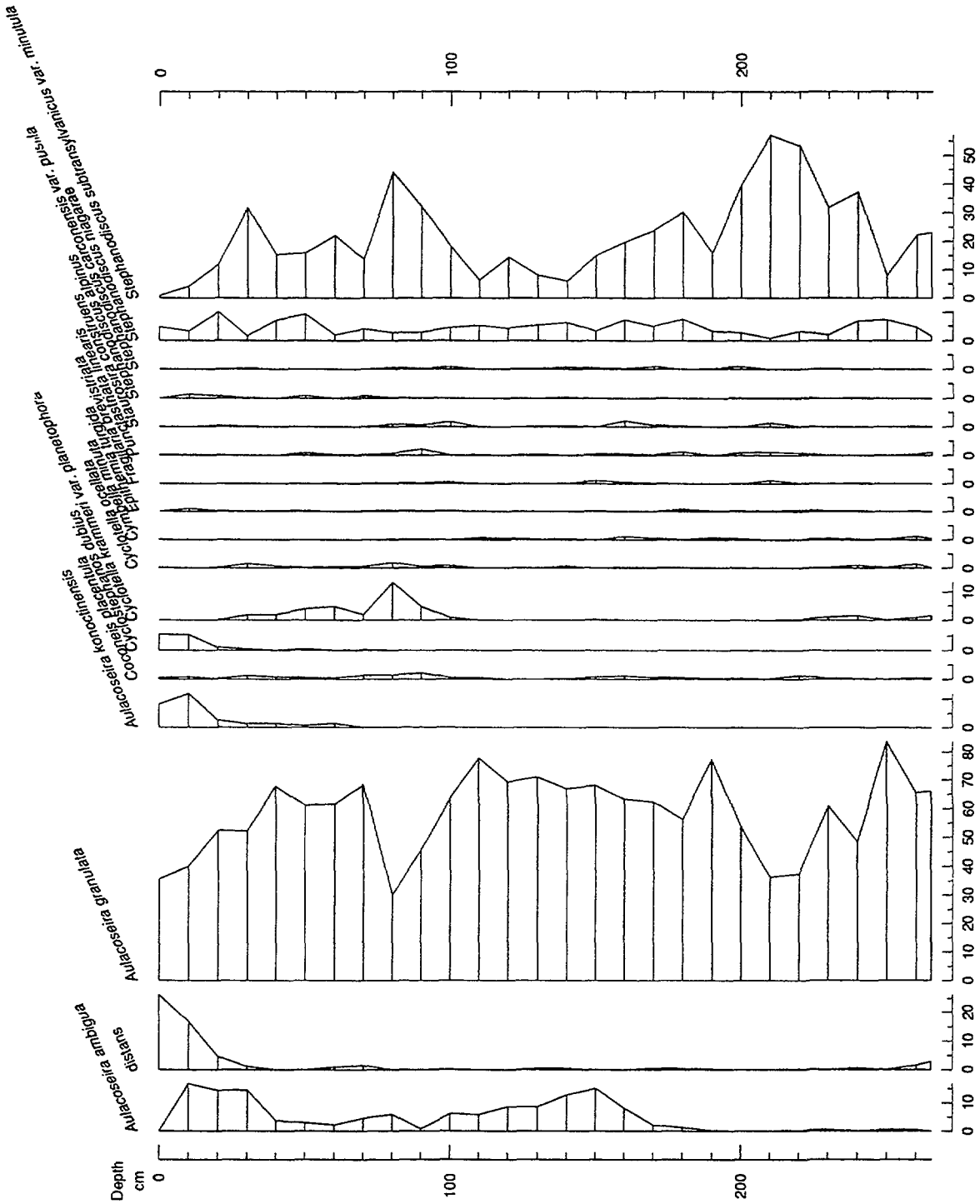

Figure 4: Diatom Biostratigraphy of LA- 05

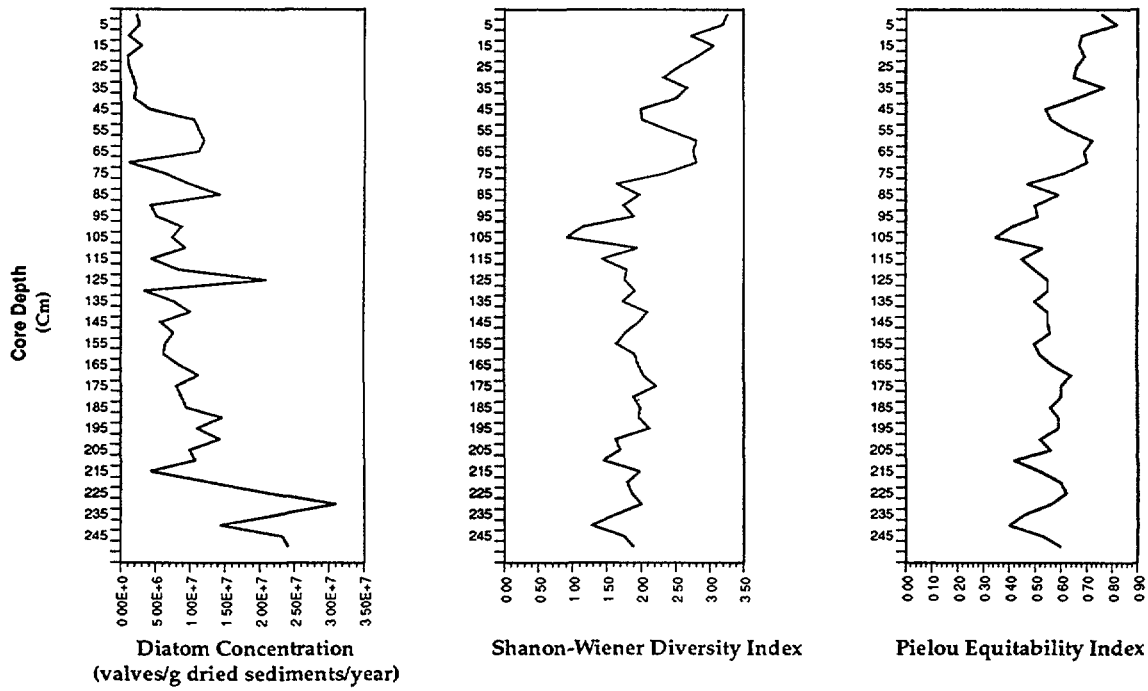

Figure 5: Diatom Concentration and Community Indices Charts for UA-02, Clear Lake CA.

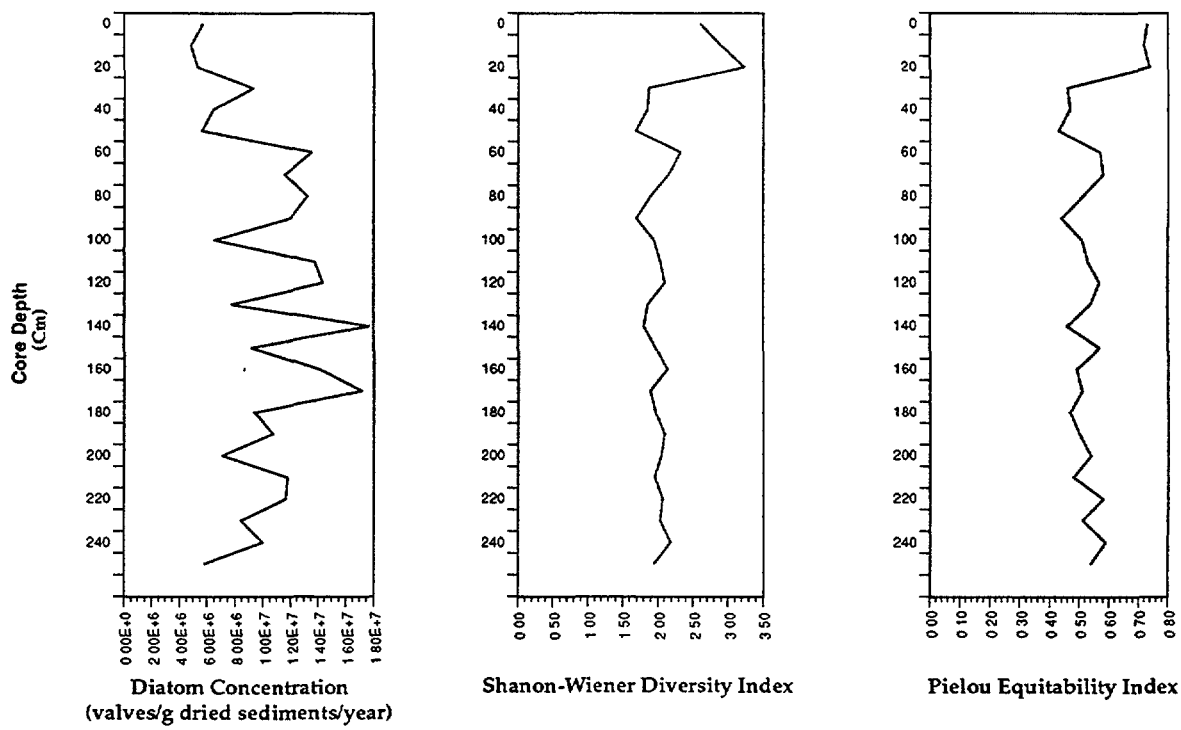

Figure 6: Diatom Concentration and Community Indices Charts for OA-03, Clear Lake CA.

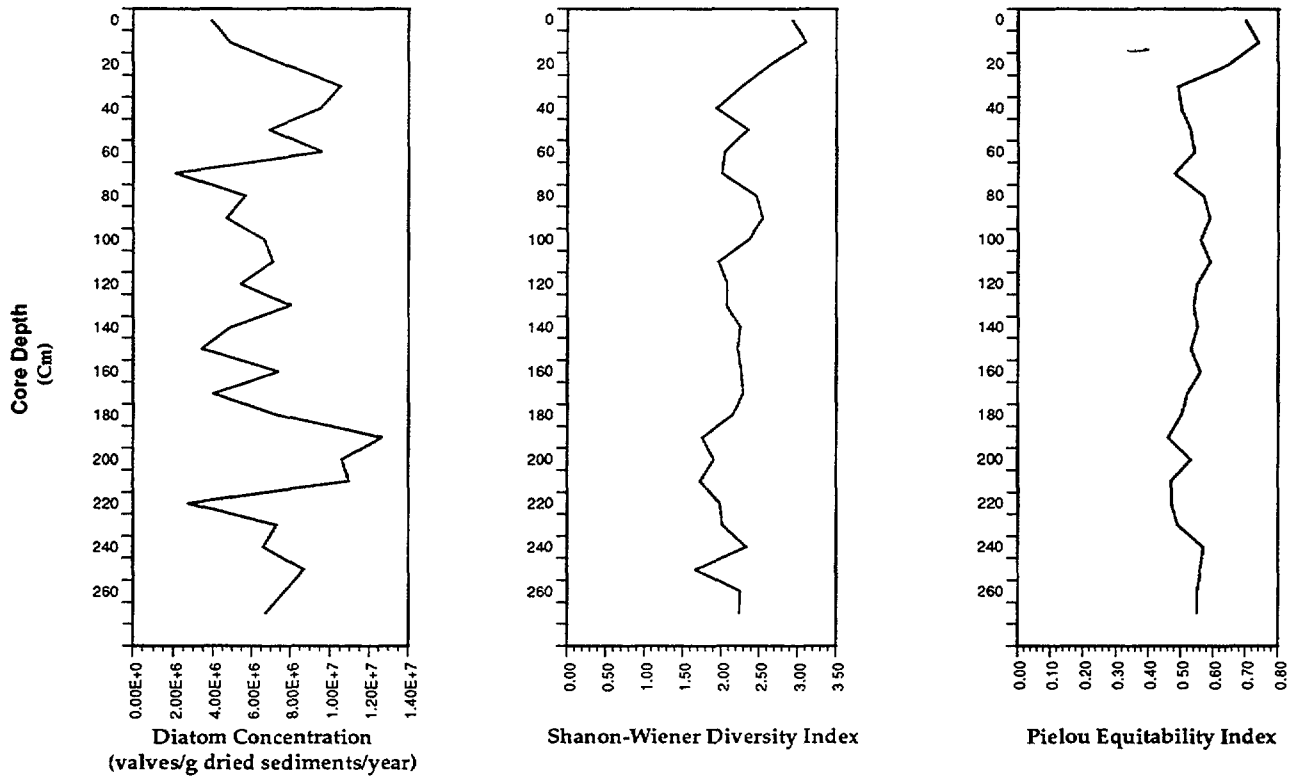

Figure 7: Sedimented Diatom Concnetration and Community Indices Charts for LA-05, Clear Lake CA

centric diatoms, which could be accomplished by using microcosms and culture experiments. Such undertakings would help the scientific community to correlate recent changes in diatom assemblages to anthropogenically derived stresses to limnic systems. Ultimately the management of aquatic ecosystems by federal and state agencies will benefit from these data. There is also a need to document diatom diversity in creeks flowing into Clear Lake. Probably dominantly pennate in nature these diatoms should be linked to the ones described in this study. Two alkaline closed basins limnic systems (Borax and Little Borax Lake) located in the close proximity of Clear Lake could be cored for diatom biostratigraphic assemblages and correlated to the Clear Lake datasets. The integration of Clear Lake diatom biostratigraphic profiles with the description of sedimented algal pigments in these cores could be integrated in a comprehensive analysis on the modern history of the Clear Lake watershed. Finally, one can envision to expand these types of limnic studies to the poorly known California Coast ranges both in ecological terms and biological taxonomy. A great help to agency and private watershed managers would be the creation of modern diatom training datasets [25] in tandem with water chemistry data. These datasets could be used in monitoring and understanding changes in aquatic ecosystems.

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OVERVIEW OF MIDDLE CREEK MARSH RESTORATION PROJECT

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Abstract: The Middle Creek Marsh Restoration Project consists of reconnecting Scotts and Middle Creek to the historic Robinson Lake wetland and floodplain areas by breaching the existing levee system to create inlets that direct flows into the historically flooded area. The County requested assistance from the U.S. Army Corps of Engineers to restore the historical wetland as an environmental restoration project. The first phase of the project, a Reconnaissance Study, is complete. Three alternative restoration projects have been selected for future study. Up to 1,218 acres of wetland habitat are proposed for restoration, including open water, seasonal wetlands, instream aquatic habitat, shaded aquatic habitat, and perennial wetlands. Additional upland habitat may be protected adjacent to the wetland and stream areas. The Project would significantly increase the habitat available for migratory waterfowl and numerous listed species. Significant water quality improvement is anticipated in Clear Lake due to the sediment and phosphorus removal capabilities of the Project. Diversion of flows through the wetland area are estimated to reduce the phosphorus load to Clear Lake by up to 28 percent. The resulting reduction in chlorophyll and organic carbon in the Lake are anticipated to be significant. Significant flood control benefits will be realized by removing up to 25 homes from the project area. Third party impacts will be primarily to existing infrastructure, primarily roads and utilities. Agricultural land will be removed from production and up to 25 homes may be relocated. Property tax revenues will decrease due to the property acquisition requirements. Improved water quality in Clear Lake will improve property values around the lake and increase tourism revenues. The Project will have an unknown impacts on vector control issues in the area.

Key Words: Clear Lake, wetland restoration, habitat restoration, water quality, flood management

INTRODUCTION

The Middle Creek Marsh Restoration Project (Project) is one step in the process of restoring damaged habitat and the water quality of the Clear Lake watershed. Reconnection of this large, previously reclaimed area, as a functional wetland is anticipated to have a significant affect on the watershed health and the water quality of Clear Lake.

The Project is located at the north end of Clear Lake in the area bounded by State Highway 20 and Rodman Slough, see Figure 1. Clear Lake is a large, natural, shallow, eutrophic lake. It is the headwaters of Cache Creek, a tributary of the Bay-Delta. The Scotts Creek and Middle Creek watersheds, which comprise approximately one half of the Clear Lake watershed, drain through Rodman Slough adjacent to the project area. These two watersheds provide 57 percent of the inflow and 71 percent of the phosphorus loading to Clear Lake. Twelve hundred eighteen acres of "reclaimed" wetlands are located in the Project area.

BACKGROUND AND PROJECT DESCRIPTION

In 1994, the EPA Clean Lakes Diagnostic/Feasibility Study for Clear Lake [1] was completed. Sediment nutrients are primarily responsible for the cultural eutrophication of Clear Lake and the resulting chronic blue-green algal blooms. The Clean Lakes Study identified a significant degradation in Clear Lake's water quality between 1920 and 1940. The Clean Lakes Study recommends numerous actions be taken to reduce the frequency and magnitude of the blue-green algal blooms. The County of Lake adopted an Implementation Plan on July 19, 1994 identifying the recommended actions and a time line for their implementation. The Plan is to improve the watershed health of the Clear Lake watershed and improve the quality of Clear Lake.

The District is currently implementing stream bank rehabilitation projects and actively encouraging the implementation of erosion control projects within the Clear Lake watershed. The District is cooperating with the U.S.D.A. Forest Service and the U.S.D.I. Bureau of Land Management in improving management of the watershed. Restoration of Middle Creek Marsh is one step in reducing the nuisance blue-green algal blooms in Clear Lake by addressing one of the largest sediment

sources within the watershed.

The Middle Creek Marsh (Robinson Lake) area was "reclaimed" in between 1918 and 1940 by constructing levees, creating a slough and reclaiming approximately 1,020 acres of lake bottom and shoreline wetlands for agricultural purposes [2]. In 1958, the U.S. Army Corps of Engineers (USACE) added to the levee system, reclaiming an additional 200 acres of shoreline wetlands. These projects resulted in the physical isolation of over 1,700 acres of marsh and floodplain from the largest tributaries of Clear Lake. Figures 2 and 3 show the 1916 and current configurationsof the project area. A recent sediment core collected by the University of California, Davis (UCD), shows an abrupt increase in sedimentation rates around 1927, corresponding to the beginning of the large scale reclamation of Robinson Lake [3].

In May 1996, the USACE began a Reconnaissance Study (Study) [4] for the environmental restoration of the historical wetland area. The Study was completed in May 1997. The Study evaluates six alternative projects, ranging from No Project to restoration of 1,218 acres of wetland habitat.

The Project consists of reconnecting Scotts and Middle Creek to the historic Robinson Lake wetland and floodplain areas by breaching the existing levee system to create inlets that direct flows into the historically flooded area. Diversion of flows through the wetland area are estimated to reduce the phosphorus load to Clear Lake by 40 percent. Based on Vollenweider's phosphorus mass-balance model, this would result in a 28 percent decrease in the phosphorus levels in Clear Lake, and a 33 percent decrease in chlorophyll levels.

The Study recommends three of the alternative projects be considered during the Feasibility Study. The alternates all include reconnecting the area adjacent to Clear Lake and Rodman Slough, with the primary difference being the northern end of the Project area. Table 1 compares the three alternatives showing the habitat, estimated water quality benefits and estimated cost of each alternative.

PROJECT BENEFITS

The Project would provide the following habitat benefits:

Restore up to 1,218 acres of the of the 7,520 acres of historic wetlands in the Clear Lake Basin that have either been lost or severely impacted. This is up to an 81 percent increase in the existing wetland habitat. Of the historic 9,300 acres of freshwater wetlands that existed in the Clear Lake Basin, approximately 7,520 acres (80 percent) have been lost or severely impacted. Restored habitat includes open water, seasonal wetlands, instream aquatic habitat, shaded aquatic habitat, and perennial wetlands. Additional upland habitat may be protected adjacent to the wetland and stream areas.

Provide a significant increase in habitat for fish and wildlife. This Project would greatly improve the bird nesting habitat and increase the available spawning habitat for native and non-native fish. The area is currently used extensively by migratory waterfowl.

Preserve the fish and wildlife resources and the cultural resources in the project area.

Several special-status wildlife species would benefit from the creation of wetland, open water, and riparian habitats in the expanded floodplain. Some species include the California red-legged frog, northwestern pond turtle, American white pelican, double-crested cormorant, western least bittern, osprey, white-tailed kite, bald eagle, northern harrier, Cooper's hawk, American peregrine falcon, California yellow warbler, yellow-breasted chat, tricolored blackbird, fringed myotis, long-eared myotis, long-legged myotis, pallid bat, and Townsend's western big-eared bat.

The Project will reduce the amount of sediment and nutrient inputs to Clear Lake producing the following water quality benefits:

Sediment is the primary nutrient source (97 percent of the total phosphorus load is sediment bound) contributing to the cultural eutrophication of Clear Lake;

Approximately 71 percent of the phosphorus entering Clear Lake is from Scotts and Middle Creeks. It has been estimated that the Project would remove up to 40 percent of phosphorus entering Clear Lake from Middle and Scotts Creeks;

Reduced phosphorus concentrations in Clear Lake would potentially reduce the chlorophyll concentrations by 33

percent. A corresponding reduction in total organic carbon would also be realized; and
Improved water quality in Clear Lake will reduce the amount of carbon being discharged to Cache Creek and the Bay-Delta.

Flood Control benefits include:

Reduce flood risk by removing structures at risk of severe flooding as a result of levee failure. The levees proposed for abandonment have settled up to three feet below design grade. The levees are at risk of overtopping during a 35 year flood event, unless emergency flood fight measures are implemented. The area was evacuated in 1983 and 1986, with evacuation imminent in 1995.
The District currently maintains the Middle Creek Flood Control Project. The Project would remove approximately three miles of substandard levees from the Flood Control Project. These levees are the most prone to failure during a major flood event. The Project would result in lower O&M and emergency response costs for the District and cooperating State and Federal agencies.

Third party benefits are:

Enhance recreation and tourism by improving the water quality in Clear Lake. In 1994, the U.S.D.A. Soil Conservation Service [5] estimated that $7 million in tourism is lost annually due to water quality issues in Clear Lake.
The Project will have an unknown, and possibly beneficial, impact on vector control issues in the area. Several hundred acres of rice fields and flood irrigated pasture will be replaced by a diverse wetland and riparian community. Natural predators may result in lower insect production in the area.

It is anticipated that the Project will impact the Clear Lake ecosystem quickly. The project area was active freshwater marsh less than 80 years ago and already has significant quantities of native wetland vegetation in the project area. The existing vegetation and the inherent soil properties will facilitate rapid re-establishment of the native habitat.

Water quality improvement in Clear Lake should be fully realized within 10 years, with some improvement almost immediately apparent. Improved regulation of instream gravel mining was implemented in 1980, with instream mining decreasing each year until 1991, when all instream mining ceased. The clarity of Clear Lake improved significantly in 1991, and has been the clearest in recent memory for the last three or four years. We anticipate the reduced phosphorus loading to Clear Lake after the Project is constructed to become apparent within a similar time frame.

THE NEXT STEP

The Reconnaissance Study evaluated six alternative plans for the project area and recommends three feasible alternative plans be evaluated in detail. The following phases remain in the Project:

Feasibility Study: The Study will evaluate each of the three alternative plans in detail to determine which is the most cost effective and cost efficient. Additional studies are necessary to fully evaluate impacts to water quality, wildlife, cultural resources, vectors, and evaluate additional sediment control measures. Environmental review as required by NEPA and CEQA will also be conducted in this phase.
Design: Detailed plans and specifications will be developed by the USACE for the alternative selected in the Feasibility Study.
Significant land acquisition will be required, including relocation of up to 25 residents. Land acquisition and relocation will be according to Federal requirements.
Construction: The Project will be constructed. The USACE will administer the construction contract, while contracting out the actual construction work.

Under current funding guidelines, approximately half of the cost for future phases of the project are the responsibility of the Project Sponsor, the District. These costs are beyond the District's ability to pay, without the development of partners. The District is currently developing partnerships to assist in completion of the Project. Partners may include the Robinson Rancheria, the California State Reclamation Board, the Environmental Protection Agency, CALFED, non-profit organizations, etc.

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Table 1 Comparision of Alternatives

Alternates	2 - Full Restoration	3 - Bloody Island to Clear Lake	4- Reclamation Road to Clear Lake
Increase in Habitat, acres			
Open Water	225	225	225
Perennial Wetland	303	303	193
Perennial Wetland/Riparian	538	457	224
Seasonal Wetland	152	21	24
Upland	480	164	147
Total (w/o upland)	1,218	1,006	666
Water Quality Improvement			
Phosphorus Removal, %	40	33	25
Chlorophyll Reduction, %	28	17	6
Estimated Cost, $K			
Feasibility Study	$900	$900	$900
Design Engineering	$684	$718	$554
Property Acquisition	$11,005	$7,300	$5,200
Construction Engineering	$485	$509	$391
Construction	$5,706	$5,910	$4,615
Total Cost, $K	**$18,780**	**$15,337**	**$11,660**

Figure 1: Project Location

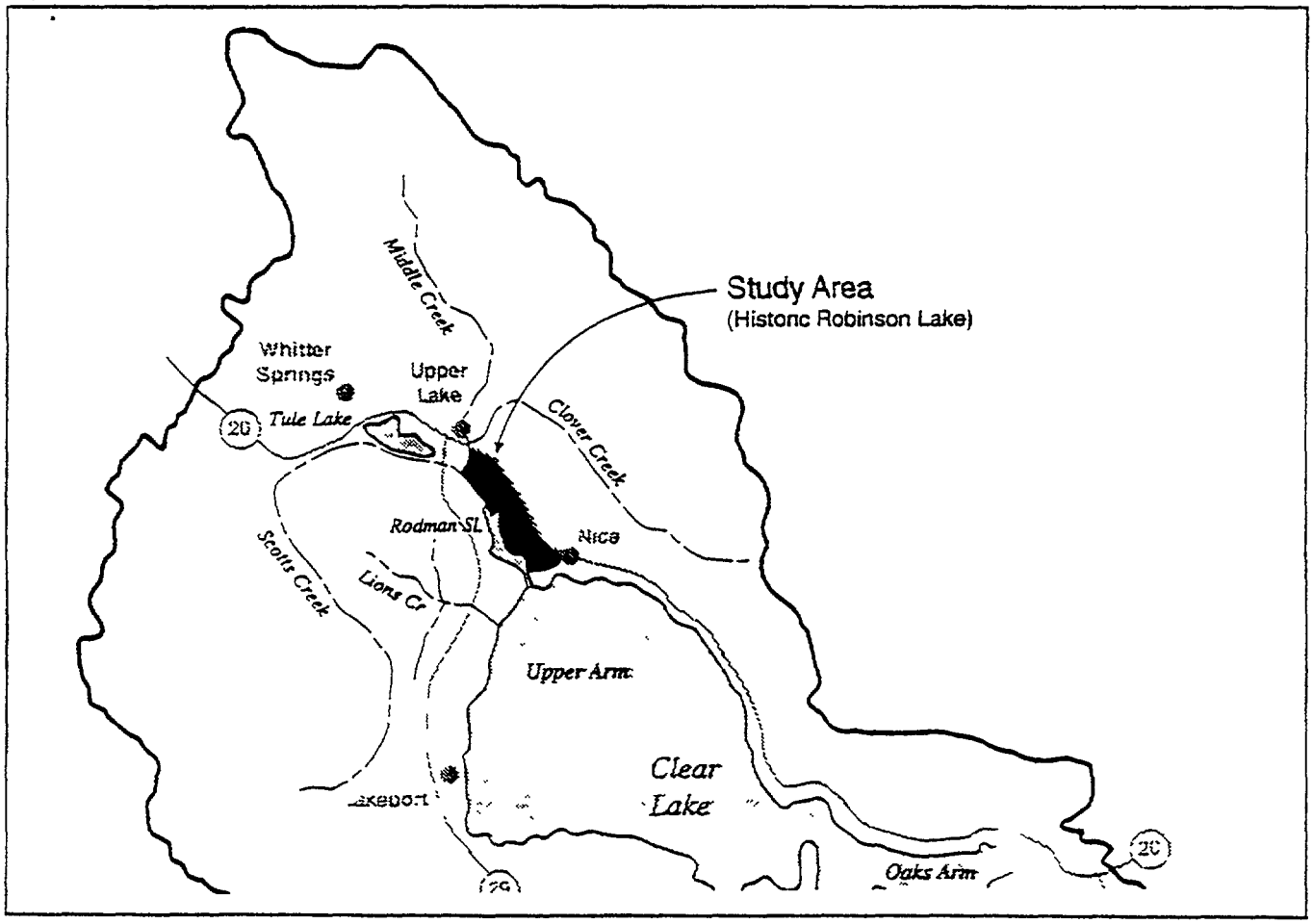


Figure 2 1916 Configuration

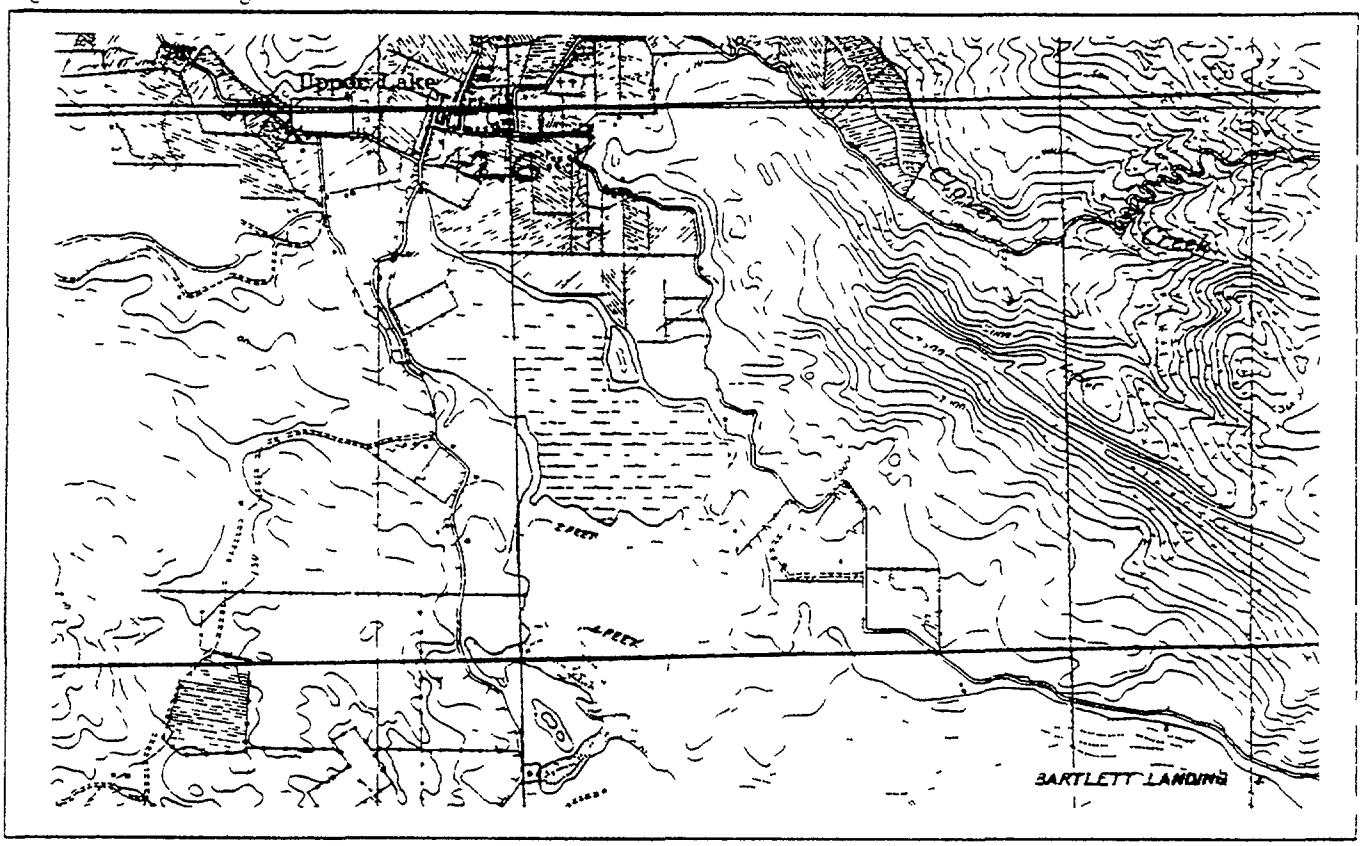
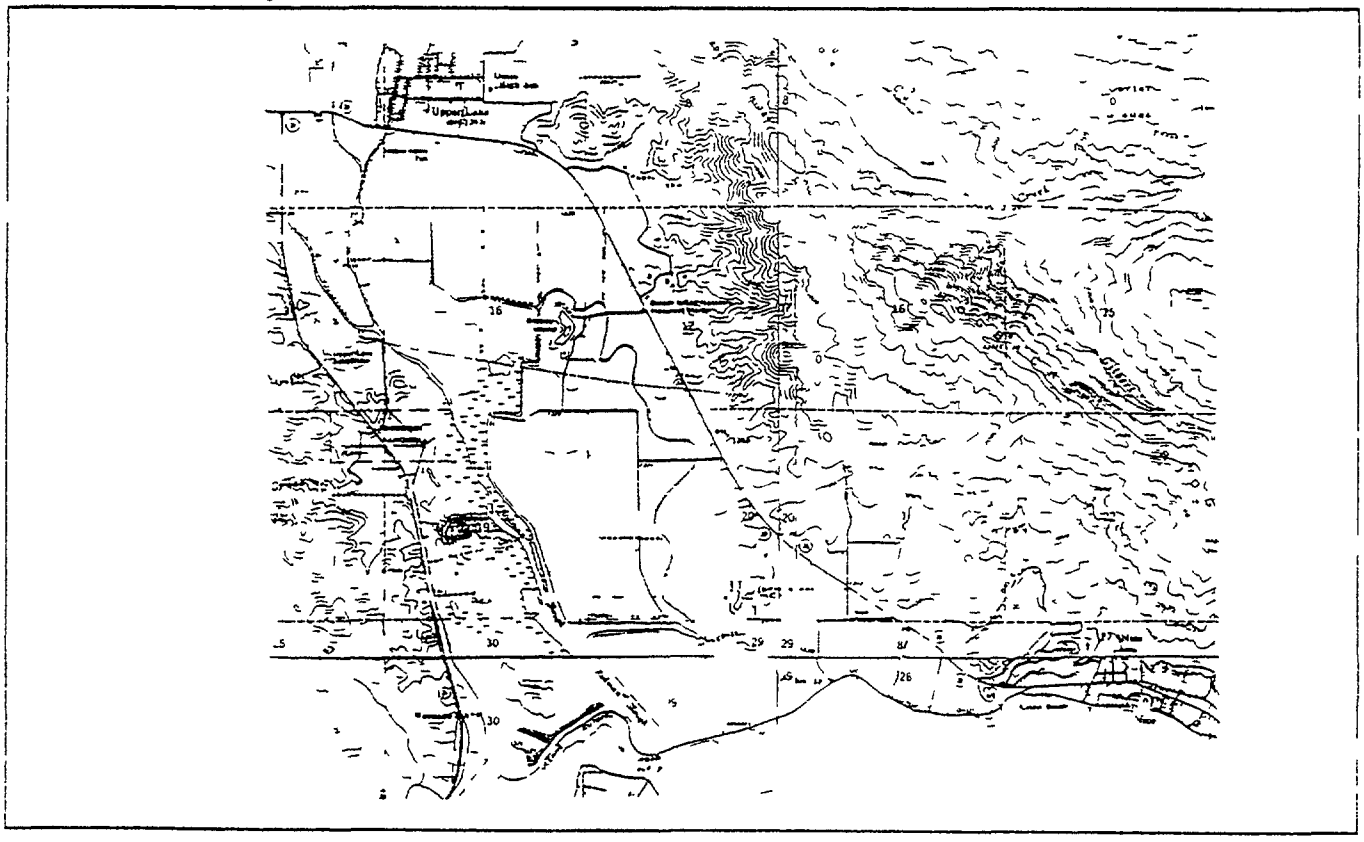


Figure 3 Current Configuration



Effects of Middle Creek Marsh Restoration on Clear Lake Water Quality

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Abstract - Water quality in Clear Lake is seriously impaired by high phytoplankton production. IFEgh phytoplankton production is primarily due to excessive phosphorus loading from diffuse sources throughout the Clear Lake catchment. The bulk (>90%) of this phosphorus is transported to the lake attached to silt and clay-sized sediment. The Middle Creek sub-catchment (i.e., the area drained by Scotts, Middle and Clover creeks) presently accounts for some 70% of the total phosphorus load to Clear Lake's Upper Arm. A previous study suggested that the restored marsh might retain up to 70% of this Middle Creek phosphorus load. By contrast, silt and clay retention coefficients calculated from HEC-6 modeling runs performed by the U.S. Army Corps of Engineers for this study suggested a phosphorus retention coefficient (Rp) of 15%. A discussion of the assumptions used to calculate these two widely differing estimates of Rp suggested that the actual retention coefficient of the restored marsh would more likely average about 40%. Calculations based on a phosphorus balance model for Upper Arm indicated that a 40% Rp for the marsh would reduce the average total phosphorus concentration (TP) of the lake's Upper Arm from its historical (I969-1991) average of 170 mg m-' to 120 mg rn-'. A global scale regression model for shallow (mean depth <5 m) temperate-latitude lakes (n=221) indicated that this roughly 30% reduction in TP would cause average chlorophyll concentration (a measure of phytoplankton production) in Upper Arm to decline from its historical (1969-1991) average of 92 mg m-' (calculated from estimates of mean biovolume and average chlorophyll content) to a new average value of 70 mg nf'. An appreciable reduction in the frequency and severity of algal blooms would accompany this reduction in average chlorophyll concentration. These results suggest that marsh restoration would lead to noticeable improvements in Clear Lake water quality, but that the goal of halving phosphorus loading to the lake will probably require additional conservation efforts in the Clear Lake watershed.

Introduction

Phosphorus loading from diffuse sources throughout the Clear Lake watershed during the wet season (November-April) is thought to be largely responsible for the excessively high levels of phytoplankton biomass that too often characterize the lake during the dry season (Richerson et al. 1994). The "algal blooms" that accompany this excessively high level of algal production seriously impair water quality by forming unsightly surface scums, fouling beaches, creating disagreeable odors, and contributing to low dissolved oxygen concentrations near the lake bed. Algal blooms may also promote processes that lead to bioaccumulation of mercury in the lake's invertebrate, fish and waterfowl communities (Suchanek et al. 1997).

A Clean Lakes Diagnostic/Feasibility Study for Clear Lake (Richerson et al. 1994) set a phosphorus reduction goal for Clear Lake that would require halving the existing loading rate of phosphorus from the watershed. Restoration of some 490 ha of marshland near the mouth of the 33,400-ha Middle Creek catchment has been proposed as one way to help accomplish this target (Richerson et al. 1994). This strategy assumes that reduced phosphorus loading will lead to reduced phytoplankton abundance, fewer and less intense algal blooms and thus better water quality.

The goal of this study was to quantitatively evaluate these assumptions by determining: (1) how much of the total phosphorus load currently entering the lake originates from the Middle Creek catchment; (2) how effectively the marsh is likely to function as a net sink for Middle Creek phosphorus loads; (3) how in-lake phosphorus concentration is likely to respond to a reduction in phosphorus loading from Middle Creek; and (4) how Clear Lake phytoplankton production is likely to respond to reduced phosphorus concentration. Only the Upper Arm was modeled because it accounts for 72% of lake surface area and 63% of its total volume and because much of the nuisance algae conditions in the lower arms are caused by algae produced in the Upper Arm and subsequently transported to the lower arms by wind or water currents. This approach simplified analytical methods and was considered adequate for the purposes of this study.

Methods and Assumptions

Phosphorus Load Reduction. Estimating the phosphorus load reduction potential of the restored marsh required first an estimate how much of the total phosphorus load to the Upper Arm is contributed by the Middle Creek watershed under existing conditions. This was done using historical hydrologic data for Upper Arm streams summarized in the Diagnostic

Report (Richerson et al. 1994) and flow-weighted mean total phosphorus concentration calculated from samples collected by Lake County between 1993 and 1996 in the Middle Creek drainage (i.e., Middle, Scotts and Clover creeks) and other streams draining into Upper Arm. The historical mean annual discharge for each group of streams was multiplied by the flow-weighted mean phosphorus concentration to estimate contribution to total mean annual phosphorus loading to Upper Arm.

The next step was to quantify an independent estimate of phosphorus retention coefficient (Rp) for the restored marsh. This estimate was calculated using output from a BEC-6 sediment transport model (Jones & Stokes Associates, Inc. 1997). The sediment transport model provided estimates of clay and silt transport at a number of transacts above within and downstream of the proposed marsh area under existing conditions and under fully restored (100-year flood plain) conditions. Sediment transport under three events were compared: one annual event after one year, ten annual events after 10 years and the 100-yr event after 10 years. For each event run, the amount of each sediment size class transported past the most downstream transect (Lucern cutoff bridge) under the fully restored condition was subtracted from the amount transported past this transect under existing conditions. The amount of silt or clay retained was multiplied by literature value estimates of average phosphorus content of each size class- specifically, 1.8 mgP g-1 for clay and 0.75 mgP g-' for silt (Syers et al. 1969) to estimate phosphorus retention. This coefficient was multiplied by Middle Creek's contribution to total phosphorus loading to Upper Arm to estimate loading reduction resulting from marsh restoration.

Prediction of In-lake Total Phosphorus Concentration. The phosphorus mass-balance model of Vollenweider (1969) was used to predict average phosphorus concentration in Upper Arm from predicted external phosphorus loading rates. This model is:

TP = Lp lz((j +P)]-,
(Equation 1)
where TP=mean total phosphorus concentration in the water column of Upper Arm (mg m-'), Lp=mean areal phosphorus loading to Upper Arm (mg m-' yr-'), z=mean depth of Upper Arm (m), cf=phosphorus sedimentation coefficient (yr-') and p=mean hydraulic flushing rate (yr-'). Mean depth, lake surface area and flushing rate were estimated from data presented in Table 3.1 of the Diagnostic Report (Richerson et al. 1994). The sedimentation coefficient was estimated from long term mean TP, annual phosphorus loading rate, mean depth and flushing rate by rearranging Equation I and solving for (i. This method indicated that sedimentation coefficient in Upper Arm averaged about 0.75 yr-'. This value was assumed constant for purposes of predicting TP under different estimates of retention coefficient for the restored marsh.

Prediction of Phytoplankton Abundance. Average seasonal (May-October) abundance of phytoplankton in Upper Arm was predicted from TP using the global scale model of Van Nieuwenhuyse (1993):

Log Chl = -2.53 + 3.29 Log TP - 0.60 (Log TP)2
(Equation 2)

where Chl = mean surface chlorophyll concentration. Chlorophyll concentration under existing conditions (I 969-199 1) was calculated from mean summer biovolume and average chlorophyll content data for blue green algae and non-blue green algae. Chlorophyll content values for each functional group were based on mean values tabulated in Reynolds (1984). Equation 2 was used because it was developed specifically for shallow temperate-latitude lakes.

Results and Discussion

Phosphorus Load Reduction. Hydrologic data for streams draining into Upper Arm indicated that project area streams (Clover, Middle and Scotts creeks) contributed on average about 57% of total inflow into Upper Arm (Table 1). Water chemistry data collected between 1992 and 1996 by Lake County indicated that flow-weighted mean total phosphorus concentration was substantially higher in project area streams (644 Mg M-3) than in other streams draining into Upper Arm (342 mg ni-'). These data indicated that the total P load to Upper Arm averages about 169 metric tons per year. The contribution of Middle Creek to this total phosphorus load was 71%; considerably higher than its contribution to hydrologic loading. This difference indicates that erosion is more intense among project area streams than among the other streams draining into the Upper Arm of Clear Lake.

Calculations based on the HEC-6 sediment transport model indicated that marsh restoration would result in the retention of about 40% of incoming silt load and 4% of incoming clay load for the annual flood. The 100-yr flood event analysis indicated similar silt retention, but considerably less clay retention (Table 2). Because most of the loading to lakes is

accomplished by "average" events over the long run (Leopold et al. 1964), the 4% and 40% values were used for purposes of predicting phosphorus loading to Upper Arm.

The retention of silt was greater than for clay-sized particles because clay-sized particles are smaller and lighter and thus settle out less rapidly than silt-sized particles. Clay-sized particles, however, have a higher average phosphorus content in part because of their greater surface area to volume ratio. Taking this differential phosphorus content into account indicated that the phosphorus retention coefficient (@) for the marsh would average about 15% (Table 2).

The 15% retention coefficient estimated from the sediment transport analysis is considerably lower than the 70% @ suggested by storm flow samples collected up and downstream of Tule Lake during the Diagnostic Study (Richerson et al. 1994). The 15% value is probably an underestimate given that the sediment transport model did not include the effects of channel complexity and vegetation in the restored marsh. Both factors would greatly increase resistance to flow and thus hydraulic residence time during high flow events. Submerged vegetation and leaf litter also tend to accumulate clay particles on their surfaces. These mechanisms would allow a greater than predicted amount of silt- and clay-sized particles to be retained in the marsh, thus reducing the amount of particulate inorganic P reaching the lake.

By contrast, the 70% estimate of @ based on storm flow samples collected during a single event probably overestimates the true value. The accuracy of the 70% value, even as a measure of the short term Rp for Tule Lake marsh, is questionable because accurate estimates of flow downstream from the marsh were not available (Richerson et al. 1994). More over, even if retention during a single event was 70%, the long term annual mean retention coefficient would almost certainly be substantially lower. This hypothesis is consistent with results presented in Table 5.15 of the Diagnostic Study report indicating that the average annual P retention coefficient for the entire Clear Lake system is 77% (range, 53-96%). It seems unlikely that a single marsh complex could retain the same percentage of incoming phosphorus as the entire Clear Lake system. Consequently, the 15% and 70% values were used in this study as upper and lower limits and a 40% value was selected as a plausible long term average value for subsequent analyses.

Prediction of Total Phosphorus Concentration. The long term average external load estimate of 169 metric tons per year (Table 1) indicated that the areal P loading rate to Upper Arm (i.e., loading per unit surface area of Upper Arm) averaged 1333 Mg m-2 yr-1. This areal loading rate decreased as the phosphorus retention coefficient assumed to characterize the restored marsh increased (Table 3). As areal loading decreased from its present value of 1333 Mg M-2 yr-' to the 671 Mg M-2 yr-' value expected with 70% retention, the average phosphorus concentration predicted by Equation I for Upper Arm declined from its historical annual mean value of 170 Mg M-3 to 85 Mg M-3 (Table 3). The expected concentration assuming a 40% long term average value of Rp was 120 mg m-'.

Prediction of Phytoplankton Abundance. Routine monitoring data from the California Department of Water Resources for the period 1969-1991 indicated that seasonal (May-October) mean biovolume of blue green and non-blue green algae in Upper Lake was highly variable. The percentage of the total biovolume attributed to blue green algae also ranged widely; from 4% to 98% of total algal biovolume. Blue green biomass has a lower chlorophyll content than green algae or other forms of algal biomass. Taking this differential chlorophyll content into account resulted in estimates of seasonal mean chlorophyll concentration that ranged between 13 and 356 mg m-', averaging some 92 Mg M-3 over the entire 22 years of available records (Table 4). These estimates provided a basis for comparison with other relatively shallow temperate latitude lakes elsewhere in the world.

Observed values of TP and estimated Chl for the Upper Arm of Clear Lake were compared to the global TP-Chl relationship for shallow lakes (Equation 2). The Clear Lake values were generally scattered along either side of the ascending limb of the model curve. This comparison indicated that Upper Arm phytoplankton abundance is probably still phosphorus limited (i.e., TP levels in Upper Arm are within a range in which Chl would be likely to respond to changes in TP). The mean and 95% confidence interval (illustrated by the crossed bars) for chlorophyll concentration under existing conditions (Figure 2) agreed well with the global model prediction. Thus, this model could be used to predict what chlorophyll concentration would be under the three retention coefficients estimated for the restored marsh.

According to the global model, the modest reduction in TP for Upper Lake expected with a 15% retention coefficient for Middle Creek marsh would result in a similarly modest reduction in phytoplankton abundance (Figure 2). By contrast, with a 70% retention coefficient for the restored marsh, the global model indicated that chlorophyll would decline from its historical average of 92 mg ff' to about 51 mg rri-3. If the marsh retains 40% of incoming P loading, the decrease in chlorophyll concentration would be less dramatic (from 92 to 70 Mg M-3), but still highly significant (Figure 2). A number of studies have shown a strong correlation between average chlorophyll concentration and the frequency and severity of

algal blooms in lakes (Rast and Lee 1978; Walker 1986). Thus, this analysis of conditions in Clear Lake indicated that even under the most plausible value of 40% retention, a substantial reduction in the frequency and severity of algal blooms in Upper Arm could be expected. This reduction would alleviate water quality problems associated with blooms in Upper Arm and reduce the amount of biomass blown in or otherwise transported from Upper Arm to Lower and Oaks arms.

Recommendations for Future Studies

The model used to predict phosphorus concentration in Upper Arm is quite sensitive to variation in the proportion of external loading contributed by project area streams and by the assumptions made about the phosphorus retention coefficient of the restored marsh. Neither estimate is very well quantified using existing data.

- To improve P loading estimates to the lake, it is recommended that all major inlet streams to Upper Arm be gaged and equipped with remote water sampling devices programmed to collect samples during at least three stages along the flood hydrograph. Experience with other
lakes

indicates that it is only with this kind of monitoring that reasonably accurate estimates of P loading can be achieved; especially in systems with highly seasonal and flashy hydrologic regimes, such as Clear Lake's inlet streams.

Two recommendations are offered for improving estimates of the marsh's retention coefficient. The HEC-6 model should be refined to include the effects of channel modifications, baffling, vegetation and other site-specific properties of the project area. These refinements should provide a more realistic estimate of retention coefficient. Output from this improved BEC-6 model should be compared with empirical estimates of retention coefficient for the Tule Lake marsh system. This system should be monitored up and downstream during floods and during the draw down season. For this purpose, it would be necessary to reactivate the D@ gage downstream of Tule Lake or to install a new temporary gage. Either gage would have to be rated against ultrasonic velocity meter (UVM) estimates of outflow or against predicted outflow values based on measured inflow, lake level and other factors using a numerical hydrodynamic model. This focused monitoring effort could be incorporated into the more routine monitoring effort required to accurately quantify phosphorus loading to Upper Arm.

Another source of error in this preliminary analysis is the estimate of chlorophyll concentration derived from biovolume estimates. Biovolume estimates are very time consuming and expensive to obtain, whereas chlorophyll concentration is relatively cheap and easy to measure. It is recommended that chlorophyll analysis be added to the routine monitoring program for Clear Lake or that a short term study be conducted to quantify a chlorophyll-biovolume relationship specific to Clear Lake rather than relying, as in this analysis, on point estimates extracted from the literature.

Finally, even though Middle Creek may account for some 70% of phosphorus loading to Upper Arm under existing conditions, restoration of the marsh may retain only 40% of this loading. Consequently, the total loading to Upper Arm would be reduced by 28%; considerably less than the 50% reduction goal defined in the Diagnostic Report. Achieving this goal will require a combination of improved watershed management and perhaps additional marsh restoration elsewhere in the Middle Creek drainage or in other inlet stream systems.

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Table 1. Estimated phosphorus loads to Upper Arm of Clear Lake from Middle Creek catchment and from other catchments draining into Upper Arm.

Contributing Area	Catchment Area (km^2)	Mean Annual Discharge[1] (cfs)	% of Total Inflow to Upper Arm (%)	Total P Concentration[2] ($mg\ m^{-3}$)	Annual P Load ($MT\ yr^{-1}$)	% of Total Load (%)
Middle Creek	334	210	57	644	121	71
Others	900	159	43	342	49	29
Total	1234	369	100	---	169	100

[1] Tom Smythe, personal communication

[2] Flow-weighted mean

Table 2. Estimate of phosphorus retention coefficient of restored marsh based on output from sediment transport model (HEC-6).[1]

Status	Model Scenario	Simulated Flow (cfs)	Simulated Clay Load[2] (MT day^{-1})	Simulated Silt Load[2] (MT day^{-1})	% Clay Retained[3] (%)	% Silt Retained[3] (%)	Phosphorus Retained[4] (%)
Without marsh	One annual event	4660	927	815	---	---	---
With marsh		4660	888	488	4.2	40.1	14.8
Without marsh	Ten annual events	4660	927	817	---	---	---
With marsh		4660	889	492	4.1	39.8	14.6
Without marsh	One 100-yr event	19180	11653	16300	---	---	---
With marsh		19180	11480	9848	1.5	39.6	12.7

[1] Source: Don Twiss, U.S. Army Corps of Engineers, Sacramento, CA

[2] Load at downstream end of Rodman Slough (mouth of Middle Creek drainage)

[3] Difference between load under existing conditions and under restored marsh conditions divided by load under existing conditions

[4] Assumes phosphorus content of 1.8 mg per gram of clay and 0 75 mg per gram of silt (Syers et al. 1969)

Table 3. Predicted mean total phosphorus concentration in Upper Arm of Clear Lake for 15%, 40%, and 70% phosphorus retention coefficient values for the restored Middle Creek marsh

Phosphorus Retention Coefficient for Restored Marsh (%)	Predicted Total P Loading Rate to Upper Arm ($mg\ m^{-2}\ yr^{-1}$)	Predicted Total P Concentration in Upper Arm[1] ($mg\ m^{-3}$)	Percentage Reduction from Historical Mean ($170\ mg\ m^{-3}$) (%)
0	1333	170	0
15	1191	150	12
40	955	120	30
70	671	85	50

[1] from Equation 1

Table 4. Seasonal (May-October) mean biovolume and estimated chlorophyll concentration associated with blue green and non-blue green algae in Upper Arm, 1969-1991[1].

Year	Total Biovolume ($mm^3 L^{-1}$)	%Blue green (%)	Blue green biovolume ($mm^3 L^{-1}$)	Non-blue green biovolume ($mm^3 L^{-1}$)	Blue green chlorophyll ($mg\ m^{-3}$)	Non-blue green chlorophyll ($mg\ m^{-3}$)	Total chlorophyll ($mg\ m^{-3}$)
1969	5.26	73.3	3.86	1.41	21.5	15.4	37
1970	9.97	79.2	7.90	2.07	44.1	22.8	67
1971	8.57	49	4.20	4.37	23.4	47.9	71
1972	15.11	98.4	14.87	0.24	83.0	2.7	86
1973	8.16	67.1	5.48	2.69	30.6	29.5	60
1974	6.64	97.1	6.44	0.19	36.0	2.1	38
1975	7.13	92	6.56	0.57	36.6	6.3	43
1976	15.75	96.4	15.18	0.57	84.7	6.2	91
1977	12.20	98.3	11.99	0.21	66.9	2.3	69
1978	5.44	81.7	4.44	1.00	24.8	10.9	36
1979	7.19	97.5	7.01	0.18	39.1	2.0	41
1980[a]							
1981	17.07	46.4	7.92	9.15	44.2	100.3	145
1982	7.70	90.7	6.99	0.72	39.0	7.9	47
1983	4.33	36.9	1.60	2.73	8.9	30.0	39
1984	40.51	83.3	33.75	6.77	188.3	74.2	263
1985	21.45	97.9	21.00	0.45	117.2	4.9	122
1986	3.92	56.8	2.23	1.69	12.4	18.6	31
1987	21.47	20.7	4.44	17.02	24.8	186.7	212
1988	1.19	3.9	0.05	1.15	0.3	12.6	13
1989	8.89	15.2	1.35	7.54	7.5	82.7	90
1990	38.53	32.2	12.41	26.13	69.2	286.6	356
1991	8.56	33.5	2.87	5.69	16.0	62.4	78

[1] Biovolume data are from the California Department of Water Resources routine monitoring program (data provided by Pete Richerson, UC-Davis). Chlorophyll content is assumed to average 5.58 ug mm^{-3} for blue green algae and 10.97 ug mm^{-3} for non-blue greens (Table 7 in Reynolds 1984).

[a] No data

Figure 1. Comparison of TP-Chl values for Upper Arm (1969-1991, see Table 4) with global TP-Chl relationship for shallow lakes

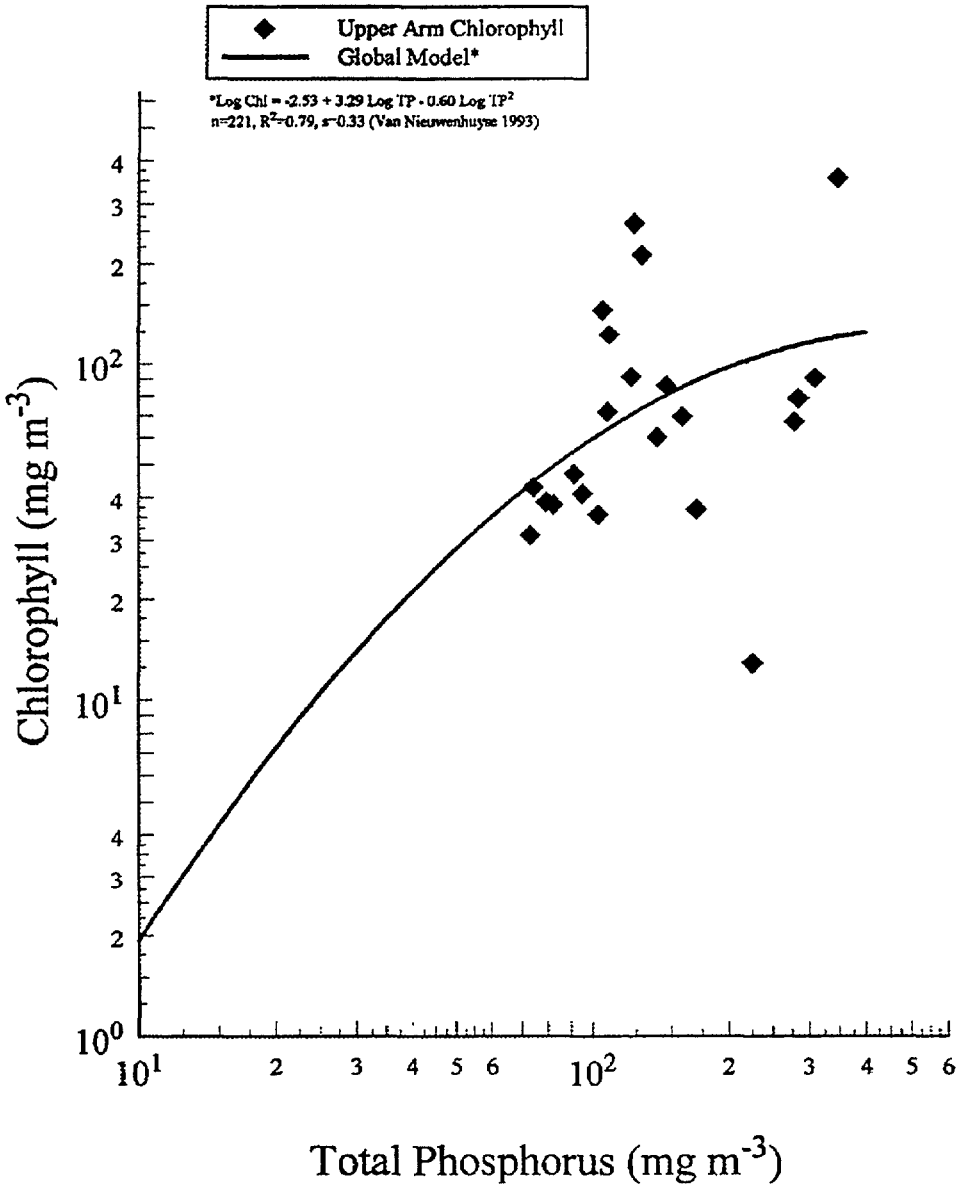
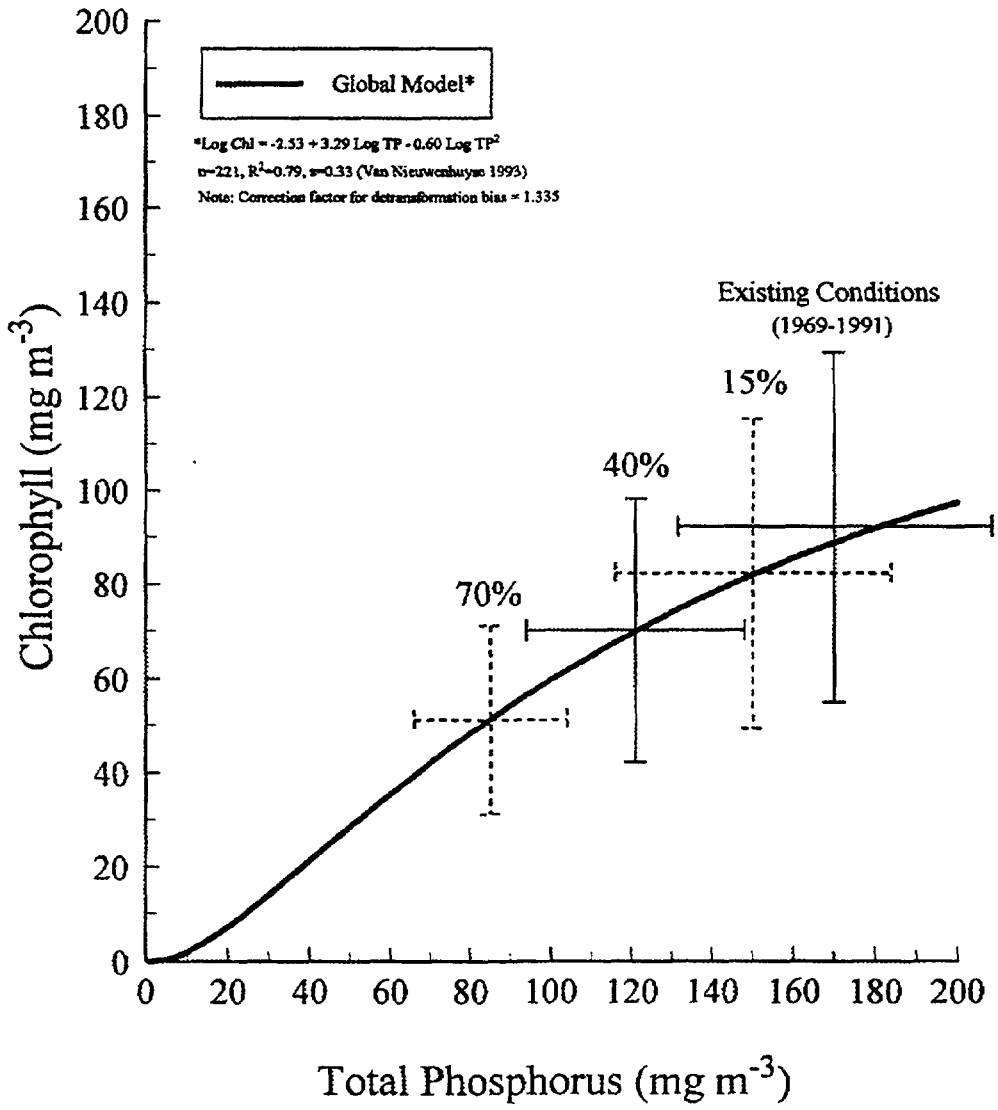


Figure 2. Variation in predicted seasonal mean (May-October) chlorophyll concentration in Upper Lake given restored marsh phosphorus retention coefficient of 15, 40, and 70%.



Created Wetlands for Wildlife Habitat and Reduction of Nutrient Inflows to Clear Lake

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Key Words: wetlands, mining, sediment, nutrient, Clear Lake

Transport of sediment and nutrients into Clear Lake have been identified as a primary cause of blue-green algal blooms. The source of much of this sediment has been traced to in-channel gravel mining within the Clear Lake Basin. During the past twelve years Lake County has implemented a policy of shifting mining out of creek channels and into terraces and flood plains to create wetlands and ponds for wildlife use. Several successful projects have been completed over the past decade. During this period, the three major creek systems where this approach has been used have experienced a significant recovery and stabilization, and numerous wetlands and ponds have been created. Lake County is currently designing wetland treatment ponds for removing nutrients from secondarily treated effluent. It is hoped that this new program will reduce the inflow of yet another potential source of nutrients into the lake.