MERCURY RESIDUES AND PRODUCTIVITY IN OSPREY AND GREBES FROM A MINE-DOMINATED ECOSYSTEM

Daniel W. Anderson,[1,3] Thomas H. Suchanek,[1,4] Collin A. Eagles-Smith,[1,5] and Thomas M. Cahill, Jr.[1,2]

[1]*Department of Wildlife, Fish, and Conservation Biology, University of California, Davis, California 95616 USA*
[2]*Department of Chemistry, University of California, Davis, California 95616 USA*

Abstract. Mercury (Hg) and reproduction and status of Western and Clark's Grebes (*Aechmophorus* sp.) and Osprey (*Pandion haliaetus*) were studied from 1992 through 2001 and then less intensely through 2006 at Clear Lake, California, USA. Remediation to reduce Hg loading from the Sulphur Bank Mercury Mine was initiated in 1992. Mercury in grebe feathers declined monotonically from ~23 mg/kg dry mass (DM) in 1967–1969 to 1 mg/kg in 2003, but then increased to 7 mg/kg in 2004–2006. Mercury in Osprey feathers varied similarly, with mean values of 20 mg/kg DM in 1992, declining to a low of 2 mg/kg in 1998, but increasing to 23 mg/kg in 2003, and 12 mg/kg in 2006. Mercury in Osprey feathers at our reference site (Eagle Lake, California) remained low (1–8 ppm) throughout the entire period, 1992–2003. Grebe productivity at Clear Lake improved from ~0.1 to 0.5 fledged young per adult during the latter part of the study when human disturbance was prevented. At that period in time, improved productivity did not differ from our reference site at Eagle Lake. Human disturbance, however, as a co-factor made it impossible to evaluate statistically subtle Hg effects on grebe productivity at Clear Lake. Osprey reproduced sufficiently to maintain increasing breeding numbers from 1992 to 2006. Mercury in Clear Lake water, sediments, invertebrates, and fish did not decline from 1992 to 2003, but a shift in trophic structure induced by an introduced planktivorous fish species may have caused significant alterations in Hg concentrations in several species of prey fishes that may have produced concomitant changes in Osprey and grebe Hg exposure. The temporary declines observed in grebe and Osprey feather residues in the late 1990s, with coincidental improvements in reproductive performance, however, could not be attributed to remediation at the mine site.

Key words: Aechmophorus sp.; *Clark's Grebe; Clear Lake, California, USA; mercury; mine site remediation; Osprey;* Pandion haliaetus; *populations; productivity; residues; Sulphur Bank Mercury Mine; Western Grebe.*

INTRODUCTION

This study is one component of an ecosystem-level project that attempted to trace the origin and pathways of mercury (Hg) from the ore body at an abandoned Hg mine through the abiotic (sediment and water) compartments to lower trophic level species (benthic invertebrates and plankton) and ultimately to higher trophic level species (e.g., fish and birds). We utilize those and other data to evaluate the fate and potential effects of mine-derived Hg on two avian groups: Western and Clark's Grebes (*Aechmophorus* sp.; hereafter referred to as "grebes") and Osprey (*Pandion haliaetus*). Our objectives were: (1) to evaluate potential changes in Hg exposure and reproductive rates in these two high-trophic-level avian piscivores from 1992 to 2001 (with less intense monitoring in subsequent years through 2006) and (2) to evaluate whether changes observed in these avian piscivores might relate to reported changes in other trophic levels and to remediation activities initiated at the mine in 1992.

Mercury affects various reproductive parameters in birds (e.g., Heinz 1979, Eisler 1987, Wren et al. 1995, Thompson 1996, Poole et al. 2002, Heinz et al. 2006). In field studies, Furness et al. (1989) and Newton and Galbraith (1991) related trends of decreasing reproductive success to elevated Hg levels in feathers of adult Golden Eagles (*Aquila chrysaetos*), but also urged caution due to other confounding variables. Mercury residues from ingested fish are transferred to bird feathers as a depuration mechanism, and most feather residues are in the organic form, methylmercury (MeHg) (Thompson and Furness 1989*a*).

The proximate cause of elevated Hg residues in biota in Clear Lake, California, USA, has been mostly point source erosion or acid mine drainage input from the Sulphur Bank Mercury Mine Superfund Site (Suchanek et al. 1998, 2000*a, b*, 2008*b, d*; Fig. 1). In 1992, the U.S.
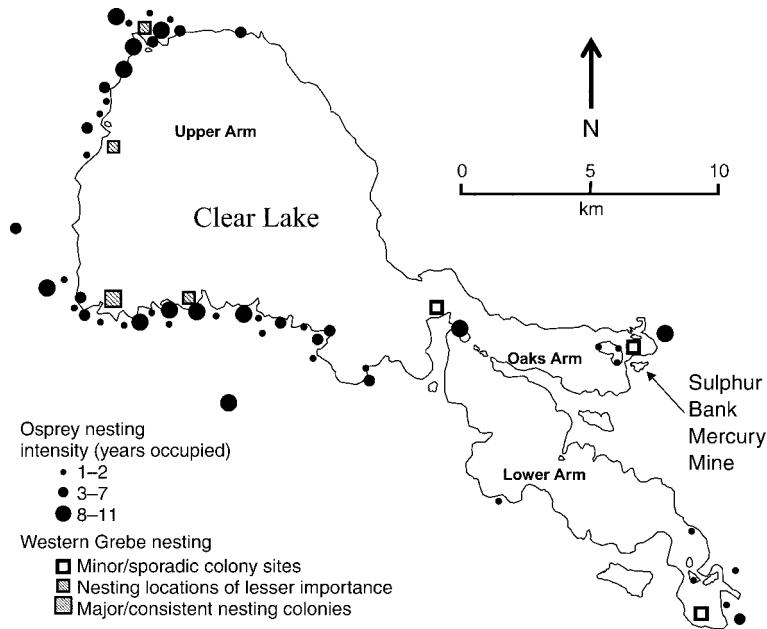
FIG. 1. Map of Clear Lake, California, USA, showing the generalized distribution and intensity of Osprey (*Pandion haliaetus*) and *Aechmophorus* grebe nesting from 1991 through 2001. The distribution of Osprey nesting closely coincides with areas of lesser shoreline development or tall, inaccessible objects that act as suitable nest platforms. Large blank areas on the map or areas with low Osprey density are areas on the lake with the most intensive development, except Anderson Marsh (southeast outlet, called Cache Creek). *Aechmophorus* grebe nesting is generally found in the less-disturbed areas of the lake, but only in areas with dense bulrush (*Scirpus* sp.) beds or rarely on stable, shallow-water cyanobacteria mats.

Environmental Protection Agency (U.S. EPA) conducted remediation activities at the mine: (1) to seal an overflow channel from an open-pit mine site excavation (Herman Pit) and (2) to reduce erosion of Hg-laden sediments from steeply sloped waste rock piles along the shoreline of Clear Lake (Suchanek et al. 2000b, 2008d).

In this quasi-experiment (see Campbell and Stanley 1966), we predicted that long-term declines in Hg residues in abiotic components of the ecosystem resulting from the remediation would translate into declining feather residues in both species and concomitant improvements in reproductive performance and population status of both Clear Lake Osprey and grebes (using management activities as a hypothesis-testing criterion [see Anderson 1998] because we had no authorization to artificially manipulate any experimental conditions).

MATERIALS AND METHODS

To minimize investigator effects, our studies were conducted without extensive tissue collecting and no Osprey soft tissues were taken. We closely monitored nesting Osprey and grebes at Clear Lake, along with their annual nest success, from 1992 through 2001. Osprey nests were surveyed initially in 1991, but detailed observations on behavior and performance started in 1992.

For Osprey, standardized population measurements (Postupalsky 1974, Poole 1989b) were determined each

year: number of nest attempts (NA, total active nests with an occupying pair for three or more weeks of any breeding season), number of successful nests (SN, those of the NA that fledged at least one young), and number of young produced per active nest (YY/NA). We considered a nestling Osprey as "fledged" when it first departed the nest (as defined by Koford et al. 1994). Osprey data were collected during four to eight surveys per season by visiting each nest vicinity and determining each nest's progress from early in the breeding season (April–May) through successful fledging of all young (June–July) or until the nest failed. Observations were conducted from distant vantage points using binoculars and spotting scopes to avoid investigator disturbance. Molted feathers (remiges and rectrices, sometimes other large feathers) were collected from near Osprey nest sites during or after nesting.

Grebe reproductive success is generally expressed on the basis of large, developed young to adult ratios at the end of or late in each breeding season, but prior to late summer/fall migration (Elbert 1996). For grebes, our open-water surveys were conducted two to five times per season to determine species compositions, brood sizes and their ages, and final adult to young ratios. Nesting populations of grebes were estimated by counting portions of each colony and projecting those densities to the entire colony and in some instances through low-level aerial photography. Progress of nesting colonies of grebes at Clear Lake, plus a comparison site in northern

California, Eagle Lake (Lassen County), was monitored throughout each breeding season in conjunction with our Osprey surveys and later in the season, for grebes only (but less frequently at Eagle Lake).

We expressed grebe productivity as the number of young per adult (YY/AD) during the final phases of the breeding season (August–September) (when maximum or near-maximum numbers of large young were present and still distinguishable from adults). Earlier surveys accounted for early-hatch young and seasonal progress. Double-brooding (which could invalidate nest success data obtained in this manner [Thompson et al. 2001]) is very rare in large grebes. For example, only very low frequencies of re-nesting or double-brooding have been reported for Red-necked Grebes (*Podiceps grisigena*; Kloskowski 2001) and *Aechmophorus* grebes (Storer and Nuechterlein 1992). Elbert (1996) and Elbert and Anderson (1998) provide further details on our survey methods and sampling strategies. Our avian sampling methodology was intended first to be internally consistent but also to be consistent with other published studies for comparative purposes. Reproductive performance is one key parameter in evaluating population-level effects of contaminants on populations (Vouk and Sheehan 1983, Kendall and Latcher 1994, and others) although it is sometimes difficult to interpret, as well as subject to a wide range of environmental variables. For example, Osprey are wide-ranging in North America and exhibit geographic variation in demography (Judge 1983, Poole 1989*b*, and others), requiring that specific comparisons be made with similar study areas and studies.

Pre- ("early") and post- ("late") remediation periods at the Sulphur Bank Mercury Mine were based on the history of remediation activities (with some time allowed for an expected change to occur). These were defined a priori as follows: early, 1992–1995; late, 1996–2001. Although most remediation was completed in fall 1992 (Suchanek et al. 2000*b*), we allowed three years for potential recovery (1993–1995). This separation was arbitrary and made prior to data inspection and analysis; in retrospect, 1995 was more similar to post-remediation or later years. After 2001, intermittent and opportunistic feather sampling was also conducted through 2006 (and is continuing) to further document long-term variation in Hg residues, not anticipated in our first 10 years of monitoring.

Although residues of Hg in feathers are subject to a number of variables (review by Wolfe et al. 1998), they remain useful as a generally noninvasive monitoring tool (Becker et al. 1993, Burger 1993, Thompson et al. 1998). Newly grown primary feathers from adult grebes were first collected by us in 1992 (Cahill et al. 1998, Elbert and Anderson 1998) and again in 1998, when blood samples were also drawn. After 2001, we monitored grebe feather residues less intensively by collecting molted primaries and secondaries.

Molted feathers provide useful samples that can be collected without injury or disturbance to the individual (Furness et al. 1986, Burger 1993). We also compared molted adult osprey feathers (rectrices, tertials, wing coverts, and remiges) collected opportunistically below active nests at Clear Lake in 1992 and 1998/1999 and at Eagle Lake in 1999, then in subsequent years as with grebes described above. Mercury residues from adult-molted feathers represented levels deposited the previous season, but due to the tendency of nesting Osprey to be highly philopatric (Poole 1989*b*) and our observation that Osprey complete significant feather molt while in the Clear Lake ecosystem each year, it was highly probable that these feathers represented Hg exposures from Clear Lake. Since some of the grebe feathers reported here represented growing or new feathers and adult grebes also complete their major molt on Clear Lake, all those collected were also likely representative of Clear Lake Hg exposure.

Henny (1988, based largely on Prevost's [1983] work) pointed out, however, that wing molts in post-fledging, juvenile vs. adult Osprey are quite different. Also in adult Osprey, as is typical for many raptors (Clark 2004), molt in the primaries and secondaries (the flight feathers of the wing) occurs as a "wave molt," which, as described for Osprey (Henny 1988, Poole et al. 2002), results in a primary and secondary molt that occurs both on the wintering and breeding grounds. Thus, we might have encountered two major sampling problems with Osprey using only feathers: (1) first-time breeders in this expanding population would be expected to have lower residues because they would have grown their primaries and secondaries off-site; (2) if sampling only primaries and secondaries, the staggered wing molt might result in sampling feathers formed off-site by adults, as well. (Newton [1979], for example, showed the high resultant Hg variation in highly migratory Osprey from Sweden.)

We dealt with the first potential problem by sampling only from nests that had been occupied at least two years previous (see Fig. 1). Regarding the second potential problem, we also felt that nonmigratory or weakly migratory Clear Lake Osprey spent enough time in the Clear Lake system to minimize the likelihood that we would be sampling feathers formed off-site. For example, D. W. Anderson (*personal observation*) has observed nest site occupancy commonly lasting up to approximately eight months at Clear Lake compared to approximately five months in the more highly migratory Osprey from Eagle Lake. Although Osprey are also common at Clear Lake year-round, we have no way of knowing if these are the same individuals that breed there (satellite telemetry studies are needed). Also, in comparing grebe and Osprey feather residue patterns post hoc, both species followed similar patterns (as will be seen in data presented herein).

Additional, "historical," pre-study feather samples of grebes were taken from preserved (air-dried) whole-wing specimens, collected from 1967 to 1969 by Herman et al.

TABLE 1. Mercury residues found in feathers of Osprey at Clear Lake, California, USA (bottom rows), compared to similar species and their possible associations with potential adverse effects on individual health and population parameters.

Location or study	Feather type	TotHg (ppm)†	Reported effect	Source
Various				
A review	various	<1–5	"background" levels	Scheuhammer (1991)
A review	various	>20	Hg "risk"	Scheuhammer (1991)
Waterbirds				
New York	various	5–40 (lab)‡	"dangerously elevated"	Burger and Gochfeld (1997)
Belted Kingfisher (*Ceryle alcyon*)				
Tennessee, USA	various nestlings	3–9	"potentially toxic"	Baron et al. (1997)
Golden Eagle (*Aquila chrysaetos*)				
Scotland	various	>8	"reproductive effect"	Furness et al. (1989), Newton and Gailbrath (1991)
Ardeids				
Florida, USA	adult primaries	9 (lab)‡	"reproductive disorders"	Beyer et al. (1997)
Bald eagle (*Haliaeetus leucocephalus*)				
Great Lakes	various nestlings	13–21	no observed effect	Bowerman et al. (1994)
British Columbia, Canada	nestling secondaries	18 ± 6	no observed effect	Weech et al. (2006)
	adult secondaries	40 ± 22	no observed effect	Weech et al. (2006)
Sparrowhawk *(Accipiter nisus)*				
Finland	adult primaries	40	sterility	Wolfe et al. (1998)
Osprey (*Pandion haliaetus*)				
Baja California	nestling primaries	<1	background	Cahill et al. (1998)
Idaho, USA	nestling primaries	2.2 ± 1.5	no observed effect	Cahill et al. (1998)
Great Lakes	nestling mantle	3–7	no observed effect	Hughes et al. (1997)
Quebec (lake), Canada	nestling wing	7 ± 4	no observed effect	DesGranges et al. (1998)
	adult various	16 ± 13	no observed effect	DesGranges et al. (1998)
Quebec (reservoir), Canada	nestling wing	37 ± 20	predict decreased survival	DesGranges et al. (1998)
	adult various	58 ± 51	no observed effect	DesGranges et al. (1998)
Clear Lake, California, USA	nestling primaries	5 ± 2	no observed effect	Cahill et al. (1997)
	adult	20 ± 11	no observed effect	Cahill et al. (1997)

Note: Most of the Hg in feathers is excreted methylmercury (MeHg; see *Materials and methods*); it represents an index of exposure during feather development.

† Parts per million (ppm) = mg/kg wet mass or dry mass, but since feathers contain so little water, wet mass and dry mass values are nearly identical. Data are expressed in ranges or as mean ± SD.

‡ The studies cited here are samplings from the field that have been compared to laboratory studies, with subsequent risk predictions for field populations of various picivorous bird species.

(1969) and now archived in the Wildlife, Fish, and Conservation Biology Museum at the University of California, Davis. Additional molted Osprey feathers were also collected in 2002–2003 and 2004–2006 at both Eagle and Clear Lakes. Similar samples from grebes were taken at Clear Lake in 2003 and 2006. These recent samples were analyzed for Hg by cold-vapor atomic absorption spectroscopy (techniques are described in Eagles-Smith et al. 2008a). The other chemical analyses of most feathers by X-ray fluorescence (XRF) has already been described by Cahill et al. (1997), where total Hg is reported, although MeHg dominates feather residues (Thompson and Furness 1989a, b, Cahill et al. 1997). Methylmercury also dominates total Hg (TotHg) residues in various Clear Lake fish species (Suchanek et al. 2008c) selected as prey by these two important piscivorous birds. Mercury concentrations derived from both of these analytical methods are comparable (see Cahill et al. 1997).

Statistical analyses mentioned in the text follow those described in either MINITAB version 9.0 (Ryan and Joiner 1994) or STATISTIX 7 (Analytical Software 2000). In all cases where feathers were sampled, sample sizes represent one feather from one individual.

RESULTS

Recent mercury concentrations in avian piscivores at Clear Lake

Osprey that nested at Clear Lake (Fig. 1) in the early 1990s had elevated Hg feather residues (Cahill et al. 1998), higher than any other avian species sampled in this ecosystem (see Wolfe and Norman 1998). Clear Lake Osprey feathers had elevated Hg residues compared to Osprey from uncontaminated areas, but were lower than at other locations where Hg toxicity problems have been reported in other piscivorous birds (e.g., Henny et al. 2002; see also Table 1). There is much inter- and intraspecific, plus environmental variation reported in effect-level relationships, and most of the reviewed field data on Hg residues are inconclusive regarding specific thresholds represented by feather

residues. Yet it is reasonable to conclude that Osprey feather Hg residues in Clear Lake in some phases of our study approached predictors of low-end, potentially toxic exposure levels (see Cahill et al. 1998 and Table 1), as they were in grebes at generally the same times (Elbert and Anderson 1998). Additionally, based on an independent total maximum daily load (TMDL) risk analysis in the major drainage system from Clear Lake, Schwarzbach et al. (2001) (see also Suchanek et al. 2008a, c) concluded that the Osprey (as well as two other prominent piscivores, Bald Eagles, *Haliaeetus leucocephalus*, and Common Mergansers, *Mergus merganser*) were not adequately "protected" from hazardous Hg exposure in this system in the early 2000s. Elbert (1996), Anderson et al. (1997), and Elbert and Anderson (1998) demonstrated during the early 1990s that Osprey productivity increased with distance from the mine. Yet after about 1994, including feather samples from fledging Osprey from Clear Lake in 1995–1996 (Cahill et al. 1998), this relationship disintegrated. A similar hypothesis for grebes had not been examined because each year, grebes centralized nesting into one or two main colonies located on the southwest shore of the lake (Fig. 1).

In 1992, Clear Lake grebe Hg residues in soft tissue (liver, muscle, kidney) and feathers were elevated in relation to two comparison sites in California (Eagle Lake and Tule Lake) and were approaching approximate threshold levels for reproductive effects in birds, based on both field and experimental studies (reviewed in Elbert 1996, Cahill et al. 1998, Elbert and Anderson 1998). Additionally, in the early 1990s, some effects on biomarkers believed to be associated with Hg effects were also reported in Clear Lake grebes (Elbert and Anderson 1998). Nevertheless, Wolfe and Norman (1998) concluded, overall, that Hg levels at Clear Lake in the early-1990s were not overtly affecting most local wildlife. Additionally, more recent studies by Weech et al. (2006) in British Columbia have reported levels of Hg in feathers of Bald Eagles and Red-necked Grebes at approximately the "high end" of what we have found at Clear Lake in two taxonomically different, but trophically similar species (Table 1). They concluded that such elevated Hg exposure did not result in any measurable behavioral or demographic effects, compared to less contaminated populations in British Columbia.

Adult Osprey at Clear Lake in 1992 also contained higher Hg levels in their feathers than did their newly produced young (Cahill et al. 1998), as predicted from other studies (Red-billed Gulls [*Larus novaehollandiae*], Furness et al. 1990; Bald Eagles, Weech et al. 2006; Osprey, DesGranges et al. 1998; Table 1), supporting the concept of age-related Hg accumulation in the Clear Lake nesting population (see Thompson et al. 1991).

Osprey residues and productivity at Clear Lake

Feather residue levels of both Osprey and grebes were highly variable from year to year and are given in Fig. 2

along with residue data of juvenile largemouth bass (*Micropterus salmoides*) from the Oaks Arm of Clear Lake. Major food items of Osprey at Clear Lake were large fish (up to 30 cm; see also Poole 1989b): largemouth bass, Sacramento blackfish (*Orthodon microlepidotus*), Clear Lake hitch (*Lavinia exilicauda*), bullheads and catfish (*Ictalurus* sp.), and common carp (*Cyprinus carpio*). Observations reported by cooperators at Clear Lake also indicated that in the later stages of the 1992–2001 period, when threadfin shad (*Dorosoma petenense*) were dominating the system (Eagles-Smith et al. 2008b), Osprey were often seen carrying these smaller fish to some nests.

Mercury residue levels in adult Osprey feathers collected in 1992–1994 (Cahill et al. 1998) were significantly higher (t test, $P < 0.01$) than feathers sampled in 1998 (early, 20.0 ± 6.7 mg/kg DM [mean $\pm$ 95% CI], $N = 12$ nests sampled; late, 2.2 ± 1.6 mg/kg DM, $N = 7$ nests), but were higher in 2003 and 2006 (Fig. 2). A 1998 reduction of Hg in Clear Lake Osprey feathers was comparable to Eagle Lake in 1999, and Hg residue values were not significantly different between the two lakes (Mann-Whitney test, $P = 0.81$; Clear Lake, 2.2 ± 0.7 mg/kg DM, $N = 7$ nests; Eagle Lake, 1.9 ± 0.5 mg/kg DM, $N = 10$ nests). We considered Osprey fledging rates normal at Eagle Lake throughout our study (Elbert 1996; YY/NA at Eagle Lake, 1992–1994, 1.00–1.18, $N = 50$–52 nests). However, 2003 and 2006 Hg residues in Clear Lake Osprey feathers had unexpectedly returned to early-study levels (t test, $P > 0.10$: early, 20.0 ± 6.7 mg/kg DM; 2003, 22.7 ± 8.3 mg/kg; 2006, 12.3 ± 2.9 mg/kg), although Eagle Lake Osprey feathers continued at somewhat lower levels (2003, 7.3 ± 4.3 mg/kg DM; see previous discussion for earlier levels; they were collected but not analyzed in 2006). Thus, Hg in Osprey feathers in 2003 and 2006 had essentially returned to earlier, "pre-remediation" concentrations, but perhaps beginning to decline again in 2006. However, in 2006, three feathers from three different nesting Osprey in Oaks Arm were significantly higher (t test, $P > 0.02$, $N = 3, 7$) than feathers from other parts of the lake (Oaks Arm, 16.2 ± 1.2 mg/kg DM vs. 11.0 ± 3.4 mg/kg DM).

Overall, the total number of active Osprey nests at Clear Lake increased steadily from 1991 through 2006. Number of nest attempts went from seven active nests in 1991 to 25 in 2001, 27 in 2002, and 31 in 2006. The increase was nearly consistent throughout the study (annual increment = 1.6 new nests/yr or ~7%/yr). The overall rate of nesting population increase did not vary significantly between early and late periods (1.6 nests/yr vs. 1.4 nests/yr, $P > 0.10$, Mann-Whitney test). This suggests that Clear Lake Osprey throughout the study period had produced excess young and that these young must have had a high survival and return rate to the Clear Lake area, enough to produce the observed population increase and perhaps also spread into other areas. Annual estimates were therefore serially correlat-
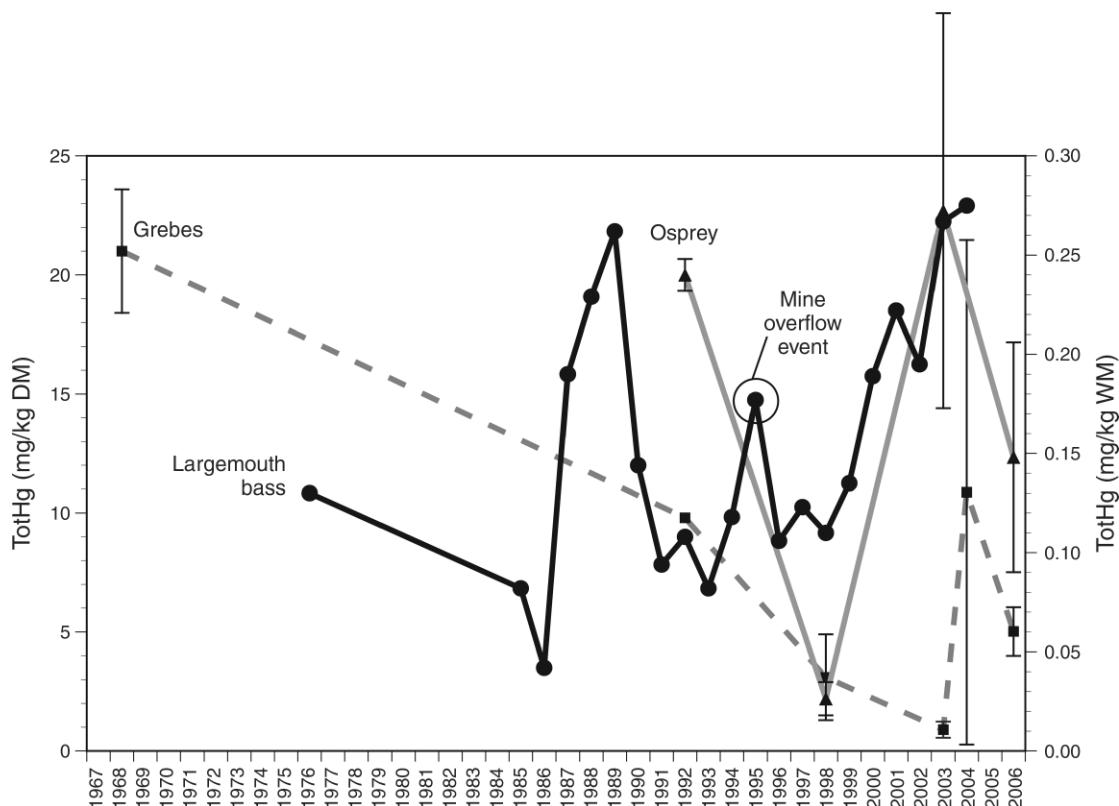
Fig. 2. Adult Osprey and grebe feather residues (left *y*-axis; means ± SE) over time plotted over residue levels in juvenile largemouth bass (right *y*-axis) from the Oaks Arm of Clear Lake (bass residue data are taken from Suchanek et al. [2008*c*]). Sample sizes are given in *Results and discussion: Osprey residues* … and *Aechmophorus* grebe residues.

ed, and any population size comparisons between pre- and post-remediation, or early vs. late, would be essentially invalid (populations in year *n* vs. year *n* + 1, *r* = 0.828, *P* = 0.003), other than to conclude that populations continued to increase throughout our study period, regardless of measured Hg residues in their feathers. The mean productivity (YY/NA) from 1992 through 2001 (*n* = 10 yr) at Clear Lake was 1.44 YY/NA. (We did not obtain productivity data in 1991, our first year of study, nor in 2002–2003.) Yet, fledging success did improve from 1.17 YY/NA during the early period to 1.62 YY/NA during the late period of the study (= post-remediation; Fig. 3; *P* < 0.03). And in 2006, from a sample of 10 successful and one unsuccessful nest attempts, productivity averaged 1.61 YY/NA. Osprey were not studied by us at Eagle Lake over the same time period, but nesting populations and productivity fluctuated, but remained stable throughout (T. Rickman, *personal comunication*).

These Clear Lake Osprey fledging rates were consistently within or above the range of normally producing populations from other areas at approximately the same latitudes (see Poole 1989*b*). Furthermore, Henny and Wight (1969) evaluated productivity and status for eastern U.S. Osprey populations and calculated that 0.95–1.30 YY/NA was sufficient to maintain a popula-
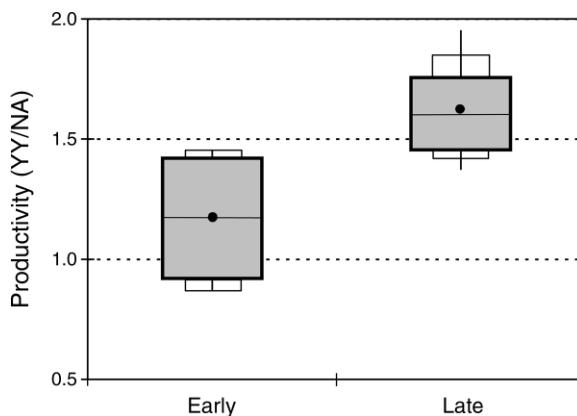

Fig. 3. Boxplots of Osprey productivity (number of young produced per active nest [YY/NA]) at Clear Lake in the early period (1992–1995) and late period (1996–2001) of the study (see *Materials and methods* for definition of time periods). Solid circles, means; internal, horizontal lines, medians; shaded boxes, the second and third quartiles; open boxes, upper/lower 95% CI; and whiskers, range. The means were significantly different (*t* test, *P* = 0.03).

tion. Spitzer (1980) and Spitzer et al. (1983), in studies of eastern Osprey populations, calculated that a long-term mean of 0.80 YY/NA represented a stationary population. In Florida, Bowman et al. (1989) reported mean Osprey fledging rates at 0.6 YY/NA in areas where they believed food was limiting (i.e., "poorer" habitats) vs. ~0.8–1.2 YY/NA in more suitable habitat. Van Daele and Van Daele (1982) reported that Osprey productivity was correlated with food availability and water level at a reservoir in Idaho (three-year mean with abundant food availability, 1.37 YY/NA). Morrissey et al. (2004) demonstrated that lower prey size, prey delivery rates, and prey biomass per nest resulted in lower Osprey nest success in British Columbia. Spatial variation in productivity like this is also characteristic of Bald Eagles, a species with nesting dispersion and feeding behavior similar to the Osprey (Elliott et al. 2005). Osprey productivity at Clear Lake, being relatively high in relation to most other areas, was thus not likely limited by Hg or food availability per se during our studies (see also Poole 1989a and Morrissey et al. 2004 for further discussion).

Recently, Henny and Kaiser (1996) studied an expanding Osprey population in the Willamette Valley of Oregon, quite similar to the Clear Lake population in that it was recently increasing from previous (organochlorine) pollution-related decimations (Poole et al. 2002), expanding into new habitat, and apparently not yet proximally limited by either nest site availability or food levels. From 1976 to 1993, Henny and Kaiser (1996) documented an increase from 13 to 78 active pairs. In 1993, the Oregon population produced an average of 1.64 YY/NA, although the nesting population had undergone an exponential rate of increase since 1976, very similar to the population expansion of a smaller size at Clear Lake.

Another piscivorous raptor also increased at Clear Lake during our studies. One Bald Eagle pair in 1986 (~1.5 km from the mine site) attempted to nest at Clear Lake, but failed (M. N. Kirven, *personal communication*). There was little eagle activity again until 1999 when Bald Eagles successfully nested for the first time at Clear Lake: two active nests each fledged two young (17.0 and 8.5 km from the Sulphur Bank Mercury Mine; a possible third nest with young was not confirmed). Bald Eagles continued to nest in the Clear Lake vicinity through 2006 and can now be considered a regular nesting species for the lake.

Aechmophorus grebe residues and productivity at Clear Lake

Mercury residues in Clear Lake grebes also varied dramatically over the course of our study (Fig. 2). The major food items of grebes were much smaller fish, perhaps dominated by inland silversides (*Menidia beryllina*) and threadfin shad, but precise food studies were not possible without additional disturbance (collections).
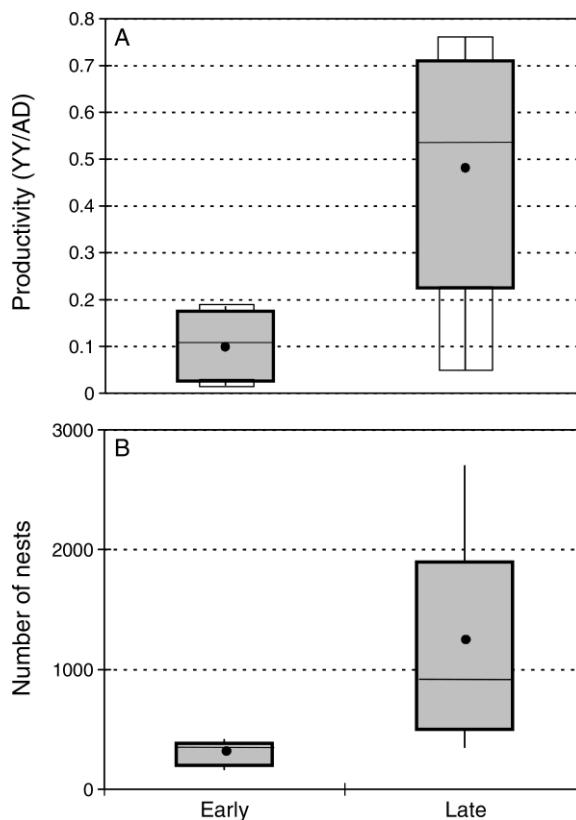

FIG. 4. (A) Grebe productivity (the number of young per adult [YY/AD]) during the early (1992–1995) and late (1996–2001) periods of study at Clear Lake (see *Materials and methods* for definition of time periods). Data are presented as in Fig. 3. Means were significantly different by *t* test ($P = 0.04$). (B) Number of nests in the same time periods. The 95% CI are not shown because the means were not significantly different by Mann-Whitney tests ($P = 0.34$).

Complete nest failure occurred once (in 1999) when water level decreases, probably due to lake draw-down for local irrigation purposes, resulted in grebe nests on dry ground. It is unusual for *Aechmophorus* grebes to even attempt to nest on land and it has been reported only once (Nero et al. 1958), as they are strongly dependent upon water level stability and stable emergent vegetation for nesting (Parmelee and Parmelee 1997). With 1999 removed from our analysis, grebe productivity and size of the nesting population improved/increased in the "post-remediation" period (Fig. 4). Because both early and late categories included years of very low productivity and nesting failures due to outside disturbances (1993–1995) and one year of severe nesting disturbances (1997), plus 1999, the final improvement in the Clear Lake nesting effort and nesting success (post-remediation) was not attributable to mine site remediation and declining Hg residues (exposure), but to active management through colony site protection from large, major disturbance events (Figs. 4 and 5).
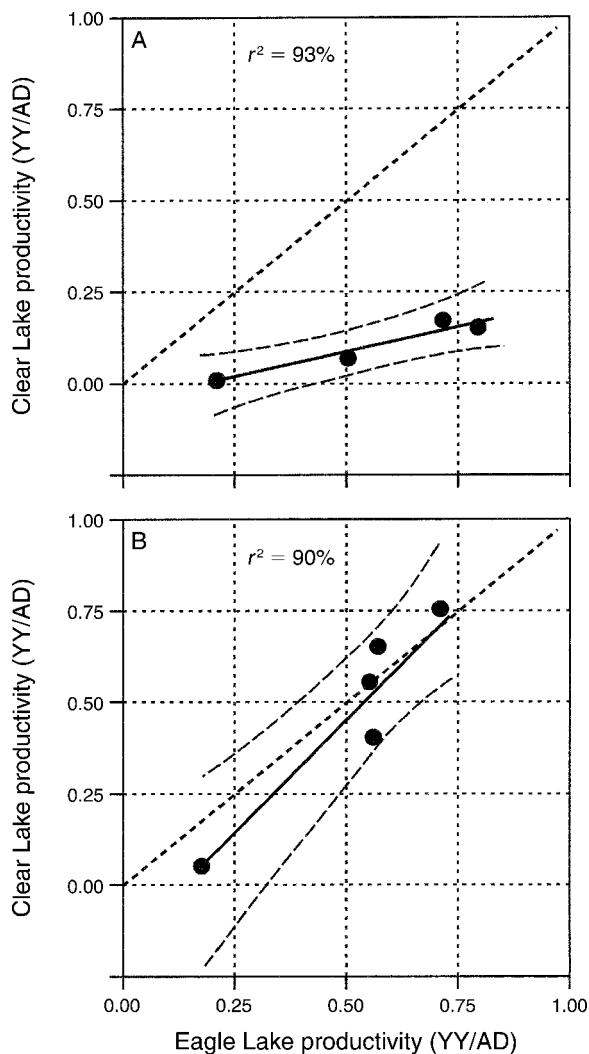
FIG. 5. (A) Grebe productivity (the number of young per adult [YY/AD]) at Clear Lake vs. our comparison site, Eagle Lake, California, USA for the "early remediation" period (1992–1995). The short-dashed line represents a 1:1 ratio, the solid line is a fitted linear regression ($P < 0.01$, $r^2 = 0.90$), and the long-dashed lines indicate 95% CI. (B) The same relationship during the "later period" (1996–2001) of study, with one year (1999) removed because of lake drawdown, which eliminated all grebe nesting. See *Materials and methods* for definition of time periods.

Yearly grebe populations were not serially correlated (autocorrelated) as were the Osprey on the same body of water ($r = -0.151$, $P = 0.699$), indicating that grebe breeding population numbers were more unpredictable and influenced by a different suite of ecological factors (namely human disturbances and regional effects of water level; see Anonymous 1999 and Fig. 5), although both species utilized contaminated fish resources from the same lake, albeit from different trophic levels.

Historic (1967–1969) grebe feathers from Clear Lake ($N = 77$ birds) were ~2.3 times higher in Hg than feathers first collected by us in this study in 1992 ($N = 12$;

historic, 22.9 ± 2.9 mg/kg DM; Fig. 2). Feather Hg residues in grebes also decreased significantly between the early (1992–1994) and late (1998) comparison periods of our intensive 10-year study (t test, $P < 0.01$: early, 9.8 ± 2.5 mg/kg DM, $N = 20$ birds; 1992 alone, 15.6 ± 4.0 mg/kg DM, $N = 12$ birds; late, 3.1 ± 1.8 mg/kg, $N = 8$ birds). Moreover, our reference site (Eagle Lake) grebe productivity comparisons for the late period (not including 1999 at Clear Lake) were highly correlated ($r = 0.949$, $P < 0.01$) at a ratio not significantly different from 1:1 (Fig. 5B). This also shows normal grebe reproduction at both lakes during recent years, but not earlier (Fig. 5A, B; but recall that these reductions were due to human disturbance factors and that with all this disturbance, it was not possible to test for Hg effects on grebes).

Regardless, several blood parameter biomarkers of potential Hg toxicity (depressed whole blood potassium and phosphorus) in grebes were reported for the pre-remediation, early period by Elbert and Anderson (1998). Notwithstanding, blood phosphorus levels in 1998 grebe blood samples were significantly higher than in 1992 (t test, $P < 0.02$: early, 4.6 ± 1.2 mg/dL, $N = 7$ individuals, late, 7.2 ± 1.5 mg/dL, $N = 6$ individuals), as were blood potassium levels (with one potassium outlier removed; t test, $P < 0.03$: early, 4.5 ± 1.0 mol/L, $N = 7$ birds; late, 6.9 ± 1.1 mol/L, $N = 5$ birds). Our internal references, calcium and sodium blood levels, however, remained constant through the two comparison periods (t tests, all $P > 0.05$: blood sodium, early, 155.0 ± 2.6 mol/L vs. late, 150.9 ± 3.4 mol/L [mean $\pm$ 95% CL]; blood calcium, early, 9.1 ± 0.3 mg/dL vs. late, 9.7 ± 0.6 mg/dL).

Recent grebe feather Hg levels remained low in 2003 (t test, $P > 0.05$: 2003; 0.90 ± 0.34 mg/kg DM), but in 2004–2006, were again elevated (7.0 ± 2.7 mg/kg; $N = 24$ birds), which was similar qualitatively to the increasing residue pattern in Osprey. Mean feather residues in Osprey and grebe feathers were correlated (Spearman's rank correlation, $R_s = 0.900$, $P < 0.04$); both fluctuated by almost one order of magnitude throughout the period from 1967–1969 through 2006 (Fig. 2).

DISCUSSION

There were no discernible population-level effects of Hg measured in Osprey or grebes at any point during our 1992–2001 studies or afterward (at least through 2006, and a detailed follow-up study is planned for 2007). Suchanek et al. (2008c) reported significant spatial trends in virtually all fish species studied at Clear Lake, with the highest Hg concentrations in the Oaks Arm (the mine location, Fig. 1); however, there were no discernible annual or long-term monotonic trends in Hg concentrations in the Oaks Arm, where Hg problems would most likely occur. Eagles-Smith et al. (2008b) showed, however, that some short-term changes in fish Hg concentrations were related to the presence/absence of invasive fish species such as the threadfin shad, suggesting that this species can potentially change trophic relationships to the point of altering the Hg
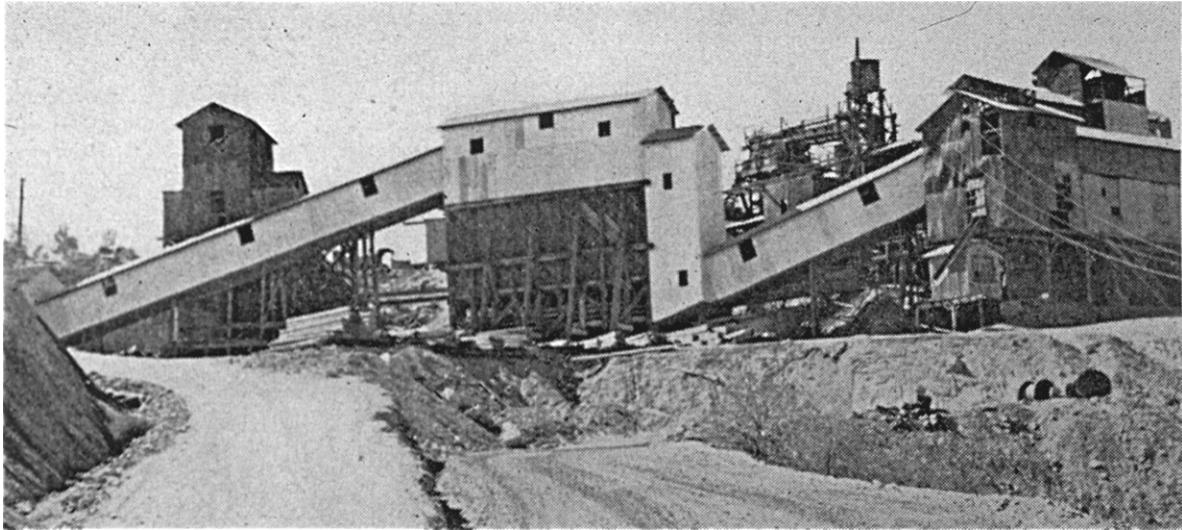
PLATE 7. Mercury retort furnace and processing facility at the Sulphur Bank Mercury Mine ca. 1947 (photo credit: Worthen Bradley; courtesy of California Geological Survey, Department of Conservation).

uptake in higher trophic species. And available long-term juvenile largemouth bass Hg residue data from Clear Lake (Oaks Arm) are somewhat consistent with the changes seen in available data on Osprey and grebe residue changes (Fig. 2). As of 2006, threadfin shad were still abundant in the lake. Isotope studies (as with Eagles-Smith et al. 2008b) on past and future Osprey and grebe feather samples are planned to elucidate the question of changing trophic interactions for avian piscivores at Clear Lake hypothesized here, especially when, and if, the threadfin shad (Eagles-Smith et al. 2008a, b) declines again.

We did not have resources to conduct long-term, detailed food habits studies to test for possible, perhaps more subtle, changes or direct connections of birds to their specific food items and potentially changing Hg exposure, nor was it our intention to create the kinds of disturbances required for such studies. However, as indicated above, there are certainly multiple, as yet unknown, factors that were important in driving changes in Hg uptake and bioaccumulation in Osprey and grebes at Clear Lake.

Aechmophorus grebes generally depart Clear Lake in the post-breeding dispersal period of their annual cycles around late September (Elbert 1996); although in a four-year period when threadfin shad were abundant, winter numbers at Clear Lake rose from the usual 1000–2000 individuals to 38 000–41 000 (1988 and 1989; Colwell et al. 1997); and it happened again in all the winters of 2003–2004 through 2006–2007. These unusual, short-term increases in wintering grebe numbers have been coincident with large increases of the threadfin shad from 1987 to 1989 and from ca. 2003 to 2005 (Eagles-Smith et al. 2008b). These events were also often followed by increases in largemouth bass that, in the first instance, lasted generally throughout the early

phases of our own study period (Colwell et al. 1997), when Hg residues were high and then later (2003, 2006) when Osprey feather Hg was elevated, as was Hg in grebe feathers in 2006. Such an increase in largemouth bass, a common prey item of Osprey, would support a similar 2003 resurgence of higher Osprey feather Hg levels (Fig. 2). Had we not sampled Osprey in 2003 and 2006 and grebes in 2006, we may have incorrectly concluded that the mine site remediation had been successful, based upon changes observed in the late-1990s and early 2000s.

In the Clear Lake ecosystem, wildlife monitoring is but one part of an effort to evaluate Hg contamination. Suchanek et al. (2000a) provide evidence that acid mine drainage is an important ongoing source of Hg loading to Clear Lake. This Hg source will likely continue into the future until some kind of more effective remediation can be found (see Suchanek et al. 2008d). Empirical observations such as those reported here, if even from a relatively long-term study like ours, must still be interpreted with caution. One wonders if a 10-year study is too little time to study Hg dynamics in piscivores in a system as dynamic as Clear Lake.

The significance of such contaminant input to future avian populations or individuals at Clear Lake is variable and perhaps still largely unknown, but if effects are to be demonstrated they might be found in the population cohort we did not study: the young birds after independence from parental care, after they no longer have the ability to excrete Hg into their developing plumage, and when they must undergo long periods of independent survival until they return to the breeding population as successful recruits (Sepulveda et al. 1999). This is the cohort most likely to be vulnerable to lower, but perhaps still ecologically or individually significant, exposures of MeHg (Henny et al. 2002), and

we recommend future studies on Clear Lake post-fledging Osprey and post-fledging grebe survival.

A prolonged drought in the Clear Lake watershed from 1986 to 1991, terminated by a severe season of cyanobacteria (bluegreen "algae") blooms (Richerson et al. 1994, Suchanek et al. 2003), also preceded the beginnings of this study, and it remains remotely possible that amelioration of those conditions was in some unknown way also related to higher Hg exposures during that drought, but prior to our studies.

But even more importantly, it has been established in modern resource management that Osprey populations respond strongly and effectively to the availability of adequate, stable nest sites and abundant food (Poole 1989*a*, *b*, Henny and Kaiser 1996); and grebe populations respond best if protected from disturbances and other man-induced perturbations (see Anonymous 1999). At Clear Lake, there was a continuing positive relationship between nesting Ospreys and the local power company. Power line managers occasionally tear down an Osprey nest, only to erect a replacement platform nearby, much as reported by Austin-Smith and Rhodenizer (1983), Poole (1989*a*), and Henny and Kaiser (1996) in other areas. Clear Lake Western and Clark's Grebes will continue to be subjected to disturbances and habitat change, but again, effective management through protection of habitat and from human-induced disturbances will be the most immediate and effective contributors to their population stability, at Clear Lake or anywhere else. There is little doubt that activities that increase undisturbed, stable, and secure long-term nest sites will greatly support contaminant remediation activities.

LITERATURE CITED

Analytical Software. 2000. Statistix 7: user's manual. Analytical Software, Tallahassee, Florida, USA.

Anderson, D. W. 1998. Evaluation and impact of multiple stressors on ecosystems: four classic case histories. Pages 3–7 *in* J. J. Cech, Jr., B. W. Wilson, and D. G. Crosby, editors. Multiple stresses in ecosystems. Lewis, New York, New York, USA.

Anderson, D. W., T. M. Cahill, Jr., T. H. Suchanek, and R. A. Elbert. 1997. Relationships between mercury and yearly trends in Osprey production and reproductive status at Clear Lake. Clear Lake Science and Management Symposium 1: 66–70.

Anonymous. 1999. Wildlife biology: the grebes come back. UC Davis Magazine 16:9.

Austin-Smith, P. J., and G. Rhodenizer. 1983. Ospreys, *Pandion haliaetus*, relocate nests from power poles to substitute sites. Canadian Field-Naturalist 97:315–319.

Baron, L. A., T. L. Ashwood, B. E. Sample, and C. Welsh. 1997. Monitoring bioaccumulation of contaminants in the belted kingfisher (*Ceryle alcyon*). Environmental Monitoring and Assessment 47:153–165.

Becker, P. H., R. W. Furness, and D. Henning. 1993. The value of chick feathers to assess spacial and interspecific variation in the mercury contamination of seabirds. Environmental Monitoring and Assessment 28:255–262.

Beyer, W. N., M. Spalding, and D. Morrison. 1997. Mercury concentrations in feathers of wading birds from Florida. Ambio 26:97–100.

Bowerman, W. W., E. D. Evans, J. P. Giesy, and S. Postupalsky. 1994. Using feathers to assess mercury and selenium risk to bald eagle reproduction in the Great Lakes region. Archives of Environmental Contamination and Toxicology 27:294–298.

Bowman, R., G. V. N. Powell, J. A. Hovis, N. C. Kline, and T. Wilmers. 1989. Variations in reproductive success between subpopulations of the Osprey (*Pandion haliaetus*) in south Florida. Bulletin of Marine Science 44:245–250.

Burger, J. 1993. Metals in avian feathers: bioindicators of environmental pollution. Review of Environmental Toxicology 5:203–215.

Burger, J., and M. Gochfeld. 1997. Risk, mercury levels, and birds: relating adverse laboratory effects to field biomonitoring. Environmental Research 75:160–172.

Cahill, T. M., D. W. Anderson, R. A. Elbert, B. P. Perley, and D. R. Johnson. 1998. Elemental profiles in feather samples from a mercury-contaminated lake in Central California. Archives of Environmental Contamination and Toxicology 35:75–81.

Cahill, T. M., B. P. Perley, and D. W. Anderson. 1997. X-ray analyses of elemental concentrations in feathers: comparison of XRF and PIXE. International Journal of PIXE 1/2:53–69.

Campbell, D. T., and J. C. Stanley. 1966. Experimental and quasi-experimental designs for research. Houghton Mifflin, Boston, Massachusetts, USA.

Clark, W. S. 2004. Wave moult of the primaries in accipitrid raptors, and its use in ageing immatures. Pages 795–804 *in* R. D. Chancellor and B.-U. Meyburg, editors. Raptors worldwide. World Working Group on Birds of Prey and Owls, Berlin, Germany.

Colwell, A. E., N. L. Anderson, and D. L. Woodward. 1997. Monitoring of dipteran pests and associated organisms in Clear Lake (California). Proceedings of the Annual Clear Lake Science and Management Symposium 1:15–32.

DesGranges, J. L., J. Rodrique, B. Tardif, and M. Laperle. 1998. Mercury accumulation and biomagnification in ospreys

(*Pandion haliaetus*) in the James Bay and Hudson Bay Regions of Québec. Archives of Environmental Contamination and Toxicology 35:330–341.

Eagles-Smith, C. A., T. H. Suchanek, A. E. Colwell, and N. L. Anderson. 2008*a*. Mercury trophic transfer in a eutrophic lake: the importance of habitat-specific foraging. Ecological Applications 18(Supplement):A196–A212.

Eagles-Smith, C. A., T. H. Suchanek, A. E. Colwell, N. L. Anderson, and P. B. Moyle. 2008*b*. Changes in fish diets and food web mercury bioaccumulation induced by an invasive planktivorous fish. Ecological Applications 18(Supplement): A213–A226.

Eisler, R. 1987. Mercury hazards to fish, wildlife, and invertebrates: a synoptic review. Biological Report 85(1.10). U.S. Fish and Wildlife Service, Laurel, Maryland, USA.

Elbert, R. A. 1996. Reproductive performance and mercury exposure of birds at Clear Lake, CA. Dissertation. University of California, Davis, California, USA.

Elbert, R. A., and D. W. Anderson. 1998. Mercury levels, reproduction, and hematology in western grebes from three California lakes, USA. Environmental Toxicology and Chemistry 17:210–213.

Elliott, K. H., C. E. Gill, and J. E. Elliott. 2005. The influence of tide and weather on provisioning rates of chick-rearing Bald Eagles in Vancouver Island, British Columbia. Journal of Raptor Research 39:1–10.

Furness, R. W., J. L. Johnston, J. A. Love, and D. R. Thompson. 1989. Pollutant burdens and reproductive success of golden eagles *Aquila chrysaetos* exploiting marine and terrestrial food webs in Scotland. Pages 495–500 *in* B.-U. Meyburg and R. D. Chancellor, editors. Raptors in the modern world. World Working Group on Birds of Prey, Berlin, Germany.

Furness, R. W., S. A. Lewis, and J. A. Mills. 1990. Mercury levels in the plumage of Red-billed Gulls *Larus novaehollandiae scopulinus* of known sex and age. Environmental Pollution 63:33–40.

Furness, R. W., S. J. Muirhead, and M. Woodburn. 1986. Using bird feathers to measure mercury in the environment: relationships between mercury content and moult. Marine Pollution Bulletin 17:27–30.

Heinz, G. H. 1979. Methylmercury: reproductive and behavioral effects on three generations of mallard ducks. Journal of Wildlife Management 43:394–401.

Heinz, G. H., D. J. Hoffman, S. L. Kondrad, and D. A. Erwin. 2006. Factors affecting the toxicity of methylmercury injected into eggs. Archives of Environmental Contamination and Toxicology 50:264–279.

Henny, C. J. 1988. Osprey *Pandion haliaetus*. Pages 7–101 *in* R. S. Palmer, editor. Handbook of North American birds. Volume 4. Yale University Press, New Haven, Connecticut, USA.

Henny, C. J., E. F. Hill, D. J. Hoffman, M. G. Spalding, and R. A. Grove. 2002. Nineteenth Century mercury: wading birds and cormorants of the Carson River, Nevada. Ecotoxicology 11:213–231.

Henny, C. J., and J. L. Kaiser. 1996. Osprey population increase along the Willamette River, Oregon, and the role of utility structures, 1976–1993. Pages 97–108 *in* D. M. Bird, D. E. Varland, and J. J. Negro, editors. Raptors in human landscapes. Academic Press, London, UK.

Henny, C. J., and H. M. Wight. 1969. An endangered osprey population: estimates of mortality and production. Auk 86: 188–198.

Herman, S. G., R. L. Garrett, and R. L. Rudd. 1969. Pesticides and the Western Grebe: a study of pesticide survival and trophic concentration at Clear Lake, Lake County, California. Pages 24–51 *in* M. W. Miller and G. G. Berg, editors. Chemical fallout: current research on persistent pesticides. C. C. Thomas, Springfield, Illinois, USA.

Hughes, K. D., P. J. Ewins, and K. E. Clark. 1997. A comparison of mercury levels in feathers and eggs of osprey (*Pandion haliaetus*) in the North American Great Lakes. Archives of Environmental Contamination and Toxicology 33:441–452.

Judge, D. S. 1983. Productivity of Ospreys in the Gulf of California. Wilson Bulletin 95:243–255.

Kendall, R. J., and T. E. Latcher, Jr., editors. 1994. Wildlife toxicology and population modeling: integrated studies of agroecosystems. Lewis, New York, New York, USA.

Kloskowski, J. 2001. Double-brooding in red-necked grebes. Waterbirds 24:121–124.

Koford, R. R., J. B. Dunning, C. A. Ribic, and D. M. Finch. 1994. A glossary for avian conservation biology. Wilson Bulletin 106:121–137.

Morrissey, C. A., G. V. Kardosi, L. K. Wilson, and J. E. Elliott. 2004. Effect of food supply and contaminants on Osprey productivity in southern British Columbia, Canada. Pages 353–368 *in* R. D. Chancellor and B.-U. Meyberg, editors. Raptors worldwide. World Working Group on Birds of Prey and Owls, Berlin, Germany.

Nero, R. W., F. W. Lahrman, and F. G. Bard. 1958. Dry-land nest-site of a Western Grebe colony. Auk 75:347–349.

Newton, I. 1979. Population ecology of raptors. Buteo Books, Vermillion, South Dakota, USA.

Newton, I., and E. A. Galbraith. 1991. Organochlorines and mercury in the eggs of golden eagles *Aquila chrysaetos* from Scotland. Ibis 133:115–120.

Parmelee, D. F., and J. M. Parmelee. 1997. Western Grebe and Clark's Grebe: habitat necessity versus phenology. Colonial Waterbirds 20:95–97.

Poole, A. F. 1989*a*. Regulation of osprey *Pandion haliaetus* populations: the role of nest-side [site] availability. Pages 227–234 *in* B. U. Meyburg and R. D. Chancellor, editors. Raptors in the modern world. World Working Group on Birds of Prey, Berlin, Germany.

Poole, A. F. 1989*b*. Ospreys: a natural and unnatural history. Cambridge University Press, Cambridge, UK.

Poole, A. F., R. O. Bierregaard, and M. S. Martell. 2002. Osprey (*Pandion haliaetus*). Number 683 *in* A. Poole and F. Gill, editors. The birds of North America. The Birds of North America, Philadelphia, Pennsylvania, USA.

Postupalsky, S. 1974. Raptor reproductive success: some problems with methods, criteria, and terminology. Raptor Research Report 2:21–31.

Prevost, Y. 1983. The moult of the Osprey. Ardea 71:199–209.

Richerson, P. J., T. H. Suchanek, and S. J. Why. 1994. The causes and control of algal blooms in Clear Lake. Clean Lakes Diagnostic/Feasibility Study for Clear Lake, California. Final report. U.S. Environmental Protection Agency, Region IX, San Francisco, California, USA.

Ryan, B. F., and B. L. Joiner. 1994. MINITAB handbook. Third edition. Duxbury, Belmont, California, USA.

Scheuhammer, A. M. 1991. Effects of acidification on the availability of toxic metals and calcium to wild birds and mammals. Environmental Pollution 71:329–375.

Schwarzbach, S., L. Thompson, and T. Adelsbach. 2001. An investigation of mercury bioaccumulation in the Upper Cache Creek watershed, 1997–1998. Environmental Contaminants Division Off Refuge Investigations Report FFS 1130 1F22, DEC ID 199710005. U.S. Fish and Wildlife Service, Sacramento, California, USA.

Sepulveda, M. S., G. E. Williams, Jr., P. C. Frederick, and M. G. Spalding. 1999. Effects of mercury on health and first-year survival of free-ranging Great Egrets (*Ardea albus*) from southern Florida. Archives of Environmental Contamination and Toxicology 37:369–376.

Spitzer, P. A. 1980. Dynamics of a discrete coastal breeding population of ospreys in the northeastern United States during the period of decline and recovery, 1969–1978. Dissertation. Cornell University, Ithaca, New York, USA.

Spitzer, P. A., A. F. Poole, and M. Scheibel. 1983. Initial recovery of breeding ospreys in the region between New York and Boston. Pages 231–241 *in* D. M. Bird, editor. Biology and management of bald eagles and osprey. Harpell, St. Anne de Bellevue, Quebec, Canada.

Storer, R. W., and G. L. Nuechterlein. 1992. Western and Clark's Grebe. Pages 1–24 *in* A. Poole, P. Stettenheim, and F. Gill, editors. The birds of North America. Number 26. The American Ornithologists' Union, Washington, D.C., USA.

Suchanek, T. H., C. A. Eagles-Smith, and E. J. Harner. 2008*a*. Is Clear Lake methylmercury distribution decoupled from bulk mercury loading? Ecological Applications 18(Supplement):A107–A127.

Suchanek, T. H., C. A. Eagles-Smith, D. G. Slotton, E. J. Harner, and D. P. Adam. 2008*b*. Mercury in abiotic matrices of Clear Lake, California: human health and ecotoxicological implications. Ecological Applications 18(Supplement):A128–A157.

Suchanek, T. H., C. A. Eagles-Smith, D. G. Slotton, E. J. Harner, A. E. Colwell, N. L. Anderson, L. H. Mullen, J. R. Flanders, D. P. Adam, and K. J. McElroy. 2008*c*. Spatiotemporal trends in fish mercury from a mine-dominated ecosystem: Clear Lake, California. Ecological Applications 18(Supplement):A177–A195.

Suchanek, T. H., L. H. Mullen, B. A. Lamphere, P. J. Richerson, C. E. Woodmansee, D. G. Slotton, E. J. Harner, and L. A. Woodward. 1998. Redistribution of mercury from contaminated lake sediments of Clear Lake, California. Water, Air, and Soil Pollution 104:77–102.

Suchanek, T. H., D. C. Nelson, R. Zierenberg, A. L. Bern, W. Shipp, P. King, and K. McElroy. 2000*a*. Influence of AMD from the abandoned Sulphur Bank Mercury Mine on methyl mercury production in Clear Lake (CA). Pages 218–224 *in* Proceedings: assessing and managing mercury from historic and current mining activities, 28 November 2000. U.S. Environmental Protection Agency, Office of Research and Development, Ada, Oklahoma, USA.

Suchanek, T. H., P. J. Richerson, J. R. Flanders, D. C. Nelson, L. H. Mullen, L. L. Brister, and J. C. Becker. 2000*b*. Monitoring inter-annual variability reveals sources of mercury contamination in Clear Lake, California. Environmental Monitoring and Assessment 64:299–310.

Suchanek, T. H., et al. 2003. Evaluating and managing a multiply stressed ecosystem at Clear Lake, California: a holistic ecosystem approach. Pages 1239–1271 *in* D. J. Rapport, W. L. Lasley, D. E. Rolston, N. O. Nielsen, C. O. Qualset, and A. B. Damania, editors. Managing for healthy ecosystems. Lewis, Boca Raton, Florida USA.

Suchanek, T. H., et al. 2008*d*. The legacy of mercury cycling from mining sources in an aquatic ecosystem: from ore to organism. Ecological Applications 18(Supplement):A12–A28.

Thompson, B. C., G. E. Knadle, D. L. Brubaker, and K. S. Brubaker. 2001. Nest success is not an adequate comparative estimate of avian reproduction. Journal of Field Ornithology 72:527–536.

Thompson, D. R. 1996. Mercury in birds and terrestrial mammals. Pages 341–356 *in* W. N. Beyer, G. H. Heinz, and A. W. Redmon-Norwood, editors. Environmental contaminants in wildlife: interpreting tissue concentrations. CRC Press, Boca Raton, Florida USA.

Thompson, D. R., and R. W. Furness. 1989*a*. Comparison of the levels of total and organic mercury in seabird feathers. Marine Pollution Bulletin 20:577–579.

Thompson, D. R., and R. W. Furness. 1989*b*. The chemical form of mercury stored in south Atlantic seabirds. Environmental Pollution 60:305–318.

Thompson, D. R., R. W. Furness, and L. R. Monteiro. 1998. Seabirds as biomonitors of mercury inputs to epipelagic and mesopelagic marine food chains. Science of the Total Environment 213:299–305.

Thompson, D. R., K. C. Hammer, and R. W. Furness. 1991. Mercury accumulation in Great Skuas *Catharacta skua* of known age and sex and its effects upon breeding and survival. Journal of Applied Ecology 28:672–684.

Van Daele, L. J., and H. A. Van Daele. 1982. Factors affecting the productivity of Osprey nesting in west-central Idaho. Condor 84:292–299.

Vouk, V. B., and P. J. Sheehan, editors. 1983. Scope 20: methods for assessing the effects of chemicals on reproductive functions. John Wiley and Sons, New York, New York, USA.

Weech, S. A., A. M. Scheuhammer, and J. E. Elliott. 2006. Mercury exposure and reproduction in fish-eating birds breeding in the Pinchi Lake region, British Columbia, Canada. Environmental Toxicology and Chemistry 25:1433–1440.

Wolfe, M. F., and D. Norman. 1998. Effects of waterborne mercury on terrestrial wildlife at Clear Lake: evaluation and testing of a predictive model. Environmental Toxicology and Chemistry 17:214–227.

Wolfe, M. F., S. Schwartzbach, and R. A. Suliaman. 1998. Effects of mercury on wildlife: a comprehensive review. Environmental Toxicology and Chemistry 17:146–160.

Wren, C. D., S. Harris, and N. Harttrup. 1995. Ecotoxicology of mercury and cadmium. Pages 392–423 *in* D. J. Hoffman, B. A. Rattner, G. A. Burton, Jr., and J. Cairns, Jr., editors. Handbook of ecotoxicology. CRC Press, Boca Raton, Florida, USA.