CHAPTER 3.0 DREDGING AND DREDGED MATERIAL CHARACTERISTICS — AN OVERVIEW

This chapter provides an overview of the dredging process and the sediment characteristics that affect disposal and reuse of dredged material. A basic understanding of how dredged sediments and any associated contaminants behave in different circumstances (for example, at upland versus aquatic placement sites) is critical to managing dredged material in a manner that minimizes potential environmental impacts and risks, and that maximizes environmental, societal, and economic benefits. The first section (section 3.1) describes the dredging process itself, including the basic types of dredging and disposal equipment, with an emphasis on how the dredging method used affects the feasibility of various management options. The next section (section 3.2) discusses the major physical, chemical, and biological characteristics of Bay area sediments in general — and dredged material in particular — that provide the basis of appropriate dredged material management.

3.1 DREDGING IN THE SAN FRANCISCO BAY REGION

Each year, over 4,000 commercial, ocean-going vessels navigate into or through the San Francisco Bay/Delta Estuary (the Estuary), carrying over 50 million tons of cargo to eight public and numerous other private ports and harbors between Sacramento and Redwood City. The Estuary has also been an important center of naval and other military operations through the years. In addition, over 1,000 commercial fishing vessels operate out of San Francisco Bay, and over 200 marinas provide slips for over 33,000 recreational boats. Together, these activities fuel a substantial maritime-related economy of over $7.5 billion annually. However, the facilities supporting these activities are located around the margins of a bay system that averages less than 20 feet deep, while modern, deep-draft ships often draw 35 to 40 feet of water or more. Extensive dredging — in the range of 2 million to 10 million cubic yards (mcy) per year — is therefore necessary to create and maintain adequate navigation channels in order to sustain the region's diverse navigation-related commercial and recreational activities. Effective management of the large volumes of dredged material generated throughout the Estuary is a substantial challenge. The following sections discuss dredging and disposal methods used in the Estuary, and the amount of dredged material anticipated to be generated over the 50-year LTMS planning period.

3.1.1 Dredging and Disposal Methods

3.1.1.1 General

This section provides a brief overview of the dredging process, including types of dredges, types of impacts that may be associated with dredging, transportation systems, and the placement or disposal practices commonly used in navigation-related dredging projects as described in the joint EPA/COE national guidance document, *Evaluating Environmental Effects of Dredged Material Management Alternatives — A Technical Framework* (USEPA and USACE 1992). The indicated references provide a more detailed description of different kinds of dredges, transport equipment, and disposal practices.

The removal or excavation, transport, and placement of dredged sediments are the primary components of the dredging process. In design and implementation of any dredging project, each part of the dredging process must be closely coordinated to ensure a successful dredging operation.

The excavation process commonly referred to as "dredging" involves the removal of sediment in its natural or recently deposited condition, using either mechanical or hydraulic equipment. (Dredging sediments in their natural condition is referred to as new work construction; dredging recently deposited sediments is referred to as maintenance dredging.) After the sediment has been excavated, it is transported from the dredging site to the placement site or disposal area. This transport operation, in many cases, is accomplished by the dredge itself or by using additional equipment such as barges, scows, and pipelines with booster pumps.

Once the dredged material has been collected and transported, the final step in the dredging process is placement in either open-water, nearshore, or upland locations. The choice of management alternatives involves a variety of factors related to the dredging process including environmental acceptability, technical feasibility, and economic feasibility of the chosen alternative.

3.1.1.2 Dredging Process, Equipment, and Techniques

The dredging equipment, techniques used for excavation and transport of the material, and the disposal alternatives considered must be compatible. The types of equipment and methods used by both the COE and private industry vary considerably throughout the United States. The most commonly used dredges are illustrated in Figure 3.1-1. Dredging equipment and dredging operations resist precise categorization. As a result of specialization and tradition in the industry, numerous descriptive, often overlapping, terms categorizing dredges have developed. For example, dredges can be classified according to the basic means of moving material (mechanical or hydraulic); the device used for excavating sediments (clamshell, cutterhead, dustpan, and plain suction); the type of pumping device used (centrifugal, pneumatic, or airlift); and others. However, for the purpose of this document, dredging is accomplished basically by only two mechanisms:

- *Hydraulic dredging* — Removal of loosely compacted materials by cutterheads, dustpans, hoppers, hydraulic pipeline, plain suction, and sidecasters, usually for maintenance dredging projects.

- *Mechanical dredging* — Removal of loose or hard compacted materials by clamshell, dipper, or ladder dredges, either for maintenance or new-work projects.

Hydraulic dredges remove and transport sediment in liquid slurry form. They are usually barge-mounted and carry diesel or electric-powered centrifugal pumps with discharge pipes ranging in diameter from 6 to 48 inches. The pump produces a vacuum on its intake side, which forces water and sediments through the suction pipe. The slurry is transported by pipeline to a disposal area. Hopper dredges are included in the category of hydraulic dredges for this report even though the dredged material is simply pumped into the self-contained hopper on the dredge rather than through a pipeline. It is often advantageous to overflow excess water from hopper dredges to increase the sediment load carried; however, this may not always be acceptable due to water quality concerns near the dredging site.

Mechanical dredges remove bottom sediment through the direct application of mechanical force to dislodge and excavate the material at almost *in situ* densities. Backhoe, bucket (such as clamshell, orange-peel, and dragline), bucket ladder, bucket wheel, and dipper dredges are types of mechanical dredges. Sediments excavated with a mechanical dredge are generally placed into a barge or scow for transport to the disposal site.

Selection of the dredging equipment and method used to perform the dredging depends on the following factors:

- Physical characteristics of the material to be dredged;
- Quantity of material to be dredged;
- Dredging depth;
- Distance to disposal area;
- Physical environment of the dredging and disposal areas;
- Contamination level of sediments;
- Method of disposal;
- Production rate required (e.g., cubic yards per hour);
- Types of dredges available; and
- Cost.

Water quality at the dredging and disposal sites is a particularly important consideration in the choice of dredging equipment. Hydraulic dredging can virtually eliminate disturbance and resuspension of sediments at the dredging site, and is often the first choice when dredging occurs in enclosed waterbodies or in locations near aquatic resources that would be especially sensitive to temporary increases in suspended solids or turbidity. However, because hydraulic dredging typically entrains additional water that is many times the volume of sediment removed, water management and water quality must be controlled at the disposal site. In contrast, mechanical dredging creates little additional water management concern at the disposal site because little additional water is entrained by mechanical dredging equipment; therefore mechanical dredging is usually the first choice when disposal site capacity limitations are a primary concern. However, typical mechanical equipment often creates more disturbance and resuspension of sediments at the dredging site.

More detailed descriptions of dredging equipment and dredging processes are available in Engineer Manual (EM) 1110-2-5025 (USACE 1983), Houston (1970), and Turner (1984).

Figure 3.1-1 Types of Dredges

3.1.1.3 The General Impacts of Dredging

This section describes briefly the types of impacts associated with dredging activities in general. Most of the impacts from dredging are temporary and localized and, with the exception of impacts associated with a changed bottom topography (potential change in local hydrodynamics and in the makeup of the benthic resources present in the dredge area), the impacts end when the dredging ends. The most substantial impacts tend to be on water quality, the potential for resuspension of contaminants buried in the sediments, and the impacts on biological resources in the dredge area. These types of impacts are therefore discussed in more detail below.

Potential Impacts on Water Quality

Water quality variables that can be affected by dredging operations include turbidity, suspended solids, and other variables that affect light transmittance, dissolved oxygen, nutrients, salinity, temperature, pH, and concentrations of trace metals and organic contaminants if they are present in the sediments (U.S. Navy 1990).

Dredging resuspends bottom sediments and thus temporarily increases the turbidity of surface waters. Chemical reactions can occur between the suspended materials and the surrounding Bay water. The primary controlling factors would be the redox potential of the seawater, the pH of the seawater and, to a lesser degree, the salinity (Pequegnat 1983). ("Redox potential" refers to the reduction-oxidation potential, which is a measure of the availability and activity of oxygen to enter into and control chemical reactions.) The fine-grained sediment fractions (clay and silt) have the highest affinity for several classes of contaminants, such as trace metals and organics, and tend to remain in the water column longer than sand because of their low settling velocities (U.S. Navy 1990). Oxygen in the seawater would promote oxidation of the organic substances in the suspended materials This, in turn, can release some dissolved contaminants, particularly the sulfides (U.S. Navy 1990).

Depending on the dredging method used, dissolved oxygen (DO) concentrations in the water column can be substantially reduced during dredging if the suspended dredged material contains high concentrations of oxygen demanding substances (e.g., hydrogen sulfide). The reduction of DO during dredging is minimal (1 to 2 ppm) and transitory in surface waters, but can be more severe in bottom waters (reduction of up to 6 ppm for 4 to 8 minutes). Most estuarine organisms are capable of tolerating low DO conditions for such short periods. Reduced DO concentrations would be expected to be localized and short term, with minimal impacts (U.S. Navy 1990).

Nutrient enrichment can increase turbidity in the water column by enhancing the growth of phytoplankton. If this occurs, it is typically a transient phenomenon with minimal local impact. In the Bay area, nutrients would be flushed out of the dredging area by tidal currents. Effects of nutrients on phytoplankton in the Bay would generally not be detectable (U.S. Navy 1990).

Depending on the location of the dredging, deepening navigation channels can increase saltwater intrusion into the Delta (since saline water is heavier than freshwater), potentially impacting freshwater supplies and fisheries. Dredging can also increase saltwater intrusion into groundwater aquifers (e.g., the Merritt Sand/Posey formation aquifer in the Oakland Harbor area), with consequent degradation of groundwater quality in shallow aquifers (U.S. Navy 1990).

Potential Impacts on Sediments

The impacts on sediments at the dredging site may include increased post-dredging sedimentation in the newly deepened areas for new work projects, local changes in air-water chemistry, and possible slumping of materials from the sides of the dredging areas.

Potential Resuspension of Contaminants

Dredging will resuspend contaminants if contamination is present in the surface sediments. Metal and organic chemical contamination is widespread in San Francisco Bay sediments due to river run-off and municipal/ industrial discharges (see section 3.2.3.2). Contaminants of particular concern in various parts of the Bay include silver, copper, selenium, mercury, cadmium, polychlorinated biphenyls (PCBs), DDT and its metabolites, pesticides, polynuclear aromatic hydrocarbons (PAHs), and tributyltin.

Dredging of contaminated sediments does present the potential for release of contaminants to the water column, and for the uptake of contaminants by organisms contacting resuspended material. However, most contaminants are tightly bound in the sediments and are not easily released during short-term resuspension. Chemical reactions that occur during dredging may change the form of the contaminant and thus alter its bioavailability to organisms. These chemical reactions are determined by complex interactions of environmental factors, and may either

enhance or decrease bioavailability, particularly of metals.

Potential Impacts on Biological Resources

The impacts of dredging on biological resources can be short term or long term, direct or indirect. There can be short-term impacts from the dredging, and long-term impacts associated with habitat modification. Short-term impacts could include local changes in species abundance or community diversity during or immediately after dredging. Long-term impacts could include permanent species abundance or community diversity changes caused by changes in hydrodynamics or sediment type, or a decline or erratic trend beyond the normal range of variability in the years following new dredging (U.S. Navy 1990). Direct impacts would be directly attributable to the dredging activity, such as a direct loss of mudflat habitat or a temporary turbidity-induced reduction in productivity in an eelgrass bed immediately adjacent to a dredging site. Indirect effects on organisms include those effects which are not immediately measurable as a consequence of dredging operations. Such effects might, for example, involve population changes in one species that are caused by dredging's effects on its predators, prey, or competitors. Indirect effects may be manifested over extended periods of time and/or at some distance away from the dredging site. The differentiation between direct and indirect effects is not always clear.

Dredging involves the removal of substrate and benthic organisms at the dredging site, resulting in immediate localized effects on the bottom life. Besides the decimation of organisms at the dredging site, there is the removal of the existing natural or established community with widely varying survival of organisms during dredged material excavation. Aside from the initial physically disruptive effects, a long-term environmental concern is the recovery (repopulation) of bottom areas where dredging has occurred (Hirsch, DiSalvo, and Peddicord 1978). Dredging thus opens the area for recolonization on a new substrate that may resemble the original substrate or be completely different in physical characteristics. Recolonization may include the same organisms or opportunistic species that have environmental requirements that are flexible enough to allow them to occupy a disturbed site (Reilly et al. 1992).

Recolonization of the dredging site can begin quickly, although re-establishment of a more stable benthic community may take several months or years after the dredging operation has occurred (Oliver et al. 1977; Conner and Simon 1979). Oliver et al. (1977) found that most of the infauna were destroyed at the center of the dredging area. Communities inhabiting highly variable and easily disrupted environments, such as those found in shallow water, recovered more quickly from dredging operations than communities in less variable environments such as in deep or offshore waters. Seasonal changes in the environment were considered most important in shallower water where the organisms are more likely to be affected by the changing seasons (Reilly et al. 1992).

Oliver et al. (1977) noted two phases of succession after a disturbance. In the first phase, opportunistic species such as some polychaetes would move into a disturbed area. The second phase involved recruitment of organisms associated with undisturbed areas around the disturbed site. Recovery at the disturbed dredging site depends on the type of environment and the speed and success of adult migration or larval recruitment from adjacent undisturbed areas (Hirsch, Disalvo, and Peddicord 1978).

The effects of habitat loss or alteration at the dredge site may extend beyond the boundaries of the dredging operations. However, dredging-induced habitat alterations are minor compared to the large-scale disturbance of benthic habitat in San Francisco Bay from naturally occurring physical forces (Reilly et al. 1992). The result of these forces is a state of non-equilibrium in benthic species composition typical of shallow estuaries. Naturally occurring habitat disturbances arise from seasonal and storm-generated waves, and from seasonal fluctuations of riverine sediment transport into San Francisco Bay. Human influences on benthic habitat include not only dredging and disposal, but also waste discharges, sediment deposition from hydraulic mining, filling of Bay margins, fresh water diversions, and introduction of exotic species. When the disturbance ceases, recolonization of the benthic substrate occurs; re-establishment of a more or less stable benthic community can take several months or years (Reilly et al. 1992).

The suspension of sediments during dredging will generally result in localized, temporary increases in turbidity that are dispersed by currents or otherwise dissipate within a few days, depending on hydrodynamics and sediment characteristics (e.g., USACE and Port of Oakland 1998). Where dredging occurs in relatively polluted areas, contaminants in the sediments are likely to be dispersed into the water column, resulting in localized, temporary increases in contaminant concentrations that may affect fish and invertebrates.

Although the increases in turbidity are transient, they can have several types of longer-term consequences for sensitive biological resources. Increased turbidity can reduce the survival of herring eggs, which are attached to hard surfaces on Central Bay shorelines, potentially resulting in reduced recruitment and, ultimately, reduced abundance of this important resource species in the Bay. In certain locations, at critical times of year, increased turbidity can affect the survival of the larval or juvenile stages of sensitive fish species, as well as the feeding and migration of adults. Short-term impacts on critical foraging areas, such as eelgrass beds, during the nesting season of marine birds such as the endangered California least tern, can affect the birds' nesting success.

The effect of dredging on fish varies to some degree with the life stage of the fish. Early life stages of fish are more sensitive than adults. Adult fish would be motile enough to avoid the areas of activity; it is assumed that fish will leave the affected areas until dredging is done. Turbidity could reduce visibility, causing difficulty in locating prey. Suspended sediments can have other impacts, including abrasion of the body and clogging of the gills. Generally, bottom-dwelling fish species are most tolerant to suspended solids, and filter feeders are the most sensitive. In San Francisco Bay, dredging between December and February could disrupt the spawning of the Pacific herring and result in mortality to eggs. Depending on the location of dredging, such activity could affect the migration of steelhead and chinook salmon. Dredging in the Central Bay during summer can affect juvenile Dungeness crabs, for which the Central Bay provides an important nursery habitat. Larval and juvenile fishes and invertebrates are also vulnerable to entrainment in dredging equipment.

Waterbirds that feed or rest in the vicinity of the dredging activity may be disturbed and, as a result, move to areas where they incur higher energetic costs or experience greater risks.

Potential Impacts on Other Resource Areas

Emissions from dredging equipment in the Bay area typically causes temporary adverse impacts on air quality, depending on the size and location of the project.

Noise from the dredge can cause significant impacts on sensitive receptors located near the dredge area.

Dredging can impact submerged cultural resources (e.g., ship wrecks) if such resources are present in the dredge area. For the ports and major navigation channels in the Bay area, this is usually not an issue because the channels have been dredged previously.

In terms of socioeconomic impacts, dredging activities have a minor beneficial impact on employment, requiring a relatively small work force which can easily be met by the large population in the Bay area. Deeper navigation channels are critical to Bay area ports' ability to compete for vessel cargo with other U.S. west coast ports, so dredging has a regional beneficial economic impact on the Bay area. Dredging has a beneficial impact also on recreational and commercial activities in that dredging helps to maintain harbors and marinas, which support fishing, boating, and associated activities.

Dredging impacts on vessel transportation are typically minimal. The dredge represents an obstacle that other vessels have to maneuver around, but the location of the dredge is posted in the *Notice to Mariners* so it can be easily avoided.

Depending on the location, dredging can affect recreational fishing but such impacts are typically temporary and insignificant.

Dredging can impact submerged utilities but, with proper notice, these utilities can be relocated to avoid impacts.

3.1.1.4 Transportation of Dredged Material

Transportation methods generally used to move dredged material include the following: pipelines, barges or scows, hopper dredges, and sometimes trucks. Pipeline transport is the method most commonly associated with cutterhead, dustpan, and other hydraulic dredges. Dredged material may be directly transported by hydraulic dredges through pipelines for distances of up to several miles, depending on a number of conditions. Longer pipeline pumping distances are feasible with the addition of booster pumps, but the cost of transport greatly increases. Barges and scows, used in conjunction with mechanical dredges, have been one of the most widely used methods of transporting large quantities of dredged material over long distances. Hopper dredges are capable of transporting the material for long distances in a self-contained hopper. Hopper dredges normally discharge the material from the bottom of the vessel by opening the hopper doors; however, some hopper dredges are equipped to pump out the material from the hopper much like a hydraulic pipeline dredge. Truck transport is typically more expensive than barge

transport; it is generally only used for transport to upland sites not accessible by water. See the discussion of impacts associated with truck transportation of dredged material in section 4.4.5.3.

3.1.1.5 Material Placement or Disposal Operations

Selection of proper dredging and transport equipment and techniques must be compatible with disposal site and other management requirements. Three main alternatives are available:

- Open-water disposal;
- Confined disposal; and
- Beneficial reuse.

Each of these alternatives involves its own set of unique considerations, and selection of a management alternative should be based on environmental, technical, and economic considerations.

Description of Open-Water Disposal

Open water disposal is the placement of dredged material at designated sites in rivers, lakes, estuaries, or oceans via pipeline or release from hopper dredges or barges. Such disposal may also involve appropriate management actions or controls such as capping. The potential for environmental impacts is affected by the physical behavior of the open-water discharge. The physical behavior of the discharge depends on the type of dredging and disposal operation used, the nature of the material (its physical characteristics), and the hydrodynamics of the disposal site.

Dredged material can be placed in open-water sites using direct pipeline discharge, direct mechanical placement, or release from hopper dredges or scows. A conceptual illustration of open water disposal using the most common placement techniques in shown in Figure 3.1-2.

Pipeline dredges are commonly used for open water disposal adjacent to channels. Material from this dredging operation consists of a slurry with a solids concentration ranging from a few grams per liter to several hundred grams per liter. Depending on material characteristics, the slurry may contain clay bails, gravel, or coarse sand material. The coarse material quickly settles to the bottom. The mixture of dredging site water and finer particles has a higher density than the disposal site water and therefore can descend to the bottom forming a fluid mud mound. Continuing the discharge may cause the mound to spread. Some fine material is "stripped" during descent and is evident as a turbidity plume. Characteristics of the plume are determined by discharge rate, characteristics of the slurry (both water and solids), water depth, currents, meteorological conditions, salinity of receiving water, and discharge configuration.

The characteristics and operation of hopper dredges result in a mixture of water and solids stored in the hopper for transport to the disposal site. At the disposal site, hopper doors in the bottom of the ship's hull are opened, and the entire hopper contents are emptied in a manner of minutes; the dredge then returns to the dredging site to reload. This procedure produces a series of discrete discharges at intervals of perhaps one to several hours. Upon release from the hopper dredge at the disposal site, the dredged material falls through the water column as a well-defined jet of high-density fluid which may contain blocks of solid material. Ambient water is entrained during descent. After it hits bottom, most of the dredged material comes to rest. Some material enters the horizontally spreading bottom surge formed by the impact and is carried away from the impact point until the turbulence of the surge is sufficiently reduced to permit its deposition.

Bucket or clamshell dredges remove the sediment being dredged at nearly its *in situ* density and place it on a barge or scow for transport to the disposal area. Although several barges may be used so that the dredging is essentially continuous, disposal occurs as a series of discrete discharges. Barges are designed with bottom doors or with a split-hull, and the contents may be emptied within seconds, essentially as an instantaneous discharge. Often sediments dredged by clamshell remain in fairly large consolidated clumps and reach the bottom in this form. Whatever

Figure 3.1-2 Plume Shapes by Dredge Types

its form, the dredged material descends rapidly through the water column to the bottom, and only a small amount of the material remains suspended. Clamshell dredge operations may also be used for direct material placement adjacent to the area being dredged (i.e., when no transport is necessary). In these instances, the material also falls directly to the bottom as consolidated clumps.

Dredge hoppers and scows are commonly filled past the point at which water overflows, in order to increase the sediment load. The gain in hopper or scow load and the characteristics of the associated overflow depend on the characteristics of the material being dredged and the equipment being used. There is little debate that the load can be increased by overflow if the material dredged is coarse grained or firm clay balls, as commonly occurs with new work dredging. For fine-grained maintenance material, there is substantial disagreement as to whether a load gain can be achieved by overflow. Environmental considerations of overflow may be related to aesthetics; or potential effects of water-column turbidity, deposition of solids, or sediment-associated contaminants.

Open water disposal sites can be either predominantly non-dispersive or predominantly dispersive. At predominantly non-dispersive sites, most of the material is intended to remain on the bottom following placement and may be placed to form mounds. At predominantly dispersive sites, the material may be dispersed either during placement or eroded from the bottom over time and transported away from the disposal site by currents and/or wave action. However, both predominantly dispersive and predominantly non-dispersive sites can be managed in a number of ways to achieve environmental objectives or reduce potential operational conflicts.

Description of Confined Disposal

Confined disposal is placement of dredged material within diked nearshore or upland confined disposal facilities (CDFs) via pipeline or other means. CDFs may be constructed as upland sites, nearshore sites with one or more sides in water (sometimes called intertidal sites), or as an island containment area as shown in Figure 3.1-3. Confined Aquatic Disposal (CAD) facilities can also be constructed (see section 3.2.6).

The main objectives inherent in design and operation of CDFs are to provide for adequate storage capacity for meeting dredging requirements; to maximize efficiency in retaining solids; and to control the release of any contaminants present in the dredged material. Basic guidance for design, operation, and management of CDFs is found in EM 1110-2-5027 (USACE 1987b).

Hydraulic dredging adds several volumes of water for each volume of sediment removed, and this excess water is normally discharged as effluent from the CDF during the filling operation. The amount of water added depends on the design of the dredge, physical characteristics of the sediment, and operational factors such as pumping distance. When the dredged material is initially deposited in the CDF, it may occupy several times its original volume. The settling process is a function of time, but the sediment will eventually consolidate to its *in situ* volume or less if desiccation (drying) occurs. Adequate volume must be provided during the dredging operation to contain both the original volume of sediment to be dredged and any water added during dredging and placement.

Some CDFs are filled by mechanically rehandling dredged material from barges filled by mechanical dredges. Material placed in the CDF in this manner is at or near its *in situ* water content. If such sites are constructed in water (nearshore CDFs), the effluent volume may be limited to the water displaced by the dredged material, and the settling behavior of the material is not important.

In most cases, CDFs must be used over a period of many years, storing material dredged periodically over the design life. The long-term storage capacity of these CDFs is therefore a major factor in design and management. Once water is drained from the CDF following active disposal operations, natural drying forces begin to dewater the dredged material adding additional storage capacity. The gains in storage capacity are therefore influenced by consolidation and drying processes and the techniques used to manage the site during and following active disposal operations.

Categories of Beneficial Reuse

Beneficial reuse includes a wide variety of options that utilize the dredged material for some productive purpose. Dredged material is a manageable, valuable soil resource, with beneficial uses of such importance that they should be incorporated into project plans and goals at the project's inception to the maximum extent possible.

Figure 3.1-3 Types of Confined Disposal Facilities

Ten broad categories of beneficial uses have been identified nationwide, based on the functional use of the dredged material or site. They include the following:

- Habitat restoration/enhancement (wetland, upland, island, and aquatic sites including use by fish, wildlife, and waterfowl and other birds);
- Beach nourishment;
- Aquaculture;
- Parks and recreation (commercial and non-commercial);
- Agriculture, forestry, and horticulture;
- Strip mine reclamation and landfill cover for solid waste management;
- Shoreline stabilization and erosion control (fills, artificial reefs, submerged berms, etc);
- Construction and industrial use (including port development, airports, urban, and residential);
- Material transfer (for fill, dikes, levees, parking lots, and roads); and
- Multiple purposes (i.e., combinations of the above).

Detailed guidelines for various beneficial use applications are given in EM 1110-2-5026 (USACE 1987a).

3.1.1.6 Feasible Reuse Options in the San Francisco Bay Area

In the Bay area, several of the general reuse options listed above have been or could feasibly be done with relatively large quantities of dredged material (other options may be feasible on a project-by-project basis, as well). In particular, dredged material could be used beneficially for new construction, levee maintenance, landfill cover, and marsh restoration. Additionally, at upland sites, facilities could be established to dry dredged material for subsequent off-site use (such facilities are referred to as "rehandling facilities"), or to confine material permanently (Confined Disposal Facilities, or CDFs). The most feasible options are described in more detail in the following paragraphs.

Wetland Restoration

Agricultural practices over many years have caused lands along the Bay to subside so that current land elevations are many feet below sea level, far below the elevation necessary to support most marsh vegetation. The perimeter dikes of these sites could be breached to introduce tidal flooding. In this case, natural siltation may be expected to result in bottom elevations suitable for wetland vegetation over a relatively long time, depending on initial site elevations and the siltation characteristics of the site. Further, until enough sediments accumulate to raise the bottom level to provide the necessary periods of inundation and exposure for marsh plants, this could result in a tidal lake at such lands.

Placing dredged materials on subsided, diked former baylands can accelerate the tidal marsh restoration process by raising ground level to the appropriate height. Before placing material, the site needs to be prepared. The construction phase typically involves constructing perimeter levees and interior dikes or peninsulas, as well as installing water control systems and an area to off-load dredged material to the site.

In the San Francisco estuary, tidal marsh has been established at three former upland disposal sites: Muzzi Marsh in Corte Madera, Marin County; Faber Tract in Palo Alto, Santa Clara County; and Salt Pond No. 3 in Fremont, Alameda County. Dredged material has also been used successfully to enhance natural resource values and management capability at managed wetlands in the Suisun Marsh. Currently, dredged material generated from improving the Oakland Harbor is being used to restore a diked historic wetland at the Sonoma Baylands site (see Appendix K.2).

Dredged materials could also be used create higher areas within tidal wetlands projects that would be inundated only by the highest tides (spring tides in the winter and storm-related extreme high tides) and would pond water from infrequent tidal inundation and rainfall. To do so would involve filling subsided land at the upper end of tidal marshes above mean higher high water (MHHW) and including depressions for ponding over the area. Additionally, dredged materials could be used to construct berms to separate tidal and seasonal wetlands on a site (without raising the elevation of the seasonal wetlands) and to create areas for ponding and drainage control on sites not associated with tidal wetland creation projects.

Potential dredged material reuse volumes (capacities) developed by BCDC for the LTMS indicate that up to 103 mcy of dredged material could be accommodated at wetland restoration sites over the 50-year planning period (BCDC 1995). Beginning in the year 2000, with the commencement of a wetland restoration project at the former Hamilton Army Airfield and adjacent properties, approximately 2.0 mcy of dredged material would be used annually to restore wetland habitat in the Bay area. Shortly thereafter, approximately 4.0 mcy of dredged material would be used annually both as part of the Hamilton restoration project and other proposed

restoration projects using dredged material such as Montezuma Wetlands. By the time the placement of dredged material is completed at these sites, it is anticipated that other sites using dredged material will be implemented and receive approximately 1.0 mcy of material per year.

Levee Repair and Rehabilitation

Vast tracts of land in the San Francisco Bay area (Bay) and the Sacramento-San Joaquin Delta (Delta) are reclaimed land that is protected from inundation by levees. Dredged materials have often been used to construct and repair Bay and Delta levees. Typically, a dragline or clamshell has been used to excavate material from either side of the proposed levee, piling the material along the proposed alignment. When sufficiently dry, the material has been graded to form the levee. Because of their similar origins, dredged materials often have similar properties as existing levee soils, improving levee stability and structural strength, and thus can be used for levee repair and maintenance.

In 1994, a demonstration project was implemented using 75,000 cy of material from the Suisun Bay and New York Slough federal channels to restore levees at Jersey Island (Contra Costa County) in the Sacramento-San Joaquin Delta. In light of existing constraints concerning the use of dredged material for Delta levee maintenance projects, including water quality issues and restricted barge access, it is estimated that approximately 26 mcy of dredged material could be used in the Delta over the next 50 years in the following manner: approximately 250,000 cy of material could be used per year during the initial years (until the year 2000); and up to 1.0 mcy of material could be used annually in subsequent years. (It should be noted that this estimate is significantly lower than the Department of Water Resources' projection, which indicates that a total of 200 mcy of dredged material could be accommodated in the Delta for levee maintenance.)

Landfill Reuse

The clays and fine silts that comprise most dredged materials from the Bay are often suitable at landfill sites (once dried) for use as cover, on-site construction, capping, or lining material. Landfills possess several characteristics which are ideal for the reuse of dredged material. Daily operations and closure procedures require substantial amounts of cover and capping material, and therefore there is the potential for utilizing a significant portion of material dredged annually from the San Francisco Bay. Because landfills are designed to contain pollutants and manage runoff, they have the added benefit of being able to accept some contaminated materials infeasible for unconfined aquatic disposal. And while liability is a potential concern for disposal of material at any site, landfills provide greater protection against liability, since thorough waste testing and gate controls are required and enforced. Additionally, in most cases dredged material will replace the use of clean soil excavated and transported from elsewhere, or other non-waste sources. Finally, because landfills are typically highly disturbed sites with limited natural resource values, the use of dredged materials at landfills is likely to impact few existing natural resources.

The Redwood Landfill in Marin County and the Tri-Cities Landfill in Alameda County are two facilities that have incorporated the use of dredged material in their closure plans. Tri-Cities Landfill is planning to use 180,000 cy of dredged material from the San Leandro Marina as capping material for eventual closure of the landfill. The material is currently stockpiled at Roberts Landing adjacent to the Marina. In addition, Redwood Landfill has accepted approximately 500,000 cy of dredged material from the Petaluma River, Gallinas Creek, and Port Sonoma-Marin. The material has been used as daily cover, for on-site construction, and as liner material. Redwood Landfill has also proposed using dredged material to construct a 2-foot liner for a sludge processing area and for levee construction and repair.

Rehandling and Confined Disposal Facilities

Rehandling facilities are mid-shipment points for dredged material that cannot be hauled directly to the site where it will be ultimately used, such as landfills. They are also locations where dredged materials can be dried or treated to remove or reduce salinity or contaminants. Typically, rehandling facilities accept relatively small volumes of material originating from specific dredging projects. In the Bay area, rehandling facilities are located at Port Sonoma-Marin, near the mouth of the Petaluma River; in the City of Petaluma, Sonoma County; and in the City of San Leandro, Alameda County.

In some cases, CDFs are needed for contaminated dredged material that cannot be reused and thus requires permanent confinement. Such facilities can be engineered similar to a rehandling facility. However, since multi-user CDFs for contaminated dredged material would have to be designed for the worst-case material that could be permitted for disposal in them,

the need for cell liners, leachate collection systems, or other contaminant control measures would also need to be considered. The potential dredged material reuse volumes developed by BCDC for the LTMS indicate that up to 298 mcy of dredged material could be processed at rehandling facilities in the Bay area over the 50-year planning period (BCDC 1995). Over the next few years, approximately 250,000 cy of dredged material is expected to be processed annually at existing rehandling facilities in the Bay area. Subsequently, existing capacity will increase over time and the volume of material processed will gradually increase: in the year 2000, approximately 500,000 cy of material will be processed annually; in 2001, approximately 1.25 mcy of material will be processed annually; and in 2005, approximately 1.75 mcy of material will be processed annually.

Construction Purposes

Naturally occurring sand deposits in the Bay have been an important source of construction material for many years, unlike Bay muds which are generally unsuitable for use as engineered fill because of their lack of structural strength. Rehandling processes do produce material from bay muds that are useful in construction activities. A cost-effective approach to rehandling bay muds, however, does not yet exist. Typically, new construction associated with water-related industries and ports involves dredging and Bay fill. In these instances, sands dredged to create new berths or to deepen navigation channels can be used to provide an engineered base for marine terminals or construction yards. The volume of material available for construction will be primarily dictated by capacity at rehandling facilities (as noted above) and whether the dried material meets the specific physical and chemical requirements of the construction project.

3.1.2 Dredging Volumes — LTMS Planning Estimates

In 1990, the COE evaluated past dredging trends and what was known about major new dredging projects in the LTMS Phase I Report (LTMS 1990b). Based on that review, it was assumed that an average of 8 mcy of sediments would be dredged each year. During the 50-year LTMS planning period (1995 to 2045), this would mean that 400 mcy of dredged material would be generated and need to be disposed. These figures — 8 mcy per year and an overall total of 400 mcy — were the initial basis of the LTMS planning effort.

Since the time of the original COE estimate, the overall dredging situation has changed significantly. In particular, several major military facilities — some of which have been associated with some of the largest dredging projects in the region — have been slated for closure. Interested parties to the LTMS planning effort requested that the LTMS agencies revisit the SFEP dredging estimates, taking into account an assumed reduced need for future dredging once the military base closures are complete. The LTMS re-evaluation of long term dredging needs (*Analysis of San Francisco Regional Dredging Quantity Estimates; Dredging Project Profiles; and Placement Site Profiles* [LTMS 1995a]) is presented in Appendix E. Appendix E also includes descriptions of each of the major dredging projects in the region. The following discussion summarizes the approach used and the resulting revised LTMS planning estimate of dredging volumes over the next 50 years. See Appendix E for details of this analysis and the references used.

3.1.2.1 Method for Re-Evaluating Dredging Volumes

To evaluate long-term dredging needs, historic dredging quantities (since 1955) were first determined, to the extent possible, based on the available dredging records of the COE, the *Sediment Budget Study for San Francisco Bay* (LTMS 1992e), and the August 1993 *Dredging and Disposal Road Map* (BCDC and USACE 1993). The records were then screened to account for technical, surveying, reporting, and regulatory differences over the years. This evaluation revealed that there has been a long-term average dredging quantity of approximately 6.84 mcy per year in the Bay area; maintenance dredging accounted for approximately 6.45 to 6.69 mcy per year of this total.

The historic dredging figures were then adjusted to account for projects associated with military bases that have closed or are slated for closure. These facilities include Mare Island Naval Ship Yard; Treasure Island Naval Station; Hunters Point Naval Shipyard; Moffett Field Naval Air Station; and Alameda Naval Air Station. Since the potential for a long-term reduction in dredging associated with these facilities is highly dependent on future uses of the facilities, three scenarios were developed. For the Low-Range Estimate of total long-term dredging, the entire average dredging volume associated with the facilities was subtracted from the historic totals. The Mid-Range Estimate of total long-term dredging subtracted 50 percent of this volume from the historic totals, reflecting continued but shallower-draft navigation use of the associated channels. For the High-Range Estimate, it was assumed that the navigation channels associated with these facilities would continue to be

dredged as they have in the past; no reduction in overall dredging was therefore made for the High-Range Estimate. An exception was the Mare Island Naval Shipyard. It is likely that closure of this facility will not eliminate the need for some level of maintenance dredging, no matter what land use the facility supports in the future. Therefore, the Low-Range Estimate assumes only a 50 percent reduction in dredging for this site, while the Mid-Range Estimate assumes a 25 percent reduction. As for the other military facilities, the High-Range Estimate assumes the entire historic volume would continue to be dredged.

Finally, an estimate of potential new work dredging projects was developed, to add to the adjusted historic volumes. Planning-level estimates of dredging volumes for authorized or proposed new work projects were first summed. These new work projects include: the Port of Oakland Phase II (-42-foot) Deepening Project; the Phase III John F. Baldwin Ship Channel Project; the Port of Richmond -38-Foot Deepening Project; the San Francisco Harbor Deepening Project; and the Port of Stockton (Avalon to New York Slough) Project.

Together, these projects would generate an estimated 24.2 mcy of dredged material over the next 15 to 20 years. Three scenarios were again developed for these new work projects to predict Low-, Mid-, and High-Range Estimates of long-term new work dredging volumes. For the Low-Range Estimate, it was assumed that only 50 percent of the volume associated with the proposed new work projects would actually be dredged. The Mid-Range Estimate assumed that the entire 24.2 mcy would be dredged. The High-Range Estimate assumed that additional, currently unknown new work projects would be proposed and constructed over the 50-year LTMS planning period, generating an additional volume of dredged material equivalent to the currently proposed projects, for a total of 48.4 mcy of dredged material.

3.1.2.2 Revised Dredging Volume Estimate for the 50-Year LTMS Planning Period

The results of the LTMS re-evaluation of long-term dredging volumes are presented in Table 3.1-1. This table shows that the SFEP estimate of 400 mcy of dredged material over the next 50 years indeed appears to be too high. Instead of an average of 8 mcy of dredging and disposal per year, the average dredging need is between a Low-Range of 3.47 mcy to a High-Range of 5.93 mcy. This equates to a 50-year total of between 173.5 to 296.5 mcy of dredged material being generated by all currently foreseeable maintenance and new work projects.

> The subject of dredged material disposal in aquatic systems is not a simple exercise in civil engineering. It involves detailed consideration of the physics and chemistry of sediments; the physics, chemistry, and toxicology of contaminants that may be associated with sediments; and the interaction of sediments, dredged material, contaminants, and estuarine hydrology with existing populations of shellfish, fish, and wildlife. The complexity of this subject, and the sparse information available for drawing specific conclusions, makes it necessary that existing estuarine resources be projected by making reasonable decisions based upon available data, while new knowledge … is accumulated (SFEP 1990).

Table 3.1-1. Revised Dredging Volume Estimate for San Francisco Bay (1995-2045)

Quantity Type	*Low Range Estimate (cubic yards/year)*	*Mid Range Estimate (cubic yards/year)*	*High Range Estimate (cubic yards/year)*
Historic maintenance and new work (1)	6,840,213	6,840,213	6,840,213
Removal of historic new work (1)	-393,062 (-100 percent of all new work)	-284,116 (-50 percent of selected new work)	-155,170 (-0 percent of selected new work)
Estimated range of historic maintenance dredging	6,447,151	6,556,097	6,685,043
Removal of dedicated disposal sites and base closures (2)	-3,223,662	-2,478,111	-1,720,195
Projected maintenance dredging	3,223,489	4,077,986	4,964,848
Addition of projected new work dredging (3)	242,000 (+50 percent)	484,000 (+100 percent)	968,000 (+200 percent)
Total	**3,465,489**	**4,561,986**	**5,932,848**
Rounded total	**3,470,000**	**4,560,000**	**5,930,000**
50-Year Projected Total Dredge Material Volume	**173,500,000**	**228,000,000**	**296,500,000**

Notes: (1) For projects with separable new work quantities, the entire quantity was deleted in all estimate ranges. For records without separable quantities, 100 percent, 50 percent, and 0 percent of the entire annual reported volume was removed for the low, mid, and high range estimates, respectively (see Table 3 in Appendix E).

(2) For projects with dedicated disposal sites, and military base closures, 100 percent, 50 percent, and 0 percent of the quantities were removed with the exception of the San Francisco Bar, which was entirely removed (ocean disposal only), and the Mare Island Straits, which had 50 percent, 25 percent, and 0 percent removed since it is known that this dredging will not cease entirely with the closure of Mare Island Naval Shipyard (see Table 3 in Appendix E).

(3) See Table 4 in Appendix E, and subsequent paragraphs.

3.2 BAY AREA SEDIMENT AND DREDGED MATERIAL CHARACTERISTICS

Materials beneath San Francisco Bay that are typically encountered during dredging projects consist of thick, unconsolidated sediments of both marine and terrestrial origin, deposited from the Pleistocene to the present day. These sediments may become contaminated by pollutants from a variety of sources. In some cases, sediment contamination may be serious enough that it poses a direct risk to the environment or to human health, such that the sediment must be removed from the Bay regardless of whether any person or port has independent plans to dredge it for navigation purposes. The state of California and the U.S. EPA have established remedial action programs for addressing such highly contaminated sediments. Discussion of the need for remediation of highly contaminated sediments is beyond the scope of this document. Instead, this EIS/EIR addresses the management of "dredged material." For the purposes of this document, dredged material is sediment that is removed for purposes other than remediation: for example, the removal of sediment for the construction or maintenance of commercial or recreational navigation channels, ship berths, marinas, or other waterways. Thus all "dredged material" consists of sediments, but not all sediments in the Bay/Delta estuary are "dredged material."

Dredged material is managed under different regulatory authorities depending on the use to which it is put, or the environment in which it is placed. For example, dredged material placed in waters of the United States

or in the ocean is regulated under the federal CWA or the MPRSA, respectively. Dredged material placed as fill in an upland location is typically a solid waste regulated under a different set of state and federal statutes (see section 4.8.1.3). Regardless of which agency or law regulates dredged material in a particular instance, management concerns vary with the placement environment (e.g., dispersive versus non-dispersive aquatic disposal sites, aquatic versus upland disposal sites, construction fill versus habitat creation uses, etc.).

The following sections provide a background on dredged material characteristics that are key to determining appropriate management techniques for aquatic or upland disposal or beneficial reuse in the San Francisco Bay area.

Important physical characteristics are addressed first (section 3.2.1), followed by a discussion of the movement and fate of sediments within the Estuary (section 3.2.2). General background on contaminants in dredged material is then presented (sections 3.2.3 and 3.2.4). More specific information is provided on contamination and toxicity, and its evaluation in Estuary sediments (section 3.2.5). Management options for contaminated dredged material (section 3.2.6) concludes this discussion.

3.2.1 Physical Characteristics

The trough-like depression that underlies San Francisco Bay is formed by Franciscan sandstone and shale bedrock (see section 4.2.2). This trough has been nearly filled with sediments, some of which has come from erosion of surrounding hills and some of which consists of later marine deposits. For example, the marine clay-silt deposit termed “old Bay mud” is

Table 3.2-1. Concentrations of Heavy Metals and Organic Compounds in Old Bay Mud and Merritt Sand Deposits

Sediment Chemistry	*Merritt Formation Sediment (a)*	*Old Bay Mud Sediment (b)*
Silver (mg/kg)	0.023-1.08	0.11
Arsenic (mg/kg)	2.93-12.60	3.28
Cadmium (mg/kg)	0.02-0.18	0.56
Chromium (mg/kg)	164-823	142
Copper (mg/kg)	8.9-43.8	27.4
Mercury (mg/kg)	0.0003-0.088	0.044
Nickel (mg/kg)	41.7-117.1	62.7
Lead (mg/kg)	3.5-10.4	10.6
Selenium (mg/kg)	0.07-0.42	0.17
Zinc (mg/kg)	33.7-100.5	68.3
Total PAH (µg/kg)	0.5-217	57
Tributyltin (µg/kg)	0.6-3.2	0.48 U
PCB (µg/kg)	2.3-4.0	20 U
Total DDT (µg/kg)	0.04-6.22	0.22 U

Notes: All values expressed in dry weight.

a. Ranges represent 13 stations with Merritt sand from the Port of Oakland Deepening Project (*Final Supplemental EIR/EIS Oakland Harbor Deep-Draft Navigation Improvements* Appendix A through E and G through L, June 1994).

b. Old Bay Mud composite is comprised of OBM sediment from four stations in the Richmond Harbor turning basin (*Ecological Evaluation of Proposed Dredged Material from Richmond Harbor Deepening Project and the Intensive Study of the Turning Basin*, June 1995, PNL-10627).

U = Undetected at or above detection limit.

present throughout most of the Bay, several feet beneath the soft, more recently deposited muds. An ancient fine-grained sand deposit known as "Merritt Sand" occurs in the vicinity of Oakland and Alameda, in places relatively close to the sediment surface. Also, natural peat deposits can be found underlying more recent Bay sediments in some areas of the North Bay and Delta. The thickness of the various historic sediment formations varies throughout the Bay/Delta estuary, but they can be several hundred feet thick overall. Figure 3.2-1 shows the general stratigraphy of sediment deposits within San Francisco Bay.

Figure 3.2-1 Stratigraphy of Sediment Deposits in San Francisco Bay

Whether of terrestrial or marine origin, the older deposits that pre-date European settlement in California generally are very hard-packed, low in moisture content, low in organic carbon (except for peat deposits), and have low concentrations of chemicals such as heavy metals and organic compounds. The chemical levels that are measurable in these historic deposits represent natural "background" levels for the sediment type. Table 3.2-1 shows typical levels of heavy metals and organic compounds measured in old Bay mud and Merritt Sand deposits. These deposits are not typically

dredged during maintenance dredging, but are often encountered during new work dredging (dredging of new navigation channels, or channel deepening projects).

The upper several feet of the sediment profile in most locations consists of more recently deposited marine and riverine sediments. The SFEP (1990) presented the following description of the classification and distribution of surficial (geologically recent) sediment deposits in the Estuary:

Sediments in the Estuary fall into three categories: sandy bottoms in the channels; shell debris over a wide expanse of the South Bay (derived from remnants of oyster beds (Wright and Phillips 1988); and soft deposits (known as "Bay Mud") underlying the vast expanses of shallow water. . . . Regions of the Estuary where currents are strong, including the deep channels of the Bay and the central channels of the major rivers in the Delta, generally have coarser sediments (i.e., fine sand, sand, or gravel). Areas where current velocities are lower, such as the shallow fringes of each sub-embayment of San Francisco Bay . . . are covered with Bay Mud (USACE 1976a). Bay Mud is comprised of silt and clay particles deposited as a result of flocculation, or "salting out," a process in which particulate matter in fresh water aggregates when mixed with more saline waters. The settling velocity of the aggregates is much greater than that of the original clay or silt particles, increasing particle deposition.

The surface Bay muds ("young Bay mud") and recent sand deposits tend to be much less densely packed, high in moisture content, and higher in organic carbon than the underlying ancient sediment formations. Figure 3.2-2 shows the generalized distribution of these sediment types in the San Francisco Bay/Delta estuary.

Physical differences between sediment types are important considerations for appropriate dredged material management. First, the deposit type and location in large part determine whether there is a likelihood that the sediments may have been exposed for a given sediment volume, and therefore greater concentrations of contaminants can potentially adsorb to the surface of silt particles. Silt particles are also readily resuspended and redistributed by even fairly low energy currents, and ultimately settle in quieter environments where pollutants and organic matter may also tend to accumulate. *Clay* (grain size less than 0.004 millimeters, or phi size greater than 8) has an even higher surface area for contaminant adsorption. Clay particles also tend to be charged, facilitating bonding of additional contaminants to their surfaces. However, their charged nature also gives them a propensity to stick together in clumps. This means that during aquatic disposal, clays tend to produce less pronounced water column plumes; this also makes it more difficult for currents to resuspend and redistribute clay from the bottom after disposal has ceased, particularly if the clay deposit was mechanically (clamshell) dredged. Third, factors such as the concentration of organic carbon and acid volatile sulfides (AVS) affect the degree to which contaminants may be associated with sediments. Organic carbon can readily absorb a variety of contaminants, including many that would not otherwise have a high affinity to attach to the surface of sediment particles. Surface sediments, particularly the finer silts and clays, can accumulate organic carbon from a variety of sources including the water column and organisms living within the sediments themselves. Whatever the source, the carbon content is generally higher in finer-grained sediments found in depositional areas (including portions of some navigation channels), where both organics and pollutants tend to accumulate. The concentration of AVS in sediments is defined as the concentration of solid phase sulfide compounds associated with metal sulfides (primarily iron and manganese monosulfides). In marine and freshwater sediments, sulfides of divalent metals form very insoluble compounds. It has been hypothesized that the quantity of AVS represents a "reactive pool" of sulfides that are able to bind and reduce the bioavailability and toxicity of the metals in sediments (DiToro et al. 1990).

Finally, the grain size class (sands, silts, clays), and the degree to which the sediment type is hard-packed, affects the degree to which the dredged material will tend to disperse or stay clumped during and after disposal. In addition, different sediment types often call for different dredging methods. For example, hydraulic suction dredging can be used with soft, unconsolidated young Bay muds, but mechanical methods such as clamshell dredging, at times even preceded by breaking up the deposit with special equipment before dredging, may be required in old Bay mud or other hard-packed formations. The dredging method also can affect dispersion or clumping during and after disposal at an open water site, as well as the area needed for upland disposal (see section 3.1).

Figure

3.2-2 General Distribution of Surface Sediment Types in the San Francisco Bay/Delta Estuary

The sediments of the Bay/Delta are dynamic, with erosion or deposition of material constantly occurring in response to complex patterns of currents and waves created by river flows, tides, and winds. The aquatic disposal of dredged sediment thus adds suspended material to a constantly changing environment, and determining the ultimate fate of disposed dredged material is a challenging task (SFEP 1990).

The majority of dredging in the Estuary is maintenance dredging of relatively soft, unconsolidated silts and clays that accumulate in existing navigation channels. Except in certain high energy areas, this material is typically comprised of 80 to 90 percent silt and clay size particles.

3.2.2 Movement and Fate of Sediments in the Estuary System

The primary source of new sediment into the Estuary system is the Sacramento River, which flows through Carquinez Strait into the northeastern end of San Pablo Bay. Other important, but much smaller sources are also in the north Bay, including the Napa, Sonoma, and Petaluma rivers. A variety of smaller streams and other drainages (including storm drains and flood control channels) can be locally important for adding new sediment to the system. Overall, these sources provide an estimated 8 mcy per year of new sediment to the Bay/Delta system (LTMS 1992e; USACE 1965). However, existing deposits of typical fine-grained surface sediments in the extensive shallow areas of the Estuary are subject to hydraulic movement (resuspension) by riverine, tidal, and wind-driven currents. Therefore, resuspended existing sediments are estimated to be 100 mcy (Krone 1974) to 286 mcy (SFEP 1992b) annually, or perhaps 10 to 30 times greater than from all the "new" sediment sources combined. Therefore, resuspended sediments account for the vast majority of suspended particulate matter and turbidity throughout the Estuary. Figure 3.2-3 is a conceptual illustration of these overall sediment movements.

SFEP (1990) included the following basic description of the dynamic environment experienced by surface sediments in the Estuary:

> With the exception of portions of Central Bay nearest the Golden Gate, the San Francisco Estuary is very shallow, with wide intertidal and subtidal regions cut by narrow, mid-Bay channels (Nichols and Thompson 1985) . . . . Greater than 40 percent of the Estuary is less than 2 m deep, and over 70 percent is less than 5 m deep (Nichols et al. 1986; Wright and Phillips 1988). The sediments of San Francisco Bay change on a time scale of days to months. The dynamic nature of the sediment compartment of the Estuary was demonstrated by the sediment survey of SAIC (1987). Most of the site studied by these investigators showed evidence of recent sediment erosion, redistribution, or deposition. On a short-term basis, Nichols and Thompson (1985) noted that sand waves standing from 20 cm to 8 m in height move with the ebb and flow of tide, resulting in a continual sediment turnover to a depth of about 40 cm every few days. On a time scale of weeks, the intertidal mud-flat environment of the Estuary may show rapid changes in elevation (Luoma and Bryan 1978; Nichols and Thompson 1985), as well as changes in sediment grain size.

More recently, in a study prepared for the LTMS (LTMS 1992e) compared the net differences between high-resolution bathymetric surveys of the Estuary taken 35 years apart. This comparison identified large-scale areas of longer-term net deposition and erosion throughout the Estuary. Figures 3.2-4 through 3.2-18 show the results of this comparison. As is apparent from these plates, deposition and erosion patterns throughout the Estuary are extremely complex and heterogeneous. The four existing disposal sites within the Estuary are all considered to be in erosional locations. The Alcatraz disposal site, in particular, is managed to maximize the erosion of dredged material disposed there in order to avoid continued mounding, which can pose a hazard to deep-draft vessels that must pass nearby. The other existing disposal sites are more fully erosional at the volumes of material disposed at them, and they have not experienced the kind of serious mounding that has occurred at the Alcatraz site.

Although the dynamic nature of the Estuary is generally known, and more information is being collected continuously, there is limited ability today to accurately predict the specific movement and ultimate fate of sediment particles from any particular source (such as dredged material disposal sites) in the Bay. Nevertheless, we have some basic information on general patterns of sediment movement. Turbidity in the central Bay is naturally less than in the south Bay or San Pablo Bay. For example, turbidity in the central Bay is naturally less than in the south Bay or San Pablo Bay. Similarly, sediment transport in the Estuary exhibits definite seasonal patterns. During the winter when freshwater flow and corresponding

Figure 3.2-3 Conceptual Illustration of Sediment Movement in the San Francisco Bay System

Figure 3.2-4 Index Map for Figures 3.2-5 through 3.2-18

Figure 3.2-5 Net Bathymetric Changes in San Francisco Bay from 1955 to 1990 — South Bay, Sections A1 & A2 (Plate 1)

Figure 3.2-6 Net Bathymetric Changes in San Francisco Bay from 1955 to 1990 — South Bay, Section B (Plate 2)

Figure 3.2-7 Net Bathymetric Changes in San Francisco Bay from 1955 to 1990 — South Bay, Section C (Plate 3)

Figure 3.2-8 Net Bathymetric Changes in San Francisco Bay from 1955 to 1990 — South Bay, Sections D & F (Plate 4)

Figure 3.2-9 Net Bathymetric Changes in San Francisco Bay from 1955 to 1990 — South Bay, Section E (Plate 5)

Figure 3.2-10 Net Bathymetric Changes in San Francisco Bay from 1955 to 1990 — North Bay, Section A (Plate 6)

Figure 3.2-11 Net Bathymetric Changes in San Francisco Bay from 1955 to 1990 — North Bay, Section B (Plate 7)

Figure 3.2-12 Net Bathymetric Changes in San Francisco Bay from 1955 to 1990 — North Bay, Section C (Plate 8)

Figure 3.2-13 Net Bathymetric Changes in San Francisco Bay from 1955 to 1990 — North Bay, Section D (Plate 9)

Figure 3.2-14 Net Bathymetric Changes in San Francisco Bay from 1955 to 1990 — San Pablo Bay, Section A (Plate 10)

Figure 3.2-15 Net Bathymetric Changes in San Francisco Bay from 1955 to 1990 — San Pablo Bay, Section B (Plate 11)

Figure 3.2-16 Net Bathymetric Changes in San Francisco Bay from 1955 to 1990 — San Pablo Bay, Section C (Plate 12)

Figure 3.2-17 Net Bathymetric Changes in San Francisco Bay from 1955 to 1990 — Suisun Bay, Section A (Plate 13)

Figure 3.2-18 Net Bathymetric Changes in San Francisco Bay from 1955 to 1990 — Suisun Bay, Sections B & C (Plate 14)

"new" sediment loads are high and winds are generally weak, sediments tends to be deposited on the mudflats of northern San Pablo Bay and other quiescent locations. In the summer when river flows and "new" sediment loads decrease dramatically, strong, frequent westerly winds over the shallow mudflats resuspend the sediments and, in conjunction with tidal currents, transport them throughout the system. In addition, although most new sediment input occurs in San Pablo Bay, and although there is less overall water circulation in south San Francisco Bay, the information available today supports the presumption that sediments from any of the major sub-basins of the Estuary can be resuspended, and soon spread widely throughout the Estuary. Some sediments leave the Estuary system by being transported out the Golden Gate; however, the quantity leaving the system during a typical year is thought to be relatively small (on average, less than the input of new sediment from rivers and other sources) compared to the total quantity cycling within the Estuary (see Figure 3.2-3).

Preliminary mathematical modeling of dredged material transport and initial deposition following disposal at several locations throughout the Estuary was conducted for the LTMS by the COE Waterways Experiment Station (WES) (Letter et al. 1994). The results of this modeling remain preliminary, and substantial model development is still needed before any such results can be used with confidence. WES modeling indicates that dredged material initially discharged at existing in-Bay disposal sites may quickly find its way into virtually every major sub-basin of the Estuary. For example, figures 3.2-19 through 3.2-21 show modeled initial deposition patterns following disposal at the Alcatraz, San Pablo Bay, and Carquinez Strait disposal sites, respectively. These modeling results are generally consistent with the LTMS (1992e) figures, that are based on empirical information, in terms of the heterogeneity of deposition and erosion patterns throughout the Estuary. However, the WES model output shows only predicted *initial* deposition locations; subsequent resuspension and further transport of the dredged material particles would be expected from any initial deposition sites that exhibit erosional characteristics at times.

Because the majority of fine sediment particles are likely to settle and resuspend a number of times in the Estuary, at least a small percentage of the sediment accumulating in navigation channels is likely to include previously dredged material that was discharged at an erosional in-Bay site, has resettled, and now has to be re-dredged. For example, tracer studies in the mid-1970s confirmed that as much as 10 percent of the sediments accumulating in the Mare Island Strait were in fact dredged material recirculated from the Carquinez disposal site (USACE 1976b). System-wide, however, the overall amount of previously dredged material that makes its way back into navigation channels to be re-dredged in this way is almost certainly much smaller.[1] The continual resuspension of sediments within the Estuary system also means it can be expected that sediments accumulating in navigation channels may have been exposed to pollutant sources in several locations, far removed from the dredging site. This helps to explain why chemical testing of sediments from some regularly dredged channels can show a fairly high degree of variability from year to year, even when there have been no nearby discharges or spills. It also helps to explain why almost all maintenance dredging projects from throughout the Bay show at least some degree of elevated (above ambient or "background") concentrations of trace contaminants (see section 3.2.3.3). By the same token, however, particles carrying pollutants also may get diluted with particles from other areas that settle in the same location, that have lower concentrations of associated contaminants. Thus the sediment from many dredging projects, even when trace pollutants are present, are not contaminated to a degree that causes toxicity or that otherwise represents any significant environmental risk. The following section presents a detailed discussion of contamination in dredged material.

3.2.3 Contaminants in Dredged Material

The vast majority of sediments in the Estuary are not polluted to a degree that poses any threat to human health or the environment. However, as noted above, the dynamic nature of the Estuary means that sediment particles may settle in one location, later to be resuspended and transported some distance, and then settle again. Because of this, even if sediment particles are not initially carrying pollutants when entering the Bay, they may have many chances to pick up pollutants from the water or air before they are removed from natural circulation (either by settling into a depositional area and becoming buried, or being carried out the Golden Gate). This section briefly discusses the natural compounds and man-made (anthropogenic) pollutants that may become associated with sediments; when "contamination" is considered to be a problem in sediments; major

Figure 3.2-19 Modeled Initial Sediment Deposition Patterns from Disposal at the Alcatraz Island Site

Figure 3.2-20 Modeled Initial Sediment Deposition Patterns from Disposal at the San Pablo Bay Site

Figure 3.2-21 Modeled Initial Sediment Deposition Patterns from Disposal at the Carquinez Site

sources of sediment contamination; and locations of contaminated sediments in the Bay/Delta estuary.

3.2.3.1 Anthropogenic vs. Non-Anthropogenic Chemicals — What is "Contamination"?

Nationwide, the most frequently reported contaminants in sediments include heavy metals (e.g., cadmium, chromium, copper, lead, nickel, mercury, selenium, silver, and zinc), metalloids (e.g., arsenic), polychlorinated biphenyls (PCBs), pesticides (e.g., DDT compounds), and polynuclear aromatic hydrocarbons (PAHs) (USEPA 1994a; SFEP 1992b). A few of these can have natural origins. Heavy metals at varying concentrations are natural constituents of the crustal rock formations in different areas. As these formations erode and eventually contribute to the sediments, the crustal concentrations of these metals are reflected in the sediment chemistry. Even some PAHs can be found in otherwise "unpolluted" sediments. For example, some PAHs (such as pyrene and perylene) are combustion products that can make their way into sediments — from runoff (Hoffman et al. 1984) or via atmospheric deposition — as a result of natural forest fires as well as human causes. Other organic compounds (such as phenols) can be formed by decomposition of organic matter in marshes and elsewhere (Sims and Overcash 1983). Information about the "background" concentrations of chemicals in Estuary sediments is therefore helpful when determining whether the measured concentrations of a chemical are high enough to indicate that the sediment may be "contaminated" by an anthropogenic source. Further discussion of chemical concentrations typically encountered in the Estuary's embayments is presented in section 3.2.3.3 and in Chapter 4.

Typically, the pollutant types noted above are the most highly concentrated types of anthropogenic contaminants in sediments because they are poorly soluble in water (hydrophobic) and have a high affinity (adsorption potential) for sediment particle surfaces or the organic matter associated with them.

These compounds are therefore readily removed from the water column by suspended particles, and are preferentially carried into the sediments as the particles settle out. The settling-out of suspended particulates is enhanced in estuaries, including the Bay/Delta Estuary, by the flocculation (aggregation of finer suspended particles into larger, more quickly settling groups) that naturally occurs where fresh and more saline waters mix. This is why sediments in general, and particularly sediments in estuaries such as the Bay/Delta, are often thought of as "sinks" for contaminants that get into the water column from point or non-point sources.

But from whatever source, a sediment is considered to be "contaminated" when it contains deleterious chemical substances at concentrations that pose a known or potential threat of adverse impact to aquatic life, wildlife, or human health (USEPA 1994a). The degree of the threat by the contaminants in a particular sediment can change depending on how the sediment is handled (e.g., buried contamination left in place may not pose a threat of ecological impact, but if that material is disturbed, such as by dredging and disposal, the contaminants may become available again and have the potential to cause adverse effects). As discussed in the sections that follow, determining whether contaminants in dredged material may pose a threat of adverse impacts is a function of determining the following: (1) the potential for the contaminants to cause adverse effects at the placement site; and (2) the practicability of control measures that may be effective in reducing or eliminating the potential adverse effects at the placement site.

3.2.3.2 Major Sources of Sediment Contamination

In general, the surficial sediments in San Francisco Bay have been deposited since industrialization began in California, and therefore may have been exposed to anthropogenic sources of pollutants. These "industrial age" sediments can be encountered in both new work and maintenance dredging. (However, the more highly contaminated sediments are usually encountered by new work projects in industrialized areas of the Bay; such dredging commonly encounters sediments that became contaminated in decades past, before today's stricter regulations on discharges were in effect.) Recent sand deposits — either riverine sand in portions of San Pablo and Suisun bays and the lower Sacramento River, or sand bars maintained by strong currents in central San Francisco Bay and the San Francisco Bar — also may be exposed to anthropogenic sources of pollutants, but typically do not accumulate significant concentrations of them, for the reasons noted above.

Existing permits authorize hundreds of millions of gallons of treated industrial and municipal effluents to be discharged into the Estuary each day. While these effluents are carefully regulated to ensure that they are not directly toxic to Estuary organisms, trace levels of various contaminants are associated with these discharges, and some of these contaminants can end up concentrating in the Estuary's sediments. For example, Figure 3.2-22 shows the locations of the largest

municipal discharges (Publicly-Owned Treatment Works, or POTWs), and their mean discharge volumes as of 1995. Similarly, a variety of industries discharge pollutants associated with sediment contamination. Table 3.2-2 lists over 40 of these classes of industries, along with the typically associated contaminants. The majority of these types of industries have historically been in operation around the Estuary. Some of the major industrial facilities still in operation as of 1995 are shown in Figure 3.2-23.

Even though industries and municipalities discharge major volumes of effluent each year, they are not the primary sources of pollutants that contribute to contamination of the Estuary's sediments today. Figure 3.2-24, from SFRWQCB (1994), shows the combined annual loadings to the Estuary of several heavy metals (arsenic, cadmium, chromium, copper, lead, and zinc) from six major sources. The sources shown are municipal and industrial effluents, riverine input, urban runoff, non-urban runoff, atmospheric deposition, and dredged material. Other sources may be important on a local level (e.g., groundwater). It can be seen from this figure that non-point pollution (especially nonurban runoff) is the source for the vast majority of these pollutants. This pattern generally holds true for other categories of chemicals, as well. It is because of these patterns that sediments near the urbanized shorelines, and especially in enclosed nearshore waters in the vicinity of storm drains and other input locations for nonpoint source pollutants, often continue to become contaminated even though permitted point sources have largely been brought under control in recent years.

It is useful to keep in mind that not all contaminants that may be associated with point and nonpoint discharges into the Estuary are necessarily contaminants of concern in the sediments. Highly water-soluble compounds will not tend to concentrate onto sediment particles in the first place. Similarly, affinity (adsorption potential) for sediment particle surfaces or the organic matter associated with them. lower molecular weight organic compounds that are highly volatile will tend to dissipate before being incorporated into the sediments. For example, "BTEX" (a mixture of benzene, toluene, ethyl-benzene, and xylene) is often a concern in upland soils excavated from around leaking underground storage tanks; hence landfills typically require information about BTEX concentrations before accepting contaminated soils for disposal. However, BTEX would rarely be found in estuarine sediments (see discussion on Current Upland Testing Practice in section 3.2.5.2). Overall, although there are limited areas of highly contaminated sediments associated with specific sources, the majority of sediments in the Estuary are characterized by low concentrations of contaminants spread through large volumes of material. In contrast, cleanup projects addressing contaminated upland soils typically encounter small volumes of highly contaminated material (and the contaminants themselves are often different).

Dredged sediments that are determined to be not suitable for unconfined aquatic disposal are very rarely classified as "hazardous." The following section gives an overview of contaminants that are typically found in Estuary sediments.

3.2.3.3 Contamination Levels in San Francisco Bay/Delta Estuary Sediments

There have been several programs in San Francisco Bay that have monitored concentrations of contaminants in sediments from various embayments. Historical sediment chemistry data collected from numerous Bay surveys performed between 1971 and 1986 have been summarized by Long and Markel (1992). Data from these surveys are presented in Table 3.2-3. This table compares mean chemical concentrations in sediments from the central areas of San Pablo Bay, central San Francisco Bay, and the south Bay with chemical concentrations in sediments from peripheral areas of these basins (marinas, harbors, ship channels, and industrial waterways) that would be expected to be more directly influenced by pollutant sources. These data indicate that, overall, the peripheral industrialized areas indeed have higher mean contaminant concentrations than do the central basins. For most compounds, the range of contaminant concentrations is also greater in the peripheral industrial areas than in central basin samples (Long et al. 1988).

Figure 3.2-22 Location of Publicly-Owned Treatment Works (POTWs) in the Bay Area

Table 3.2-2 Industries Associated with Sediment Contamination

Figure 3.2-23 Location of Industrial Discharge Sites in the Bay Area

Figure 3.2-24 Combined Pollutant Loadings to the Bay/Delta by Source Type

Table 3.2-3. Mean Concentrations of Selected Toxicants in Surficial Sediments from Three Basins and Four Peripheral Area of San Francisco Bay
(Adapted from Long and Markel 1992)

	BASINS			**PERIPHERY**			
	San Pablo Bay	***Central SF Bay***	***South SF Bay***	***Oakland Inner Harbor***	***Islais Creek Harbor***	***Redwood Creek***	***Richmond Harbor***
Trace Metals (ppm, dry weight)							
Mercury	0.45	0.35	0.65	0.57	1.30	0.42	0.40
Cadmium	0.71	0.79	1.44	0.67	2.23	2.47	0.65
Copper	45	33	33	72	78	66	36
Lead	32	34	30	97	102	87	39
Chromium	280	81	84	ND	140	91	123
Silver	0.45	0.72	0.57	ND	4.69	ND	ND
Organics (ppb, dry weight)							
Total PAHs	2,600	3,900	2,700	7,200	62,700	ND	ND
Total DDTs	9	16	3	120	3	ND	260,700*
Total PCBs	27	71	28	361	305	ND	ND

* Includes stations within the United Heckathorne/Lauritzen Canal Superfund site. Mean concentration of DDT compounds for other areas of Richmond Harbor is less than 200 ppb.

More recent data sets documenting Bay-wide trends in sediment contamination have been collected as part of the state's San Francisco Bay Regional Monitoring Program and the Bay Protection and Toxic Cleanup Program (BPTCP) (described in more detail in Chapter 4). Overall, the median, maximum, and minimum concentrations of selected sediment contaminants from monitoring locations representing ambient conditions in the three main basins of the Bay (areas removed from known sources of contamination) are presented in Table 3.2-4. Generally, the medians and ranges of ambient contaminant concentrations observed are consistent among the three basins.

Concentrations of trace metals are consistently low in each of the basins and are similar to historical data (discussed earlier) reported by Long et al. (1988) (Table 3.2-3). Likewise, median concentrations of the pesticide DDT and total PCB are consistently low (< 4.5 ppb and < 11.2 ppb, respectively) in samples from all three basins, although the maximum concentrations of both chemicals measured in the north and central Bay sediments are higher than those from the south Bay. PAHs are the only contaminants whose ambient concentrations appear to be both elevated in some of the basins and variable between basins. Median values for summed PAHs (HPAHs

Table 3.2-4. Ambient Concentrations of Selected Contaminants in San Francisco Bay Sediments from Recent Monitoring Programs

Chemical	**SOUTH BAY # OF SITES = 11**				**CENTRAL BAY # OF SITES = 9**				**NORTH BAY # OF SITES = 13**			
	Median	*Min*	*Max*	*N*	*Median*	*Min*	*Max*	*N*	*Median*	*Min*	*Max*	*N*
Silver	0.4	0.1	1.2	39	0.2	0.0	0.4	31	0.2	0.0	0.5	43
Arsenic	8.9	0.8	14.2	38	9.6	0.7	29.4	31	7.7	0.6	20.6	43
Cadmium	0.2	0.0	0.7	39	0.2	0.0	0.3	31	0.2	0.1	0.6	43
Chromium	93.3	9.4	213.0	39	75.7	8.5	238.0	31	81.5	6.9	209.0	43
Copper	38.3	16.6	94.6	39	32.7	8.0	56.5	31	46.0	13.2	71.9	43
Mercury	0.3	0.1	0.5	38	0.2	0.0	0.4	31	0.2	0.0	0.4	48
Nickel	76.9	14.9	130.8	39	72.8	12.2	107.0	31	76.3	12.9	135.0	43
Lead	23.1	10.6	45.4	39	21.5	8.4	33.7	31	23.1	5.6	115.0	43
Selenium	0.3	0.1	1.3	38	0.3	0.0	0.9	31	0.2	0.1	3.3	48
Zinc	109.0	44.8	221.8	39	87.9	39.0	154.0	31	120.1	34.9	180.0	43
Sum DDTs	2.7	0.0	12.5	34	4.5	0.0	63.1	31	3.7	0.1	70.3	48
Sum HPAH	2,226.4	1,239.6	6,837.1	34	1,954.3	1.9	5,844.0	31	508.5	33.2	2,705.9	49
Sum LPAH	204.6	12.0	1,065.6	34	239.2	0.0	646.5	31	49.7	0.0	246.0	48
Total PCB	9.2	8.6	25.3	10	11.2	0.0	38.7	9	8.8	3.8	117.0	22

Notes: Units: metals mg/kg (ppm) dry weight.
organics µg/kg (ppb) dry weight.
N = number of samples.
All 1993 Status and Trends data; SFBWQCB 1994 unpublished data from RWQCB 1994-95 Bay Protection and Toxic Cleanup Program Reference Study.

FINAL REPORT: LTMS CONTAINMENT SITES COMMITTEE

TASK: Develop a list of potential disposal sites capable of handling "contaminated" or "unsuitable" dredge material.

The Containment Sites Task Committee held two meetings, the first on December 14, 1992, and the second on January 19, 1993, and reached a general consensus about its work. Four major areas of substantive work were involved, and the Committee reached a consensus in each area, as follows:

1. *Locate major (probable) areas and amounts of "contaminated" or "unsuitable" dredged materials.*

   - After discussion with the Regional Board and review of their work on the Bay Protection and Toxic Cleanup Program (BPTCP), we concluded that it was appropriate to adopt a "planning" estimate of 10 million cubic yards that will need to be dredged over the next 10 years. This only involves material removed as part of dredging, and does not include clean-up of hot spots. This estimate was based on an estimate of 2 million cubic yards of unsuitable material in the Oakland deepening project, 1 million cubic yards of unsuitable material in the Richmond deepening project, and 500,000 cubic yards each year (about 10-20 percent) of maintenance material that might be expected to fail tests for in-Bay or ocean disposal. This reserves up to 3 million cubic yards of unsuitable material capacity for the Navy's deepening projects and for other projects and hot spots. This estimate should be updated when the Regional Board adopts a final report under the BPTCP.

2. *Develop alternative strategies for addressing "contaminated" or "unsuitable" materials, e.g., (a) leaving such materials in-place, (b) confined disposal — either upland or aquatic, and (c) treatment solutions.*

   - The committee judged that all of these options may be suitable strategies. Unfortunately, too little is known about the location, quantity, and degree of contamination of most material to be able to select the appropriate disposal option. Thus, the committee spent most of its efforts on confined disposal.

3. *Determine whether any of the sites now under consideration could handle dredged materials and, if so, in what amounts.*

   - The Committee concluded that approximately 6 million cubic yards of disposal capacity was available as reuse for daily cover in Redwood Landfill, approximately 10 million cubic yards of capacity may be available in the Montezuma Slough project, and approximately 10 million cubic yards of disposal may be available in the borrow pit near Bay Farm Island. Other sanitary landfills and other drying and/or rehandling sites can also handle unsuitable material, but these three sites appear to be the most advanced of sites now under consideration.

4. *Recommend at least three sites that should be brought on-line to handle "contaminated" or "unsuitable" sediment disposal needs.*

   - The Committee decided not to recommend specific sites, largely because several specific sites appear to be heading toward environmental review and permitting. Instead, the Committee established the following hierarchy of preference for disposal site types. This hierarchy reflects the relative certainty of confinement in the disposal site, and the ease of management.

     First Choice: The preferred disposal location of the Committee is for upland disposal in landfills. The Committee understands that landfill capacity and permitting are issues for this option. However, the Committee concluded that this option provided the greatest certainty of containment, ease of management, and provided the additional benefit of also acting as daily cover.

     Second Choice: The Committee concluded that confinement in wetlands represented a suitable disposal option if done properly. In particular, the Committee believed it was essential to make sure that channels would not erode the placed sediments. The Committee considered this alternative to be less certain than landfill disposal because construction might involve hydraulic placement with more opportunity for runoff and because biological activity would disturb a portion of the covering soils.

     Third Choice: The Committee concluded that confinement at in-water capping sites represented a suitable disposal option if done properly. However, in-water capping raises complex technical issues including the question of material loss during initial placement, long-term stability of the cap, and consistency with applicable laws. The Committee also concluded that the LTMS should be the forum for consideration of this option, and that such consideration should take place in the in-Bay Work Group, as an explicit part of their work program.

and LPAHs) ranged from 558 ppb in the north Bay to 2,431 ppb in the south Bay. Relatively high concentrations of HPAHs (in excess of 4600 ppb) were repeatedly observed at several sampling locations in the south Bay.

While such overall trends in the basins are readily discernable, contaminant distributions and sediment toxicity can be very patchy in all areas of the Estuary. Areas around shipyards and naval facilities — where in decades past it was common to simply dump wastes such as used solvents, paint, and other chemicals off the sides of ships or docks, or down drains leading to the Bay — often have highly contaminated sediments. For example, the Hunters Point Naval Shipyard is a Superfund cleanup site today, and some nearby sediments have accumulated pollutants — most notably heavy metals — to the point of exceeding hazardous waste criteria (PRC 1994). Similarly, onshore industrial facilities often dumped pollutants into the Bay, contaminating sediments in the vicinity. Past operation at the United Heckathorn facility in the Port of Richmond contaminated nearby sediments with DDT and other compounds; this site has also been the subject of a Superfund cleanup action (Lincoff et al. 1994). Likewise, elevated levels of petroleum-derived contaminants have been observed in sediments from Castro Cove, a site that has historically received discharges from a nearby oil refinery (SF RWQCB 1994).

3.2.3.4 Efforts to Reduce Sediment Contamination

Throughout the Estuary there are many other such examples of significant site-specific sediment contamination resulting from identifiable local sources. However, important strides have been made in the last several years to control such identifiable sources. As described above in section 3.2.3.2, the primary sources of sediment contamination now (with the exception of accidental spills directly into the Estuary) are nonpoint sources (runoff from urban and agricultural areas, stormwater discharges, and atmospheric deposition). These kinds of sources, combined with the Bay's natural resuspension processes, often result in much less intensive but more wide-spread and "patchy" sediment contamination. As a result of remedial action programs (such as BPTCP and Superfund that address the most significant areas of sediment contamination from past activities), better regulation of point source discharges under the NPDES program jointly administered by EPA and the state of California, and increasing attention to non-point source discharges by many programs at the federal, state, and local levels, levels of contaminants in Bay area surficial sediments would be expected to continue to decrease over time. However, it must also be expected that both new work and maintenance dredging projects will continue to occasionally encounter "unsuitable" material (sediments that cannot go to unconfined aquatic disposal sites, but instead need some form of specific management for their contamination) throughout the planning time frame of this EIS/EIR. For planning purposes, this LTMS EIS/EIR assumes that 20 percent of all dredged material will be unsuitable for unconfined aquatic disposal. This percentage is based on a review of available sediment quality information on recent projects, as well as major new work (harbor deepening) projects that can reasonably be anticipated (see the following text box on the LTMS Containment Sites Committee). It already appears evident that this estimate is conservative,[2] and the 20 percent figure probably overestimates to some degree the long-term volume of dredged material needing special handling.

3.2.3.5 Determining When "Contamination" is a Problem in San Francisco Bay/Delta Estuary Sediments

The potential for contaminants in dredged material to cause an adverse biological effect at a placement site is related to the bioavailability of the contaminants present, and to the opportunity for organisms of concern to be either directly or indirectly (e.g., via groundwater) exposed to them. The term "bioavailable" is used here broadly to refer to a contaminant whose concentration and chemical form make it available for uptake by an organism, so that the contaminant can then (directly or after being metabolized to a more toxic chemical compound or form) cause an adverse biological impact. The bioavailability of contaminants in sediments can change dramatically depending on the placement environment, and depending upon a wide variety of factors that can vary from sediment to sediment (such as organic carbon content, salinity, pH, oxidation/reduction potential, and particle size). The following text box describes some of the major chemical factors controlling bioaccumulation (the uptake of contaminants into an organism). Also, see section 3.2.4 (Exposure Pathways and Potential Risks) below for a more detailed discussion of how the type of placement environment can affect contaminant bioavailability.

The opportunity for organisms (or other resources of concern) to be exposed to bioavailable contaminants at toxicologically significant concentrations is often a more site-specific matter. Contaminants that are in a bioavailable form may not represent an adverse effect if organisms cannot be exposed to them. For example,

sediments that would be of concern for placement in an unconfined aquatic disposal site — due to the presence of elevated concentrations of contaminants that are bioavailable to marine organisms — may be fully suitable for placement at a properly constructed landfill, where contaminants can be contained and organism exposure is minimized. These same sediments may also be suitable for beneficial reuse of various kinds. The type and degree of testing needed (if any) must be based on the potential exposure pathways that are determined to be of concern for a particular project and its potential disposal sites. See section 3.2.5 (Role of Sediment Evaluation) below for a discussion of the testing frameworks for aquatic and upland placement environments.

If evaluation of sediment quality (including any testing data) shows that there is the potential for unacceptable adverse effects at the proposed placement site, control measures can be considered for reducing or eliminating the risk. See section 3.2.4.5 below for a discussion of the kinds of control measures that may be appropriate for various contaminant pathways of concern in each type of placement environment. If potential control measures would not be effective in adequately reducing the risk of adverse contaminant- related effects, an alternative disposal option must be selected if the sediments must be dredged (it is sometimes possible to avoid dredging problematic sediments by reconfiguring the project — however, the environmental acceptability of leaving contaminated sediments in place must also be considered.

3.2.4 Contaminant Exposure Pathways and Potential Risks in Different Placement Environments

In order for contaminants associated with sediment particles to cause a biological effect, an organism must be exposed to the contaminants in a bioavailable form. Organisms can be exposed to contaminants in sediments directly (e.g., via ingestion of or direct contact with the sediment), or indirectly (e.g., via contaminated surface or groundwater, or by eating other organisms that have taken up contaminants from the sediments). This section provides a basic description of the contaminant "exposure pathways" and potential risks that are associated with disposal of dredged material in the various types of placement environments (ocean, in-Bay, nearshore/wetland and upland) considered in this EIS/EIR.

Overall, dredging and aquatic disposal can have effects on organisms in the water column, or in the benthos. The location of the dredging and disposal sites in relation to resources of concern, and whether an aquatic disposal site is erosional or depositional, are important in determining which of these pathways may be of most concern. There are also important overall differences when dredged materials are placed in upland versus aquatic sites: upland sites represent very different geochemical environments, in which the behavior of sediment-associated contaminants can be dramatically different than under aquatic conditions. Sediments placed in upland locations can affect a different mix of organisms, and can have effects on surface water quality, groundwater quality, and air quality.

There is also a difference between the different placement environments in the ability to engineer disposal sites to appropriately manage the relevant contaminant exposure pathways. Generally, it is not possible to control organism exposure to dredged material or to limit organism access at dispersive unconfined aquatic sites. At non-dispersive unconfined aquatic sites, organism exposure can be limited but organism access typically cannot (other than indirectly, by locating sites to avoid important habitat areas). In contrast, design features can be included at confined aquatic disposal sites and at many upland/wetland reuse (UWR) sites to limit both organism exposure and organism access.

A basic understanding of how the exposure pathways differ between the placement environments is essential to determining the need for specific management restrictions, and designing and implementing placement site design features that are truly effective at minimizing or eliminating potential impacts. The following sections discuss the main contaminant exposure pathways for each major placement environment, and potential control measures for them.

Major Chemical Properties Controlling Propensity of Contaminants to Bioaccumulate from Sediments
(Adapted from USEPA and USACE 1994)

Hydrophobicity

Literally, "fear of water;" the property of neutral (i.e., uncharged) organic molecules that causes them to associate with surfaces or organic solvents rather than to be in aqueous solution. The presence of a neutral surface such as an uncharged organic molecule causes water molecules to become structured around the intruding entity. This structuring is energetically unfavorable, and the neutral organic molecule tends to be partitioned to a less energetic phase if one is available. In an operational sense, hydrophobicity is the reverse of aqueous solubility. The octanol/water partition coefficient (K_{ow}) is a measure of hydrophobicity. The tendency for organic molecules to bioaccumulate is related to their hydrophobicity. Bioaccumulation factors increase with increasing hydrophobicity up to a log K_{ow} of about 6.00.

Aqueous Solubility

Chemicals such as acids, bases, and salts that speciate (dissociate) as charged entities tend to be water-soluble and those that do not speciate (neutral and nonpolar organic compounds) tend to be insoluble, or nearly so. Solubility favors rapid uptake of chemicals by organisms, but at the same time favors rapid elimination, with the result that soluble chemicals generally do not bioaccumulate to a great extent. The soluble free ions of certain heavy metals are exceptional in that they bind with tissues and thus are actively bioaccumulated by organisms.

Stability

For chemicals to bioaccumulate, they must be stable, conservative, and resistant to degradation (although some contaminants degrade to other contaminants that do bioaccumulate). Organic compounds with structures that protect them from the catalytic action of enzymes or from non-enzymatic hydrolysis tend to bioaccumulate. Phosphate ester pesticides do not bioaccumulate because they are easily hydrolyzed. Unsubstituted polynuclear aromatic hydrocarbons (PAHs) can be broken down by oxidative metabolism and subsequent conjugation with polar molecules. The presence of electron-withdrawing substituents tends to stabilize an organic molecule. Chlorines, for example, are bulky, highly electro-negative atoms that tend to protect the nucleus of an organic molecule from chemical attack. Chlorinated organic compounds tend to bioaccumulate to high levels because they are easily taken up by organisms and, once in the body, they cannot be readily broken down and eliminated.

Stereochemistry

The spatial configuration (i.e., stereochemistry) of a neutral molecule affects its tendency to bioaccumulate. Molecules that are planar tend to be more lipid-soluble (lipophilic) than do globular molecules of similar weight. For neutral organic molecules, planarity can correlate with higher bioaccumulation unless the molecule is easily metabolized by an organism.

3.2.4.1 Exposure Pathways in Aquatic Placement Environments

Dredging, and dredged material disposal at aquatic sites, can result in adverse effects in two basic "compartments": in the water column, and on the bottom (benthos). Figure 3.2-25 depicts these exposure pathways for open water disposal. Water column effects occur when sediment particles are disturbed from the bottom and resuspended during dredging and disposal, and are usually limited to the immediate period when dredging and disposal activities occur. Benthic effects can result from physical burial of benthic organisms at the disposal site, and long-term exposure of local organisms to the sediments on the bottom after disposal has ceased.

Typical in-place estuarine sediments are dark in color and reduced, with little or no oxygen (anaerobic conditions). Reduced, anaerobic conditions favor the partitioning of contaminants onto the sediment particles or the organic matter associated with them. Thus the bulk of sediment contaminants may not be directly available to many aquatic organisms. During typical dredging and disposal operations, the exposure of anaerobic sediments to oxygenated water is sufficiently short term that the reduced characteristics of the sediment do not change appreciably. At a non-dispersive aquatic disposal site the dredged material quickly settles to the bottom, where anaerobic conditions are quickly restored or maintained.

Water Column vs. Benthos

As discussed in section 3.2.3.1, specific contaminants typically are associated with sediments because they are either hydrophobic or otherwise are easily scavenged from the aqueous phase. The same processes that preferentially bind these contaminants to sediment particles make it relatively difficult for the contaminants to disassociate from the particulates and go back into the aqueous phase during dredging and aquatic disposal operations. Those contaminants that do disassociate from sediment particles in a disposal plume usually do so for only very short periods of time before they are re-scavenged by other suspended particles. The degree of water column effect will be directly related to the extent and speed of dilution of the water column plume and/or resettling of the resuspended sediment. Water column impacts are therefore evaluated by comparisons with water quality standards and evaluation of the potential for short-term toxicity, considering the mixing that may occur at the dredging or disposal site in question.

Potential water column effects can usually be managed by selection of appropriate dredging/disposal methods and discharge rates (for a listing of control measures see section 3.2.4.5), in conjunction with designation of an appropriate mixing zone. Mixing zones are areas (designated by the relevant state for inshore and state waters, and by federal criteria for offshore ocean waters) outside of which water quality standards must be met and beneficial uses of the waterbody must be protected. In general, mixing zones may not be so large as to inhibit the movement or migration of aquatic species, or to allow degraded water quality to extend throughout a significant portion of a water body. Most states (including California) have both numerical and "narrative" water quality standards that must be met at all points outside the boundaries of the mixing zone. The narrative criteria for California state that, in addition to meeting all relevant numerical water quality criteria, plumes outside mixing zones cannot include "toxic substances in toxic amounts." Therefore, pre-disposal testing for potential water column impacts evaluates both water quality (against numeric criteria) and short-term suspended particulate phase toxicity, considering the dilution characteristics of the specific disposal site and any specific mixing zone designated for that site. Section 3.2.5 describes sediment testing approaches in more detail.

Because most fish species are able to actively avoid the immediate vicinity of dredging and disposal areas, and because water column plumes during dredging or disposal are usually local and temporary (diluting to background levels within minutes to a few hours after dredging or disposal operations cease), the water column pathway rarely results in significant direct impacts to most aquatic organisms except in certain, limited circumstances. These could include the following:

- Continuous dredging or discharging near specific resources of concern;
- Dredging of highly contaminated sediments or sediments with an unusually high oxygen demand;
- Dredging or discharging within constricted areas where water column mixing would be inadequate; or
- Dredging or discharging at locations and during times where increased suspended particulates would have a direct effect on particular species of concern (such as herring spawning sites, when spawning herring or incubating eggs are actually present).

Figure 3.2-25 Contaminant Pathways for Open Water Disposal

Based on the long-term experience of disposing approximately 400 mcy of dredged material per year at hundreds of aquatic disposal sites nationwide, the water column is rarely found to be the primary pathway of concern.

In contrast, benthic exposure to "undiluted" (solid phase) dredged material after disposal can be long-term. On-site benthic infauna and epifauna can be exposed long enough that any contaminants in the dredged material can directly affect them, or they can accumulate bioavailable contaminants to such a degree that animals that prey on them may be adversely affected. Therefore, potential impacts as a result of benthic exposure are typically considered by evaluating both longer-term solid phase toxicity, and bioaccumulation (section 3.2.5 describes sediment testing approaches in more detail). Solid phase toxicity testing evaluates whether the sediments may be toxic to organisms directly exposed to them. Bioaccumulation testing provides an indication of the potential bioavailability of contaminants in the dredged material, which in turn aids in the evaluation of whether (given the location and characteristics of the particular disposal site) there is the potential for trophic transfer of contaminants to other organisms that are not necessarily directly exposed to the dredged material (i.e., food web effects).

Dispersive vs. Non-Dispersive Aquatic Sites

The basic aquatic contaminant exposure pathways described above apply to both dispersive and depositional disposal sites. However, the applicability and effectiveness of potential control measures differs substantially between dispersive sites (such as the existing in-Bay disposal sites) and depositional sites (such as the SF-DODS).

Dredged material does not remain on the bottom for long periods of time at the existing, predominantly dispersive in-Bay disposal sites. Instead, fine sediments are resuspended and transported away from these sites. These resuspended sediments may resettle and resuspend again several times before either leaving the Estuary system through the Golden Gate, or finally settling in a depositional site (see section 3.2.2). The testing methods for evaluating whether water column restrictions (control measures) are needed to reduce adverse effects of the suspended particulate phase of dredged material (section 3.2.5) are also appropriate for evaluating whether sediments resuspended from dispersive sites may pose a contaminant-related risk. In most cases, if the original disposal of the dredged material did not require contaminant-related water column control measures then it is unlikely that water column control measures would be required for subsequent resuspension, due to the likelihood of increased dilution during each subsequent resuspension event. Thus, the water column pathway is not necessarily of any greater concern at dispersive versus depositional sites. (Note that this refers only to *contaminant-related* effects. Physical effects — such as potential local or embayment-wide increases in turbidity that might be associated with high levels of disposal at dispersive sites — may still be of concern.)

In contrast, the benthic pathway can be of greater concern at dispersive sites compared to depositional ones. Although this may at first seem contradictory (after all, fine dredged material by definition does not remain on the bottom at dispersive sites), the difference relates to the inability to monitor the ultimate fate of dredged material placed at dispersive sites, and to confirm that adverse benthic effects are not occurring at some other, unanticipated location. As noted in section 3.2.2, there is very limited ability today to accurately predict where or how much dredged material resuspended from in-Bay sites will ultimately settle. However, it is known that many contaminants will tend to be at higher concentrations in the fine particle size fractions of sediment (section 3.2.1), and that the fine fractions (particularly the silts) are the most easily eroded from dispersive disposal sites. It is therefore theoretically possible that the overall concentrations of contaminants measured in a whole sediment sample can underestimate the risk posed by preferential settling of hydraulically sorted fines at depositional locations away from the disposal site.[3]

Unfortunately, it is virtually impossible to confirm that contaminants from dredged materials so dispersed are or are not resulting in adverse off-site effects. By the same token, it is often impossible to prove that any off-site effects that are measured are in fact a result of dredged material from a dispersive site, rather than from some other source. This uncertainty must be viewed as a risk that adverse environmental effects could occur. From a dredged material management standpoint, two considerations become the focus of addressing this risk. First, is the dredged material "clean" enough that, even if fines preferentially settled in a single location, adverse impacts are unlikely? And second, are alternative disposal sites available and practicable to use that would manage the dredged material with less risk? Because there is little ability to manage contaminant-related risk at dispersive sites by other means, the primary effective control measure that can be used to address potential adverse benthic effects related to dispersive sites is to avoid, in the first place,

disposal of significant quantities of dredged materials that contain appreciable concentrations of contaminants.

At predominantly depositional sites, dredged material is expected to remain on the bottom, within the boundaries of the disposal site. This makes it much more possible to monitor site performance and to confirm that unacceptable adverse effects are not occurring, or to take corrective action if necessary. If adverse effects are indicated in the vicinity of a depositional disposal site, it can be determined much more readily than at a dispersive site whether this is due to dredged material disposal or some other cause. A listing of potential management and control measures applicable to depositional sites is presented in section 3.2.4.5.

3.2.4.2 Exposure Pathways in Upland Placement Environments

When dredged material is placed in an upland environment (i.e., a site with no tidal action), important physical and/or chemical changes occur once disposal operations cease and the sediments begin to dry (Francingues et al. 1985). As it dries and cracks form, the dredged material will oxidize and become lighter in color. Accumulations of salt will develop on the surface and the edge of cracks. Rain will tend to dissolve the salts and remove them in surface runoff, and accumulations of some now-oxidized metals may be carried away with the runoff as well. As drying proceeds, organic complexes (which had sequestered many contaminants away from organisms in situ, anaerobic sediments) oxidize and decompose. Sulfide complexes also oxidize to sulfate salts, and acidity may increase (pH may drop) dramatically. Lowered pH can directly affect the speciation and reactivity of various heavy metals (generally making them more soluble and reactive, and therefore more bioavailable and toxic). Lowered pH also directly affects the toxicity of ammonia produced by decomposing organic matter. These transformations can promote the release of contaminants into surface water and groundwater (via leachate), and organisms exposed to these water sources, or to the site itself, may readily take up these released contaminants. However, recent studies of dredged material placement for wetland creation have demonstrated that drying for purposes of maximizing site capacity does not necessarily promote the release of contaminants or their bioavailability (LTMS 1995d). Nonetheless, site management measures, such as resaturation of dried sediments prior to the restoration of tidal action, can be taken to minimize the bioavailability of contaminants. Volatilization of some contaminants into the air may also occur from dewatered dredged material placement sites, resulting in an additional potential exposure pathway. From a human health standpoint, fugitive dust can be a pathway of particular concern. In certain circumstances fine particles of dredged material, with any associated adsorbed contaminants, can be blown from upland placement sites if the surface of the dredged material is allowed to dry completely. This "fugitive dust" can be inhaled or ingested by on-site workers and people living, playing, or working nearby. Fortunately, fugitive dust can be easily controlled by standard operating procedures (principally, keeping the surface of the site moist when the dredged material is exposed). Figure 3.2-26 shows the exposure pathways potentially associated with dredged material placement in upland environments.

These differences (compared to the behavior of dredged material in an aquatic environment) lead to different mechanisms by which organism may be exposed to contaminants in dredged material, as well as to differences in the types of resources that may be exposed. For example, upland placement of dredged material can potentially affect:

- *Surface water quality* (and any organisms exposed to the affected water body). Depending on the specific placement site, the receiving water body may be a river, slough, or the Bay. Surface water quality may be affected by return effluent during initial filling of the upland site with dredged material; rainwater runoff from the site after the dredged material has been initially dewatered; or seepage from the site into other adjacent surface waters.

- *Groundwater quality* (and any organisms ultimately exposed to the groundwater). Groundwater impacts are avoidable by both appropriate siting of upland facilities (i.e., avoid areas where underlying groundwater quality is high, and/or is used for drinking water or other domestic purposes), and by proper engineering of the upland facility itself (e.g., impermeable liners and/or leachate collections systems where appropriate).

Figure 3.2-26 Contaminant Pathways for Upland Confined Disposal Facilities

- *Wildlife attracted to site while it is flooded* (e.g., in the early stages of the drying process, when sediments are still settling and consolidating and before overlying water has been decanted).

- *Other wildlife using the site after the sediments have dried* (e.g., exposure to or through invertebrates colonizing the site).

- *Plant uptake of contaminants* from the dried sediments (especially certain metals that can be taken up into plant tissues from the surface, oxygenated layer of the sediment deposit). Bioaccumulation of contaminants into plant tissues can be of concern for wildlife who may be exposed to the contaminants by eating the plants.

- *Air quality* (volatilization of some compounds from the surface layers of the sediment deposit, odor, fugitive dust — these are discussed further in Chapter 6).

- *Human health* (via direct exposure, or indirect exposure via air quality impacts or water quality impacts). However, risks to human health from dredged material at upland sits is highly dependent on the type and level of contaminants in the material and site-specific factors.

Although exposure pathways for upland placement sites may seem more complex than for aquatic sites, it is also important to note that it is often more possible to engineer effective control measures at upland or nearshore sites than it is to do so at unconfined aquatic sites. In contrast to dispersive unconfined aquatic disposal sites in particular, operational and design features can generally be incorporated into upland placement sites to address any of the pathways listed, should they be of concern on a projects-specific basis.

3.2.4.3 Exposure Pathways in Nearshore Placement Environments

Nearshore placement sites (i.e., diked historic baylands or diked baylands now restored to tidal action) combine the characteristics of upland and aquatic sites, and all of the exposure pathways of those environments can come into play. Similarly, nearshore sites are intermediate between upland and aquatic sites in terms of the ability to engineer control measures to address the contaminant exposure pathways. Figure 3.2-27 shows the typical exposure pathways for nearshore placement sites.

Much of the dredged material placed at nearshore disposal sites will remain saturated and anaerobic, thereby minimizing the geochemical changes that occur with upland placement, and that can lead to increased contaminant solubility or mobility. On the other hand, there will generally be less initial dilution of the water and any suspended solids that may be decanted back into the adjacent water body during disposal (as the site is being filled), compared to unconfined aquatic disposal at deeper water sites. The ability to address the potential contaminant exposure pathways at nearshore sites also falls between that of upland and open water sites, as discussed below.

3.2.4.4 Ability to Take Corrective Site Management Measures in Different Placement Environments

Most of the impacts that can potentially be associated with dredged material placement are best addressed *before* disposal occurs, by selecting an appropriate site. Sites that avoid sensitive resources, that have few potential contaminant pathways of concern, and/or that include features to help control the potential pathways, will minimize initial impacts as well as reduce the need to take corrective measures later. Nevertheless, a variety of tools or management responses are available at any placement site if corrective measures are found to be necessary *after* dredged material has been disposed.

The ability to take corrective measures, should a concern arise at a dredged material placement site, varies among the placement environments. If the concern is an unacceptable level of contamination, removing problem material is rarely feasible under any circumstances at open water sites. Instead, capping (and if necessary, armoring the cap against erosion by placing coarser material on top) is often the only feasible means of isolating material of concern once it is on the bottom. Even this is generally only practical at non-dispersive sites. At dispersive sites little can be done, because in most cases it will not be possible to determine where problem material from the site has ultimately settled.

In contrast, at upland sites, there is generally the ability to take several steps (including re-excavation of the problem material and re-engineering the site, or removing the material to a new site) if unexpected problems arise. Actual steps taken would depend on the site and the specific problem identified (see section 3.2.4.5).

Figure 3.2-27 Contaminant Pathways for Nearshore Confined Disposal Facilities

At nearshore sites, control of return water quality and quantity is also generally feasible. And, should serious problems develop so that it becomes necessary to remove the sediments and re-engineer the site, access for heavy equipment is possible. However, especially if the nearshore site was used for habitat restoration, or has otherwise developed important habitat values, there may be more ancillary consequences of correcting problems at a nearshore site (e.g., potential release of contaminants into adjacent areas) compared to an upland site.

If the problem needing correction at an open water site is not related to contaminants but is instead physical in nature, various management actions are possible. For example, as discussed in Chapter 2 (section 2.2), mounding has been an ongoing problem at the Alcatraz site, and it has taken active management by the COE regarding the timing, rates, locations within the site, and methods of disposal to keep this problem from worsening. In the 1980s, the mound itself had to be physically re-dredged to reduce the navigational hazard posed by the site. In the early 1990s mounding problems reappeared and, in 1993, additional active site management steps were outlined in COE Public Notice 93-3. Under these measures, and given the relatively low volumes of material placed at the site in 1993 and 1994, dispersion appears to be keeping up with disposal so that mounding is not causing a navigation hazard today. At other dispersive in-Bay sites, the same degree of active management has not been necessary. At SF-DODS, mounding is not of significant concern, but potential off-site deposition of substantial quantities of dredged material would be. If this should be identified as a result of ongoing site monitoring, a variety of management actions are possible there, as well.

These have been identified in the final rule designating the site, and may include moving the surface discharge point within the overall site so that dredged material continues to deposit where desired; restricting the timing of discharges relative to currents; restricting the rate and/or volume of discharge so that significant off-site deposition does not occur; or discontinuing use of the site. However, removing or re-dredging deposited material in the vicinity of SF-DODS would most likely be infeasible.

Physical problems at upland and nearshore sites could include not achieving proper elevations at a habitat restoration site; or an upland site developing a fugitive dust problem during drying. These kinds of physical problems can generally be readily addressed at upland and nearshore sites. For example, before tidal action is allowed to return at a tidal wetlands restoration site, regrading can be done if necessary to achieve proper elevations for marsh vegetation; and fugitive dust can be controlled by standard operating procedures which require that the surface of drying dredged material be sprayed to keep it moistened.

3.2.4.5 Summary of Potential Management Actions and Control Measures for Contaminant Pathways of Concern

A variety of management actions are possible for cases where evaluation of contaminant pathways indicates that ecological impact criteria will not be meet using conventional disposal techniques. The primary consideration in selecting any of these control options is to identify the site-specific exposure pathway(s) of concern and to choose the management option that best addresses those exposure pathways. This section presents examples of the potential management actions and controls for the various exposures pathways associated with aquatic, upland, and nearshore disposal areas. These controls are summarized by contaminant pathway in Table 3.2-5. Where appropriate, these are reflected in the mitigation measures included as companion policies common to all action alternatives, presented in Chapter 5. For more detailed information on specific control measures, see the technical framework document for dredged material management (USEPA and USACE 1992).

Water Column Pathway Controls

In the limited circumstances where the water column pathway is determined to be of concern, there are several available control measures that may be applied to reduce potential adverse effects. Controls for water column effects at the dredging site include, for example, restricting the time or rate of dredging; requiring the use of silt curtains or closed "environmental" clamshell buckets; requiring the use of hydraulic dredges (which minimize mechanical disturbance at the dredging site but increase the volume of water and suspended material that must be managed at the *disposal* site); and prohibiting overflow from hopper dredges. Controls for water column effects at the disposal site include reducing water-column dispersion by using clamshell dredging with discharge from barges or submerged diffusers; constraining the location, rate, and timing of disposal; and placing dredged material in geotextile bags to reduce water column exposure during disposal.

Table 3.2-5. Potential Management Actions and Control Measures, by Contaminant Pathway

Water Column Pathway Controls	
At the Dredging Site	• Restricting the time or rate of dredging
	• Requiring the use of silt curtains or closed "environmental" clamshell buckets
	• Requiring the use of hydraulic dredges
	• Prohibiting overflow from hopper dredges
At the Disposal Site	• Reducing water-column dispersion by using clamshell dredging with discharge from barges or submerged diffusers
	• Constraining the location, rate, and timing of disposal
	• Placing dredged material in geotextile bags to reduce water column exposure
Benthic Pathway Controls	
• For some depositional sites, adjusting the surface release zone to ensure sediments are depositing at the desired bottom location	
• Using lateral containment measures (e.g., existing subaqueous depressions or constructed dikes) to restrict the bottom area affected by dredged material placement	
• Placing sediment in a thin layer over a wide disposal area can offset physical effects on benthos due to burial	
• At depositional sites, covering or capping the deposited material with cleaner dredged material	
Upland Pathway Controls	
• Managing settling time and discharge rates to improve return water quality	
• Treating return water (e.g., by flocculation)	
• Controlling seepage by using impermeable liners or retrofitting slurry walls around the site	
• Using leachate collection systems	
• Controlling dust by keeping the surface of the dredged material moist	
• Avoiding locating upland sites near resources that would be sensitive to odors	
Nearshore Pathway Controls (1)	
• If tidal transport of dredged material offsite is a concern, closing off the openings in the dikes and managing the area as a nearshore confined disposal facility may be possible	
Note: 1. All of the pathways and control measures listed for both the water column (aquatic) and upland disposal pathways might apply to nearshore disposal sites.	

Benthic Pathway Controls

A variety of modifications in dredged material placement operations can be instituted to control contaminant exposures to benthic pathways if monitoring shows that site performance is not optimal. For example, for some depositional sites, the surface release zone can be adjusted to ensure that sediments are depositing at the desired bottom location. Lateral containment measures, such as existing subaqueous depressions or constructed dikes, can also be used to restrict the bottom area being affected by dredged material placement. Conversely, thin-layer placement over a wide disposal area is a management action that may offset physical effects on benthos due to burial. Finally, if material is discharged at a depositional disposal site that causes unacceptable impacts (e.g., off-site toxicity, or bioaccumulation and food web effects), the deposited material of concern can often be covered or capped with cleaner dredged material, reducing the exposure of organisms to it.

This measure is, however, rarely possible at dispersive sites. Generally, depositional sites can be more effectively managed to minimize contaminant-related risks associated with dredged material than can dispersive sites.

Upland Pathway Controls

There are several possible migration pathways of contaminants out of upland disposal sites, including effluent discharges to surface water, surface runoff, leachate into groundwater, and air quality effects from volatilization or fugitive dust. Several measures exist to minimize exposures from these pathways, including

managing settling time and discharge rates for the site to improve return water quality; treating return water (e.g., by flocculation); controlling seepage by using impermeable liners or retrofitting slurry walls around the site; using leachate collection systems; controlling dust by keeping the surface of the dredged material moist; and avoiding locating upland sites near resources that would be sensitive to odors.

Nearshore Pathway Controls

All of the pathways and control measures described above for both the aquatic and upland disposal pathways might apply to nearshore disposal sites. One additional control measure would apply to nearshore facilities that are subject to tidal action (e.g., the proposed Montezuma Wetlands project). In cases where tidal transport of dredged material offsite becomes a pathway of concern, it may be possible to close off the openings in the dikes and manage the area as a nearshore confined disposal facility.

3.2.5 Role of Sediment Evaluations (Testing)

A major purpose of sediment quality testing is to assess whether the bioavailability of and exposure to contaminants in a specific dredged material have the potential to adversely effect sensitive, representative organisms at the disposal site. As required under both the CWA and the MPRSA (the "Ocean Dumping Act"), the EPA and COE have set forth consistent, standardized procedures for evaluating potential effects associated with dredged material disposal at open water sites, and at certain beneficial use sites. These evaluations focus on the specific exposure pathways and biological endpoints of concern, and should provide sufficient information for decisionmaking. In some cases, a rigorous testing regime is required to adequately characterize the ecological risk associated with a particular dredged material. This level of effort is often necessary because, for most unconfined disposal sites, once dredged material is disposed it is difficult to control the exposure of organisms to the material, or to remove it or re-engineer the site to rectify any adverse impacts.

Sediment testing is a key aspect of ensuring that unacceptable adverse effects do not occur as a result of dredged material disposal at a particular location. For proper site management, testing must be used in conjunction with appropriate *interpretation standards* (which will differ for different disposal methods or sites); disposal activity must follow all *site use requirements* (such as specific timing or volume restrictions that may be placed on specific sites); and site performance must be confirmed by appropriate *site monitoring*. Each of these are essential aspects of an overall Site Management and Monitoring Plan.

Sediment testing is, however, only one element of the overall decisionmaking process for determining whether a permit will be issued for a proposed discharge of dredged material. A range of other requirements must also be met. For example, under Section 404 of the CWA the proposal generally must be shown to be the least environmentally damaging alternative that is "practicable" to perform. Thus, if beneficial reuse options that would have less adverse environmental impacts are available and otherwise practicable to a project proponent at a given time, unconfined aquatic disposal would not be permitted even though the dredged material was shown to meet aquatic disposal standards.

The following sections provide a general background on what is involved in sediment testing and how the results are used in making suitability decisions for the disposal of dredged material. Detailed descriptions of past and current testing practices specific to the San Francisco Bay area are included to illustrate the more important regional issues and considerations.

3.2.5.1 Testing for Aquatic Disposal

Sediments may contain contaminants that, if bioavailable and present in elevated concentrations, can cause adverse environmental impacts. Dredging and dredged material disposal activities may release or redistribute these contaminants in the aquatic environment. Nationwide, the majority of dredged material disposal occurs in inland and near coastal waters (and thus falls under the jurisdiction of the CWA). As mentioned earlier, it is often difficult to control exposures of organisms to dredged material at unconfined aquatic disposal sites; this is particularly true at dispersive locations. While capping with additional, clean dredged material is usually the main corrective measure that can be taken at depositional sites, capping is usually impractical and/or ineffectual at dispersive sites. Therefore, for unconfined aquatic disposal in general, and for disposal at dispersive sites in particular, it is especially important that a comprehensive sediment evaluation be conducted to ensure that any potential for adverse effects is identified.

In this section, we describe the fundamental regulatory and technical bases of dredged material testing for aquatic disposal. This testing uses a tiered, effects-based, and reference-based evaluation structure to

make suitability decisions that are based on adequate information, that address appropriate exposure pathways, and that are as cost-effective as possible in collecting the information required. Although the specific tests (e.g., species and endpoints), interpretation values (e.g., suitability criteria), and degree and frequency with which testing is needed (e.g., full chemical and biological testing) may change as more information becomes available, the basic framework for dredged material evaluations should remain the same.

Conceptual Framework

The overall framework for evaluating dredged material management alternatives is provided in the EPA/USACE document. *Evaluation Environmental Effects of Dredged Material Management Alternatives — A Technical Framework* (USEPA and USACE 1992). Comprehensive guidance regarding sediment quality testing for offshore (i.e., ocean) disposal is established by the Ocean Dumping regulations (40 CFR Part 227) and provided in detail in the joint EPA/COE ocean disposal testing manual, titled *Evaluation of Dredged Material Proposed for Ocean Disposal — Testing Manual*, popularly known as the Green Book (USEPA and USACE 1991). Similar comprehensive national sediment testing guidance for inland waters was recently published, titled *Evaluation of Dredged Material Proposed for Discharge in Waters of the United States — Inland Testing Manual* (USEPA and USACE 1998). Both the Green Book and the ITM are tiered under the Framework Document (as well as under their respective legislation and regulations).

Ocean Disposal

Since it was finalized in 1977, all dredged material testing for ocean disposal has followed the comprehensive guidance laid out in the Green Book. (The most recent update to the Green Book was conducted in 1991.) Procedures outlined in this manual are designed to meet basic MPRSA requirements for evaluation of potential contaminant-related impacts that may be associated with the discharge of dredged material at marine disposal sites. The Green Book uses a testing approach that is effects-based, reference-based, and tiered (a detailed description of each of these concepts is given below). This approach is designed to ensure that adequate information is generated to satisfy regulatory requirements, without forcing applicants to incur unnecessary testing expense.

The evaluation procedure outlined in the Green Book begins with determining whether testing is even necessary based on the availability of sufficient existing information. If existing data are inadequate to serve as the basis for a suitability determination, additional steps must be taken to collect the necessary information. The following discussion will focus on those subsequent steps that involve chemical and biological testing of dredged material. The testing framework outlined in the Green Book involves three basic components:

1. To evaluate the degree of contamination using bulk chemical analysis of sediments;

2. To determine acute toxicity in the water column and sediment using suspended-phase (elutriate) and solid-phase (whole sediment) bioassays; and

3. To evaluate the potential for bioavailability of compounds that may lead to chronic and/or sublethal effects, or effects at higher trophic levels, using solid-phase bioaccumulation tests.

The degree of testing for any given project is based on several factors: a reason to believe that the sediments may be contaminated (as determined in the tiered evaluation process discussed below), the size of the dredging project, the nature of the proposed disposal site (e.g., dispersive or non-dispersive), and the nature of nearby resources that may be affected by the disposal. The extent and nature of the testing performed will also depend on the exposure pathways of concern at the disposal site relative to the contaminants of concern in the dredged material.

In-Bay Disposal

Although sediment testing guidelines for disposal at aquatic sites within San Francisco Bay have evolved considerably over the past decade, testing has historically been less comprehensive than the requirements for ocean disposal. Early guidelines for sampling and testing of sediments for disposal within San Francisco Bay were provided in the COE Public Notice (PN) 78-1 (released on July 30, 1978) and later in PN 87-1 (released in June 1987 by the COE, EPA, and Regional Water Quality Control Board [RWQCB]). Routine testing requirements outlined in PN 87-1 were limited to bulk sediment chemistry and a single elutriate bioassay for state water quality certification purposes. Furthermore, under these testing guidelines, reference samples (used as the point of comparison for determining whether sediments to be disposed are “clean” enough) were taken from the disposal site itself for comparison with the proposed dredged material (see section on Reference-Based Testing below for further discussion of reference

sampling issues). Because one project's disposed material became the next project's "reference," overall contamination levels at the Alcatraz site increased over time, to the point that "reference" samples for later projects were themselves toxic to marine organisms in bioassay tests.

The acknowledged limitations of PN 87-1 testing led to the preparation of interim testing guidelines for in-Bay disposal presented in the COE PN 93-2 (released jointly by the COE, EPA, San Francisco Bay Conservation and Development Commission [BCDC], and RWQCB in February 1993). PN 93-2 was a significant improvement over the approach taken in PN 87-1, because it moved the reference sampling site for Alcatraz from the disposal mound itself to an "environs" area contiguous with the site but off the mound of previously dumped dredged material. The environs approach was intended to stop the documented degradation of Alcatraz that was worsened by using the site itself as the reference. PN 93-2 also expanded the routinely required bioassay testing to include the benthic exposure pathway, using a solid-phase amphipod test. However, even this increased level of testing remained less comprehensive in comparison with that required for ocean disposal. For example, under PN 93-2 only one bioassay test species each is required for the water column and benthic exposure pathways. Furthermore, bioaccumulation testing is only required in special circumstances when acute exposures do not provide sufficient information to evaluate the potential impacts of the dredged material.

PN 93-2 testing guidelines were explicitly published as interim measures, and apply to dredged material testing for in-Bay disposal only until superseded by implementation of the recently published EPA/COE national testing manual for inland waters, titled *Evaluation of Dredged Material Proposed for Discharge in Waters of the U.S. — Inland Testing Manual* (USEPA and USACE 1998). The Inland Testing Manual (ITM) updates and replaces the 1976 COE document, *Ecological Evaluation of Proposed Discharge of Dredged or Fill Material into Navigable Waters*, and adopts the same basic framework as the Green Book, including the tiered testing approach, multi-species benthic and elutriate testing, and 28-day bioaccumulation testing. It is expected once a separate rulemaking process is completed, that it will also include comparison of benthic test results with those of an off-site reference sediment. Both EPA and the COE have acknowledged earlier inconsistencies in testing requirements between inland and ocean environments and that there is a need for comprehensive evaluation wherever dredged material is disposed. At the time of writing this EIS/EIR, the draft ITM has been circulated for public comment, and it is expected that the final version will be implemented in 1998. Now that the ITM has been adopted as a national testing guidance for inland waters, the agencies will prepare a Regional Implementation Manual (RIM) that will draw upon both the Green Book and ITM guidance to provide detailed dredged material testing requirements for San Francisco Bay area projects. However, the overall testing framework included in the ITM and Green Book, as described in the following discussions, will be reflected in any such regional guidance.

Effects-Based Testing

Effects-based management (as opposed to management based on pre-existing numerical standards) involves bioassay testing using sensitive aquatic organisms as an indication of whether contamination associated with dredged material may cause adverse biological effects. Biological evaluations are particularly important for sediments because chemical measurements alone are usually inadequate to predict the bioavailability, and therefore toxicity, of sediment associated contaminants (see, for example, Power and Chapman 1992; Long and Chapman 1985; Lamberson et al. 1992; Hoffman et al. 1994). It is well documented that a given bulk concentration of contaminant(s) may be toxic in one sediment and not in another due to a variety of abiotic variables governing bioavailability (such as partitioning into pore water, the chemical form of the compound, the presence of other ions, organic content, and oxidation state of the sediment) (USEPA 1993b). Biological effects testing provides an important complement to chemical analysis because it gives a direct measure of organism response, integrating the biological and chemical interactions of the suite of contaminants that may be present in a dredged material sample (USEPA and USACE 1994).

The effects-based framework for testing presented in the Green Book/ITM is based on multi-species testing using appropriately sensitive organisms. In order to adequately assess the possible impact of contaminants on aquatic communities, it is recommend that testing be performed using a suite of species to account for the variable sensitivity among organisms for different chemicals. Currently, use of at least three sensitive species is recommended for the water column (elutriate), and at least two species for the whole sediment exposure pathways. The general types of bioassays used to evaluate these pathways are discussed below.

Water column (elutriate) toxicity tests are designed to mimic the short-term exposures in the water column that are associated with active dredging and disposal operations. There are standardized protocols of the American Society for Testing and Materials (ASTM) for numerous species and endpoints including bivalve and echinoderm larval development, and survival of mysid shrimp and juvenile fish (ASTM 1989; Ward et al. 1995). Elutriate results are used primarily to evaluate compliance with state water quality standards and federal water quality criteria, after allowing for appropriate mixing at the disposal site. In addition, these tests provide useful information in the overall evaluation of potential sediment toxicity.

Generally, the greatest potential for environmental effects from the disposal of dredged material is associated with the benthic exposure pathway (see section 3.2.4). Bottom dwelling (benthic) animals living and feeding on or in deposited material for extended periods represent the most likely pathways for adverse ecological effects from contaminated sediment. Thus, the emphasis of dredged material evaluations is usually on estimating effects associated with exposure of benthic organisms to contaminants in bedded sediment. Acute toxicity to various benthic species is used as a measure of the potential for direct effects to exposed organisms, while tissue bioaccumulation is a measure of the bioavailability and thereby the potential for chronic or food web effects (including human health effects from eating contaminated seafood) of sediment contaminants in longer-term exposures (see discussion of Tier III under Tiered Testing below for further information on these tests).

Reference-Based Testing

Reference-based testing refers to the practice of comparing biological effects and chemical data from the dredged material to those from a reference sediment selected to represent an appropriate and acceptable level of environmental quality at the disposal site. What is appropriate to use as a reference will differ depending upon the nature of the disposal site, the sediments being tested, and the disposal site management approach. In general, reference sediment is a sediment that is substantially free of contaminants, that is as similar as practical to the grain size of the dredged material and the sediment at the disposal site, and that reflects the conditions that would exist in the vicinity of the disposal site had no dredged-material disposal ever occurred (USEPA and USACE 1994). For depositional sites, the reference should be located in an environment similar to but out of the influence of the disposal site itself, whereas for dispersive sites, the reference should represent the off-site area in which the dredged material ultimately deposits. In the latter case, the reference may be a site representing the changing conditions of the general water body (e.g., the most appropriate reference for Alcatraz and other in-Bay sites may be site[s] that reflects ambient conditions in each embayment).

The Ocean Dumping Program has always used an off-site reference as a comparison for suitability determinations. In contrast, the Section 404 program has in the past required that reference samples be collected from the disposal site itself. One problem with the onsite reference approach is that ongoing disposal will (by definition) create different reference conditions for every project. Over the long term, comparisons made to an ever-degraded reference can lead to increased site degradation. Exactly this problem occurred at the Alcatraz disposal site in recent years, where chemical and biological testing performed as part of the permitting program indicated markedly increased levels of contamination and acute toxicity in reference samples taken from the disposal mound. This led, in turn, to even more contaminated sediments being authorized for disposal at the site as the next project would effectively be compared against the prior project's sediment as the new reference condition. To address ongoing degradation at Alcatraz, the agencies redefined the reference for Alcatraz, in PN 93-2, to be a series of stations located outside of the disposal mound (the "Alcatraz Environs"). However, the environs reference is still influenced by sediments disposed at the Alcatraz site, and does not necessarily appropriately reflect background conditions in the central Bay.

Another issue associated with reference testing at in-Bay locations is that currently used reference sediments often have grain-size and other critical physical characteristics (e.g., organic carbon content) that are very different from those of the dredged material being tested. Thus, it is often the case that chemical and biological testing results from dredged material having a high silt and clay component are compared to results from a reference sample that is primarily comprised of sand. Currently, there are several efforts underway to identify and characterize in-Bay sites that could serve as references for natural background conditions in regional monitoring programs. The RWQCB, for example, through the BPTCP, has been conducting a Reference Site Study to identify and characterize fine-grain sediment reference sites in the Bay. Furthermore, the Regional Monitoring Program has been measuring sediment toxicity throughout the Bay at sites with a range of grain sizes, several of which may be able to be used as reference sites for dredged material evaluations.

Tiered Testing

The tiered approach to sediment testing promotes cost-effectiveness by focusing the least effort on disposal operations where the potential (or lack thereof) for unacceptable adverse impact is clear, and expending the most effort on those operations requiring more extensive investigation to characterize the potential for impacts. For any particular project, it is necessary to proceed to more detailed (and expensive) testing in higher tiers only when the previous tier did not result in adequate information for a decision to be made. This following paragraphs summarize each of the tiers as they are currently described in the Green Book and ITM.

TIER I. "Tier I" involves the examination of readily available, existing chemical and biological information (including that from all previous sediment testing) to determine whether there is a reason to believe that the dredged material needs to be tested for potential adverse effects. Information that may be considered as part of the Tier I evaluation includes recently collected chemical and biological data from the site and/or adjacent areas, known sources of contamination such as discharges or spills, and information on changes in land use adjacent to the site that might influence sediments (USEPA and USACE 1994). Some dredged material will be excluded from any need for testing when there is no reason to believe that it would be a carrier of contaminants.[4] Such dredged material typically is characterized by large particle size (sand, gravel, or rock), and is found in areas of high current or wave energy that are far removed from known existing and historical sources of pollution. Although most dredged material from San Francisco Bay does not meet exclusion criteria, it may not require testing if existing information is adequate to determine suitability. Furthermore, information collected at this tier may also be used for the identification of contaminants of concern relative to any testing performed in later tiers.

TIER II. Evaluations performed under "Tier II" provide screening information based on sediment and water chemistry data. Specifically, this can include evaluating compliance with state water quality standards using a numerical mixing model and estimating the potential for benthic impacts due to nonpolar organic chemicals using a calculation of theoretical tissue bioaccumulation. Though this screening information is useful for focusing additional testing efforts, at present, it is not generally adequate by itself to support suitability determinations for aquatic disposal (USEPA and USACE 1991, 1994).

When national sediment quality criteria (SQC) or state sediment quality standards or objectives for individual chemicals are proposed and finalized, they are expected to be incorporated into Tier II benthic impact evaluations. Comparison of sediment chemical data to numerical sediment criteria may be useful as a screening tool to streamline any additional testing required (as is currently practiced by the Puget Sound Dredged Disposal Analysis (PSDDA) program in Washington). However, due to the complexities of sediment/chemical/organism interactions and the potential for unpredictable interactive effects of contaminant mixtures, numerical criteria are not expected to completely replace effects-based testing, including bioassays (*Federal Register*, January 1994). Currently, national SQC have been proposed for only five priority pollutant chemicals (endrin, dieldrin, fluoranthene, phenanthrene, and acenaphthene). Site-specific numerical screening criteria (Apparent Effects Thresholds [AETs]) are currently being developed by the RWQCB for San Francisco Bay sediments and may be potentially useful for screening dredged-material to determine the level of testing required (see also discussion in section 3.2.5.3).

TIER III. If the evaluation of existing information and standards is not adequate to determine dredged material suitability, a "Tier III" evaluation is necessary. This tier is comprised of comprehensive chemical and biological testing of the sediment proposed for discharge to assess the potential effects of contaminants on appropriately sensitive and representative organisms. Standardized bioassay tests are available for many different aquatic species, representing various feeding/life strategies and biological effects endpoints of concern. Detailed presentation of these protocols can be found in ASTM (1990), USEPA (1994b), and Ward et al. (1995) methods manuals. Measured effects endpoints include acute toxicity in both sediment and water column exposures, and the bioaccumulation of contaminants in tissue. Although chronic/sublethal tests for sediments are under development, none are yet considered suitable for routine use nationwide.

Because dredged material potentially contains a myriad of contaminants that may adversely impact aquatic organisms, testing using a suite of species is necessary to fully assess the potential impact of dredged material on the aquatic community. A minimum of two sensitive species, together representing the three important functional characteristics (filter feeder, deposit feeder, and burrower), are recommended for the water column and whole sediment toxicity tests. There is flexibility in the guidance to tailor the choice of which tests to perform based on the exposure pathways

of concern at the proposed disposal site. Within the constraints of experimental conditions and the effects endpoints measured, these biological evaluations provide for a quantitative comparison of the potential effects of dredged material to the reference station. Generally, dredged material is considered unsuitable for unconfined aquatic disposal when test organism mortality is statistically greater than reference and exceeds mortality in the reference sediment by at least 10 percent (20 percent for tests using amphipod species).

Body burdens of chemicals are of concern for both ecological and human health reasons. To assess the potential for contaminants to bioaccumulate, 28-day tests have been developed using two species having adequate tissue biomass and the ability to ingest sediments. It is important to remember that tissue bioaccumulation itself is not an adverse impact. Rather, bioaccumulation is used as an indication of the bioavailability of sediment-associated contaminants. Concentrations of contaminants of concern in tissues of benthic organisms are compared to applicable Food and Drug Administration (FDA) and other human health standards such as local fish consumption advisories. Such comparisons are important, even though the particular test species may not be a typical human food item, because certain contaminants can be transferred through aquatic food webs, and because uptake to designated levels of concern may indicate the potential for accumulation in other species (USEPA and USACE 1994). The residue-effects information that would facilitate direct ecological evaluation using bioaccumulation data is not available for many contaminants of concern.[5] Consequently, the following additional factors are considered in determining the potential for adverse impacts associated with benthic bioaccumulation: toxicological importance of the bioaccumulated contaminants, magnification over reference, number of contaminants and the magnitude of their bioaccumulation, and the propensity for contaminants to biomagnify within aquatic food webs.

TIER IV. For the majority of projects, Tiers I-III are expected to be adequate for determining the suitability of dredged material for unconfined aquatic disposal. In those cases when lower tiered testing is judged to be insufficient to make complete factual determinations, then a special, project-specific "Tier IV" evaluation is necessary. Tier IV involves non-routine sampling or testing, designed to provide specific information that could not be obtained from application of the routine methods in Tiers I-III. For example, toxicity determinations in this tier can involve more intensive laboratory or field testing, or field assessments of resident benthic communities. Recently developed procedures such as toxicity identification evaluations (TIE) and chronic tests may be used in this tier. In all cases, a Tier IV evaluation will generate the specific information needed for decisionmaking; there is no tier beyond Tier IV.

Additional Sampling and Analysis Considerations

Collecting representative samples involves detailed site-specific consideration of the material to be dredged. Careful consideration of numerous factors should be given in the sampling scheme for any project, including historical data, sediment heterogeneity, dredge depth, volume to be dredged, number and geographical distribution of sites, and potential sources of pollution. Minimum sediment sampling guidelines were outlined in PN 93-2 (to be updated in the RIM) that indicate the number of samples that should be collected for a project of a given volume. Compositing sediment samples from an area into a smaller number of samples is allowed for testing purposes. However, samples should only be composited together when they are from a contiguous portion of the project area and when there is reason to believe that these sediments are exposed to the same influences and pollutant sources. To ensure appropriate sampling, all sampling and analysis plans should be coordinated with the appropriate agencies before any sampling or testing begins.

Physical and chemical tests are conducted at a minimum on each composite sample. Detailed guidance on sampling and analysis procedures is given in the Green Book and ITM. Currently, routine sediment physical and chemical analysis is performed for the list of contaminants in Table 3.2-6. Chemicals appear on this list based on their toxicological significance, persistence, and presence in San Francisco Bay sediments.

Table 3.2-6. Routine Sediment Physical and Chemical Analysis

Parameter	*Target Detection Limit (1)*
Conventionals	
Grain size	NA
Total organic carbon	0.1 percent
TRPH	20
Total volatile solids	0.1 percent
Total and water soluble sulfides	0.1
Total solids/water content	0.1 percent
Metals	
Silver	0.1
Arsenic	0.1
Cadmium	0.1
Chromium	0.1
Copper	0.1
Mercury	0.02
Nickel	0.1
Lead	0.1
Selenium	0.1
Zinc	1.0
Organic Compounds	
Phthalate esters	0.01
PAHs (2)	0.02
PCBs (3)	0.02
Pesticides (4)	0.002
Butyltins (5)	0.001

Notes: 1. Reported as mg/kg dry weight, unless otherwise noted.
2. All compounds on EPA Method 610 list.
3. Reported as Arcolor equivalents 1242, 1248, 1254, 1260, and total PCB.
4. All compounds on EPA Method 608 list.
5. Mono-, di-, and tributyltin.

Based on Tier I information, the required testing for any particular project may include additional chemicals or, conversely, fewer than those listed in Table 3.2-6.

3.2.5.2 Testing for Upland Disposal

This section describes current requirements, and a more systematic testing framework under development by the LTMS agencies, for disposal of dredged material at upland sites. A variety of additional upland/wetland sediment tests have also been developed nationally, and are available for non-routine, site-specific use. These include tests on effluent discharge quality (to evaluate the need for controls on return water); tests to estimate surface runoff quality (to evaluate the need for runoff controls such as collection and treatment); tests to estimate leachate quality (to evaluate the need for controls to address the potential for groundwater contamination); and upland plant and animal bioassays (if projected land use is such that this is a concern). Development of detailed engineering designs for specific new upland sites is outside the scope of this EIS/EIR. The interested reader is referred to USEPA and USACE (1992) and the references contained therein for further information about these non-routine tests.

Current Upland Testing Practice

In general, upland testing needs differ from aquatic testing because geochemical conditions in the two placement environments differ, and because there are different potential exposure pathways. For disposal at landfills and other upland sites, an important concern is with contaminants that may become soluble and mobilize into groundwater or surface water. In general, the soluble portion of contaminants is a small fraction of the total contaminant load. Unfortunately, there is no way to easily predict the soluble portion of contaminants from the measured total concentrations. Therefore, since typical aquatic disposal tests measure only total metals concentrations, data from aquatic testing programs are often inadequate for determining the suitability of dredged material for upland or landfill disposal.[6] Instead, landfill disposal testing guidelines generally require that if the concentration of contaminants measured using total metals analysis methodology (which measures all forms of the metal present, including soluble and non-soluble) exceeds the Soluble Threshold Limiting Concentration (STLC) hazardous waste numerical criteria by a factor of 10,

then substantial concentrations of the metal *may* be soluble and direct measurement of the actual soluble fraction is required.

For dredged material to be disposed at an upland site such as a landfill, it generally must be tested under current landfill testing criteria developed to address material from contaminated soil sites including leaking underground storage tank sites. Tests that are typically required include total and soluble metals, and total organics including BTEX, PCBs, pesticides, chlorinated solvents, and total recoverable petroleum hydrocarbons (TRPH) as waste oil or diesel. Such tests are often required by landfills in addition to, or without any review of the information available from previous testing, and without specific consideration of the differences between dredged material and upland soils. For example, a number of the contaminants routinely tested for are highly volatile, and it is unlikely that they would occur at elevated concentrations in sediments.

The environmental concerns regarding the placement of dredged material at upland sites cannot be addressed generically. Each type of placement environment represents a unique set of concerns. Consequently, project sponsors must work independently with the various agencies involved to develop project-specific testing protocols, sampling frequencies, and Waste Discharge Requirements for each project. The types of tests required and the sampling frequencies also vary with each landfill. There currently are no standard tests used by all landfills for acceptance of any kind of waste. This is largely due to the engineering differences at each landfill. In most cases, a discharger has been required to take a given number of samples and conduct a modified statistical analysis using methods outlined in EPA guidance (EPA 1990) to show that the material (1) is not hazardous; and (2) meets the landfill's specific acceptance requirements. Landfill acceptance criteria are determined by the landfill and approved by the relevant agencies based on the landfill's attenuation factors, and the landfill's proximity to groundwater (especially drinking water aquifers).

Proposed "LTMS Sediment Classification Framework"

As a basis for the establishment of regulatory guidance more specifically tailored to dredged material placement in upland environments, the LTMS agencies have developed a draft comprehensive Sediment Classification Framework that describes the suitability of dredged material for different kinds of disposal options, based on degree of contamination. Under this system, the least contaminated material is (chemically) suitable for the broadest range of disposal options, while the most contaminated material (meeting established hazardous waste criteria) must receive very specific handling. Appendix F presents this draft Sediment Classification Framework. It shows the general relationship between material that is "suitable for unconfined aquatic disposal" (SUAD material) or "not suitable for unconfined aquatic disposal" (NUAD material), and the various existing solid waste categories that apply to upland disposal or reuse. Appendix F also shows how these categories relate to the three existing "classes" of landfills.

The draft Sediment Classification Framework does not represent new regulation. Instead, it is a presentation of how the existing laws, policies, and definitions affecting dredged material disposal relate to each other. The Sediment Classification Framework can, however, serve as a useful basis for development of more consistent dredged material management policies, particularly with respect to testing and approval of material proposed for placement in upland disposal or reuse sites such as existing landfills.

3.2.5.3 Testing for Nearshore Disposal

Nearshore sites can have exposure pathways similar to both aquatic and upland sites (see section 3.2.4.3). Therefore, testing for placement in nearshore sites can involve some of the aquatic and upland tests described in sections 3.2.5.1 and 3.2.5.2, respectively. The specific tests needed will depend on site-specific issues of concern. Currently, there are no nationally standardized tests specific to nearshore environments that are appropriate for routine regulatory program use. However, in the San Francisco Bay area, interim screening guidelines have been developed by the RWQCB for wetland placement of dredged material (Wolfenden and Carlin 1992). The RWQCB interim screening guidelines use a combination of chemical screening levels and bioassay testing to identify when dredged material may be acceptable for use in nearshore disposal or reuse sites; in particular, the interim guidelines specify when dredged material can be considered for either "wetland cover" or "wetland noncover" placement. In general, SUAD sediments are considered appropriate for "wetland cover," while NUAD sediments must be isolated from the aquatic environment as "non-cover" material. Officially-designated hazardous waste, and other highly contaminated sediments, generally do not qualify for either "cover" or "non-cover" placement in nearshore wetland restoration sites.

A variety of other upland/wetland sediment tests have been developed nationally that can be used in non-routine, site-specific circumstances. These includes tests on effluent discharge quality, tests to estimate surface runoff quality, tests to estimate leachate quality, and upland or wetland plant and animal bioassays. These kinds of tests are not typically used for routine regulatory program purposes because they tend to be more appropriate for research or "Tier IV" applications, and because they tend to be too expensive and time-consuming to conduct except in association with large projects. Nevertheless, such testing has occasionally been conducted for projects in the San Francisco region. The interested reader is referred to USEPA and USACE (1992) and the references contained therein for further information about these non-routine tests.

3.2.5.4 Opportunities to "Streamline" Testing Needs

As indicated earlier, it is not expected that the basic sediment evaluation framework or approach for dredged material (discussed in sections 3.2.5.1 and 3.2.5.2) will fundamentally change over time, due to the need for comprehensive evaluation that considers site-specific exposure pathways and project-specific contaminants of concern. However, this framework provides for substantial flexibility to address local concerns, and local experience that is accumulated over time. There are many possibilities for streamlining the sediment evaluation process, from both an overall management standpoint and a project-specific standpoint. Some of the possibilities presented below are already in practice or in the early stages of development in the San Francisco Bay area. These streamlining options, and others that may be identified in the future, would be implemented through the periodic review process for the LTMS Management Plan, as discussed in Chapter 7.

Development of an Interagency Dredged Material Management Office (DMMO) to coordinate decisionmaking relative to dredging permits (e.g., combined application, sampling and analysis plan approval, suitability determination, disposal options). The goal of the DMMO would be to establish a permitting framework that reduces redundancy and unnecessary delays in permit processing and increases consensus decision-making among staff of the member agencies (COE, EPA, RWQCB, BCDC, and State Lands Commission). Thus one result of the DMMO approach would be to increase consistency regarding when and how much testing is required of applicants. Another product of the DMMO would be a combined database to share regulatory and technical information among the agencies, applicants, and interested parties. One important long-term goal of the DMMO that would significantly streamline permit coordination is the creation of a single, interagency dredge permit.

A consolidated Regional Implementation Manual (RIM) for the testing of dredged material for aquatic disposal will be developed by EPA and the COE, with the input from other regulatory agency. Under this RIM, required biological and chemical testing will be consistent for disposal in both ocean and in-Bay environments.

More systematic use of the tiered approach to dredged material evaluation will be included in the RIM, based on the GB/ITM. Thus there will be less testing needed for some individual projects, once a multi-year track record (Tier I) has been established for them demonstrating consistently clean material (e.g., yearly channel maintenance projects). Development of appropriate numerical sediment quality screening values (Tier II) could help to minimize the volume of sediment that must be tested using Tier III bioassays (e.g., San Francisco sediment quality criteria values are currently under development by the SFBRWQCB for use on a regional basis). In addition there can be more systematic application of available models as an affordable screen for potential ecological risk (e.g., calculation of Theoretical Bioaccumulation Potentials from sediment chemistry samples can minimize the need for more costly bioaccumulation testing).

Improve coordination of upland testing requirements. Agencies have already made progress toward more consistent upland testing requirements. The Sediment Classification Framework (section 3.2.5.2) could also serve as a basis for further streamlining by other agencies. For example, the state Integrated Waste Management Board could consider the equivalent of a general permit for the use of certain defined categories of dredged material as an alternate source of daily cover at landfills.

Over-design any new sites to minimize testing. Locate and design placement sites so that exposure to potential contaminants is already controlled to reduce testing needs. For example, less testing would need to be conducted by individual project proponents if their material is proposed to be used for landfill daily cover, wetland non-cover, or in confined aquatic disposal sites where most pathways of concern were already addressed in the design of the disposal site.

3.2.6 Management of Contaminated Dredged Material

This section discusses the kinds of management options that are appropriate for handling contaminated (NUAD-class) dredged material. These discussions apply to sediments that are not classified as Hazardous Waste; remediation or management of in-place sediments that have Hazardous Waste levels of contamination is outside the scope of this EIS/EIR.

Appropriate dredged material management involves a comprehensive evaluation of sediment quality, available disposal or placement options, control measures tailored to address specific issues of concern (project-specific contaminants of concern, site-specific exposure pathways), monitoring needs, and the ability to take corrective site management actions if necessary. It is important to keep in mind that the presence of contaminants *per se* does not automatically mean that a sediment is unsuitable for a particular disposal option. As discussed in section 3.2.3.4, the great majority of sediments dredged from the San Francisco Bay/Delta Estuary would not pose a threat of significant adverse effects at most potential disposal sites, even though many of these sediments contain levels of contaminants that are somewhat elevated over natural "background" and basin-wide ambient values. However, when sediment contamination is high enough to require specific management, it is important that appropriately designed sites are available.

There are currently few multi-user sites available for the disposal of contaminated sediment. No multi-user confined disposal facilities (CDFs) and no confined aquatic disposal (CAD) sites currently exist in the region. When portions of a dredging project are determined to be unsuitable for unconfined aquatic disposal (NUAD-class material), sponsors often have the option of retesting the material at a higher resolution (e.g., with more closely spaced sampling) in order to identify the minimum volume of material requiring confined disposal. Once the problem area has been delineated, in some cases the sponsor will elect to leave that material in place if the project can be made to function without dredging that particular location. Other sites require that the problem material be removed to facilitate the use of the site.

If NUAD-class material must be dredged, disposal opportunities are currently limited to upland disposal into landfills (such as the Redwood Landfill in Marin County), discharge into a confined upland site arranged for by the individual project sponsor (for example, one that can be established on their own property, such as the Port of Oakland's Galbraith site), or in some cases, reuse as fill in an otherwise approved construction project.

There are three main approaches that can be taken to manage dredged sediments that do not qualify for unconfined aquatic disposal. Each of these is discusses in the subsections that follow. The first approach discussed is isolation of the dredged material in a CAD site. The second is isolation of the dredged material at a confined upland disposal site. Confinement at properly designed aquatic or upland sites is generally technologically feasible and appropriate for management of dredged material that ranges in quality, and a number of disposal options are discussed under each of these general headings. The third option available for dealing with contaminated sediments is treatment to reduce contamination levels or to render the contaminants unavailable. Treatment can allow sediments that would otherwise require high-cost disposal to be suitable for lower cost disposal options. Although treatment is usually expensive, and in general is not feasible for large volumes of dredged material or material with relatively low concentrations of contaminants, it remains a viable option for small volumes of highly contaminated material. Again, a number of treatment options are discussed under the general treatment heading.

3.2.6.1 Confined Aquatic Disposal

Confined Aquatic Disposal (CAD) is a term used to describe the general category of options that relate to the sequestering of contaminated sediments in the aquatic environment, so that they are physically isolated from aquatic organisms and so that they remain in a saturated and chemically reduced state. In the CAD process, contaminated material is sequestered (usually by placing it in an environment that is low energy, or "depositional") and then capping the contaminated material with clean material so that it is isolated and aquatic organisms are not exposed to it. Several CAD projects have been successfully constructed internationally and around the country, including on the west coast in Los Angeles Harbor and Puget Sound. However, CAD has not been conducted to date in the San Francisco Bay area. The COE and EPA are currently finalizing a major national guidance document on CAD. This document, *Guidance for Subaqueous Dredged Material Capping* (Palermo et al. 1995) addresses many of the detailed siting, design, and environmental impact issues associated with CAD projects. In addition, the lead author of the national guidance document has prepared an evaluation paper for LTMS on issues specific to any consideration of

CAD in the San Francisco Bay area. This evaluation is presented as Appendix G, *Confined Aquatic Disposal (CAD) in San Francisco Bay — General Discussion of Environmental Impacts and Issues*. The following paragraphs provide an overview of some of the issues in Appendix G, and in the COE/EPA national guidance document.

Types of CAD

The options under this general heading include reuse as non-cover material in wetland creation/restoration projects, disposal into a confined site such as a submerged pit, depression, or other lateral confinement (true CAD sites), level bottom capping (CAD without structural lateral controls) and the creation of nearshore structures such as marine terminals, harbors, parks, or other fill projects where the sediments to be isolated will remain saturated and reduced. Nearshore CAD sites (such as tidal wetlands sites) can potentially be placed in high energy areas as long as the associated containment structure (marine wharf, breakwater, levee, etc.) is designed specifically for that environment.

Siting and Design Issues

There are a number of potential risks that must be addressed when considering CAD projects. These relate primarily to (1) whether an appropriate site has been chosen so as to minimize impacts to aquatic resources during construction and/or due to any loss of existing environmental values; and (2) whether all appropriate design and operational measures have been identified, considering the physical characteristics of the chosen site. It must be well documented that the site can adequately isolate the contaminated material, and that any change in contour caused by filling the CAD site will not change its character to an erosional one. In addition, the site must be set aside in an area that will remain free of dredging, shipping, mooring, or other activities that could compromise the ability of the cap to isolate the NUAD material at the site. Similarly, the engineering and initial site investigations must be rigorous and conservative to ensure that all appropriate design needs have been identified and incorporated. Long-term monitoring and management of the site may also be required. If the cap is found to be insufficient or failing over time, mechanisms must be in place to identify and rectify the problems.

Cap Design

A generalized cap design includes a 1-foot cover thickness as a chemical seal to prevent long term release through diffusion of contaminants, and an additional 2-foot cover thickness as a biological seal to prevent burrowing aquatic organisms from being exposed to the contaminated material. Mixtures of silt and clay in the initial foot of cap to act as an effective chemical seal and a sand final cap to help prevent erosion are often incorporated into cap designs at open water CAD sites. However, these are only general guidelines for cap design; site-specific information on the physical and biological environment is needed to determine the appropriate design criteria to take into account project-specific conditions (Palermo et al. 1995).

Material Appropriate for Use at a CAD Site

There is sometimes a public perception that CAD is the dumping of Hazardous Waste into the aquatic environment. This is not the case; in fact, Hazardous Waste must be handled in very specific ways (see section 3.2.5.2), and is not appropriate for disposal at CAD sites. However, NUAD Category I and II material, and in some circumstances NUAD Category III material, would generally be suitable for non-cover material in CAD sites (when NUAD Category III material contains high concentrations of soluble or highly toxic contaminants, it would not be suitable for CAD).

CAD requires that contaminated material be dredged and placed in a manner consistent with the environmental risk posed by the material. For example, if the contaminants in a material are shown to be leachable to the extent that water quality objectives would be exceeded during initial placement, then special precautions such as silt curtains, or placing the material into geotextile tubes prior to disposal, may be required. If application of the available management tools would not adequately minimize risks, the material would not be suitable for placement at the CAD site. Any CAD site proposed for the Bay area in future years would require that all appropriate siting and design studies be rigorously conducted, and it will be important to conduct a focused public outreach effort to identify and fully address public concerns.

Potential Benefits of CAD

Sequestering certain contaminants in the marine environment can be considered beneficial. Certain metals, for example, can become soluble and therefore available under the acidic conditions that can occur in landfills or other upland sites. Where disposal in the upland environment could result in acidic conditions, the buffering capacity of the estuarine environment would significantly reduce the risk that metals would be

released (see section 3.2.4). In addition, through the aerobic biological breakdown process, contaminants such as PCB and DDT can be converted to even more toxic intermediates than the parent compounds. Material sequestered in the aquatic environment undergoes mainly anaerobic degradation, which occurs much more slowly and can result in less toxic intermediates.

Once completed, in many cases CAD sites can be converted into beneficial habitat for fish and wildlife, including special status species. One example would include a multi-user CAD site that is designed to become a nesting island after capping. Another would be a CAD site that becomes a shallow water foraging habitat for the California least tern; such a project was recently constructed in the Los Angeles Harbor (Pier 400 Design Consultants 1995). Similarly, wetland habitat creation can be accomplished using NUAD material as non-cover material (a form of CAD), as has been proposed for the Montezuma Wetlands Project (USACE and Solano County 1994). Coupling habitat improvement with CAD would be consistent with existing plans and policies, and could serve the dual purpose of reducing contaminant risk while improving environmental quality by restoring or creating important habitats.

3.2.6.2 Confined Upland Disposal

Confined upland disposal is a term used to describe the general category of options that relate to the sequestering of contaminated sediments in the upland environment. Material is removed from the aquatic environment and sequestered in an upland site that is designed to manage the physical and chemical pathways associated with the material. The appropriate design for an upland disposal site depends on the extent of the contamination in the dredge material, the material's physical properties, and the location of the upland disposal site. Land placement of dredged material presents a set of testing and policy issues different from aquatic disposal, because the contaminant exposure pathways and other management concerns differ for upland sites. The action of removing NUAD-class dredged material from the aquatic system and placing it on land does not, by itself, necessarily reduce the potential for environmental impacts. Instead, land placement presents a new set of environmental concerns, associated with oxidation/acidification, dust and odor nuisances, and leaching of heavy metals and salts. On the other hand, land placement presents an opportunity to reuse dredged material in beneficial projects. These opportunities include reuse as daily cover, liner, and levee material in landfills; and reuse as fill in approved construction projects. If beneficial reuse cannot be accomplished on a particular project, remaining options include disposal at dedicated upland CDFs or Class I, II (Subtitle D), or III landfills.

Landfill Reuse and Disposal

Due to environmental concerns and site volume limitations associated with in-Bay disposal of dredged material, there has been particular interest in disposal or reuse of dredged material at landfills. Although placement of dredged material in landfills often faces several obstacles, projects undertaken at four Bay area landfills[7] demonstrate that reuse of dredged material can be environmentally and economically feasible here. A report prepared by BCDC (1995a) for LTMS on the potential for dredged material reuse at landfills contained the following conclusions and recommendations:

1. Sixteen of the 127 landfills studied were identified as highly feasible for accepting dredged material for reuse projects. These 16 landfills have a capacity to accept over 5 mcy of dredged material per year over the 50-year planning period for reuse as landfill daily cover and capping material.

2. Rehandling facilities need to be established to dry dredged material for reuse or disposal in landfills.

3. Segregation by grain size to obtain low permeability material should be a priority in consideration of the final design of the rehandling facility and during dredged material placement operations.

4. Testing requirements for upland reuse and disposal are different than those for aquatic disposal. Testing guidelines need to be developed for upland reuse and disposal.

5. Guidelines on the pollutant levels appropriate for disposal and reuse projects should be developed.

Items (4) and (5) involve establishing coordinated policies for testing and interpretation, as discussed in section 3.2.5.2.

Reuse as Construction Fill

The primary consideration in reusing NUAD-class material as fill in approved construction projects is ensuring that the potential exposure pathways of concern are adequately addressed in the design of the

construction project. Often, constructions fills that will be paved or otherwise capped with an impermeable surface can adequately control for infiltration and leachate into groundwater, without the need for special liners or other control mechanisms, especially if the NUAD-class dredged material can be incorporated into the overall fill project in such a way that it is surrounded on all sides by clean fill. Also, dredged material must typically be dewatered (e.g., at a rehandling facility) before it can be use as fill in an upland construction project; however, dewatering first is not always necessary for nearshore fills.

Dredged material being considered for reuse as construction fill must also have acceptable engineering qualities for the particular project — for example, fine-grained silts may not be physically suitable if the fill must bear heavy loading. Sands are generally more versatile for use in fill projects; several sand mining companies dredge natural sand deposits in the Estuary specifically to sell the material for aggregate or for fill.

Dedicated Confined Disposal Facilities

A dedicated CDF is a site constructed specifically for disposal of dredged material. While CDFs can be used for disposal of either SUAD- or NUAD-class dredged material, it is anticipated that CDFs in the Bay area would be used primarily for NUAD-class dredged material that cannot be disposed at other sites, and cannot be reused. The availability here of acceptable sites for unconfined aquatic disposal of SUAD material, in addition to an emphasis on reuse of material whenever possible, would make confined disposal of SUAD material unlikely.

There are no multi-user CDFs at this time in the Bay area. However, any CDFs constructed in the future for NUAD-class material will have to address many of the same siting and design issues as rehandling facilities. In particular, CDFs would have to be designed to contain and isolate the worst-case material that could be permitted for disposal there. It is therefore likely that new CDFs would include some form of liners, surface water control, and other measures to address the potential contaminant exposure pathways associated with the particular site.

3.2.6.3 Treatment

In treatment processes, contaminants in sediments are destroyed, significantly reduced, or converted into less reactive or available forms. Treatment in itself is not a disposal option. However, in some cases treatment can reduce the volume of sediment requiring disposal at more expensive or restrictive sites or can make the material suitable for some kinds of beneficial reuse, thereby reducing potential liability concerns for the dredger. Treatment in general is consistent with agency policies regarding the reduction of wastes, recycling, and minimizing landfill disposal. It is also possible that treatment methods such as soil washing to reduce salinity could make SUAD materials from marine-influenced areas suitable for use on Delta levees.

In general, dredged NUAD sediments have characteristics unique from contaminated soil, including higher water content and significantly lower contaminant concentrations, combined with larger volumes of material. These characteristics often preclude many existing soil remediation techniques from being applicable and/or cost effective for use with dredged NUAD-class sediments, as discussed in detail in the LTMS report *Analysis of Remediation Technologies for Contaminated Dredged Material* (LTMS 1993a). The kinds of potential treatment technologies evaluated in that report include:

- Biological treatment;
- Alkaline stabilization;
- Incineration;
- Encapsulation;
- Other chemical treatment methods; and
- Salinity reduction.

Presently, the most cost effective method of disposal for NUAD material may be to reuse it in a beneficial application that would require minimal or no pretreatment. Reuse as landfill daily cover or liner material, isolation within wetland restoration projects, or isolation in upland or nearshore construction projects are currently more practical than remedial treatment. However, new technologies may be developed that would facilitate the practicality of remediation of NUAD material in the future.

Even today, treatment of contaminated sediments may be practical and feasible for the rare project with high concentrations of a single pollutant. However, a variety of issues would affect any such decision. There is often a diminishing return for treatment costs when contaminant concentrations are not very high. For dredged material, treatment is usually most effective in reducing highly contaminated material to moderately contaminated material. Treatment of moderate or low concentrations of contaminants becomes a more difficult and intensive effort. Agencies and project proponents must consider numerous factors including the following: current contaminant concentrations; target concentrations; differences in disposal costs;

reduction of liability; treatability of the contaminants present; effectiveness of treatment (no treatment process is guaranteed to achieve target levels in all cases); delays in obtaining approval for the treatment process; delays caused by the treatment itself; availability of appropriate places to carry out the treatment process; and other concerns. The economics of treatment can be complex but, in general, the more costly the initial disposal option, the more attractive treatment becomes. For example, if material is designated as a Hazardous Waste, the landfill fees alone could range from $90 to $150 per ton. In such an instance treatment that costs, for example, $60 per cubic yard, with subsequent disposal into a Class III or II landfill at greatly reduced costs, can make treatment desirable if allowable under regulation. However, certain materials cannot be treated easily and, under existing laws, Hazardous Waste is subject to state Department of Toxic Substances Control (DTSC) permit requirements.

In the long run, treatment is a potentially valuable tool. From a policy standpoint, treatment is encouraged when it would be feasible and effective. However, the value of treatment must be assessed on a project-by-project basis.

3.3 STATUS OF DREDGED MATERIAL DISPOSAL TODAY

In the past, management of the dredged material generated by projects throughout the Estuary was effectively piecemeal and reactionary, rather than comprehensive and planned. Options to reduce unconfined aquatic disposal within the Estuary were limited by the general lack of alternative placement sites for large quantities of material. Opportunities to realize environmental benefits by reusing dredged material as a resource — rather than handling it as a waste to be disposed of — were also limited by the lack of available reuse sites, the lack of coordinated agency policies, and financial disincentives to dredging project sponsors. The planning and financial responsibilities for appropriate management of dredged materials that could not be disposed at unconfined aquatic sites (NUAD-class materials) were typically left to project sponsors to address on their own. Together, these problems have helped to make dredging, and disposal of dredged material, expensive, unpredictable and, in the eyes of the public, environmentally questionable.

To a large extent these problems remain today, and the purpose of this programmatic Policy EIS/EIR is to develop and select an overall Long-Term Management Strategy that addresses these kinds of concerns. However, in some ways, the situation is already improved. The recent designation of an appropriate ocean disposal site has given the region its first true, large-scale, multi-user alternative to disposal within the waters of the Estuary. Beneficial reuse of dredged material has also been occurring to a much greater extent: several million cubic yards of sediment from the Port of Oakland Deepening Project have gone into construction of endangered species habitat at the Sonoma Baylands Wetlands Enhancement Project, and to the upland Galbraith site, which will be returned to a recreational site (a golf course) following dewatering of the dredged material. In addition, a demonstration project for reusing dredged material for levee maintenance and stabilization was recently conducted at Jersey Island in the Delta. Even NUAD-class dredged material has been beneficially reused for daily cover and other uses at the Redwood Landfill. There is also the potential to leverage funding with other programs that have overlapping interests and goals, such as the use of dredged material for habitat and/or levee projects pursuant to the Bay-Delta CALFED program (see section 2.2.5). Nevertheless, the great majority of dredged material from San Francisco Bay area dredging projects continues to be disposed at the existing in-Bay sites today. The current disposal sites, their current management, and the distribution of dredged material placement within each environment, are described in Chapter 4 (Affected Environment) and in Chapter 5 (Development of Alternatives).

3.4 SUMMARY

This chapter has presented a basic description of the dredging process, the important physical and chemical factors that determine whether disposal of dredged sediments is of concern in the different placement environments, the approaches used to evaluate dredged material, and the numerous ways in which

dredged material can potentially be disposed and reused safely. The next chapter presents a description of each of the affected environments — the Estuary, the uplands, and the ocean — and identifies those specific resources that are potentially affected, beneficially or adversely, by dredged material disposal and reuse.

Footnotes for Chapter 3:

1. Considering the total volume of dredged material disposed at in-Bay sites annually compared to the volume of sediment resuspended and transported by waves and currents, and assuming that settlement is equally likely to occur anywhere within the system, resettled previously-dredged material probably makes up no more than about 5 percent of all the sediment that needs to be dredged annually in the Bay/Delta. It is possible, however, that previously-dredged sediments may be a significant source in very specific, local situations such as the Mare Island Strait ship channel.

2. For example, the Containment Site Committee report estimated that 10 mcy of dredged material needing contaminant-related management restrictions would be generated in the next 10 years. This included an assumed 2 mcy from the Port of Oakland -42-foot deepening project, and 1 mcy from the Port of Richmond -38-foot deepening project. Actual volumes of sediments needing management restrictions from these projects were later found to be lower: approximately 1.1 mcy and 0.2 mcy, respectively.

3. It is also likely that fines resuspended from other locations throughout the estuary, some of which will have greater contaminant loadings and some of which will have less, will mix with and settle in the same depositional areas, significantly diluting the fines originating from dredged material. Nevertheless, any contaminants in dredged material eroded from dispersive in-Bay sites will add to the overall "background" contamination at depositional sites throughout the estuary, and maximize the potential for aquatic organisms to be exposed to them, rather than removing them from the system.

4. The ITM recommends that the interval between re-evaluation of Tier I data should not exceed 3 years or the dredging cycle, whichever is longer. If there is reason to believe that conditions have changed, then the time interval for re-evaluation may be less than 3 years (USEPA and USACE 1994).

5. Recently, agency efforts have intensified to compile, into a comprehensive database, information on the adverse biological effects associated with tissue residues of contaminants. This information will be used in interpreting bioaccumulation data as they become available.

6. An exception is when return water flows from an upland site back into a water body. This circumstance is regulated under Section 404 of the CWA, and typical aquatic tests do address this issue.

7. BCDC (1994) discusses dredged material reuse projects at the Redwood and Tri-Cities landfills. In addition, dredged material has been reused as capping material at West Winton and Winton Avenue landfills in Hayward, California.