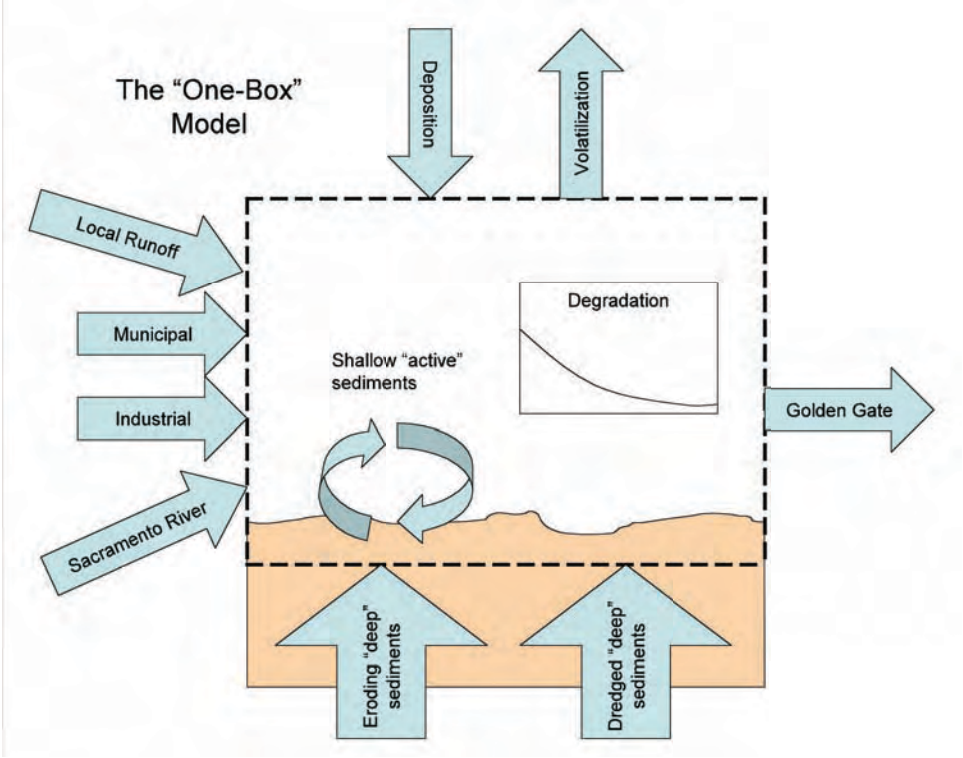

Figure 3-1. One-box model (figure courtesy of Dan Cloak)

3.2 Sources and Pathways

DDTs, chlordanes, and dieldrin enter San Francisco Bay from several sources (Davis *et al.*, 2001) (Figure 3-2):

- Agricultural and urban **watersheds** with histories of pesticide application.
- **Wastewater effluent**.
- **Atmospheric deposition**.
- Erosion of **historic deposits** within the Bay.
- Dredging and disposal of **dredged material**.

Some sources and loadings of legacy pesticides may be controllable, while others are not.

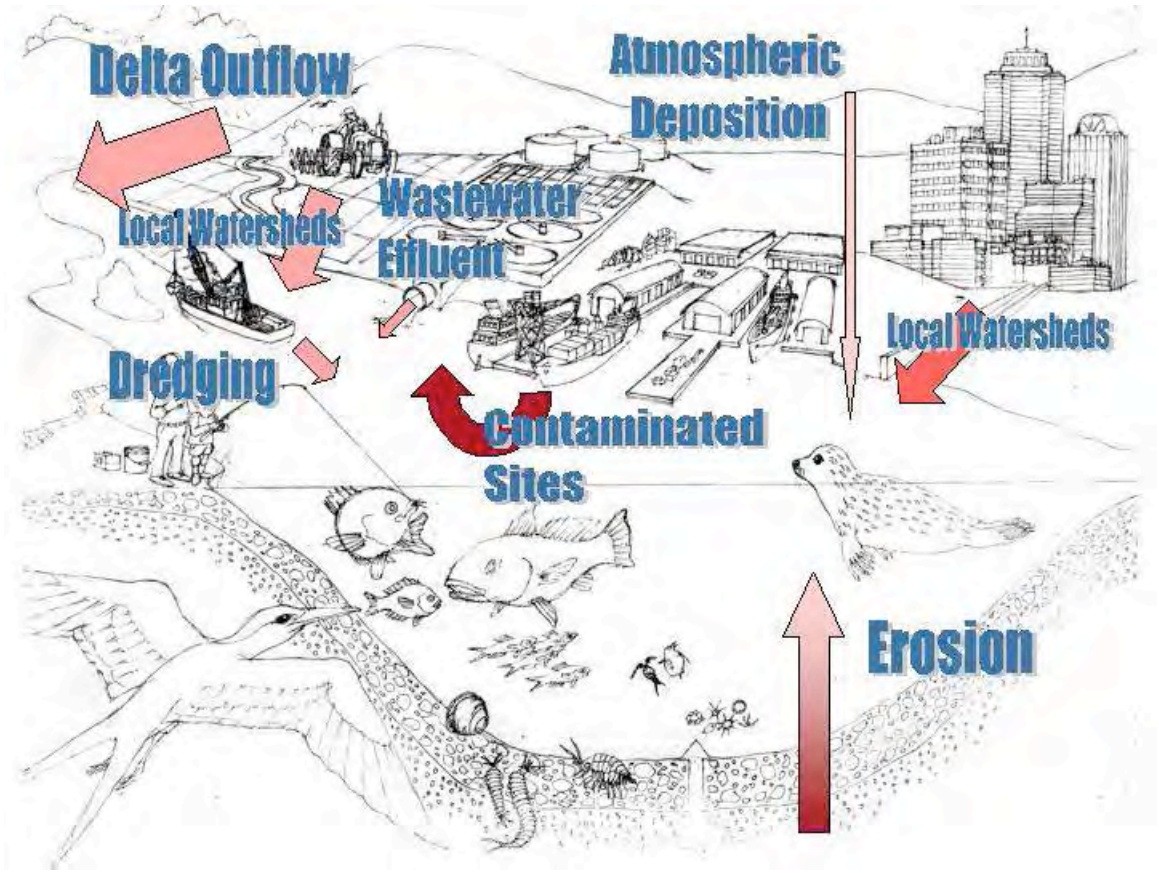

Figure 3-2. Sources of legacy pesticides to San Francisco Bay

3.2.1 Watersheds

Because of the widespread historic use of the legacy pesticides throughout the country and the enormity of the area that feeds the Bay, almost 60,000 square miles, runoff from the watersheds, particularly the Central Valley, is a major pathway for legacy pesticides to reach the Bay. Historic use left persistent residues of pesticides in soils and in the sediments of floodplains, banks, and beds of channels throughout California and the Bay Area (Mischke *et al*., 1985; Law and Goerlitz, 1974; KLI, 2002; Salop *et al.*, 2002). Pesticide residues primarily associated with particles entrained in surface runoff and resuspended in channels are transported to the Bay during large storm events (Bergamaschi *et al*., 2001; Leatherbarrow *et al*., 2002; McKee *et al*., 2004; Leatherbarrow *et al*., 2004).

Central Valley

The intense agricultural activity in the Central Valley left pesticide residues in the soils, stream sediments, water and biota (Mischke *et al*., 1985; Gilliom and Clifton, 1990; Pereira *et al*., 1996; Kratzer, 1999; Brown, 1997). Urban use of pesticides was also common and occurred more recently than agricultural use, as

agricultural restrictions to the pesticides preceded their overall bans. In particular, total urban use of chlordanes may have exceeded agricultural use (Nowell *et al.*, 1999). Consequently, while the contemporary occurrence and distribution of DDTs and dieldrin in the Central Valley are typically associated with historic agricultural applications, the presence of chlordanes is more likely related to historic use for termite and ant control in residential and commercial applications.

In 1985, concentrations of DDTs and dieldrin in sediments from the San Joaquin River watershed were among the highest in the nation (Gilliom and Clifton, 1990). The stations with the highest concentrations were located in westside tributaries that primarily carried agricultural surface runoff and overflow from the Delta-Mendota Canal. Pereira *et al.* (1996) observed a similar pattern, with high concentrations of DDTs and dieldrin in water, suspended sediments, sediments, and biota of a westside tributary, Orestimba Creek, which is bordered by apple orchards, field crops, and row crops. In contrast, maximum chlordane concentrations in suspended sediments were measured in samples from Dry Creek, which receives urban runoff from Modesto.

Local Watersheds

Inputs of legacy pesticides from the other watersheds that feed San Francisco Bay besides the Central Valley also reflect historic and current land use. Much of the area directly adjacent to the Bay was used for agriculture before the post-World War II period of rapid population growth and urbanization. Two studies conducted in the 1970s and 1980s found that DDT residues were ubiquitous and persistent in agricultural soils and tributary sediments throughout the Bay Area (Law and Goerlitz, 1974; Mischke *et al.*, 1985). More recent monitoring conducted in 2001 found that concentrations of DDTs in sediments from urbanized regions of the watersheds were greater than those from non-urbanized, non-agricultural open space, with concentrations ranging as high as 4,010 µg/kg (KLI, 2002).

The urban influence on chlordane and dieldrin distribution in the local watersheds has also been evident. Law and Goerlitz (1974) detected chlordanes in 92% of sediment samples from tributaries to the Bay, with no spatial differences between the northern and southern regions. After that study, pesticide use declined—statistics from the California Department of Pesticide Regulation (CDPR) indicate a particularly rapid decline in chlordane use from 1989-1990 (Table 3-1).

Table 3-1. Chlordane use in the Bay Area, 1989-1990 (data from CDPR, 2003)

	Amount applied (kg)	Counties applying chlordane
1989	240	Alameda, Contra Costa, San Mateo, Santa Clara
1990	78	Santa Clara

Recent measurements (KLI, 2002) found concentrations of chlordanes in sediments as high as 11,300 µg/kg in urban, industrial locations, with much lower

concentrations in sediments from open space. In the same study, dieldrin concentrations were as high as 70 µg/kg, with no dieldrin detected in samples from the non-urban open space locations (KLI, 2002).

Legacy pesticide contamination is widely distributed in the Bay watersheds, and complete elimination of watershed loadings is not feasible. Contaminated soils and sediments from the watersheds will continue to wash into the Bay and constitute a continuing input that will delay recovery. However, as the pesticides degrade, loads from the watersheds are expected to decline.

3.2.2 Wastewater Effluent

Municipal wastewater treatment plants receive inputs of legacy pesticides from various sources, including water supply, stormwater runoff, human and food waste, landfill leachate, and hazardous waste disposal (EIP, 1997). In a study of sampling and analysis methodologies, there was great variation in concentrations of pesticides in samples collected in 1999 and 2000 from four Bay Area municipal wastewater treatment plants (Table 3-2) (Yee *et al.*, 2001). (The purpose of the study was not monitoring, so calculation of means for comparison with water quality standards is not possible.)

Table 3-2. Range of concentrations of legacy pesticides in municipal wastewater.

Pesticide	Concentration range (pg/liter)
DDTs	4 to 1,900
Chlordanes	Less than ~1-3 to 1,800
Dieldrin	Less than ~1-3 to 450 pg/l

Legacy pesticide contamination of the human food supply due to the global distribution of these chemicals and their accumulation in the human food web, especially meats and dairy products, will cause a continued level of loading from municipal wastewater that would be very difficult to control. Small quantities of pesticides may also occur in industrial discharges.

3.2.3 Atmospheric Deposition

Much of the input of legacy pesticides from the atmosphere to San Francisco Bay is an indirect result of deposition onto the land surface in the watershed, and those inputs are considered to be part of the runoff inputs from the watershed. There is some local re-deposition of pesticides that are volatilized, evaporated, or eroded from surface soils, water, and sediment. There is also some direct atmospheric deposition resulting from long-range transport in air masses. This input is not controllable and will contribute to continued loading of legacy pesticides to the Bay via direct and indirect atmospheric deposition.

3.2.4 Erosion of Sediment Deposits

While deposition of sediments can be a sink for legacy pesticides in some areas of the Bay, as of 1990, Suisun, San Pablo, and South bays were areas of net erosion (Capiella *et al*., 1999; Jaffe *et al.*, 1998; Foxgrover *et al.*, 2003). Considering that Suisun and San Pablo bays are in close proximity to inputs from the Central Valley, continued erosion will potentially uncover more contaminated layers of historically deposited pesticides.

Remobilization of sediments from highly contaminated areas, or "hot spots," may contribute potentially significant inputs to the Bay. One known location of former pesticide use is the United Heckathorn site on Richmond Harbor (Pereira *et al.*, 1994; Anderson *et al.*, 2000). The United Heckathorn facility received technical grade pesticides, primarily DDT, from chemical manufacturers and prepared and packaged them for final sale. Despite on-land soil and subtidal sediment cleanup, one part of the site, the Lauritzen Channel, remains contaminated with DDTs and dieldrin.

3.2.5 Dredging and Dredged Material Disposal

Sediment is dredged from Bay channels and ports and disposed of in and outside of the Bay. In the recent mercury TMDL report, Johnson and Looker (2003) estimated that there is greater out-of-Bay disposal of dredged sediment than in-Bay disposal, resulting in an overall net loss of sediment from the Bay. However, on a more localized regional or Bay-segment level, dredged material disposal may contribute to net addition of pesticide mass. For example, dredged material disposed of at Alcatraz Island may increase the mass of sediment and associated pesticides in the Central Bay.

3.3 Loads

Estimated loads of legacy pesticides to the water column and active sediment layer of San Francisco Bay are approximately 60 kg/year DDTs, 30 kg/year chlordanes, and 10 kg/year dieldrin (Table 3-3). The estimates have some large uncertainties. The amount of pesticides available for transport from the watershed is the largest factor. Limited information of historic pesticide use and loading, as well as considerable variability in the hydrologic and geomorphic processes in the watersheds, preclude making a definitive estimate of the pesticide mass being stored within the watershed.

Table 3-3. Estimated loads (best estimate and range) of legacy pesticides to San Francisco Bay (kg/year).

Pathway	DDTs	Chlordanes	Dieldrin
Central Valley	15 (5 – 40)	2 (0.7 – 5)	5 (2 – 13)
Local watersheds	40 (9 – 190)	30 (7 – 160)	3 (0.7 – 15)
Municipal wastewater	0.2 (0.02 – 2)	0.1 (0.003 – 2)	0.06 (0.008 – 0.4)
Industrial wastewater	<0.2	<0.1	<0.06
Atmospheric deposition	1 (0.02 – 2)	0.9	1 (0.2 – 2)
Erosion of sediment deposits	9 (0.2 – 18)	2 (0 – 4)	0.2 (0 – 0.6)
Dredged material	-2 (-3 – -0.03)	-0.3 (-0.6 – 0)	-0.03 (-0.1 – 0)
Total Best Estimate	**60 (10 – 250)**	**30 (10 – 170)**	**10 (3 – 30)**

This section of the report describes the calculations, assumptions, and uncertainties associated with the values presented in Table 3-3.

3.3.1 Loads from Watersheds

Central Valley

Pesticide loads from the Sacramento and San Joaquin rivers that drain the Central Valley were estimated from preliminary contaminant data collected as part of a RMP special study conducted in 2002 and 2003 at Mallard Island, a site located approximately five kilometers downstream from the confluence of the two rivers (Leatherbarrow *et al.*, 2004). Since 1994, the United States Geological Survey (USGS) has been collecting continuous turbidity data on 15-minute intervals at the Mallard Island site. This continuous turbidity data set, used in conjunction with regressions between suspended sediment and pesticide concentrations, allowed for extrapolation of continuous records of suspended sediment, DDT, and chlordane (methods described in McKee *et al.*, 2002; McKee and Foe, 2002). Best estimate loads in Table 2-4 were derived from the median and ranges of annual loads estimated from 1995-2003 using two methods:

- Regression between turbidity (and suspended sediment concentrations) and pesticides.
- Flow-weighted mean concentrations of pesticides (SFEI, unpublished data).

Variability in Delta outflow and sediment transport led to a range of contaminant load estimates that spanned an order of magnitude. The maximum pesticide load, in 1995, occurred because of above-average outflow from the Delta (52,000 Mm^3) that was approximately six times greater than flow in 2001 (8,600 Mm^3).

Several sources of uncertainty in the calculations were described in detail by McKee *et al.* (2002) and McKee and Foe (2002): averaging of suspended sediment concentration data on a daily time step to estimate daily loads, error in the Delta outflow calculation, cross-sectional variability of suspended sediment concentrations, tidal influence, and regression errors between turbidity, suspended sediment concentrations, and pesticide concentrations. Estimates of error associated with pesticide loading estimates for individual years were ±38% for DDTs, ±39% for chlordanes, and ±44% for dieldrin.

Another source of uncertainty is that data used for estimating loads from the Central Valley were collected during two years of below-average Delta outflow (based on a 30-year average from 1971 to 2000). Calculating loads for years prior to 2002 relies on the assumption that the relationships between turbidity, suspended sediment, and pesticides remained constant over the entire range of Delta outflows. In reality, these relationships may vary at higher flows that carry sediment and freshwater from varying sources in the Central Valley. Monitoring pesticide concentrations downstream of the large rivers during periods of above-average Delta outflow would help characterize the pesticide concentrations and transport processes observed over the full range of variability in sediment transport and freshwater runoff from the Central Valley.

Local Watersheds

Estimating pesticide loads from the combined Bay Area watersheds is inherently difficult, due to limited available data and insufficient techniques for extrapolating from existing data and accounting for different land uses, hydrology, and other watershed characteristics. Bay Area stormwater management agencies used pesticide concentrations in bed sediments from stormwater conveyance systems and the SIMPLE model to derive preliminary estimates of DDT and chlordane loads (KLI, 2002; Salop *et al.*, 2002). Best estimates (and ranges) were 9.2 (0.9-20) kg DDT and 22 (19-102) kg chlordane, with 98% of the total attributed to urban sources. There are considerable uncertainties associated with the estimates derived from the SIMPLE model (Davis *et al.*, 2000; KLI, 2002). For example, the study focused on urban sources of the pesticides—no data exist to facilitate estimating loads from agricultural sources.

Pesticide loads from local watersheds were also estimated using preliminary data collected by McKee *et al.* (2004) in the lower Guadalupe River watershed in 2003 and extrapolated to all watersheds based on the overall sediment and water budgets in the Bay. The Guadalupe River watershed represents an area that was historically agricultural and converted to predominantly urban land uses during the period that the pesticides were used. Similar to Central Valley load estimates, estimated loads for local watersheds were derived using two types of data:

- Linear regression between suspended sediment concentrations and pesticides.
- Flow-weighted mean concentrations of pesticides.

Linear relationships between total pesticide concentrations in water and suspended sediment concentrations in 22 Guadalupe River samples provided an estimate of pesticide concentrations associated with suspended particulate material entering the Bay from a local watershed. Slopes of the regressions resulted in approximate suspended sediment-normalized concentrations of DDTs, chlordanes, and dieldrin, 46, 41, and 3.7 µg/kg, respectively (McKee *et al*., 2004).

The best available estimates of sediment transport to the Bay range from approximately 0.56 to 1.0 million metric tons (McKee *et al.*, 2003). Applying the suspended sediment concentration-normalized pesticide concentrations from Guadalupe River to the range of annual sediment loads from the combined local watershed area resulted in annual pesticide loads from the local watersheds of 26-46 kg DDTs, 23-41 kg chlordanes, and 2.0-3.5 kg dieldrin.

In the Guadalupe River water samples, flow-weighted mean concentrations of total DDTs, total chlordanes, and dieldrin were 48 ng/L, 40 ng/L, and 3.7 ng/L, respectively. Annual freshwater flow from local watersheds ranges from approximately 180 Mm^3 in dry years to 3,930 Mm^3 in wet years (McKee *et al.*, 2003). Using an average annual flow of 920 Mm^3, annual pesticide loads were estimated to be approximately 44 kg DDTs, 37 kg chlordanes, and 3.4 kg dieldrin. These loads were consistent with the suspended sediment concentration-derived loads discussed above, while the range of local watershed pesticide loads presented in Table 3-3 reflects the variability expected between dry and wet years. Best estimates of loads were derived from the two methods of estimation.

Using the same methods, estimated chlordane loads were of similar magnitude to estimates calculated by the SIMPLE model (KLI, 2002); however, DDT loads were approximately an order of magnitude higher than SIMPLE model estimates. Lower DDT loads estimated by the SIMPLE model may be due to an underestimate of sediment loads by the model (McKee *et al.*, 2003) and the fact that non-urban sites were not well characterized in the studies by KLI (2002) and Salop *et al.* (2002). This discrepancy may not have greatly affected chlordane loads, since chlordane was primarily associated with urban land uses.

Using Guadalupe River data to estimate loads relies on the assumption that runoff from local watersheds has pesticide concentrations that are similar to those found in the Guadalupe River samples. In fact, there is great variability in pesticide concentrations (KLI, 2002; Salop *et al.*, 2002). The extent of variability remains an important unknown and introduces significant uncertainties when extrapolating from Guadalupe River data to other watersheds or applying the SIMPLE model on a regional scale. The lack of available data from other local watersheds and lack of more sophisticated modeling preclude estimating pesticide loads from the local watersheds with known accuracy or precision.

3.3.2 Loads from Wastewater Effluent

Estimates of pesticide loads from municipal wastewater were based on concentration ranges in Yee *et al.* (2001) and an estimated combined effluent discharge of 600 million gallons per day (MGD) (D. Yee, SFEI, personal communication). Contaminant data from industrial dischargers were not readily available. However, the magnitude of industrial discharge is much lower than municipal discharge (Hetzel, 2004; Johnson and Looker, 2003), and the loads from industrial discharges were simply assumed to be less than loads from municipal discharges.

3.3.3 Loads from Atmospheric Deposition

There are no local data on atmospheric deposition of pesticides to San Francisco Bay. However, ranges of wet- and dry-depositional fluxes of legacy pesticides have been estimated for other water bodies, including the Great Lakes (Chan *et al*., 1994) and Galveston Bay in Texas (Park *et al*., 2001). Chan *et al.* (1994) estimated that wet depositional fluxes in the Great Lakes ranged from 0.02-1.3 $g/km^2/yr$ for DDE and 0.2-1.9 $g/km^2/yr$ for dieldrin. The magnitudes of these fluxes were consistent with total (wet+dry) fluxes estimated by Park *et al.* (2001) for Galveston Bay: 1.9 $g/km^2/yr$ for DDTs, 0.75 $g/km^2/yr$ for chlordanes, and 0.79 $g/km^2/yr$ for cyclodienes, including dieldrin. If the magnitudes of atmospheric flux were similar in San Francisco Bay, resulting atmospheric loads would be approximately 0.02-2 kg/yr of DDTs, 0.9 kg/yr of chlordanes, and 0.2-2 kg/yr of dieldrin over the surface water area of the Bay (1.1×10^9 m^2).

3.3.4 Loads from Historic Sediment Deposits

Pesticide loads introduced from erosion of buried sediment were estimated using methods and assumptions outlined by Johnson and Looker (2003) in the mercury TMDL report for San Francisco Bay. These estimates were based on bathymetric studies of regions in Suisun Bay (Capiella *et al.*, 1999) and San Pablo Bay (Jaffe *et al.*, 1998) that were undergoing erosion as of 1990. Loading estimates from bed sediment were calculated using the following assumptions:

- There is an annual net loss of 1,100 Mkg of sediment from Suisun and San Pablo Bays.
- Eroded sediment is 50% water and 50% sediment by weight and comprises 740 kg of dry sediment per cubic meter of wet volume.
- Eroding material has approximately the same concentrations of pesticides as surface sediment monitored by the RMP.
- Eroded material remains within the Bay.

These assumptions do not account for varying pesticide concentrations with sediment depth, nor do they consider transport of eroding sediment out of the

Bay. USGS is currently developing a sediment-transport model that may be used to refine the assumptions.

Similar to Suisun and San Pablo bays, the South Bay underwent net erosion during 1956-1983 (Foxgrover *et al.*, 2003). Over this time period, approximately 70 Mm^3 (962 Mkg) of sediment eroded (an annual average of approximately 2.6 Mm^3). The total estimate of sediment erosion in the Bay is approximately 2,100 Mkg of sediment.

Pesticide loads from erosion of buried sediment were estimated using the estimate of sediment erosion and a range of surface sediment pesticide concentrations measured at ambient water RMP stations from 1991 to 1999 (excluding stations in sloughs and tributaries). The best estimate and range of loads were based on average concentrations ± one standard deviation. Average concentrations used to estimate loads of total DDTs, total chlordanes, and dieldrin were 4.1 µg/kg, 0.71 µg/kg, and 0.08 µg/kg, respectively.

Load estimates in Table 3-3 do not account for erosion and lateral mixing of especially highly contaminated sediment from areas such as the Lauritzen Channel near the Richmond shoreline. These areas also influence the extent to which bed sediment contributes to future loading; however, data for estimating pesticide loading from such areas are not readily available. This is a large uncertainty, since these areas may continue to erode.

3.3.5 Loads from Dredged Material

Pesticide loads to the Bay from dredged material disposal were estimated based on methods and assumptions used in the mercury TMDL report for San Francisco Bay (Johnson and Looker, 2003). The following assumptions were used:

- An annual average of 2.3 Myd^3 of sediment were disposed in the Bay out of 3 Myd^3 dredged.
- Dredged sediment is 50% water and 50% sediment by weight and comprises 570 kg of dry sediment per cubic meter of wet volume.
- Dredged material has approximately the same concentrations of legacy pesticides as surface sediment monitored by the RMP.
- Dredged material that is disposed of in the Bay remains within the Bay.

Net loads of pesticides from dredged material disposal were estimated from average concentrations (± one standard deviation), (Table 3-3). The dynamics associated with remobilizing or exposing contaminated sediments through dredging, the resulting magnitudes of pesticide loading to the Bay, and overall effects on water quality are unknown.

3.4 Processes

The fate of legacy pesticides in the water, sediments, and biota of San Francisco Bay is dependent upon the physical, chemical, and biological traits of the pesticides and of the San Francisco Bay environment (Figure 3-3). Processes include:

- Dissolved/solid partitioning.
- Bioaccumulation.
- Sediment transport and hydrodynamics.
- Sediment storage, mixing, and remobilization.
- Degradation in sediments.
- Degradation in water
- Volatilization.

Small differences in solubility and bioaccumulation rates of the individual pesticides affect their persistence in Bay sediment and biota and the extent to which they bioconcentrate. Sediment transport and hydrodynamics of the system affect the geographical distribution of the pesticides and the residence times of pesticides in the water column. Sediment storage, mixing, and remobilization are also major factors determining long-term fate. Degradation rates, although slow, also are important over the long-term and vary among the individual compounds.

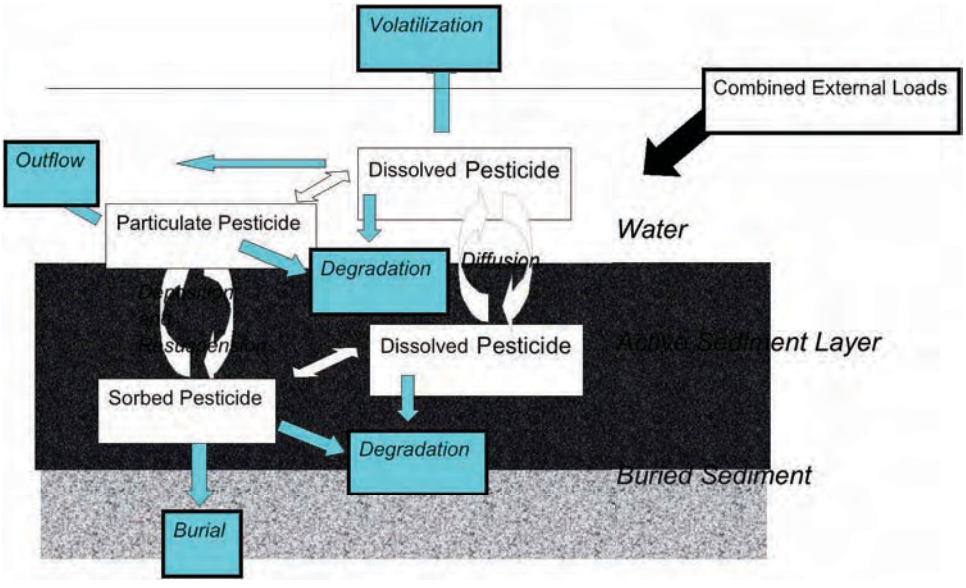

Figure 3-3. Fate of legacy pesticides in San Francisco Bay

3.4.1 Dissolved/Solid Partitioning

The chemical properties of the legacy pesticides greatly affect their fates in San Francisco Bay. All organochlorine pesticides have low solubility in water, and they are found associated with particles and sediments (Table 3-4). Solubility of prominent DDT compounds (p,p'-DDE and p,p'-DDD) and chlordanes (alpha- and gamma-chlordane and cis- and trans-nonachlor) are particularly low. Association with particles (measured as K_{OC}, the soil organic carbon partition coefficient) is greatest for DDTs and least for dieldrin.

Most of the pesticide load entering the Bay in surface runoff is associated with suspended particles. Short transit times between the sources within the watersheds and the receiving waters in the Bay do not necessarily allow sufficient time for equilibrium conditions to be reached. Therefore, association with particles can be greater than predicted by equilibrium models (Bergamaschi *et al.*, 2001; Domagalski and Kuivila, 1993).

Table 3-4. Chemical properties of legacy pesticides, summarized by Nowell et al. (1999). K_{OC}=soil organic carbon partition coefficient; K_{OW}=octanol-water partition coefficient; BCF=bioconcentration factor

Compound	Solubility mg L^{-1} (@T° C)	Log K_{OC}	log K_{OW}	log BCF	Soil Half-lives (days)
o,p'-DDD	0.1 (25)	5.36	5.06 - 6.22	4.73	730 - 5,690
p,p'-DDD	0.05 (25)	5.38	5.06 - 6.22	4.73	730 - 5,690
o,p'-DDE	0.0013 (nr)	5.58	5.69 - 6.96	4.73 - 5.26	730 - 5,690
p,p'-DDE	0.065 (24)	5.95	5.69 - 6.96	4.73 - 5.26	730 - 5,690
o,p'-DDT	-	5.63	5.98 - 6.00	4.73	2,390
p,p'-DDT	0.0077 (20)	5.63	5.98 - 6.00	4.73	110 - 5,480
Chlordane	0.06 (25)	4.78	6	4.15	365
Nonachlor	0.06 (nr)	4.86	5.66	4.34	15
Heptachlor	0.056 (25-29)	4.38	4.4 - 5.5	3.0 - 4.32	250
Heptachlor epoxide	0.275-0.35 (25)	3.89	3.65	2.93 - 4.16	4.7 - 79
Oxychlordane	0.7 (25)	2.48	2.6	1.28	-
Dieldrin	0.14 (25)	4.08	3.69 - 6.2	3.67	1,000

Monitoring data show the extent to which organochlorine pesticides partition between suspended particulate matter and the dissolved phase in the water column in San Francisco Bay (Figure 3-4). According to RMP data, DDTs are predominantly associated with particles throughout the Bay. For chlordanes, compounds are predominantly associated with particles in samples from Coyote Creek, Guadalupe River, and Petaluma River. Dieldrin, which is the most soluble of the legacy pesticides, predominantly occurs in the dissolved phase at every station in the Estuary. (One caveat to these relationships is that the RMP

operationally defines the dissolved fraction as the portion of the sample that passes through a 1-µm pore size filter. Since pesticides are sorbed onto small particles such as colloids, which pass through the filters, the fraction associated with the particulate fraction is underestimated.)

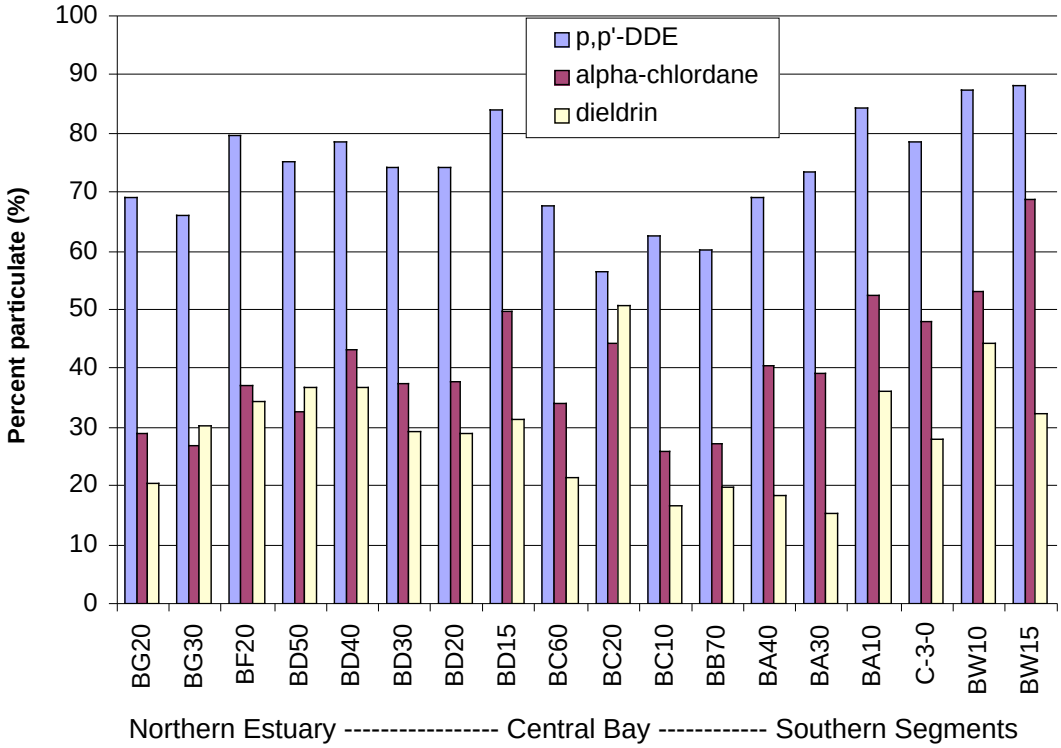

Figure 3-4. Percent contribution of particulate pesticide concentrations in water samples. Mean average values are presented for individual RMP monitoring stations from 1993 to 2001.

3.4.2 Bioaccumulation

All organochlorine pesticides are lipophilic and partition into the fats of living animals. The pesticides biomagnify, reaching higher concentrations with each step in the food chain (Figure 3-5). DDTs, particularly p,p'-DDE, bioaccumulate to a greater extent than chlordanes and dieldrin (measured as K_{OW}, the octanol-water partition factor, and BCF, bioconcentration factor, Table 3-4). Their tendency to bioaccumulate is the greatest concern for possible impairment of the beneficial uses of San Francisco Bay by the legacy pesticides.

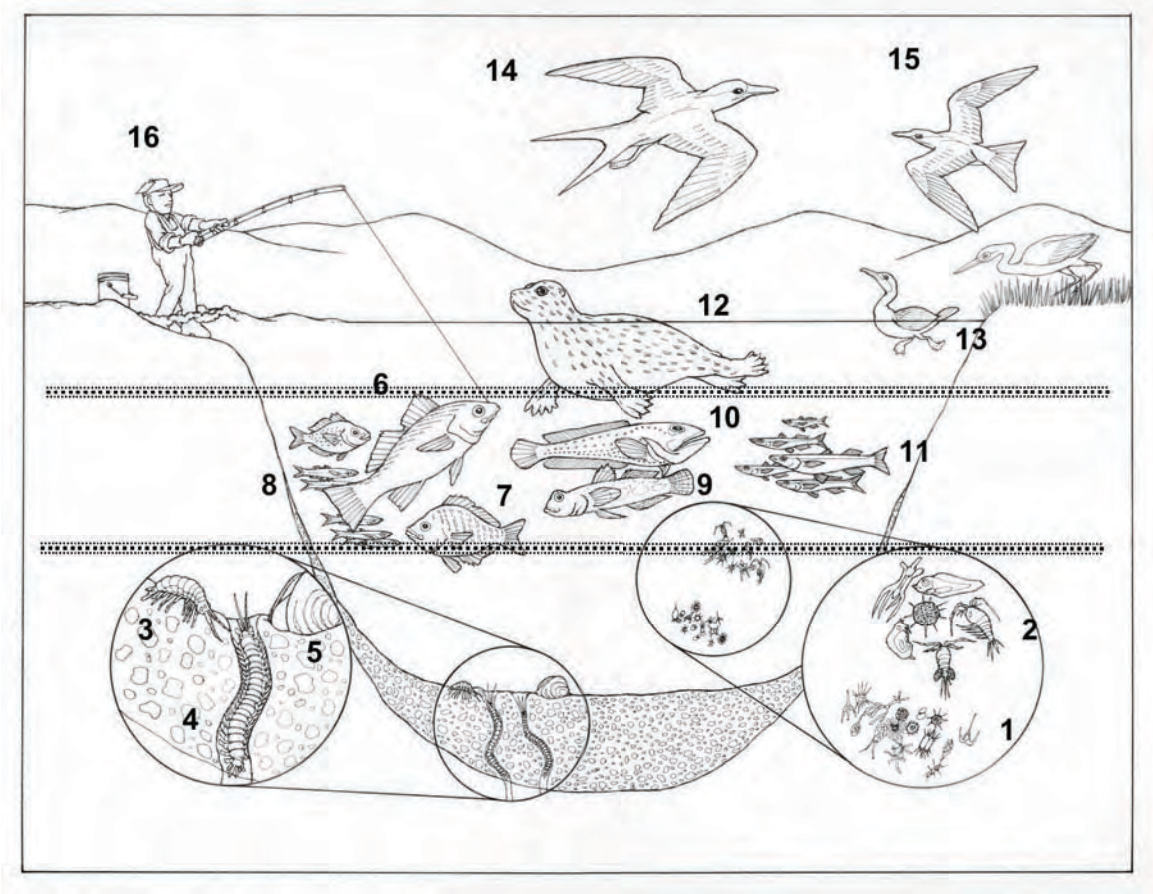

Figure 3-5. San Francisco Bay food web: (1) phytoplankton are consumed by (2) zooplankton and small invertebrates such as (3) amphipods, (4) worms, and (5) clams; (6-11) fish consume zooplankton and invertebrates; (12-16) fish are consumed by humans and wildlife species.

Food-web models are tools for linking legacy pesticide concentrations in sediment and water with concentrations in important indicator species (sport fish, birds, and seals). A food-web model for PCB movement into Bay sport fish has been developed, and the predictions of the model have been found to be in reasonable agreement with the observed concentrations (Gobas and Wilcockson, 2003). Given the strong chemical similarities between the legacy pesticides and PCBs, the general lessons learned from the PCB modeling also apply to legacy pesticides. An important finding of the study was that the structure of the food web is an important influence on concentrations found in sport fish—concentrations of PCBs in jacksmelt were higher in large fish that had consumed clams and polychaete worms than in smaller fish that had fed on phytoplankton and zooplankton. The lipid content of indicator species is another important factor. RMP fish sampling has shown that species with higher lipid content accumulate higher legacy pesticide concentrations (Greenfield *et al*., 2003). Seasonal variation in legacy pesticide concentrations in white croaker was also associated with the seasonal variation in lipid content, which reflects the reproductive cycle in this species.

3.4.3 Sediment Transport and Hydrodynamics

Sediment transport and hydrodynamics are important factors in determining the distribution of legacy pesticides throughout San Francisco Bay. Transport and distribution of pesticides associated with suspended sediments are influenced by highly variable processes, including freshwater runoff, salinity, tidal flow, and wind.

Surface runoff from the Central Valley via the Sacramento and San Joaquin rivers comprises the largest portion of freshwater flow and sediment transport to the Bay (McKee *et al.*, 2003; Kron, 1979). Consequently, the northern segments of the Bay are well-flushed, especially during large storm events. In contrast, southern segments of the Bay receive little freshwater flow, resulting in longer residence times for water, sediment, and associated pesticides.

Once legacy pesticides reach the Bay, they can be remobilized and redistributed through tidal action and wind-driven waves. These processes account for most of the variability in suspended sediment concentrations observed in the Bay (Schoellhamer *et al.*, 2003). The strongest tidal events occur during the spring tides associated with new and full moons, while the strongest winds occur during the spring and summer.

The highest concentrations of legacy pesticides in the system are found along the margins of the Bay, in areas in close proximity to urban landscapes. These areas, including Oakland Harbor, Richmond Harbor, San Leandro Bay, and the South Bay sloughs, are depositional environments, affected by runoff from urban watersheds. In areas where tidal and wind-driven mixing are insufficient to transport the deposited sediment out into the open areas of the Bay, pesticides will persist in the sediments and be available for local resuspension and possible uptake into the biota.

The residence time of the Bay and its subembayments is a key influence on the ecosystem. Leatherbarrow *et al.* (2003) estimated that outflow through the Golden Gate was a more important process of pesticide removal for dieldrin than for DDTs or chlordanes. Dieldrin is more soluble in water, making it more available for outflow. Consequently, hydrodynamics exert a greater influence on dieldrin than on the other pesticides.

3.4.4 Sediment Storage, Mixing, and Remobilization

Sediment storage, mixing, and remobilization are major determinants of the long-term fate of legacy pesticides in San Francisco Bay. Within depositional areas, pesticide residues are stored in the sediments. Data from cores taken from these depositional areas provide information on inputs, transport, and fate (*e.g.*, Venkatesan *et al.*, 1999, for San Pablo and Richardson bays; Daum *et al.*, 2000, for San Leandro Bay).

A core from San Pablo Bay had generally higher concentrations of DDTs and dieldrin relative to a core from Richardson Bay, while concentrations of chlordanes were more uniform between the cores (Figure 3-6). These findings reflect the inputs of DDTs and dieldrin from agricultural applications in the Central Valley to San Pablo Bay and the more urban application of chlordanes. Concentrations of DDTs and chlordanes were much higher in cores from shallow San Leandro Bay than in the cores from the deeper San Pablo and Richardson bays, illustrating the large contributions of pesticides from local watersheds and/or contaminated sites to the Bay margins.

In the San Pablo and Richardson bays cores, the subsurface maximum concentrations of DDTs were at least an order of magnitude greater than the subsurface maxima for chlordanes and dieldrin, while the difference was only a factor of two or three in the San Leandro Bay core. These patterns may indicate more recent use of chlordane and dieldrin in urban applications but are also confounded by varying loading and depositional processes in the sites. They also show that on a Bay-wide scale, there are greater stores of DDT than chlordane and dieldrin.

Organochlorine pesticides preferentially partition to sediments, including the active, surface layer as well as the buried deposits. The dynamics and depths of the active sediment layer are highly variable throughout the Estuary and not well-characterized (Fuller *et al.*, 1999; Davis, 2003). Recent modeling studies of contaminant fate have shown that the active sediment layer is one of the most influential, yet least understood, factors that affect the long-term fate of contaminants in the Bay (Davis, 2003; Greenfield and Davis, 2003; Leatherbarrow *et al.*, 2003). The depth of the active layer can vary from as little as three to more than 50 cm (Leahy *et al.*, 1976). Despite such great variability, a best estimate of 15 cm has been used in recent modeling of persistent organic contaminants in the Bay. Leatherbarrow *et al.* (2003) estimated that pesticide storage in an active sediment layer of 15 cm is approximately 350 kg total DDTs, 45 kg total chlordanes, and 13 kg dieldrin. These estimates are based on Bay-wide estimates of pesticide concentrations, which are highly variable. Therefore, the estimates have high degrees of uncertainty. Further characterization of the active sediment layer is necessary to improve the understanding of existing storage of legacy pesticides in Bay sediments.

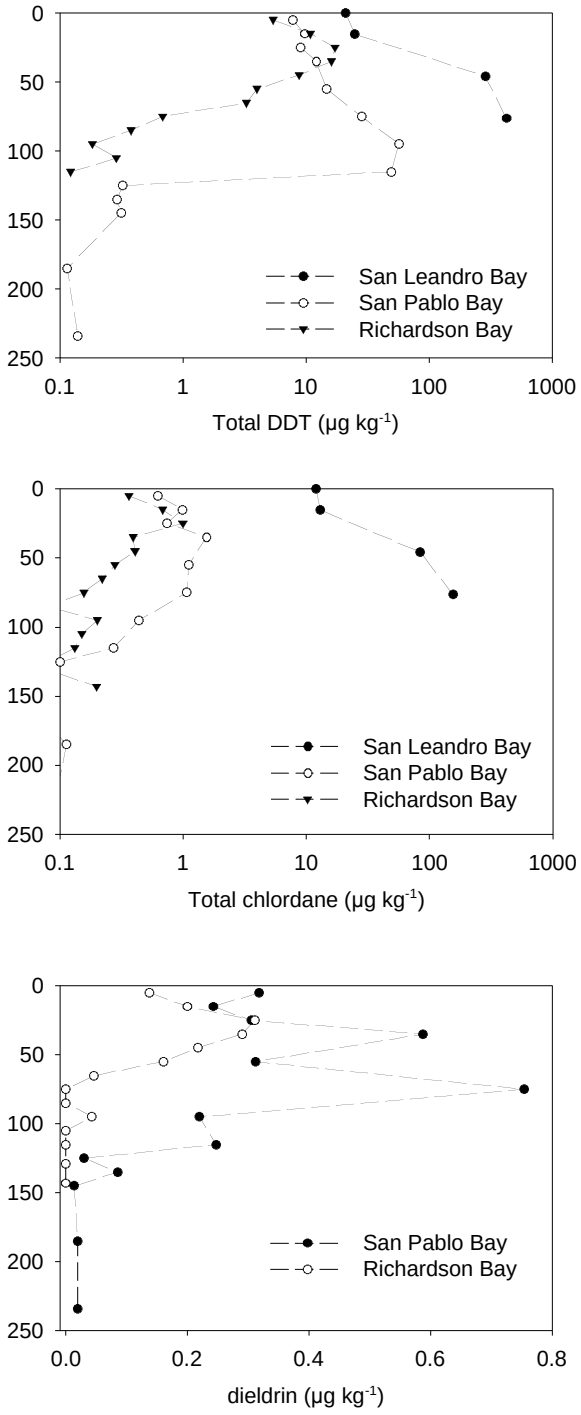

Figure 3-6. DDTs, chlordanes, and dieldrin in sediment cores from San Pablo, Richardson, and San Leandro bays (Note logarithmic scales for DDTs and chlordanes)

Estimating the mass of legacy pesticides that are buried in the Bay but that could be exposed by erosion or dredging is confounded by the lack of data. The two cores from the Bay, one from Richardson Bay and one from San Pablo Bay (Venkatesan *et al.*,1999), estimated inventories of total DDTs over the entire depths of the cores as 557 ng/cm^2 in the Richardson Bay core and 3,453 ng/cm^2 in the San Pablo Bay core. Extrapolating to the entire Bay (1.1 x 109 m^2), results in a range of buried DDT mass of 6,000 to 38,000 kg. Non-DDT pesticide inventories ranged from 1,154 ng/cm^2 to 4,069 ng/cm^2 in Richardson and San Pablo bays, respectively. Dieldrin concentrations composed approximately 2-3% of total pesticides in the cores, while total chlordanes made up an average of 10% of total pesticides in the San Pablo Bay core and 18% in the Richardson Bay core. Using these percentages, the range of masses stored in buried deposits in the Bay are approximately 3,400-8,300 kg chlordanes and 380-2,500 kg dieldrin.

Spatial variability in contamination and depositional patterns introduces large uncertainties in estimating Bay-wide storages of pesticides in buried sediments from only two cores. The cores were collected offshore and provide only a rough estimate of pesticide storage in offshore or ambient Bay sediments. The lack of data from Bay margins precludes making an estimate of pesticide storage along the shorelines and in the sloughs.

Despite the uncertainties, existing data indicate that further erosion in the Bay could remobilize sediments that are more contaminated than those in the current active layer. Exposure of buried sediments along the Bay margins of particularly contaminated areas, such as Oakland Harbor or San Leandro Bay, could reintroduce sediments with high concentrations of pesticides. This possibility may be especially important with regard to sport fish, as the highest concentrations of chlordanes and dieldrin measured in the fish from the 2000 RMP collections were shiner surfperch and white croaker from Oakland Harbor and San Leandro Bay.

3.4.5 Degradation in Soils and Sediment

The long-term persistence of DDTs, chlordanes, and dieldrin in watershed soils is well-documented (*e.g.*, Gilliom and Clifton, 1990; Mischke *et al.*, 1985; Spencer *et al.*, 1996; Stewart and Chisholm, 1971; Castro and Yoshida, 1971). Spencer *et al.* (1996) found that total DDT concentrations measured in the top 75 cm of agricultural soil samples collected in California in 1994 were approximately 10-28% of the concentrations measured in 1971. Assuming first order reaction rates, this corresponds to a half-life of approximately 7 to 13 years. At these rates, approximately 2-11% of total DDT applied in 1965 still remained in watershed soils in 2003.

The degradation of DDT to DDD and DDE poses an added complexity in understanding the total degradation rates. DDT readily undergoes reductive dechlorination under anaerobic conditions, and the flooding of soils promotes the degradation of DDT to DDD (Castro and Yoshida, 1971). In aerobic environments, DDT is dehydrochlorinated to DDE. Both DDD and DDE are much more recalcitrant in aerobic and anaerobic soils (Castro and Yoshida, 1971; Strompl and Thiele, 1997). As a result, in estuarine sediment, transformation rates of DDT compounds decrease in the order: DDT > DDD > DDE (Huang *et al*., 2001).

Degradation rates have been shown to increase with increasing moisture in soil (Spencer *et al.*, 1995; Castro and Yoshida, 1971; Ghadiri *et al.*, 1995) and marine sediment (Kale *et al.*, 1999), suggesting that degradation rates are higher in the Bay than in watershed soils. However, degradation rates of organochlorine pesticides in marine and estuarine sediment have not been well-studied.

Leatherbarrow *et al*. (2003) compiled literature estimates of degradation rates in soil and sediment to derive applicable rates for a mass budget model of organochlorine pesticide fate in San Francisco Bay (Table 3-5).

Table 3-5. Best estimates of half-lives of legacy pesticides in soil and sediment

Pesticide	Half-life (years)
DDT	9
Chlordane	2.3
Dieldrin	2.8

There was considerable uncertainty associated with the degradation rate estimates, which typically spanned an order of magnitude. Estimated degradation rates for dieldrin spanned two orders of magnitude.

3.4.6 Degradation in Water

Degradation of legacy pesticides is faster in water than in soils or sediments. However, given that an estimated 97-99% of the mass of DDTs, chlordanes, and dieldrin in the Bay is associated with the actively mixed sediment layer, degradation in water is not thought to be a major removal process (Leatherbarrow *et al*., 2003). In the water column, degradation occurs by direct and indirect photolysis and hydrolysis. Hydrolysis of legacy pesticides is not expected to be important (Mackay *et al*., 1997).

While photolysis rates of DDT and DDD are not expected to be important (Callahan *et al.*, 1979), rates of DDE photolysis that would essentially remove all DDE from a water body within one day have been reported (Zepp and Cline, 1977). The persistence of p,p'-DDE and other DDT compounds in the water column and sediment of San Francisco Bay indicate that rates of degradation are

probably much slower than the reported values. Persistence of p,p'-DDE in other surface water bodies has been explained by its sorption to sediment (Zepp *et al.*, 1977), which can decrease photolysis rates in the water column (Miller and Zepp, 1979; Oliver *et al.*, 1979). Moreover, attenuation of sunlight in natural waters decreases photolysis rates of organic contaminants within the top few centimeters (Zepp and Cline, 1977).

3.4.7 Volatilization

Volatilization of pesticides from the water column is expected to be an important pathway of pesticide removal from the Bay, based on the mass budget model (Leatherbarrow *et al.*, 2003). No data have been collected to directly study air-water exchange of legacy pesticides within the Bay; however, Henry's law constants compiled from the literature were used to estimate volatilization rates. (Henry's law states that the mass of a gas that dissolves in liquid is proportional to the pressure of the gas.) Varying Henry's law constants had only a minor effect on model output, indicating that uncertainty in this parameter is less important than others for estimating long-term fate of legacy pesticides in the Bay.

Of DDT compounds, p,p'-DDE has the highest volatility, increasing with soil moisture (Spencer *et al.*, 1996). Due to increased degradation and volatility of p,p'-DDE with increased moisture, agricultural areas that have been plowed and irrigated have shown greater long-term declines in total DDT concentrations than areas that have not been similarly managed (Spencer *et al.*, 1996). Since most of the total DDT residues measured in San Francisco Bay sport fish is in the form of p,p'-DDE, watershed management efforts that promote the degradation and volatilization of more volatile species, such as p,p'-DDE, could lead to faster reductions in the pesticide mass entering the Bay and more rapid declines in fish tissue concentrations.

3.5 Recovery of the Bay

The recovery of San Francisco Bay from legacy pesticide contamination was evaluated using the one-box mass budget model, which considers inputs and losses to the water column and the sediment active layer, which interacts with the overlying water. The underlying mass of buried sediment is considered a long-term sink. The model accounted for five major pathways of addition to or removal of pesticides from the Bay: loading to the Bay, outflow to the Pacific Ocean, volatilization to the atmosphere, permanent burial to the sediment, and degradation (Figure 3-7). The loading term encompasses all inputs, including runoff from the Central Valley and the local watersheds, wastewater effluent, atmospheric deposition, and erosion or dredging of bottom sediments. The model also accounted for transfer of pesticides between water and the sediment active layer.

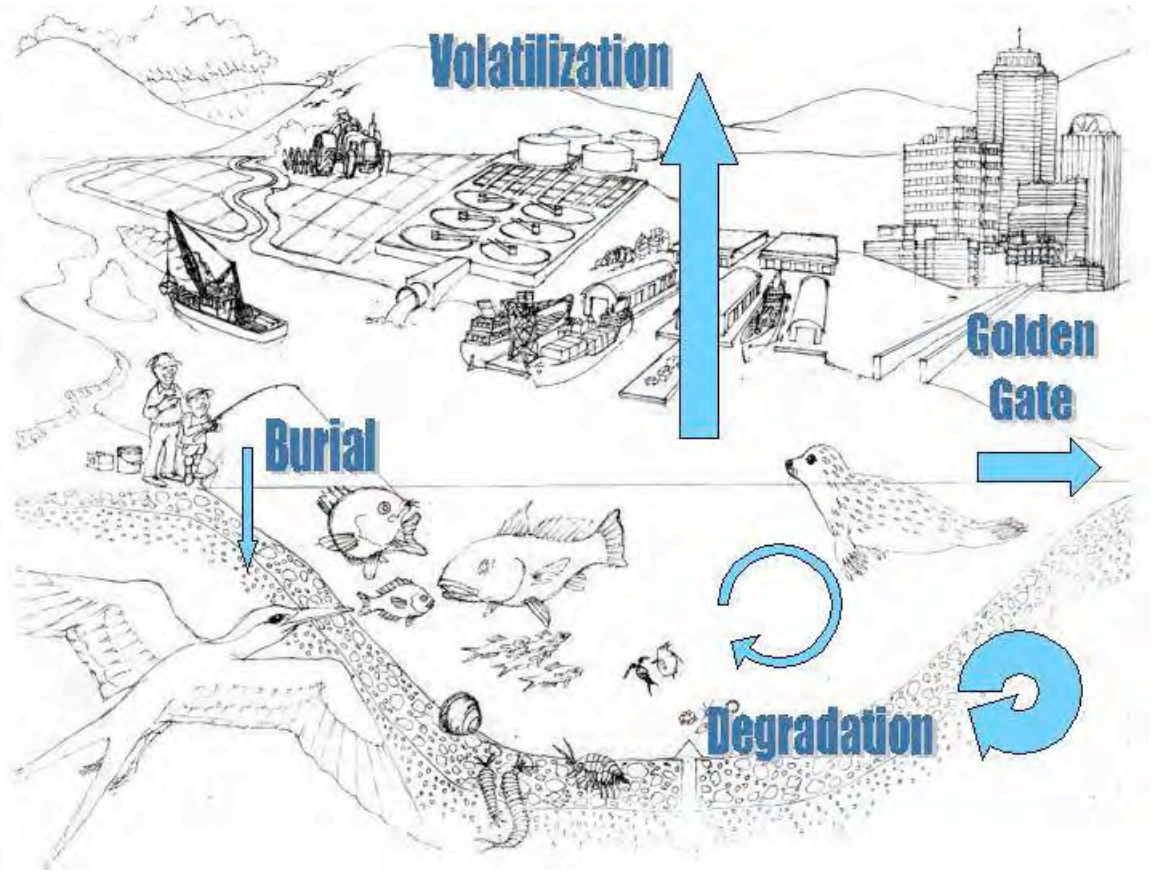

Figure 3-7. Loss pathways

Significant uncertainties were introduced to the model by using Bay-wide estimates of spatially variable parameters, such as concentrations in sediments and the depth of the active layer. Other uncertainties, such as those associated with loading and degradation, added to the overall uncertainty in model output. The cumulative effect of these uncertainties has not been defined. Consequently, the modeling exercise was only an initial attempt to integrate existing information. As improved information becomes available, more sophisticated modeling approaches can be used.

3.5.1 Current Inventory

Because DDTs, chlordanes, and dieldrin are sparingly soluble in water, most of the current mass of legacy pesticides in San Francisco Bay (excluding the buried sediment) resides in the active layer of the sediments rather than the water column (Leatherbarrow *et al.*, 2003; Davis *et al.*, 2003) (Table 3-6). Assuming a 15-cm active-sediment-layer depth, pesticide mass in sediments comprises 97-99% of the total mass of pesticides in the system, with an estimated 347 kg DDTs, 45 kg chlordanes, and 12 kg dieldrin.

Table 3-6. Concentration and mass of legacy pesticides in water and sediments

	Concentration		Mass		
	Water pg/l	**Sediment** µg/kg	**Water** kg	**Sediment** kg	**Total** kg
DDT	660	4.2	3.6	347	350
Chlordane	160	0.54	0.88	45	45
Dieldrin	77	0.15	0.42	12	13

3.5.2 Removal Pathways

The model compared the relative importance of the pathways of legacy pesticide removal from the Bay: outflow, degradation in water and sediment, volatilization, and burial (Figure 3-8). In all modeling scenarios, the rate of burial was assumed to be zero, based on bathymetric studies that have shown net erosion of sediments from the Bay in recent decades (*e.g.*, Foxgrover *et al.*, 2003).

The model results indicated that degradation in sediment is the major pathway of removal. Over a 10-year period with no loading, 72% of p,p'-DDE mass would be removed from the Bay, mostly through degradation in sediment. Similarly, the model predicted that 98% of the initial alpha-chlordane mass would be removed in 10 years, with most removed through degradation in sediment. Lower affinity of dieldrin for sediment would lead to removal processes in the water column having a greater effect than for other pesticides. All of the dieldrin mass was estimated to be removed within 10 years.

As previously noted, degradation rates of legacy pesticides in Bay sediments are important to estimates of long-term fate, but they are poorly characterized. For example, increasing the half-life of DDT over a plausible range of 2 to 16 years decreases the mass of DDT removed from the Bay in five years from 86% to 37%. However, even using the slowest reported degradation rates, degradation is the most important removal pathway for DDTs and chlordanes. Volatilization and outflow may be the more important removal pathways for the more soluble dieldrin.

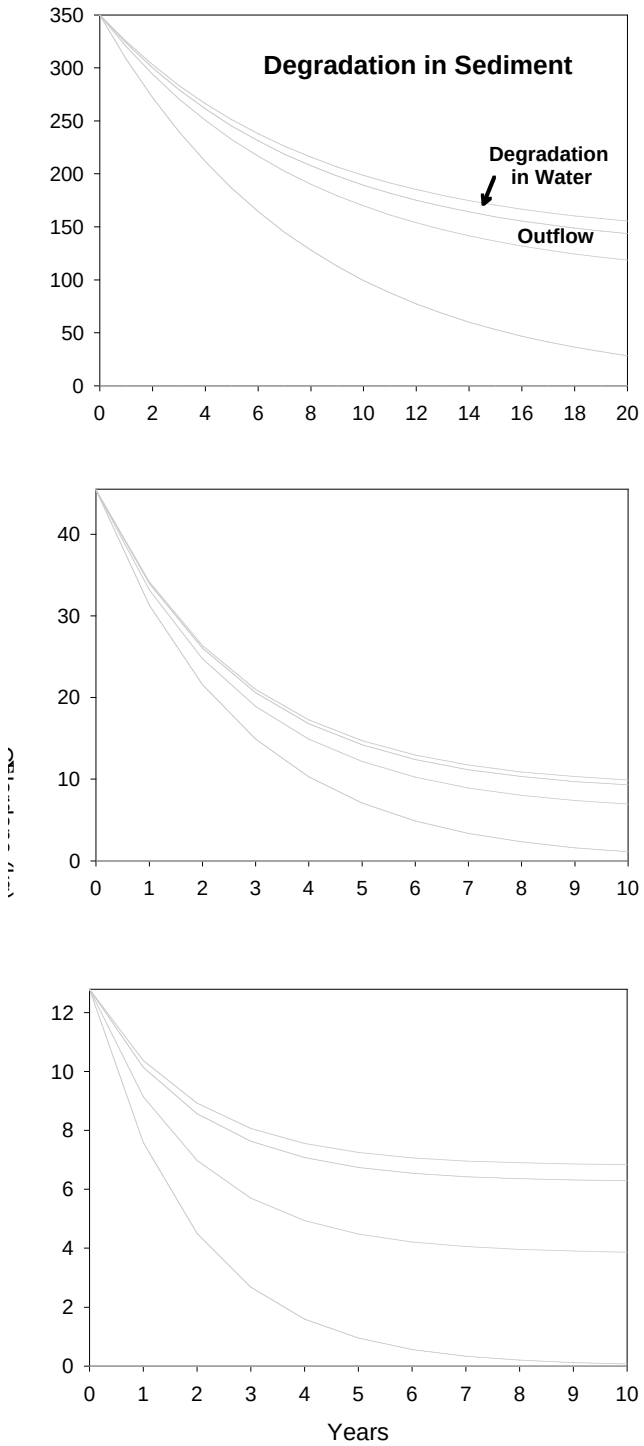

Figure 3-8. Removal pathways for legacy pesticides

3.5.3 Removal Rates

The model was used to compare estimated recovery times of the Bay under various loading scenarios. Under a scenario with no new pesticide loading, the model estimated that DDT was the most persistent of the legacy pesticides, with a half-life of about five years (Figure 3-9). The half-life of chlordane was about two years, and the half-life of dieldrin was about one year, reflecting its higher degradation rate.

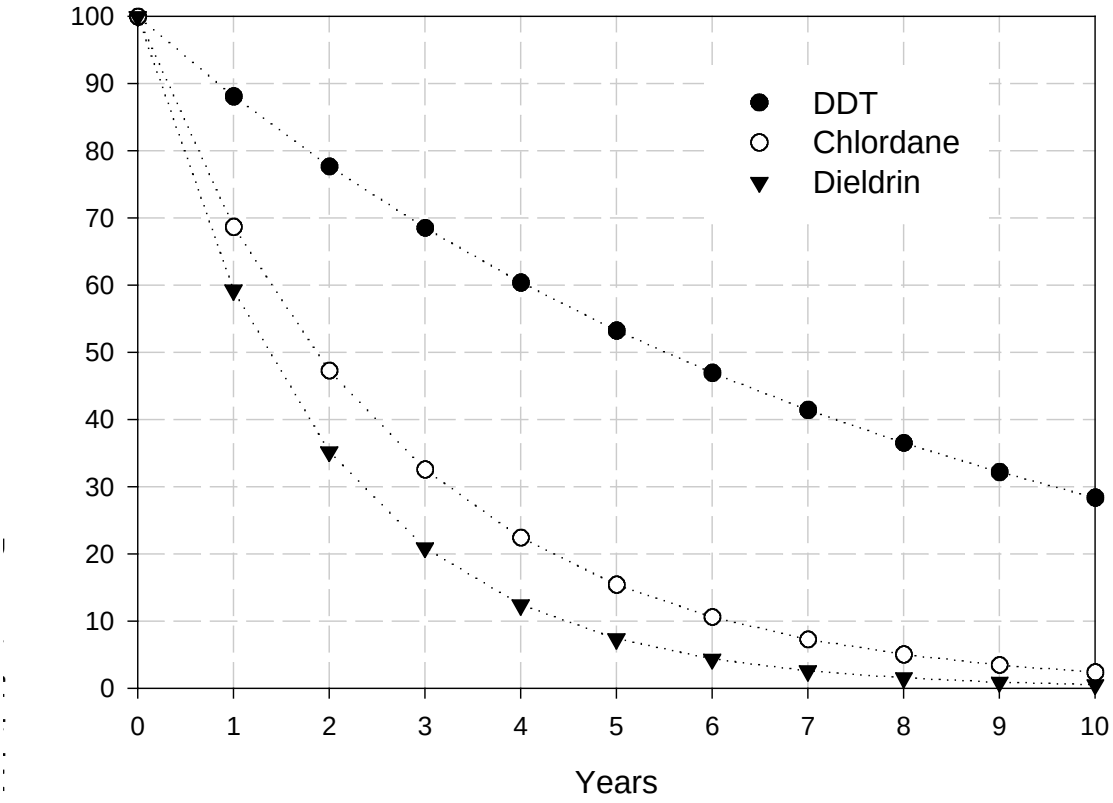

Figure 3-9. Half-lives of legacy pesticides under conditions of no new loading

The model estimated that, under conditions of no loading, San Francisco Bay would be cleared of legacy pesticides within one to three decades. In the absence of loading, 95% of the current mass DDTs, chlordanes, and dieldrin would be removed from the Bay within 25, 8, and 6 years, respectively.

Under scenarios of continued loading to the Bay, the model estimated that the Bay would eventually reach steady states of inputs and outputs (Figure 3-10). Annual loads of about 60 kg DDTs and 20 kg chlordanes and dieldrin would be sufficient to prevent any decrease in the current mass of pesticides in the Bay. These loads are of similar magnitude to the best estimates of current pesticide loads calculated for this report, 60 kg DDTs, 30 kg chlordanes, and 10 kg dieldrin. Although there are significant uncertainties in both the load estimates and the model outputs, the

results suggest a question as to whether pesticide mass (and concentrations) in the Bay will decline.

Actual trends in pesticide concentrations in the Bay were evaluated using bivalve data collected by the State Mussel Watch Program and the RMP (Figure 3-11). For all three pesticides, there have been obvious declines in concentrations over time, but these declines are less apparent since the early 1990s (Gunther *et al.*, 1999; Leatherbarrow *et al.*, 2003). Half-lives estimated from bivalve data were two to eight times longer than those measured by the model (under the scenario of no additional loading). The longer half-lives estimated from bivalve data provide evidence of continued pesticide loading to the Bay during the last two decades. Bivalve data also support model predictions that p,p'-DDE will be more persistent than other pesticides.

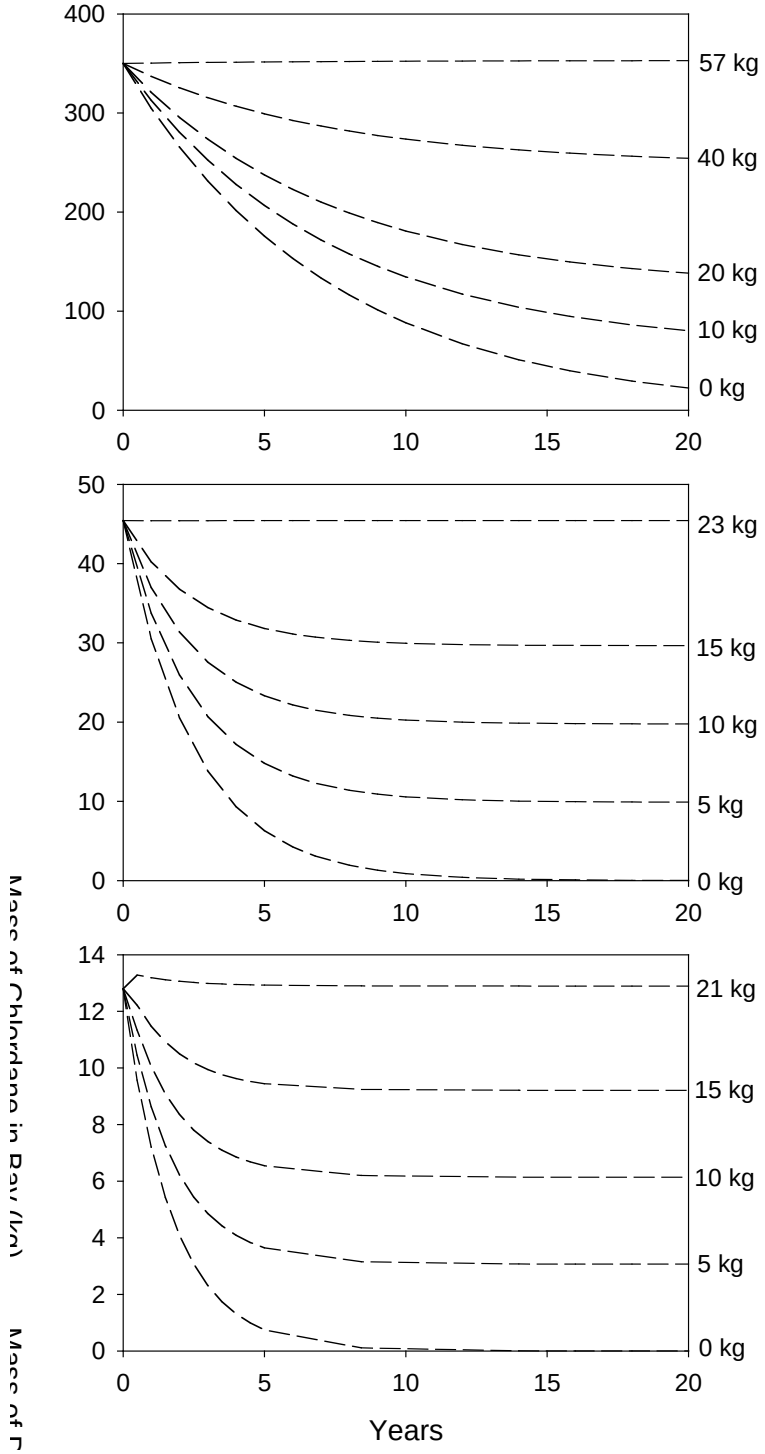

Figure 3-10. Model estimates of the change of legacy pesticide mass in San Francisco Bay with varying pesticide loading. (DDTs were modeled as p,p'-DDE. Chlordanes were modeled as alpha-chlordane.)

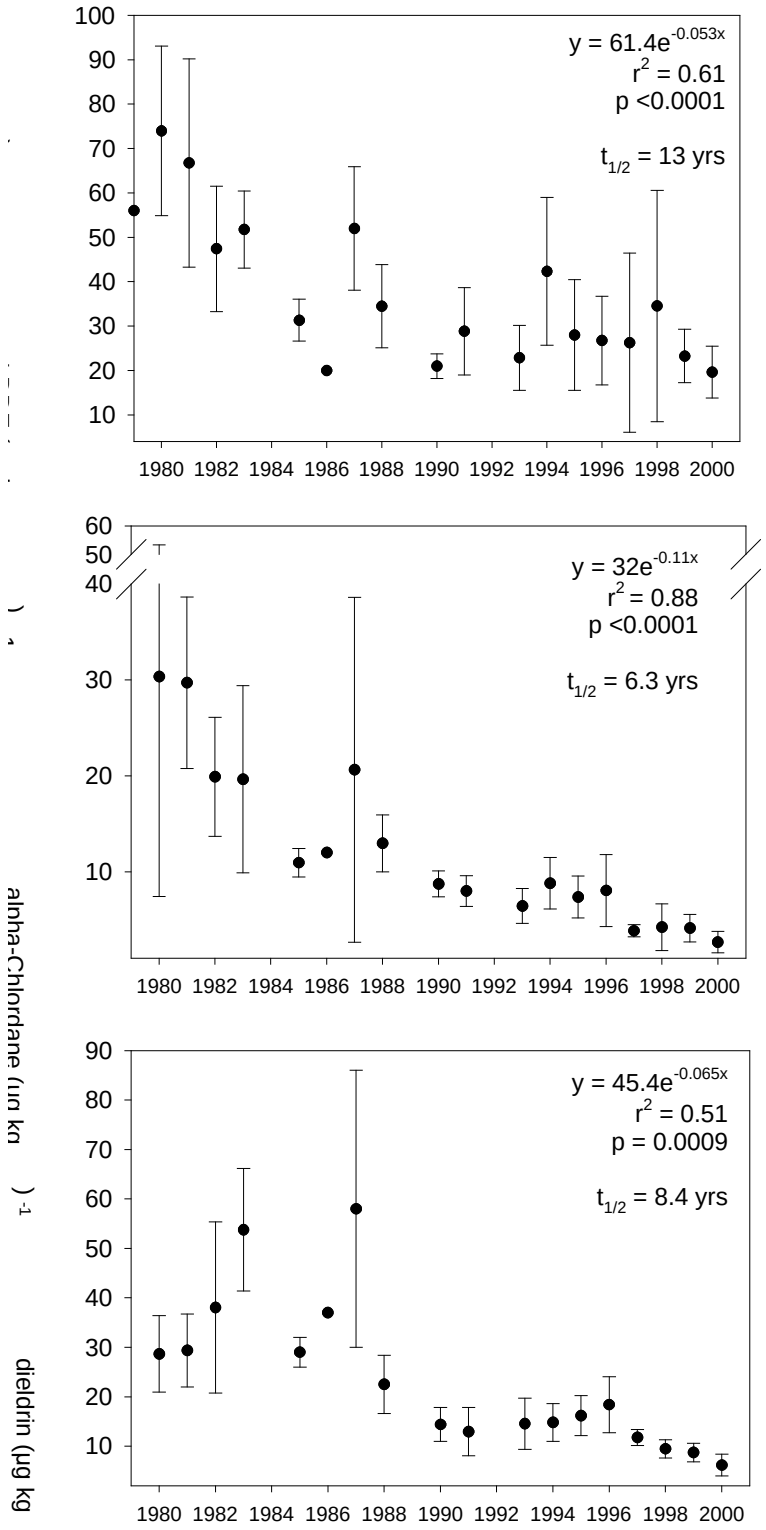

Figure 3-11. Pesticide concentrations in bivalves

4. Information Gaps

This section summarizes the uncertainties in this report's conclusions and suggests some potential future projects to obtain additional data and conduct more analysis of the sources, fate, transport, and effects of legacy pesticides. In other documents or forums, the CEP will develop appropriate strategies for addressing legacy pesticides in the Bay and its watersheds. These strategies may include:

- Data collection or analysis.
- Implementation of corrective actions.
- Formulating and refining management questions and setting priorities for the above two activities.
- Determining an ongoing process for integrating all of the above.

There may be control measures, remediation, and regulatory actions that can and should begin now, even with existing uncertainties. CEP partners are committed to identifying these actions. Future CEP data gathering and technical analysis should focus on determining the potential effectiveness and actual effects of actions to reduce or eliminate impairment and to restore beneficial uses of the Bay.

Understanding the fate of the legacy pesticides and the level of impairment that they cause to San Francisco Bay is hampered by a variety of data gaps and uncertainties:

- Lack of established criteria for determining impairment.
- Uncertain understanding of trends in pesticide concentrations.
- Lack of understanding of sediment "hot spots."
- Uncertain understanding of the large runoff events from the Central Valley.
- Uncertain understanding of loads from small tributaries.
- Model uncertainties.

4.1 Impairment Criteria

The beneficial use of the Bay that is most probably impaired by legacy pesticides is its value for sport fishing. Data from the water column and fish tissue provide some indications of this impairment. However, a good definition of impairment to sport fishing in San Francisco Bay depends upon calculation of screening values, and there are no standards for these values. There are also no established criteria for interpreting sediment or wildlife data. OEHHA, the San Francisco RWQCB, and USEPA should work together to establish criteria for determining impairment.

4.2 Trends in Pesticide Concentrations

Monitoring data collected over a span of decades indicate that there have been large declines in concentrations of legacy pesticides in fish tissues. Trends over a shorter, more recent period are not as apparent. While pesticide loading to the Bay is probably declining, erosion of sediment may expose contaminated layers that had been buried. However, the best records of trends in sediments come from just two sediment cores taken by the USGS in the early 1990s. Additional information is needed to determine whether concentrations and loads of legacy pesticides in the Bay are continuing to decline, remaining somewhat stable, or even increasing. This is a primary data gap for which information is needed.

Possible actions include:

- **Continued monitoring of concentrations of pesticides in fish, bird eggs, water, and sediment** is necessary to determine trends and regularly inform managers about the status of impairment of the Bay.
- **Collection and analysis of sediment cores** from depositional areas would provide information on recent trends in pesticide concentrations. Cores collected from erosional areas would help determine the potential for future re-introduction of pesticides. Cores collected from near-shore environments and the individual segments of the Bay would provide information on geographic variation.

4.3 Near-Shore Locations and Hot Spots

Monitoring has shown that areas of contamination exist in near-shore locations in the Bay and in localized areas within the watersheds. The location of such "hot spots" within urban areas may especially affect subsistence anglers, who depend upon fishing for food. For contaminated sites within the Bay, there is uncertainty as to whether processes such as erosion or dredging are contributing to overall loading of pesticides to the Bay. Many sites that are contaminated by legacy pesticides are also affected by other contaminants of concern, such as PCBs. As a result, management actions that successfully reduce PCB contamination will also reduce contamination by legacy pesticides.

Studies of water and sediment in close proximity to hot spots may provide information on the extent to which remobilization of near-shore sediments influences impairment of the Bay. These studies may assist managers in deciding whether to direct management resources to in-Bay remediation or to source reduction efforts in the watersheds.

4.4 Runoff from the Central Valley

The Central Valley is a significant source of legacy pesticides to San Francisco Bay because of historic use of pesticides in its predominantly agricultural land and the large magnitude of sediment and freshwater flow from the Sacramento and San Joaquin rivers. An RMP special study of loading to the Bay from the rivers was conducted during years of relatively low to moderate Delta outflow and relatively low concentrations of suspended sediments. During larger storm events, different sources of pesticides within the watersheds might be activated. Characterization of the large runoff events would provide better information for understanding this large source of pesticides to the Bay.

It is difficult to predict the occurrence of wet *vs.* dry years, and it is especially difficult to predict the occurrence of the largest storm events. Therefore, successful characterization of large runoff events would require appropriate preparedness to conduct sampling whenever such an event occurred. Ideally, a monitoring plan would include sampling every year, which would provide a good understanding of interannual variability. A more efficient plan should provide for sampling on an opportunistic basis, focusing on large flood events. The number of years that such a study would last is difficult to predict.

4.5 Loadings from Small Tributaries

Tributaries that drain the local watersheds are important pathways for legacy pesticides in terms of the magnitude of loads and also in determining practical management actions. Currently, there is uncertainty in estimates of loads from small tributaries. Data from the Guadalupe River have indicated that concentrations of legacy pesticides are correlated with sediment and discharge in a predominantly urban watershed. How these relationships differ in watersheds of varying land use and other characteristics and how they may change in response to management actions is not known. In particular, there are no recent data for pesticide concentrations in agricultural parts of the Bay Area. Also, as management actions are planned and implemented in local watersheds, it will be necessary to assess their effectiveness.

Possible actions include:

- **Characterization of watersheds with varying land use.** Concentrations of legacy pesticides and suspended sediments should be measured in the water column during runoff events for representative watersheds. This information would afford a better understanding of the range of variability in pesticide transport processes and their influence on concentrations and loading. The information would provide greater ability to extrapolate from existing data to the Bay as a whole.
- **Determination of the effectiveness of management actions.** Methods for evaluating the success of local management actions should be

established. This action may require a review of methods that other geographic areas have used to determine success in clean-up efforts. Assessment of success resulting from the PCB TMDL may also apply to legacy pesticides.

4.6 Modeling Uncertainties

A mass budget modeling study of legacy pesticides in the Bay integrated existing information to provide an initial understanding of recovery rates of the Bay. However, the current one-box model lacks the spatial and temporal resolution necessary for accurate predictions of pesticide fate and transport in response to fluctuations in sediment transport and hydrodynamics in different parts of the Bay. There are several sources of uncertainty that may be reduced by further investigation:

- The model's annual time step neglects the effect of shorter-term variability in sediment transport and hydrodynamics.
- Better Bay-wide estimates of spatially variable parameters, such as pesticide concentrations in the sediments and the depth of the active layer, are essential for estimating the current reservoir of pesticide storage in the Bay and the time scales for recovery.
- Degradation rates in sediments are not well defined but are important factors for determining recovery.
- Linkage between concentrations and loads of legacy pesticides in the water and the sediments and those in the biota has not been established.

Possible actions include:

- **Development a five-box model of pesticide fate and transport.** A five-box model is currently being developed to predict the transport and fate of PCBs on a daily time step in five major regions of the Bay. If the modeling exercise is successful for PCBs, it could be applied to legacy pesticides as well and would provide greater temporal and spatial information.
- **Development a food-web model.** A food web model has been developed to link concentrations of PCBs in water and sediment samples with concentrations in important indicator organisms. The legacy pesticides share many chemical attributes with PCBs, and it is possible that this model could be adapted for use with pesticides.

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