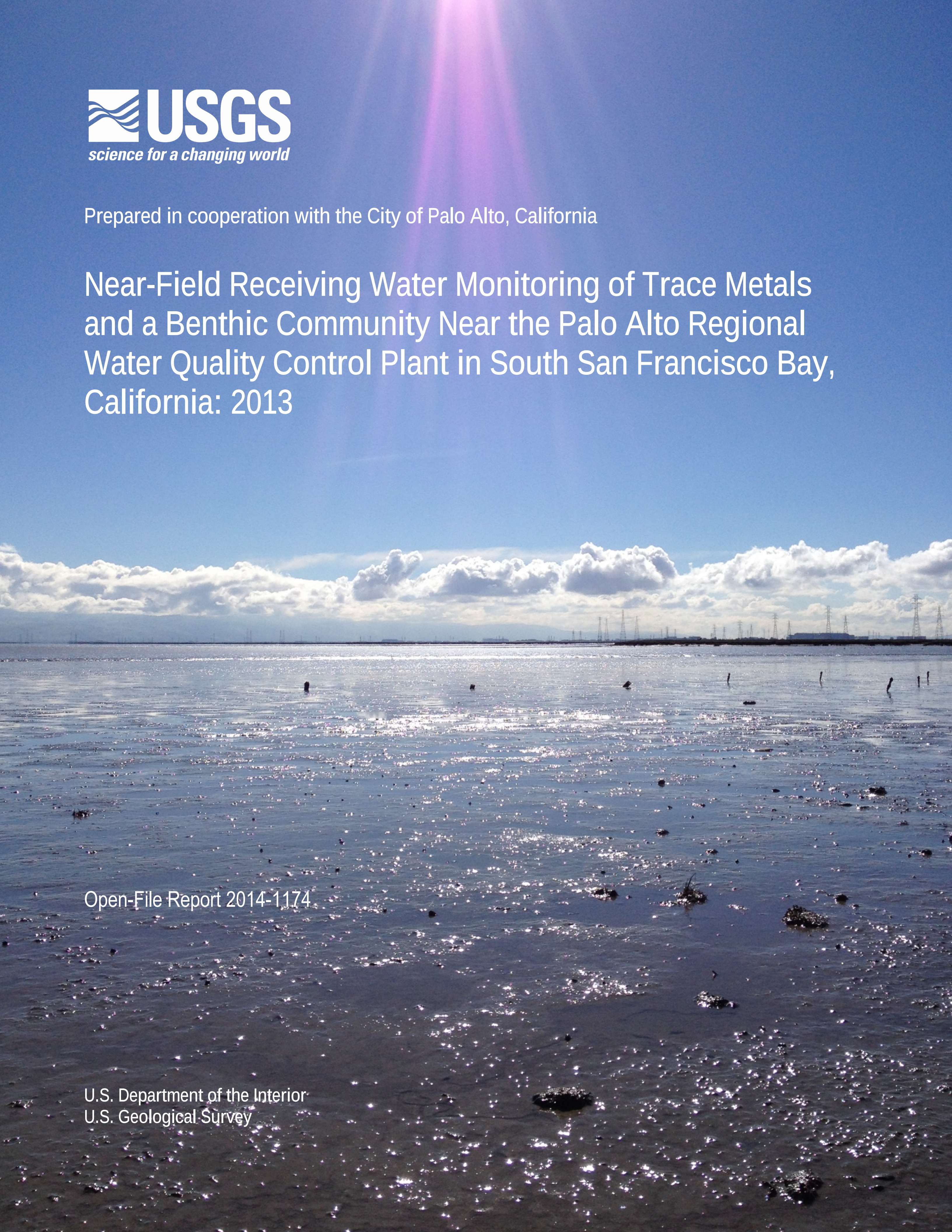


USGS
science for a changing world

Prepared in cooperation with the City of Palo Alto, California

Near-Field Receiving Water Monitoring of Trace Metals and a Benthic Community Near the Palo Alto Regional Water Quality Control Plant in South San Francisco Bay, California: 2013

Open-File Report 2014-1174

U.S. Department of the Interior
U.S. Geological Survey

Cover:
Mudflat east of Sand Point, Baylands Nature Preserve, Palo Alto, Calif. Photo courtesy of Jessica L. Dyke, USGS, April 2, 2014.

Prepared in cooperation with the City of Palo Alto, California

Near-Field Receiving Water Monitoring of Trace Metals and a Benthic Community Near the Palo Alto Regional Water Quality Control Plant in South San Francisco Bay, California: 2013

By Jessica L. Dyke, Daniel J. Cain, Janet K. Thompson, Amy E. Kleckner, Francis Parchaso, Michelle I. Hornberger, and Samuel N. Luoma

Open-File Report 2014-1174

U.S. Department of the Interior
U.S. Geological Survey

U.S. Department of the Interior
SALLY JEWELL, Secretary

U.S. Geological Survey
Suzette Kimball, Acting Director

U.S. Geological Survey, Reston, Virginia: 2014

For product and ordering information:
World Wide Web: http://www.usgs.gov/pubprod
Telephone: 1-888-ASK-USGS

For more information on the USGS—the Federal source for science about the Earth,
its natural and living resources, natural hazards, and the environment:
World Wide Web: http://www.usgs.gov
Telephone: 1-888-ASK-USGS

Suggested citation:
Dyke, J.L., Cain, D.J., Thompson, J.K., Kleckner, A.E., Parchaso, F., Hornberber, M.I., and Luoma, S.N., 2014, Near-field receiving water monitoring of trace metals and a benthic community near the Palo Alto Regional Water Quality Control Plant in south San Francisco Bay, California, 2013: U.S. Geological Survey Open-File Report 2014-1174, 81 p., *http://dx.doi.org/10.3133/ofr20141174*.

ISSN: 2331-1258 (online)

Contents

Figures

Tables

Conversion Factors

Multiply	By	To obtain
foot (ft)	0.3048	meter (m)
gallon (gal)	3.785	liter (L)
inch (in)	2.54	centimeter (cm)
inch (in)	25,400	micrometer (µm)
meter (m)	1,000,000	micrometer (µm)
mile (mi)	1.609	kilometer (km)
ounce (oz)	28.35	gram (g)
part per million (ppm)	1	microgram per gram (µg/g)
milligram per kilogram (mg/kg)	1	microgram per gram (µg/g)

Temperature in degrees Celsius (°C) may be converted to degrees Fahrenheit (°F) as follows:
°F=(1.8×°C)+32
Temperature in degrees Fahrenheit (°F) may be converted to degrees Celsius (°C) as follows:
°C=(°F-32)/1.8
Concentrations of chemical constituents in water are given either in milligrams per liter (mg/L) or micrograms per liter (µg/L).
NOTE TO USGS USERS: Use of liter (L) as a special name for cubic decimeter (dm^3) is restricted to the measurement of liquids and gases. No prefix other than milli should be used with liter.

Abbreviations and Acronyms

Abbreviations and Acronyms	Meaning
mL	milliliter
µΩ	microohm
µg/g	microgram per gram
mg/kg	milligram per kilogram
µm	micrometer
CI	Condition Index
ERL	Effects Range-Low
ERM	Effects Range-Median
ICP-OES	Inductively Coupled Plasma-Optical Emission Spectrophotometry
IRMS	Isotopic Ratio Mass Spectrophotometry
MDL	Method Detection Limit
MLLW	Mean Low Low Water
MRL	Method Reporting Level
NIST	National Institute of Standards and Technology
NPDES	National Pollutant Discharge Elimination System
PARWQCP	Palo Alto Regional Water Quality Control Plant
RWQCB	California Regional Water Quality Control Board
SFEI	San Francisco Estuary Institute
USEPA	U.S. Environmental Protection Agency
USGS	U.S. Geological Survey

Near-Field Receiving Water Monitoring of Trace Metals and a Benthic Community near the Palo Alto Regional Water Quality Control Plant in South San Francisco Bay, California: 2013

By Jessica L. Dyke, Daniel J. Cain, Janet K. Thompson, Amy E. Kleckner, Francis Parchaso, Michelle I. Hornberger, and Samuel N. Luoma

Executive Summary of Past Findings

U.S. Geological Survey (USGS) personnel have assessed trace-metal concentrations in sediments and sediment-dwelling species since 1977 at an intertidal site in the vicinity of the discharge of the Palo Alto Regional Water Quality Control Plant (PARWQCP). They have also profiled that area's benthic community structure since 1974. Ancillary biotic and abiotic factors that could affect metal concentrations and benthic community structure—exotic species invasions, pelagic food availability, and weather anomalies—have also been measured during this time.

Initially, these studies found exceptionally high concentrations of copper (Cu) and silver (Ag) in mud-dwelling animals in this area, with strong seasonal variability. Additional studies identified the PARWQCP as a point source for Cu and Ag and established the clam *Macoma petalum* as a biological indicator of metal exposure. The annual mean concentrations of Cu and Ag in *Macoma petalum* were 287 mg/kg and 105 mg/kg, respectively, in 1980. These levels exceeded tissue concentrations reported in the literature for this species and were much greater than seen elsewhere in San Francisco Bay. Elevated metal concentrations coincided with reduced reproductive activity in *M. petalum.* Related studies supported the theory that elevated Ag concentrations inhibited the development of reproductive tissue. The benthic community also showed signs of environmental stress during this time. Opportunistic organisms (capable of fast invasion and propagation in disturbed environments) dominated the community. These organisms thrived on the surface of the mud in tubes or as shelled animals, brooded their young, and fed on waterborne particles.

Concentrations of Cu and Ag in both sediments and clams declined significantly during the 1980s as the PARWQCP improved its waste treatment facilities and conducted source control programs. The downward trends in Cu in sediments and in the tissues of *M. petalum* correlated with reduced Cu discharge from the PARWQCP. Coincident with the decline in Cu and Ag in the sediment and clams, the reproductive activity of the clam greatly increased. The composition of the benthic community also shifted during this period. Opportunistic species became less dominant, and nonopportunistic species became more persistent. Other environmental factors that vary seasonally and annually (for example, sediment composition, grain-size distribution, organic content, and ambient water salinity) were not associated with the observed temporal trends in metal concentrations, inferred metal effects on species, and benthic community changes. The only unidirectional change in an environmental factor during this period (1980–1990) was the decline in metal concentrations in discharge from the waste treatment plant.

Following the significant reductions in the 1980s, concentrations of Cu and Ag in sediments and clams have remained relatively low and stable. Concentrations have fluctuated modestly and without a sustained temporal trend. However, Ag in sediments remains greater than what may be considered the regional background (0.09 mg/kg). This persistent, low level of contamination likely derives from Ag introduced to the site before the 1990s. The concentrations of Cu and Ag in *M. petalum* have fluctuated as much as four-fold. Concentration minima for Cu observed during this period (1991, 2000–2005, and 2008–2012) were comparable to what can be considered baseline concentrations for this species in San Francisco Bay (20–30 mg/kg).

Two lines of evidence suggest that the effluent of the PARWQCP is no longer the main driver of temporal patterns in metals at the sampling site. First, since the 1990s, annual variations in Ag and Cu in *M. petalum* have not correlated with discharge of Cu and Ag from PARWQCP. Second, temporal patterns in Ag and Cu at the site are generally similar to patterns for other major and minor elements that derive from terrigenous inputs and multiple discrete and diffuse anthropogenic sources to South San Francisco Bay. Thus, metal concentrations in sediments and tissue of *M. petalum* are more likely a combination of inputs from the PARWQP and other regional sources, cycling of contaminants stored within sediments, and regionally-scaled physical and biogeochemical processes controlling the distribution and bioavailability of metals.

As concentrations of Ag and Cu in *M. petalum* declined, reproductive activity increased both in terms of the percentage of individuals that were in a reproductively active stage and the frequency of reproductive activity during the year. Overall, the reproductive status of the population has improved and stabilized over the 20 years of reduced exposure to Ag and Cu at the site.

Over the same period, the composition of the infaunal community shifted from a dominance of surface-dwelling, brooding species to species with various life-history characteristics. In particular, species that lay their eggs in the mud and feed by burrowing through and consuming the mud, which were rare in the community in the 1970s and 1980s, have increased in abundance. This pattern continued through 2007, with the less opportunistic species becoming more dominant in abundance. A disturbance occurred on the mudflat in early 2008 (possible causes include sediment accretion or freshwater inundation) that resulted in the loss of the benthic animals, except for those deep-dwelling animals like *M. petalum*. Animals immediately returned to the mudflat, which indicates that the disturbance was not due to a persistent toxin or to anoxia. Benthic community data in 2009 show that the animals that have returned to the mudflat are those that can respond successfully to a physical, nontoxic disturbance. The most recent community surveys show a mix of animals that consume the sediment, filter feed, brood their young, and have pelagic larvae that must survive life on the sediment at a young age. The 2008 defaunation event allowed an examination of the response of the community to a natural disturbance and a comparison of this recovery to the long-term recovery observed in the 1970s, when the decline in sediment pollutants was the dominating factor. Today, the community at this site is very similar to the benthic community observed by Thompson and Parchaso (2013) throughout south San Francisco Bay: although small filter feeding species are numerically dominant, there is a significant proportion of the community that feeds on surface and subsurface sediment particles. This does not occur when the sediment has a high concentration of toxicants.

When this study started in the late 1970s, the site was already heavily contaminated with metals. Although the authors assume that the biological conditions reflected the consequences of elevated metal exposures, there is a scarcity of preexisting data to evaluate impacts due to elevated metals. However, the long-term record contained in this study provides a unique opportunity to document biological response when the stress of metal exposure is relaxed. The data make a compelling case that the mitigation of Ag and Cu in waste-water effluent during the 1980s allowed for biological recovery and the establishment of a more diverse and stable infaunal community.

Abstract

Trace-metal concentrations in sediment and in the clam *Macoma petalum* (formerly reported as *Macoma balthica*), clam reproductive activity, and benthic macroinvertebrate community structure were

investigated in a mudflat 1 kilometer south of the discharge of the Palo Alto Regional Water Quality Control Plant (PARWQCP) in South San Francisco Bay, Calif. This report includes the data collected by U.S. Geological Survey (USGS) scientists for the period January 2013 to December 2013. These data serve as the basis for the City of Palo Alto's Near-Field Receiving Water Monitoring Program, initiated in 1994.

Following significant reductions in the late 1980s, silver (Ag) and copper (Cu) concentrations in sediment and *M. petalum* appear to have stabilized. Data for other metals, including chromium (Cr), mercury (Hg), nickel (Ni), selenium (Se), and zinc (Zn), have been collected since 1994. Over this period, concentrations of these elements have remained relatively constant, aside from seasonal variation that is common to all elements. In 2013, concentrations of Ag and Cu in *M. petalum* varied seasonally in response to a combination of site-specific metal exposures and annual growth and reproduction, as reported previously. Seasonal patterns for other elements, including Cr, Ni, Zn, Hg, and Se, were generally similar in timing and magnitude as those for Ag and Cu. In *M. petalum,* all observed elements showed annual maxima in January–February and minima in April, except for Zn, which was lowest in December. In sediments, annual maxima also occurred in January–February, and minima were measured in June and September. In 2013, metal concentrations in both sediments and clam tissue were among the lowest concentrations on record. This record suggests that regional-scale factors now largely control sedimentary and bioavailable concentrations of Ag and Cu, as well as other elements of regulatory interest, at the Palo Alto site.

Analyses of the benthic community structure of a mudflat in South San Francisco Bay over a 40-year period show that changes in the community have occurred concurrent with reduced concentrations of metals in the sediment and in the tissues of the biosentinel clam, *M. petalum,* from the same area. Analysis of the *M. petalum* community shows increases in reproductive activity concurrent with the decline in metal concentrations in the tissues of this organism. Reproductive activity is presently stable (2013), with almost all animals initiating reproduction in the fall and spawning the following spring. The community has shifted from being dominated by several opportunistic species to a community where the species are more similar in abundance, a pattern that indicates a more stable community that is subjected to fewer stressors. In addition, two of the opportunistic species (*Ampelisca abdita* and *Streblospio benedicti*) that brood their young and live on the surface of the sediment in tubes have shown a continual decline in dominance coincident with the decline in metals; both species had short-lived rebounds in abundance in 2008, 2009, and 2010. *Heteromastus filiformis* (a subsurface polychaete worm that lives in the sediment, consumes sediment and organic particles residing in the sediment, and reproduces by laying its eggs on or in the sediment) showed a concurrent increase in dominance and, in the last several years before 2008, showed a stable population. *H. filiformis* abundance increased slightly in 2011–2012 and returned to pre-2011 numbers in 2013. An unidentified disturbance occurred on the mudflat in early 2008 that resulted in the loss of the benthic animals, except for those deep-dwelling animals like *Macoma petalum*. Animals immediately returned to the mudflat in 2008, which was the first indication that the disturbance was not due to a persistent toxin or to anoxia. The reproductive mode of most species present in 2013 is reflective of the species that were available either as pelagic larvae or as mobile adults. Although oviparous species were lower in number in this group, the authors hypothesize that these species will return slowly as more species move back into the area. The use of functional ecology was highlighted in the 2013 benthic community data, which show that the animals that have now returned to the mudflat are those that can respond successfully to a physical, nontoxic disturbance. Today, community data show a mix of animals that consume the sediment, filter feed, have pelagic larvae that must survive landing on the sediment, and brood their young. USGS scientists continue to observe the community's response to the 2008 defaunation event because it allows them to examine the response of the community to a natural disturbance (possible causes include sediment accretion or freshwater inundation) and compare this recovery to the long-term recovery observed in the 1970s when the decline in sediment pollutants was the dominating factor.

Introduction

Determining spatial distributions and temporal trends in trace metals in sediments and benthic organisms is common practice for monitoring environmental contamination. These data can be the basis for inferring ecological implications of metal contamination. Another common method of environmental monitoring is to examine the community structure of sediment-dwelling benthic organisms (Simon, 2002). Spatial and temporal changes in community structure reflect the response of resident species to environmental conditions, although the underlying cause(s) for the response may be difficult to identify and quantify. Integrating measurements of metal exposure and biological response can provide a more complete view of anthropogenic disturbances and the associated effects on ecosystem health.

Environmental Exposure to Trace Metals

Sediment particles can strongly bind metals, effectively removing them from solution. As a result, sediments may accumulate and retain metals released to the environment. Contaminated sediments may become a chronic source of metals to the environment. Thus, concentrations of metals in sediments serve as a record of metal contamination in an estuary, with some integration over time. Fluctuations in the record may be indicative of changes in anthropogenic releases of metals into the environment.

Metals in sediments are also indicative of the level of exposure of benthic animals to metals through contact with, and ingestion of, bottom sediments and suspended particulate materials. However, geochemical conditions of the sediment affect the biological availability of the bound metals. Assimilation of bioavailable sediment-bound metal by digestive processes and the relative contribution of this source of metals relative to metals in the aqueous phase are not well understood. Thus, in order to better estimate bioavailable metal exposures, the tissues of the organisms themselves may be analyzed for trace metals. Benthic organisms concentrate most metals to levels higher than those that occur in solution. Therefore, the record of metal concentrations in clam tissue can be a more sensitive indicator of anthropogenic metal inputs than the sediment record. Different species concentrate metals to different degrees. However, if one species is analyzed consistently, the results can be used to indicate trace-element exposures to the local food web.

Biological Response to Trace Metals

Contaminants can adversely affect benthic organisms at several organizational levels. For example, responses to a pollutant at the cellular or physiological level of an individual can result in changes at the population level, such as reductions in growth, survival, and reproductive success. Community level responses to population level impairment can include overall shifts in species abundance, favoring metal-tolerant species, which can result in changes in predator/prey interactions and competition for available resources. Changes in the benthic community can ultimately result in changes at the ecosystem level due to that community's importance in the cycling of carbon in aquatic environments (Alpine and Cloern, 1992, provides a local example).

In all aquatic environments, benthic organisms may be exposed to contaminants at all life stages through a variety of routes—sediment, water, and food (Wang and Fisher, 1999, provides a summary of the potential transport of trace elements through food). Toxicant exposure is related to contaminant concentration as well as duration. Even at low contaminant levels, long-term exposure can affect benthic organisms. The added complexity of synergistic or antagonistic effects between different contaminants, and between contaminants and natural stressors, makes causal relationships difficult to identify and quantify, even on a site-specific basis. However, a time-integrated picture of ecosystem response to contaminant loading can be provided by field studies that link changes in exposure at multiple time scales (in this case seasonal to decadal) to changes at individual, population, and community levels.

RWQCB and NPDES

The California Regional Water Quality Control Board (RWQCB) has prescribed a Self-Monitoring Program with its reissuance of the National Pollutant Discharge Elimination System (NPDES) permits for South San Francisco Bay dischargers. The recommendation includes specific receiving-water monitoring requirements.

Since 1994, the Palo Alto Regional Water Quality Control Plant (PARWQCP) (fig. 1) has been required to monitor metals and other specified parameters in sediments and the clam *M. petalum* at an inshore location in South San Francisco Bay, Calif. In addition to the required monitoring, PARWQCP has undertaken monitoring of the benthic community as a whole. The monitoring protocols have been designed to be compatible with or complement the RWQCB's Regional Monitoring Program. Monitoring efforts are being conducted by the U.S. Geological Survey (USGS) and are coordinated with more than 30 years of previous data collections and investigations by the USGS at this inshore location.

Objectives

The data collected during this study include trace-metal concentrations in sediments and clams, clam reproductive activity, and benthic community structure. These data and those reported earlier (for example, Hornberger and others, 2000a; Luoma and others, 1991, 1995a, 1996; Moon and others, 2005; Shouse and others, 2003, 2004; Thompson and others, 2002; Cain and others, 2006; Dyke and others, 2011, 2012) were used to meet the following objectives:

- Provide data to assess seasonal and annual trends in trace-element concentrations in sediments and clams, reproductive activity of clams, and benthic community structure at a site designated in the RWQCB's Self-Monitoring Program guidelines for PARWQCP.
- Present the data within the context of historical changes in South San Francisco Bay and within the context of other locations in the Bay published in the international literature.
- Coordinate inshore receiving water monitoring programs for PARWQCP and provide data compatible with relevant aspects of the Regional Monitoring Program. The near-field data will augment the Regional Monitoring Program as suggested by the RWQCB.
- Provide data that could support other South San Francisco Bay issues or programs, such as development of sediment quality standards.

Approach

Despite the complexities inherent in monitoring natural systems, the adopted approach has been effective in relating changes in near-field contamination to changes in reproductive activity of a clam (Hornberger and others, 2000b) and in benthic community structure (Kennish,1998). This study, with its basis in historical data, provides a context within which future environmental changes can be assessed.

Metal concentrations were monitored in sediments and a resident clam species, *Macoma petalum*. Analysis of trace-metal concentrations in the sediments provides a record of metal contamination of the site. The concentration and bioavailability of sediment-bound metals are affected by hydrology and geochemical factors (Thomson-Becker and Luoma, 1985; Luoma and others, 1995b). Thus, ancillary data, including grain-size distribution, organic carbon, aluminum and iron content of the sediment, regional rainfall, and surface salinity were collected to interpret seasonal, annual, and interannual variation in metal concentrations. The tissue of *M. petalum* provides a direct measure of exposure to bioavailable metals.

Biological response of the benthic community to metal exposure was examined at three levels of organization: individual, population, and community. At the individual level, concentrations of metals in the tissues of *M. petalum* were compared with physiological indicators. Two common animal responses to environmental stress are reduced reproductive activity and reduced growth. Growth and reproduction in *M. petalum* occur on fairly regular seasonal cycles. Seasonally, a clam of a given shell length will increase somatic tissue weight as it grows during the late winter and spring. Reproductive tissue increases during the early stages of reproduction and subsequently declines during and after reproduction. These cycles can be followed with the condition index (CI), which is an indicator of the physiological condition of the animal. The condition index (CI) is the total soft-tissue weight of a clam standardized to shell length. Interannual differences in growth and reproduction, expressed in the CI, are affected by the availability and quality of food, as well as other stressors, such as pollutant exposure and salinity extremes. An earlier study (Hornberger and others, 2000b) has shown that reproductive activity of *M. petalum* has increased with declining metal concentrations in animals from this location. Therefore, CI and reproductive activity of *M. petalum* appear to be useful indicators of physiological stress by pollutants at this location and continue to be monitored for this study.

At the population level, trends of the dominant benthic species were examined to see if certain species have been more affected than others by environmental change. It has been shown that most taxonomic groups have species that are sensitive to elevated Ag (Luoma and others, 1995b) and that some crustacean and polychaete species are particularly sensitive to elevated sedimentary Cu (Morrisey and others, 1996; Rygg, 1985). In addition, the benthic community was examined for changes in structure: that is, shifts in the species composition of the macroinvertebrate community resulting in a change in the function of the community. The authors hypothesized that a shift in community composition and potentially in the function of the benthic community in the ecosystem would result from changes in the concentrations of specific metals or from a composite of all contaminants for several reasons. First, prior studies have shown that South San Francisco Bay benthic communities were dominated by opportunistic species in the 1980s (Nichols and Thompson, 1985a). These opportunistic species might become less dominant as environmental stressors decrease. Second, environmental pollutants may differentially affect benthic species that use different feeding and reproductive modes. An intertidal mudflat community, such as this study site, should include a combination of species that feed on particles in the water column, on settled and buried food particles in the mud, and on other organisms. The absence of any one of these feeding groups may show limitations on species as a result of environmental stressors that target specific feeding groups. For example, pollutants attached to sediment particles are more likely to affect species that consume the sediment as part of their feeding mode or those species that lay their eggs in the sediment.

Previous analysis of this community has shown no correlation between changes in the community and measured environmental parameters (salinity, air and water temperature, delta outflow, precipitation, chlorophyll *a*, sediment total organic carbon, and biological oxygen demand; Shouse, 2002). Therefore, the community data are compared only to trace-metal data in this report.

Study Site

The Palo Alto site (PA) includes the benthic community sampling site and the *M. petalum* and sediment sampling site, both adjacent to Sand Point in Palo Alto Baylands Nature Preserve on a mudflat on the western shore of San Francisco Bay (not a slough) (fig. 1). The site is 1 kilometer southeast of the intertidal discharge point of the PARWQCP. The sampling locations are approximately 12 meters (m) from the edge of the marsh and 110 centimeters (cm) above mean low low water (MLLW).

The sediment and biological samples from this location reflect a response of the receiving waters to the effluent just beyond the location of discharge. Earlier studies (Thomson and others, 1984) have shown that dyes, natural organic materials in San Francisquito Creek, and waters in the PARWQCP discharge move predominantly south toward Sand Point, thereby affecting the mudflats in the vicinity of Sand Point. Thomson and others (1984) showed that San Francisquito Creek and the Palo Alto Yacht Harbor were minor sources of most trace elements compared to the PARWQCP. Based on spatial and temporal trends of Cu, Ag, and Zn in clams and sediments, the PARWQCP appeared to be the primary source of elevated metal concentrations at the PA site in the spring of 1980 (Thomson and others, 1984; Cain and Luoma, 1990). Metal concentrations in sediments and clams (*M. petalum*), especially Cu and Ag, have declined substantially since the original studies as more efficient treatment processes and source controls were employed (Hornberger and others, 2000b). Frequent sampling each year was necessary to characterize those trends because there was significant seasonal variability (Cain and Luoma, 1990; Luoma and others, 1985). This report characterizes data for the year 2013, thereby extending the long-term record at this site.

Previous reports (Luoma and others, 1995a, 1996, 1997, 1998; Wellise and others, 1999) also included data for a site in South San Francisco Bay that was influenced by discharge from the San Jose/Santa Clara Water Pollution Control Plant. Samples were collected from this site from 1994 to September 1999. Comparison of data from this site to the Palo Alto site allowed differentiation of local and regional long-term metal trends.

Methods

Sampling Frequency and Duration

In dynamic ecosystems such as San Francisco Bay, the environmental effects of anthropogenic stressors are difficult to distinguish from natural seasonal changes. Sustained sampling at frequent intervals can characterize seasonal patterns, capture episodic events, and identify longer term trends, thereby increasing the probability that anthropogenic effects can be identified. Analyses of early community data (1974 through 1983; Nichols and Thompson, 1985a, 1985b) showed that benthic samples need to be collected at monthly to bimonthly intervals to distinguish between natural and anthropogenic effects. Therefore, data reported herein are based on samples collected, with a few exceptions, on a monthly basis from the exposed mudflat at low tide between January and December, 2013. Samples collected in the field include surface sediment, the deposit-feeding clam *M. petalum*, surface water, and sediment cores for community analysis. Surface water, surface sediment, and *M. petalum* were not collected during the months of July, August, and November. Cores for benthic community analyses were not collected during the months of January, October, and December. Data on sediments, *M. petalum*, and surface water have been collected continuously since 1977, while community data were collected during 1974–1990 and 1998 to the present (2013).

Measurements of Metal Exposure

Sediment

Sediment samples were scraped from the visibly oxidized (brownish) surface layer (top 1–2 cm) of mud. This surface layer represents recently deposited sediment and detritus, or sediment affected by recent chemical reactions with the water column. The sediment also supports microflora and fauna, a nutritional source ingested by *M. petalum.* Sediment samples were immediately taken to the laboratory and sieved through a 100-micrometer (µm) mesh polyethylene screen with distilled water to remove large grains that might bias interpretation of concentrations. The mesh size was chosen to match the largest grains typically found in the digestive tract of *M. petalum*. All sediment data reported herein were determined from the fraction that passed through the sieve (<100 µm), termed the “silt/clay fraction.” Previous studies have shown little difference between metal concentrations in sieved and unsieved sediments when silt/clay-type sediment dominates at a site. However, where sand-size particles dominate the bed sediment, differences in metal concentrations can be substantial. Sediments in extreme South San Francisco Bay can vary spatially and temporally in their sand content (Dyke and others, 2011). Where sand content varies, sieving reduces the likelihood that differences in metal concentrations are the result of sampling sediments of different grain size. Some differences between the USGS and the Regional Monitoring Program results (San Francisco Estuary Institute, 1997) reflect the bias of particle size on the latter's data.

To provide a measure of bulk sediment characteristics at a site and thus provide some comparability with bulk sediment determinations such as that employed in the Regional Monitoring Program (San Francisco Estuary Institute, 1997), the fraction of sediment that did not pass through the sieve (≥100 µm) was determined. This fraction is termed the sand fraction. Bulk sediment samples were sieved to determine the percent sand and percent silt/clay (<100 µm) (appendix 1, table 1). The percentage of the bulk sediment sample composed of sand-size particles (percent sand) was determined by weighing the fraction of sediment that did not pass through the sieve (≥100 µm), dividing that weight by the total weight of the bulk sample, and multiplying the quotient by 100. The percentage of silt/clay in the sediment was determined similarly by weighing the sediment that passed through the sieve (grain size <100 µm).

The silt/clay fraction was dried at 70 degrees Celsius (ºC), weighed, and then subsampled to provide replicates weighing 0.4 to 0.6 g. These were redried (70 ºC), reweighed, and then digested by hot acid reflux (10 milliliters (mL) of 16 normal (N) nitric acid) until the digest was nearly clear. This method provides a "near-total" extraction of metals from the sediment and is comparable to the recommended procedures of the U.S. Environmental Protection Agency (USEPA) and to the procedures employed in the Regional Monitoring Program. It also provides data comparable to the historical data available on San Francisco Bay sediments. Although near-total analysis does not result in 100-percent recovery of all metals, recent comparisons between this method and more rigorous complete decomposition show that trends in the two types of data are very similar (Hornberger and others, 1999). After extraction, samples were evaporated until dry, then reconstituted in dilute hydrochloric acid (10 percent or 0.6 N). The hydrochloric acid matrix was specifically chosen because it mobilizes Ag into solution through the creation of Ag-chloro-complexes. Sediment extracts were allowed to equilibrate with the hydrochloric acid (minimum of 48 hours) before they were filtered (0.45 µm) into acid-washed polystyrene vials for elemental analysis. Another set of replicate subsamples from the silt/clay fraction were directly extracted with 12 mL of 0.6 N hydrochloric acid (HCl) for 2 hours at room temperature. This partial extraction method extracts metals bound to sediment surfaces and is operationally designed to obtain a crude chemical estimate of bioavailable metal. The extract was pressure filtered (0.45 µm) before elemental analysis.

Total organic carbon (TOC) concentrations were determined using a continuous flow isotope ratio mass spectrophotometer (IRMS) (table 1). Before the analysis, sediment samples were acidified with 12 N HCl vapor to remove inorganic carbon (Harris and others, 2001).

Water pooled on the surface of the mudflat was collected in a bottle and returned to the laboratory, where it was measured for salinity with a handheld refractometer.

Clam Tissue

Specimens of *M. petalum* were collected by hand on each sampling occasion. Typically, 60–120 individuals were collected, representing a range of sizes (shell length). As they were collected, the clams were placed into a screw-cap polypropylene container (previously acid-washed) containing site water. These containers were used to transport the clams to the laboratory.

In the laboratory, the clams were removed from the containers and gently rinsed with deionized water to remove sediment. A small amount of mantle water was collected from randomly selected clams for the determination of salinity with a refractometer. The salinity of the mantle water and the surface water collected from the site were typically within 1 part per thousand (ppt) of each other. Only surface water values are reported here. Natural, sand-filtered seawater (obtained from U.C. Santa Cruz, Long Marine Labs, Santa Cruz, California) was diluted with deionized water to the measured salinity of the site water. Clams were immersed in this water and moved to a constant temperature room (12 ºC) for 48 hours to allow for the egestion of sediment and undigested material from their digestive tracts. Clams were not fed during this depuration period. After depuration, the clams were returned to the laboratory and further prepared for chemical analysis.

Elemental Analysis, Excluding Mercury and Selenium

The shell length of each clam was measured with electronic calipers and recorded digitally. Clams were separated into 1- or 2-millimeter (mm) size classes (for example, 10.00–10.99 mm or 10.00–11.99 mm). The soft tissues from all of the individuals within a given size class were dissected from the shell and collected in preweighed 20-mL screw-top borosilicate glass vials to form a single composite sample for elemental analysis. The sample for each collection was thus composed of 9 to 13 composites, with each composite consisting of 1 to 35 clams of a similar shell length. The vials were capped with a glass reflux bulb and transferred to a convection oven (70 ºC). After the tissues were dried to constant weight, they were digested by reflux in subboiling 16 N nitric acid. The tissue digests were then dried and reconstituted in 0.6 N hydrochloric acid for trace-metal analysis.

Analysis for Mercury and Selenium

Samples collected during winter (December, January, and February), spring (April), and summer (June and September) were analyzed for total mercury (Hg) and selenium (Se). Approximately 40 clams were selected from the collection. The only criterion for selection was that the range of sizes (shell length) within this group was representative of the larger collection. Otherwise, the selection of individuals was random. Selected individuals were grouped according to size to form 3 composites, each containing a minimum of ~1.25 g wet weight. To meet this requirement, especially for the smaller clams, the 1-mm size classes were usually combined to form broader size classes (within 3–4 mm of each other, as appropriate). Once the composites were formed, the clams were dissected as described above, and the soft tissue was placed into preweighed 30-mL screw-top polycarbonate vials. These vials were closed and transferred to a freezer (–20 ºC). Once frozen, the samples were freeze-dried. Freeze dried samples were analyzed for total mercury by acid-digestion, BrCl oxidation, purge and trap, and cold vapor atomic fluorescence spectrometry according to the EPA Method1631, Revision E (2002), and for selenium by acid digestion, hydrogen peroxide oxidation, hydride generation inductively coupled plasma mass spectrometry (HG-ICP-MS) according to a method modified from Liber (2011) and Elrick & Horowitz (1985).

Analytical

Sediment and tissue concentrations of aluminum (Al), chromium (Cr), copper (Cu), iron (Fe), nickel (Ni), silver (Ag), and zinc (Zn) were determined using Inductively Coupled Plasma Optical Emission Spectrophotometry (ICP-OES). Analytical ICP-OES results are available upon request.

Quality Assurance

The polypropylene containers used in the field, depuration containers, glass-reflux bulbs, and all glassware and plastic used for metal analysis were first cleaned to remove contamination. Cleaning consisted of a detergent wash and rinse in deionized water, followed with a 10-percent hydrochloric-acid wash and thorough rinse in double-deionized water (approximately 18 mega-ohm (MΩ) resistivity).

Materials were dried in a dust-free positive-pressure environment, sealed, and stored in a dust-free cabinet.

Samples prepared for ICP-OES analysis (that is, all elements except Se and Hg) were accompanied by procedural blanks and standard reference materials (SRMs) issued by the National Institute of Standards and Technology (NIST). Analysis was preceded by instrument calibration, followed by quality-control checks with prepared quality-control standards before, during (approximately every 10 samples), and after each analytical run. Metal recoveries of sediment digests were evaluated with NIST 2709a San Joaquin soils and NIST 2711a Montana II Soil (appendix 3). Metal recoveries for soft tissue digests were evaluated with NIST 2976 Mussel Tissue and NIST 1566b Oyster Tissue (appendix 4). Results were consistent within methods, and most elements were within the ranges of certified values, with a few exceptions. The near-total extraction method only recovered on average 45–51 percent of the Al in NIST 2709a and 2711a. Zinc recoveries were much lower in NIST 1566b (59.7 percent) compared to NIST 2976 (95.7 percent), while nickel recovery was much better in NIST 1566b (95.0 percent) compared to NIST 2976 (68.9 percent). Lead recovery in NIST 1566b was quite high at 207 percent likely because the low certified concentration yielded analytical results near the method detection limit.

Method detection limits (MDL) and reporting levels (MRL) were determined using the procedures outlined by Glaser and others (1981), Childress and others (1999), and U.S. Environmental Protection Agency (2004) (appendix 5). A full quality-assurance/quality-control plan is available upon request.

A variety of standard reference materials were prepared according to the method used for the determination of Se and Hg. Observed concentrations fell within the range of certified values for these materials (appendix 6).

Other Data Sources

Precipitation data (fig. 2) for San Francisco Bay are reported from a station at San Francisco International Airport (station identification SFF) and were obtained from the California Data Exchange Center (http://cdec.water.ca.gov/).

Biological Response

Condition Index

The condition index (CI) is a measure of the clam’s physiological state derived from the relation between soft tissue weight and shell length and reported as the soft tissue dry weight (grams) for a clam of a particular shell length (mm). Specifically, for each collection, the relation between the average shell length and tissue dry weight of the composites was fit with a linear regression, and from that regression, the tissue dry weight was predicted for a normalized shell length of 25 mm.

Reproductive Activity

A minimum of 10 clams of varying sizes (minimum of 10 mm) were processed for reproductive activity concurrent with samples for metal analyses. Clams were immediately preserved in 10-percent formalin at the time of collection. The visceral mass of each clam was removed in the laboratory, stored in 70-percent ethyl alcohol, and then prepared using standard histological techniques. Tissues were dehydrated in a graded series of alcohol, cleared in toluene (twice for 1 hour each), and infiltrated in a saturated solution of toluene and Paraplast® for 1 hour, and two changes of melted Tissuemat® for 1 hour each. Samples were embedded in Paraplast® in a vacuum chamber and then thin sectioned (10 µm) using a microtome (Weesner, 1960). Sections were stained with Harris’ hematoxylin and eosin and examined with a light microscope. Each individual was characterized by size (length in mm), sex, developmental stage, and condition of gonads, thus allowing each specimen to be placed in one of five qualitative classes of gonadal development (previously described by Parchaso, 1993).

Community Analysis

Samples for benthic community analysis were collected with hand-held core 8.5-cm in diameter and 20-cm deep. Three replicate samples were taken arbitrarily, within a square-meter area, during each sampling date.

Benthic community samples were washed on a 500-µm screen, fixed in 10-percent formalin, and then later preserved in 70-percent ethanol. Samples were stained with rose bengal solution. All animals in all samples were sorted to species level where possible (some groups, such as the oligochaetes, are still not well defined in the Bay), and individuals for each species were enumerated. Taxonomic work was performed in conjunction with a private contractor familiar with the taxonomy of San Francisco Bay invertebrates (Susan McCormick, Colfax, California) (appendix 8). McCormick also compared and verified her identifications with previously identified samples.

Results

Salinity

Surface-water salinity in the Bay is related to the seasonal weather pattern in Northern California, which is characterized by a winter rainy season that has been defined as months with rainfall amounts greater than 0.25 inches (November through April) and a summer dry season (May through October) (fig. 2). The average annual rainfall during the period of record (1994–2013) is 22.3 inches. The severe statewide drought during 2013 was reflected in the cumulative annual rainfall measured at station SFF. For the year, 5.59 inches was measured, by far the lowest annual precipitation during the period of this study.

Surface-water salinity typically exhibits a seasonal pattern that is roughly the inverse of regional rainfall (fig. 3, table 1). The relatively weak seasonal variation in salinity in 2013 reflected the drought conditions decribed above. With rainfall in late 2012, salinity declined during January–March (19–20 ppt). Thereafter, salinity increased to 27 ppt in April, reached a maximum in September (30 ppt), and declined slightly during the rest of the year.

Sediments

Metal concentrations in surface sediments from Palo Alto typically display an annual periodicity of seasonal patterns that generally corresponds to the relative abundance of fine particles (figs. 4C–8C). Thomson-Becker and Luoma (1985) suggest that this intra-annual variation is related to changes in the size distribution of sediment particles caused by deposition of fine-grained particles in the winter and their subsequent wind-driven resuspension in the summer and fall. Because metal concentrations vary as a function of the ratio of surface area to volume of a particle, metal concentrations in fine-grained particles are typically higher on a weight basis than in larger particles. Thomson-Becker and Luoma (1985) showed that the composition of surface sediments was dominated by fine-grained particles, accompanied by high Al and Fe concentrations, during the period of freshwater input (low salinities through April), reflecting annual terrigenous sediment inputs from runoff. Coarser sediments dominated later in the year because the seasonal diurnal winds progressively stir the water and the fine sediments into suspension through the summer. This typical seasonal pattern of variation in sediment properties was mostly repeated in 2013 (fig. 4, appendix 1). The percentage of silt/clay in the sediment sharply increased from 58 percent in December 2012 to the annual maximum of 97 percent in February 2013 following heavy winter rains in November–December 2012. Fine-grained sediment remained dominant (greater than 86 percent) until it dropped again to an annual low of 46 percent in June. The percentage of silt/clay rose again as summer shifted to winter, reaching 93 percent in December 2013. The concentrations of Al and Fe followed the same general trend as the silt/clay size particles (maximum concentrations occurred in January–February) (fig. 4, table 1), as described above, reflecting the contribution of clays rich in Al and Fe.

Surface sediments from Palo Alto in 2013 contained about 1.24 percent (by weight) total organic carbon (TOC) (table 1). Carbon content varied slightly during the year and correlated positively with the percentage of fine-grained (<100 µm) sediment particles ($p < 0.05$). Total organic carbon was highest in January and February (1.42 percent and 1.50 percent, respectively) and lowest in the summer (0.856 percent in September).

The metals chromium (Cr) and nickel (Ni) are highly enriched in some geologic formations within the watershed. In North San Francisco Bay, studies of sediment cores indicated that concentrations of these elements similar to those reported here were derived from natural geologic inputs (Hornberger and others, 1999; Topping and Kuwabara, 2003). Inputs of minerals bearing Cr and Ni appear to vary seasonally as indicated by the varying concentrations of these metals in surface sediments. Typically, maximum concentrations coincide with winter/spring maxima in fine sediments, consistent with inputs from the surrounding watershed, whereas minimum concentrations occur during the late summer/fall (fig. 5, table 1). However, identification of inputs is complicated by seasonal changes in the particle size distribution of sediments decribed above. The minimum concentrations of Cr and Ni occurred in the summer/fall of 2013 (87.5 mg/kg of Cr in June and 65.2 mg/kg of Ni in September). Nickel reached its annual maxima in February of 2013 (89.0 mg/kg), and while Cr was also high in February (116 mg/kg), its maximum occurred in May (118 mg/kg). The concentration range and timing of variation for Cr and Ni in 2013 is typical of the record (1994–2013). Although average concentrations of Cr tended to increase during 2006–2010, approaching the record high Cr concentrations in 2003, this trend was not sustained in 2011–2013. Nickel has been fairly stable throughout the record.

Concentrations of Cu and Zn in sediments are shown with sediment guidelines set by the National Oceanic and Atmospheric Administration (Long and others, 1995) in figures 6 and 7 and table 1. Long and others (1995) defined values between ERL (Effects Range-Low) and ERM (Effects Range-Median) as concentrations that are occasionally associated with adverse effects on sensitive species (21–47 percent of the time for different metals). Values greater than the ERM were frequently associated with adverse effects on sensitive species (42–93 percent of the time for different metals). It is important to note, however, that these effects levels were derived mostly from bioassay data and are not accurate estimates of site-specific sediment toxicity. During 2006–2007, Cu concentrations increased to concentrations similar to those observed before 2000, apparently reversing a trend of declining concentrations during the intervening years. Since 2007, Cu concentrations have gradually decreased. In 2013, the average annual copper was 34.3 ± 1.6 (SEM) mg/kg, practically equivalent to the ERL (34 mg/kg). Copper concentrations were below the ERL in March, June, September, and December. The typical seasonal pattern was evident: Cu was highest in January–February (41.3 mg/kg and 38.7 mg/kg respectively) and May (39.2 mg/kg), and lowest in the dry summer/fall months of June (28.1 mg/kg) and September (27.9 mg/kg) (fig. 6, table 1). The higher partial-extractable Cu concentrations in 2011–2013 appear to reverse the slight downward trend observed from 2006 to 2010. Partial-extractable and near-total Zn concentrations were below the Zn ERL (150 mg/kg) for all of 2012–2013 (fig. 7, table 1). Maximum annual Zn concentrations occurred in February and May (123 mg/kg), and the minimum Zn concentration occurred in June (93.9 mg/kg). Average annual Zn concentrations in 2013 were the lowest observed since 2002.

The annual average of silver extracted from sediments using the partial-extraction method has steadily increased since the record low recorded in 2008–2009 (0.20 mg). The average silver concentration in 2013 was 0.35 mg/kg, equal to the average of the Ag values observed during 1994–2007. Like other trace elements, Ag concentrations have displayed a seasonal pattern over the record (fig. 8C); 2013 data display the typical seasonal pattern of higher values in the winter/spring and lower values in the fall/winter (fig. 8, table 1). Annual maxima were observed in February and December (0.45 mg/kg and 0.44 mg/kg, respectively), and annual minima occurred in May, June, and September (0.28 mg/kg for each month).

Mercury concentrations in 2013 were within the range usually observed in San Francisco Bay (0.2–0.4 mg/kg). The 2012 and 2013 annual averages of 0.22 mg/kg have been the lowest of the record (1994–2013). Concentrations of Hg in sediment showed a weak seasonal pattern. High concentrations (0.25 mg/kg) occurred in both January and February, and lowest concentrations (0.19 mg/kg) occurred in June and September (fig. 9, table 1).

Selenium concentrations in 2013 (annual average 0.40 mg/kg) increased slightly from 2012 values (fig. 9, table 1). The maximum Se concentration occurred in January at 0.52 mg/kg, and minimum concentrations occurred in June (0.31 mg/kg) and September (0.30 mg/kg). Sedimentary Se hasn’t exhibited a sustained temporal trend over the period of the record. Concentrations have varied annually during the record and since 2004 have alternated from relatively high concentrations (2004, 2006, 2008, and 2011) to relatively low concentrations (2005, 2007, 2009–2010, and 2012).

Clam Tissue

Metal concentrations in the soft tissues of *Macoma petalum* reflect a combination of metal exposures from water and food and the diluting and concentrating effects of gaining and losing tissue mass. Exposures to Cu and Ag at Palo Alto are of special interest because of the high tissue concentrations observed at this site in the past (figs. 10 and 11, appendix 7). During 1977–1987, the ranges in annual concentrations of Cu and Ag were 95–287 mg/kg and 45–106 mg/kg, respectively. Since 1987, concentrations have been considerably lower, 24–71 mg/kg for Cu and 1.8–20 mg/kg for Ag. Concentrations have been particularly low and stable since 1997, except for a 2-year period in 2005–2006 when concentrations increased modestly. Annual mean concentrations of Cu and Ag for 2013 were 28.2±4.7 mg/kg and 2.38±0.47 mg/kg, respectively; these are within the range of concentrations observed since 1997.

Intra-annual variations in Ag and Cu concentrations in clam soft tissues display a consistent seasonal signal characterized by fall/winter maxima and spring/summer minima. The amplitude of this seasonal cycle varies from year to year. For example, the winter maxima and the magnitude of seasonal Ag and Cu concentrations during 1994–1997 and in 2007 were relatively large and bracketed years of less variability (figs. 12 and 13). These trends most likely reflect the interaction of the changing exposure regime of the site (the long-term decline in metal concentrations) with the annual growth cycle of *M. petalum* (Cain and Luoma, 1990). In 2013, Ag and Cu concentrations were highest in January (4.81 mg/kg and 53.1 mg/kg, respectively). Concentrations decreased over the winter and reached their annual minima of 0.825 mg/kg Ag and 10.9 mg/kg Cu in April (table 2). Both Cu and Ag concentrations followed a typical season pattern—low through the summer (June) and thereafter steadily increasing.

As with Cu and Ag, tissue concentrations of Cr (fig. 14, table 2), Ni (fig. 15, table 2), and Zn (fig. 16, table 2) also exhibited seasonal cycles. The seasonal cycles of Cr and Ni were very similar in terms of their timing and magnitude throughout the record (1994–2013). Maximum concentrations occurred in the winters of 1996–1997 and 2006–2007, whereas 2000–2002 was a period of relatively low winter-maximum concentrations. However, neither element exhibited a clear temporal trend (either downward or upward) in concentration. The annual average chromium concentration in 2013 was the lowest value in the historical record at 1.60 mg/kg. The annual average concentration of Ni (4.60 mg/kg) was well below the record average for Ni (5.31 mg/kg), but higher than the 2012 record low annual average (3.84 mg/kg). Maximum concentrations of Cr and Ni occurred in winter of 2013: Ni in January (6.92 mg/kg) and Cr in February (2.67 mg/kg). Typical seasonal patterns of Zn concentrations were evident in most years, yet weakly expressed in 2006, 2010–2011, and 2013. Zinc concentrations were notably highest throughout the year during 1994–1997. Zinc exhibited a slight long-term decline of seasonal maxima through 2005, but was high again in 2006. Concentrations of Zn over the past 6 years (2008–2013) are similar to the relatively lower concentrations observed during 2000–2005. In 2013, Zn began the year at a maximum of 328 mg/kg in January, then decreased to a near-minimum concentration of 181 mg/kg in April, increased to 238 mg/kg in May, but then steadily decreased to the annual minimum of 171 mg/kg in December. Wellise and others (1999) observed that seasonal and interannual patterns of Cr, Ni, and Zn in *M. petalum* at Palo Alto were generally similar to those observed near the San Jose/Santa Clara Water Pollution Control Plant, indicating that regional-scale processes may be more important than treatment plant inputs in controlling the bioavailability of these elements.

The average Hg concentration in *M. petalum* in 2013 was the third lowest in the historical record at 0.22 mg/kg (fig.17, table 2). Seasonal variation of Hg in 2013 displayed the typical pattern of fall/winter maxima and spring/summer minima, with a maximum of 0.29 mg/kg in January and a minimum of 0.13 mg/kg in April. Concentrations then increased throughout the rest of the year. A long-term trend in Hg concentration is not evident (fig. 17a).

Selenium concentrations in *M. petalum* in 2013 displayed the typical seasonal pattern of higher concentrations in the winter and lower concentrations in spring/summer. Selenium was highest in February (5.05 mg/kg) then declined to the annual minima from April– September (3.22–3.53 mg/kg) (fig. 18, table 2). Average Se concentrations had shown an increasing trend from 2007 to 2010, but average selenium concentrations in 2011–2013 were the lowest since 2002–2003, reversing the trend. Long-term trends in the Se data are not evident.

Data on the condition index (CI) for *M. petalum* at Palo Alto extends back to 1988 (fig. 19, table 2). As previously discussed, the data fluctuate seasonally in relation to growth and reproductive cycles, and annual cycles differ in magnitude. Condition index values typically build to a maximum in the spring (pre-spawning period) then taper off throughout the year (fig. 19C). The seasonal variation of CI in 2013 was typical for the record, and the annual average CI (136 mg) was consistent with values observed since 2007. The maximum CI in April (231 mg) coincided with the minima for most observed metal concentrations in *M. petalum* in 2013, and the minimum CI (91 mg in January) coincided with the maxima for most observed metal concentrations in *M. petalum*.

Reproduction of *Macoma petalum*

Earlier studies (Hornberger and others, 2000b; Shouse and others, 2004) found that low reproductive activity in *M. petalum* in the late 1970s coincided with highly elevated concentrations of Ag (and perhaps Cu) in the soft tissues. During this period, *Macoma* exhibited extended periods (as long as 2 months) of reproductive inactivity. Following the decline in tissue concentrations of Ag and Cu in the 1980s, reproductive activity of *M. petalum* improved (fig. 20). Furthermore, the low reproductive activity observed during the late 1970s has not been observed during the entire period of reduced metal exposures. The temporal coincidence of these events indicates that reproductive activity was related to the concentration of metals in the animal. This finding has implications for the reproductive success of the population.

The time series of reproductive activity (figs. 20 and 21) shows that *M. petalum* continues to be highly reproductive relative to the 1970s with a high percentage of the animals being reproductively active at any one time during the normal seasonal cycle of reproduction. That cycle begins in fall, with spawning occurring the following spring (see table 3 for detailed reproduction data for 2013). A closer look at the last 6 years of data demonstrates this seasonality of reproduction—unlike the earlier periods, animals do not stay reproductively inactive for longer than a month or two.

Benthic Community

Estimates of species diversity and total animal abundance are simple metrics that are used in assessing environmental stress on biological communities. Species diversity at the Palo Alto site, as estimated by a time series of number of species, continues to show an upward trend (with one exception) since the last very wet year in 1998 (fig. 22). Total animal abundance has varied significantly during the sampling period (fig. 23). The difficulty with these types of metrics is that they do not consider the possibility that one species can take the place of another or that high abundance is based on one species. Depending on the characteristics of a species new to the community or newly dominant in the community, the community structure and function may change as a result of this change in species composition or dominance. The details of changes in species composition are important because they may reflect the relative ability of species to accommodate environmental stress and redistribute site resources. In general, the species composition at the study site has changed little since 1998, although there have been seasonal eruptions of several species in some years.

Three common bivalves (*Macoma petalum, Mya arenaria*, and *Gemma gemma*) have not shown any consistent trend over the 40-year period from 1974 to 2013 (figs. 24, 25, and 26). Significant seasonal and interannual variability has been displayed in species abundances for all species found at the Palo Alto site. The three common bivalves illustrate this variability well; *Gemma gemma* has been particularly volatile since 2005. *Gemma gemma* abundance dropped to near zero in late fall 2007 and has not regained its previous high density, although the abundance has now rebounded to levels seen in the early 2000s.

Six species have shown trends in their abundance since the 1970s, and these trends continued through 2013. The first species, *Ampelisca abdita*, is a small crustacean that lives above the surface of the mudflat in a tube built from selected sediment particles. *A. abdita* showed a general decline in abundance (fig. 27) after 1998. That pattern mostly continues through today. The second species to show a significant trend is the small polychaete worm *Streblospio benedicti*, which also builds a tube above the surface of the mudflat. As with *A. abdita*, *S. benedicti* abundance has declined through the study years and, over the past 5 years, the species has settled into a seasonal pattern of increasing fall abundance followed by a winter decline (fig. 28). The abundance of the small burrowing crustacean *Grandiderella japonica*, a deposit feeder, became more seasonally consistent after 2000 (fig. 29), although the population abundance was particularly low in 2011. This species has shown a consistent peak in abundance in the fall since 1999 with the exception of 2011. *Neanthes succinea*, a burrowing polychaete that feeds on surface deposits and scavenges for detrital food, showed large seasonal fluctuations in abundance through the 1980s. *N. succinea* abundance became more stable in the late 1990s and remained so until 2005, when the abundance decreased (fig. 30). *N. succinea* abundance declined further in 2011. In 2012 and 2013, *N. succinea* abundance rebounded to pre-2011 numbers. Two species showed an increase in abundance within the time series. The first was the polychaete worm *Heteromastus filiformis* (fig. 31), a sub-surface deposit feeding, burrowing species that lives deep in the sediment (usually 5–20 cm below the surface of the mudflat). Abundance increased sharply in 1985 and then partially receded in the late 1980s. Abundance remained higher than in the late 1970s until 2008, when there was a large decline in *H. filiformis* abundance (fig. 31). *H. filiformis* abundance increased slightly in 2011, 2012 and 2013; 1980s abundance data indicate that this is a species that increases slowly, possibly because of their egg-laying mode of recruitment. The second species showing an increase was *Nippoleucon hinumensis*, a small, burrowing, surface-deposit-feeding crustacean, which appeared in the dataset in 1988 (fig. 32) following its introduction into the bay in 1986 (Cohen and Carlton, 1995). Another nonindigenous species, *Potamocorbula amurensis,* a filter-feeding bivalve that first appeared in the benthic community in significant numbers in April 2005 and persisted into 2006 with peaks in abundance occurring in spring and fall, has continued to be present in 2013. A complete list of the benthic species found at the Palo Alto site in the year 2013 is shown in appendix 8. The benthic species name changes (as of 2013) for appendix 8 are shown in appendix 9.

A sudden drop in animal abundance was observed in February 2008. Very few animals were found at the site, and the mudflat community was evidently stressed by some event between the January and February sampling. Possible causes of the stress include sedimentation or freshwater inundation. There was a large storm on January 25, 2008, with rainfall rates exceeding 0.5 cm/hr for more than half the day, including during the low-tide period. No obvious changes in the sediment surface were observed, but sediment changes can occur and be incorporated quickly in this tidal environment. Other possible causes of benthic community death or exodus include a toxic event or anoxia. It is unlikely that either of these occurred, because *M. petalum* were present in the deep sediment in February 2008, and animals were found again at the site in March 2008. This would not happen with toxicity or anoxia. The timeline for recovery from anoxia can be estimated on the basis of observations following an anoxic event at this site in 1975. Macroalgae were deposited on the mudflat surface and began to decay, and the resulting bacterial consumption of oxygen led to anoxia. The benthic community took many months to recover from this anoxic event. Animals that returned after the disturbance in 2008 include those species with pelagic larvae and mobile adults, as would be expected. Nonmobile brooders returned to the site in 2009, concurrent with an increase in abundance of the brooding clam *G. gemma* and the brooding polychaete *Streblospio benedicti*. This trend continued into 2013, when brooders and oviparous species were more than half of the top 10 most abundant species (fig. 39).

As stated earlier, multivariate analyses of population data of the dominant species with environmental parameters did not reveal any relations, except with the concentration of Ag and Cu in the sediment and in the tissue of *M. petalum* (using data reported by David and others, 2002). Therefore, this update will consider only those metals. Metal concentration and abundance of species with the most susceptible mode of feeding and reproduction will be compared over the period of the study. One such susceptible species, the worm *H. filiformis*, increased in abundance with the decrease in Ag and Cu until 2008 (fig. 31). This was of interest because *H. filiformis* has continual tissue contact with the sediment at the exterior of its body, as well as within its body, as a result of its lifestyle of burrowing through the sediment and consuming a diet of mud and organic particles. In addition, this is one of the few species in the present community that reproduces exclusively by laying its eggs in the sediment. The larvae hatch after 2 or 3 days and spend a very short period (2–3 days) in the plankton before settling back to the mud as juvenile worms (Rasmussen, 1956). The short planktonic period limits the species' speed of expansion into new areas. The authors hypothesize that once a few individuals successfully arrived at the study site, *H. filiformis* increased in abundance because either the adult worms or the eggs now found the environment agreeable. This species is not likely to move into an area quickly after an environmental stressor because of its mode of reproduction and short planktonic larval period. A large spike in *H. filiformis* abundance was observed in January 2008 because of the settling of larvae, but these larvae did not survive the event that occurred before the February sampling. So far, the species has not returned in high numbers to the study site. The dynamics of recovery for this species will continue to be monitored closely.

Two species that have shown the opposite trend of *H. filiformis*, the crustacean *A. abdita* (fig. 27) and the worm *S. benedicti* (fig. 28), have declined in abundance coincident with the decline in metals. These species have very similar life-history characteristics that make them less susceptible to high Ag and Cu concentrations in sediment. Both species live on the surface of the sediment in tubes that are built from sediment particles. They feed on particles in the water column or on particles that have settled to the sediment surface, brood their young, and produce young that are capable of either swimming or settling upon hatching. These opportunistic characteristics make these species ideal for invading a disturbed or stressed environment; thus, they are capable of rapid increase in population size and distribution. It is not surprising that both species immediately responded to the near-empty community in February 2008 and have subsequently declined (figs. 27 and 28). This abundance pattern is consistent with what is expected of an opportunistic species.

Other species share the characteristics highlighted in our discussion of *H. filiformis, S. benedicti*, and *A. abdita*; the species with similar characteristics have been combined into plots that examine the percent of abundance represented by each feeding and reproductive mode (figs. 33–36). Because the natural spatial variability (that is, the large standard deviations around the monthly means) and seasonal variability of invertebrate abundance can be quite large, the average percentages for the month of August of each characteristic reproductive and feeding mode are shown in figures 33–36. To interpret these plots, the life-history characteristics must first be examined to determine if there is some mechanism by which this organism could be responding to a decrease in Ag or Cu in the environment. It is likely that Ag, but probably not Cu, adversely affected reproduction of all animals, but it was most obvious in species with juveniles that were not transported from outside the area as pelagic larvae. Therefore, the species having oviparous and mixed (species capable of oviparity and brooding) reproductive modes (Ahn and others, 1995; Hornberger and others, 2000b) are worth examining in the early years of the study. The gradual increase in abundance of this group through 1983 occurred concurrent with the gradual reduction of metals in the environment during that time. In the present environment of much lower metal concentrations, these species respond to a variety of stresses, and their percentage of brooding and oviparous individuals in the community reflects those stresses (figs. 33 and 34). Although the percentage is volatile, it never stays as low as it was in the early 1970s. The authors interpret this as being a reflection of the general health of the benthic environment. In a similar manner, we can examine the feeding modes of the majority of the individuals (figs. 35 and 36). High concentrations of either Cu or Ag are unlikely to be healthy for species that ingest the sediment in order to consume the interstitial and attached carbon; thus, it is reasonable to expect species that consume particles from the water column to be more protected from the contaminants in the sediment. Filter feeding species are usually the dominant group throughout the data-set, and the subsurface deposit feeders are the group that shows the largest increase in dominance after the 1970s. This is consistent with the conceptual model posed here.

The change in function of the benthic community over time can be examined by ranking the top 10 species by abundance and plotting the log (n + 1) of mean abundance against the rank of each species. The plot for 2013 (fig. 37) is indicative of a healthy benthic community with species dominance, as revealed by abundance, not showing large differences among the top 10 species. An examination of similar plots for August of three hydrologically dry years during this study (1977, 1989, and 2002) shows that the shape of the curve has changed greatly and that the curve is now similar to that seen in 2002. The series of lines shows a community that was heavily dominated by three species in 1977 and 1989, and a community with 1–2 dominant species in 2002 and 2013. The 1977 community plot is the most extreme and reflects a bimodal species distribution, with three species dominating the community and the remainder having similar but relatively low abundances. The 1977 community was dominated by opportunistic species. In contrast, the 2013 community plot is dissimilar because the opportunists are present but display less dominance in the community (the most opportunistic species ranks fourth in 2013).

It is informative to examine the rank-abundance plots within the context of the life-history characteristics of each species to determine if shifts in plot shape coincide with a shift in community structure and function that might be indicative of a healthier environment. Two critical life history characteristics are shown: feeding mode in fig. 38 and reproductive mode in fig. 39. The 1977 community was dominated by filter feeding species (species that consume particles in the water column), species that have the option of either filter feeding or feeding on the sediment surface (mixed feeders), and two species that feed on food particles on the sediment surface. In 1989, the species composition had shifted such that filter feeding species, subsurface deposit and surface deposit feeding species (those that ingest sediment and strip the food off of the sediment in their gut) dominated the community. In 2002, a shift was observed towards species that could either filter feed or deposit feed (mixed feeders) and those species that feed on subsurface sediment. The most recent data (2013) show the community to be almost evenly composed of a mix of surface deposit feeding species and filter feeding species. Over the period of this study, a shift has occurred from a community dominated by species that feed either in the water column or on recently settled food particles on the sediment surface to a mixed community of species that feed directly on the subsurface sediment, those capable of feeding in the water column, and those feeding on the sediment surface. The species that returned following the defaunation event in January/February 2008 have maintained this pattern. Thus, it is unlikely that any sediment-borne pollutant caused the collapse of the community in early 2008.

An examination of these rank-abundance plots using reproductive mode as the descriptor for each point is equally informative (fig. 39). The dominant species in 1977 were species that brood their young and release fully functional juveniles into the environment. In 1989, there were still several brooders, but there were also two species that lay their eggs in the sediment. Although brooding species remain in the 10 most abundant species, species that spawn their gametes into the water column in combination with those that lay eggs in the sediment (oviparous) have equal presence. It is possible that some of the metal contaminants found in the sediment in the 1970s at this location limited the success of species that consumed the sediment for food, laid eggs in the sediment, or depended on water-borne larvae to repopulate the community. The reproductive mode of most species present in 2013 is brooding with some egg laying (oviparous).

Summary

Long-Term Observations

Since 1974, USGS personnel have monitored and conducted basic research on the benthic sediments and biological community in the vicinity of the discharge of the Palo Alto Regional Water Quality Control Plant (PARWQCP). The time series presented here update previous findings (for example, Hornberger and others, 2000a; Luoma and others, 1991, 1995a, 1996; Moon and others, 2005; Shouse and others, 2003, 2004; Thompson and others, 2002; Cain and others, 2006; Dyke and others, 2011, 2012) with additional data from January 2013 through December 2013 to create a record spanning 40 years. This long-term dataset includes sediment chemistry and tissue concentrations of metals (1977–2013 for Cu and Ag, 1994–2013 for other metals), condition index (1988–2013), and reproductive activity in *M. petalum* and population dynamics of benthic invertebrate species (1974–2013). The time series encompasses the period when exceptionally high concentrations of Cu and Ag were found in *M. petalum* (1970s) and the subsequent period when those concentrations declined. The sustained record of biogeochemical data at this site provides a rare opportunity to examine the biological response to metal contamination within this ecosystem.

Studies during the 1970s showed that sediments and *M. petalum* at the Palo Alto site contained highly elevated levels of metals, especially Ag and Cu, as a result of metal-containing effluent being discharged from the Palo Alto Regional Water Quality Control Plant (PARWQCP) to South San Francisco Bay. In the early 1980s, the point-source metal loading from the nearby PARWQCP was significantly reduced as a result of advanced treatment of influent and source mitigation. Coincident with declines in metal loadings, concentrations of metals in the sediment and in the clam *M. petalum* (serving as a biomonitor of metal exposures) also declined, as previously described by Hornberger and others (2000b). Interannual trends in clams and sediments were highly correlated with Cu loadings from PARWQCP (concurrent loading data for Ag were not available). Metal levels in sediments and clams responded relatively quickly to changes in metal loading; the reduction in metal loadings by the PARWQCP resulted in a reduction in metal concentrations in both the sediment and *M. petalum* within a year (Hornberger and others, 2000b).

Biological responses to metal inputs to South San Francisco Bay were assessed at different levels of organization. These responses are interpreted within the appropriate temporal context. Because metal exposures were already high when the study began, interpretations are based on observed changes in biological attributes as metal inputs declined. In general, discernable responses at the organism level (that is, reproductive activity, a manifestation of a cellular or physiological change) to metal exposure may occur within a relatively short time, whereas population and community level responses take longer to develop. Stable changes in the benthic community may take a relatively long period of time to be expressed because of the normally high degree of intra-annual variability of benthic community dynamics, which reflects the cumulative response to natural and anthropogenic disturbances. It is therefore critical that sampling frequency and duration be conducted at temporal scales appropriate to characterize the different biological responses.

During the first 10 years of this study, when the metal concentrations were high and declining, the benthic community was largely composed of nonindigenous, opportunistic species that dominated because of their ability to survive the many physical disturbances on the mudflat (Nichols and Thompson, 1985a, 1985b). These disturbances included sediment erosion and deposition and aerial exposure at extreme low tides, as well as less well defined stresses. The possible effects of metal exposure as a disturbance factor were not considered in the analyses by Nichols and Thompson because the decline in metal concentrations in *M. petalum* and sediment had just begun.

However, data collected throughout the period of declining metal exposure have revealed biological responses to this metal decline. Reproductive activity improved within a year or two of reduced metal exposure, and responses at the population and community levels were observed afterward. Identification of these responses was possible because the frequency of sampling allowed long-term trends related to metal contamination to be identified within the context of repeating seasonal cycles and unrelated intra-annual variation.

The ecology of the Palo Alto mudflats is part of the larger South San Francisco Bay, which has been undergoing some changes in recent years. During 1999–2005, USGS scientists noticed an increase in phytoplankton biomass in the southern bay. Sampling in the deeper water of the southern bay showed that the bivalves were mostly absent from the system during this increase in primary production. Cloern and others (2007) indicate that the cause of the decline in bivalves was an increase in fish predators resulting from increased offshore upwelling activity. The higher reproductive success of demersal fish, crabs, and shrimp during this period resulted in a higher number of juveniles moving into the South San Francisco Bay to grow. Since 2005, scientists have seen the large bivalve populations fluctuate more than in previous years, and these fluctuations have been reflected in changes in phytoplankton biomass in the system (primarily through an increase in phytoplankton biomass in late summer and fall). The value of these findings in greater South San Francisco Bay to this study is twofold. First, it reinforces the importance of the benthic community in structuring the ecosystem function. Second, it shows that the high intertidal community at the Palo Alto site has not been demonstrably affected by these greater South San Francisco Bay influences during these years. This finding solidifies the authors' confidence that the changes observed in the benthic community are in large part due to local factors.

2013 Observations

Throughout 2013, Cu and Ag concentrations in sediments and soft tissues of the clam, *M. petalum*, remained representative of the concentrations observed since 1991, following the significant reductions in concentrations during the 1980s that coincided with reductions in the discharge of these elements from PARWQCP. Since 1991, annual mean Cu and Ag concentrations have fluctuated modestly and without any extended trends. This is also true for other elements. For example, sedimentary selenium (Se) concentrations have been variable from year to year and showed no sustained temporal trend. Selenium concentrations in surface sediment declined in 2009–2010 from the record high concentrations observed in 2008, more than doubled during 2011, and in 2012–2013 decreased to about half the annual average observed in 2008. In another example, annual average concentrations of Cu and Ag in *M. petalum* were relatively low from 1997 to 2005, increased notably in 2006–2007, and returned to 1997–2005 levels during 2008–2013. Silver and Cu annual averages in *M. petalum* (as well as Cu in sediment) were slightly lower than 2012 concentrations, whereas annual average Ag in sediments has been steadily increasing since 2010. The most recent results (2013) show that Ag and Cu in *M. petalum* are only 2 percent and 12 percent, respectively, of the maximum values observed during 1978–1980. Concentrations of Ag and Cu in sediments in 2013 were 22 percent and 40 percent, respectively, of the record high concentrations observed in 1979. Interannual variation in Cu and Ag in *M. petalum* from 1991 to 2006 did not correlate with discharge of Cu and Ag from PARWQCP (Lorenzi and others, 2007), indicating that, as with other elements of regulatory interest, including Cr, Ni, and Zn, regional-scale factors now largely influence sedimentary and bioavailable concentrations (see, for example, Luoma and others, 1998). Abiotic factors that affect the seasonal and year-to-year patterns in sedimentary and tissue concentrations may include precipitation, nonpoint-source runoff, cycling of legacy contamination, accelerated erosion of salt marsh banks in recent years, and periods of accretion and erosion of sediment on the mudflat.

The long-term dataset demonstrates various adverse effects of contaminants on benthic organisms. Decreasing particulate concentrations of trace metals in the local environment have benefited resident populations of invertebrates, as evidenced by increased reproductive activity in *M. petalum* that has been sustained through 2013. In early 2008, the benthic community declined, with few animals present in February. This decline was likely the result of a natural stressor, such as a sedimentation or freshwater event, and the composition of the benthic community supports that supposition. Mobile animals such as *M. petalum* that were capable of burrowing down to avoid the stressor probably did so, but many other species either relocated or were killed. This natural disturbance gives scientists the opportunity to observe mudflat community recovery from a natural stressor and to compare this recovery to that observed during the long-term decline in metals. Shifts in species abundance at Palo Alto have been interpreted to be a response to decreasing sediment contaminants. These community changes have included a shift from species that live on the surface, filter food out of the water column or consume particles on the sediment surface, and brood their young to a community dominated by species that live on and below the surface, consume the sediment directly to harvest food particles, and spawn and lay eggs in the sediment. The 2008–2013 data reveal a community that had a short-term physical stressor but not one that was subject to unhealthy sediment. In 2013, the species abundance data continue to show signs of a community that has recovered and stabilized, and the constancy of functional groups reflects the stability of the ecosystem. This “natural experiment” has given USGS scientists a great opportunity to test various hypotheses on the benthic community response to different stressors. Future data will further refine the understanding of the response of this benthic community to natural and anthropogenic stressors.

Value of Long-Term Monitoring

This study highlights the importance of long-term ecosystem monitoring. The decadal time series produced during the course of sustained efforts at this site have made it possible to describe trends, identify previously undocumented phenomena, and pose hypotheses that have guided past detailed explanatory studies and can guide future studies. Monitoring studies cannot always unambiguously determine the causes of trends in metal concentrations or benthic community structure. The strength and uniqueness of this study is the integrated analysis of metal exposure and biological response at intra- and interannual time scales over multiple decades. Changes and trends in community structure that may be related to anthropogenic stressors, as was seen in this study, can be established only with a concerted and committed effort of sufficient duration and frequency of sampling. Such rare field designs allow biological responses to natural stressors to be characterized and separated from those to stressors introduced by humans. Through interpreting time-series data, it has been possible to separate anthropogenic effects from natural annual and interannual variability. The data from the recent record (that is, within the past decade) increasingly appear to be indicative of an integrated regional ecological baseline with indicators of metal contamination and greater physiological well-being of aquatic life and benthic community structure. Changes are occurring in the South San Francisco Bay watershed. For example, implementation is beginning in the South Bay Salt Ponds Restoration Program; with unknown implications (positive or negative) for all of South San Francisco Bay. Nanomaterials, many of which include metal-based products in forms for which environmental researchers have little or no experience, are being used in an increasing number of common consumer products. The long-term, detailed, integrated ecological baseline that has been established at this sampling site will be uniquely valuable in assessing the response of the environment as human activities in the watershed continue to change.

Selected References

Ahn, I.Y., Kang, Y.C, and Choi, J.W., 1995, The influence of industrial effluents on the intertidal benthic communities in Panweol, Kyeonggi Bay (Yellow Sea) on the west coast of Korea: Marine Pollution Bulletin, v. 30, p. 200–206.

Alpine, A.E., and Cloern, J.E., 1992, Trophic interactions and direct physical effects control phytoplankton biomass and production in an estuary: Limnology and Oceanography, v. 37, p. 946–955.

Begon, M., Harper, J.L., and Townsend, C.R., 1986, Ecology; individuals, populations and communities: Sunderland, Massachusetts, Sinauer Associates Inc., 876 p.

Cain, D.J., and Luoma, S.N., 1990, Influence of seasonal growth, age and environmental exposure on Cu and Ag in a bivalve indicator, *Macoma balthica* in San Francisco Bay: Marine Ecology Progress Series 60, p. 45–55.

Cain, D.J., Parchaso, F., Thompson, J.K., Luoma, S.N., Lorenzi, A.H., Moon, E., Shouse, M.K., Hornberger, M.I., and Dyke, J.L., 2006, Near-field receiving water monitoring of trace metals and a benthic community near the Palo Alto Regional Water Quality Control Plant in South San Francisco Bay, California; 2005: U.S. Geological Survey Open-File Report 2006–1152, 128 p.

Cain, D.J., Thompson, J.K., Dyke, J.L., Parchaso, F., Luoma, S.N., and Hornberger, M.I., 2009, Near-field receiving water monitoring of trace metals and a benthic community near the Palo Alto Regional Water Quality Control Plant in South San Francisco Bay, California; 2008: U.S. Geological Survey Open-File Report 2009–1193, 120p.

Childress, C.J.O., Foreman, W.T., Connor, B.F., and Maloney, T.J., 1999, New reporting procedures based on long-term method detection levels and some considerations for interpretations of water-quality data: U.S. Geological Survey Open-File Report 99–193, 19 p.

Cloern, J.E., Jassby, A.D., Thompson, J.K, and Hieb, K., 2007, A cold phase of the east Pacific triggers new phytoplankton blooms in San Francisco Bay: Proceedings National Academy of Science, v. 104, no. 47, p. 18561–18565.

Cohen, A.N., and Carlton, J.T., 1995, Biological study, nonindigenous aquatic species in a United States estuary; a case study of the biological invasions of the San Francisco Bay and Delta: Washington, D. C., U.S. Fish and Wildlife Service.

David, C.P.C., Luoma, S.N, Brown, C., Cain, D.J, Hornberger, M., and Lavigne, I.R., 2002, Near-field receiving water monitoring of trace metals in clams (*Macoma balthica*) and sediments near the Palo

Alto Water Quality Control Plant in South San Francisco Bay, California; 1999–2001: U. S. Geological Survey Open-File Report 02–453, 105 p.

Dyke, J.L., Parchaso, F., Thompson, J.K., Cain, D.J., Luoma, S.N., and Hornberger, M.I., 2010, Near-field receiving water monitoring of trace metals and a benthic community near the Palo Alto Regional Water Quality Control Plant in South San Francisco Bay, California; 2009: U.S. Geological Survey Open-File Report 2010-1188, 142 p.

Dyke, J.L., Parchaso, F., Thompson, J.K., Cain, D.J., Luoma, S.N., and Hornberger, M.I., 2011, Near-field receiving water monitoring of trace metals and a benthic community near the Palo Alto Regional Water Quality Control Plant in South San Francisco Bay, California; 2010: U.S. Geological Survey Open-File Report 2011-1163, 114 p.

Dyke, J.L., Parchaso, F., Thompson, J.K., Cain, D.J., Luoma, S.N., and Hornberger, M.I., 2012, Near-field receiving water monitoring of trace metals and a benthic community near the Palo Alto Regional Water Quality Control Plant in South San Francisco Bay, California; 2011: U.S. Geological Survey Open-File Report 2012-1165, 108 p.

Elrick, K.A., and Horowitz, A.J., 1985, Analysis of rocks and sediments for arsenic, antimony, and selenium, by wet digestion and hydride generation atomic absorption: U.S. Geological Survey Open-File Report 85-497, 14 p.

Glaser, J.A., Foerst, D.L., Mckee, G.D., Quave, S.A., and Budde, W.L., 1981, Trace analyses for wastewaters: Environmental Science and Technology, v. 15, no. 12, p. 1426–1435.

Harris, D., Horwáth, W.R., and van Kessel, C., 2001, Acid fumigation of soils to remove carbonates prior to total organic carbon or CARBON-13 isotopic analysis: Soil Science Society of America Journal, v. 65, p. 1853–1856.

Hornberger, M.I., Luoma, S.N., Van Geen, A., Fuller, C.C., and Anima, R., 1999, Historical trends of metals in the sediments of San Francisco Bay, California: Marine Chemistry, v. 64, p. 39–55.

Hornberger, M.I., Luoma, S.N., Cain, D., Parchaso, F., Brown, C., Bouse, R., Wellise, C.J., and Thompson, J., 2000a, Bioaccumulation of metals by the bivalve *Macoma balthica* at a site in South San Francisco Bay between 1977 and 1997; long-term trends and associated biological effects with changing pollutant loadings: U. S. Geological Survey Open-File Report 99–55, 42 p.

Hornberger, M., Luoma, S., Cain, D., Parchaso, F., Brown, C., Bouse, R., Wellise, C., and Thompson, J., 2000b, Linkage of bioaccumulation and biological effects to changes in pollutant loads in South San Francisco Bay: Environmental Science and Technology, v. 34, p. 2401–2409.

Kennish, J.K., 1998, Pollution impacts on marine biotic communities: New York, CRC Press, 310 p.

Liber, K, 2011, Cold Digestion of Invertebrates for the Selenium Project. Method developed C.W. and E.F. 2007. Revised Mar 2011 MK: Water Quality Laboratory, Toxicology Center, University of Saskatchewan.

Long, E.R., MacDonald, D.D., Smith, S.L., and Calder, F.D., 1995, Incidence of adverse biological effects within ranges of chemical concentrations in marine and estuarine sediments: Environmental Management, v. 19, p. 81–97.

Lorenzi, A., Cain, D.J., Parchaso, F., Thompson, J.K., Luoma, S.N., Hornberger, M.I., Dyke, J.L., Cervantes, R., and Shouse, M.K., 2007, Near-field receiving water monitoring of trace metals and a benthic community near the Palo Alto Regional Water Quality Control Plant in South San Francisco Bay, California; 2006: U.S. Geological Survey Open-File Report 2007–1199, 121 p.

Lorenzi, A., Cain, D.J., Parchaso, F., Thompson, J.K., Luoma, S.N., Hornberger, M.I., and Dyke, J.L., 2008, Near-field receiving water monitoring of trace metals and a benthic community near the Palo Alto Regional Water Quality Control Plant in South San Francisco Bay, California; 2007: U.S. Geological Survey Open-File Report 2008–1180, 120 p.

Luoma, S.N., Cain, D.J., and Johansson, C., 1985, Temporal fluctuations of silver, copper and zinc in the bivalve *Macoma balthica* at five stations in South San Francisco Bay: Hydrobiologia, v. 129, p. 109–120.

Luoma, S.N., Cain, D.J., Brown, C. and Axtmann, E.V., 1991, Trace metals in clams (*Macoma balthica*) and sediments at the Palo Alto mudflat in South San Francisco Bay; April, 1990–April, 1991: U. S. Geological Survey Open-File Report 91–460, 47 p.

Luoma, S.N., Cain, D.J., Brown, C., and Hornberger, M., 1992, Trace metals in clams (*Macoma balthica*) and sediments at the Palo Alto mudflat in South San Francisco Bay; May 1991–May 1992: U. S. Geological Survey Open-File Report 92–456, 51 p.

Luoma, S.N., Cain, D.J., Brown, C., and Hornberger, M., 1993, Trace metals in clams (*Macoma balthica*) and sediments at the Palo Alto mudflat in South San Francisco Bay; June 1992–June 1993: U. S. Geological Survey Open-File Report 93–500, 52 p.

Luoma, S.N., Cain, D.J., Brown, C., and Hornberger, M., 1995a, Near-field receiving water monitoring of trace metals in clams (*Macoma balthica*) and sediments near the Palo Alto and San Jose/Sunnyvale Water Quality Control Plants in South San Francisco Bay; June 1993–October 1994: U. S. Geological Survey Open-File Report 95–299, 83 p.

Luoma, S.N., Ho, Y.B, and. Bryan, G.W., 1995b, Fate, bioavailability and toxicity of silver in estuarine environments: Marine Pollution Bulletin, v. 31, p. 44–54.

Luoma, S.N., Cain, D.J., Brown, C., Hornberger, M., and Bouse, R. 1996, Near-field receiving water monitoring of trace metals in clams (*Macoma balthica*) and sediments near the Palo Alto and San Jose/Sunnyvale Water Quality Control Plants in South San Francisco Bay; December 1994–December 1995: U. S. Geological Survey Open-File Report 96–203, 90 p.

Luoma, S.N., Cain, D.J., Brown, C., Hornberger, M., and Bouse, R. 1997, Near-field receiving water monitoring of trace metals in clams (*Macoma balthica*) and sediments near the Palo Alto and San Jose/Sunnyvale Water Quality Control Plants in South San Francisco Bay; 1996: U. S. Geological Survey Open-File Report 97–585, 91 p.

Luoma, S.N., Wellise, C, Cain, D.J., Brown, C., Hornberger, M., and Bouse, R., 1998, Near-field receiving water monitoring of trace metals in clams (*Macoma balthica*) and sediments near the Palo Alto and San Jose/Sunnyvale Water Quality Control Plants in South San Francisco Bay; 1997: U. S. Geological Survey Open-File Report 98–563, 88 p.

Moon, E., David, C.P.C., Luoma, S.N, Cain, D. J, Hornberger, M., and Lavigne, I.R., 2003, Near field receiving water monitoring of trace metals in clams (*Macoma balthica*) and sediments near the Palo Alto Water Quality Control Plant in South San Francisco Bay, California; 2002: U.S. Geological Survey Open-File Report 03–339, 78 p.

Moon, E., Luoma, S.N, Cain, D.J., Hornberger, M., and David, C.P.C., 2004, Near field receiving water monitoring of trace metals in clams (*Macoma balthica*) and sediments near the Palo Alto Water Quality Control Plant in South San Francisco Bay, California; 2003: U.S. Geological Survey Open-File Report 2004–1213, 84 p.

Moon, E., Shouse, M.K., Parchaso, F. Thompson, J.K., Luoma, S.N., Cain, D.J. and Hornberger, M.I., 2005, Near-field receiving water monitoring of trace metals and a benthic community near the Palo Alto Regional Water Quality Control Plant in South San Francisco Bay, California; 2004: U.S. Geological Survey Open-File Report 2005–1279, 118 p.

Morrisey, D.J., Underwood, A.J., and Howitt, L., 1996, Effects of copper on the faunas of marine soft-sediments; an experimental field study: Marine Biology, v. 125, p.199–213.

Nichols, F.N., and Thompson, J.K., 1985a, Persistence of an introduced mudflat community in South San Francisco Bay, California: Marine Ecology Progress Series, v. 24, p. 83–97.

Nichols, F.N., and Thompson, J.K., 1985b, Time scales of change in the San Francisco Bay benthos: Hydrobiologia, v. 129, p. 121–138.

Parchaso, F., 1993, Seasonal reproduction of *Potamocorbula amurensis* in San Francisco Bay, California: San Francisco, California, San Francisco State University, M.S. thesis.

Rasmussen, E., 1956, The reproduction and larval development of some polychaetes from the Isefjord, with some faunistic notes: Biologiske Meddelelser, published by det Kongelige Danske Videnskabernes Selskab, v. 23, no. 1, p. 1–84.

Rygg, B., 1985, Effect of sediment copper on benthic fauna: Marine Ecology Progress Series, v. 25, p. 83–89.

San Francisco Estuary Institute, 1997, RMP, regional monitoring program for trace substances; 1996: Richmond, California, San Francisco Estuary Institute, 349 p.

Shouse, M.K., 2002. The effects of decreasing trace metal concentrations on benthic community structure: San Francisco, California, San Francisco State University, M.S. thesis, 177 p.

Shouse, M.K., Parchaso, F., and Thompson, J.K. 2003, Near-field receiving water monitoring of benthic community near the Palo Alto Water Quality Control Plant in South San Francisco Bay; February 1974 through December 2003: U.S. Geological Survey Open-File Report 2003–224, 39 p.

Shouse, M.K., Parchaso, F., and Thompson, J.K., 2004, Near-field receiving water monitoring of benthic community near the Palo Alto Water Quality Control Plant in South San Francisco Bay; February 1974 through December 2003: U.S. Geological Survey Open-File Report 2004–1210, 39 p.

Simon, T.P., 2002, Biological response signatures: Boca Raton, Florida, CRC Press, 608 p.

Thompson, J.K., and Parchaso, F., 2012, Benthic invertebrate community assessment as a phytoplankton consumer and fish and bird prey source before and after the start of the restoration: South Bay Salt Pond Restoration Project Cooperative Agreement #2009-0211, 90 p.

Thompson, J.K., Parchaso, F., and Shouse, M.K., 2002, Near-field receiving water monitoring of benthic community near the Palo Alto Water Quality Control Plant in South San Francisco Bay; February 1974 through December 2000: U.S. Geological Survey Open-File Report 02–394, 117 p.

Thomson, E.A., Luoma, S.N., Johansson, C.E., and Cain, D.J., 1984, Comparison of sediments and organisms in identifying sources of biologically available trace metal contamination: Water Research, v. 18, p. 755–765.

Thomson-Becker, E.A., and Luoma, S.N., 1985, Temporal fluctuations in grain size, organic materials and iron concentrations in intertidal surface sediment: Hydrobiologia, v. 129, p. 91–109.

Topping, B.R., and Kuwabara, J.S., 2003, Dissolved nickel and benthic flux in south San Francisco Bay; a potential for natural sources to dominate: Bulletin of Environmental Toxicology and Chemistry, v. 71, p. 46–51.

U.S. Environmental Protection Agency, 2002, Method 1631, Revision E: Mercury in Water by Oxidation, Purge and Trap, and Cold Vapor Atomic Fluorescense Spectrometry: Washington, D.C., U.S. Environmental Protection Agency, EPA–821–R–02–019, 38 p.

U.S. Environmental Protection Agency, 2004, Revised assessment of detection and quantitation approaches: Washington, D.C., U.S. Environmental Protection Agency, EPA–821–B–04–005, 254 p.

Wang, W., and Fisher, N.S., 1999, Delineating metal accumulation pathways for marine invertebrates: Science of the Total Environment, v. 237, p. 459–472.

Weesner, F.M., 1960, General zoological microtechniques: Baltimore, Maryland, The Waverly Press, Inc., 230 p.

Wellise, C., Luoma, S.N., Cain, D.J., Brown, C., Hornberger, M., and Bouse, R., 1999, Near-field receiving water monitoring of trace metals in clams (*Macoma balthica*) and sediments near the Palo Alto and San Jose/Sunnyvale Water Quality Control Plants in South San Francisco Bay; 1998: U.S. Geological Survey Open-File Report 99–455, 101 p.

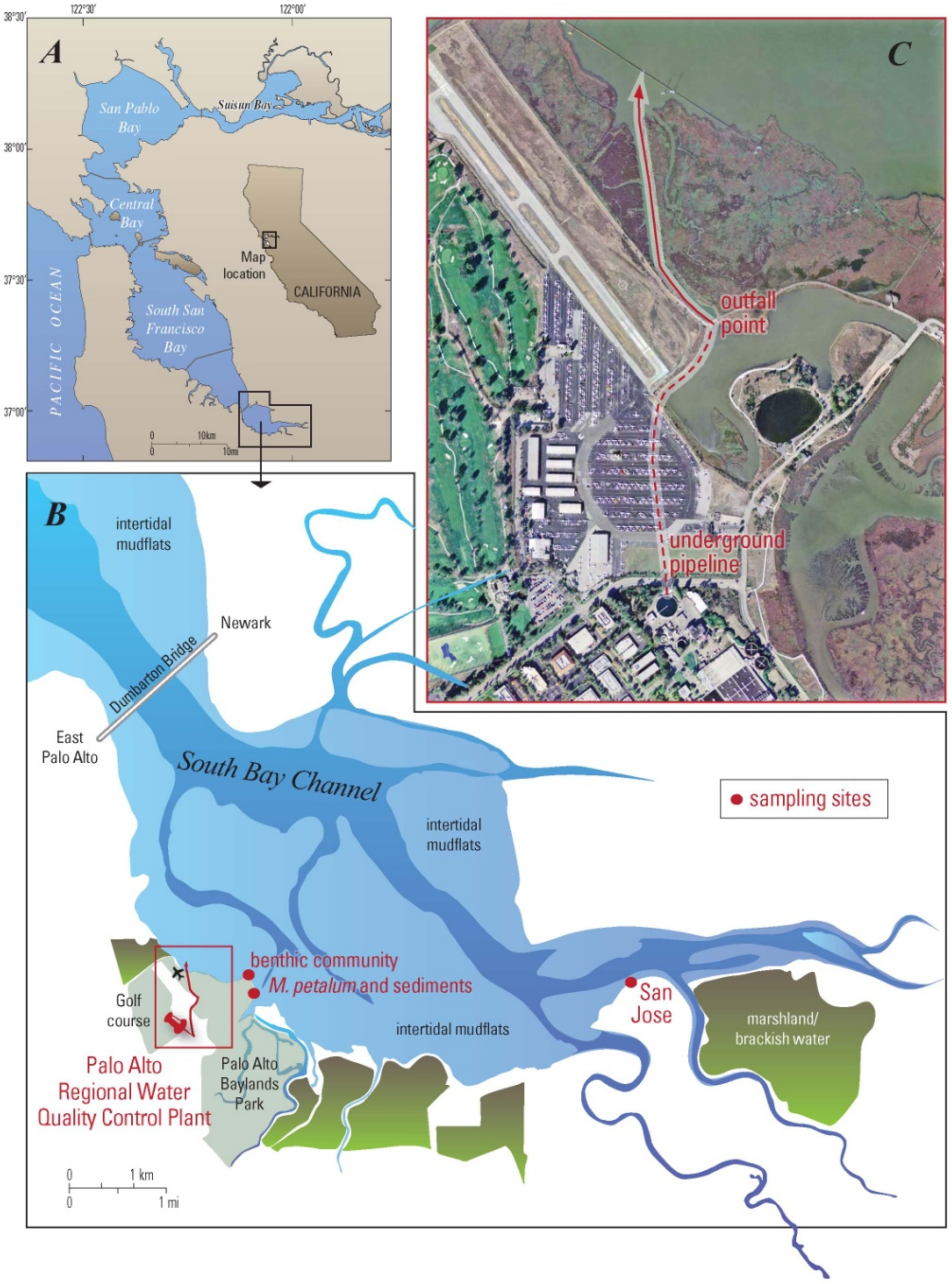


Figure 1. Location of the Palo Alto sampling site in South San Francisco Bay, Calif. *A*. Sampling area within the greater San Francisco Bay region. *B*. The intertidal mudflats are shaded light blue, subtidal in dark blue, and marshland/brackish water in green/brown. The benthic community and M. petalum and sediments points make up the Palo Alto sampling site. The San Jose sampling site (inactive) is also shown for reference. *C*. Effluent from the Palo Alto Regional Water Quality Control Plant (red thumbtack, insert B) is discharged via underground pipe (dashed red line) until it reaches the mouth of a small channel that connects to the intertidal mudflat approximately 1 km northwest of the sampling sites.

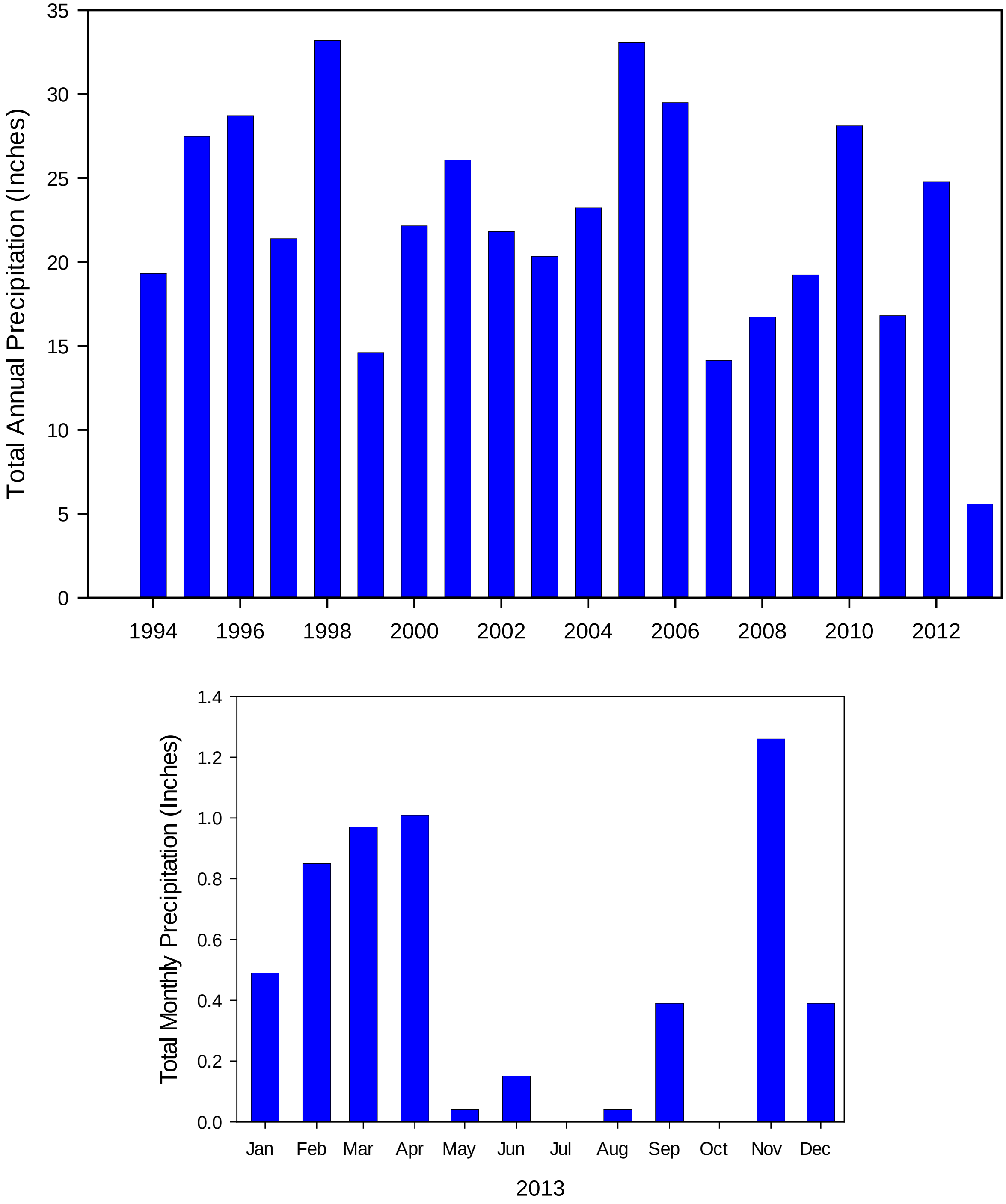


Figure 2. Total annual precipitation (1994–2013) and total monthly precipitation (2013) recorded at San Francisco WB AP in San Mateo County, Calif. The station (identification SFF) is operated by the National Weather Service.

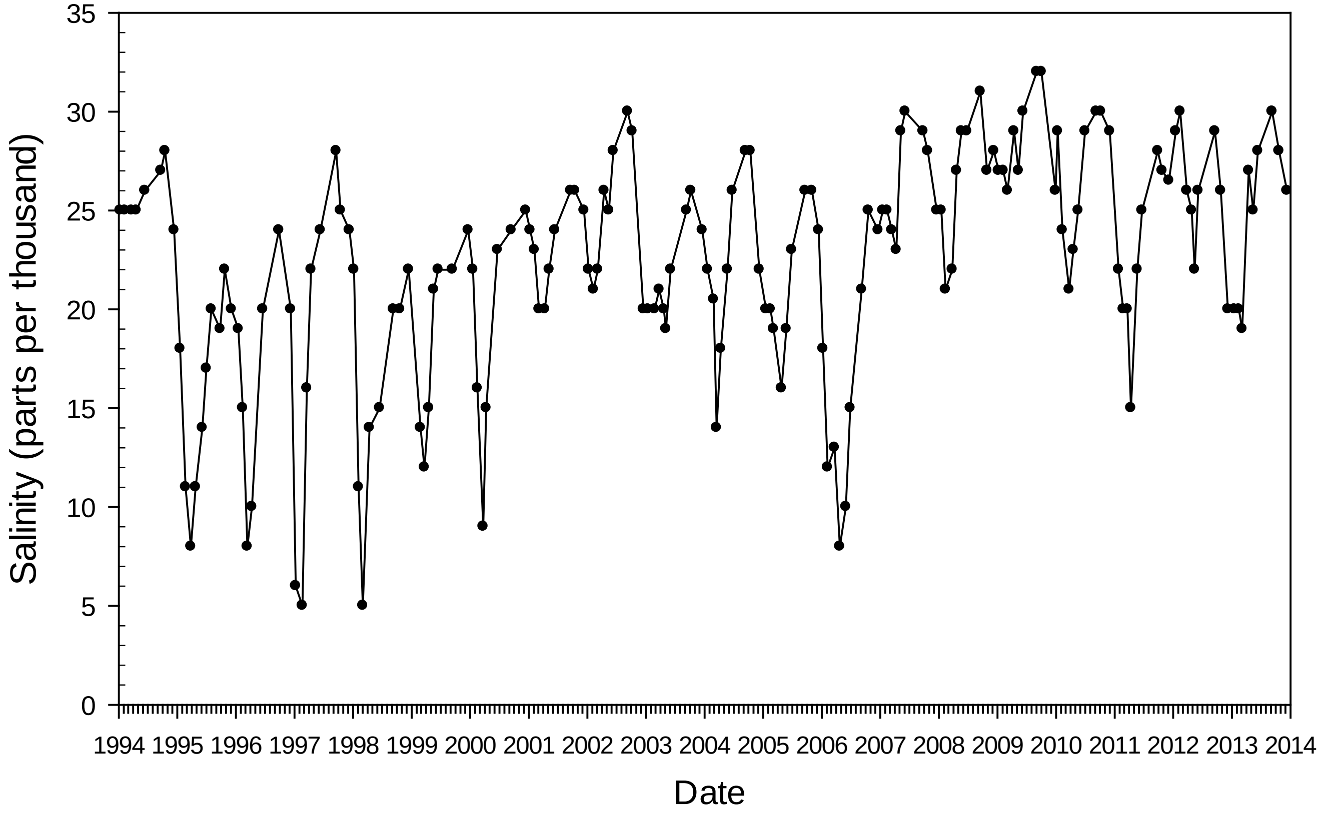


Figure 3. Surface-water salinity at the Palo Alto site, Calif.,1994–2013.

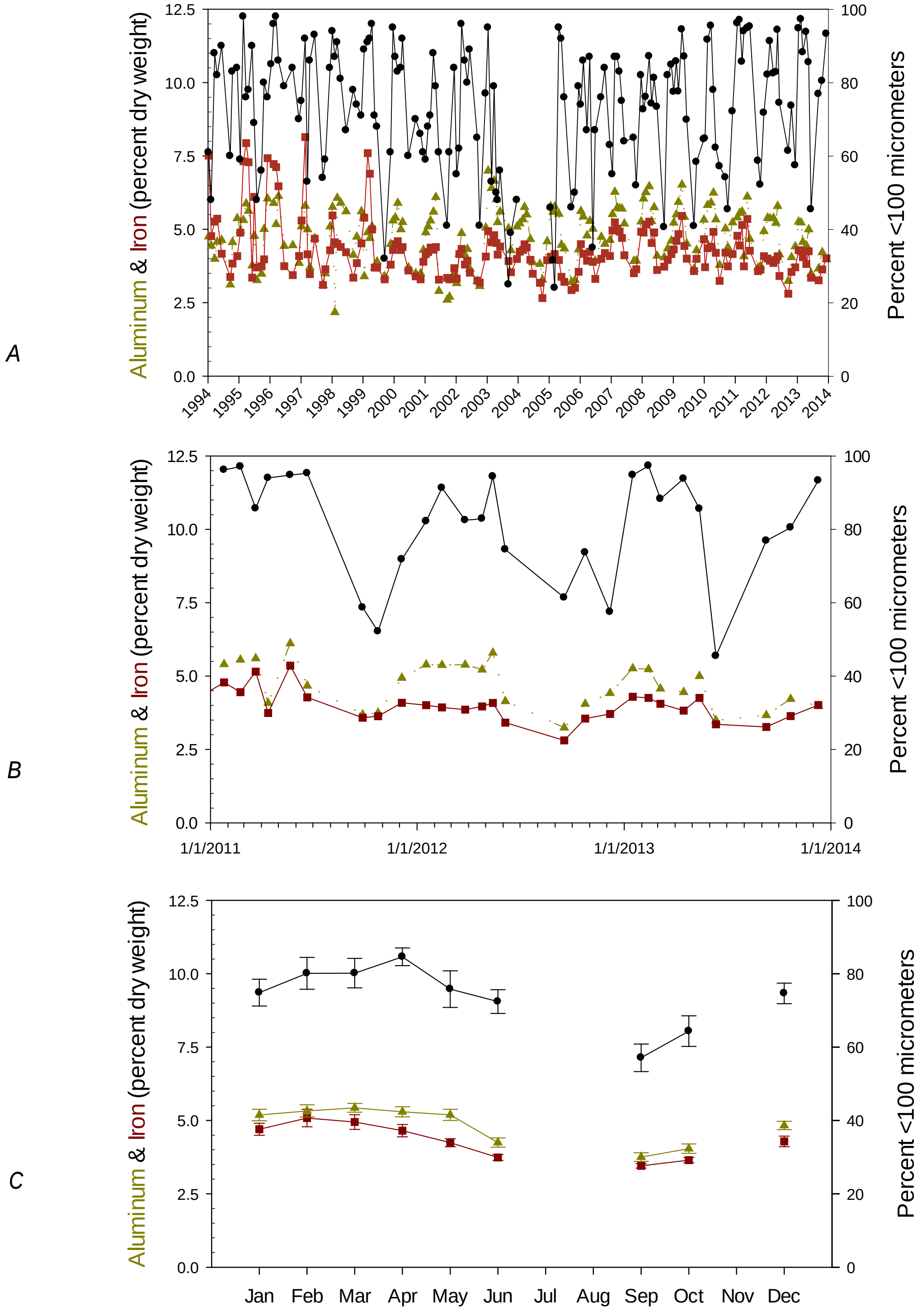


Figure 4. Aluminum, iron, and silt/clay in sediments, Palo Alto, Calif.,1994–2013. *A*. Percent aluminum (▲), iron (■) (extracted by near-total digest), and silt/clay (<100 μm) (●). Data on percent fines for 2004 contain unquantifiable biases due to errors in sample processing and, therefore, have been censored. *B*. Data for the past 3 years (2011–2013). *C*. The monthly mean of all samples collected from 1994–2013, illustrating the general seasonal variation in Al, Fe, and percent fine sediments. Collections are not made in July, August, and November. The error bar is the standard error of the mean (SEM).

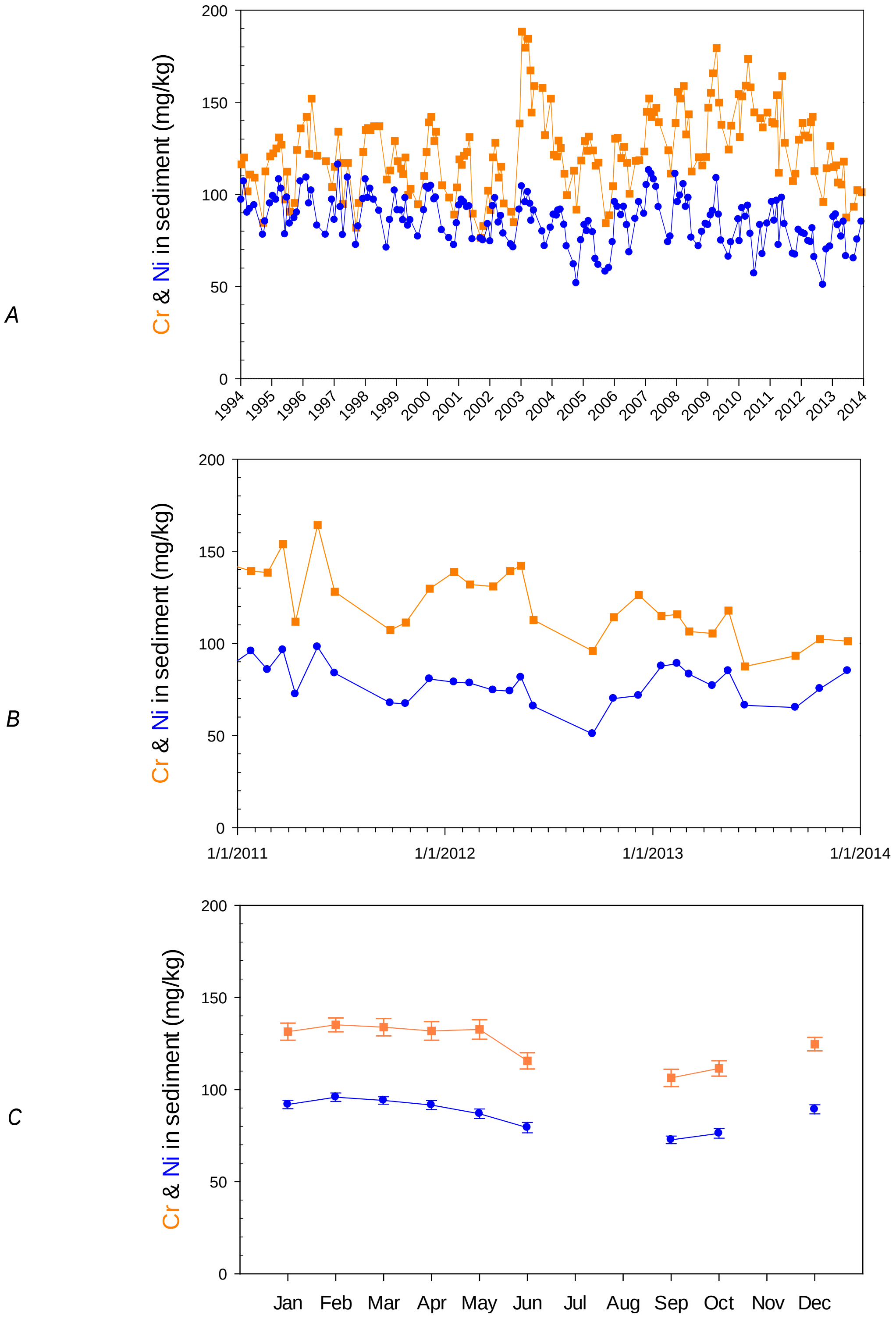


Figure 5. Chromium and nickel in sediments, Palo Alto, Calif., 1994–2013. *A*. Concentrations of chromium (Cr) (■) and nickel (Ni) (●) extracted by near-total digest. *B*. Data for the past 3 years (2011–2013). *C*. The monthly mean of all samples collected from 1994–2013, illustrating the general seasonal variation in Cr and Ni. Collections are not made in July, August, and November. The error bar is the standard error of the mean (SEM).

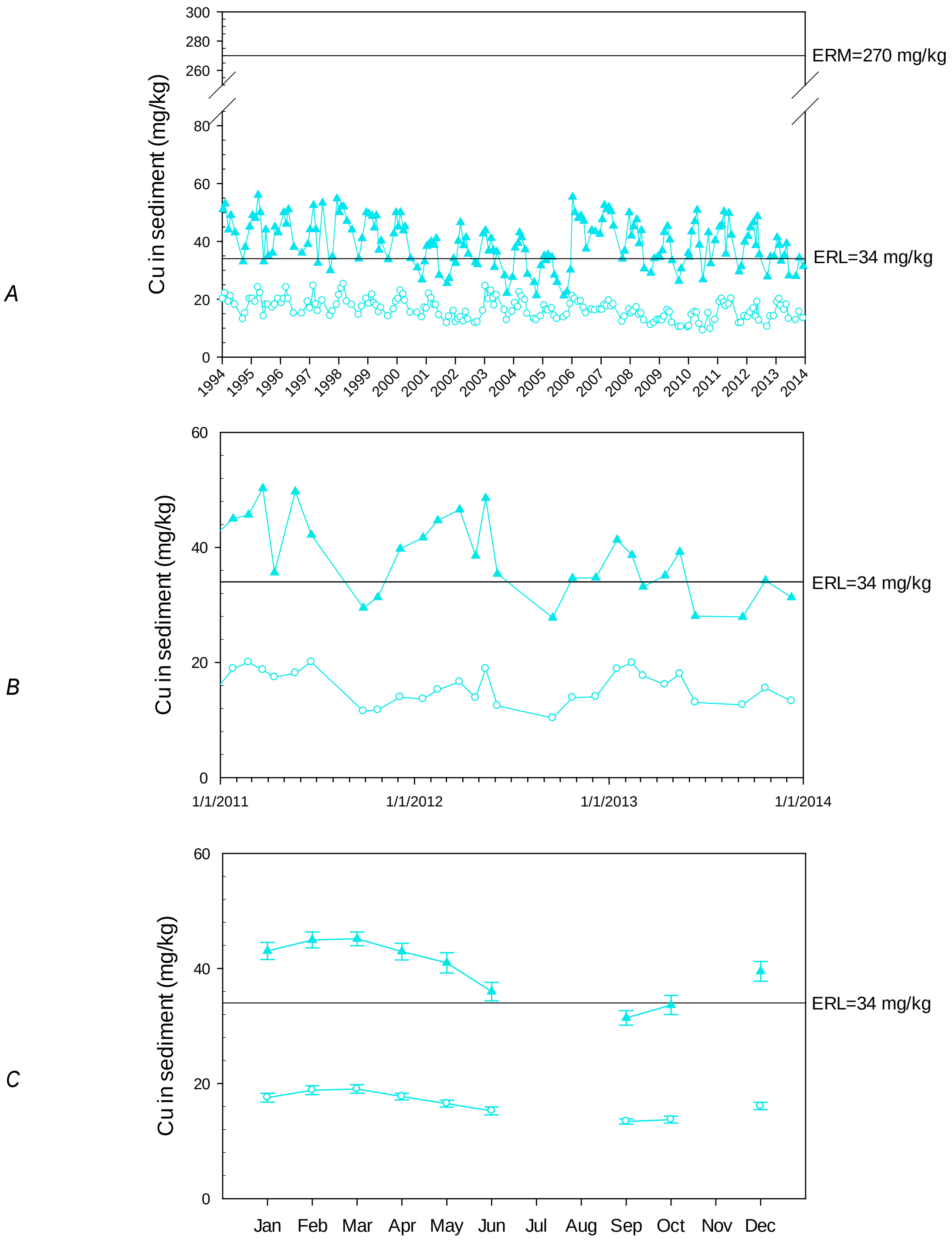


Figure 6. Copper in sediments, Palo Alto, Calif., 1994–2013. A. Near-total (▲) and partial-extractable (○) copper. B. Data for the past 3 years (2011–2013). C. The monthly mean of all samples collected from 1994–2013, illustrating the general seasonal variation in Cu. Collections are not made in July, August, and November. The error bar is the standard error of the mean (SEM). The ERL is the concentration below which the expected incidence of adverse effects is low (9 percent). The ERM is the concentration above which the expected incidence of adverse effects is high (84 percent).

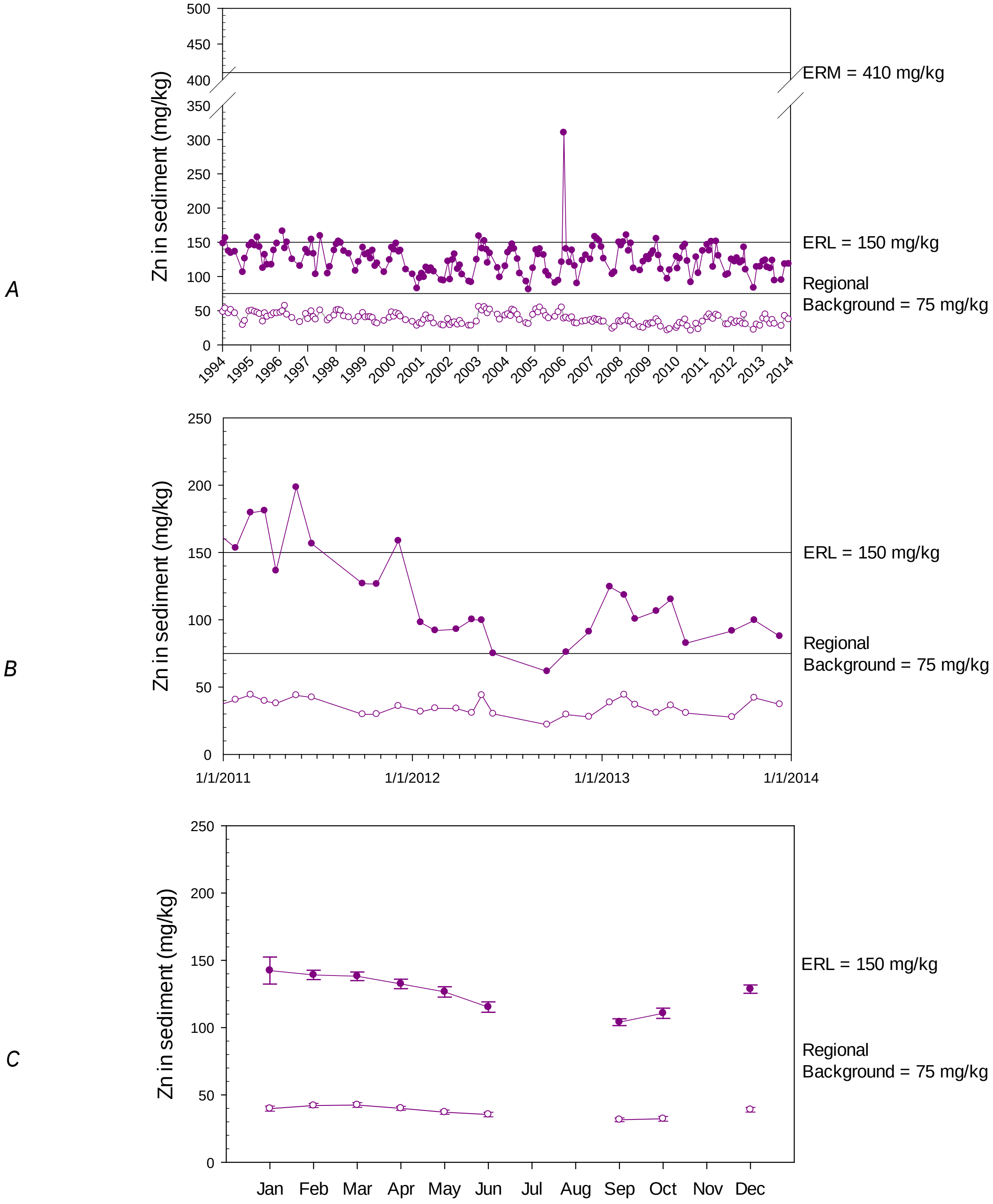


Figure 7. Zinc in sediments, Palo Alto, Calif., 1994–2013. A. Near-total (●) and partial-extractable (○) zinc. B. Data for the past 3 years (2011–2013). C. The monthly mean of all samples collected from 1994–2013, illustrating the general seasonal variation in Zn. Collections are not made in July, August, and November. The error bar is the standard error of the mean (SEM). The ERL is the concentration below which the expected incidence of adverse effects is low (6 percent). The ERM is the concentration above which the expected incidence of adverse effects is high (70 percent).

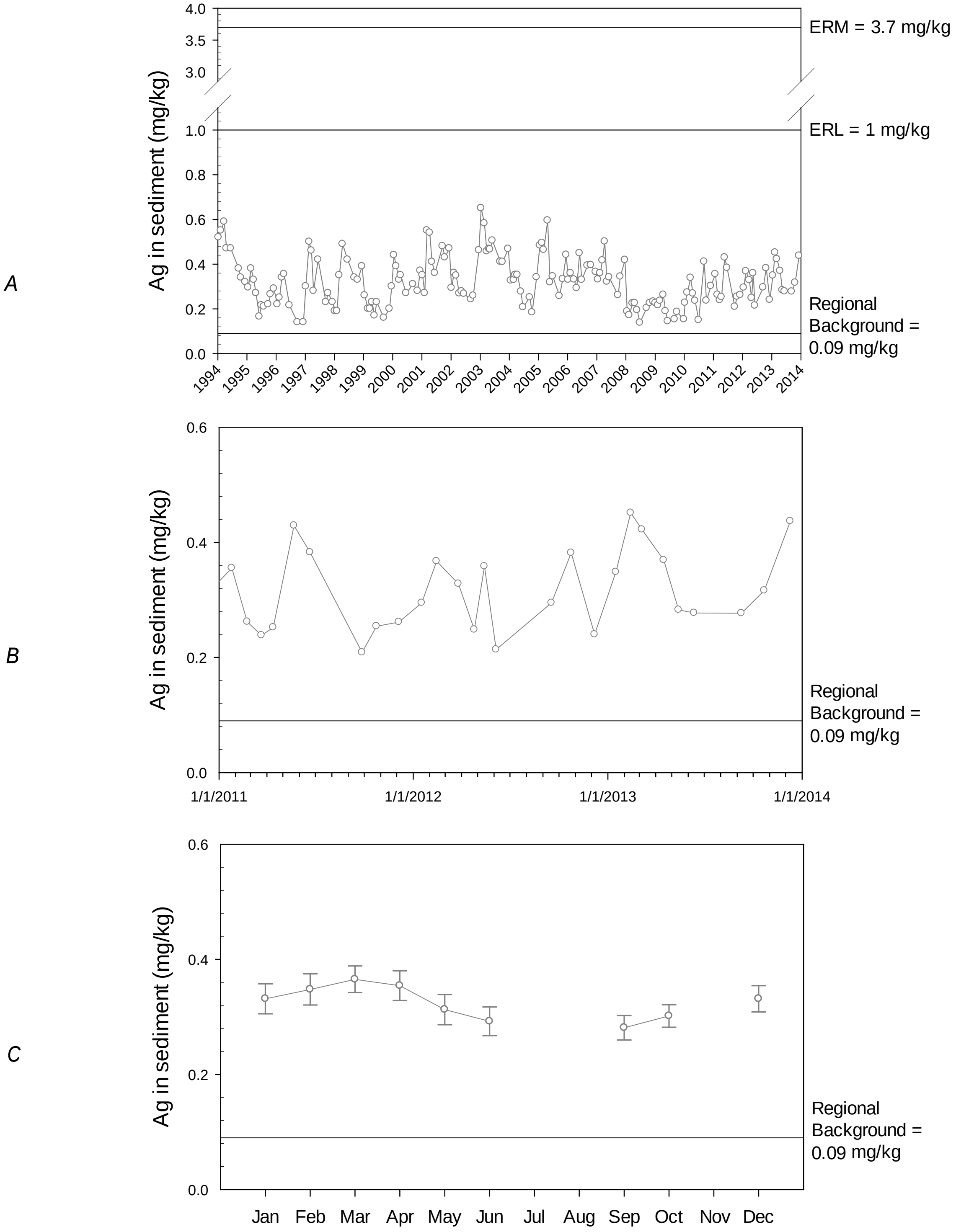


Figure 8. Silver in sediments, Palo Alto, Calif., 1994–2013. *A*. Data represent partial-extractable silver (treatment with 0.6 N hydrochloric acid). *B*. Data for the past 3 years (2011–2013). *C*. The monthly mean of all samples collected from 1994–2013, illustrating the general seasonal variation in Ag. Collections are not made in July, August, and November. The error bar is the standard error of the mean (SEM). The ERL is the concentration below which the expected incidence of adverse effects is low (3 percent). The ERM is the concentration above which the expected incidence of adverse effects is high (93 percent).

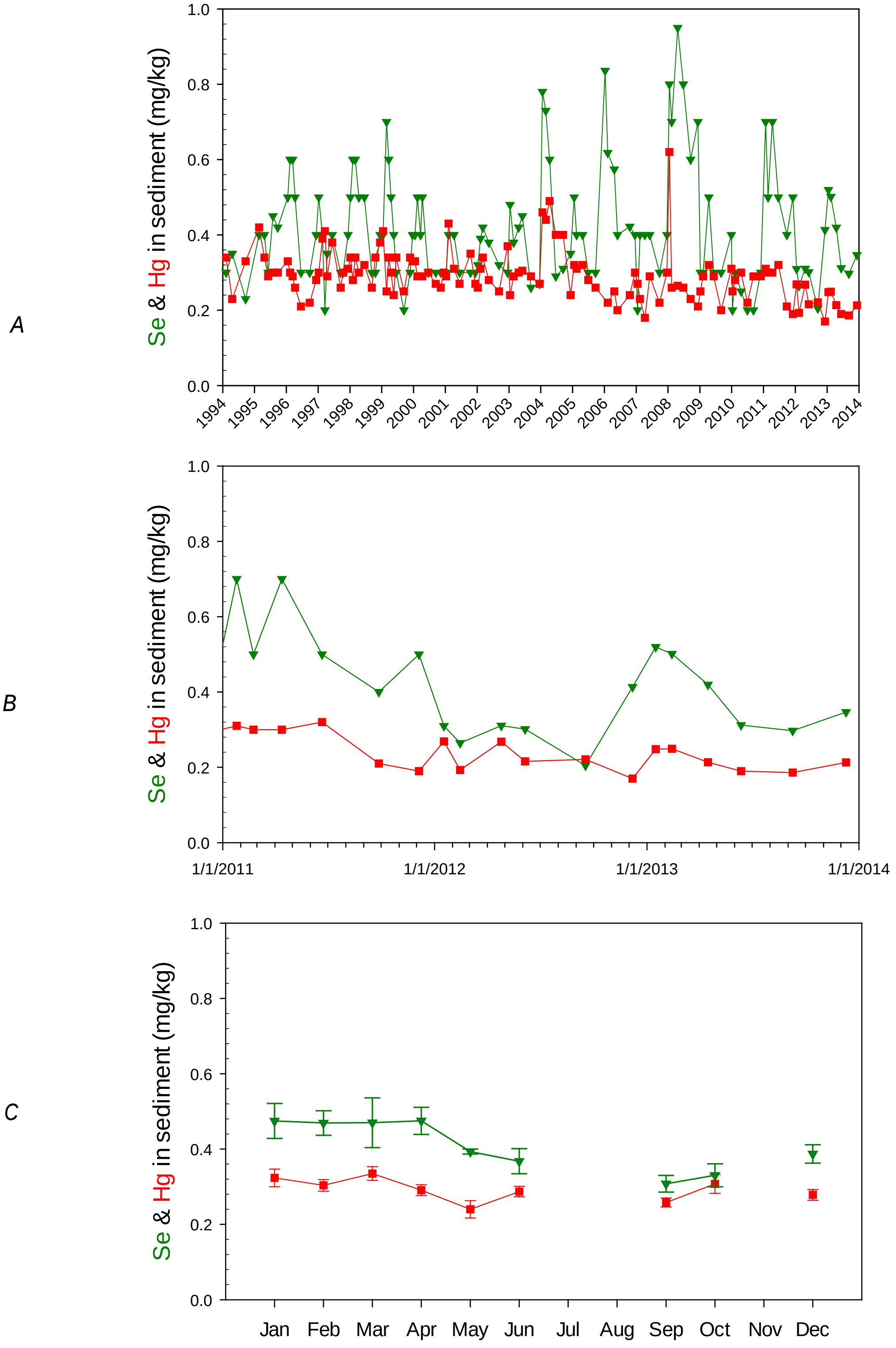


Figure 9. Selenium and mercury in sediments, Palo Alto, Calif., 1994–2013. *A*. Selenium (▼); mercury (🟥). *B*. Data for the past 3 years (2011–2013). *C*. The monthly mean of all samples collected from 1994–2013, illustrating the general seasonal variation in Se and Hg. Collections are not made in July, August, and November. The error bar is the standard error of the mean (SEM).

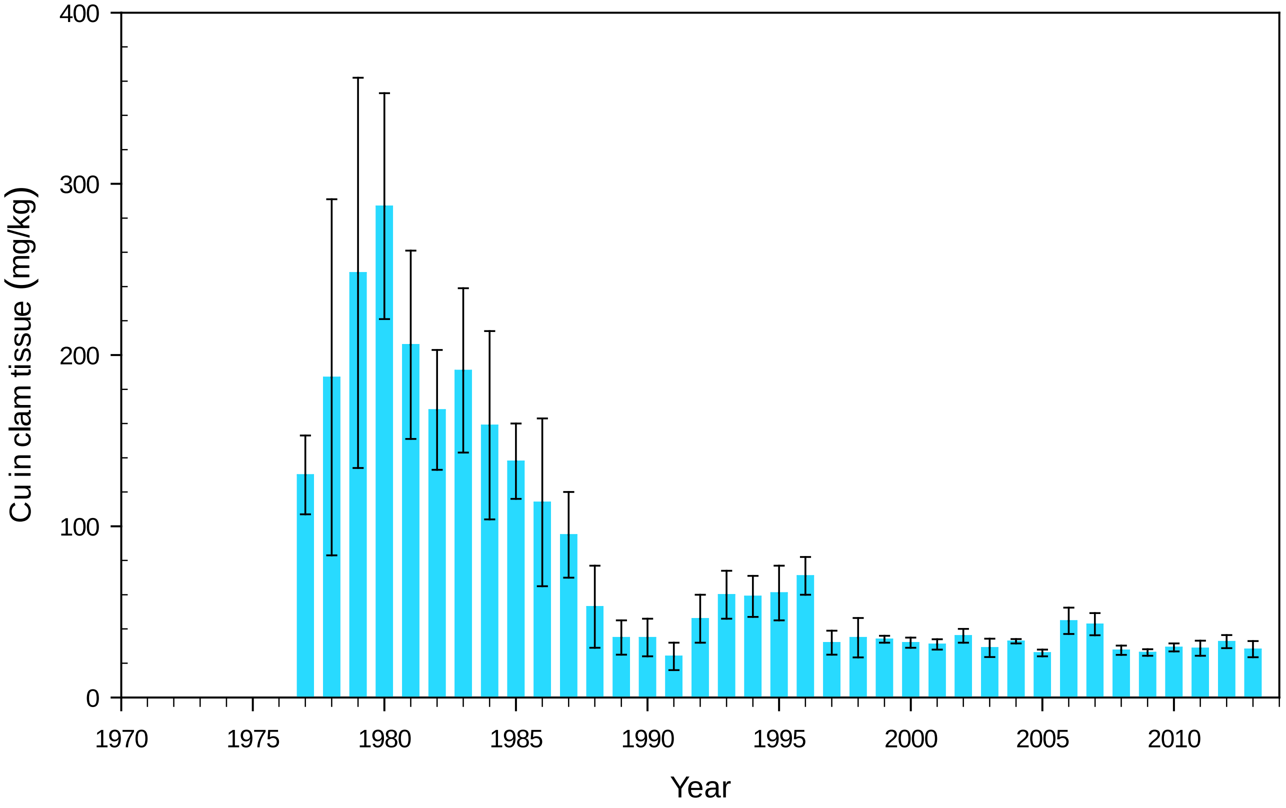


Figure 10. Annual mean copper concentrations in the clam *Macoma petalum*, Palo Alto, Calif., 1977–2013. Values are the annual (grand) means for 7 to 12 separate samples per year and error bars are standard errors of those means (SEM).

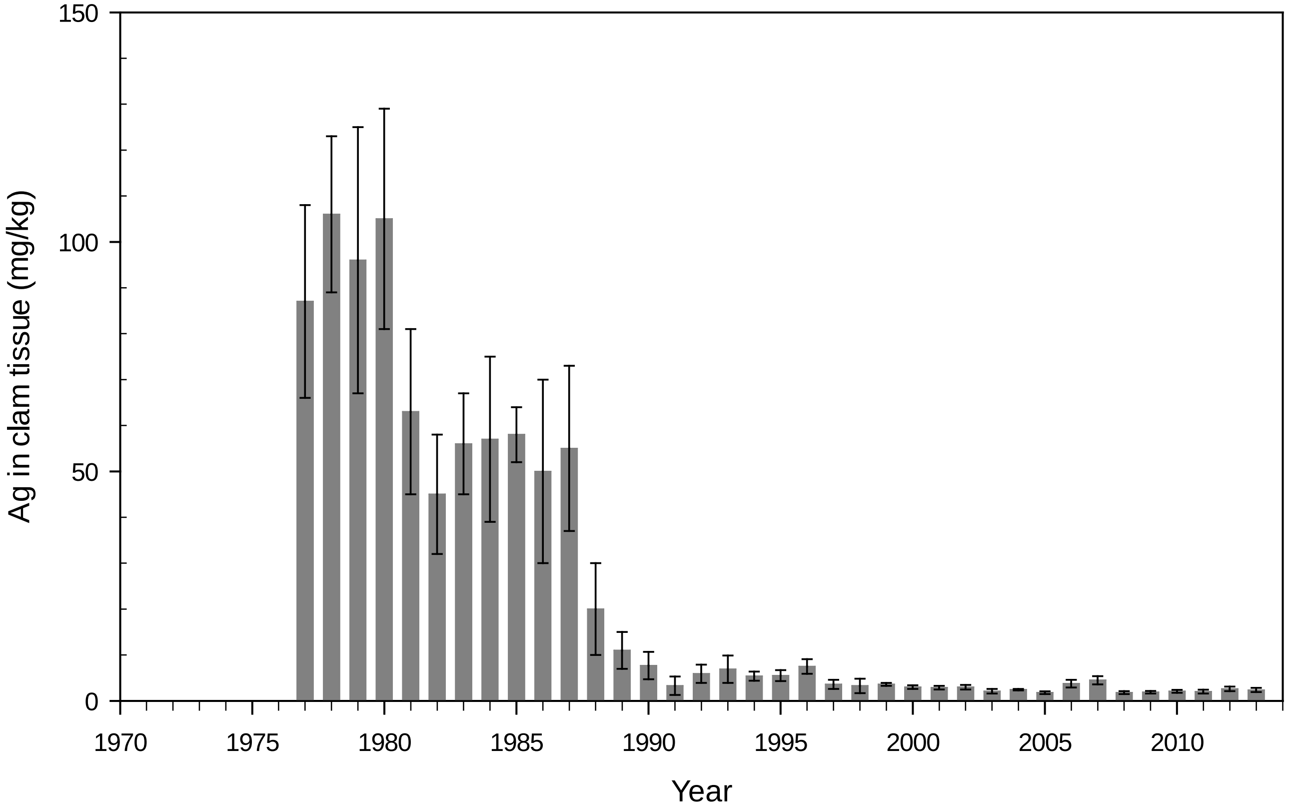


Figure 11. Annual mean silver concentrations in the clam *Macoma petalum*, Palo Alto, Calif., 1977–2013. Values are the annual (grand) means for 7 to 12 separate samples per year and error bars are standard errors of those means (SEM).

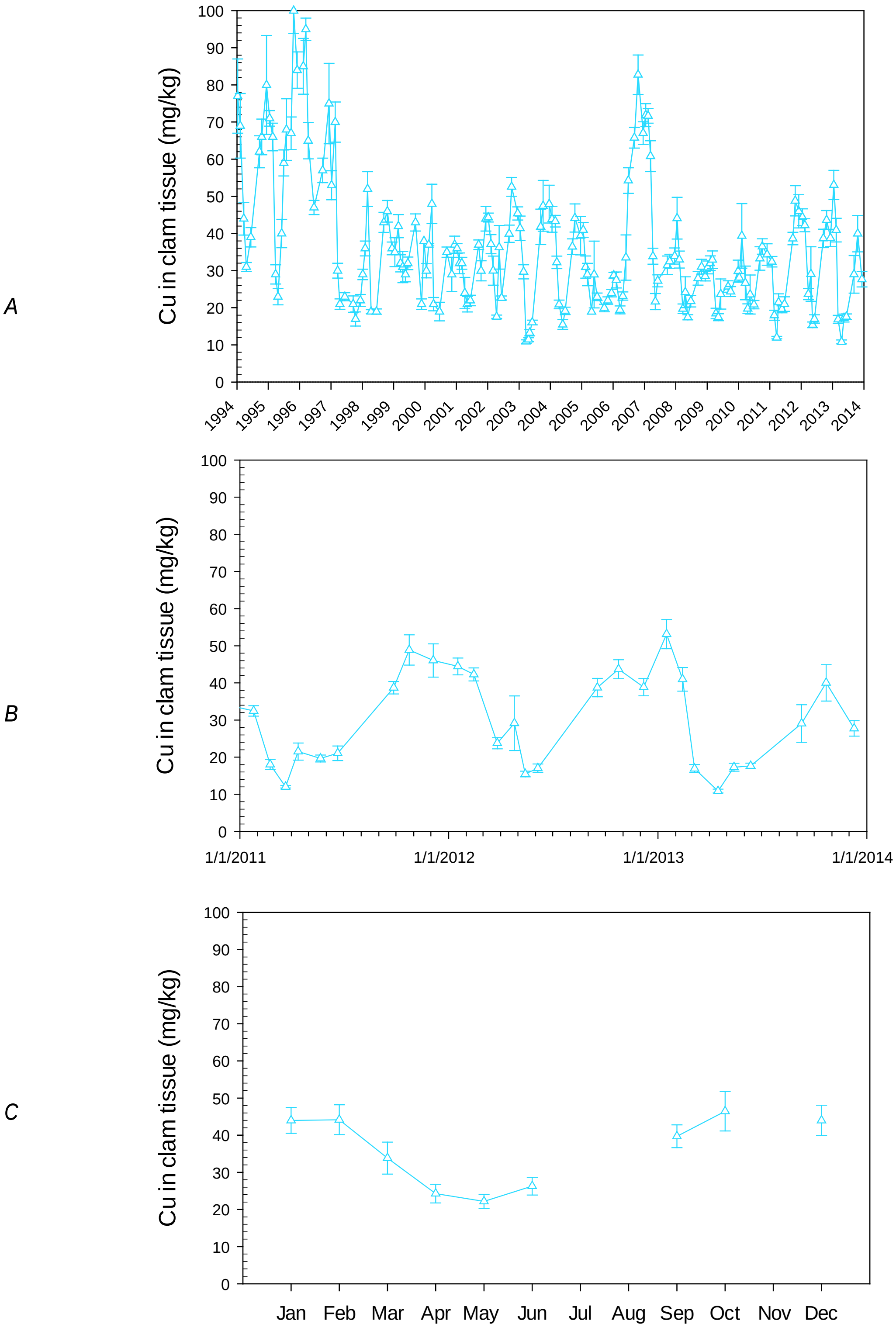


Figure 12. Copper concentrations in the clam *Macoma petalum*, Palo Alto, Calif., 1994–2013. *A*. Each value is the mean concentration for the sample collected on a given date. The error bar is the standard error of the mean (SEM). *B*. Data for the past 3 years (2011–2013). *C*. The monthly mean of all samples collected from 1994–2013, illustrating the general seasonal variation in Cu. Collections are not made in July, August, and November. The error bar is the standard error of the mean (SEM).

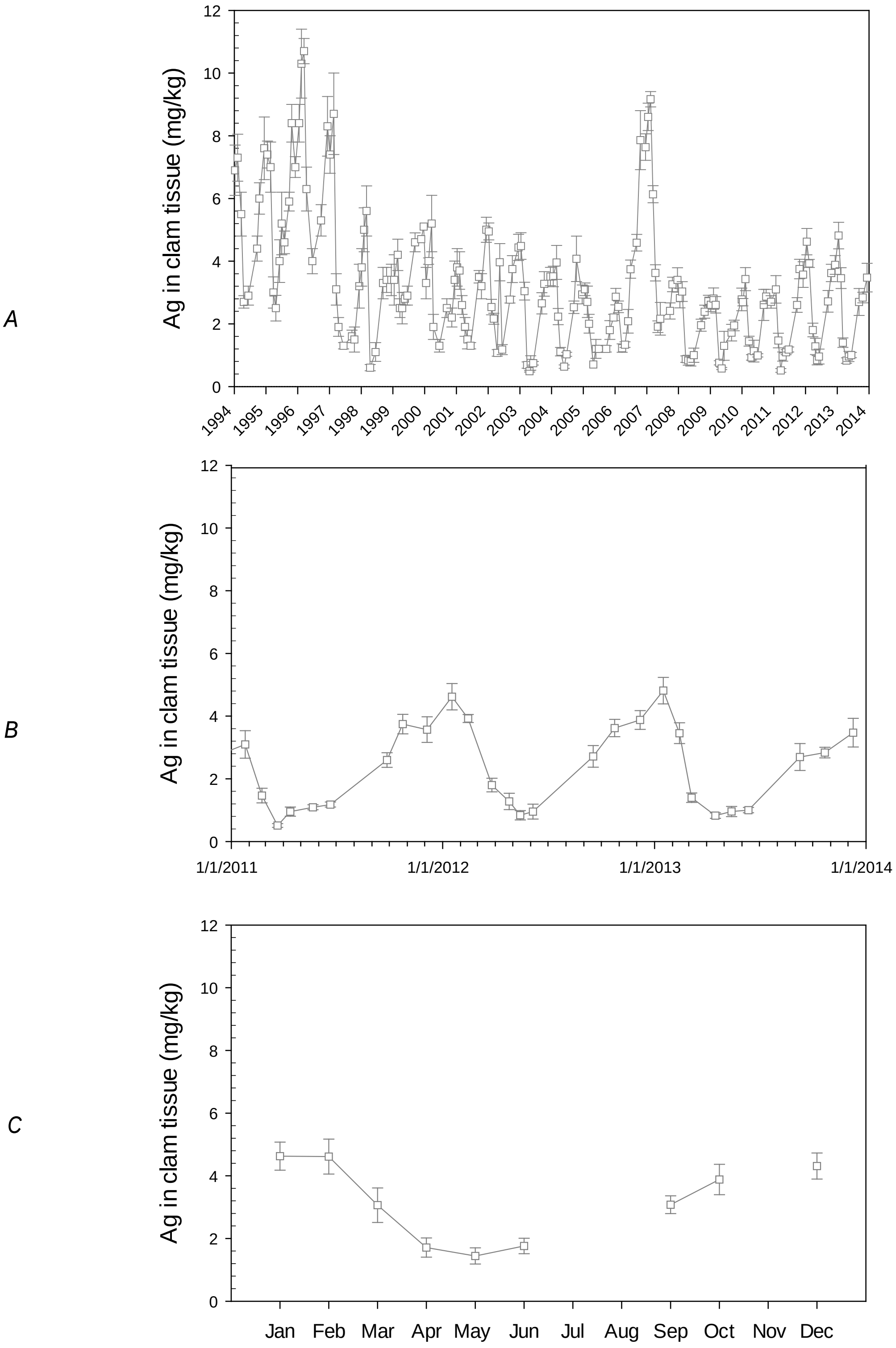


Figure 13. Silver concentrations in the clam *Macoma petalum*, Palo Alto, Calif., 1994–2013. *A*. Each value is the mean concentration for the sample collected on a given date. The error bar is the standard error of the mean (SEM). *B*. Data for the past 3 years (2011–2013). *C*. The monthly mean of all samples collected from 1994–2013, illustrating the general seasonal variation in Ag. Collections are not made in July, August, and November. The error bar is the standard error of the mean (SEM).

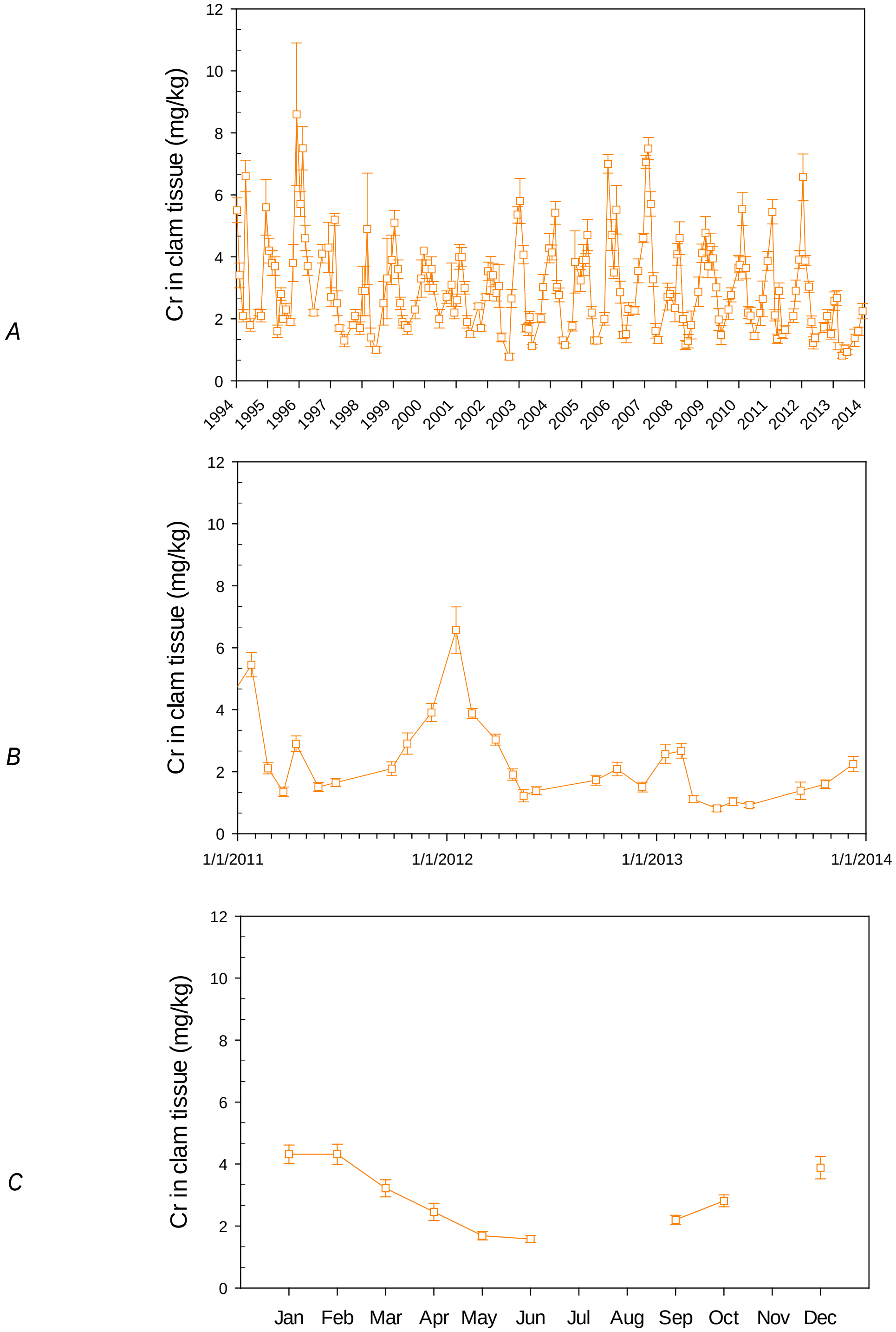


Figure 14. Chromium concentrations in the clam *Macoma petalum*, Palo Alto, Calif., 1994–2013. *A*. Each value is the mean concentration for the sample collected on a given date. The error bar is the standard error of the mean (SEM). *B*. Data for the past 3 years (2011–2013). *C*. The monthly mean of all samples collected from 1994–2013, illustrating the general seasonal variation in Cr. Collections are not made in July, August, and November. The error bar is the standard error of the mean (SEM).

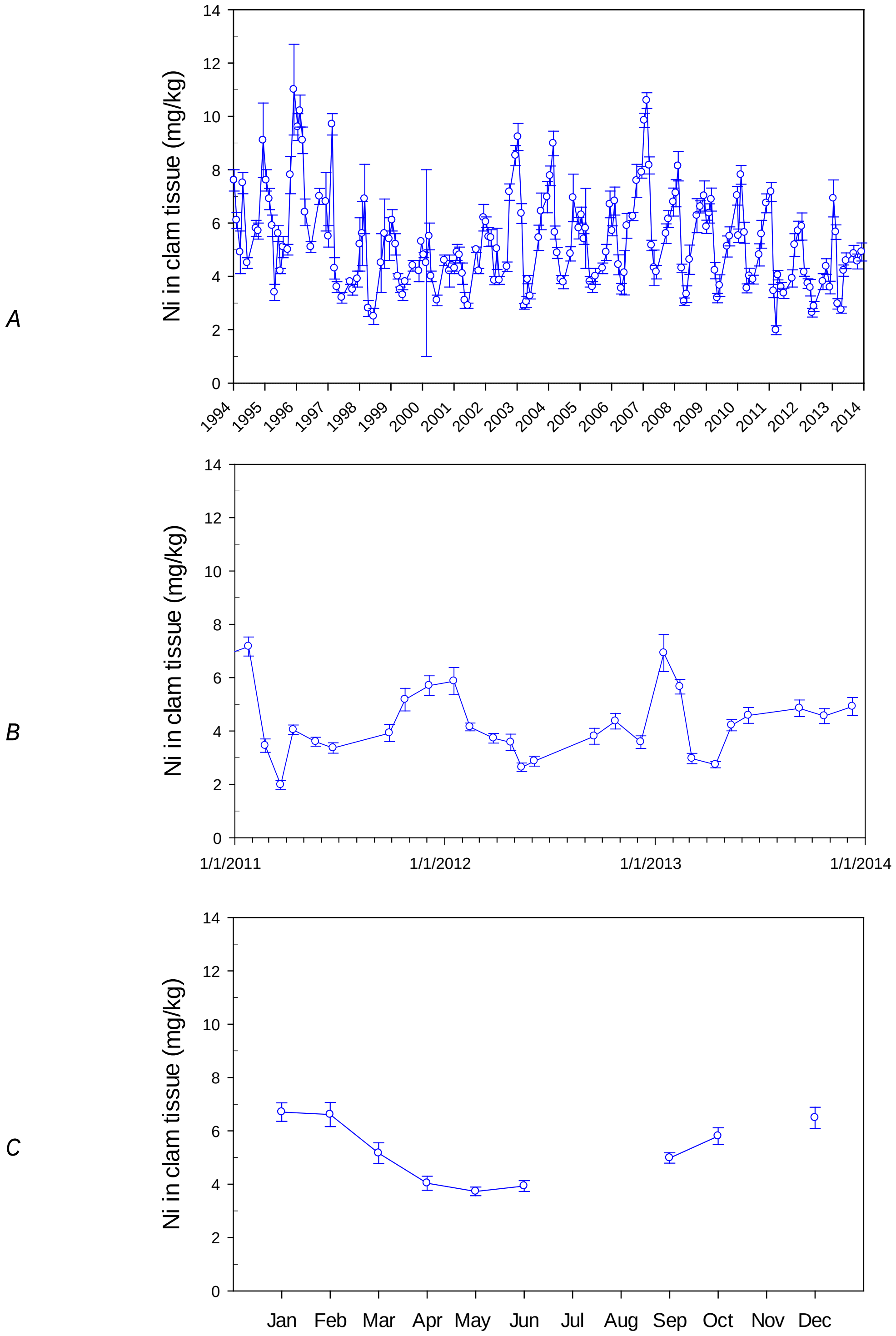


Figure 15. Nickel concentrations in the clam *Macoma petalum,* Palo Alto, Calif., 1994–2013. *A*. Each value is the mean concentration for the sample collected on a given date. The error bar is the standard error of the mean (SEM). *B*. Data for the past 3 years (2011–2013) *C*. The monthly mean of all samples collected from 1994–2013, illustrating the general seasonal variation in Ni. Collections are not made in July, August, and November. The error bar is the standard error of the mean (SEM).

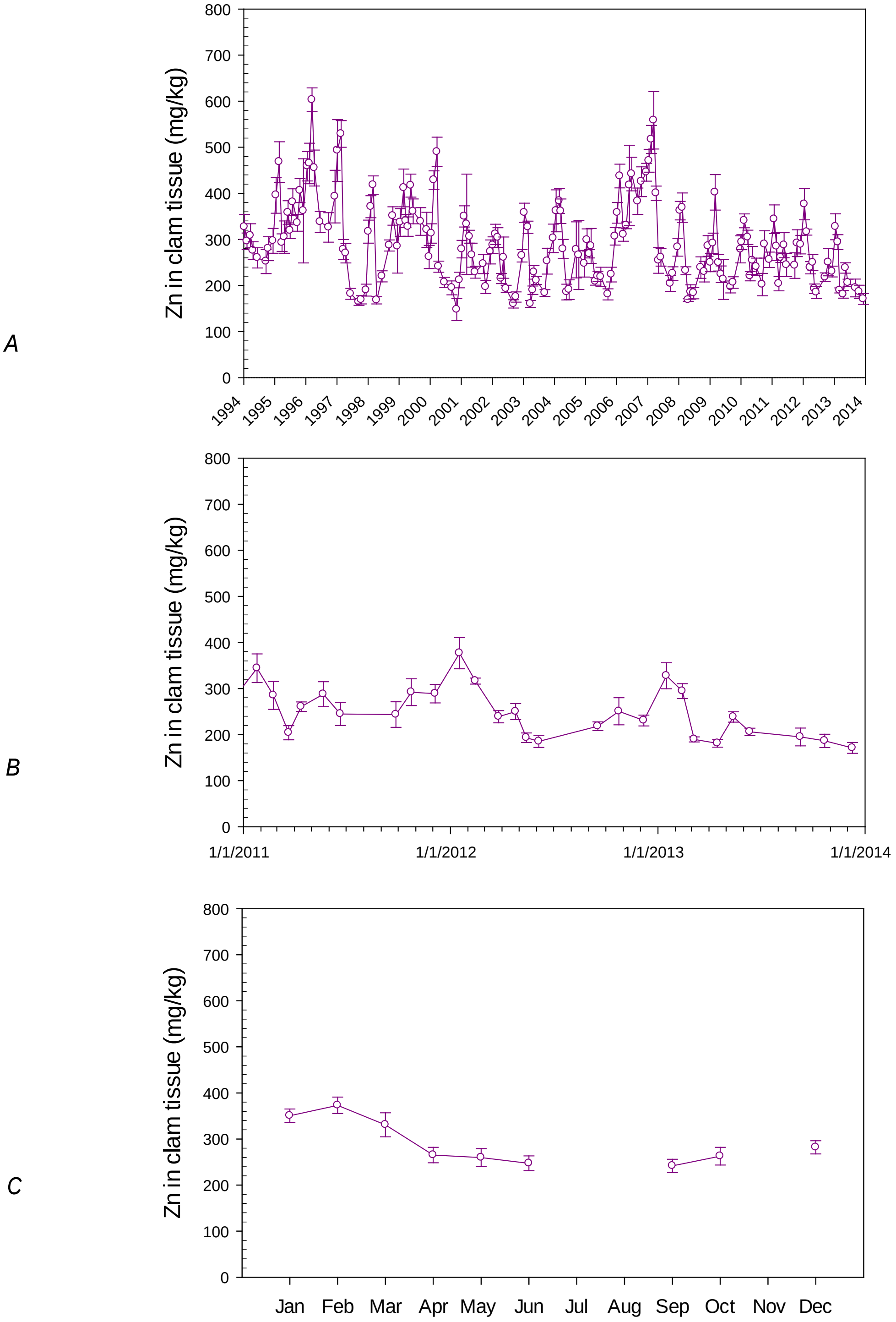


Figure 16. Zinc concentrations in the clam *Macoma petalum,* Palo Alto, Calif., 1994–2013. *A*. Each value is the mean concentration for the sample collected on a given date. The error bar is the standard error of the mean (SEM). *B*. Data for the past 3 years (2011–2013). *C*. The monthly mean of all samples collected from 1994–2013, illustrating the general seasonal variation in Zn. Collections are not made in July, August, and November. The error bar is the standard error of the mean (SEM).

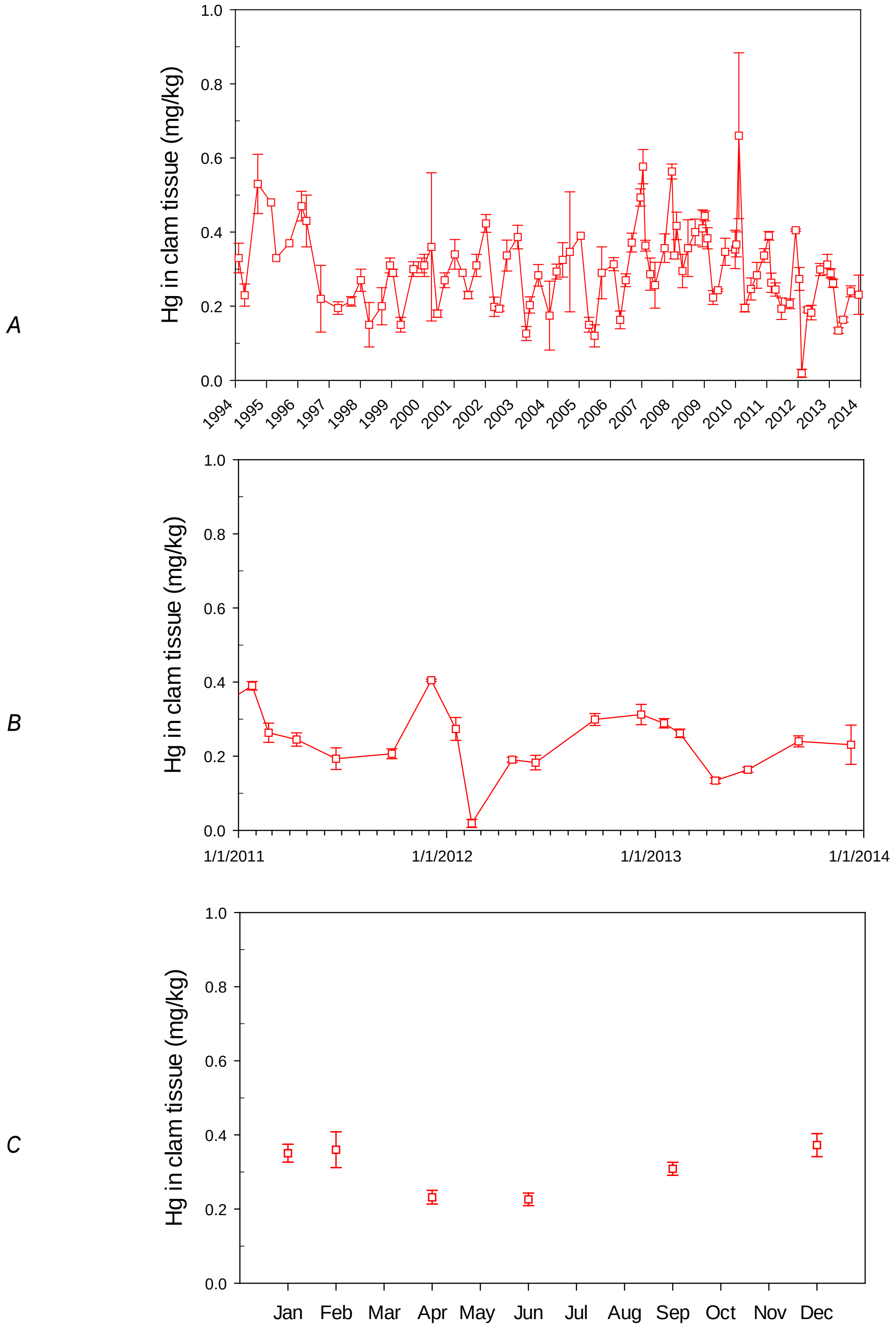


Figure 17. Mercury concentrations in the clam *Macoma petalum,* Palo Alto, Calif., 1994–2013. *A*. Each value is the mean concentration for the sample collected on a given date. The error bar is the standard error of the mean (SEM). *B*. Data for the past 3 years (2011–2013). *C*. The monthly mean of all samples collected from 1994–2013, illustrating the general seasonal variation in Hg. Collections are not made in March, May, July, August, October, and November. The error bar is the standard error of the mean (SEM).

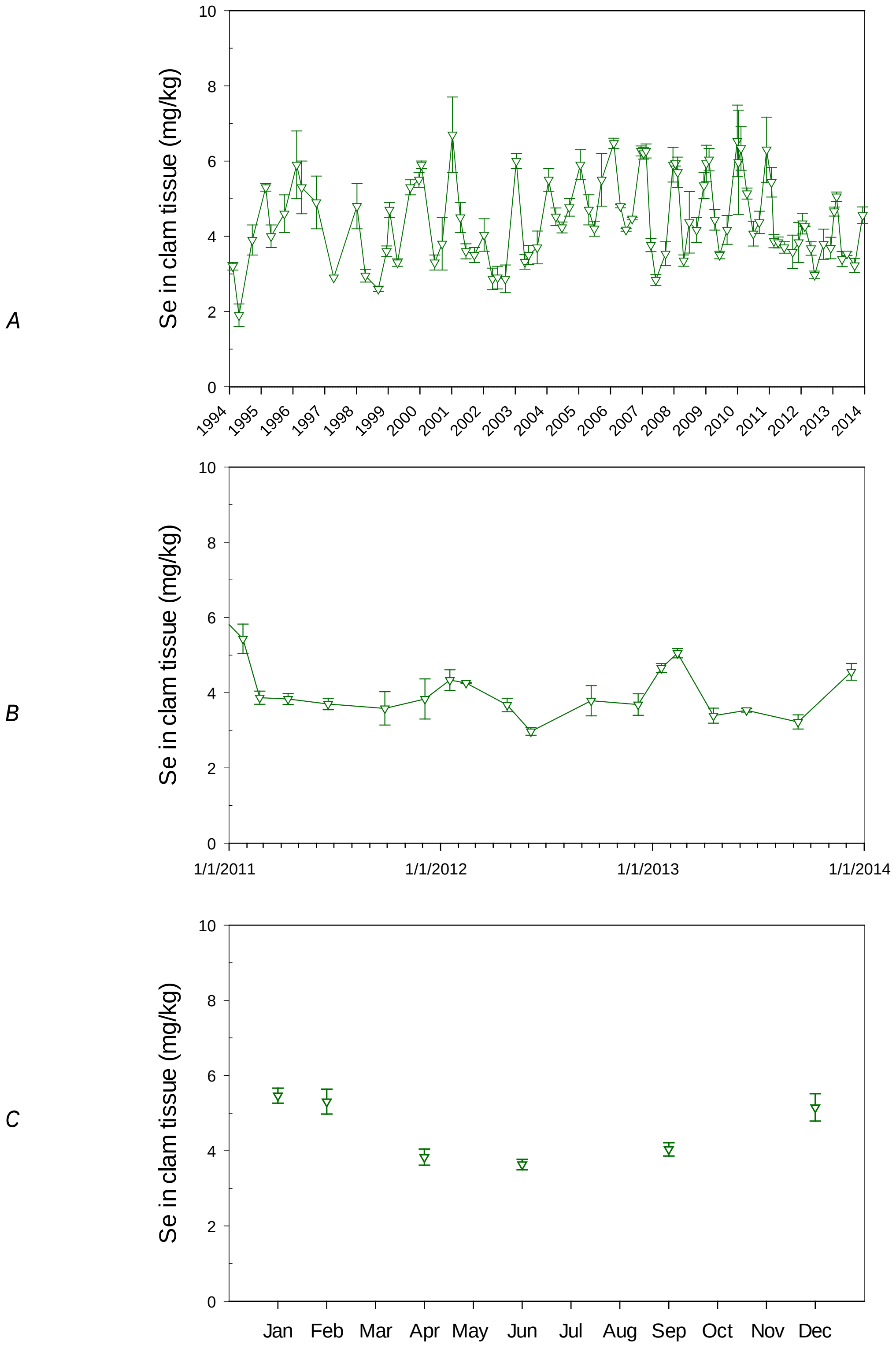


Figure 18. Selenium concentrations in the clam *Macoma petalum,* Palo Alto, Calif., 1994–2013. *A*. Each value is the mean concentration for the sample collected on a given date. The error bar is the standard error of the mean (SEM). *B*. Data for the past 3 years (2011–2013). *C*. The monthly mean of all samples collected from 1994–2013, illustrating the general seasonal variation in Se. Collections are not made in March, May, July, August, October, and November. The error bar is the standard error of the mean (SEM).

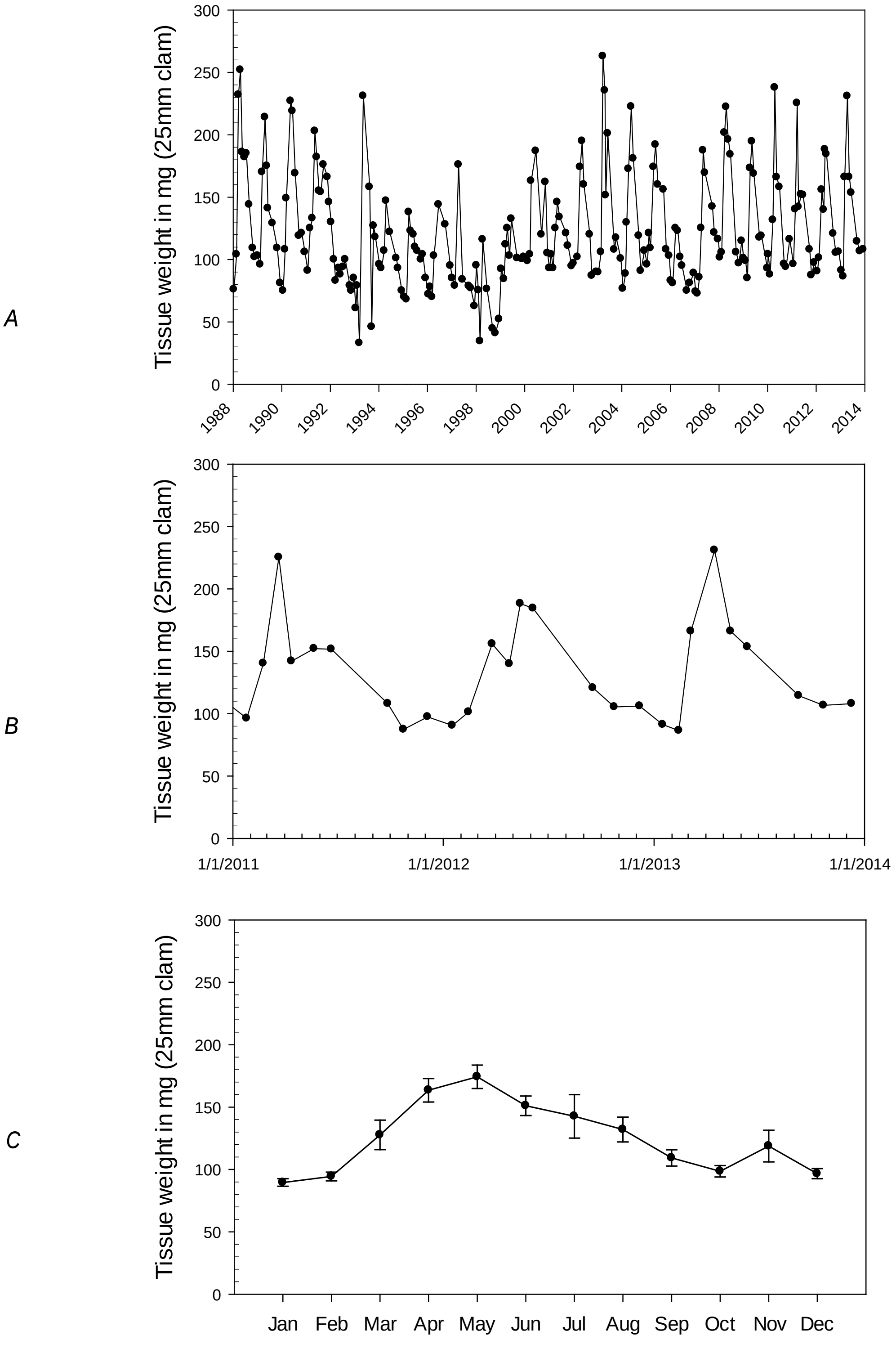


Figure 19. Condition index of the clam *Macoma petalum,* Palo Alto, Calif., 1988–2013. *A*. The condition index (CI) is defined as the weight (milligrams) of the soft tissues for an individual clam having a shell length of 25 mm. *B*. Condition index over the past 3 years (2011–2013). *C*. The monthly mean of all samples collected from 1988–2013, illustrating the general seasonal variation in condition index. The error bar is the standard error of the mean (SEM).

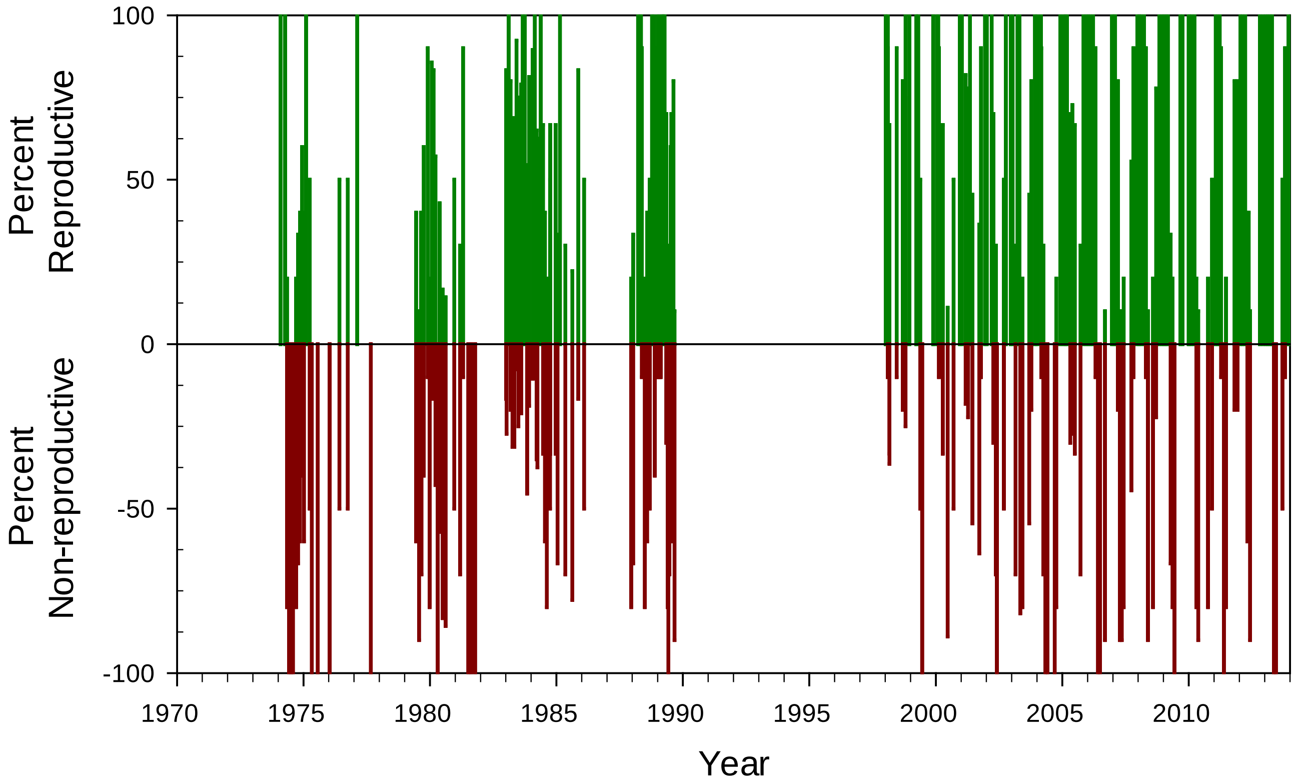


Figure 20. Reproductive activity of the clam *Macoma petalum,* Palo Alto, Calif., 1974–2013. Values are the percent of individuals that were either reproductively inactive (non-reproductive; shown in red) or in various stages of reproduction (reproductive; shown in green). The percent of non-reproductive individuals is reported as a negative value.

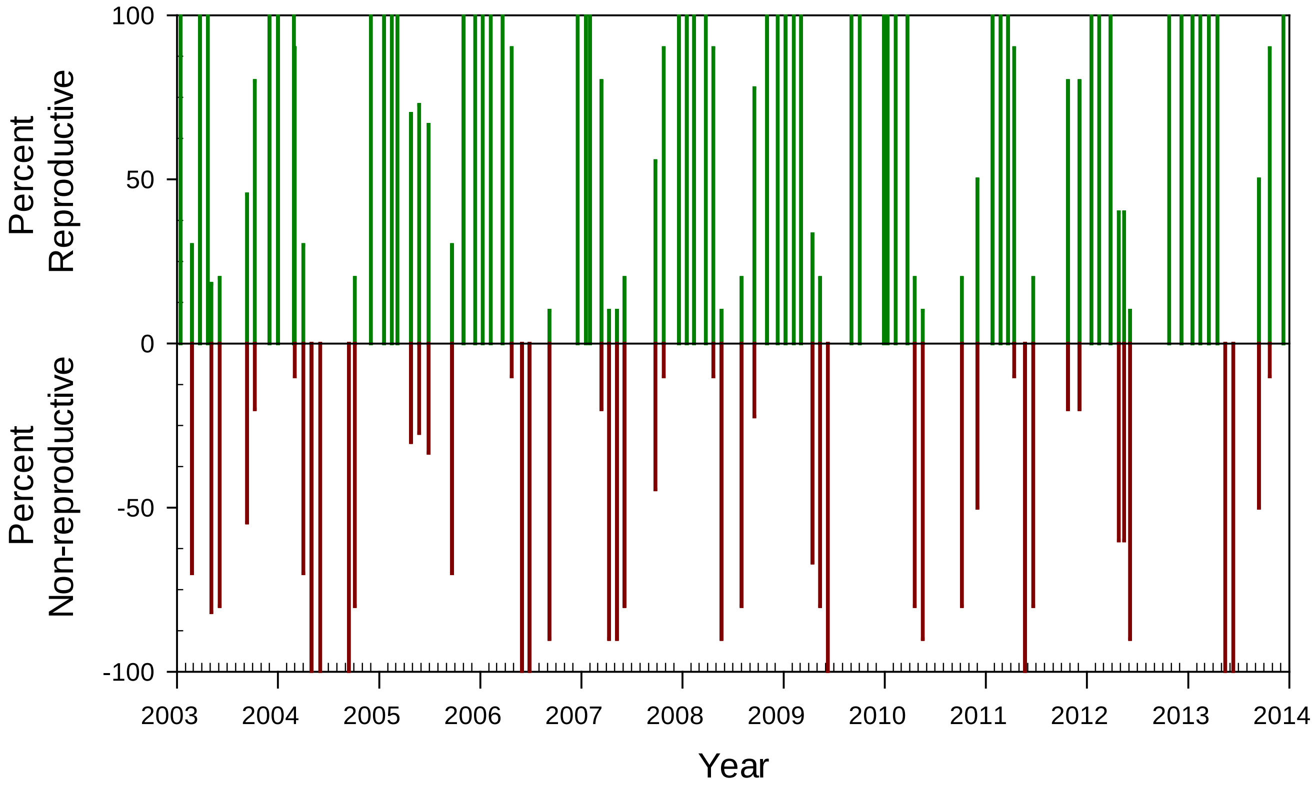
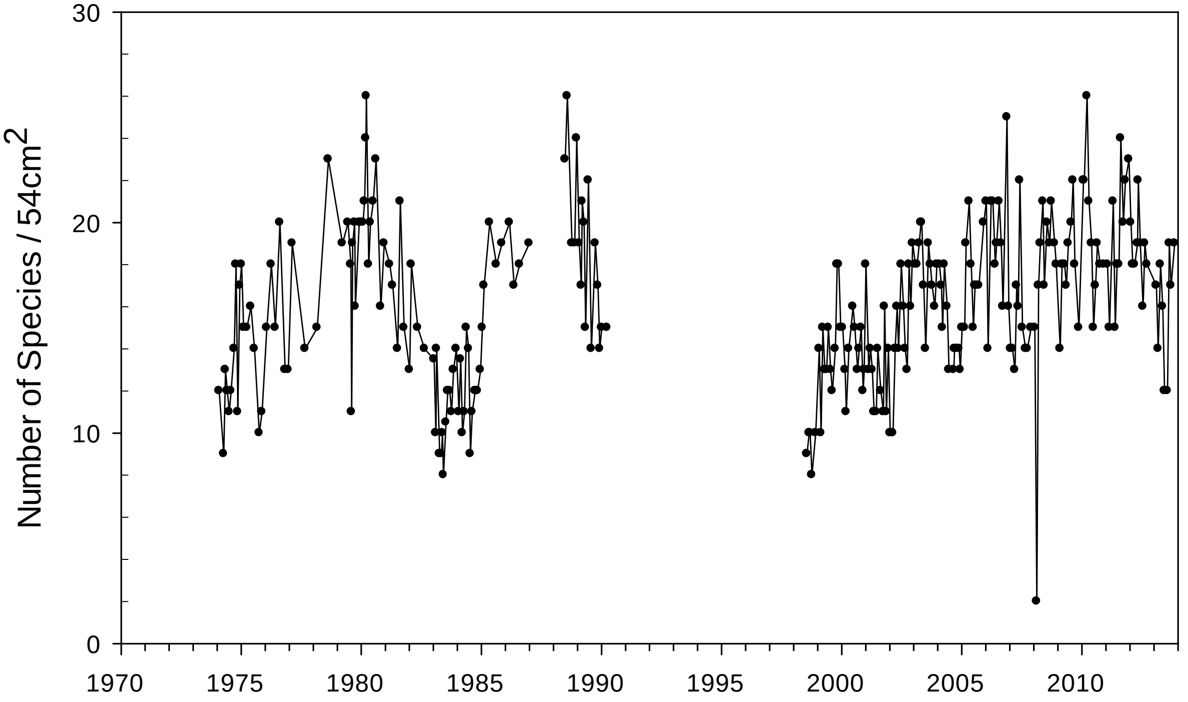


Figure 21. Reproductive activity of the clam *Macoma petalum,* Palo Alto, Calif., 2003–2013. Values are the percent of individuals that were either reproductively inactive (non-reproductive; shown in red) or in various stages of reproduction (reproductive; shown in green). The percent of non-reproductive individuals is reported as a negative value.



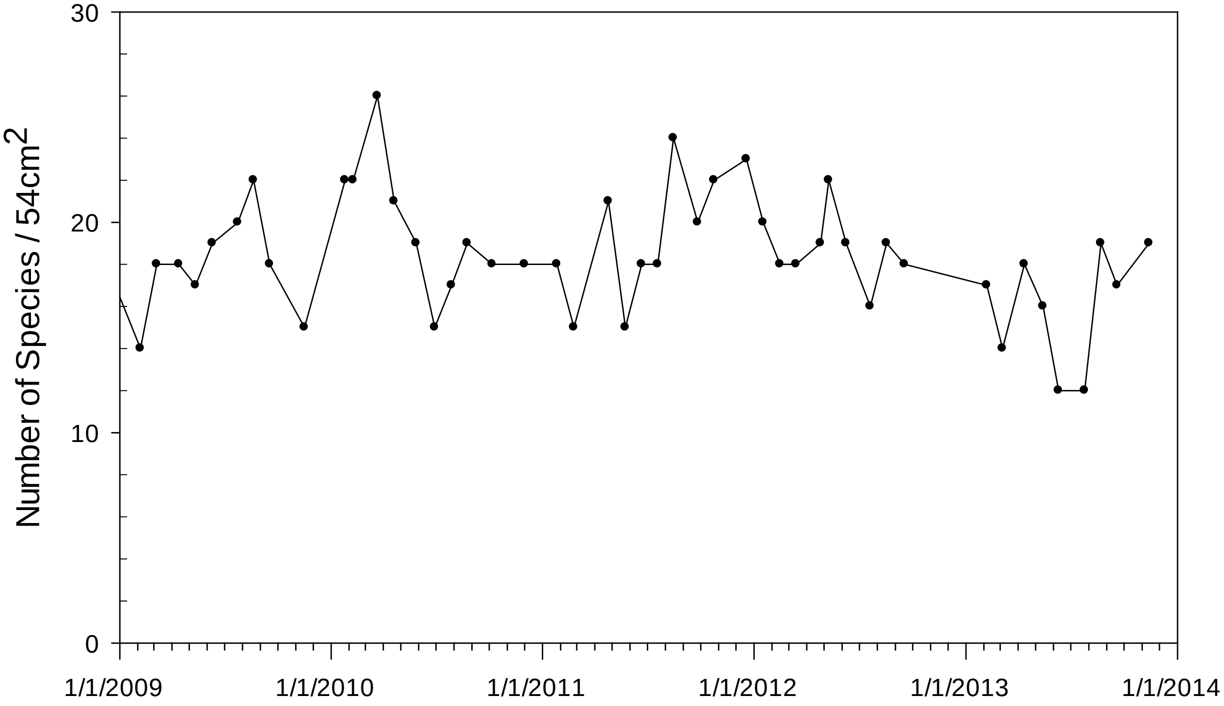


Figure 22. Total number of species present at the Palo Alto site, Calif., 1974–2013. Collections were not made between 1991 and 1998.

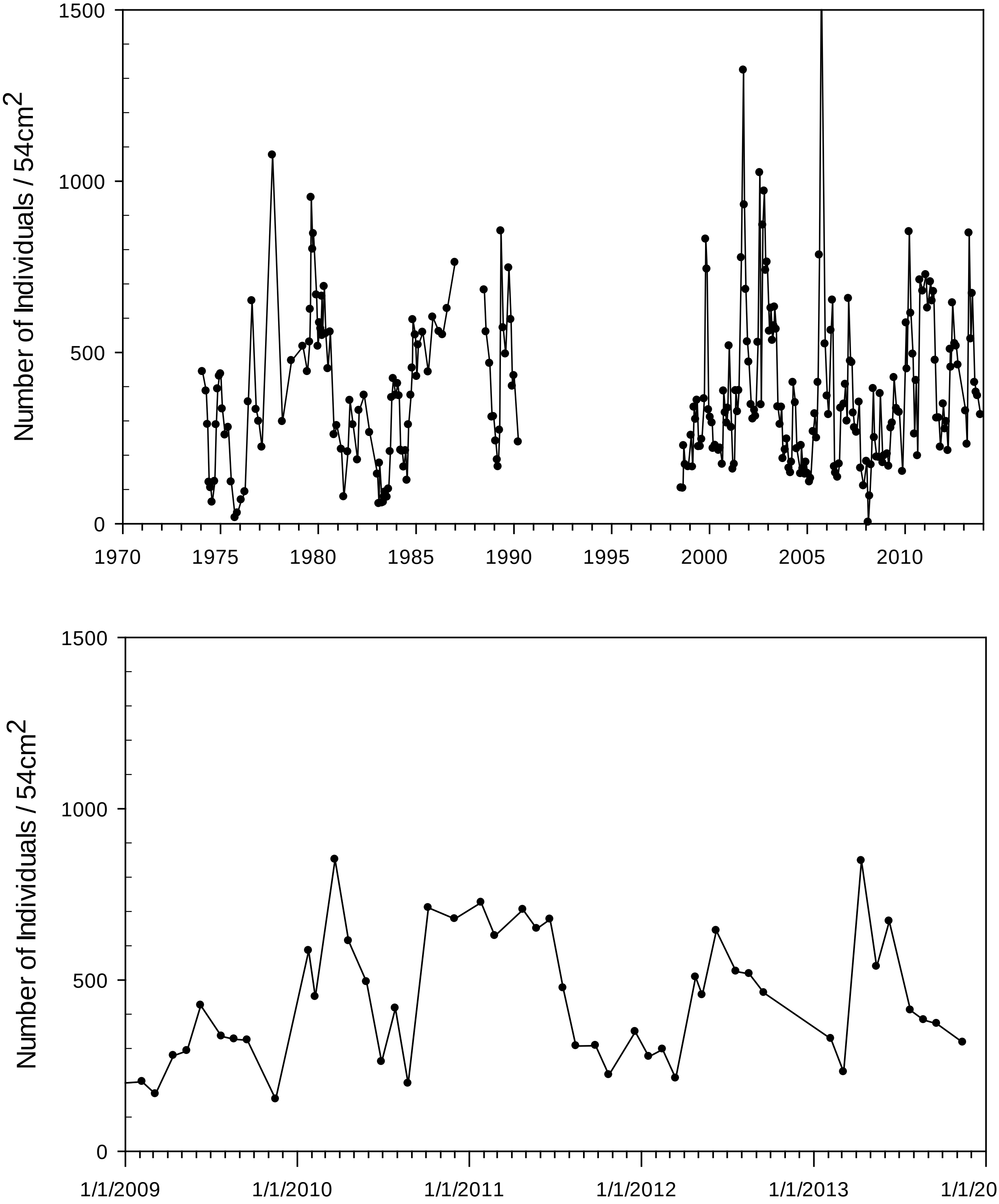


Figure 23. Total average number of individuals present at the Palo Alto site, Calif., 1974–2013. Collections were not made between 1991 and 1998.

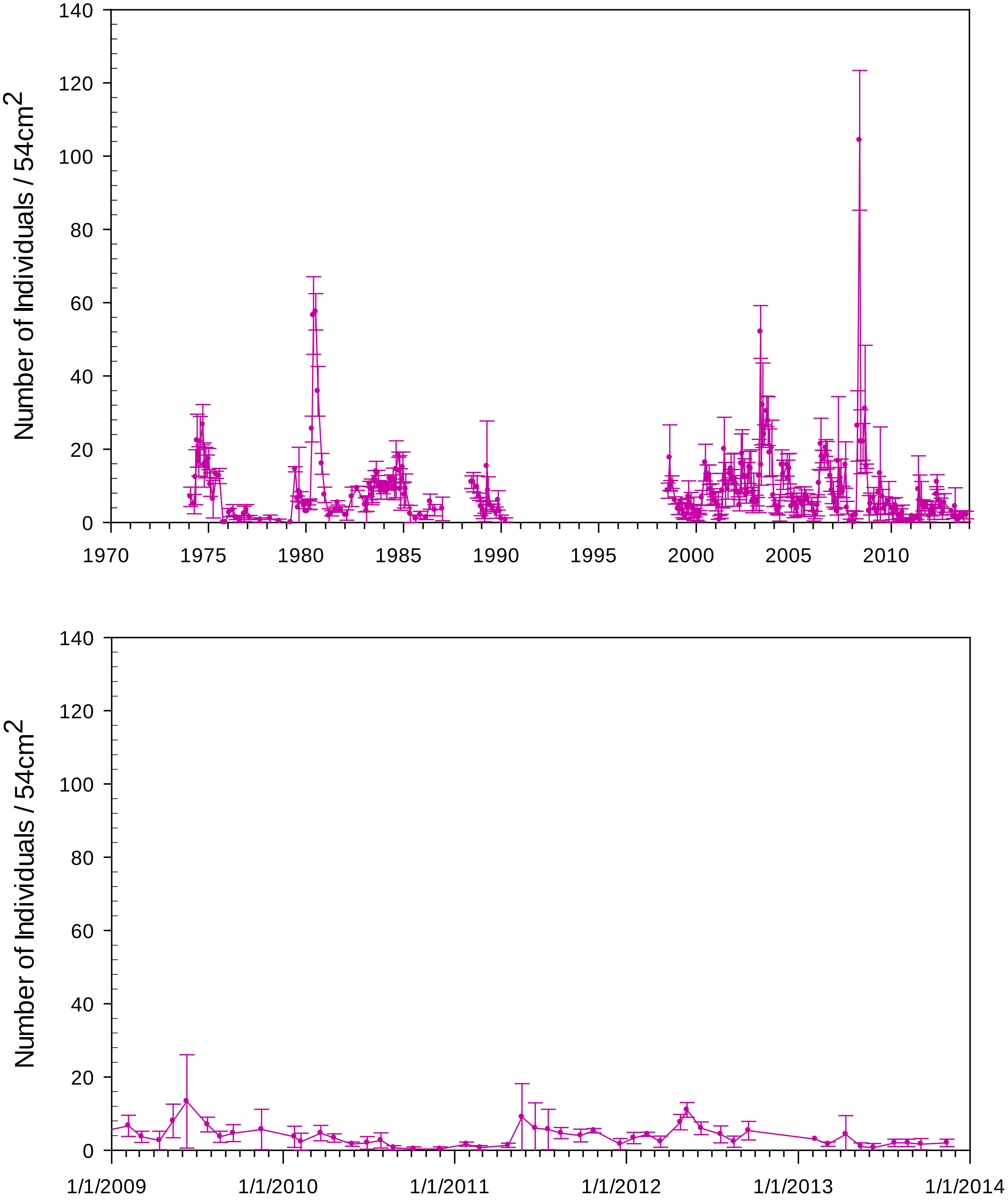


Figure 24. Monthly average abundance of *Macoma petalum*, Palo Alto, Calif., 1974–2013. Error bars represent standard deviation from three replicate samplings. Collections were not made between 1991 and 1998.

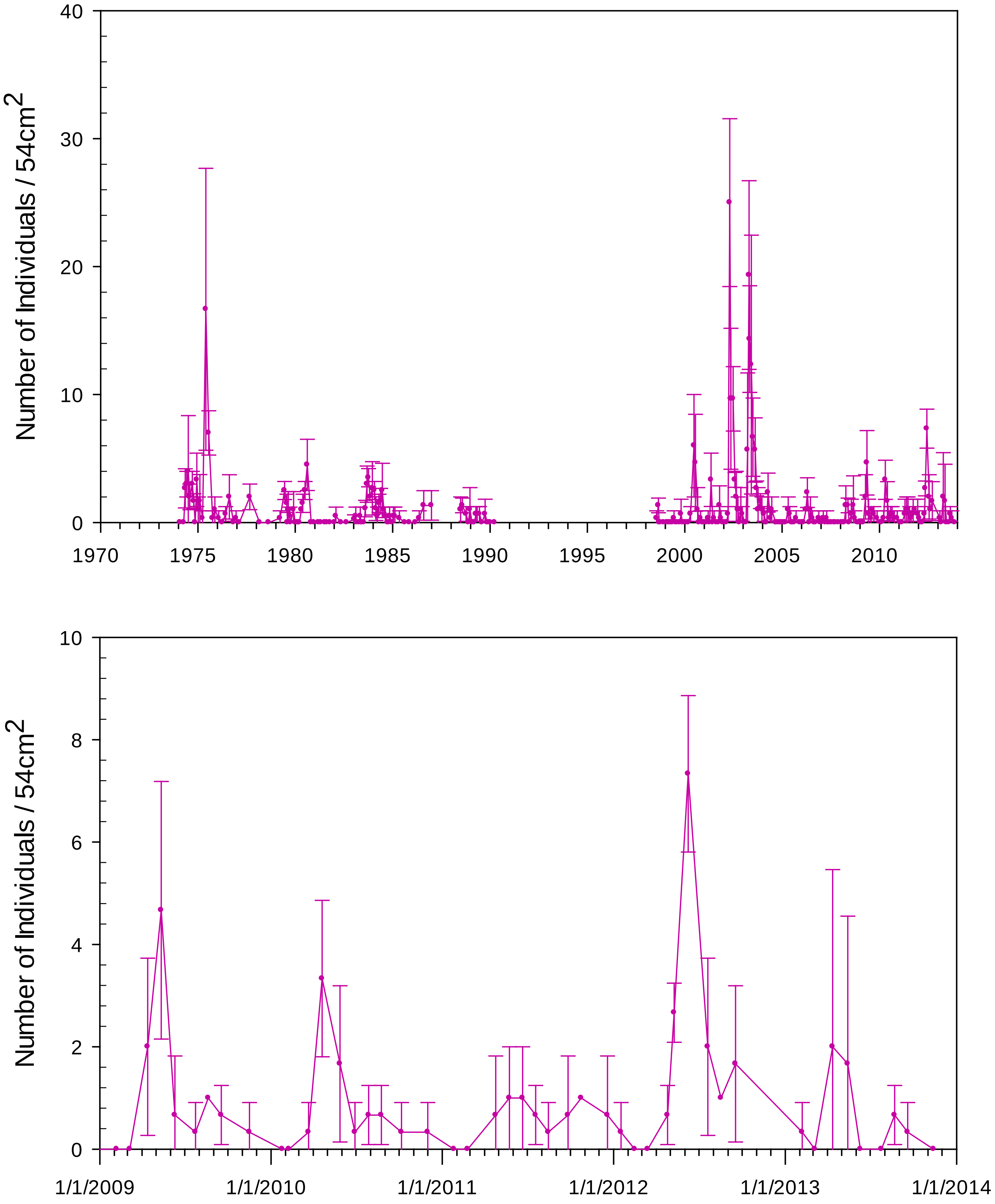


Figure 25. Monthly average abundance of *Mya arenaria,* Palo Alto, Calif., 1974–2013. Error bars represent standard deviation from three replicate samplings. Collections were not made between 1991 and 1998.

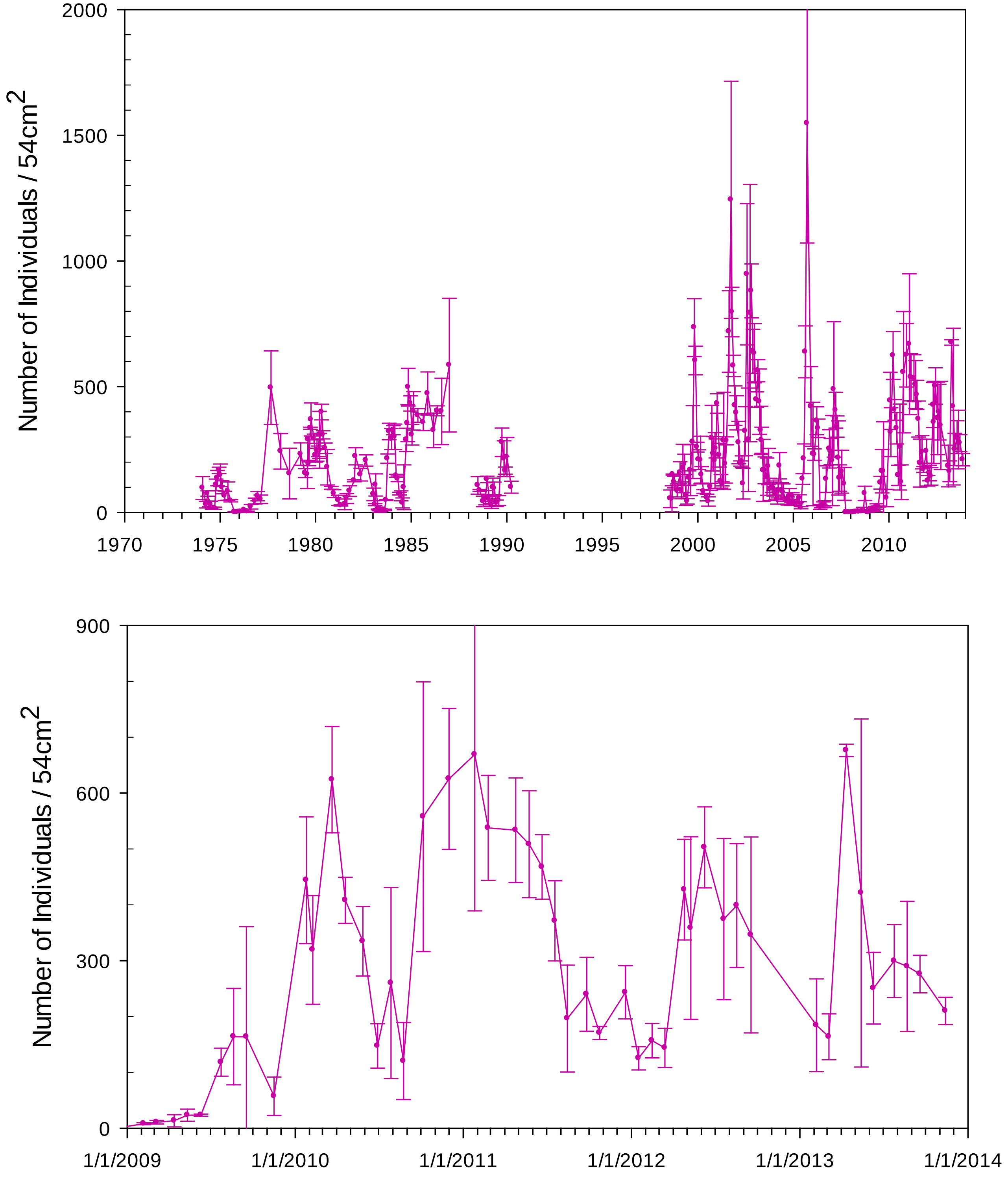


Figure 26. Monthly average abundance of *Gemma gemma,* Palo Alto, Calif., 1974–2013. Error bars represent standard deviation from three replicate samplings. Collections were not made between 1991 and 1998.

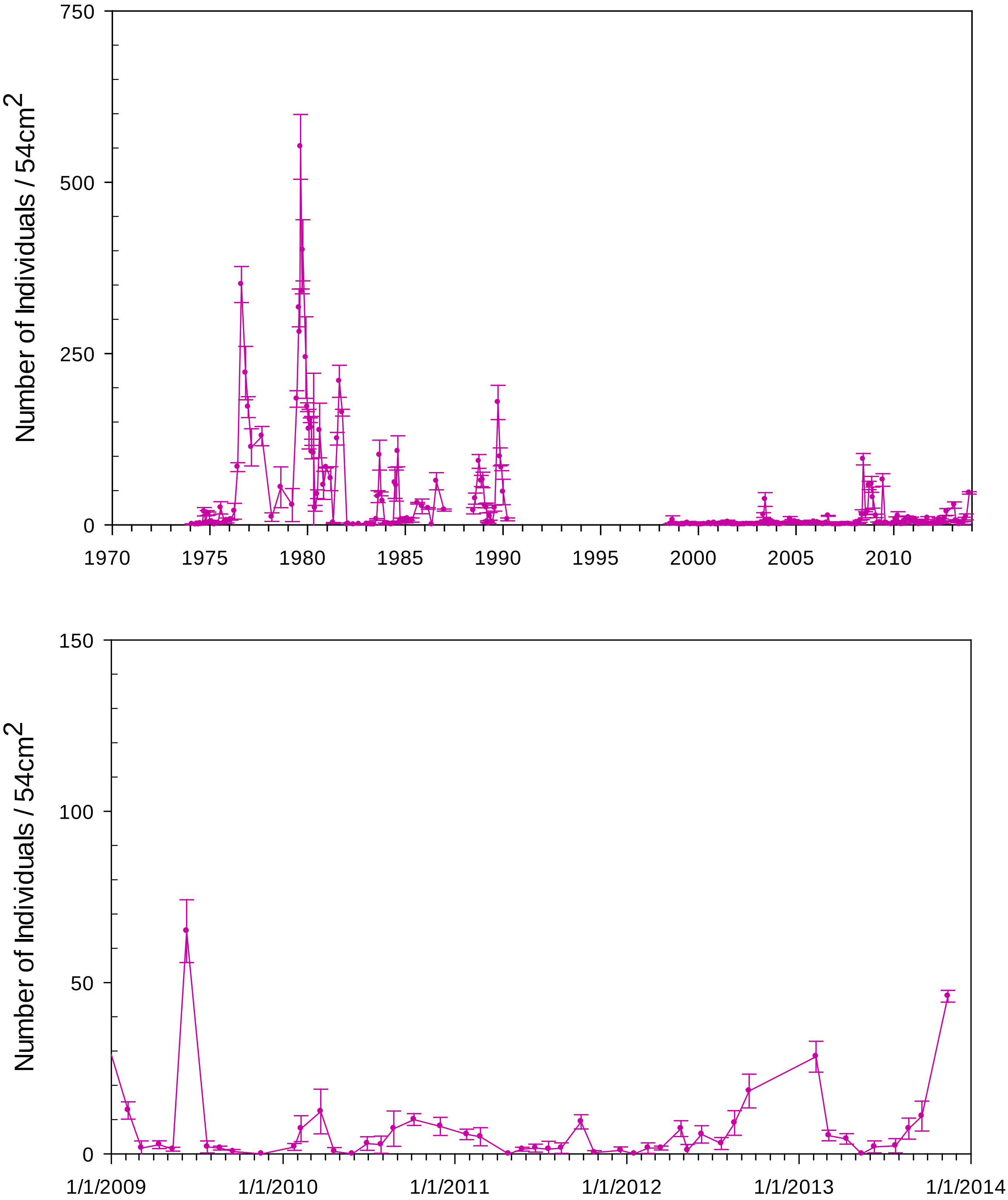


Figure 27. Monthly average abundance of *Ampelisca abdita,* Palo Alto, Calif., 1974–2013. Error bars represent standard deviation from three replicate samplings. Collections were not made between 1991 and 1998.

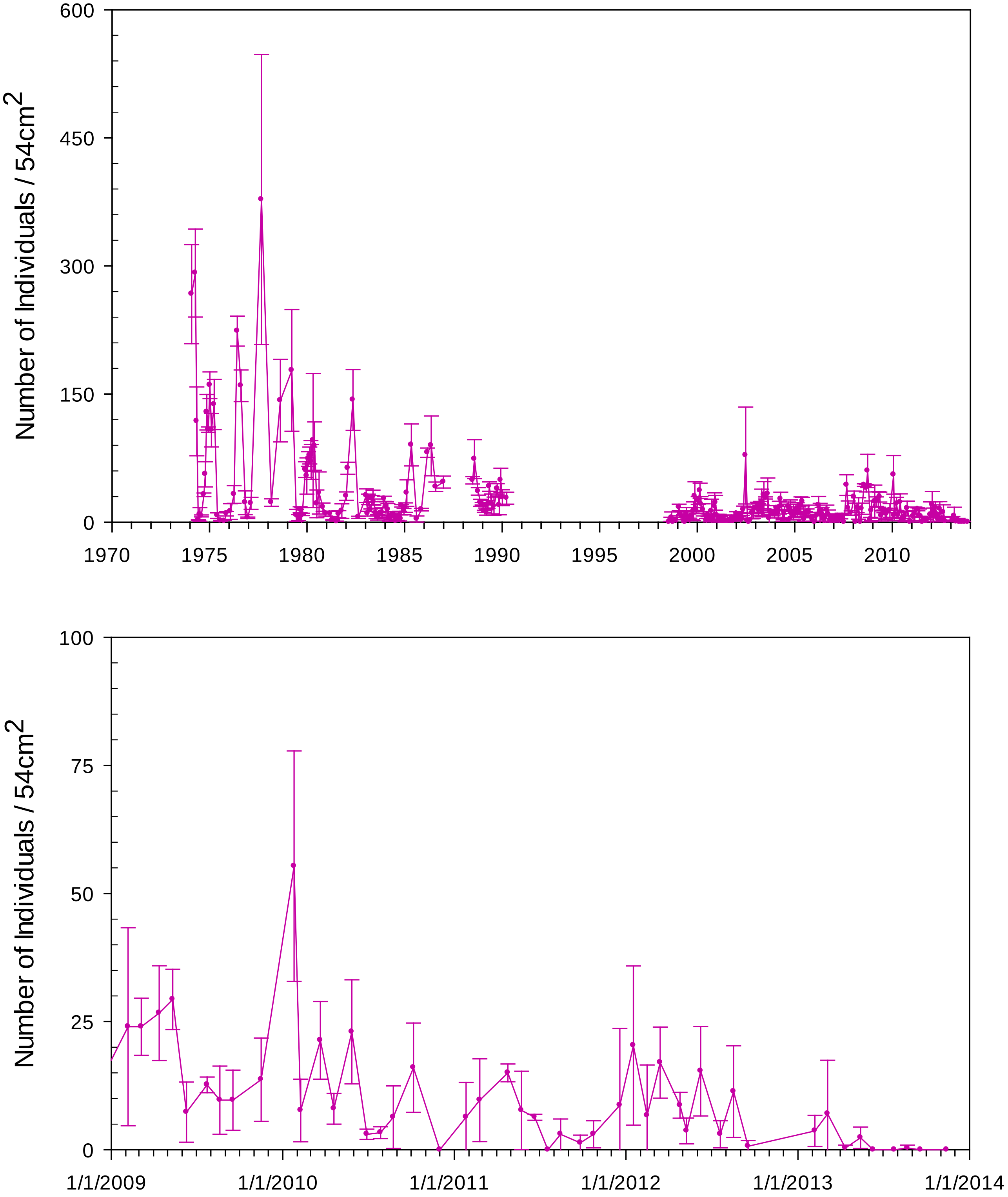
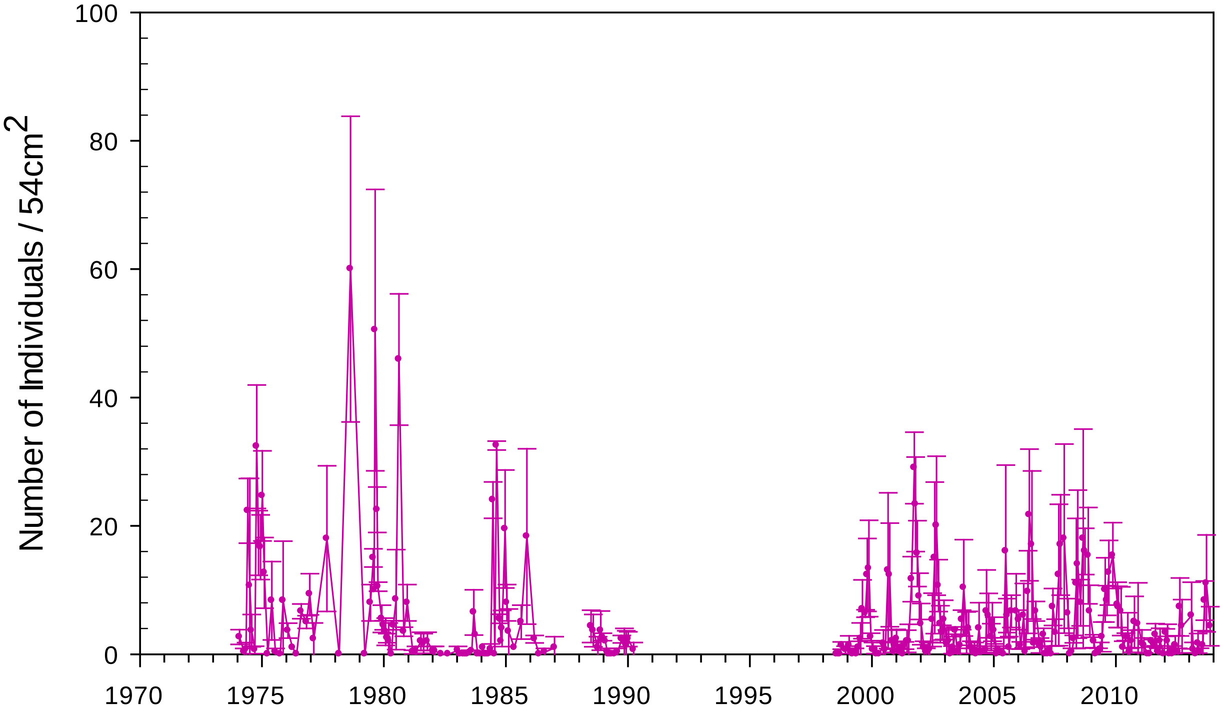


Figure 28. Monthly average abundance of *Streblospio benedicti,* Palo Alto, Calif., 1974–2013. Error bars represent standard deviation from three replicate samplings. Collections were not made between 1991 and 1998.



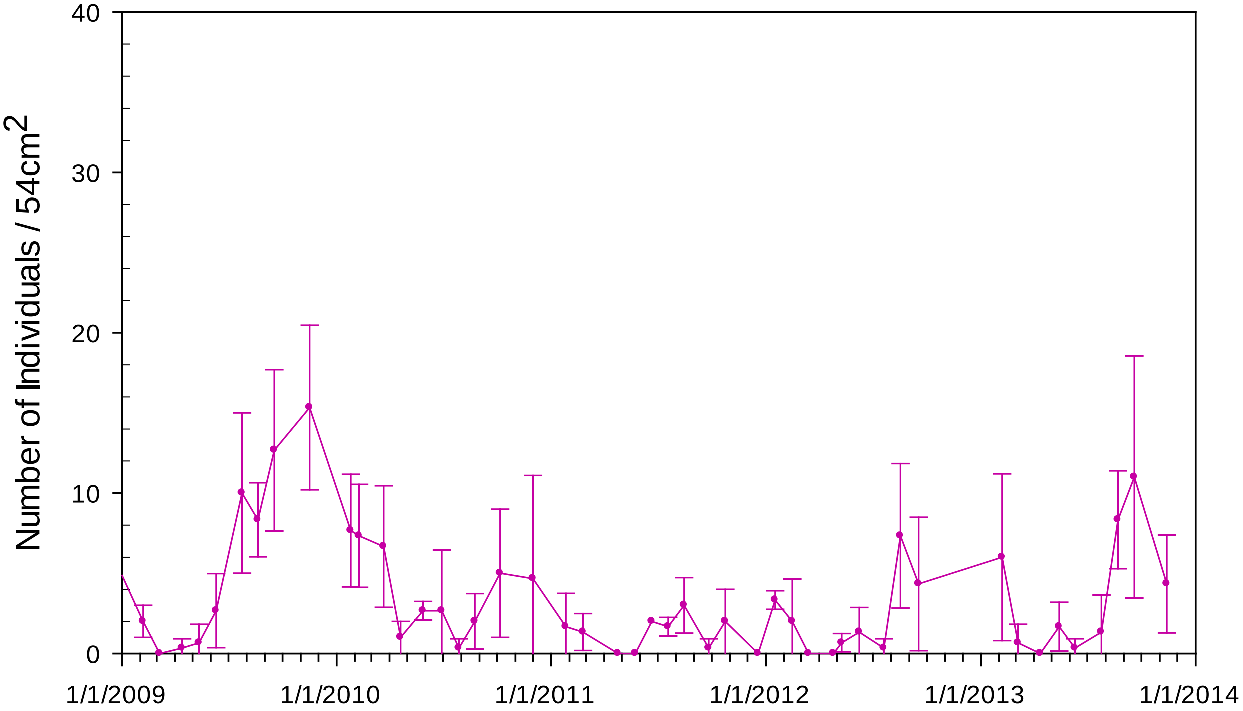


Figure 29. Monthly average abundance of *Grandiderella japonica,* Palo Alto, Calif., 1974–2013. Error bars represent standard deviation from three replicate samplings. Collections were not made between 1991 and 1998.

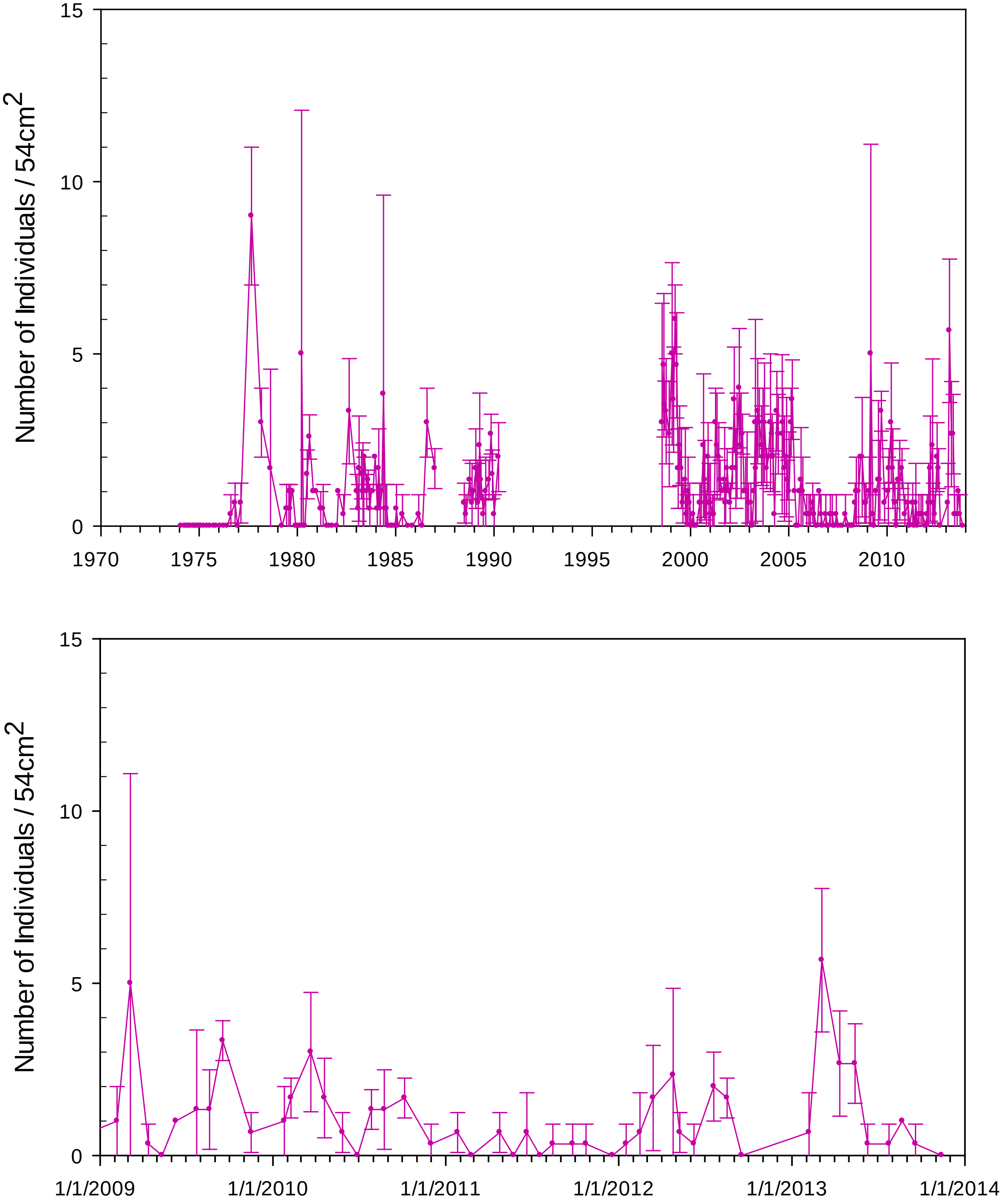


Figure 30. Monthly average abundance of *Neanthes succinea,* Palo Alto, Calif., 1974–2013. Error bars represent standard deviation from three replicate samplings. Collections were not made between 1991 and 1998.

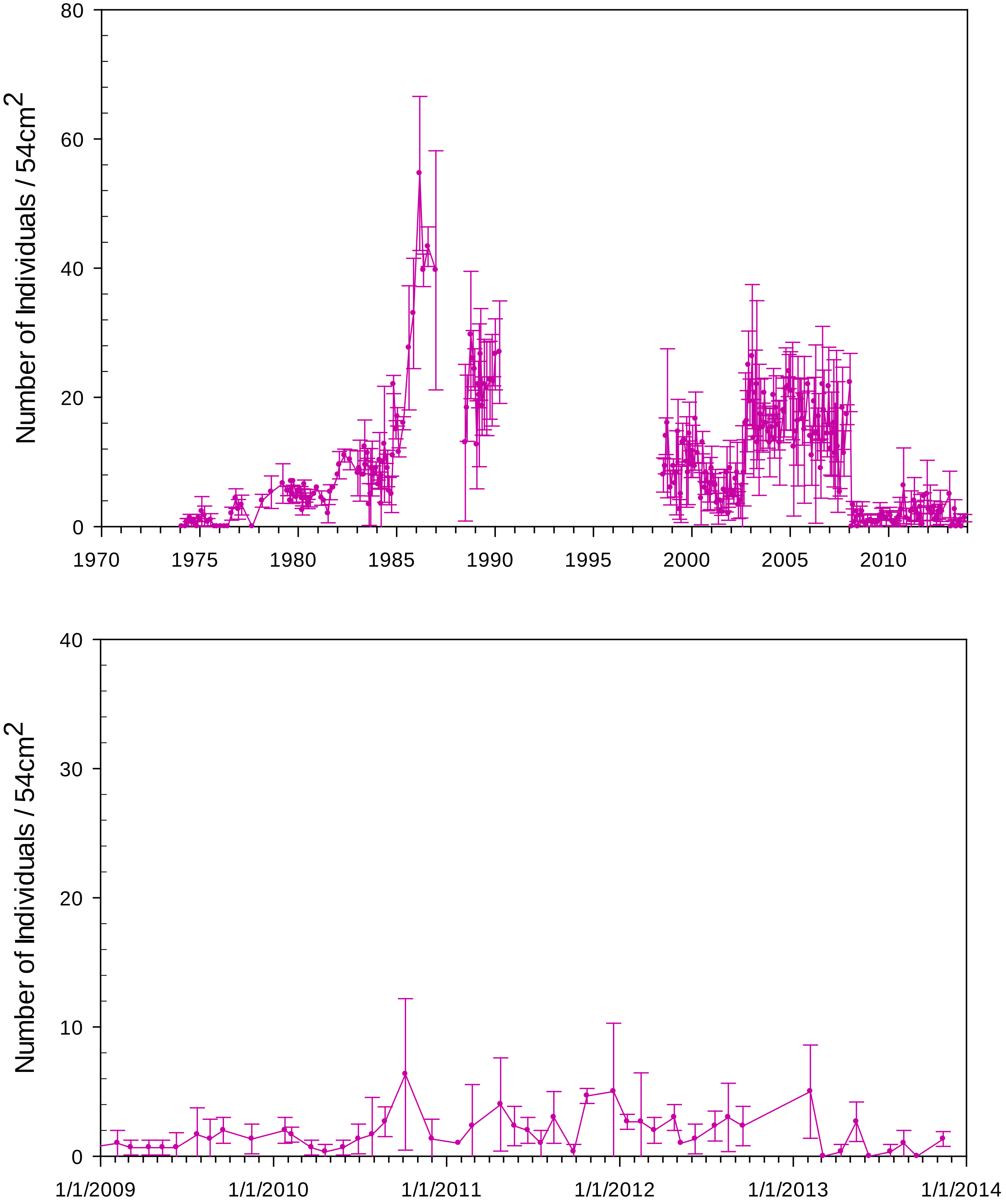


Figure 31. Monthly average abundance of *Heteromastus filiformis,* Palo Alto, Calif., 1974–2013. Error bars represent standard deviation from three replicate samplings. Collections were not made between 1991 and 1998.

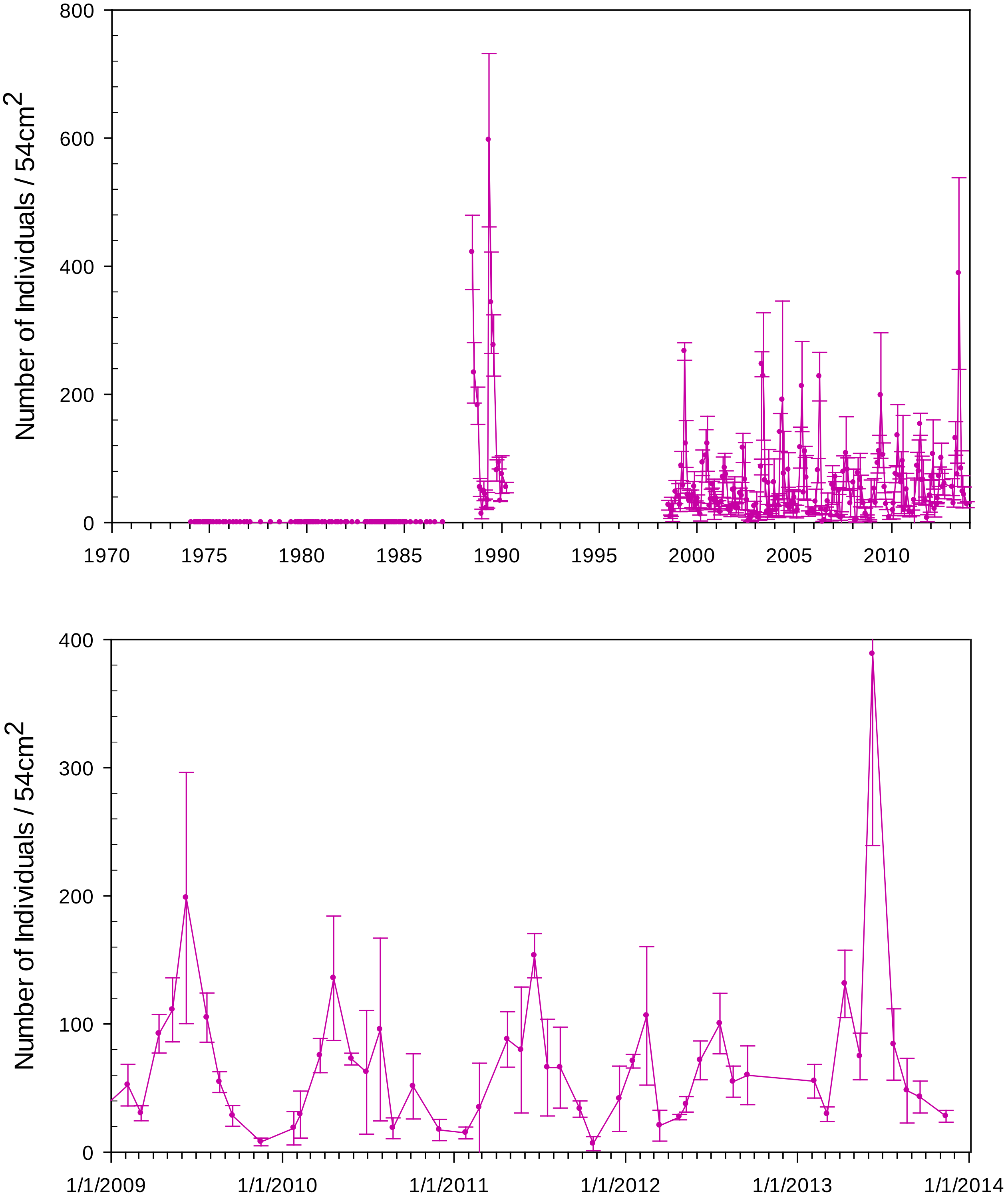


Figure 32. Monthly average abundance of *Nippoleucon hinumensis,* Palo Alto, Calif., 1974–2013. Error bars represent standard deviation from three replicate samplings. Collections were not made between 1991 and 1998.

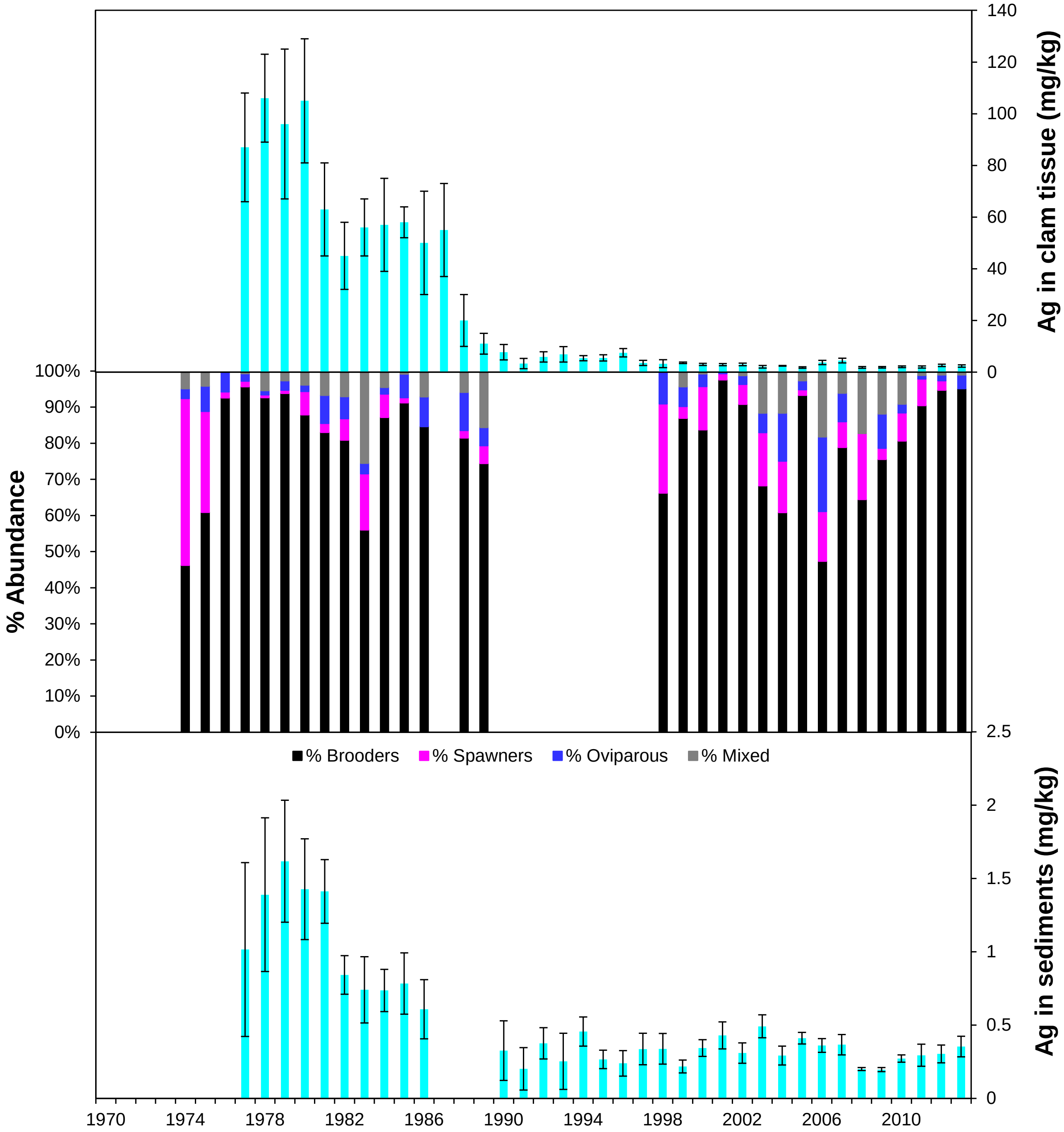


Figure 33. Reproductive mode annual abundance with silver concentrations in the clam *Macoma petalum* and in sediment, Palo Alto, Calif., 1974–2013. Annual abundance data is from August of each year. The reproductive mode of the top ten ranked species for each year is shown: ***Brooder***: broods young and release juveniles as fully functional “miniature adults”; ***Oviparous***: lays eggs in or on sediment; ***Spawner***: releases gametes into water column and juveniles settle out of plankton onto sediment surface after growth in the plankton.

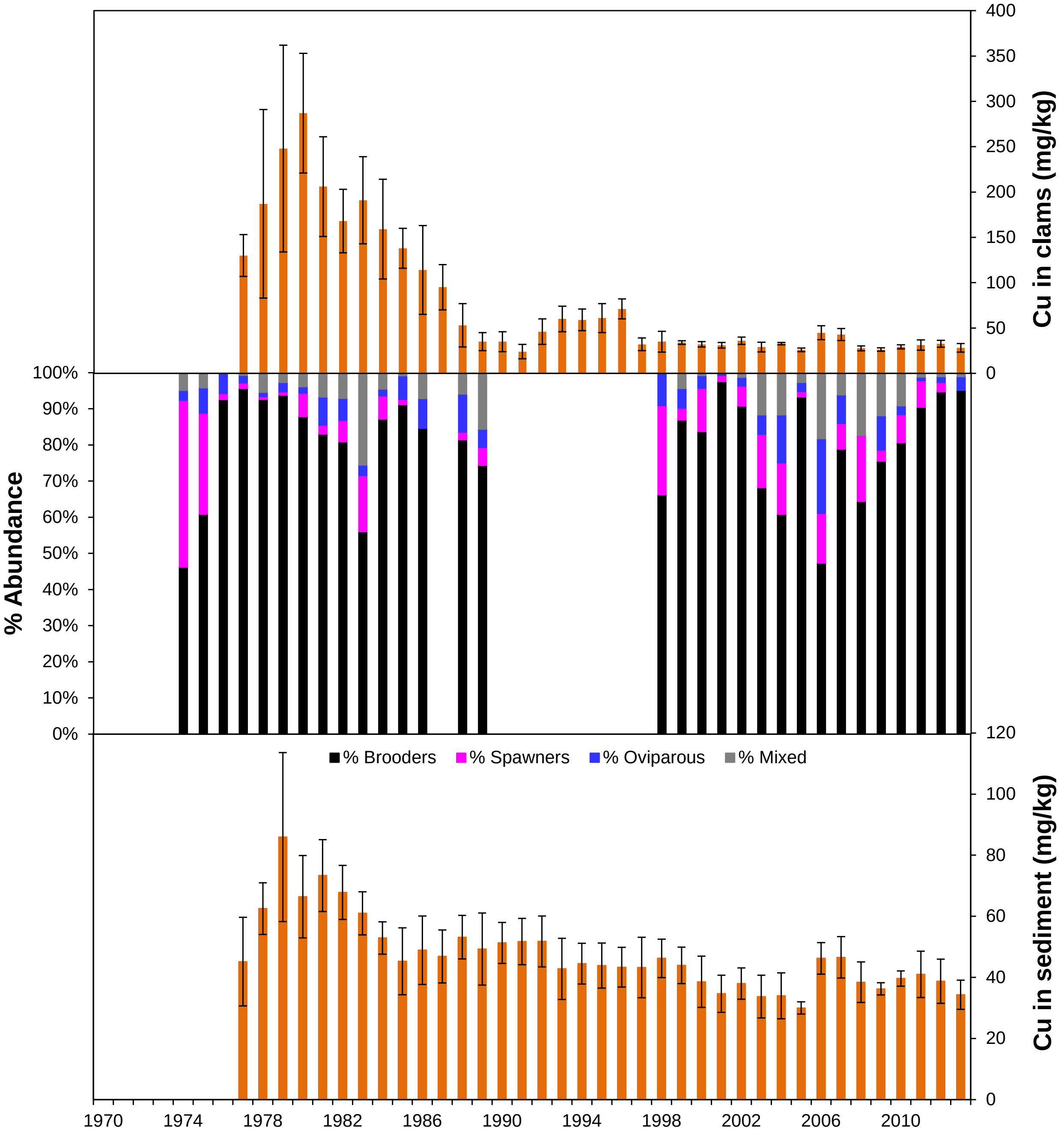


Figure 34. Reproductive mode annual abundance with copper concentrations in the clam *Macoma petalum* and in sediment, Palo Alto, Calif., 1974–2013. Annual abundance data is from August of each year. The reproductive mode of the top ten ranked species for each year is shown: *Brooder*: broods young and release juveniles as fully functional “miniature adults”; *Oviparous*: lays eggs in or on sediment; *Spawner*: releases gametes into water column and juveniles settle out of plankton onto sediment surface after growth in the plankton.

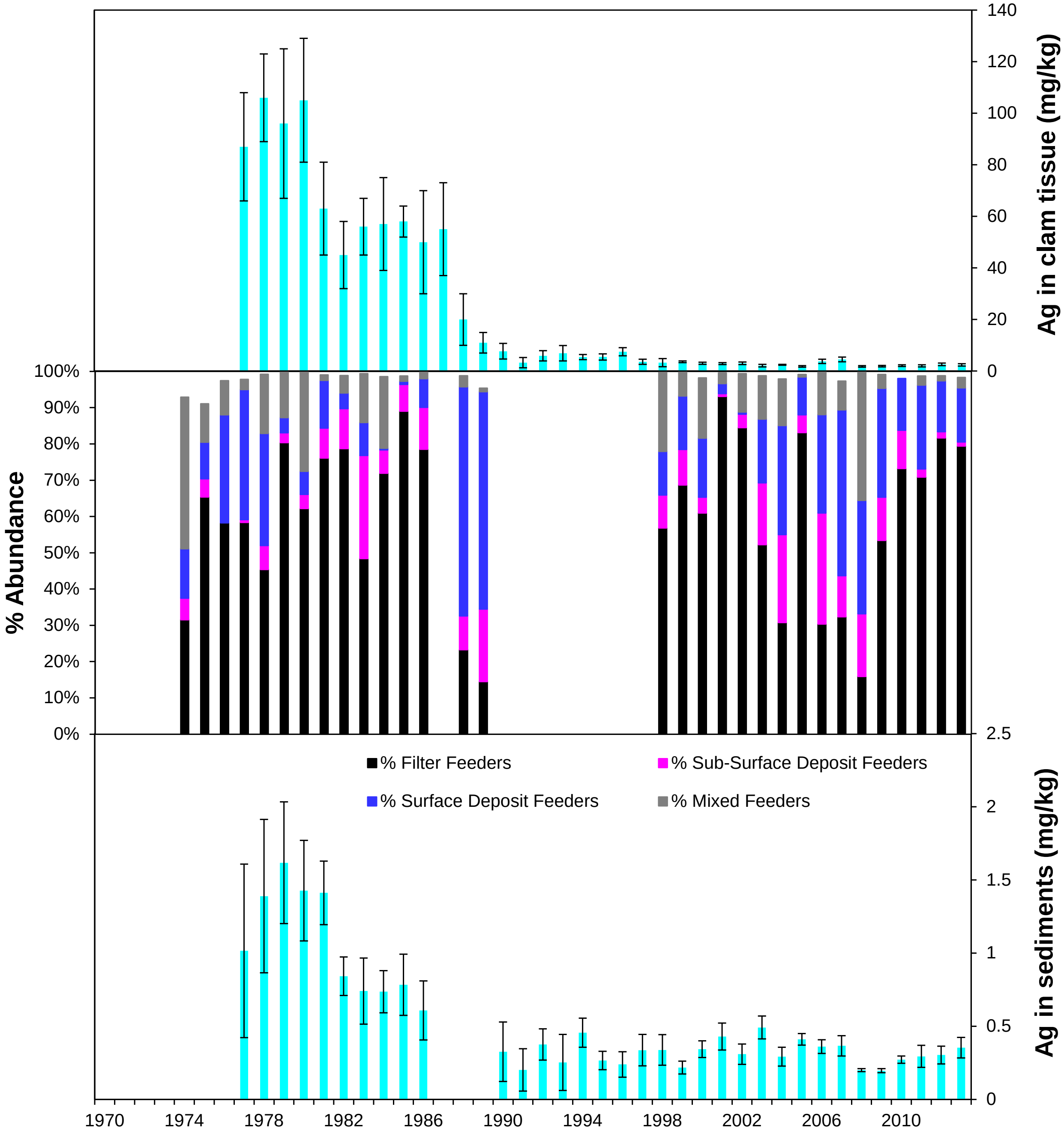


Figure 35. Feeding mode annual abundance with silver concentrations in the clam *Macoma petalum* and in sediment, Palo Alto, Calif., 1974–2013. Annual abundance data is from August of each year. The feeding mode of the top ten ranked species for each year is shown: *Filter*: filters food particles from water column; *Subsurface Deposit*: ingests subsurface sediment and removes food from sediment in gut; *Surface Deposit*: ingests food particles on surface sediment; *Mixed*: capable of filter feeding and surface deposit feeding.

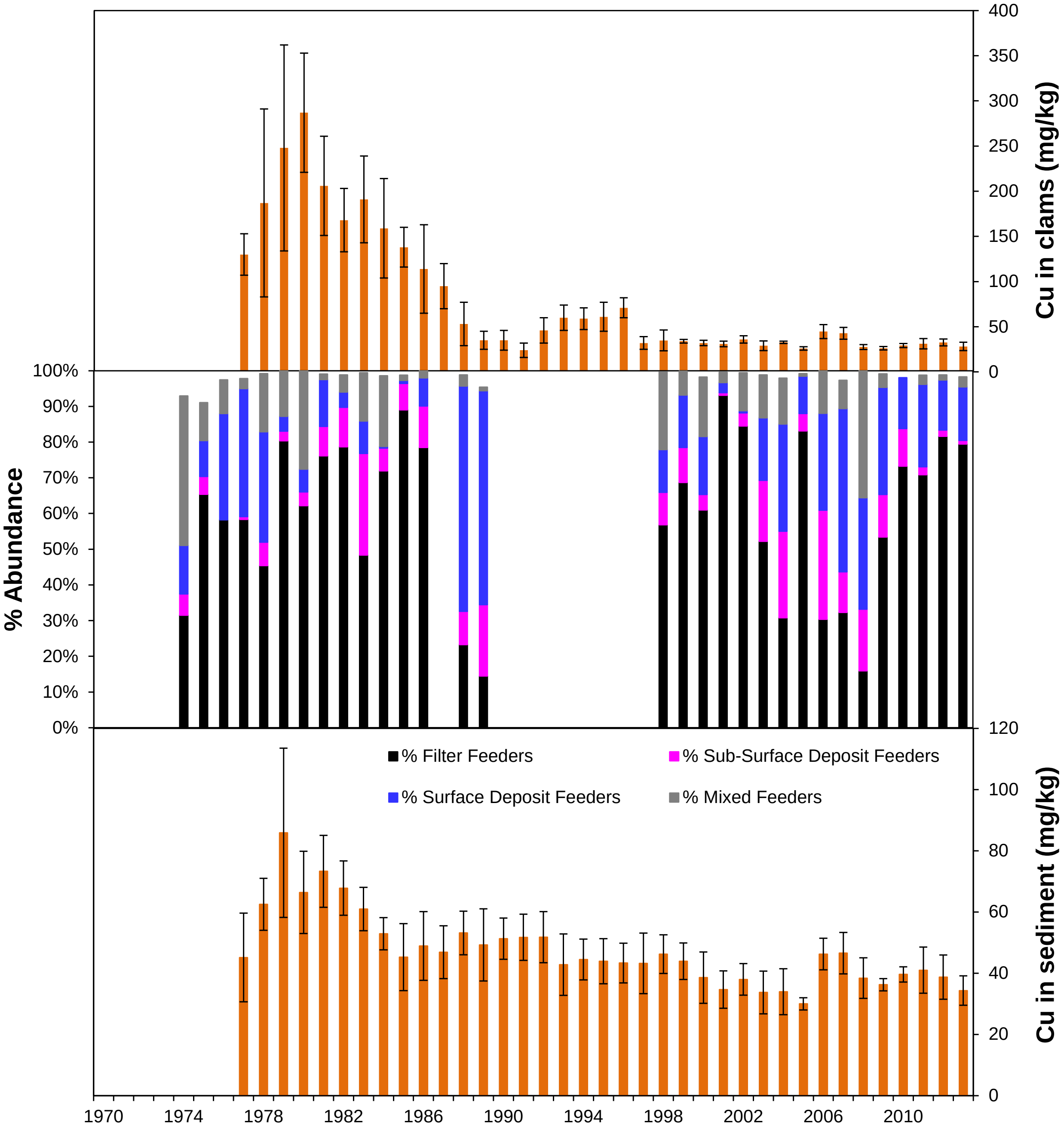


Figure 36. Feeding mode annual abundance with copper concentrations in the clam *Macoma petalum* and in sediment, Palo Alto, Calif., 1974–2013. Annual abundance data is from August of each year. The feeding mode of the top ten ranked species for each year is shown: ***Filter***: filters food particles from water column; ***Subsurface*** **Deposit**: ingests subsurface sediment and removes food from sediment in gut; ***Surface Deposit***: ingests food particles on surface sediment; ***Mixed***: capable of filter feeding and surface deposit feeding.

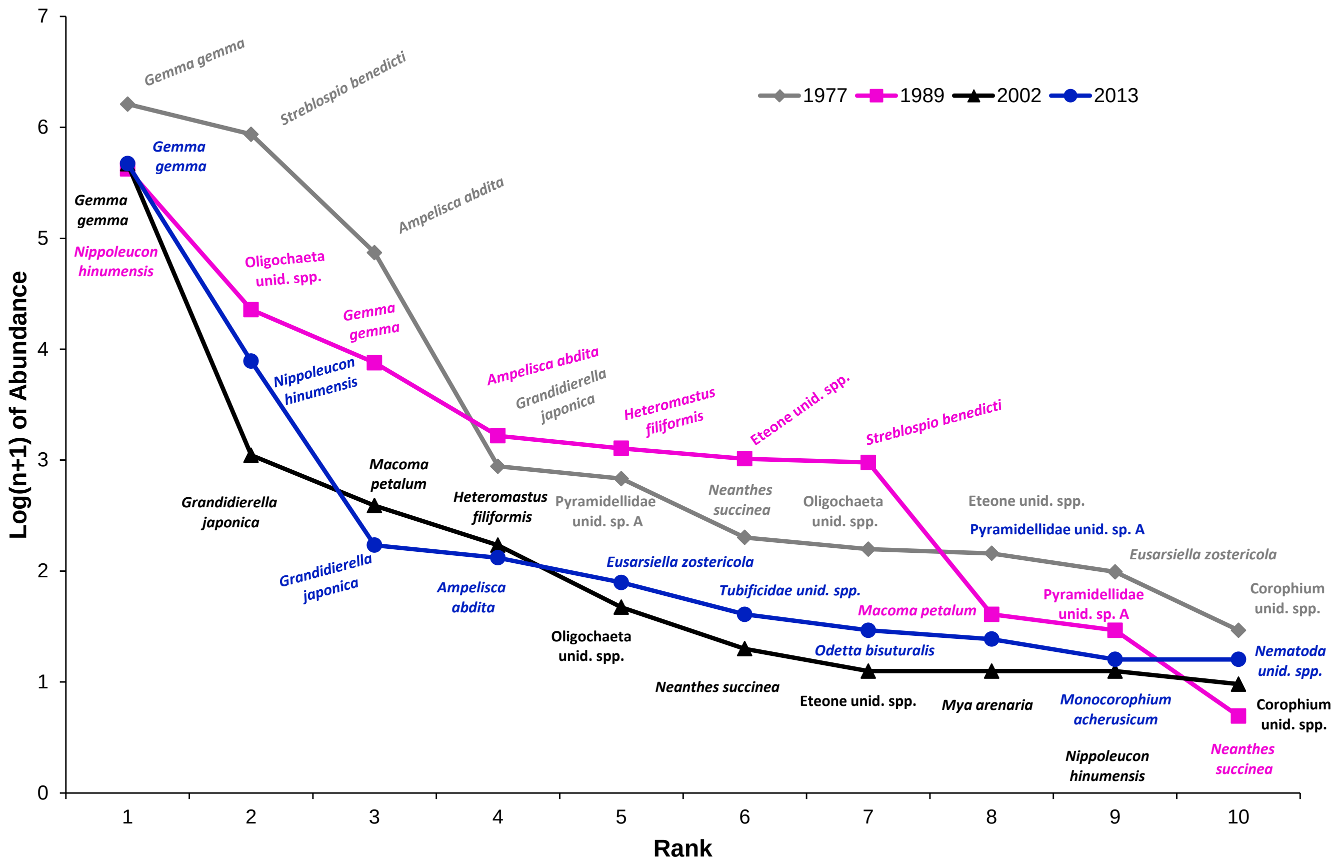


Figure 37. Species rank-abundance for the benthic community, Palo Alto, Calif., 2013.

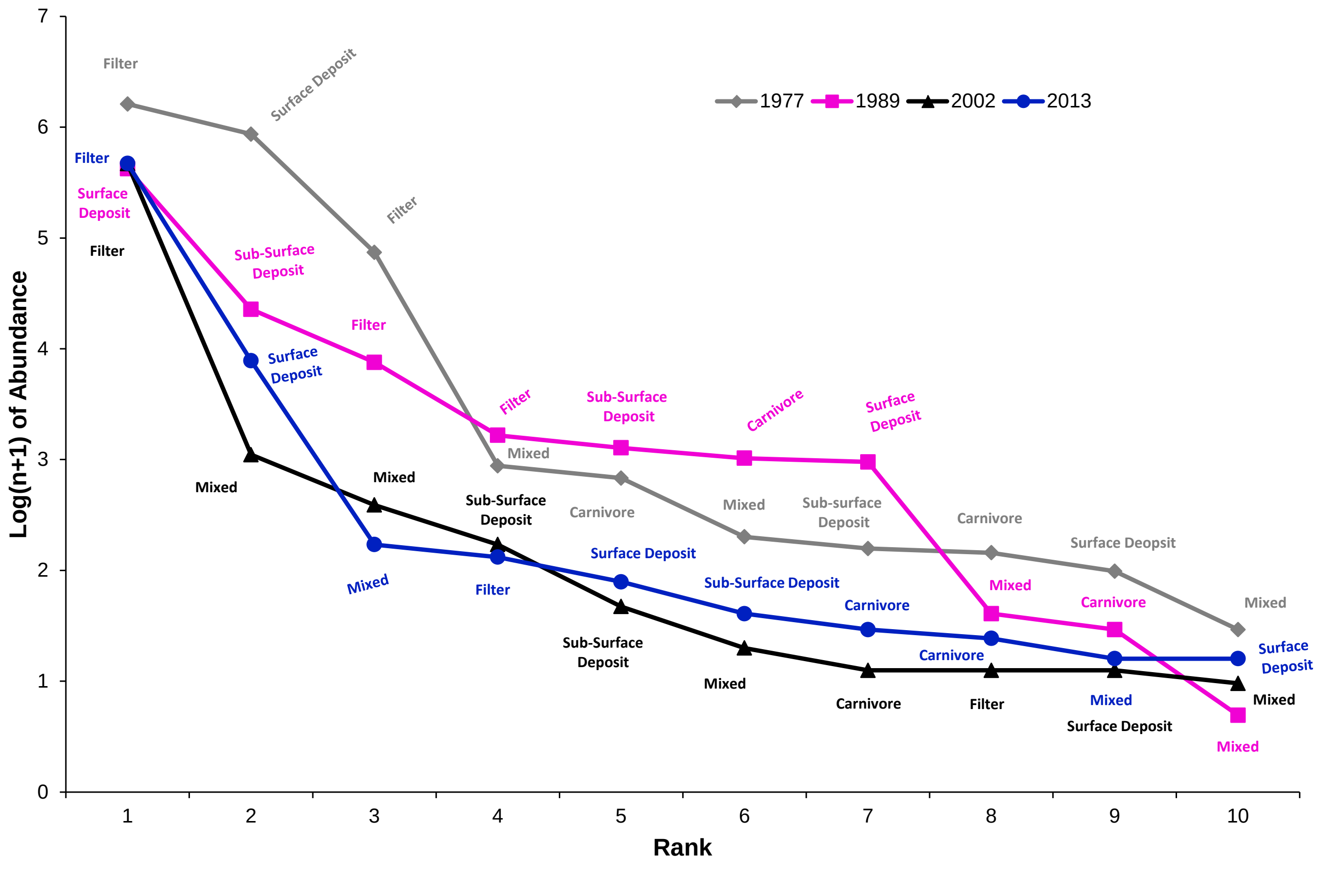


Figure 39. Species rank-abundance identified by feeding mode, Palo Alto, Calif., for 1977, 1989, 2002, and 2013. The feeding mode for each species at each rank is shown: *Filter*: filters food particles from water column; *Subsurface Deposit*: ingests subsurface sediment and removes food from sediment in gut; *Surface Deposit*: ingests food particles on surface sediment; *Mixed*: capable of filter feeding and surface deposit feeding; *Carnivore*: predator on other fauna.

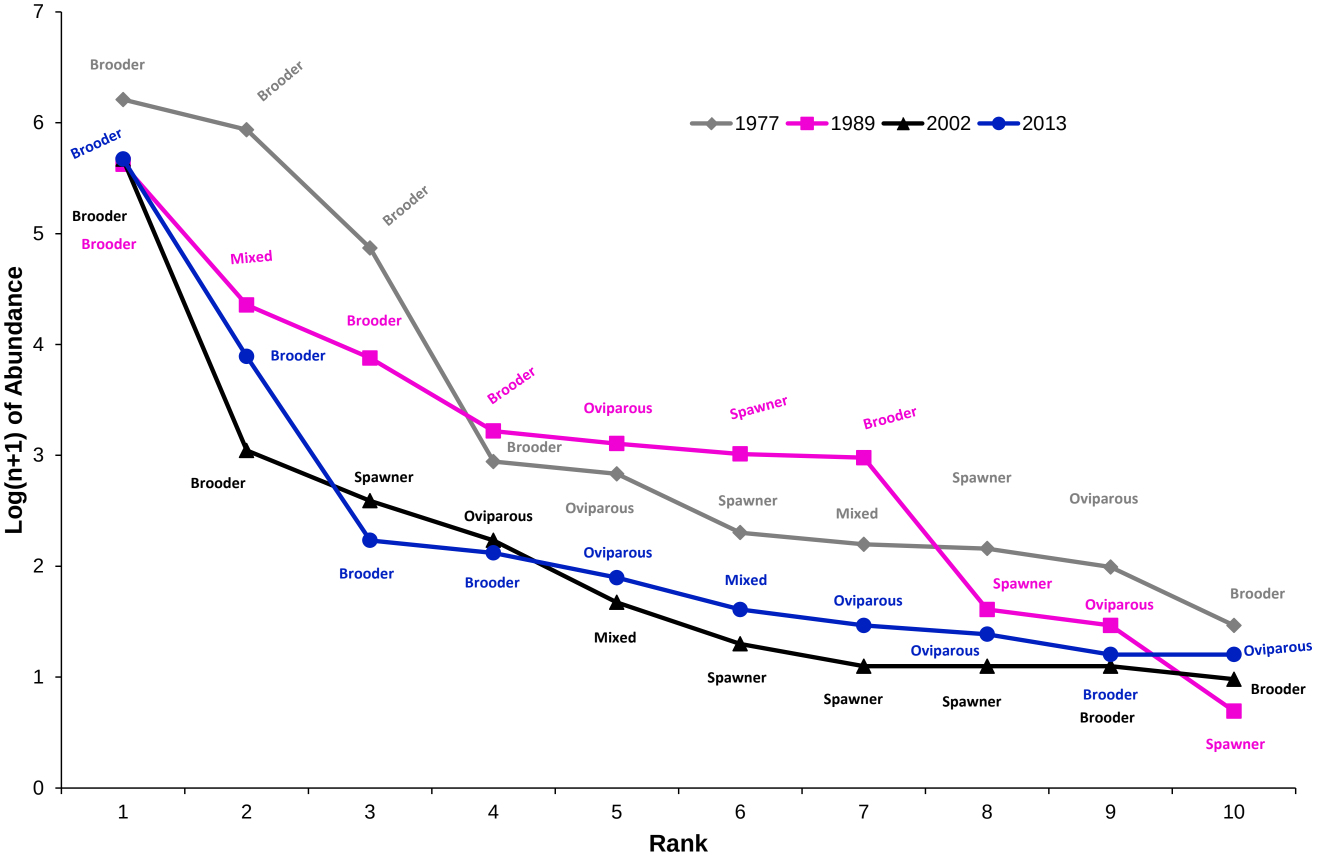


Figure 40. Species rank-abundance data identified by reproductive mode, Palo Alto, Calif., for 1977, 1989, 2002, and 2013. Reproductive mode for each species at each rank is shown: *Brooder*: broods young and release juveniles as fully functional “miniature adults”; *Oviparous*: lays eggs in or on sediment; *Spawner*: releases gametes into water column and juveniles settle out of plankton onto sediment surface after growth in the plankton.

Table 1. Sediment characteristics, salinity, and concentrations of trace metals in sediments, Palo Alto, Calif., 2013.

[Units for Al, Fe, total organic carbon (TOC), and silt/clay are percent of dry weight. Silt/Clay is operationally determined as ≤100 micrometer grain size. Salinity is reported in units of parts per thousand (ppt) for water pooled at the sediment surface during low tide. Elemental concentrations for the monthly samples are reported as the mean ±1 standard deviation (std) for replicate subsamples (n=2). Units are milligram per kilogram dry weight. Results for TOC, silt/clay, and salinity are for a single (n=1) measurement. Means for monthly samples were summarized and reported as the annual mean ± the standard error (SEM) (n=6 or 9). All concentrations are based on near-total extracts, except for silver (Ag), which is based on partial extraction (see text section in Methods). ND means No Data.]

Date	Ag	Cr	Cu	Hg	Ni	Se	Zn	Al (%)	Fe (%)	TOC (%)	Silt/Clay (%)	Salinity (ppt)
1/16/13	0.35±0.0001	115±1	41.3±0.7	0.25	87.6±3.9	0.52	122±5	5.3±0.004	4.3±0.1	1.42	95	20
2/13/13	0.45±0.01	116±1	38.7±0.2	0.25	89.0±0.1	0.50	123±2	5.2±0.1	4.3±0.04	1.50	97	20
3/6/13	0.42±0.003	106±5	33.2±0.8	ND	83.2±0.8	ND	113±2	4.6±0.2	4.1±0.004	1.39	88	19
4/16/13	0.37±0.001	105±3	35.1±0.2	0.21	77.0±1.2	0.42	112±2	4.5±0.1	3.8±0.1	1.13	94	27
5/14/13	0.28±0.001	118±5	39.2±1.4	ND	85.0±1.0	ND	123±4	5.0±0.2	4.3±0.2	1.39	86	25
6/12/13	0.28±0.007	87.5±5.5	28.1±0.2	0.19	66.4±0.3	0.31	93.9±0.4	3.5±0.3	3.4±0.1	0.965	46	28
9/9/13	0.28±0.001	93.3±0.9	27.9±0.1	0.19	65.2±0.7	0.30	94.6±0.8	3.7±0.03	3.3±0.04	0.856	77	30
10/22/13	0.32±0.003	102±2	34.3±0.3	ND	75.4±0.6	ND	118±1	4.2±0.003	3.6±0.1	1.25	81	28
12/10/13	0.44±0.001	101±5	31.3±3.7	0.21	85.1±0.4	0.35	118±3	4.0±0.4	4.0±0.1	1.29	93	26
Annual Mean:	0.35	105	34.3	0.22	79.3	0.40	113	4.4	3.9	1.24	84	25
SEM:	0.02	3	1.6	0.01	3.0	0.04	4	0.2	0.1	0.07	5	1

Table 2. Concentrations of trace metals in the clam *Macoma petalum*, Palo Alto, Calif., 2013.

[Monthly data are the mean and standard deviation for replicate composites (n=11–13, n=3 for Se and Hg). Means for monthly samples were summarized and reported as the annual mean ± the standard error (SEM) (n=6 or 9). All concentrations are based on near-total extracts. Elemental concentrations are milligram per kilogram soft tissue dry weight. The condition index (CI) is the soft tissue weight in milligrams of a clam of 25-millimeter shell length. ND means No Data.]

Date	Ag	Cr	Cu	Hg	Ni	Se	Zn	Condition Index
01/16/13	4.81±1.46	2.56±0.81	53.1±13.6	0.29±0.02	6.92±2.39	4.65±0.21	328±98	91
02/13/13	3.45±1.20	2.67±0.57	41.0±11.5	0.26±0.02	5.66±0.98	5.05±0.21	294±58	86
03/06/13	1.40±0.55	1.11±0.40	16.9±4.0	ND	2.97±0.70	ND	190±20	166
04/16/13	0.825±0.286	0.817±0.327	10.9±1.7	0.13±0.01	2.74±0.39	3.39±0.35	181±27	231
05/14/13	0.956±0.586	1.03±0.43	17.3±3.8	ND	4.22±0.76	ND	238±40	166
06/12/13	1.00±0.31	0.930±0.229	17.7±2.7	0.16±0.01	4.58±1.07	3.53±0.09	206±29	153
09/09/13	2.69±1.49	1.39±0.74	29.1±17.6	0.24±0.03	4.85±1.08	3.22±0.33	195±67	114
10/22/13	2.84±0.58	1.60±0.48	40.0±17.0	ND	4.56±0.97	ND	186±50	107
12/10/13	3.47±1.52	2.25±0.82	27.8±6.9	0.23±0.09	4.92±1.12	4.55±0.38	171±39	108
Annual Mean:	2.38	1.60	28.2	0.22	4.60	4.07	221	136
SEM:	0.47	0.24	4.7	0.02	0.42	0.32	18	16

Table 3. Reproduction data for *Macoma petalum*, Palo Alto, Calif., 2013.

[Data are percentage of clams in each stage of reproduction. Reproductive = the percentage of clams in Active, Ripe, and Spawning stages. Non-Reproductive = the percentage of clams in Inactive and Spent stages. Spent means the clams have released all their gametes. N refers to the number of clams that were analyzed. ND means No Data.]

Date	Inactive	Active	Ripe	Spawning	Spent	N	Repro-ductive	Non-Repro-ductive
01/16/13	0	30	70	0	0	10	100	0
02/13/13	0	0	100	0	0	10	100	0
03/16/13	0	0	100	0	0	10	100	0
04/16/13	0	0	10	90	0	10	100	0
05/14/13	60	0	0	0	40	10	0	-100
06/12/13	30	0	0	0	70	10	0	-100
09/13/13	50	50	0	0	0	10	50	-50
10/22/13	10	80	10	0	0	10	90	-10
12/10/13	0	10	90	0	0	10	100	0

Appendixes

Appendix 1. Statistical summary of percentage of sediment in samples composed of clay- and silt-sized particles, collected from Palo Alto, Calif., 1994–2013.

[Statistical results are for percent fine-grained particles (silt and clay, ≤100 micrometer) observed each month from 1994–2013. Data for percent fines for 2004, which contain unquantifiable biases due to errors in sample processing, are not included in these statistical calculations. # of samples refers to the number of times data were collected in that respective month during the span of collection years.]

MONTH	# OF SAMPLES	MAXIMUM	MINIMUM	MEAN	MEDIAN
January	18	96	46	76	76
Febrzuary	18	98	32	82	87
March	18	98	24	80	85
April	18	96	50	85	86
May	13	95	35	76	80
June	18	95	46	71	69
July	1	69	69	69	69
August	1	48	48	48	48
September	18	84	25	57	57
October	14	84	39	63	60
November	3	66	50	59	61
December	18	95	48	74	74

Appendix 2. Mercury and selenium concentrations (mg/kg dry weight) determined in sample splits of surface sediments and *Macoma petalum* collected at Palo Alto, Calif., 2013.

[Two sediment samples and four clam tissue samples were split and analyzed for Se and Hg. The split results are shown here.]

DATE	SEDIMENT		*M. PETALUM*	
04/16/13				
12/10/13				

Appendix 3. Results of the analyses of National Institute of Science and Technology (NIST) Standard Reference Material 2709a (San Joaquin Soil) and 2711a (Montana II Soil) for elements.

Constituent	Number of analyses	Dilution ratio	Certified Concentration (µg/g)	Mean SRM recovery, in percent	95-percent confidence interval for SRM recovery (percent)
SRM 2709a - Near-total metal extraction					
Aluminum	18	1:10	73,700	51.2	49.3 – 53.1
Arsenic	18	1:10	10.5	68.0	66.4 – 69.7
Cadmium	18	1:10	0.371	54.8	52.7 – 56.8
Chromium	18	1:10	130	79.6	77.7 – 81.6
Copper	18	1:10	33.9	81.4	79.3 – 83.5
Iron	18	1:10	33,600	87.6	86.2 – 89.1
Lead	18	1:10	17.3	66.1	65.1 – 67.1
Manganese	18	1:10	529	87.3	85.7 – 88.9
Nickel	18	1:10	85	80.3	78.9 – 81.6
Silver	18	1:10	unknown		–
Vanadium	18	1:10	110	81.5	78.3 – 84.7
Zinc	18	1:10	103	87.5	86.0 – 89.0
SRM 2711a - Near-total metal extraction					
Aluminum	18	1:10	67,200	44.7	43.1 – 46.3
Arsenic	18	1:10	107	82.8	81.9 – 83.6
Cadmium	18	1:10	54.1	98.3	97.4 – 99.3
Chromium	18	1:10	52.3	96.2	93.4 – 99.0
Copper	18	1:10	140	101.4	100 – 102
Iron	18	1:10	28,200	84.1	82.7 – 85.5
Lead	18	1:10	1,410	92.1	91.3 – 92.9
Manganese	18	1:10	675	77.1	76.2 – 78.1
Nickel	18	1:10	21.7	80.4	79.5 – 81.4
Silver	18	1:10	6	58.7	56.9 – 60.5
Vanadium	18	1:10	81	81.3	78.3 – 84.4
Zinc	18	1:10	414	88.8	87.8 – 89.8
SRM 2711a - Partial metal extraction					
Aluminum	18	1:10	67,200	3.39	3.38 – 3.40
Arsenic	18	1:10	107	54.2	54.0 – 54.3
Cadmium	18	1:10	54.1	80.6	80.5 – 80.7
Chromium	18	1:10	52.3	1.63	1.62 – 1.65
Copper	18	1:10	140	54.1	53.9 – 54.2
Iron	18	1:10	28,200	3.28	3.26 – 3.30
Lead	18	1:10	1,410	70.4	70.3 – 70.5
Manganese	18	1:10	675	41.1	40.9 – 41.3
Nickel	18	1:10	22	13.7	13.6 – 13.7
Silver	18	1:10	6	63.7	63.5 – 63.9
Vanadium	18	1:10	81	6.38	6.36 – 6.40
Zinc	18	1:10	414	30.5	30.4 – 30.6

Appendix 4. Observed and certified values for inorganic elements in NIST Standard Reference Material 2976 (Mussel Tissue) and 1566b (Oyster Tissue) prepared in 2013.

[MRL means Method Reporting Level. Samples were not diluted prior to analysis.]

Constituent	Number of analyses	Certified Concentration (µg/g)	Mean SRM recovery, in percent	95-percent confidence interval for SRM recovery (percent)
SRM 2976				
Chromium	20	0.5	109	105 – 113
Copper	27	4.02	90.7	89.7 – 91.6
Lead	27	1.19	77.1	76.0 – 78.2
Nickel	27	0.93	68.9	68.2 – 69.7
Silver	27	0.011	<MRL	--
Vanadium	27	unknown	unknown	unknown
Zinc	27	137	95.7	94.6 – 96.9
SRM 1566b				
Chromium	26	unknown	unknown	unknown
Copper	26	71.6	93.5	93.2 – 93.7
Lead	26	0.308	207	204 – 211
Nickel	26	1.04	95.0	94.2 – 95.7
Silver	26	0.666	95.8	94.2 – 97.3
Vanadium	26	0.577	95.4	94.0 – 96.7
Zinc	26	1,424	59.7	59.1 – 60.3

Appendix 5. Method detection limits (MDL) and reporting levels (MRL) for ICP-OES methods.

[Concentration is reported as microgram per milliliter (µg/mL).]

METHOD	MARKER	AG	AL	CD	CR	CU	FE	MN	NI	PB	V	ZN
Sediment	MDL	0.0005	0.0046	0.0002	0.0019	0.0008	0.0030	0.0004	0.0003	0.0024	0.0002	0.0003
	MRL	0.0011	0.0093	0.0004	0.0038	0.0016	0.0060	0.0009	0.0006	0.0047	0.0005	0.0005
Tissue	MDL	0.0004	0.0095	0.0004	0.0016	0.0010	0.0106	0.0011	0.0004	0.0020	0.0047	0.0021
	MRL	0.0008	0.0190	0.0008	0.0031	0.0020	0.0211	0.0023	0.0009	0.0040	0.0094	0.0041

Appendix 6. Observed and certified concentrations of mercury and selenium in standard reference materials (SRM) analyzed in 2013.

[Concentration is reported as mg/kg. The mercury recovery results for NIST-2976 and NIST-1566b are the result of a different sample run than the mercury recovery results for PACS-2. The NIST-2976 and NIST-1566b standards were re-run after initially low recoveries: 68 percent and 31 percent, respectively.]

SRM	Mercury				Selenium		
	Observed	Certified	Percent Recovery		Observed	Certified	Percent Recovery
PACS-2							
NIST-2976							
NIST-1566b							
TORT-2							

Appendix 7. Annual mean silver and annual mean copper in sediments and the clam *Macoma petalum*, Palo Alto, Calif., 1977–2013.

[Mean, Median, Minimum, and Maximum are calculated from all data between 1977–2013. The 2013 column presents the current 2013 data. Values for the 2013 column are annual (grand) means standard errors of those means, the rest of the columns are statistics from the grand means of 1977–2013. Samples were collected between January and December of each year. Units are milligrams per kilogram dry weight of soft tissue for the clam (*Macoma petalum*) and milligram per kilogram dry weight for sediment. HCl refers to hydrochloric acid extractable copper and silver.]

Type of Analysis	2013	Mean	Median	Minimum	Maximum
Sediment—HCl	16±1	23	19	13	55
Sediment—Total	34±2	45	44	30	86
Clams	28±5	79	43	24	287
Sediment—HCl	0.35±0.02	0.52	0.36	0.20	1.6
Clams	2.4±0.5	24	5.4	1.8	106

Appendix 8. Complete list of benthic species found at Palo Alto in the year 2013.

[Three samples are taken at each sampling event. The mean and standard deviation of the three samples are shown.]

TAXON	2/6/2013		3/5/2013		4/12/2013		5/14/2013		6/10/2013		7/25/2013		8/22/2013		9/19/2013		11/13/2013	
	Mean	std dev	Mean	std dev	Mean	std dev	Mean	std dev	Mean	std dev	Mean	std dev	Mean	std dev	Mean	std dev	Mean	std dev
PHYLUM ANNELIDA																		
Class Oligochaeta																		
Oligochaeta unid. spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Class Polychaeta																		
Cirratulidae unid. spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.6	0.0	0.0	0.0	0.0
Eteone unid. spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Euchone unid. spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Glycinde armigera	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Glycinde unid. sp. SF1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Harmothoe imbricata	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

TAXON	2/6/2013		3/5/2013		4/12/2013		5/14/2013		6/10/2013		7/25/2013		8/22/2013		9/19/2013		11/13/2013	
	Mean	std dev	Mean	std dev	Mean	std dev	Mean	std dev	Mean	std dev	Mean	std dev	Mean	std dev	Mean	std dev	Mean	std dev
Maldanidae unid. spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Marphysa sanguinea	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Neanthes succinea	0.7	1.2	5.7	2.1	2.7	1.5	2.7	1.2	0.3	0.6	0.3	0.6	1.0	0.0	0.3	0.6	0.0	0.0
Polychaeta unid. spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Polydora cornuta	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Polydora unid. spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Pseudopolydora kempi	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Sabaco elongatus	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Sphaerosyllis californiensis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Sphaerosyllis erinaceus	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Sphaerosyllis unid. sp. A	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Streblospio benedicti	3.7	3.1	7.0	10.4	0.3	0.6	2.3	2.1	0.0	0.0	0.0	0.0	0.3	0.6	0.0	0.0	0.0	0.0
Spionidae unid. spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
PHYLUM ARTHROPODA																		
Class Arachnida																		
Acari	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Class Copepoda																		
Calanoida unid. spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Harpacticoida unid. spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Class Insecta																		

TAXON	2/6/2013		3/5/2013		4/12/2013		5/14/2013		6/10/2013		7/25/2013		8/22/2013		9/19/2013		11/13/2013	
	Mean	std dev	Mean	std dev	Mean	std dev	Mean	std dev	Mean	std dev	Mean	std dev	Mean	std dev	Mean	std dev	Mean	std dev
Chironomidae unid. spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Class Malacostraca																		
Americhelidium unid. spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Ampelisca abdita	28.3	4.5	5.3	1.5	4.3	1.5	0.0	0.0	2.0	1.7	2.3	2.1	7.3	3.1	11.0	4.4	46.0	1.7
Ampithoe unid. spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Callianassidae unid. spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Caprella californica	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Corophium alienense	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Corophium heteroceratum	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Corophium spinicorne	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Corophium unid. spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Corophiidae unid. spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Crangon nigricauda	0.0	0.0	0.0	0.0	0.3	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cumella vulgaris	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Eochelidium cf. miraculum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.6
Eogammarus confervicolus	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Gnorisphaeroma oregonensis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Grandidierella japonica	6.0	5.2	0.7	1.2	0.0	0.0	1.7	1.5	0.3	0.6	1.3	2.3	8.3	3.1	11.0	7.5	4.3	3.1
Hemigrapsus oregonensis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Melita nitida	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

TAXON	2/6/2013		3/5/2013		4/12/2013		5/14/2013		6/10/2013		7/25/2013		8/22/2013		9/19/2013		11/13/2013	
	Mean	std dev	Mean	std dev	Mean	std dev	Mean	std dev	Mean	std dev	Mean	std dev	Mean	std dev	Mean	std dev	Mean	std dev
Melita unid. spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Monocorophium acherusicum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.3	1.5	0.0	0.0	0.7	0.6
Monocorophium insidiosum	2.0	1.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Monocorophium unid. spp.	0.7	1.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.3	1.2	1.0	1.7	2.0	2.0
Mysidacea unid. spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Nippoleucon hinumensis	55.3	13.1	29.7	5.7	131.3	26.3	74.7	18.2	388.7	149.5	84.0	27.8	48.0	25.2	43.0	12.5	28.0	4.6
Sinelobus unid. spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Sphaeromatidae unid. spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Synidotea laevidorsalis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Synidotea unid. spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.6
Class Ostacoda																		
Eusarsiella zostericola	5.7	1.2	7.0	2.6	11.0	3.5	3.7	1.5	2.7	0.6	2.3	1.2	5.7	4.0	7.3	2.5	10.0	2.0
Cyprideis unid. spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Class Thecostraca																		
Amphibalanus improvisus	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Balanomorpha unid. spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.6	0.0	0.0
Cirripedia unid. spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
PHYLUM CNIDARIA																		
Class Anthozoa																		
Actiniaria - attached	0.7	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.3	1.5	0.3	0.6

TAXON	2/6/2013		3/5/2013		4/12/2013		5/14/2013		6/10/2013		7/25/2013		8/22/2013		9/19/2013		11/13/2013	
	Mean	std dev	Mean	std dev	Mean	std dev	Mean	std dev	Mean	std dev	Mean	std dev	Mean	std dev	Mean	std dev	Mean	std dev
Actiniaria - burrowing	0.0	0.0	1.0	1.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.6	0.0	0.0
Actiniaria unid. spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
PHYLUM MOLLUSCA																		
Class Bivalvia																		
Bivalvia unid. spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Gemma gemma	184.3	83.0	163.7	41.0	676.3	11.0	421.0	311.5	250.7	64.1	299.3	65.5	289.7	116.4	276.0	33.6	210.0	24.4
Macoma petalum	3.0	0.0	1.7	0.6	4.3	5.1	1.0	1.0	0.7	1.2	2.0	1.0	2.0	1.0	1.7	1.5	2.0	1.0
Macoma unid. spp.	0.0	0.0	0.0	0.0	1.7	1.5	0.7	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Musculista senhousia	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Mya arenaria	0.3	0.6	0.0	0.0	2.0	3.5	1.7	2.9	0.0	0.0	0.0	0.0	0.7	0.6	0.3	0.6	0.0	0.0
Potamocorbula amurensis	3.0	2.0	0.3	0.6	0.7	1.2	2.0	1.0	2.0	1.0	0.7	0.6	0.3	0.6	2.0	1.0	2.0	1.0
Rochefortia grippi	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Rochefortia unid. spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Tellinidae unid. spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Class Gastropoda																		
Gastropoda unid. sp. B	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Ilyanassa obsoleta	0.0	0.0	0.0	0.0	0.3	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.6
Odetta bisuturalis	5.0	3.6	2.0	1.0	0.3	0.6	2.0	2.0	0.3	0.6	0.3	0.6	3.3	1.5	1.0	1.7	1.0	0.0
Philine unid. spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Pyramidellidae unid. sp. A	0.0	0.0	0.0	0.0	0.7	1.2	0.0	0.0	0.7	1.2	0.0	0.0	3.0	1.0	1.0	0.0	0.7	0.6

TAXON	2/6/2013		3/5/2013		4/12/2013		5/14/2013		6/10/2013		7/25/2013		8/22/2013		9/19/2013		11/13/2013	
	Mean	std dev	Mean	std dev	Mean	std dev	Mean	std dev	Mean	std dev	Mean	std dev	Mean	std dev	Mean	std dev	Mean	std dev
Urosalpinx cinerea	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
PHYLUM NEMATODA																		
Nematoda unid. spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.3	1.2	0.0	0.0	0.7	1.2
PHYLUM PLATYHELMINTHES																		
Class Turbellaria																		
Turbellaria unid. sp. A	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Turbellaria unid. spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

Appendix 9. Benthic species name changes as of 2013.

Current Species Name:	Formerly Documented As:
PHYLUM ANNELIDA	
Class Oligochaeta	
Naididae unid. spp.	Naididae
Oligochaeta unid. spp.	Oligochaeta, Unid. Oligochaeta family
Tubificidae unid. spp.	Tubificidae
Class Polychaeta	
Capitella capitata complex	Capitella "capitata"
Cirratulidae unid. spp.	Tharyx spp. ?, Cirratulidae
Eteone unid. spp.	Eteone spp., Eteone ?californica
Euchone unid. spp.	Euchone spp.
Glycera unid. spp.	Glycera spp.
Glycinde picta	Glycinde polygnatha
Glycinde unid. sp. SF1	Glycinde sp. SF1
Glycinde unid. spp.	Glycinde spp.
Maldanidae unid. spp.	Unid. Malanidae
Polychaeta unid. spp.	Unid. Polychaeta, Polychaeta
Polydora cornuta	Polydora lighti, Polydora ligni
Polydora unid. spp.	Polydora spp.
Sphaerosyllis unid. sp. A	Sphaerosyllis spp.
Spionidae unid. spp.	Unid. Spionidae, Spionidae Unidentified
PHYLUM ARTHROPODA	
Class Copepoda	
Calanoida unid. spp.	Calinoida
Harpacticoida unid. spp.	Harpacticoida
Class Insecta	
Chironomidae unid. spp.	Chironomidae
Class Malacostraca	
Americhelidium unid. spp.	Americhelidium spp., Synchelidium spp.
Ampithoe unid. spp.	Ampithoe spp.
Callianassidae unid. spp.	Callianassidae, Callianassidae unidentified
Corophium unid. spp.	Corophiidae - unidentified
Corophiidae unid. spp.	Corophium spp.
Eogammarus confervicolus	Anisogammarus confervicolus
Gnorisphaeroma oregonensis	Gnorisphaeroma oregonensis, Gnorimosphaeroma oregonensis
Melita unid. spp.	Melita spp.
Monocorophium acherusicum	Corophium acherusicum
Monocorophium insidiosum	Corophium insidiosum
Monocorophium unid. spp.	Corophium ?insidiosum, Corophium spp. (female & juvenile), Corophium spp. (male), Monocorophium spp.
Mysidacea unid. spp.	Mysidacea
Nippoleucon hinumensis	Hemileucon hinumensis

Current Species Name:	Formerly Documented As:
Sinelobus unid. spp.	Sinelobus stanfordi, Sinolobus spp., Sinolobus stanfordi, Tanais spp.
Sphaeromatidae unid. spp.	Sphaeromatidae (juv.), Sphaeromatidae unid., Dynamella spp.
Synidotea unid. spp.	Synidotea spp.
Class Ostacoda	
Eusarsiella zostericola	Sarsiella zostericola
Cyprideis unid. spp.	Cyprideis spp.
Class Thecostraca	
Amphibalanus improvisus	Balanus improvisus
Balanomorpha unid. spp.	Balanus ?aquila, Balanus spp., Unid. Balanomorpha, Balanomorpha - unidentified
Cirripedia unid. spp.	Cirripedia
PHYLUM CNIDARIA	
Class Anthozoa	
Actiniaria unid. spp.	Anthozoa, Unid. Actiniaria
PHYLUM MOLLUSCA	
Class Bivalvia	
Bivalvia unid. spp.	Unid. Bivalvia
Macoma petalum	Macoma balthica
Macoma unid. spp.	Macoma spp.
Potamocorbula amurensis	Corbula amurensis
Rochefortia unid. spp.	Rochefortia spp.
Tellinidae unid. spp.	Tellinidae
Class Gastropoda	
Gastropoda unid. sp. B	Unidentified Gastropoda B
Ilyanassa obsoleta	Nassarius obsoletus
Odetta bisuturalis	Boonea bisuturalis, Odostomia fetella
Philine unid. spp.	Philine spp.
Pyramidellidae unid. sp. A	Odostomia spp., Pyramidellidae, Unidentified Gastropod A
PHYLUM NEMATODA	
Nematoda unid. spp.	Nematoda
PHYLUM PLATYHELMINTHES	
Class Turbellaria	
Turbellaria unid. sp. A	Planariidae A
Turbellaria unid. spp.	Turbellaria

Turbellaria unid. sp. A	Planariidae A
Turbellaria unid. spp.	Turbellaria