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Marine Chemistry 64 (1999) 1–6

MARINE CHEMISTRY

The impact of human activities on sediments of San Francisco Bay, California: an overview

Alexander van Geen [a,*], Samuel N. Luoma [b,1]

[a] Lamont-Doherty Earth Observatory of Columbia University, Palisades, NY 10964, USA
[b] US Geological Survey, MS465, 345 Middlefield Road, Menlo Park, CA 94025, USA

Received 31 July 1997; revised 14 October 1998; accepted 14 October 1998

Abstract

This note introduces a set of eight papers devoted to a detailed study of two sediment cores from San Francisco Bay with an overview of the region and a chronology of human activities. Data used in this study to constrain the range of sediment ages at different depths include ^{234}Th, ^{210}Pb, ^{137}Cs, $^{239,240}Pu$, and ^{10}Be concentrations in the sediment and the ^{14}C age of shell fragments. In order of first detectable appearance in the record, the indicators of contamination that were analyzed include PAHs > Hg > Ag, Cu, Pb, Zn > DDT, PCB > foraminiferal Cd/Ca. This study also documents a large memory effect for estuarine contamination caused by sediment mixing and resuspension. Once an estuary such as San Francisco Bay has been contaminated, decades must pass before contaminant levels in surface sediment will return to background levels, even if external contaminant inputs have been entirely eliminated. © 1999 Elsevier Science B.V. All rights reserved.

Keywords: San Francisco Bay; sediment; estuarine contamination

1. Background

This collection of papers is the outcome of an interdisciplinary study that utilized sediment core material to document the influence of human activities on San Francisco Bay. The project, a cooperative effort between the US Geological Survey and researchers at several universities, was partially funded by the NOAA/ORCA Status and Trends Program. Contributions include studies of sediment sources, deposition, and mixing (Fuller et al., 1999; van Geen et al., 1999), a reconstruction of metal contamination patterns (Hornberger et al., 1999), a study of Pb sources based on Pb isotopes (Ritson et al., 1999), a study of past contamination in the water column for Cd based on foraminifera (van Geen and Luoma, 1999), reconstructions of organic contaminant inputs (Pereira et al., 1999; Venkatesan et al., 1999), as well as a biomarker study (Hostettler et al., 1999). In this introductory paper, we provide an overview of the study area and relate results from the study to a chronology of human activities in northern California.

San Francisco Bay is one of the few large estuaries on the western coast of North America. With a watershed of 163,000 km^2, it drains 40% of California's surface area. It is a relatively shallow estuary (mean depth of 6 m) and has a large tidal range of up

* Corresponding author. Tel.: +1-914-365-8644; Fax: +1-914-365-8154; E-mail: avangeen@ldeo.columbia.edu
[1] E-mail: snluoma@usgs.gov.

PII: S0304-4203(98)00080-2

to 1.7 m at the mouth (Conomos, 1979). Broad expanses of bay floor are incised by narrow channels (Fig. 1). The annual riverine sediment load to northern San Francisco corresponding to 1960 flow conditions has been estimated at 4.4×10^9 kg/year (Krone, 1979). This corresponds to a potential volume of sediment accumulation of 8.3×10^6 m^3/year, assuming a dry weight of sediment deposited of 530 kg/m^3 (Krone, 1979). For a dry sediment density of 2.2 g/cm^3, this corresponds to a mean porosity of 76%. The potential linear sedimentation rate corresponding to this input averaged over the 890 km^2 area of the estuary is ~ 0.9 cm/year. In principle, this is more than sufficient to keep up with a relative sea level rise of 0.2 cm/year documented for the past century by a tide gauge record near the mouth of San Francisco Bay (Krone, 1979) and for the past 8000 years by ^{14}C-dated marsh deposits (Atwater et al., 1979).

Human activities have drastically changed river inflows, sediment loads, and the geomorphology of the bay, however (Nichols et al., 1986). In a perturbation that lasted from at least 1852 to 1914, sediment transport to San Francisco Bay was increased by nearly an order of magnitude (from 1.5 to 14×16^6 m^3/year) due to debris created by hydraulic mining for gold (Gilbert, 1917). Large deposits of these sediments still reside in San Francisco Bay. By comparing historical bathymetric surveys, Jaffe et al. (1998) estimate that 350×10^6 m^3 of sediments were deposited in San Pablo Bay between 1856 and 1951; but over two-thirds of this input was hydraulic mining debris that accumulated in only 31 years between 1856 and 1887. At the same time, the area of tidal marsh, nearly double the area of the bay itself in 1850, has been reduced to less than 4–8% of its original area by filling and the construction of levies (Atwater et al., 1979). At least in part because of accelerated erosion and the reduction in size of the estuary, navigation channels have to be dredged periodically (e.g., 5.4×10^9 kg/year between 1975 and 1985, Gunther et al., 1987) at a rate which is comparable to the riverine sediment input. Because the dredge material has been disposed of mostly near the mouth of San Francisco Bay, a substantial proportion has subsequently been redistributed within the estuary (Conomos, 1979).

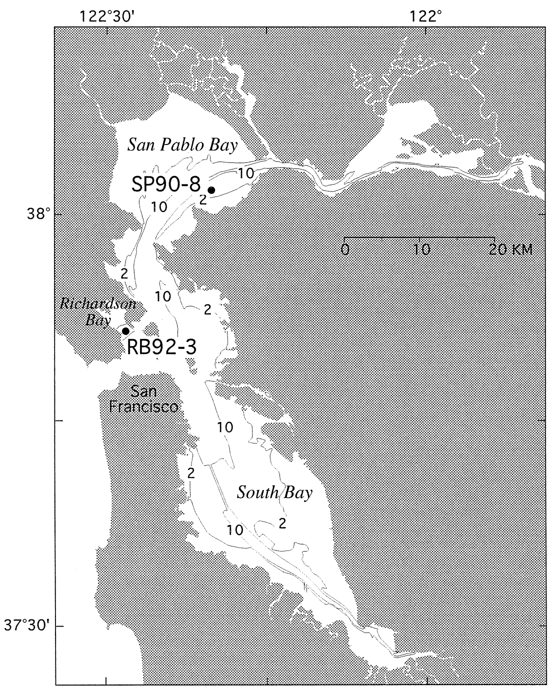

Fig. 1. Location of the two main sediment cores that were studied. Bathymetry in meters.

One of the largest water management projects in the world was established in the watershed beginning in the 1930s, with most major components of the federal water management system completed by 1951. More than 60% of river flow is now diverted for agricultural and urban consumption before it reaches the estuary (Nichols et al., 1986). Fourteen major dams have been constructed on the incoming rivers. The dams have reduced the frequency of peak river flows and have decreased sediment input to the estuary (Peterson et al., 1993). This may explain why between 1951 and 1983, sediment erosion has replaced sediment deposition in much of San Francisco Bay (Jaffe et al., 1998).

San Francisco Bay is a highly urbanized and industrialized estuary today; more than 7.5 million people presently live within the greater Bay Area (Perkins et al., 1991). The bay is the ultimate receptacle for much of their waste. Watershed activities that contribute(d) contamination to the bay include mining for Hg, Au, and Cu, and extensive agricultural use of pesticides. Historic hazardous waste sites, military bases, and harbors are potential sources of localized contaminant input, in addition to as many as 50 small creeks that carry untreated urban runoff into the bay during periods of heavy rainfall. A large number of municipal and industrial dischargers release their wastes into the estuary. Fifty publicly owned municipal waste treatment plants discharge 3.2 billion l/day of wastewater into the bay and incoming rivers (Monroe and Kelly, 1992). Six refineries discharge > 100 million l/day of process water, a dozen other large industrial dischargers together release another 100 million l/day of wastewater, and 50 smaller industrial facilities (power plants, oil terminals, chemical, metal and paper facilities) release 225 million l of waste water. Between 4500 and 36,000 metric tons of toxic pollutants enter the estuary with these discharges, as best can be estimated (Monroe and Kelly, 1992). These loadings are similar to other large industrialized estuarine ecosystems for which comparable data exist (e.g., Nixon, 1995).

Although waste water discharges have continually increased through time, inputs of at least some contaminants have declined. After 1970 substantial investments were made in advanced waste treatment and chemicals such as chlorinated hydrocarbon pesticides (dichlorodiphenyltrichloroethane, DDT) and polychlorinated biphenyls (PCBs) were banned from use. A number of heavy industries have closed around the bay since 1970 (including the large Selby lead smelter, automobile manufacturers, and a large photo processing plant) and mining activities have ceased. Analyses of effluents by dischargers suggest loadings of pollutants such as metals have declined. The papers that follow present some of the first data on the response of contaminant levels in the sediment to these changes in inputs.

2. Some key observations

In order to document the impact of human activities on sediment in San Francisco Bay, we selected cores from two depositional areas for detailed dating and contaminant analysis. Selected studies were conducted on several additional cores from North Bay and some surface sediments. One area studied in detail is Richardson Bay (RB, Fig. 1), a small protected embayment near the mouth of the estuary, where sediments have been accumulating through much of the Holocene (van Geen et al., 1992). The second area is San Pablo Bay, an additional ~15 km landward into the estuary (SP, Fig. 1). Sedimentation patterns have been rather different at the two locations (Fuller et al., 1999). Radionuclide distributions, variations in sediment texture, and historical bathymetry indicate that rapid deposition alternated with periods of erosion at the SP core. Data from the RB core best supports a synthesized chronology of disturbance (Fig. 2), both because the core is centrally located and because sediment accumulation has been continuous at this site. No records have yet been obtained from South San Francisco Bay because of the high degree of sediment mixing and resuspension (Fuller, 1982). While every paper in this series describes heretofore unrecognized features of the history of sediment disturbances in San Francisco Bay, viewing the body of work as a whole also provides some insights.

Radiometric dating of the cores by various techniques shows that sediment accumulation increased by an order of magnitude relative to the past several centuries, at least in Richardson Bay (van Geen et al., 1999; Fuller et al., 1999). Because of physical and/or biological mixing, each horizon in the core contains different proportions of sediments deposited over several decades. Despite the increase in sedimentation, physical and/or biological mixing processes had a substantial effect on the vertical distribution of contaminants (Fuller et al., 1999). This is illustrated in Fig. 2a showing the depth distribution of sediment deposited in a sequence of 20-year intervals calculated from the mixing/accumulation model of Fuller et al. (1999) for the RB core. The model results show that no that more than half the sediment at any depth was deposited within a single 20-year interval during the past century. A notable consequence of the intense mixing regime is the predicted delay in recovery from contamination that extends over decades even after a drastic reduction in contaminant input (Fuller et al., 1999). Despite intense mixing, a chronology of multi-contaminant input was retained in the RB core.

Downcore profiles show that the rise in concentration of polyaromatic hydrocarbons (PAHs) in the sediment preceded all other disturbances (Pereira et al., this volume, Fig. 2b). It is possible that land was cleared by fire following the arrival of the Spanish

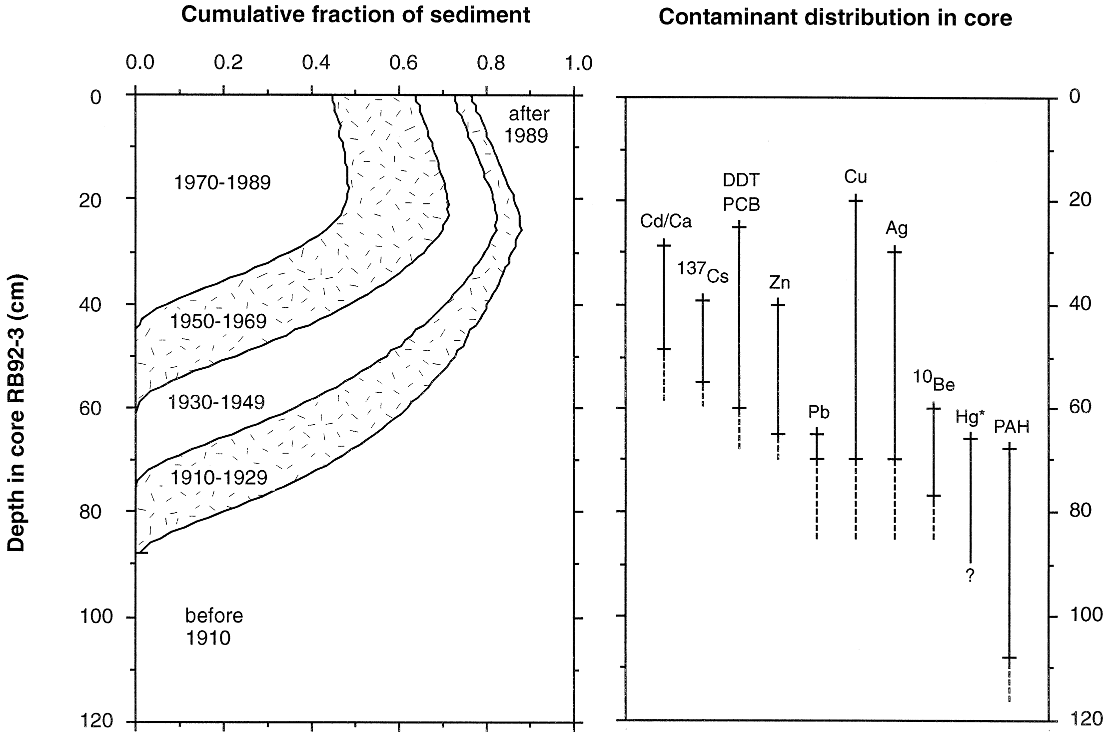

Fig. 2. Chronology of disturbance and contaminant input in San Francisco Bay based on record from Richardson Bay. The panel to the left shows the depth distribution of sediment deposited in sequential 20-year intervals calculated from the model of Fuller et al. (1999). Dashed lines in the right panel connect the shallowest interval where a contaminant was not detected to the nearest interval where it was detected. The solid line connects the depth interval where a contaminant was first detected to the depth of maximum concentrations. The data used for this figure are from Fuller et al. (1999), van Geen et al. (1999), Hornberger et al. (1999), Pereira et al. (1999) and Venkatesan et al. (1999). *Although the onset of Hg contamination does not precede increases in other metal concentrations in RB92-3, it clearly does in SP90-8.

colonizers in 1769 or that other early human activities left a combustion-related PAH signature. Mercury was the next contaminant to appear in the sediments, probably in association with the input of hydraulic mining debris (Hornberger et al., 1999). Hg was used in California, as in developing nations today, to amalgamate and extract gold. Sediments deposited during the period of 1940–1980 contain a variety of contaminants: metals of mining and industrial origin (Hornberger et al., 1999; Ritson et al., 1999); DDT from upstream in the watershed and PCBs with more widely distributed sources (Venkatesan et al., 1999); complex hydrocarbons, including PAHs, mostly originating from combusted petroleum products (Hostettler et al., 1999; Pereira et al., 1999). The RB chronology suggests that the interval between 1950 and 1970 was the period of most severe overall contamination of bay sediments. After 1970, the mixture of contaminants reaching the sediments began to change and, in some cases, the level of input declined. For example, the sharp decline in DDT concentrations over the past two decades follows the deconvoluted mixing profile predicted for a greatly diminished input starting at the time of the 1970 ban (Fuller et al., 1999; Venkatesan et al., 1999). A slower rate of decline is observed for PAHs and most metal contaminants based on ratios of concentrations at the surface of the RB core to concentrations in sediments deposited around 1960 (Table 1). Such sediment profiles show that, in addition to vertical mixing of sediments, the rate of decline of certain contaminant concentrations has been limited by continuing inputs, dispersal of contamination from large historic sources (e.g., a Pb smelter, Ritson et al., 1999) and, perhaps, enhanced trapping of some contaminants due to diversion of

Table 1

	Ratio of inventories San Pablo/Richardson Bay in sediment containing ^{137}Cs	Ratio of concentrations near surface and at ^{137}Cs maximum in Richardson Bay
Total organic carbon[a]	2.4	1.0
Silt and clay[b]	3.9	1.0
^{137}Cs[b]	9.4	0.6
$^{239,240}Pu$[b]	3.2	0.6
Excess ^{210}Pb[b]	2.0	1.0
Excess Ag[c]	2.2	0.8
Excess Cu[c]	2.9	0.9
Excess Hg[c]	2.6	0.8
Excess Pb[c]	2.2	0.8
Excess Zn[c]	2.7	0.9
PAH[a]	0.4	0.7
DDT[d]	6.2	0.4
PCB[d]	3.5	0.6

Data from [a]Pereira et al. (1999).
[b]Fuller et al. (1999).
[c]Hornberger et al. (1999).
[d]Venkatesan et al. (1999).

fresh water inflows for agriculture (e.g., Cd, van Geen et al., 1999).

Inventories of many contaminants (PCBs, DDT, Hg and other metals) are factors of 2–6 greater in SP than in RB sediments (Table 1). In some cases such as DDT, the inventory difference is so large that it can be attributed to a source located upstream in the watershed. For other pollutants (e.g., PCBs) however, the contrast is probably a reflection of the different depositional environments, as indicated by finer grain size and higher organic content of the bulk sediment in SP relative to RB. In the case of metal contaminants, sediments were sieved before analysis to minimize such biases. Some metal sources were unquestionably highly localized (e.g., Pb, Ritson et al., this volume), some were probably more widely distributed in the watershed (e.g., Hg), while others were associated with industries in the upper estuary (Hornberger et al., 1999). It is worth noting that the PAH inventory is surprisingly high in RB sediments relative to sediments of comparable age in SP (Pereira et al., 1999).

3. Conclusion

Considered together, these multi-faceted studies show a remarkable correspondence between the intensity of specific human activities and the impact of these activities on sediment contamination, despite the strong physical and biological mixing that characterizes the system. A significant observation applicable to estuaries in general is the large memory effect caused by sediment mixing and redistribution. A large estuary widely contaminated by human activities can take decades to recover even if sources of contamination have been eliminated. At the same time the effectiveness of regulation is illustrated by the decline in concentrations of some contaminants in the face of continued economic growth. The study illustrates the importance of early implementation of waste treatment technologies all over the world as estuaries are subjected to growing pressure from urban and industrial expansion.

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