Special Publication SJ94-SP16

St. Johns River, Florida
Water Quality Feasibility Study
Phase I Interim Report

Volume V

A Review of Sediment Analysis, Management Techniques, and Sediment Quality Data for the Lower St. Johns River Basin

U.S. Army Corps of Engineers
Jacksonville District
South Atlantic Division

St. Johns River Water Management District
Palatka, Florida

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A REVIEW OF SEDIMENT ANALYSIS, MANAGEMENT TECHNIQUES AND SEDIMENT QUALITY DATA FOR THE LOWER ST. JOHNS RIVER BASIN

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FINAL REPORT

Submitted to the

Jacksonville District Office
U.S. Army Corps of Engineers
400 West Bay Street
Jacksonville, FL 32202-4407

Contract Number: DACW17-92-C-0008
St. Johns River Water Quality Management Study
Task 4 - Sediment Management

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July 1993

EXECUTIVE SUMMARY

The Lower St. Johns River Basin (LSJRB) is a complex ecosystem subject to a variety of natural and anthropogenic effects. The system is comprised of the main St. Johns River and twelve major tributaries where tidally-driven ocean water is mixed with freshwater from inland drainage. The river sediments consist primarily of inorganic substances of marine origin mixed with fine particulate matter resulting from decomposing organic materials.

Intensive human interference in the form of urbanization, industrialization, agricultural practices and recreational activities have resulted in elevated concentrations of various pollutants at a number of locations throughout the riverine system. There are approximately 360 domestic and 49 industrial permitted point sources which release more than 2 billion gallons per day into the surface waters of the LSJRB. There are also about 70 point sources discharging directly into the groundwater. Due to the ability of aquatic sediments to function as sources and sinks for suspended and dissolved pollutants, concerns have been raised about sediment quality and its impact on environmental conditions of the LSJRB.

Understanding and quantifying the water-sediment dynamics and environmental issues in the LSJRB require an extensive database of physicochemical and biological parameters. The database should include information on hydrometeorology, geomorphology, sediment characterization, hydrodynamics, land-uses, natural habitats, aquatic flora and fauna, and any other relative information on the riverine system and its drainage basins.

Development of a comprehensive sediment quality management study for environmental protection of the LSJRB involves three major components: estimation of the sediment mass-balance, characterization of the physicochemical and biological characteristics of the aquatic ecosystem, and assessment of the overall environmental conditions.

Presently, there is not any quantitative data pertaining to the movement of the suspended and bottom sediments in the river and its tributaries other than the navigation channel dredging records. However, this information is not sufficient to establish a reliable mass balance of the transported sediments. Knowledge of the sediment mass balance is essential for assessment of the pathways of transported sediments and the sediment-bound contaminants.

Regarding sediment quality, six field studies have been conducted since 1982, at various sampling locations throughout the LSJRB system. These studies document the existence of "enriched" levels of nutrients, trace metals (i.e., cadmium, chromium, copper, mercury, nickel, lead and zinc), polychlorinated biphenyls (PCBs), polynuclear aromatic hydrocarbons (PAHs), aliphatic hydrocarbons, chlorinated pesticides and coprostanol in the sediments and aquatic biota of the main river and its tributaries. These pollutants are indicative of agricultural and industrial runoff, residues of fuel combustion, boat traffic and leachate from septic tanks. Locations with elevated contaminant concentrations were found near the Jacksonville metropolitan area, at the confluence of the Cedar and Ortega Rivers, and at Julington Creek, Durbin Creek, Rice Creek, Trout River, Arlington River and Pottsburg Creek.

In spite of these studies, the existing sediment quality data and the bioassessment records of the Lower St. Johns River Basin are too few by far and cover a relatively short period of time. Thus, they are inadequate for a comprehensive analysis that would support any drastic remedial action or any major sediment management plan other than control of the point sources. However, the mere fact that elevated contaminant concentrations have been documented in the estuary and its tributaries justifies development and implementation of a continuous and systematic sediment quality monitoring program.

Such a program would establish a broad database of the historic changes and trends of sediment quality that could be used in future environmental studies or for development of a comprehensive sediment quality management/remedial plan. Initially the program would be focused on the sites with documented high contaminant concentrations, and it later would extend to other locations suspected of having low environmental quality. In addition to sediment data, hydrologic, hydrodynamic and anthropogenic activities data must be collected concurrently. These data are necessary for the calibration and verification of any modeling efforts which would simulate the dynamics and sediment quality of the LSJRB.

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July 1993

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CHAPTER I

SEDIMENT DYNAMICS AND MODELING

A. BACKGROUND AND GENERAL DESCRIPTION OF THE LOWER ST. JOHNS RIVER BASIN

1.1 Location and Geography.

The St. Johns River drainage basin extends from central-east Florida to north-east Florida between 27.7° to 30.5° north latitude and 80.6° to 82.1° west longitude. The river flows in a general south-to-north direction for approximately 300 miles from its headwaters inland of Ft. Pierce to its mouth east of Jacksonville (Figure 1.1). The river borders or crosses through ten different counties, i.e., Brevard, Orange, Seminole, Volusia, Lake, Marion, Putnam, Clay, St. Johns and Duval. Also, a very small area of the riverine basin lies within the Alachua, Baker and Bradford counties. Along its upstream stretch, the basin encompasses a number of major and minor lakes including Lake Hellen Blazes, Sawgrass Lake, Lake Washington, Lake Winter, Lake Poinsett, Lake Harney, Lake Monroe, Lake George and Crescent Lake. The system includes a variety of diverse elements such as wildlife refuges, the Ocala National Forest, state parks and other recreation areas, historical sites, agricultural lands, rural developments, industrial complexes and urban communities. The urban communities and some of the rural ones affecting the water quality of the Lower St. Johns River Basin (LSJRB) are:

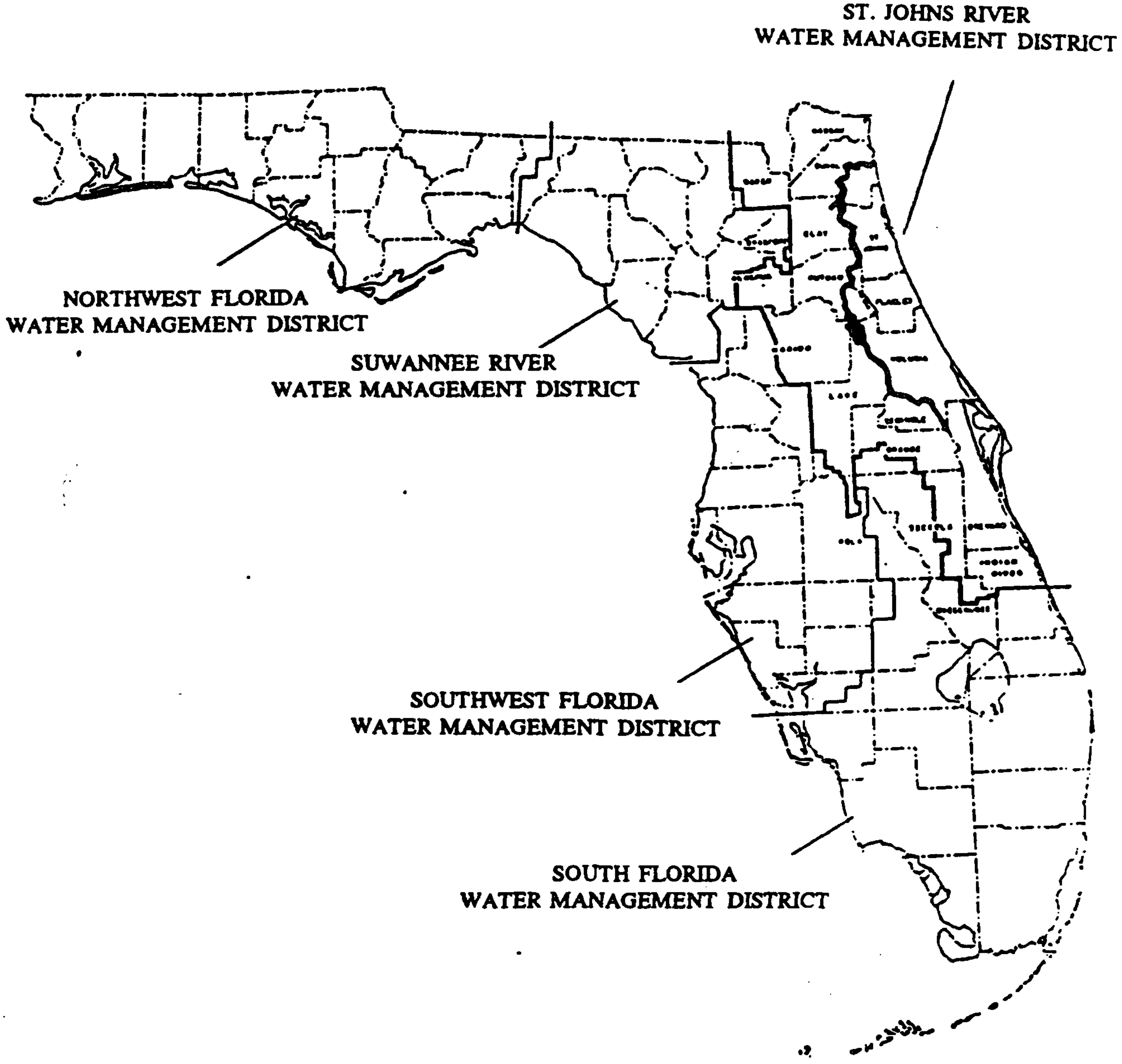


Figure 1.1 Geographic map of the St. Johns River (Bergman, 1992).

Jacksonville, Orange Park, Middleburg, Green Cove Springs, Palatka, Pomona Park, Keystone Heights, Hastings, and Bunnell. Industrial activities within the LSJRB include electric power generation, pulp and paper production, food processing, chemical factories, general manufacturing and maritime works. In addition, there are intensive agricultural, silviculture and cattle grazing activities.

1.2 Hydraulics and Drainage.

The lower 101 miles of the main river, downstream of Oklawaha Canal (near Palatka) is considered for management purposes to be the Lower St. Johns River Basin (Figure 1.2). The drainage area of this lower basin is about 2,200 square miles, which represents 22 percent of the total area of the St. Johns River Water Management District (Campbell et al. 1989). The total area of the drainage basin of the St. Johns River and its main tributary, the Oklawaha River, is approximately 12,400 square miles, i.e., about one-sixth of the total area of the State of Florida (St. Johns River Water Management District, 1990). The drainage area of the tidally affected section of the river is about 7,200 square miles.

The bed of the estuary is below sea level and tidal fluctuations penetrate to a distance of 200 miles upstream. The average water surface gradient of the river is 0.1 ft per mile, i.e., $S_o = 1.89 \times 10^{-5}$ (U.S. Army Corps of Engineers, 1986). The gradient at the lower part of the river is extremely small, i.e., 0.05 ft per mile or $S_o = 0.95 \times 10^{-5}$ (Snell and Anderson, 1970; Keller and Schell, 1992). The width of the river north of Palatka (downstream) ranges from 1.5 to 3 miles. Near the City of Jacksonville, the river narrows to about a quarter of a mile. The widest section of the river is across the

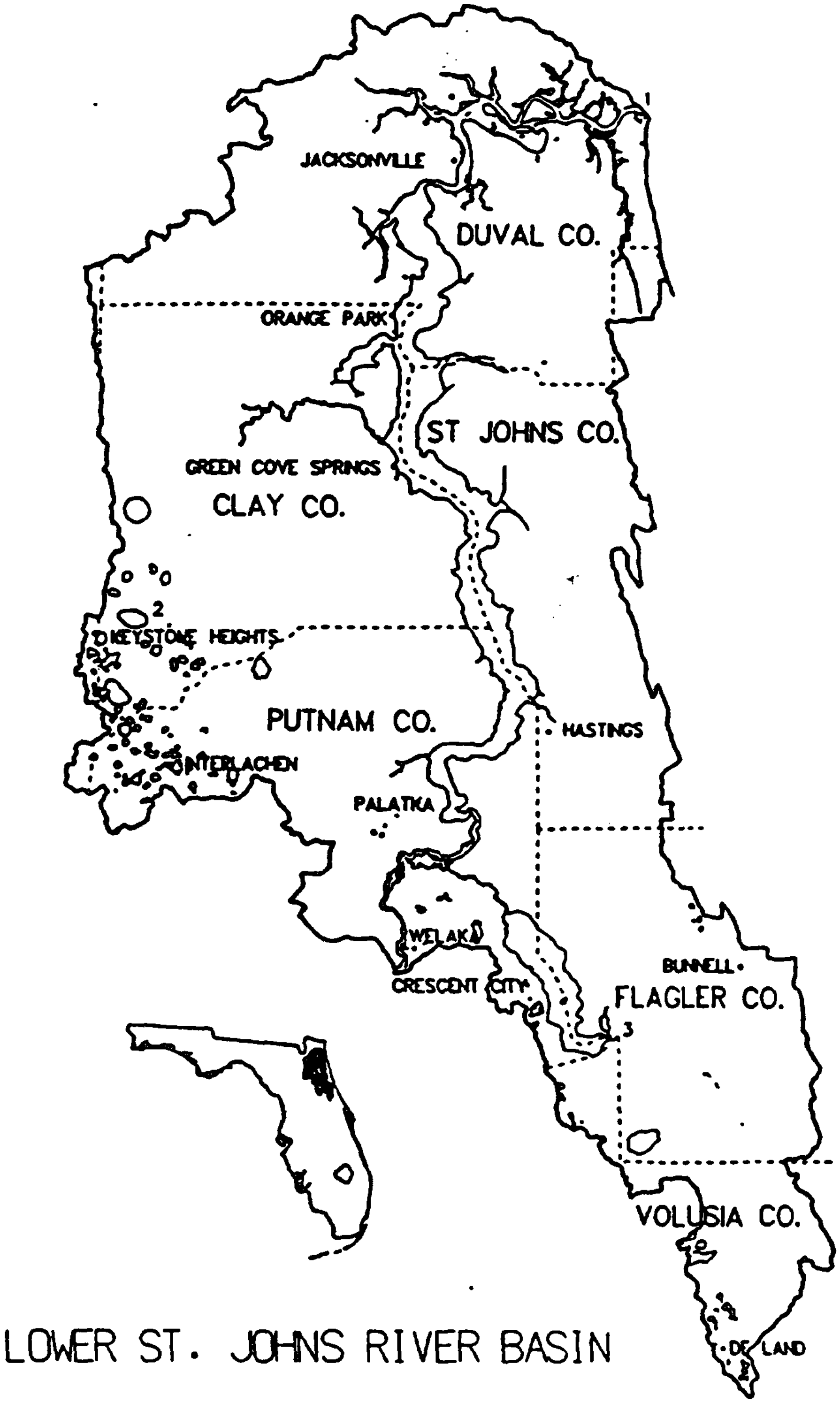


Figure 1.2 Map of the Lower St. Johns River Basin (Campbell et al. 1989).

mouth of the Ortega River. The main tidal constituent at Mayport is semi-diurnal (M2-tide), with a period of 12.42 hours and an average tidal range at the river's mouth of about 4.9 ft Near Palatka, the mean tidal range is reduced to 1.2 ft. Between Orange Park and Palatka it has been reported that the tide maintains the characteristics of a standing wave, while downstream and upstream of these locations the flow has the typical reversing motion tidal characteristics (Haight, 1938).

The Lower St. Johns River receives drainage water from twelve main tributaries and the land adjacent to its main course. The names of these tributaries along with their drainage area, inflowing minor streams, length, discharge, land use and water quality are given in Table 1.1 (Bergman, 1992). For management purposes, the LSJRB is divided by the U.S. Army Corps of Engineers into three planning regions, i.e., the East Region, the West Region, and the North Region (Figure 1.3).

From a twelve year record (1969 to 1982), the average discharge at Palatka (river mile 83.1) is estimated as 5,945 cfs, with a maximum daily downstream discharge of 31,300 cfs and a maximum daily reverse flow of 20,400 cfs. From a twenty-two year period of data (1954-76) the average flow at the City of Jacksonville (river mile 25) is estimated as 5,687 cfs with a maximum daily downstream discharge of 64,000 cfs and a maximum daily reverse flow of 62,700 cfs. Finally, at the mouth of the St. Johns River the maximum daily flood flow is found to be about 61,100 cfs and the maximum ebb flow about 51,040 cfs (U.S. Army Corps of Engineers, 1986). The net downstream velocity of the river is very small. At State Road 44 (SR44) near DeLand (river mile 142) the average velocity is approximately 0.44 fps. At the river's mouth, during ebb or flood, flow velocities can reach 4 fps.

Table 1.1 Characteristics of the LSJR drainage tributary sub-basins (Bergman, 1992).

Drainage Basin Basin Area (sq mi)	Tributary	Length (mi)	Discharge mean-max-min	Land Use and Pollution Issues
Arlington River (32.4)	*Arlington River	1.9		Rapidly growing residential developments (50%). Water quality is fair to poor.
	*Pottsburg Creek	7.5		
	*Lt. Pottsburg Cr.	3.2		
	*Strawberry Creek	3.3		
	*Silversmith Creek	2.1		
Black Creek (496.5)	*Black Creek	13.3		Silviculture and natural vegetation cover, largely undeveloped. Water quality is good to fair.
	Bradley Creek	5.3		
	*Peters Creek	6.7		
	Little Black Cr.	10.1		
	Double Branch	2.3		
	North Prong	2.3		
	South Prong	2.8		
	*N. Fork Black Cr.	28.2	200-11200-10	
	Big Branch	5.0		
	Long Branch	4.2		
	Yellow Water Cr.	11.0		
	*S. Fork Black Cr.	22.3	148-10300-4	
	Bull Creek	8.5		
	Greens Creek	10.7		
	Ates Creek	10.5		
Broward River (26.8)	Broward River	4.0		Residential, industrial, agricultural, airport (15%). Water quality is fair.
	*Cedar Creek	5.5		
	*Little Cedar Cr.	4.2		
Crescent Lake (605.0)	Dunns Creek	8.5		Silviculture (57%), row crops (15%), cattle operations (8%). Water quality is good. Algae blooms are common.
	Haw Creek	4.8		
	Middle Haw Creek	9.9	77-1810-0	
	Little Haw Creek	7.2	84-1600-0	
	Black Branch	3.8		
Deep Creek (91.0)	Deep Creek	10.4	9-227-0	Mainly agricultural production. Water quality is good.
	Sixteenmile Creek	7.0		
Dunn Creek (23.3)	Dunn Creek	2.0		Forest (80%), agricultural, urban, industrial, wetlands. Water quality is fair.
	Terrapin Creek	0.8		
	Rushing Branch	1.7		
	Caney Branch	2.3		
Etonia Creek (355.0)	*Rice Creek (Main)	6.4	45-2000-0	Mainly agricultural production, paper mill. Water quality is good to poor.
	Etonia Creek	17.5	100-1160-35	
	*Simms Creek	14.8	47-1300-4	
	Rice Creek	16.1		
Julington Cr. (104.3)	Julington Creek	8.5		Mainly undeveloped, natural cover, agriculture/silviculture (40%) Water quality is fair.
	Flora Brench	1.0		
	Big Davis Creek	3.8	11-735-0	
	*Durbin Creek	4.0		
	Sampson Creek	1.1		
	Oldfield Creek	2.2		
	Sweetwater Creek	2.3		
McCullough Cr. (61.8)	McCullough Creek	3.2		Mainly agricultural. Water quality is good.
Ortega River (99.2)	*Ortega River	16.0	36-2500-0	Residential (33%), industrial, Commercial, vacant lands. Water quality is fair to poor.
	*Fishing Creek	3.4		
	McGirts Creek	7.4		
	*Cedar River	8.0		
	*Butcher Pen Creek	1.7		
	*Wills Branch	4.5		
Sixmile Creek (121.8)	*Sixmile Creek	4.9		Mainly agricultural. Water quality is fair.
	Mill Creek	2.3		
	*Trout Creek	3.6		
	Turnbull Creek	15.0		
St. Johns River (210.0)	Newcastle Creek	1.2		Residential, commercial, industrial, undeveloped. Water quality is good to fair.
	Jones Creek	2.4		
	Ginhouse Creek	2.9		
	Mt. Pleasant Creek	5.0		

Continued:

Table 1.1 Characteristics of the LSJR drainage tributary sub-basins (Bergman, 1992) (Continued):

Drainage Basin Basin Area (sq mi)	Tributary	Length (mi)	Discharge mean-max-min	Land Use and Pollution Issues
	Tiger Pond Creek	1.6		
	Greenfield Creek	5.0		
	*McCoy Creek	3.4		
	*Hogan Creek	2.4		
	Long Branch	1.8		
	Deer Creek	0.9		
	Big Fishweir Creek	1.8		
	Lt. Fishweir Creek	1.6		
	Deep Bottom Creek	1.1		
	Goodbys Creek	1.7		
	Christopher Creek	1.4		
	Craig Creek	1.7		
	Miller Creek	1.0		
	Governors Creek	8.4		
	Clarkes Creek	7.6		
	Cedar Creek	3.5		
	Camp Branch	9.1		
	Tocoi Creek	3.6		
	Dog Branch	1.8		
	Mill Branch	2.0		
Trout River (94.0)	*Trout River	19.2		Residential, commercial, agricultural. Water quality is poor.
	*Moncrief Creek	5.5		
	Blockhouse Creek	2.7		
	Half Creek	2.7		
	Gulley Branch	1.8		
	Little Trout R.	1.8		
	*Ribault River	7.6		
	*Sixmile Creek	6.5		
	West Branch	1.9		

Note: An asterisk (*) indicates tributaries with water and/or sediment quality problems.

The St. Johns River exhibits typical characteristics of an estuary, where saline water from the ocean mixes freely with fresh water from inland drainage. Stratification between fresh and salt water (i.e., formation of saline wedge) is observed from the mouth to a distance of 40 miles upstream. The water becomes well-mixed near the Duval-Clay county line. Under drought conditions, sea water intrusion extends upstream until it reaches Palatka. Further upstream from Palatka, salinity increases due to chlorides introduced from groundwater seepage and salt water springs (Odum, 1953; Brody, 1992).

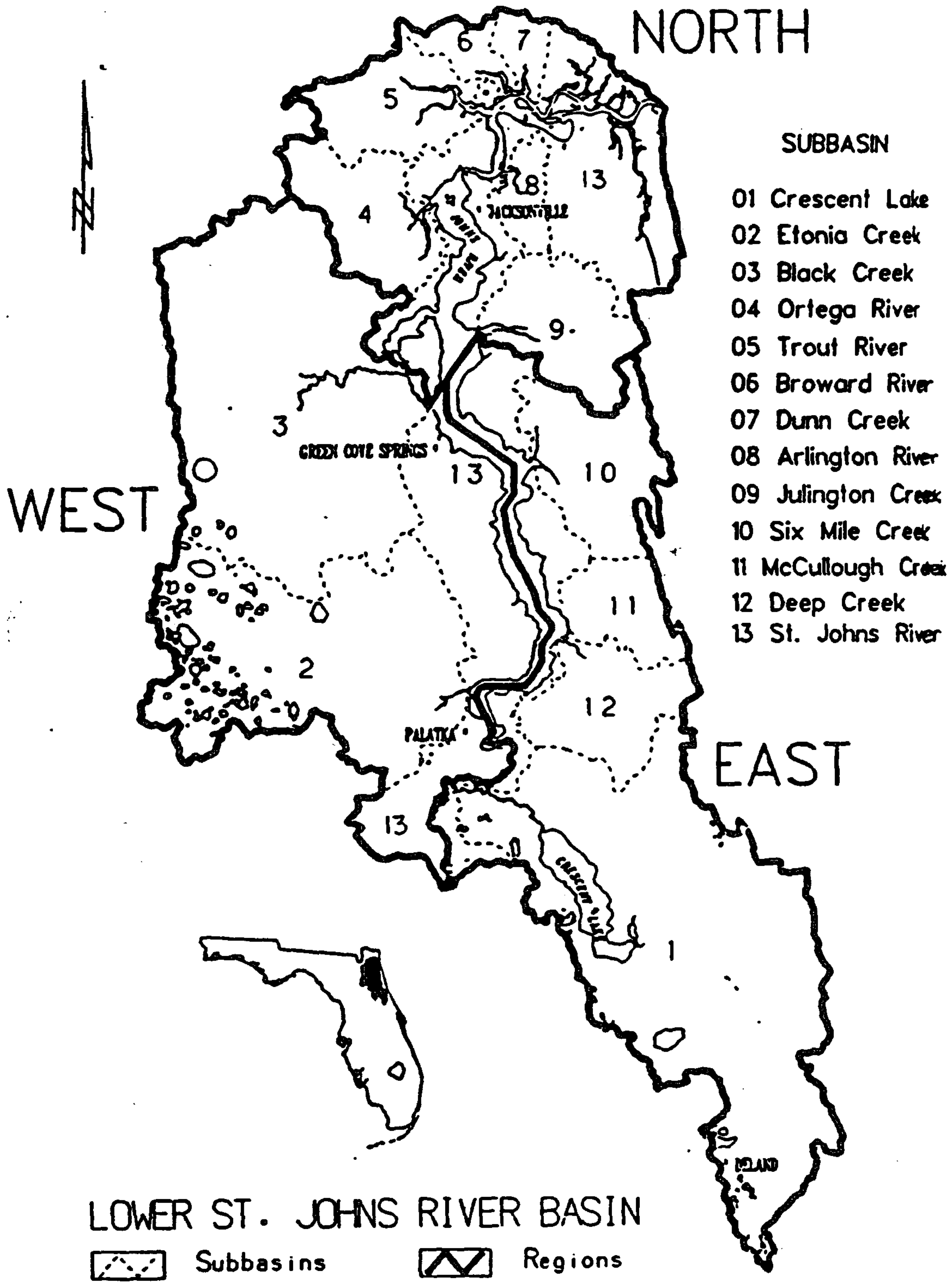


Figure 1.3 U.S. Army C.O.E. planning regions of the LSJRB (Campbell et al. 1989).

1.3 Dredging and Navigation.

For navigation purposes, a channel of adequate dimensions is maintained from the mouth of the river to Lake Harney. The dimensions of this artificial channel vary along the river. Thus, the navigation channel is 38 ft deep (Mean Low Water - MLW) along the first 20 river miles to White Shells Cut Range, 34 ft deep to Commodore Point and 30 ft deep to the Florida East Coast Railroad Bridge in Jacksonville. The width of the channel from its mouth to the City of Jacksonville varies from 400 to 1,200 ft. Upstream from Jacksonville, the channel is maintained at 13 ft deep and 200 ft wide from Jacksonville to Palatka, 12 ft deep and 100 ft wide from Palatka to Sanford, and 5 ft deep and 100 ft wide from Sanford to Lake Harney (U.S. Army Corps of Engineers, 1989). The spoil material from maintenance dredging is presently disposed of on Bartrum and Blount islands, which were created totally from dredge spoil.

1.4 Hydrology and Climate.

The Lower St. Johns River basin is classified as humid subtropical and is located in the transitional zone between the humid continental climate of the northern Florida and the subtropical climate of southern Florida. In summer the average daily temperature maximum and minimum are respectively 90°F and 72°F. In winter, frontal activities may cause temperatures to fall below freezing throughout the basin at an average frequency of 10 to 15 times. However, freezing temperatures are primarily limited during night and the early morning hours (Campbell et al. 1989). Freezes may occur in northern parts of the basin anytime between November and March.

The prevailing wind direction in the basin is generally from the northeast during the September to January period, and from the southwest or southeast in February through August. The northeaster winds are characterized by sustained velocities of 20 to 30 mph and a light rain which typically last for 3 to 4 days. The average monthly wind speeds range from 6 to 10 mph. However, during episodic events (e.g., thunderstorms, tropical storms, hurricanes) wind speeds can reach hurricane strength wind velocities (120 mph). Since the directions of prevailing winds are aligned approximately with the longitudinal axis of the main river, winds have a strong effect on the hydrodynamics of this wide and shallow riverine system.

Annual precipitation within the basin is characterized by a "wet" season from June through September and a "dry" season from October through May. Most of the annual precipitation occurs during the "wet" season in the form of short duration, but high intensity thunderstorms. Based on rainfall records for the years 1947-1976, the basin's annual precipitation ranged from 48.99 to 59.14 inches, with an annual maximum of 84.95 inches and an annual minimum of 27.44 inches (U.S. Army Corps of Engineers, 1986). In the southern part of the basin, short intensive rainfall events exceeding 1 inch per event were not unusual. Snow rarely fell in the basin (i.e., once every few years). Analysis of precipitation records indicated a higher correlation between rainfall in the north-south direction than in the east-west direction (Bergman, 1992). This shows that frontal activities follow mostly a north-to-south direction. Evapotranspiration losses from the basin are estimated at an annual rate of 35 to 40 inches.

1.5 Geomorphology and Sediments.

The geomorphological features of the drainage basin are relatively low and flat.

Generally, the LSJRB is bounded by three geological districts, i.e., the Sea Island District along the north-northwest, the Eastern Flatwoods District along the east, and the Central Lake District along the south-southwest (Brooks, 1981) (Figure 1.4). Each of these three districts is divided into smaller subdistricts. The Sea Island District includes the Okefenokee Upland (maximum elevation 240 ft), the Duval Upland (maximum elevation 100 ft), and the Northern Coastal Strip (maximum elevation 95 ft). The Eastern Flatwoods District includes the Palatka Anomalies and the Volusia Ridge Sets (maximum elevation 55 ft). The Central Lake District includes the Interlachen Sand Hills (maximum elevation 220 ft), the St. Johns Offset, and the Crescent City-DeLand Ridge (maximum elevation 100 ft) (Bergman, 1992).

It is believed that the Lower St. Johns River valley was formed during the time of the Talbot shoveling when the mean sea level was approximately 42 ft above the current level. During that time, most of what is now Duval and western St. Johns counties was inundated, forming a 20 ft deep lagoon. Sand bars and barrier islands separated this lagoon from the ocean (Struthers, 1981). After recession of the sea level, thousands of years of surface hydrogeologic processes shaped the present system of the LSJRB stream network. The majority of the surficial layers of the river bed and the surrounding watershed areas contain Pleistocene and Holocene sediments (Figure 1.5). Pockets of Miocene sediments exist east of Green Cove Springs, near Palatka, at Crescent City, and at Deland (American Association of Petroleum Geologists, 1991). In addition, it has been documented that the bottom of the man-made navigation channel has penetrated into deposits of Miocene or Pliocene sediments (Boehnke et al. 1983).

Soils in the LSJRB vary from well-drained sands in the uplands (soils of the

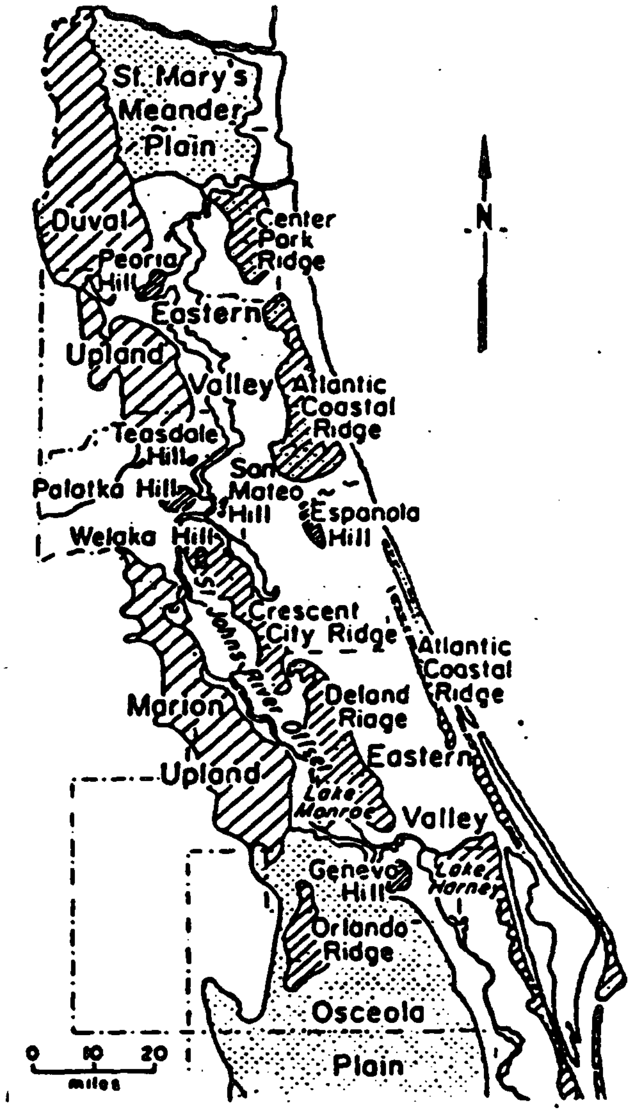


Figure 1.4 Geological districts of the LSJRB (U.S. Army C.O.E. 1986).

PERIOD EPOCH	GROUP	PENINSULAR FLORIDA EAST	FORMATION SYMBOLS BEDDED SEQUENCE			TERRACE DEPOSITS	
HOLOCENE		ALLUVIUM AND SHORE DEPOSITS	Q	Q			Qi
PLEISTOCENE		SILVER BLUFF	Q	Qlf		Qsb	Qsbi
		PRINCESS ANNE	Q	Qlf		Qpa	Qpai
		PAMICO		Qft		Qpm	Qpmi
		TALBOT	Qmp	Qkl	Qap	Qta	Qtai
		PENHOLOWAY	Qt	Qc		Qpe	Qpei
		WICOMICO	Qt	Qc		Qw	Qwi
PLIOCENE			Tpc	QTp			
		CHARLTON	Tmu	Tpm			
MIOCENE	CHOCTAW-HATCHEE						
	ALUM BLUFF	HAWTHORN	Tmm				
OLIGOCENE	VICKSBURG	PRE-HAWTHORN ROCKS	To				

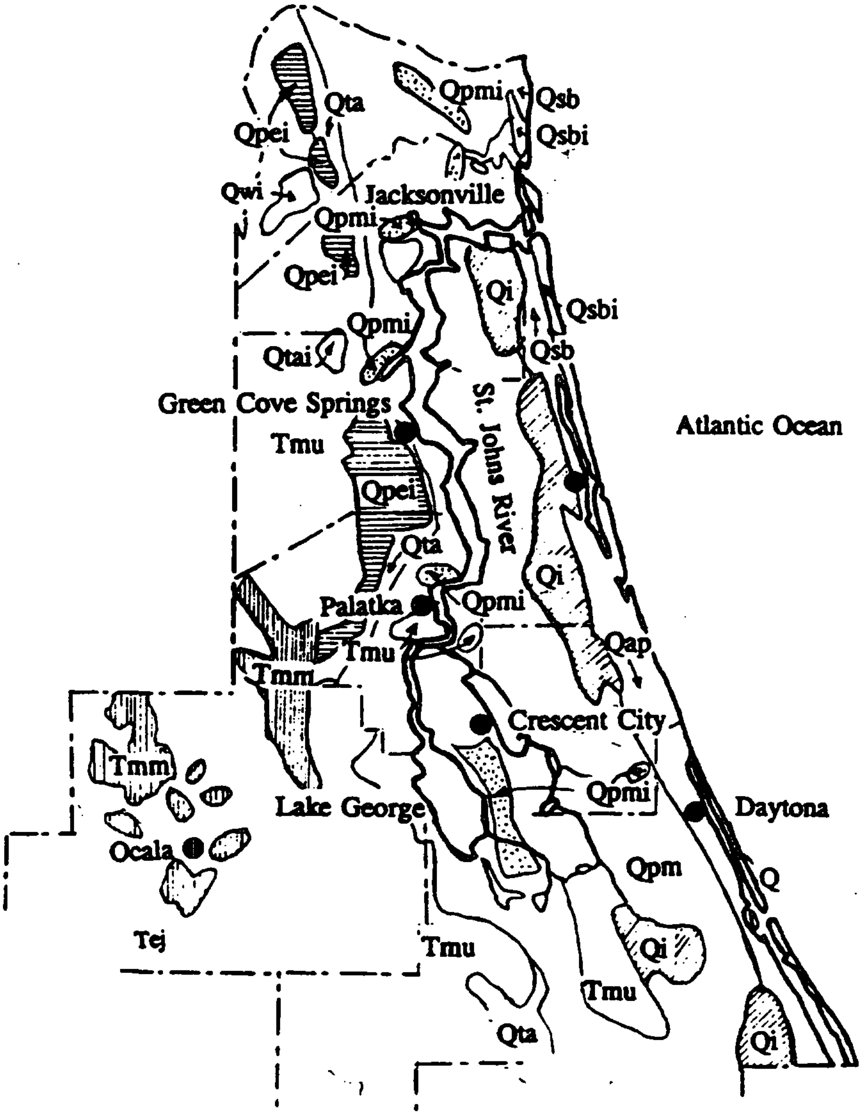


Figure 1.5 Surficial sediments of the LSJRB (American Association of Petroleum Geologists, 1991).

Central Ridge), to mostly saturated organic peat and muck in the lowlands (soils of the Flatwoods or soils of Organic Origin) (Fernald and Patton, 1984). Using the A, B, C, D soil classification established by the Soil Conservation Service (SCS), it has been found that in all of the subdrainage areas excluding Arlington River, soils belong predominantly in category D (Table 1.2). Soils in category A are very well drained thick sands, sometimes with loamy subdeposits; soils in category B are moderately well drained very thick sands with loamy or clayey subdeposits; soils in category C are somewhat poorly drained, comprised of primarily sands with subsoils from loamy, clayey or weakly cemented sands; soils in category D are very poorly drained and they range from sandy soils with loamy subsoils, to purely organic soils in layers of more than 50 inches thick. Within the LSJRB, large peat deposits can be found near Palatka, in the Oklawaha River Valley, near Lake Jessup, in Julington Creek and in Pablo Creek.

Table 1.2 Soil classification according to SCS for the LSJR sub-basins (CDM, 1989).

	Category			
Sub-basin	A	B	C	D
Arlington River	70%	2%		28%
Black Creek	24%	24%		52%
Broward River	38%	12%		50%
Crescent Lake		Poorly to very poorly drained		
Deep Creek		Poorly drained		
Dunn Creek	40%	12%		48%
Etonia Creek		Moderately to poorly drained		
Julington Creek	23%	11%	6%	60%
McCullough Creek		Well to poorly drained		
Ortega River	22%	8%		70%
Sixmile Creek		Well to poorly drained		
SJR downstream of Trout River	25%			75%
SJR upstream of Trout River	28%	9%		63%
Trout River	11%	13%		76%

1.6 Water Quality.

The major land use in the LSJRB is forest land (i.e., silviculture and natural forests). However, the Jacksonville metropolitan area and other residential, agricultural and industrial units generate substantial amounts of pollutants leading to degradation of surface water quality. Pollution is produced by both point and non-point sources. Point sources include 48 industrial complexes; 116 A, B and C domestic waste water plants; and another 206 permitted sources each of them producing an average of 25,000 gallons per day (See Chapter III, Table 3.1). Thus, there are at least 370 known point sources releasing pollutants directly into the riverine system (U.S. Army Corps of Engineers, 1986). Besides the point sources, household and agricultural chemicals (e.g., nutrients, pesticides, herbicides, insecticides, etc.) are released into the river through surface runoff and groundwater seepage. Tributaries with significant water quality problems were identified in Table 1.1.

1.7 Public Concerns and Legislature.

Sewage discharges, industrial wastes, seepage of agricultural chemicals, health warnings to the public, littering of the riverine water and banks, mangrove losses, destruction of benthic communities, fish-kills, algae blooms and numerous other incidents, have aroused serious public concerns regarding the quality of the Lower St. Johns River system (Keneagy, 1991a; 1991b). Most environmental degradation is caused by various human activities especially in Duval county and the Jacksonville metropolitan area.

In order to assess the environmental conditions of the St. Johns River, the Energy and Water Development Appropriation Act, 1984 (Public Law 98-50) authorized the U.S. Army Corps of Engineers to conduct reconnaissance investigations. Subsequently, the Corps produced in 1986 a report entitled "St. Johns River Basin - Florida Interim Water Quality Management" which identified data deficiencies and suggested plans for detailed studies to provide more information. The report noted the lack of technical information pertaining to riverine water quality. It also emphasized the need for a thorough investigation into temporal and spatial variations of the physical, chemical and biological components of the St. Johns River. Such an investigation would require synoptic studies pertaining to basin hydrology, estuarine hydrodynamics, geological and geomorphological details, groundwater movement, overland runoff, tributary inflows, sediment transport, benthic and sessile communities, point and non-point contamination sources, physicochemical aspects, and microbial information for the water column and surficial sediments.

The St. Johns River Water Management District initiated a comprehensive reconnaissance study of the LSJRB which includes: basin hydrology; surface water hydrology; hydrodynamics of surface water; water quality; river sediment characteristics and quality; biological resources; economic values; and intergovernmental management. This study will identify the need for more research and assimilation of information leading to development of a management strategy for the estuary.

1.8 Purpose and Objectives of this Study.

This study is part (Task # 4 - Sediment Management) of a broad ongoing project pertaining to St. Johns River Water Quality Management. The purpose of this study is to provide a better understanding of sediment properties and dynamics and to: document their importance in environmental degradation; identify problems and data needs; and develop guidelines for establishment of future detailed studies and management procedures. The study will summarize the state-of-the-art on: sediment dynamics and modeling; contaminant decomposition, precipitation or partitioning in aquatic ecosystems; environmental impact assessment methods; and techniques for establishment of quantitative environmental indices. Pollution effects on benthic and sessile communities will be assessed, while information will be collected for point and non-point pollution sources within the Lower St. Johns River Basin. Emphasis will be placed on sediment management as related to urban expansion, industrial activities and agricultural practices. The study will entail a collection of existing data and information, evaluation and analysis of available resources, and suggestions for further studies. Some information that may not be directly applicable to the LSJRB will also be included for completeness and continuity of the study.

B. PHYSICOCHEMICAL CHARACTERISTICS AND SEDIMENT DYNAMICS

1.9 Introduction.

The extent of contamination of aquatic ecosystems can often be determined by analyzing sediments for metals and organic pollutants. Contrary to the once popular belief that sediments are acting only as pollutant traps, resuspended contaminated sediments can substantially contribute to the degradation of an ecosystem (Hunsaker and Carpenter, 1990). Indeed, mobile sediments in an estuarine system may reduce navigability by shoaling, undermine the integrity of the foundation of marine structures by scouring, or increase turbidity levels and contaminant concentrations of the ambient water during resuspension events (Partheniades, 1973a). Suspended or dissolved pollutants in aquatic environments can pose a very serious problem to cohesive sediment bottoms (Medina and McCutcheon, 1989). Due to their small particle size (high specific surface) cohesive sediment particles have an affinity for dissolved or suspended pollutants. Understanding water-sediment interaction is very important for environmental quality assessment of aquatic ecosystems. The impact of sediments on aquatic ecosystems is now being considered in rule-making. For example, on April 27, 1991, Washington became the first state which adopted standards for contamination of marine sediments. These standards developed by the Washington Department of Ecology apply to 47 chemicals and include five different biological tests for quantifying the effects of polluted sediments in Puget Sound. In Florida, the Department of Environmental Regulation (FDER) completed a report for evaluation of contamination of Florida's coastal areas by using an effects-based numerical chemical sediment quality assessment (MacDonald, 1992).

Within the LSJRB urban growth, intensive agricultural practices and industrial activities have significantly increased the rates of sedimentation and chemical pollution during the past 40 years. In certain tributaries concentration levels of nutrients, metals, and organic pollutants exceed the levels allowable by the FDER.

1.10 Physicochemical Characteristics.

1.10.1 Sediment Classification.

Sediments originate from weathering of rocks by physical forces (abrasion, temperature variations, etc.), chemical reactions (oxidation, vapor, CO_2, etc.), and/or biological processes (e.g., animal burrowing, plant roots) (Garde and Ranga Raju, 1985). Besides sediments of mineralogical composition there are sediments originating from decomposition of organic matter. Individual sediment particles are primarily characterized by their specific gravity (γ_s), particle size (D), and particle shape. Most sediments are comprised of a variety of parent rock minerals. For alluvial sediments, the specific gravity usually ranges between 2.56 to 2.76 (Table 1.3). If there are not enough field data to calculate a local value for specific gravity, an average value of γ_s = 2.65 is commonly used.

Table 1.3 Specific gravity of some common minerals (Garde and Ranga Raju, 1985).

Mineral (Parent Rock)	Specific gravity (γ_s)	Mineral (Parent Rock)	Specific gravity (γ_s)
Barite	4.5	Basalt	2.7-3.2
Clay shale	2.7-2.9	Corundum	3.9-4.0
Feldspar	2.7	Flint	2.7
Garnet	3.2	Granite	2.5-3.1
Hornblende	3.2	Limestone	2.5-2.8
Magnetite	5.2	Quartz	2.7
Sandstone	2.2-2.5	Serpentine	2.4-2.7
Seynite	2.6-2.8	Tourmaline	3.5-4.3
Trachyte	2.6-2.8	Zircon	4.6

Several classifications have been developed for particle size but perhaps the best-known is the Wentworth scale (Vanoni, 1975). This scale uses a geometric series with base 2, (i.e., $D = 2^n$, n = 0 to 12) (Table 1.4).

In the LSJRB, sediment particle diameters largely fall within the last two particle groups, i.e., from very coarse sand to very fine clay. For large size particles shape can

Table 1.4 Wentworth scale for particle size diameters (Vanoni, 1975).

Name	Size in mm	Remarks
Very large boulders	4096-2048	Size of individual particles easy to measure. Difficult to establish particle size distribution.
Large boulders	2048-1024	
Medium boulders	1024-512	
Small boulders	512-256	
Large cobbles	256-128	Important to local scouring and resistance to flow. Less important to bed load.
Small cobbles	128-64	
Very coarse gravel	64-32	
Coarse gravel	32-16	
Medium gravel	16-8	
Fine gravel	8-4	
Very fine gravel	4-2	
Very coarse sand	2-1	Overall very important to sediment transport in alluvial channels. Particle diameter is measured by sieving or visual accumulation tubes.
Coarse sand	1-1/2	
Medium sand	1/2-1/4	
Fine sand	1/4-1/8	
Very fine sand	1/8-1/16	
Coarse silt	1/16-1/32	Important to total sediment load, density currents, consolidation, channel stability and development of berms. Particle size is measured by microscopic indirect methods.
Medium silt	1/32-1/64	
Fine silt	1/64-1/128	
Very fine silt	1/128-1/256	
Coarse clay	1/256-1/512	
Medium clay	1/512-1/1024	
Fine clay	1/1024-1/2048	
Very fine clay	1/2048-1/4096	

be described using an orthogonal triaxial system, i.e., by approximating the particle by an ellipsoid with main axes a, b, and c, where $a \geq b \geq c$ (Figure 1.6). Based on these three axes particles are classified as spheroid, disk, roller and blade (Vanoni, 1975) (Table 1.5).

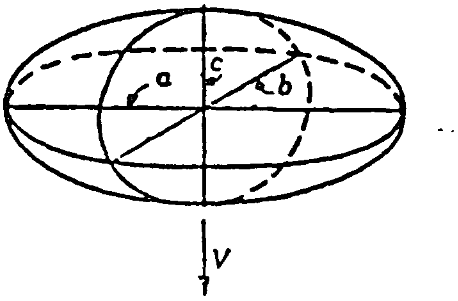


Figure 1.6 Triaxial particle shape (Vanoni, 1975).

Table 1.5 Triaxial particle shape classification (Vanoni, 1975).

Spheroid:	1.0 ≥ b/a ≥ 2/3	and	1.0 ≥ c/b ≥ 2/3
Disk:	1.0 ≥ b/a ≥ 2/3	and	2/3 ≥ c/b ≥ 0
Roller:	2/3 ≥ b/a ≥ 0	and	1.0 ≥ c/b ≥ 2/3
Blade:	2/3 ≥ b/a ≥ 0	and	2/3 ≥ c/b ≥ 0

Other features indicative of particle shape are the sphericity and the roundness. Sphericity describes how close the shape of a particle compares to that of a sphere, while roundness indicates the presence of corners and edges on particle surface. Sphericity depends on mineralogical composition, while roundness on particle weathering and abrasiveness (Garde and Ranga Raju, 1985). For small particles it is very difficult to quantify and utilize information based on individual particle sizes. Therefore, bulk properties such as density and mean fall velocity are commonly used to describe particle distribution.

1.10.2 Particle Frequency Distribution. Sediment frequency distribution is usually expressed in terms of particle diameter versus percentage by weight, plotted on a histogram or frequency distribution curve (FDC) (Figure 1.7). The main characteristics of a FDC are its spread, skewness, and kurtosis. Spread represents the uniformity of a sample; the smaller the spread the more uniform the sample is. The mid-point of the range is the median diameter D_{50} while the center of gravity of the area under the FDC gives the mean diameter D_m. For a symmetric frequency distribution curve $D_{50}=D_m$. Uniformity is a very important feature of sediments because it is directly related to porosity and void ratio. Porosity (n) and void ratio (e) are defined respectively as

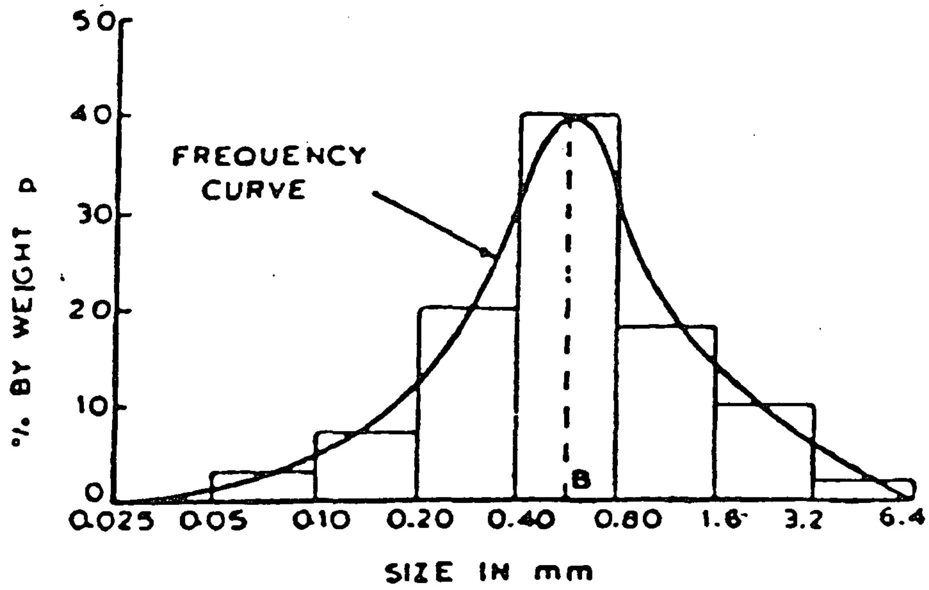


Figure 1.7 Histogram and frequency distribution curve (Garde and Ranga Raju, 1985).

$$n = V_e/V_B = V_v/(V_e+V_s) \quad \ldots\ldots (1.1)$$

$$e = V_e/V_s \quad \ldots\ldots (1.2)$$

where V_e is volume of voids, V_s is volume of solids, and V_B is the bulk volume. For nonuniform sediments, porosity is less than the porosity of uniform sediments of similar particle diameter. Skewness quantifies the asymmetry of the FDC and can be either positive (skewed to the right) or negative (skewed to the left). A positively skewed FDC indicates that the majority of sediment particles have a diameter larger than the median diameter. Kurtosis quantifies the pointness of the peak of a FDC. The importance of skewness and kurtosis of particle FDC on sediment transport dynamics has not as yet been determined.

From a frequency distribution curve equation f(D), a cumulative distribution curve F(D) can be generated as follows:

$$F(D) = \int_0^D f(r)\,dr \quad \ldots\ldots (1.3)$$

where r is a dummy variable of integration. The cumulative distribution function (CDF) is plotted on normal coordinate paper, on semi-logarithmic paper, on semi-probability paper, or on logarithmic-probability paper. The logarithmic scale is always used for the particle diameter and the probability scale for the particle distribution. The reason for using logarithmic and/or probabilistic paper is that particle distribution of riverine sediments usually approximates to a Gaussian (normal) distribution. Therefore, plotting on a logarithmic-probabilistic paper gives a linear distribution (Vanoni, 1975). This is particularly true for the F(x) section between particle diameters of $D_{84.1}$ and $D_{15.9}$, where D_x denotes the particle diameter that is larger than x% of the sample (Figure 1.8).

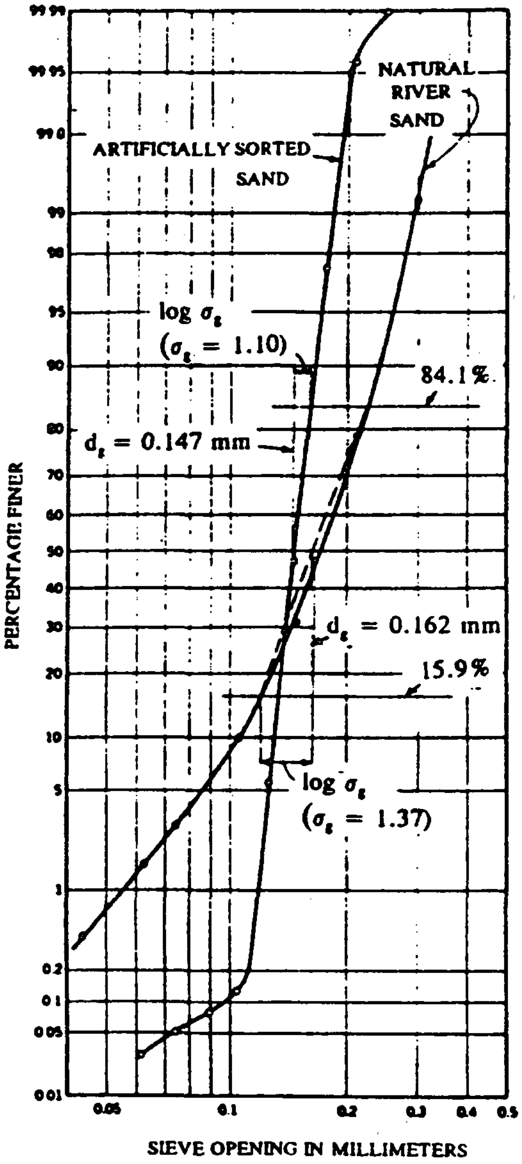


Figure 1.8 Cumulative particle distribution curve (Vanoni, 1975).

1.10.3 Cohesive Sediments. Particle diameter and its distribution drastically affect sediment dynamics. Based on particle size, two major sediment groups are recognized: granular and cohesive. The separation line between these two groups is fine silt (D=15 μm). The basic difference between granular and cohesive sediments is the type of forces that govern their dynamic behavior. Granular materials are subject only to mechanical forces, i.e., inertial, gravitational, drag, and buoyancy (Ibad-zade, 1987). Cohesive sediments, in addition to the mechanical forces are subject to electrochemical forces. These electrochemical forces include primary bonds (ionic and covalent), secondary bonds such as Van der Waal's, residual electric surface forces, double-layer forces, etc. (Van Olphen, 1963). Therefore, the physicochemical behavior of cohesive particle sediments in aquatic ecosystems is much more complex than the behavior of granular materials. Estuarine bottoms, which are protected from direct current, tidal or wind action, may contain high percentage of fine material. Fine material (less than 60 μm) is comprised of fine silt, clay minerals and decomposed organic matter.

Clay particles have a diameter of less than 2 μm and a crystalline shape with a primarily sheet-like structure. However, needle-like and tube-like clay particles also exist. There are two main building blocks of clay minerals: the octahedral and the tetrahedral units (Chamley, 1989). The octahedral unit is comprised of aluminum (Al), iron (Fe), or magnesium (Mg) atoms enclosed equidistantly by six oxygen atoms (O_2) or hydroxyl molecules (OH), located on the vertices of a octahedral configuration (Figure 1.9). The tetrahedral unit is comprised of a silicon (Si) atom enclosed equidistantly by four oxygen atoms (O_2) or hydroxyl molecules (OH) located on the vertices of a tetrahedral configuration (Figure 1.10). Both units form sheet-like structures. The most common clay minerals along with their schematic molecular structure and chemical formulas are given in Table 1.6.

1.10.3.1 Aggregation-Flocculation. Due to the electrochemical forces, cohesive sediments tend to aggregate and flocculate. The mechanisms that promote aggregation-

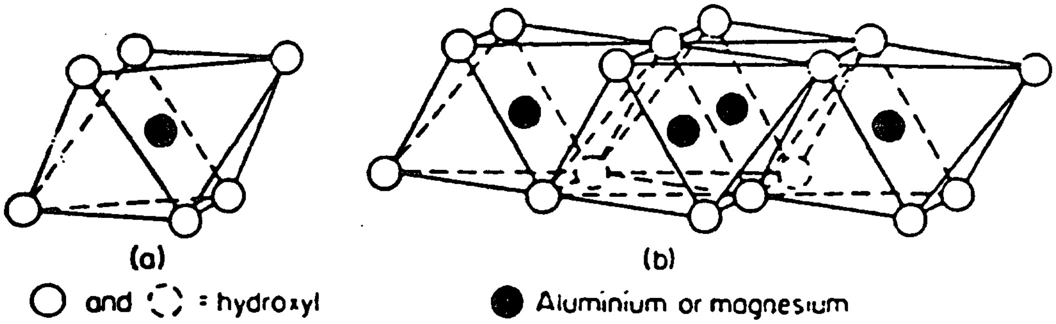


Figure 1.9 Octahedral unit structure of clay minerals (Partheniades, 1973b).

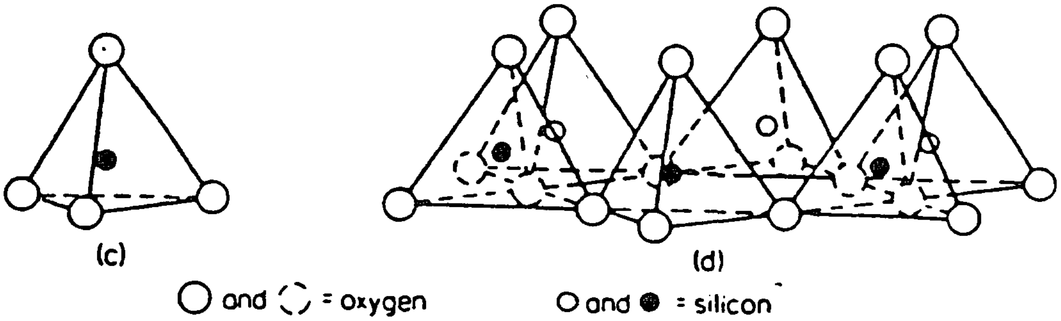


Figure 1.10 Tetrahedral unit structure of clay minerals (Partheniades, 1973b).

Table 1.6 Clay minerals (Van Olphen, 1977).

Name	Molecular structure	Chemical formula
Kaolinite	□◂□◂	$Al_4Si_4O_{10}(OH)_8$
Montmorillonite	◂▸□◂▸	$(Al_{1.67}\overset{\overset{Na_{0.33}}{\uparrow}}{Mg_{0.33}})Si_4O_{10}(OH)_2$
Illite	◂•▸□◂•▸	$K_y(Al_4Fe_4Mg_4Mg_6)Si_{8-y}Al_y)O_{20}(OH)_4$

◂ : octahedral unit; □ : tetrahedral unit; • : K molecules

flocculation are: Brownian motion, differential settling, and velocity gradients. Brownian motion develops weak and low density aggregates that can easily break under flow shear. Differential settling, caused by differences in particle fall velocities, develops weak and low density aggregates. The stronger aggregates are developed by moderate velocity gradients. High velocity shear causes aggregate disintegration (Krone, 1978). In high concentration environments (turbidity maximum), Dyer et al. (1990) observed a decrease in floc size, which they attribute to floc disruption through particle collision. In estuarine environments, it is very hard to establish a representative cohesive particle diameter because there is a substantial difference between the size of the individual particle and the sizes of aggregates and flocs (Eisma, 1990). The higher the order of aggregation, the less is the density of the aggregate.

During flocculation, particles are bonded together either edge-to-edge, or edge-to-face. A schematic representation of an edge-to-face flocculent at microscopic level is given in Figure 1.11. Possible aggregation-flocculation micro-structure states of

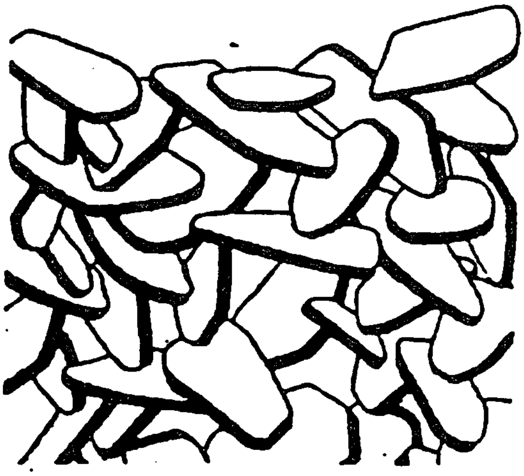


Figure 1.11 Edge-to-face particle aggregation.

estuarine sediments are as follows: a) dispersed and deflocculated, b) aggregated but deflocculated, c) edge-to-face flocculated but dispersed, d) edge-to-edge flocculated but dispersed, e) edge-to-face flocculated and aggregated, f) edge-to-edge flocculated and aggregated, g) edge-to-face and edge-to-edge flocculated and aggregated (Figure 1.12). Williams (1986) presented a mathematical model for representation of the various modes of particle interaction by using the free energy of interaction, V_T (i.e., the sum of the Van der Waals' attractive energy, V_A, and the double-layer coulombic energy, V_R). The processes of aggregation-flocculation depend on the chemical features of the soil and the

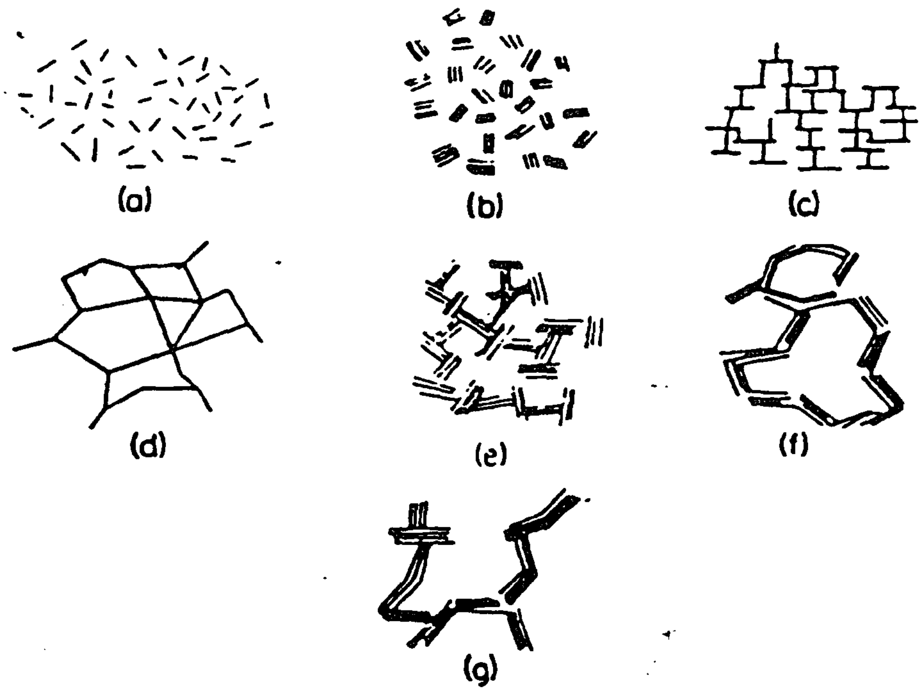


Figure 1.12 Possible aggregate-flocculate particle states.

ambient water. Salinity and organic matter promote aggregation (Partheniades, 1973a). Suspended particles are primarily attracted by general Van der Waal's forces. These particles remain in a stable, dispersed state by means of repulsive electric forces which counteract the attractive forces. The repulsive forces become less effective with increasing concentration of salt ions (Van Olphen, 1977). Since clay suspensions are considered as hydrophobic colloidal suspensions (sols), the rate of flocculation is substantially increased in the presence of even small amounts of salt. In salt-free environments flocculation still occurs but at an extremely low rate. Other chemical factors affecting particle flocculation are the pH of water, the sodium adsorption ratio (SAR), and the cation exchange capacity (CEC). In natural estuarine waters, pH is typically about 8. However, the presence of dense algal mats can increase pH to about

10 during photosynthesis or decrease it to around 5.5 during respiration (Montague, 1986). pH drastically affects the erodibility of abiotic sediments. Raising pH from 5.6 to 8.2 the critical shear stress for erosion was increased, in salt water, by an order of magnitude (Arulanandan, 1974). The sodium adsorption capacity is a measure of the ratio of exchangeable sodium ions to those of calcium and magnesium found in the diffusive layer of clay surfaces, i.e.,

$$SAR = Na^{+}/[\tfrac{1}{2}(Ca^{++}+Mg^{++})]^{1/2} \quad \dots\dots (1.4)$$

The higher the SAR, the greater is the mutual repulsion between clay particles (Ariathurai and Arulanandan, 1986). The range of SAR is from 0 to ∞; for SAR values greater than 14, clay particles are in a diffused state since Na has a lower valence and larger hydrated radius than Ca or Mg. Cation exchange capacity is defined as the capacity of negatively charged surfaces to attract cations and neutralize their negative charge. This is measured in milli-equivalents per 100 grams of clay mineral (meq/100 g). Typical CEC values for clay minerals are: kaolinites 3-8 meq/100 g, montorillonites 80 meq/100 g, illites 40 meq/100 g. However, CEC is a property of the sediments and not of the water system, while sediment deposition depends on the properties of both sediments and water (Partheniades, 1973b). In very low suspended sediment concentration environments, the effects of aggregation-flocculation are insignificant.

Organic material (humic substances) in aquatic ecosystems is comprised of decomposed flora and fauna. The four major types of organic matter include fulvid acids, humic acids, humins, and yellow organic acids (Jonasson, 1977). Organic substances are characterized by a large surface area, high cation exchange capacity (CEC), and a high negative charge.

Quantitative assessment of sediment dynamics requires knowledge of a number of physical parameters such as density, porosity, shear strength, angle of repose, coefficient of consolidation, effective cohesion, liquid and plastic limits, etc. Since the value of these parameters falls within a narrow range, in many practical situations an average value can be preselected (Table 1.7).

Table 1.7 Common properties of sediments.

Granular Material:	Gravels	Sands
Relative density	2.50-2.80	2.60-2.70
Bulk density (kg/l)	1.45-2.30	1.40-2.15
Dry density (kg/l)	1.40-2.10	1.35-1.90
Porosity (%)	20-50	23-35
Shear strength (kPa)	200-600	100-400
Angle of repose	35-45°	32-42°
Cohesive Matter:	Silts	Clays
Relative density	2.64-2.66	2.55-2.75
Bulk density (kg/l)	1.82-2.15	1.50-2.15
Dry density (kg/l)	1.45-1.95	1.20-1.75
Void ratio (%)	35-85	42-96
Liquid limit (%)	24-35	>25
Plastic limit (%)	14-25	>20
Coefficient of consolidation (m^2/yr)	12.2	5.0-20.0
Effective cohesion (kPa)	75	20-200
Effective angle of repose	32-36°	-
Organic Soils and Fill:	Peat	Coarse fill
Relative density	1.30-1.70	1.80-2.70
Bulk density (kg/l)	0.91-1.05	1.20-2.40
Dry density (kg/l)	0.07-0.11	1.05-2.00
Void ratio (%)	12.7-14.9	35-100
Liquid limit (%)	-	23-45
Plastic limit (%)	-	0-35
Moisture content (%)	650-1100	6-14
Effective cohesion (kPa)	20	20-50
Effective angle of repose	5°	28-40°

1.11 Sediment Dynamics.

Sediment dynamics is a very complicated phenomenon. In general, sediment dynamics is a time-dependent, multi-phase, nonlinear phenomenon subject to physical, chemical and biological effects. Sediments continuously undergo cycles of erosion, transport, deposition and resuspension. Of course, the sediment particles that participate in these cycles exchange continuously (i.e., particles continuously move in and out of the cycles). Based on their transportation mode, riverine sediments are classified as bed load or suspended load. In bed load, the sediment particles move by rolling, dragging, or making short jumps. In suspended load, particles are transported while in suspension (Simons and Senturk, 1977). There is no clear distinction between bed load and suspended sediments other than the time period that particles move continuously in suspension.

Based on their place of origin, sediments are classified as bed load or wash load materials. Bed load material is the part of the sediment that has the same composition as that of the soils in the riverine bed. On the other hand, wash load is the part that has compositions different from those in the riverine bed (Simons and Senturk, 1977). Bed load is comprised mostly of granular material while suspended sediments from cohesive inorganic and organic sediments. The technical approach of studying granular or cohesive sediments differs substantially.

1.11.1 Mechanics of Granular Materials. The main physical parameters that govern the mechanics of granular sediment transport depend on water and sediment physical characteristics as well as on riverine geometry. The parameters pertaining to the water phase are as follows: water density (ρ) and dynamic viscosity (μ); the parameters pertaining to the sediment phase are: sediment density (ρ_s), and particle diameter (D); and the riverine related characteristics are: width (B), depth (h) and bottom slope (S). In addition to these parameters, gravity (g) is very important. By using dimensional analysis (π-theorem), the following dimensionless parameters pertaining to sediment transport can be established (Yalin, 1977).

$$X_1 = \rho u_* D/\mu = u_* D/\nu \quad (1.5)$$

$$X_2 = \rho u_*^2/(\gamma_s' D) = \tau/(\gamma_s' D) \quad (1.6)$$

$$X_3 = \rho_s/\rho \quad (1.7)$$

$$X_4 = D/h \quad (1.8)$$

$$X_5 = D/B \quad (1.9)$$

where u_* is the shear velocity and γ_s' is the buoyant particle specific gravity. The shear velocity is defined according to the bottom shear stress τ as follows:

$$u_* = (\tau/\rho)^{1/2} = (ghS)^{1/2} \quad (1.10)$$

and the buoyant particle specific gravity as

$$\gamma_s' = g(\rho_s - \rho) = \gamma_s - \gamma \quad (1.11)$$

Parameter X_1 is the particle shear Reynolds number, Re_*, which demonstrates the relative importance of shear forces versus viscous forces. Parameter X_2 is called the mobility number and quantifies the balance between the traction and resistance forces. The physical meaning of the dimensional parameter, X_3, is related to particle ballistics, while parameters, X_4 and X_5, show the influence of riverine depth and width. Based on the these five parameters some property P pertaining to the mechanics of granular material transport can be expressed in a functional form as follows:

$$P = P(X_1, X_2, X_3, X_4, X_5) \quad (1.12)$$

1.11.1.1 Particle Incipient Motion. Using Equation 1.12, particle incipient motion can be effectively established. During incipient motion since no inertia is involved, the parameter X_3 can be neglected. Also, since the phenomenon is restricted to the bottom, both X_4 and X_5 can also be neglected. Thus, the resulting functional relationship for initiation of particle motion is reduced to

$$X_2 = f(X_1) \qquad (1.13)$$

For small X_1, since both u_* and ν are finite quantities, $D \rightarrow 0$. Thus,

$$X_2 = C_1/X_1 \qquad (1.14)$$

so that D cancels. Experimentally it was found that $C \approx 0.1$. For large X_1, viscosity effects are minimal so that ν is no longer a characteristic parameter (e.g., estuarine systems). But ν appears only in the X_1, so that X_2 has to be a constant independent of X_1,, i.e.,

$$X_2 = C_2 \qquad (1.15)$$

Experimentally it was found that for sand and gravel $C_2 \approx 0.05$. The aforementioned relationship between the shear particle Reynolds number (X_1) and the mobility number (X_2) establishes the well-known Shield's curve for particle incipient motion (Figure 1.13). One should keep in mind however, that this curve represents conditions of initiation of a general bottom movement and not of sporadically transported particles. Also, both theoretical and experimental results assume uniform and steady state flow conditions. An extension of the Shield's curve for extreme values of the particle shear Reynolds number was investigated by Wang and Shen (1985). The same authors also investigated incipient motion under wave conditions and the mobility number was modified in the following form:

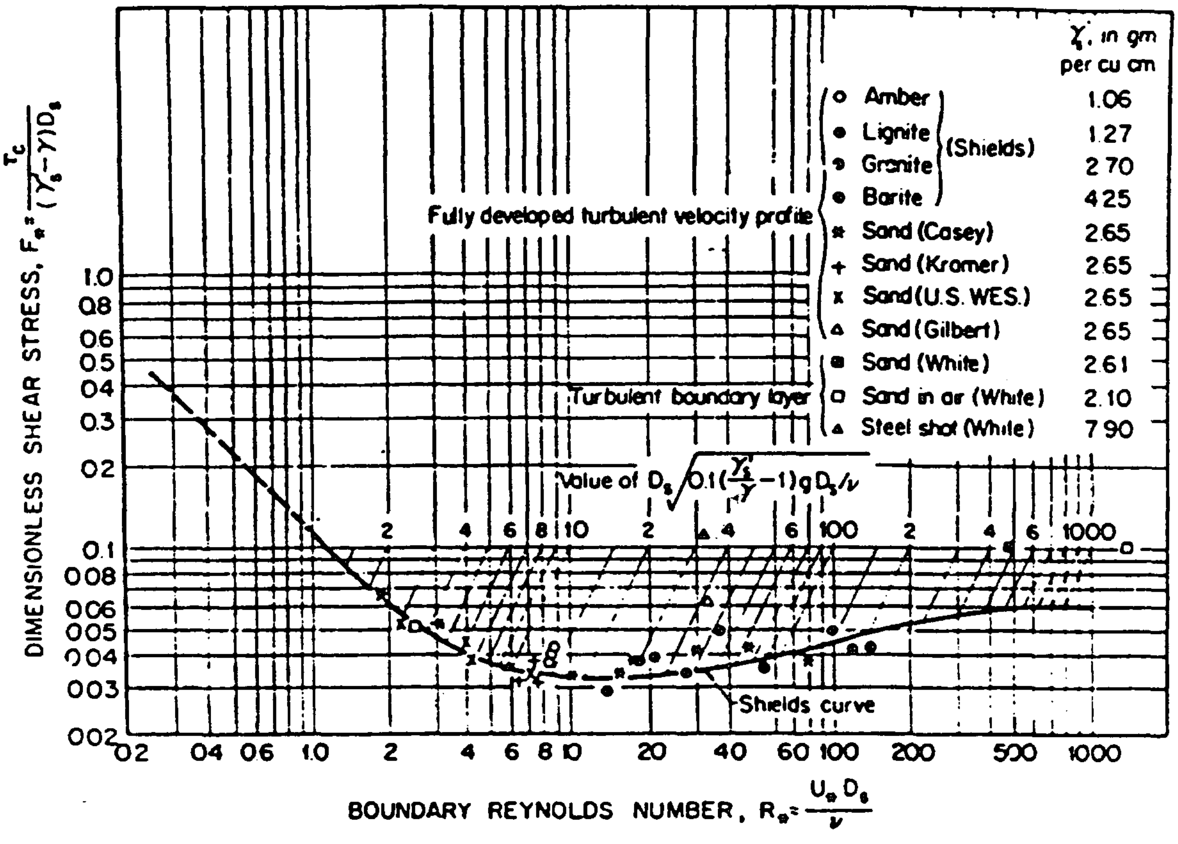


Figure 1.13 Shield's curve for particle incipient motion (Vanoni, 1975).

$$X_2 = \tau/(\gamma_s'D) = f\rho u_o^2/(2\gamma_s'D) \quad \ldots\ldots\ldots\ldots \quad (1.16)$$

where u_o is the amplitude of the orbital velocity and f is the wave friction factor. The velocity amplitude u_o is estimated as follows:

$$u_o = \pi\eta/[T\sinh(2\pi h/L)] \quad \ldots\ldots\ldots\ldots \quad (1.17)$$

where η is the wave height, T is the wave period, L is the wave length, and h is the mean water depth. Setting

$$a_o = \eta/[2\sinh(2\pi h/L)] \tag{1.18}$$

the friction factor can be expressed as follows:

$$f = f(a_o,\gamma) \text{ for rough bed, and} \tag{1.19}$$

$$f = f(a_o,u_o,\nu) \text{ for smooth bed.} \tag{1.20}$$

The above functional relations can be defined experimentally. The Reynolds number is modified as follows:

$$Re_* = (fu_o^2/2)^{1/2}D/\nu \tag{1.21}$$

For $3x10^3 \geq Re_* \geq 10^2$ the Shield's curve is the same for both unidirectional and wave motion; but for $Re_* < 10^2$, the particles are more resistant to erosion for wave motion than for unidirectional flow. This analysis neglects any electrochemical effects of the water-sediment system, since these effects are insignificant for granular materials.

1.11.1.2 **<u>Bedload Discharge.</u>** Once the sediment incipient motion is determined, then the next step is to estimate the rates of bed load transport. This can be achieved by a variety of existing methods. The most commonly used methods are those based on the Schoklitsch formula, the Meyer-Peter-Muller formula, the Bagnold's formula, and the Einstein-Brown and Van Rijn methods.

The Schoklitsch formula estimates the bed load by weight (q_{bw}) based on some "excessive" discharge, i.e.,

$$q_{bw} = 2500\ S^{3/2}(q-q_c) \tag{1.22}$$

where S is the bed slope, q is the flow discharge, and q_c is the critical flow discharge defined as follows:

$$q_c = 0.26(\gamma_s'/\gamma)^{5/3}D^{3/2}/S^{7/6} \tag{1.23}$$

Schoklitsch's formula is expressed in metric units.

The Meyer-Peter-Muller formula is written in terms of the submerged weight of the transported materials q_{bw}' in a dimensionless form as follows:

$$q_{bw}'/(\rho u_*^3) = 8[1 - 0.047\gamma_s'D/(\rho_s u_*^2)]^{3/2} \tag{1.24}$$

where the submerged weight is estimated as

$$q_{bw}' = q_{bv}\gamma_s' \tag{1.25}$$

and q_{bv} is the bed load discharge by volume.

The Bagnold's formula is based on energy considerations and yields

$$q_{bw} = (8.5e_b/S)(\tau - \tau_c)u_* \tag{1.26}$$

where e_b is an experimental coefficient and τ_c is a critical shear stress for initiation of erosion (Yalin, 1977). The value of e_b/S varies from 0.09 to 0.5 for particle diameter between 0.1 and 4 mm. The Meyer-Peter-Muller and Bagnold's formulas are both valid for small riverine slopes and for fine to medium sand beds.

Einstein (1950) was the first to introduce stochastic concepts in sediment transport mechanics. The Einstein-Brown method utilizes a dimensionless expression:

$$\Phi = f(1/\Psi) \tag{1.27}$$

where Φ is the sediment transport rate function and Ψ is the inverse of the mobility number, i.e.,

$$\Phi = q_{bw}/[\gamma_s K_E(\gamma_s'D^3/\rho)^{1/2}] \tag{1.28}$$

$$\Psi = \gamma_s'D/\tau \tag{1.29}$$

The constant K_E is defined as follows:

$$K_E = [2/3 + 36\rho\nu^2/(\gamma_s'D^3)]^{1/2} - [36\rho\nu^2/(\gamma_s'D^3)]^{1/2} \tag{1.30}$$

The relationship between Φ and $1/\Psi$ is presented in Figure 1.14 (Raudkivi, 1976). Using the $\Phi \sim 1/\Psi$ curve, the bed load discharge can be estimated directly.

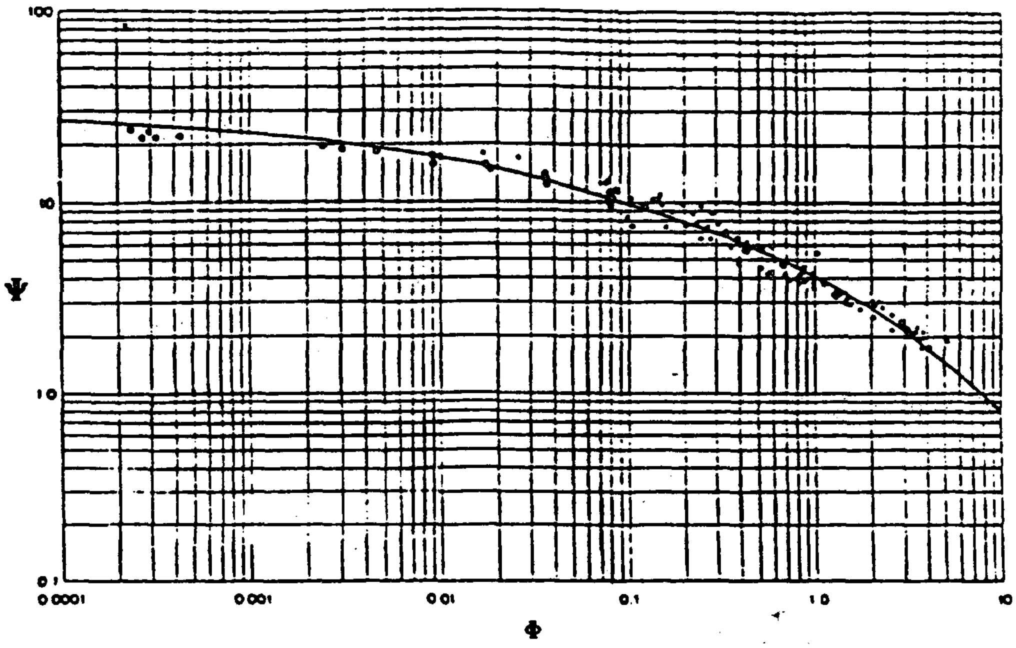


Figure 1.14 Transport rate function Φ versus mobility number Ψ (Raudkivi, 1976).

One of the latest methods for estimation of bed load and suspended granular materials in alluvial channels was developed by Van Rijn (1982). Van Rijn's bedload estimation is based on the relation

$$q_{bv}/[(\gamma_s'/\rho)^{1/2}D_{50}^{3/2}] = 0.053T^{2.1}/D_*^{0.3} \quad \ldots\ldots\ldots\ldots \quad (1.31)$$

where

$$D_* = D_{50}[\gamma_s'/(\rho\nu^2)]^{1/3} \quad \ldots\ldots\ldots\ldots \quad (1.32)$$

$$T = (u_*^2 - u_{*c}^2)/u_{*c}^2 \quad \ldots\ldots\ldots\ldots \quad (1.33)$$

The critical shear velocity u_{*c} can be estimated by using the dimensionless parameter Θ and Table 1.8, where

$$\Theta = u_{*c}^2/\{[(\rho_s/\rho)-1]gD_{50}\} \quad \ldots\ldots\ldots\ldots \quad (1.34)$$

Table 1.8 Parameter Θ versus D. (Van Rijn, 1982).

For	$D. \leq 4$	$\Rightarrow$	$\Theta \approx 0.240 D.^{-1.00}$
For	$4 < D. \leq 10$	$\Rightarrow$	$\Theta \approx 0.140 D.^{-0.64}$
For	$10 < D. \leq 20$	$\Rightarrow$	$\Theta \approx 0.040 D.^{-0.10}$
For	$20 < D. \leq 150$	$\Rightarrow$	$\Theta \approx 0.013 D.^{0.29}$
For	$150 < D.$	$\Rightarrow$	$\Theta \approx 0.055$

Van Rijn's formula requires the use of metric units and is valid for particle diameters between 200 and 2000 mm. A detailed discussion on the performance of the various bed load methods can be found in Simons and Senturk (1977).

1.11.1.3 Bedforms. Another important issue that should be considered during studies of sediment dynamics in alluvial channels is bed formation. Bed forms are observed in sandy soils and depend strongly on the flow regime (Table 1.9). Since flow conditions within the main St. Johns River are within the subcritical range, the bed forms are anticipated to be either ripples or dunes. However, this may not be true for the

Table 1.9 Types of bedforms (Simon and Senturk, 1977).

Bedform Type	Flow Regime	Remarks
Ripples	Subcritical	Particles migrate downstream; less than 1 ft in length; water surface is not effected.
Dunes	Subcritical	Particles migrate downstream; much larger than ripples; surface water waves out of phase with bed form waves.
Antidunes	Supercritical	Particles can migrate upstream, downstream, or form a standing wave; surface water waves are in phase with bed form waves; very unstable.

upstream regions of some tributaries immediately after a storm event. Simons and Richardson (1966) developed a graph where the various bed form regimes are identified based on the stream power and median particle diameter D_{50} (Figure 1.15).

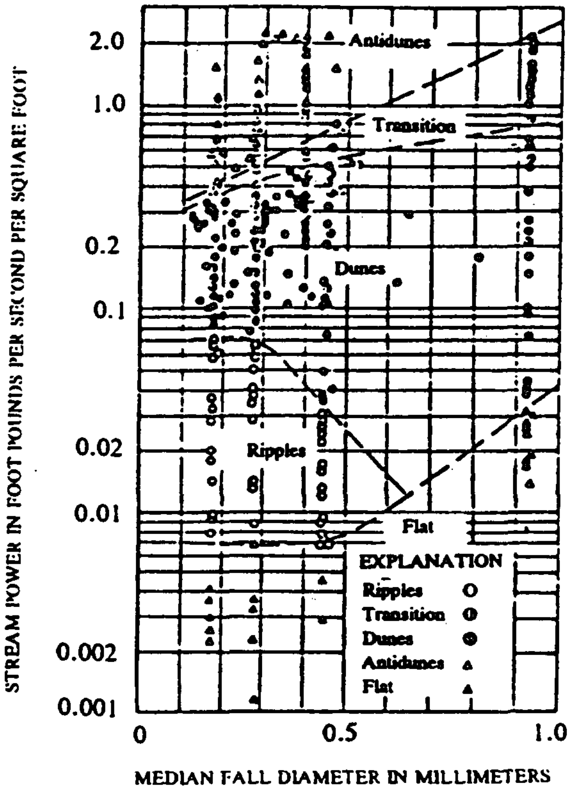


Figure 1.15 Simon and Richardson classification of bed forms (Simon and Richardson, 1966).

The stream power is defined as the product between shear stress τ and mean velocity u. A comprehensive analytical study on ripple and dune formation has been given by Haque and Mahmood (1985, 1986). These investigators considered that the most discriminating features of ripples and dunes are the separation point, the stagnation point, and the free-streamline, and based their analysis on the free-streamline method of Helmholtz (Figure 1.16).

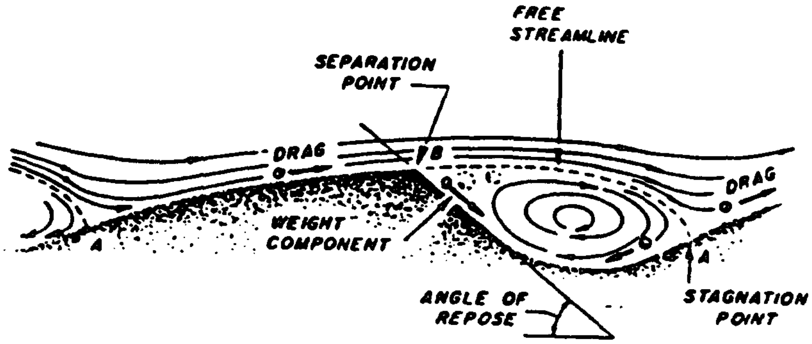


Figure 1.16 Schematic sketch for ripples and dunes (Haque and Mahmood, 1986).

Ripple formation under wave action was studied by Vongvisessomjai (1984). It was concluded that ripple geometry depends on the ripple function Ψ_R defined as follows:

$$\Psi_R = F_{d*}^{\alpha} A_*^{\beta} \tag{1.35}$$

where

$$F_{d*} = u_w^2/[(\gamma_s/\gamma - 1) g D_m] \tag{1.36}$$

$$A_* = a_w/D_m \tag{1.37}$$

and u_w is the wave-induced velocity amplitude, a_w is the orbital amplitude of water particle motion outside of the laminar boundary layer, and α, β are experimental exponents. Then, the ratio of ripple wave height a_R to ripple wave length L_R is given by

$$a_R/L_R = A\,\Psi_R^{r} \tag{1.38}$$

where A and r are experimental constants. Understanding of riverine bed formation is essential for a correct estimation of flow resistance. Indeed, besides the skin friction, bed forms induce an additional resistance on the flow. There are two main approaches that have been used to estimate bed form effects. One approach is based on a modified energy gradient and the other on a modified hydraulic radius approach. A representative example of the former approach is the Engelund and Hansen method (1967), and for the latter approach the Einstein-Barbarossa method (1952).

The algorithm of the Engelund and Hansen method is as follows: Given the depth (h) and slope (S) of the river, a parameter θ is computed as

$$\theta = hS/\{[(\rho_s/\rho)-1]D\} \quad \ldots\ldots \quad (1.39)$$

Then another parameter θ' is estimated based on the following relations

$$\theta' = 0.06 + 0.4\theta^2 \ \text{ for } \ \theta \leq 0.8 \quad \ldots\ldots \quad (1.40a)$$

$$\theta' = 0.4\theta^2 \ \text{ for } \theta > 0.8 \quad \ldots\ldots \quad (1.40b)$$

if the bed forms are dunes ($\theta' \leq 0.5$), and

$$\theta' = \theta \quad \ldots\ldots \quad (1.41)$$

if the bed forms are antidunes. Then an equivalent depth h' is estimated using the equation

$$\theta' = h'S/\{[(\rho_s/\rho)-1]D\} \quad \ldots\ldots \quad (1.42)$$

Finally, the velocity is obtained from the relation

$$\theta = \theta' + [u^2/(2gL)](a/h)2 \quad \ldots\ldots \quad (1.43)$$

where L is the wave length and a is the amplitude of the bed form.

The algorithm for the Einstein-Barbarossa method is as follows: given the flow discharge Q and the bed slope S, a hydraulic radius R_h' is assumed and the function Ψ' is estimated as follows:

$$\Psi' = \gamma_s'D_{35}/(R_h'S) \quad \ldots\ldots \quad (1.44)$$

Then from Figure 1.17, the value of $u/u.''$ is obtained and subsequently R_h'' is calculated

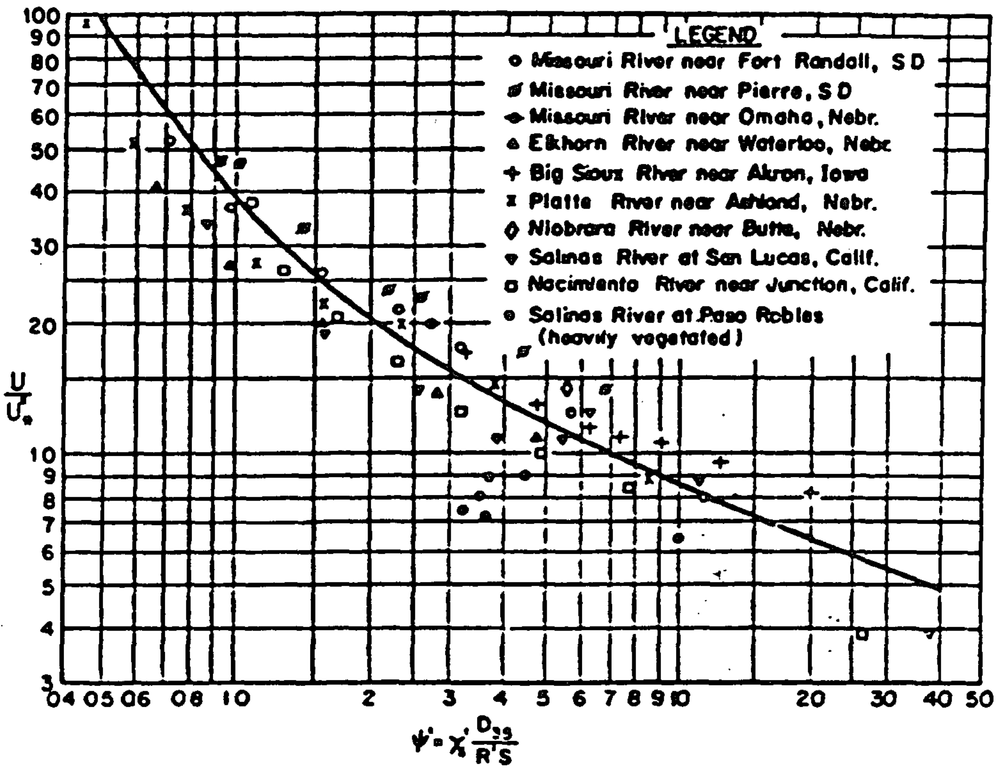


Figure 1.17 Einstein-Barbarossa's function (Einstein-Barbarossa, 1952).

by using the relationship

$$R_b'' = u_*''^2/(gS) \qquad (1.45)$$

The algorithm is repeated until

$$R_b = R_b' + R_b'' \qquad (1.46)$$

Bedforms are not found in beds comprised of cohesive materials. The aforementioned discussion on granular materials is applicable to areas where sandy bottoms are exposed, i.e., near the inlet, the dredged navigation channel, and sections where strong current velocities occur. For these areas of exposed sandy beds, the analytical methods for granular sediments can be applied even if they are tidally influenced. This can be achieved by separating the tidal cycle into ebb and flood flow.

1.11.2 **Cohesive Sediment Dynamics.** The behavior of cohesive sediments is more complicated than that of the granular material. Due to their small particle size, cohesive sediments resuspend easier, remain in suspension for long time periods, and bind to each other forming aggregates and flocculates. The sedimentation cycles of cohesive sediments involve erosion, advective transport, diffusion-dispersion, aggregation-flocculation, deposition, consolidation, and resuspension (Scarlatos and Partheniades, 1986). While in suspension, cohesive sediments do not have a uniform vertical concentration profile. Suspended cohesive sediments in aquatic systems are normally found in a stratified state comprised of seven different layers (Ross and Mehta, 1989; Mehta, 1989). Counting from the top, these layers include: the mixed layer mobile suspension, the stratified mobile suspension, the lutocline shear layer, the mobile hyperpycnal layer (fluid mud), the stationary mud, the deforming cohesive bed, and the stationary cohesive bed (Figure 1.18). The mixed layer mobile suspension behaves as a Newtonian fluid and the particles are in a dispersed state. The stratified mobile suspension demonstrates increasingly non-Newtonian behavior and the effect of flocculation on particle settling. In the following three layers (i.e., lutocline, fluid mud and stationary mud), the rheological properties are highly non-Newtonian and settling is hindered by a weak soil matrix. Particles in the upper five layers are suspended and do not have any effective stress. In the two bottom layers (i.e., deforming and stationary bed) there is a measurable effective stress introduced by a strong soil structure. Within these two layers, dewatering and consolidation effects take place.

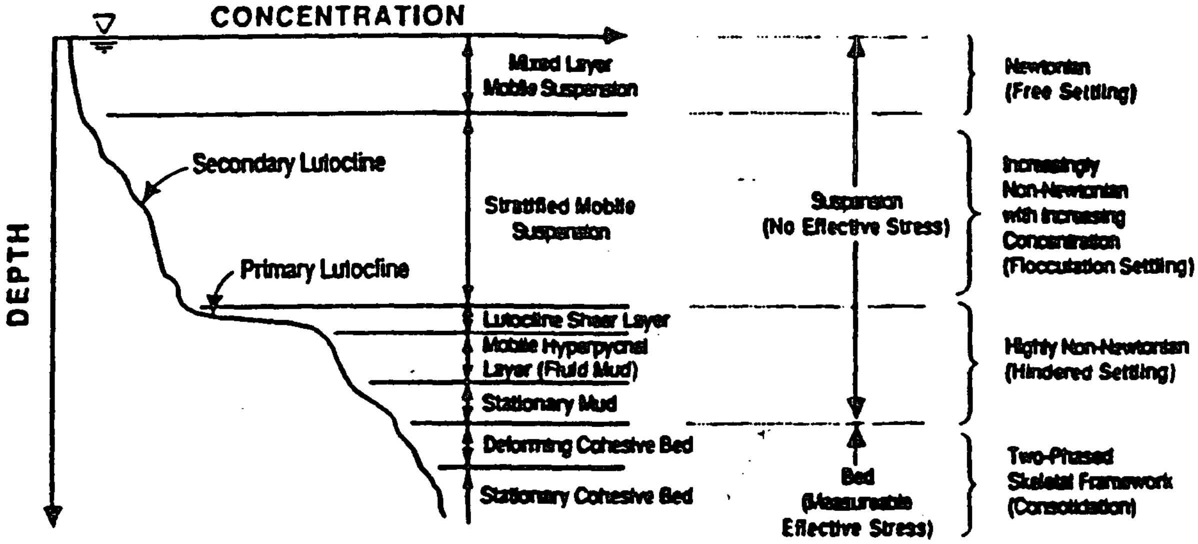


Figure 1.18 Typical suspended sediment profile (Mehta, 1989).

The lutocline is the zone of steep suspended sediment concentration gradient. Suspended particles within the fluid mud layer are supported partially by turbulent flow fluctuations, and partially by a weak soil matrix established between the aggregates. Recognizing the existence of the fluid mud layer is very important for environmental studies, since fluid mud resuspends very easily and can cause substantial pollution.

1.11.2.1 Sediment Erosion. Erosion of settled beds of cohesive sediments depends on the magnitude of the bottom shear stress. Once the shear stress exceeds a critical value (τ_c), the material from the bottom starts to erode and entrain into the water column. Erosion processes show different behavior for uniform beds and for beds resulting from differential deposition. This is due to the fact that in the former case properties are uniform along the bed thickness, while for the latter case erosive resistance increases with increasing depth (Partheniades, 1986). Physicochemical properties, benthic communities, and time history of the deposited bed are all important to the erosion processes (Montague, 1986). Increased NaCl concentrations result in increased bottom yield stresses of clay deposits (Williams, 1986). Also, the bottom shear stress of clay minerals increases with increased consolidation time (Partheniades, 1986). The temperature effect on erosion of kaolinite clays by deionized water was studied by Raudkivi and Hutchison (1974). From their experiments it was concluded that erosion rate is a U-shaped function of temperature (i.e., initially, the erosion rate decreases with increasing temperature until it reaches a minimum value for a critical temperature; after that temperature the erosion rate increases with further increasing temperature). However, experiments conducted by Kelly and Gularte (1981) on illite clays, eroded by saline water, have shown a monotonically decreasing rate of erosion for increasing temperature. The same investigators demonstrated the existence of an exponential relationship between critical shear stress and the number of interparticle bonds. Another factor affecting the erosivity of cohesive sediments is the sodium adsorption ratio, SAR. Experiments on Yolo-Loam (i.e., mixture of sand, silt, montmorillonite, kaolinite, mica and vermiculite) indicated that under the same shear stress, the rate of erosion increases with increasing SAR (Arulanandan et al. (1975). For organic materials, erosive properties are expected to depend on the redox potential. However, there is not any available quantitative relation between the rate of erosion and the redox potential.

For uniform beds, the erosion rate (S_e) is linearly related to the excessive bed shear stress (Arulanandan, 1974),

$$S_e = M(\tau/\tau_c-1)/h \quad \text{for } \tau > \tau_c \quad \ldots\ldots\ldots\ldots \quad (1.47)$$

where M is the erodibility constant that incorporates the physicochemical features of the bed.

For stratified beds, erosion tests revealed a rate of erosion which exponentially decreases in time (Mehta, 1986),

$$S_e = C_s b \exp(-bt) \quad \ldots\ldots\ldots\ldots \quad (1.48)$$

where C_s and b are parameters depending on sediment characteristics. Additionally, C_s is linearly related to the bottom shear stress. The above rates of erosion are valid only for settled, consolidated beds. For modeling purposes, the bed can be treated as a stratified system comprised of layers of different thickness and erosive characteristics. Erosion of the fluid mud layer follows a different pattern from the one observed in consolidated beds. Generally, resuspension of fluid mud can be simulated as an interfacial instability problem between two fluids of different density. Under parallel flow conditions, breaking of the interface resembles the Kelvin-Helmholtz or Holmboe instability modes (Scarlatos and Mehta, 1990).

1.11.2.2 <u>**Sediment Deposition.**</u> Similarly to the erosion rates, the rate of deposition (S_d) is related to some shear stress as (Krone, 1962):

$$S_d = -w(1-\tau/\tau_{dm})C/h \quad \text{for } \tau < \tau_{dm} \quad \ldots\ldots\ldots\ldots \quad (1.49)$$

where τ_{dm} is the threshold shear stress for deposition to occur, C is the sediment concentration, and w is the particle fall velocity. The above formula has been extended in order to incorporate the effects of different particle size groups, i.e., aggregates and flocculates, as (Krone, 1986)

$$S_d = \sum_{i=1}^{i=N} [w_i(1-\tau/\tau_{dmi})/h]C_i \qquad (1.50)$$

where the sub-index i indicates each of the N different particle groups, and C_i is the concentration of each subgroup. Therefore,

$$C = \sum_{i=1}^{i=N} C_i \qquad (1.51)$$

where C is the total suspended sediment concentration. Another expression for the deposition of cohesive sediments was developed by Partheniades (1971). Partheniades' analysis is based on results from a non-advective system (recirculating channel) and requires a variety of physical inputs. If treated properly however, Partheniades' expression can be applied to advective systems (Scarlatos and Partheniades, 1986).

An interesting feature of cohesive sediment deposition is the dependence of the particle fall velocity on the suspended sediment concentration (hindering settling). Experimental data supporting the concept of hindered settling are presented in Figure 1.19 (Thorn, 1981). Deposition is also affected by the salinity content of the ambient water. However, in spite of the fact that laboratory experiments have shown a well-defined direct relationship between fall velocity and salinity content (Krone, 1962), *in-situ* measurements in Thames estuary were inconclusive (Figure 1.20). At this point it should be emphasized that modeling of the erosion and deposition processes require extensive field data and laboratory analysis of water and sediment samples in order to characterize the various physicochemical properties of the system and quantify the pertinent parameters.

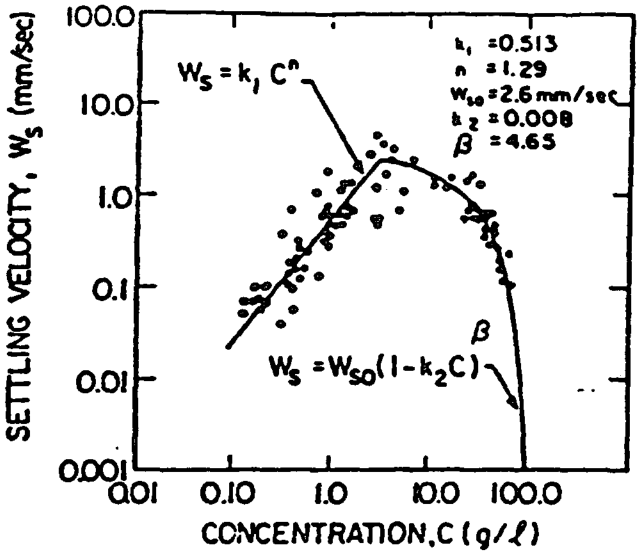


Figure 1.19 Hindered settling (Mehta, 1986).

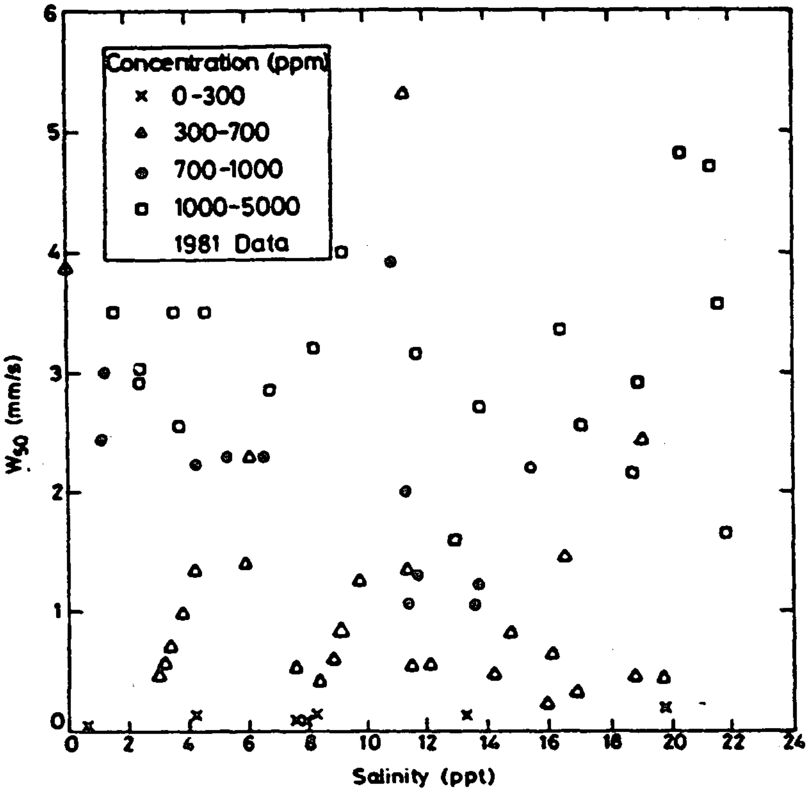


Figure 1.20 Salinity effects on settling velocity (Burt, 1986).

1.12 Sediment Modeling.

1.12.1 Cohesive Sediment Dynamics Modeling.

For quantification of riverine dynamics, a number of mathematical models have been developed. These models are based on different assumptions, have different input requirements, and their applicability and performance varies for each case. All of these models require information of the hydrodynamic field. The hydrodynamics of the system can be provided either from field measurements and/or historical records, or by numerical solution of the hydrodynamic equations, i.e., mass continuity and momentum balance equations (Pinder and Gray, 1977).

All of the sediment transport models are based on a generalized mass balance equation. Due to the complexity of the system, bed load and suspended matter are treated separately. For estimation of the bed load discharge any of the aforementioned formulas and approaches can be utilized. For detailed estimation of the suspended sediment transport, the full advection-dispersion equation should be applied. In one-dimensional form this equation reads

$$\partial C/\partial t + u\partial C/\partial x - \partial[D_L\partial C/\partial x]/\partial x + S_d = S_e \quad \ldots\ldots\ldots\ldots\ldots \quad (1.52)$$

where C is the suspended sediment concentration and D_L is the dispersion coefficient. In the following, various mathematical models for simulation of cohesive sediment transport are discussed.

Sayre (1969) and Jobson and Sayre (1970) studied, both theoretically and experimentally, the vertical distribution of silt and fine sand particles under turbulent shear flow in uniform two dimensional open channel flow. The study involved advection only; erosion and deposition were not included. As initial conditions, they considered an instantaneous plane source uniformly distributed along the cross section. Their theory is valid only for particle fall velocities within the Stoke's range. However, they found that for particles of sizes less than 100 μm, the Reynolds analogy holds, i.e., momentum and mass diffusivities can be considered to be equal (Schmidt number = 1).

One of the first attempts to include erosion, deposition, and transport into a single model was done by Odd and Owen (1972), and Owen and Odd (1972). This was a one-dimensional, two-layered mathematical model of mud transport for the well-mixed Thames estuary. The regular configuration of the estuary allowed a simplified geometric description of a rectangular cross section with an exponentially varying width. The main purpose of the model was to predict the siltation regime in a system of various tidal barriers proposed for flood protection. Based on the fact that flow and sediment concentration change significantly over the depth and especially near the bottom, Odd and Owen assumed two bottom layers in which internal circulation, induced by density currents took place. The properties in each layer were taken as uniform. The lower layer was of constant thickness and much smaller than the upper layer which had variable thickness. The equations of water mass continuity and momentum were solved by using a finite difference scheme, while for the sediment balance the method of characteristics was applied. The rate of deposition was taken from the experimental work of Krone (1962), and for the rate of erosion a linear expression was used (Arulanandan, 1974). The disadvantage of the model is that it neglects advection and that depth-averaging causes distortion of the inertia effects of tidal currents.

Ariathurai (1974) developed a time-dependent, two-dimensional model using the advection-dispersion equation. The physical domain was subdivided into a finite element grid and solution was obtained by means of Galerkin's weighted residual method. Erosion and deposition were included in the form of source and sink terms. The expressions for these terms were similar to the ones used by Odd and Owen. Depending on the situation, the model could be used either as depth-averaged or as width-averaged. Flow dynamics, diffusion coefficients and certain sediment properties must be provided as input in this model. Also, the settling velocity of the suspended phase must be known *a priori*. By assigning appropriate settling velocities for each time step, the model can account also for particle aggregation. Another important feature introduced in this model was the differentiation between surface erosion, i.e., layer by layer, and mass erosion which occurs when the stress exceeds the bulk strength of the bed. The bed was assumed to be comprised of various layers with variable shear strength. Consolidation effects were incorporated according to the experimental relation proposed by Bosworth (1956). The validity of the model was tested against data from the Savannah estuary. Agreement between simulated results and field data demonstrated the applicability of the model. From a numerical point of view the solution scheme is unconditionally stable. However, the accuracy of the solution depends on the Peclet number, discretization of the flow field and time step. Improvement of the solution can be attained by introducing the water elevation as one of the dependent variables and not assume it as constant (lid-type model).

Christodoulou et al. (1974) developed another model for simulation of sediment dispersion in coastal waters. This was a quasi-, three-dimensional analytical model with constant flow depth, where a sinusoidal tidal current was superimposed at any arbitrary angle to the net flow. A separation of variables technique was applied so that the vertical distribution of sediment concentration was treated separately. The vertical concentration was estimated from the average concentration value. The equilibrium concentration profile along the vertical direction was computed by assuming a single layer shear flow. Sediment was introduced into the system from a vertical line source and each particle that reached the bottom was assumed to stick to it. The processes of scouring and aggregation were not included in this model. For the settling velocities of all sediment fractions but clay, Stokes law was utilized. For the clay, fall velocities were taken from laboratory tests in settling tubes. Dispersion coefficients were directly related to the shear velocity. This model worked very satisfactorily when applied for the Massachusetts Bay. However, it is recommended only for simulation of systems with slowly varying velocity fields.

Another attempt at modeling the circulation of fine sediments in estuaries was done by Hunter (1975). Hunter developed a three-dimensional pseudo-dynamic numerical model for simulation of suspended sediments in the upper Chesapeake Bay. His model was able to predict, from the sediment concentrations, the source of any suspended material that was subjected to advection, diffusion and deposition, but no resuspension. The suspended material was restricted to the passive fraction of the sediments in order to avoid primary production of suspended matter. However, as Hunter pointed out the results were not very useful as they did not provide any new insight into the sediment transport problem.

An improvement of Ariathurai's model was done by Ariathurai et al. (1977) by introducing isoparametric quadrilateral finite elements. However, although the numerical approach became more sophisticated, there was no improvement in the understanding of the physical description of the phenomenon since the rates of erosion and deposition were the same as those used in the original model.

Schubel and Carter (1977) developed a simple, "control-volume" type model for prediction of net exchange of suspended sediments between Chesapeake Bay, its tributaries, and the ocean. Water, suspended sediment, and salt were transported along an upper and lower layer within a single control volume. Advection was the main transport process for water and sediment; salt was assumed to be transported by both advection and diffusion. The advective and diffusive fluxes of salt were estimated first. Then the advective water flux through the upper layer was computed and finally the suspended sediment in both layers was determined. With this simple model, the authors demonstrated that the ocean is a sediment source while the tributaries are sediment sinks for the Chesapeake Bay. This model is not recommended for general use.

Another investigation on numerical modeling of fine sediment transport was done by Kuo et al. (1978). This is a two-dimensional, time-dependent model predicting sediment motion in the turbidity maxima zone of an estuary. The governing equations of this model are the equations of mass continuity and momentum balance. The equations were averaged over the lateral dimension. The solution was obtained by a finite difference scheme. The output of the model estimated tidal velocities, salinity and suspended sediment concentrations. The rates of erosion and deposition were given by the same expressions that were utilized in Ariathurai's model. The tidal part of the model worked efficiently for the cases that was tested for. The part of the model pertaining to sedimentation can be used only if the various parameters of the suspended material are well established and evaluated.

Tuxford (1980) and O'Connor and Tuxford (1980) presented a two-dimensional cohesive sediment model based on the advection-dispersion equation in which both erosion and deposition were included. In their model they expressed the deposition rate in a more improved form since it referred to near bottom and not to a depth-averaged concentration. For erosion they considered a surface erosion mechanism along with sediment re-entrainment that was approximated as a percentage of the deposited material. The numerical solution was based on an alternative-direction implicit finite difference scheme which was reported to be very efficient. Application of the model for simulation of a laboratory study case showed good agreement with the observed data.

Onishi (1981) developed a vertically-averaged, two-dimensional sediment transport model. In his study, both cohesive and granular sediments were included. The expressions for erosion and deposition were taken from Arulanandan (1974) and Krone (1962), respectively. The bed was divided into a number of layers of "normal" thickness. Each layer consisted of sediments with three different particle sizes and specific characteristics. The top layer always had a thickness equal or less than the "normal" thickness. Thus, during the sedimentation processes either a new top layer was forming when the old top layer exceeded the "normal" thickness, or when all of the top layer was eroded the layer immediately under was starting to erode too. The whole mechanism was assumed to be regulated by the capacity of the granular material to erode or deposit, so that the cohesive sediments were supposedly covered and protected by the non-cohesive ones. The model simulated successfully the sedimentation processes in James River.

Partheniades (1980) suggested guidelines for simulation of cohesive sediment erosion and deposition according to functional relations derived from experimental data (Mehta and Partheniades, 1979). In accordance with these guidelines Scarlatos (1981) developed a one-dimensional model for cohesive sediment transport. The model was based on the advection-dispersion equation which included the source and sink terms suggested by Partheniades. The solution was obtained by utilizing the Crank-Nicholson finite difference scheme. Application of the model requires calibration of certain parameters from field or laboratory data. The model was tested for the Savannah estuary. The simulation results have shown agreement with field data.

Cole and Miles (1983) developed a horizontally-averaged, two-dimensional numerical model for simulation of well-mixed suspended sediments in estuarine and coastal waters. The model utilizes a simple expression for deposition (Krone, 1962), while erosion is not included explicitly. Simulation of sedimentation within the Conwy estuary in England, using Cole and Mile's model showed satisfactory results. However the model is not of general use because erosion and deposition processes are not properly represented.

Hayter (1983) presented the most complete model to date of cohesive sediment transport which accounts for erosion, deposition and bed consolidation. The model is two-dimensional, depth-averaged and was developed according to the computational guidelines of the Ariathurai's model (1974). In Hayter's model, the bed is discretized into three different layers: unconsolidated stationary suspensions, partially consolidated, and fully consolidated beds. Erosion is considered for each layer separately. The consolidation algorithm accounts for self-weight consolidation in the upper two bed layers by increasing the bed density and bed shear strength while decreasing bed thickness with time. The model has been verified by a series of laboratory experiments and showed close agreement between simulated and measured data.

Sheng (1983) presented a three-dimensional model for simulation of coastal currents and sediment dispersion. For the erosion rates Sheng used the exponential function given by Mehta (1986), while for deposition he considered the effects of vegetation canopy, laminar sublayer, and biochemical resistance on particle fall velocity. The flow turbulence is modeled according to the second-order closure scheme proposed by Donaldson (1972). Sheng applied his model for simulation of a section of the Mississippi River. The results showed agreement with field data.

Maa (1986) developed a model for estimation or erosion rates under wave action. In his model Maa, introduced the rheological effects of the bed. Indeed, for hyperconcentrated flows, the behavior of the water-sediment mixture is no longer Newtonian but resembles either a Bingham or pseudoplastic fluid (Williams and Williams, 1989). Compared to experimental data the model performed very satisfactorily.

A recent model for estimation of resuspension of fluid mud under parallel flow conditions was developed by Scarlatos and Mehta (1991). This model is based on vortex dynamics, where the rates of entrainment were quantified by estimating vortex interaction, growth, and stretching along the lutocline.

All of these cohesive sedimentation models were developed for estuarine and coastal waters; thus they are all applicable to the Lower St. River Johns River. Before their adoption for the LSJR however, they would require a number of laboratory and field data for adjustment of the various parameters involved. The selection of an appropriate sedimentation model for the Lower St. Johns River should depend on the particular needs and goals of the study. Before selection, special consideration should be given to the following issues:

- o time period of simulation (daily, weekly, seasonal, annual, multi-year),
- o spatial resolution (near-, meso-, or far-field; microscopic, macroscopic),
- o available information and additional data needs of the water-sediment system (physical, chemical, microbial characteristics),
- o laboratory facilities for data analyses,
- o number and degree of complexity of constants and parameters to be calibrated,
- o hydrodynamic information and modeling (currents, turbulence, boundary layer),
- o hydrologic input of boundary conditions (winds, ET, precipitation, groundwater seepage, tributary inflow),
- o stochastic or deterministic character of the hydrologic input data,
- o user-friendliness and transparency (i.e., explanations on assumptions made, methodology applied, applicability, etc.) of the model, and
- o computational limits.

1.12.2 Remarks on Modeling of Turbulent Flows. Quantitative simulation of turbulent flows is a very difficult subject. The governing equations are highly nonlinear, time-dependent partial differential equations. Traditionally, turbulent flow problems were thought to be of entirely stochastic nature and they were studied by means of probabilistic methods. In the last two decades, coherent turbulent structures were observed and deterministic approaches were applied for the solution of these flows. Hydrodynamic models of high Reynolds number turbulent flows are still open to question. The main difficulty in the solution stems from the closure problem of the Navier-Stokes equations. Navier-Stoke equations written in tensor form read as follows:

$$(\partial/\partial t-\nu\nabla^2)u_i = -(\partial u_i/\partial x_j)u_j - \rho^{-1}\,\partial P/\partial x_i \quad (i,j = 1,2,3) \quad \dots\dots\dots\dots \quad (1.53)$$

In a highly symbolic fashion, the same equations can be written as

$$L_o u = L_1 uu + L_2 P \quad \dots\dots\dots\dots\dots\dots\dots\dots \quad (1.54)$$

where L_o, L_1, L_2 are operators. The pressure P is related to the velocity u through the continuity equation. By averaging each term, < >, we obtain a relation for the mean velocity

$$L_o\langle u\rangle = L_1\langle uu\rangle + L_2\langle P\rangle \quad \dots\dots\dots\dots\dots\dots \quad (1.55)$$

Therefore, in principle the solution for <u> depends on the second moment <uu>. By repeating this process, i.e., multiplying by u and taking the time-average, we can obtain a hierarchy of moment equations

$$L_o\langle uu\rangle = L_1\langle uuu\rangle + L_2\langle uP\rangle \quad \dots\dots\dots\dots\dots \quad (1.56a)$$

$$L_o\langle uuu\rangle = L_1\langle uuuu\rangle + L_2\langle uuP\rangle \quad \dots\dots\dots\dots \quad (1.56b)$$

..... etc .....

Thus, there is a set of n equations for n+1 moments, since each moment depends on the one of higher order. The process of closing this moment hierarchy constitutes the so-called closure problem. The two main closure models are the k-ϵ model and the Reynold stress (second order) model. These models are based on single point statistics which implicitly imply that there is an orderly transport from large vorticity scales to energy dissipating (viscous) scales which are taken as isotropic. Therefore, single point closure models cannot apply to mixing of two different turbulent streams with significantly different turbulent scales. Another important limitation of the single point models is their assumption of local flow homogeneity. This assumption is strongly violated whenever there is a body within the flow and the energy carrying eddies are of equal or greater size than this body (Taulbee, 1989). In order to simulate a full scale turbulence, the number of mesh points N are given approximately as (McComb, 1990)

$$N \approx 0.77\, R_e^{2.25} \qquad (1.57)$$

From this expression one can easily see that for large Reynolds numbers, which is the case for estuarine flows, the capacity of any available computer system will easily overflow. To overcome this difficulty large-eddy simulation models were developed and applied. Alternatives to the Navier-Stokes equations for simulation of turbulence include the discrete vortex approach and lattice gas models (McComb, 1990).

Concluding, it should be emphasized that any model regardless of how sophisticated and complete it may appear to be, is always a gross approximation of the natural system. Also, many of the governing factors of the various phenomena are still not well understood, while the majority of the knowledge pertaining to sediment dynamics is derived from laboratory experiments conducted under limited number of controlling parameters.

C. DATA NEEDS

1.13 Laboratory Data.

In order to properly describe sediment properties and dynamics, a number of laboratory and field data should be collected. The laboratory data that are usually required for a complete characterization of sediments (particularly cohesive) are the following: dry density, particle size grading, mineralogy, organic matter content, settling velocity, yield value, viscosity, consolidation, critical shear stress for erosion, and slope of repose of emerged and submerged deposits (Hamm and Migniot, 1990).

1.13.1 Laboratory Methods. Density, mineralogy, and organic content can be identified with standard laboratory methods. Particle size grading for particles greater than 1/16 mm can be accomplished by sieving or visual accumulation tubes (VA tubes). For particle sizes less than 1/16 mm, other direct or indirect methods can be used such as Coulter counter, pipet, microscope, etc. The particle diameter of untreated cohesive sediment samples is representative of the floc size. In order to estimate the size of deflocculated particles, mud samples should be treated by ultrasonic waves (i.e., break interparticle bonds) and the organic matter must be removed by oxidation.

The settling velocity, w_s, should be estimated as a function of the suspended sediment concentration, C, and water salinity, s. The yield stress value, τ_y, which indicates the rheologic behavior of the bottom should be measured by a viscosimeter (e.g. Brookfield LVT) and expressed as a function of the dry sediment density.

Consolidation can be studied in settling columns (20 cm to 4 m high). The density can be monitored by various techniques, e.g., Gamma ray densimeter. The rate of consolidation is a function of the dry sediment density, initial concentration, and initial height. Estimation of the critical shear stress for erosion requires flume testing.

1.14 Field Data.

Besides the information provided by the laboratory analysis of water-sediment samples, a lot of information is also required from direct field (in-situ) measurements. Field data requirements involve: hydrodynamics, water properties, suspended sediment concentration profiles, and bottom characteristics.

Hydrodynamic data should include, water elevations, current profiles, wave climate, and dye or radioactive tracer experiments. In addition, tributary inflows, pattern of sheet overland runoff, ET and groundwater exchange must be identified. The data can be either discrete or continuous. However, in order to obtain a comprehensive picture of the dynamic behavior of the riverine system, the data should include both normal and extreme (episodic event) conditions, and also be synoptic.

Water properties should include, salinity, temperature and pH as minimum requirements. Suspended sediment profiles should identify the extent of the layers of mobile and stationary suspensions. This can be accomplished by sampling the water column and analyzing the sediment content by weight.

The ASCE Task Committee on Analysis of Laboratory and Field Sediment Data Accuracy and Availability presented a number of papers pertaining to data collection, needs and analyses of riverine sediments (Glysson, 1989) or hyperconcentrated debris-mud flows (Julien, 1989; Bradley, 1989). However, in their reports they did not include any guidelines for estuarine sediments.

1.14.1 **Field Methods for Data Collection.** A thorough description of methods for field sampling of bed load and suspended material is provided in the USGS manual (Guy and Norman, 1982). The manual describes both the equal-discharge-increments (EDI) and the equal-transit-rate (ETR) methods. In the EDI method samples are collected at the centroids of equal discharge cross-section increments, while in the ETR method samples are being taken at each of equally spaced cross-section verticals.

In-situ observations of suspended particles (visualization), through transparent settling tubes, are also very useful for assessing the state of aggregation-flocculation. Bottom characteristics can be identified in the field by echo sounding (sonar), rheological probes, ultrasonic densimetric probe, gamma densimetric and other devices. Field information should also identify areas of sedimentation problems, i.e., excessive shoaling or scouring.

1.15 Cost Estimates for Establishing Sediment Mass Balances.

Based on the discussions presented in sections 1.11 to 1.14, a preliminary cost analysis for development of a comprehensive sediment mass balance assessment in the LSJRB is estimated as follows. The cost for the study is divided into four main categories: field data collection, laboratory analyses, data interpretation and mathematical modeling (Table 1.10). The cost estimates in Table 1.10 are based on a projected number of sixteen sampling stations (twelve in the tributaries and four in the main river). The monetary figures are rough estimates which can vary between different engineering/

Table 1.10 Cost estimates for development of sediment budget for the LSJRB.

Activity		Approximate Unit Cost	Number of Units	Total Cost Per Activity
Field Data Collection:	Per station:	$ 500	16	$ 8,000
Laboratory Analysis:	Per sample:	$ 150	48	$ 7,200
Data Interpretation:	Per station:	$ 500	16	$ 8,000
Mathematical Modeling:	Hydrodynamics	$ 50,000	1	$ 50,000 (*)
	Sediments	$ 50,000	1	$ 50,000 (*)
				$123,200

NOTE: (*) : One-Time Cost Item

scientific companies and laboratories. More specifically, for the given budget the following services will be provided: The field data collection efforts will include waterways bathymetry, flow measurements and sediment sampling at pre-designated cross-sections. The laboratory analysis will be focused on the physicochemical properties of the aquatic sediments (both bottom samples and suspended material). Analysis of the sediment-bound contaminants is not included in this budget. The interpretation of the collected field data will provide a preliminary assessment of the sediment mass balance at each station, while the mathematical models will simulate the water/sediment dynamics throughout the entire LSJRB.

D. PRELIMINARY ANALYSIS AND RECOMMENDATIONS

1.16 Preliminary Analysis.

Based on the information and data from the previous sections, a preliminary analysis of the prevailing sedimentary conditions of the Lower St. Johns River Basin can be done. The bottom slope of the main riverine channel is very small. Thus, it is very unlikely to have any bed load discharge under normal flow conditions. Assuming an average bottom slope of $S = 1.89 \times 10^{-5}$, an average depth of $h = 3$ m and a specific gravity of sediments $\gamma_s = 2.65$, the critical particle diameter for erosion can be estimated from Eq. 1.6 as $D = X_2[(\rho_s - \rho)/\rho]/(\rho h S)$. By trial and error, the critical particle diameter, D_c (i.e., all particles with $D < D_c$ will erode), can be defined by using the Shields curve. This particle diameter under the aforementioned physical conditions was found to be approximately 0.6 mm. This conclusion does not apply to bank erosion and is not valid for cohesive beds. Bank erosion is possible to occur due to various causes such as wave action from boat wakes, animal burrowing on the banks, human related activities, etc.

Erosion rates within the tributaries may differ from the ones in the main channel, since the gradient of the tributaries is higher. For example a rough estimate of the slope of the Etonia Creek gives approximately So $\approx$ 1.5 x 10^{-3}, which is two orders of magnitude higher than the slope of the main channel.

A qualitative assessment of the riverine response can be established by using the relationship (Simons and Senturk, 1976),

$$QS_o \propto Q_s D_{50} \quad \text{(1.58)}$$

where Q is the water discharge and Q_s is the transport sediment discharge. Writing the same relation as follows:

$$Q_s \propto QS_o/D_{50} \quad \text{(1.58a)}$$

it can be easily concluded that an increase in water discharge or bed slope will result in an increase of transported material, while an increase of the particle diameter will have the opposite effect.

For mud beds, typical values of critical shear stress for erosion range from 0.2 to 5.0 N/m^2. Assuming a velocity of 0.14 m/s (0.44 fps), and a bottom resistance of C_z = 45 m$^{1/2}$/s, the bottom shear stress is estimated as $\tau \approx 0.1$ N/m^2 (less than the critical).

Measurable erosion is anticipated to occur after intensive precipitation events and during natural habitat disturbances, urbanization, highway construction, deforestation, etc. However, the effects of the latter will cease after the disturbance is stopped. Also, wind generated currents much have predominant effect on the rates of erosion.

Another situation that may effect sediment transport rates is the ephemeral streams subjected to cycles of wetting and drying. These streams can produce relatively high volume of transported material during the first hours of a rainfall event.

Sedimentation conditions near the ocean inlet should be treated differently than the rest of the river. Near the inlet sediment composition is primarily sandy since strong currents wash out all of the fine material.

1.17 Recommendations.

A comprehensive understanding of the dynamic processes and sedimentation budget of the Lower St. Johns River Basin requires detailed information of the prevailing hydrologic and hydrodynamic forces. Assuming that the hydrologic and hydrodynamic information is available, sedimentary field data and laboratory analysis should be conducted. The action plan should include the following items:

o Identification of the most problematic areas based on reported shoaling, scouring, bank failures, etc.

o Monitoring of bed changes in the problematic areas by bathymetry mapping, diver observations, sonar surveying, etc.

o Placement of bottom sediment traps within the main river and in the major tributaries to identify any bed load discharge.

o Establishment of a systematic suspended sediment data collection program. This will include weekly or monthly sampling of the water column for temperature, salinity, pH, and total suspended material. The stations should be located upstream from the mouth of each of the twelve main tributaries. The same data should also be collected at four locations distributed evenly along the length of main river. For identification of diurnal sedimentation patterns, some of the data should be collected on a continuous time basis.

- o Establishment of a special data collection program targeted to episodic events, as well as to areas with new development activities.
- o Analyses of the sediment samples and identification of the physical properties of the sediments and their vertical profiles.
- o Estimation of transported sediment volume from each tributary based on water discharge and suspended sediment data.
- o Development of a sediment budget model based on a simple mass balance relation, i.e., inflow - outflow = accumulation.
- o Development of an advection-dispersion type of mathematical model for simulation of sediment dynamics. Calibration and verification of the model would be based on the collected data.

CHAPTER II

SEDIMENT REMEDIATION AND ENVIRONMENTAL IMPACT ASSESSMENT

E. REMEDIATION TECHNIQUES FOR POLLUTED ESTUARINE SEDIMENTS

2.1 Sediments as Pollution Sources and Sinks.

Solutes in water are very rarely in a perfectly dissolved state (Medine and McCutcheon, 1989). As a result, portions of the solutes attach to solid surfaces. Organic, hydrophobic-persistent chemicals, metals, nutrients, and radionucleides have a particularly strong affinity for solid surfaces (Hart, 1982). Fine grain sediments tend to adsorb pollutants on their surfaces. These sediments deposit in quiescent sections along rivers, waterways, lakes and estuaries, or they are ultimately transported and settle in the oceans (Partheniades, 1973). Thus, depending on the dynamics of the ecosystem, pollutants in aquatic environments may be temporarily or permanently stored within bottom sediments. Natural forces (e.g., tides, wind waves, etc.) and human-induced activities (e.g., boating, dredging, etc.) may disturb the surficial sediment deposits and cause entrainment and release of the accumulated pollutants. Therefore, even if the source of polluting elements and chemical compounds has been eliminated, recovery of the ecosystem in affected areas may occur very slowly or not at all unless the contaminated sediments are removed (Medine and McCutcheon, 1989).

2.1.1 Exchange Processes of Particulate-Bound Pollutants. During sediment transport, particle-bound pollutants are effected by a variety of physical, chemical, and biological processes including:

- advective motion (due to tides, currents, etc.),
- diffusion and dispersion (spreading due to Brownian motion, turbulence, and the nonuniformity of the velocity profile),
- bioturbation (i.e., stirring of bottom sediment by benthic organisms),
- adsorption/desorption (i.e., solute attachment/detachment on particulate surface by electrochemical forces),
- hydrolysis (i.e., chemical decomposition caused by water: $AB + H_2O = AOH + HB$),
- oxidation (i.e., increase of the positive electro-charge due to augmentation of the valence number of an atom or ion as a result of the loss of one or more electrons),
- reduction (i.e., the antonym of oxidation due to addition of hydrogen, removal of oxygen, etc.),
- ionization (i.e., removal of one or more outer electrons from an atom or molecule),
- complexation (i.e., reaction of two species to form a third species),
- dissolution and precipitation (i.e., formation of precipitates),
- volatization (i.e., a physicochemical process where a solid or liquid mass changes into gas),
- photolysis (i.e., chemical reaction or decomposition promoted by light), and
- biological degradation (i.e., decomposition by microbial action).

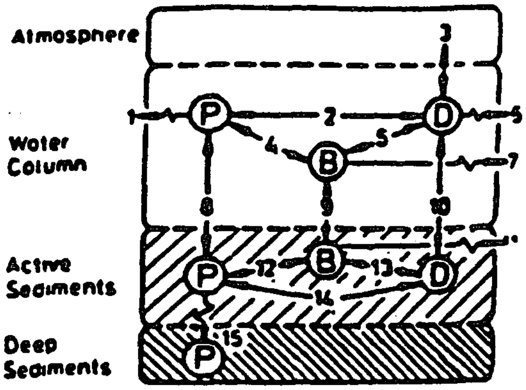
Forstner (1987) summarized the main processes affecting the exchange of pollutants

Table 2.1 Processes involved in pollutant exchange in aquatic ecosystems (Forstner, 1987).

Process	Aqueous species	Particulate phases
Physical:	Advection	Resuspension (8)*
Bioturbation	Diffusion	Settling (8)*
	Photolysis (1,6)*	Burial (15)*
Chemical	Dissolution	
	Desorption	
	Complexation	
Biological:	Decomposition (7,11)*	Food web transfer (9)*
Species	Adsorption/Release	Filtering/Digestion (5)
Transformation	Cell wall exchange	Pellet generation (4)*

* The numbers in parenthesis refer to processes indicated in Figure 2.1.

between water and sediments (Table 2.1). The transfer and interactions within the air-water-sediment system of three toxic chemical phases, i.e., particulate, dissolved, and abbreviated food web, are given schematically in Figure 2.1 (Eadie et al. 1983).



P = particulate phase; D = dissolved phase; D = abbreviated food web

PROCESS: 1 & 6 = Photolysis; 2 & 14 = Adsorption/Desorption, Complexation; 3 = Air-Water Exchange; 4 = Grazing and Fecal Pellet Generation; 5 = Filtering; 7 & 11 = Biological Decomposition; 8 = Setting and Resuspension; 9 = Food-web Dynamics; 10 = Advective and Diffuse Mixing; 12 & 13 = Benthos-Sediment Interaction; 15 = Burial and Bioturbation.

Figure 2.1 Compartments and interactions of toxic chemicals (Eadie et al. 1983)

This figure complements Table 2.1 by providing not only the processes but also the interactions between the pollutant phases.

2.1.2 Release or Burial of Pollutants in Oxygenated and Anoxic Sediments. The study of biochemical processes in sediments can be simplified by assuming that the sediment-water system is divided into three horizontal layers: the oxygenated (oxic) layer, the anoxic layer, and the interface between the oxic and anoxic layers. In estuaries, the oxic-anoxic interface usually lies within the bottom sediments, while in deep, stagnant water bodies, the oxic-anoxic interface may occur within the water column. The oxygenated layer is often rich in organic matter which promotes the growth of large populations of bacteria. These bacteria in turn, degrade suspended particulates thereby remobilizing adsorbed chemicals. If the oxic-anoxic interface is within the water column this remobilization results in a net flux of pollutants from the water column towards the bottom sediment. The opposite phenomenon occurs when the oxic-anoxic interface is located within the sediments; i.e., pollutants are released from the sediments into the water column (Salomons et al. 1987). Generally, the oxic-anoxic interface regulates the exchange of trace metals between sediments and the water column. When compared to bioturbation and consolidation fluxes, molecular diffusion flux of pollutants across the oxic-anoxic interface can be considered negligible since pollutant concentration gradients are relatively small.

Within the oxic layer, trace metal mobility is strongly affected by pH changes (e.g., as can be caused by algal blooms) and/or metal uptake by organisms. In the anoxic layer, trace metal behavior is mainly controlled by the redox (reduction/oxidation) potential. The main redox processes in the anoxic layer are methanogenesis and reduction of sulphate, iron and manganese hydroxides (Eckenfelder, 1989). Methane is produced by the action of anaerobic methane-forming bacteria which convert organic acids into methane, CH_4, carbon dioxide, CO_2, and other stable compounds. Depending on the various chemical compounds present in bottom sediments, redox reactions follow a certain sequence (Stumm and Baccini, 1978) (Figure 2.2).

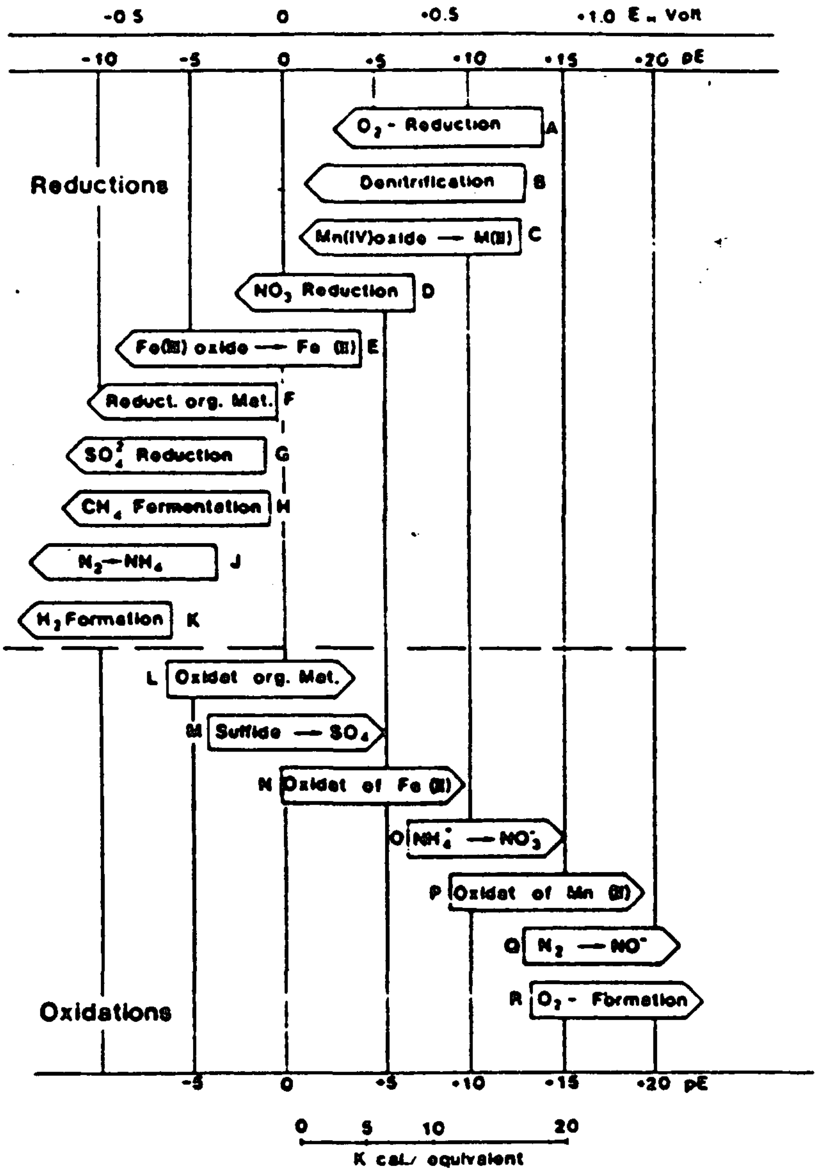


Figure 2.2 Sequence of redox reactions (Stumm and Baccini, 1978).

In estuarine waters, diffused sulphate reacts with iron to form iron sulphides while organic matter participates are degraded (Salomons et al. 1987). Therefore, in the surficial sediments the iron hydroxides are converted to iron sulphides and the degradable organic compounds are depleted while sulphate is still available within the pore water. As a result, the methanogenic state is attained only in relatively deep bottom layers (e.g., of the order of 50 cm or more) due to the sequence of redox reactions (Figure 2.2).

In fresh waters, the methanogenic state is reached much closer to the surficial sediment layers. This is due to the fact that sulphide concentrations are low, and all of the reducible iron is reduced to iron carbonate. By considering the availability of sulphate and reducible iron, Salomons et al. (1987) evaluated the processes controlling pH conditions within anoxic sediment layers and provided estimates of the pH for zero ionic strength at 25°C. According to their results, the pH values will be approximately: 7.25 if both sulphate and reducible iron are present; 6.75 if sulphate is present and no reducible iron is available; 7.05-7.1 if sulphate is exhausted and only reducible iron is available; and 6.2-6.5 if both sulphate and reducible iron are absent.

Complex organics such as fats, carbohydrates, and proteins are converted biologically into simpler compounds, mostly fatty acids. This conversion is accomplished by anaerobic bacteria (acid-formers) which are tolerant to pH and temperature changes, and grow much fatter than the methane-formers that carry out the second stage of decomposition (Masters, 1991).

Hydrocarbons are subject primarily to microbial degradation. Microbes, slowly but preferentially, proceed into breaking the hydrocarbons to n-alkanes, then to branched alkanes and finally to cyclic alkanes (Wakeman and Farrington, 1980). In general, persistent organic compounds (i.e., PAHs, PCBs and Cl-pesticides) adsorbed by bottom sediments can be mobilized through the following biomechanisms:

- o physical mixing with surficial sediments by benthic organisms, so that there is a greater chance for resuspension, solubilization and oxygenated conditions that favor microbial action,
- o water pumping through sediments by benthic organisms, thereby accelerating solubilization and removal of soluble compounds and creating oxic conditions underneath the surficial sediment layer, and
- o ingestion and metabolism of the organic compounds by benthos and fish.

2.1.3 <u>Physicochemical Phases of the Water-Sediment System.</u> Elements and molecules may partition into any one of the following phases of the sediment- water system (Engler, 1980):

- o interstitial water/sediment phase,
- o mineral exchange phase,
- o reducible phase,
- o organic phase, and
- o residual phase.

Interstitial water refers to the pore water of the sediment phase, which is in equilibrium with the silicate and organics. The mineral exchange phase is the part of an element that can be removed from the molecules by application of any ion exchange extractor (e.g., HCl). The reducible sediment phase is the phase comprised of iron and manganese hydroxides. The organic phase consists of compounds which become soluble after decomposition of the organic matter. Finally, the residual phase is comprised of weathered primary and secondary minerals and is very stable. The residual phase is the one that normally has the highest concentrations of metals. Pollutants may be bound in the listed sediment phases as dissolved materials, adsorbed particles, surface coatings, and/or by chemical association.

2.2 Sediment Remediation Assessment.

Although once a popular belief, the theory that sediments act as irreversible sinks for precipitating pollutants has been completely discredited (Knezovich et al. 1987; Salomons, 1985). It is now known that sediments may not only take up pollutants, but they may also release pollutants back to the water column under certain conditions as it was discussed earlier. Therefore, if untreated, contaminated sediments can cause environmental problems.

Both prevention of sediment-related pollution and remediation of contaminated sediments require an initial evaluation and a preliminary study plan. Such a preliminary plan involves three phases (Forstner, 1987):

o the reconnaissance phase,

o the investigation phase pertaining to the physicochemical and biological processes, and

o the bioavailability assessment phase.

2.2.1 Reconnaissance Phase. The reconnaissance phase involves a preliminary evaluation of the problem, the data requirements, and the available technical approaches. This phase is important for planning and implementing the investigation phase. In the reconnaissance phase, the tasks that should be included are:

o developing a detailed definition of data collection objectives (Why do we need the data?

How are we going to use it?),

o planning and implementation of sample collection program (Where, when, and what data should we collect?), and

o selection of equipment and analytical methods (How do we collect, store and analyze the data?).

Therefore, the reconnaissance phase provides compilation and pre-analysis of the existing data, and suggests action plans for further studies and analyses.

2.2.1.1 Quality Control/Quality Assurance for Contaminated Sediments. Since conclusions reached and decisions made during the reconnaissance phase are essential for designing an efficient and effective investigation plan, a quality control/quality assurance, QC/QA, program must be employed. For sediment quality studies, the QC/QA program must be focused mainly on sampling, preparing and analyzing the sediment samples. Generally, determination of the contaminant concentration in sediment samples requires two steps (DePinto et al. 1984):

o contaminant extraction, and

o chemical analysis.

Sample preparation during extraction and analysis should be performed with great care. For example, anoxic samples can be oxidized by exposure to aerobic conditions. Also, changes in the speciation of labile phases of anoxic samples may possibly occur during freezing and/or drying of the samples. Laboratory analysis on split and spiked sediment samples should follow the analytical procedures developed and recommended by the U.S. Environmental Protection Agency and the U.S. Army Corps of Engineers (U.S. E.P.A. and U.S. Army Corps of Engineers, 1991), and the American Society of

Testing and Materials (1990).

2.2.2 **Investigation Phase.** The purpose of the investigation phase is to model the extent of potential sediment-related hazards to biota based on hydrologic, hydrodynamic, geomorphologic, geologic, chemical, biological and socioeconomic data. The investigation phase includes assimilation and quantitative analysis and modeling of the information obtained from the data pre-analysis. This phase requires application of statistical methods (e.g., regression, trend analysis, etc.), and the development of deterministic or stochastic models for description of the physicochemical processes involved. These models should be able to simulate the spatial and temporal dynamics of the water and sediments as well as the chemical speciation, complexation, partitioning, and precipitation of the different chemicals involved. Depending on the spatial extent of the sediment contamination and the characteristic times of the various phenomena, different models may be required, such as near-field, meso-field, or far-field, and short-term or long-term models. For example, the flow field near the outlet of a point source can be effectively described by means of buoyant jet dynamics, while description of the field away from the source requires application of the full set of hydrodynamic equations.

Pertaining to the St. Johns River, the investigation phase can be either for the entire estuary or for one or more of its tributaries separately. Presently however, the former approach is more suitable for the hydrology/hydrodynamics of the system, while the latter is for the sediment quality/quality studies. This is due to the fact that the existing sediment quality/quantity data are very limited and they cannot support a broad, quantitative sediment transport and sediment pollution study.

2.2.3 Bioavailability Assessment Phase. The bioavailability assessment phase is the final part of a preliminary ecological evaluation. Bioavailability assessment should focus on the estimation of the bioavailable forms of the various contaminants (Allan, 1984). Indeed, the absolute amount of contaminants in an ecosystem is of minor importance compared to the amounts of chemicals that are readily available to the biota (Reynoldson, 1987). Bioavailability can be assessed by three methods:

o chemical extraction,
o multi-organism benthic bioassay, and
o combination of chemical extraction and toxicity test.

2.2.3.1 Chemical Extraction. Chemical extraction from sediments can provide an estimation of the pollutant amount that can be mobilized by changes of the pH, salinity, or redox conditions and/or by organic complexing processes. Chemical extraction should be conducted preferably by chemical leaching rather than by heat treatment since the former provides information on reaction kinetics and digenetic processes such as diffusion and aging (Forstner et al. 1984). A relatively good indication of the mobility and potential bioavailability of metal-contaminated sediments can be achieved by using a weak acid-reducing extractor. However, the leachate may not give an accurate estimate of the chemical fraction that will be available to the biota. One of the most common extraction procedures for estimation of short-term chemical release is the elutriation test (i.e., release of adsorbed pollutants under the action of fluid media). Laboratory testing has indicated that the oxygen content and the solid:liquid ratio are the most important factors influencing elutriation test results. Experimentally a 1:4 of solid to liquid ratio gives reliable results for marine, estuarine, and freshwater systems.

Elutriation test results are excellent predictors of pollutant releases from the sediments into the water column.

2.2.3.2 Toxicity Tests. Pollutant uptake by aquatic biota under natural conditions is a very complex process which is strongly affected by the natural hydrologic and biological cycles of the ecosystem. Field or laboratory bioassays should include at least three different types of organisms; i.e., phytoplankton or zooplankton species, shellfish (e.g., crustaceans or mollusks), and fish (Hebert and Schwartz, 1983). Bacteria and protozoa were found to be unsuitable for routine bioassays because of the difficulty in interpretation of the results relative to higher organisms. On the other hand, algae and zooplankton can be effectively used for simulation or toxicity tests (Shuba et al. 1977). Toxicity tests should be conducted for both the liquid and the solid phases (sediments). Special attention should be given to sediment tests because sediment-bound contaminants facilitate bioaccumulation and create sublethal toxicity. Bioaccumulation from historically contaminated sites should be quantified from biota taken directly from the contaminated site rather than from control animals exposed to short-term laboratory conditions (10 days) (Metcalf, 1977). Bioassays should also be conducted on plants (Lee et al. 1982; Munawar and Munawar, 1987). In general, prediction of the effects of sediment geochemistry upon pollutant bioavailability is a very difficult and complex process (Luoma, 1989). Besides the conventional toxicity tests that require relatively large quantities of a sample, a limited sample bioassay (LSB) method has been developed and successfully used in the assessment of bioavailability of sediment-bound contaminants (Munawar et al. 1989). More reliable information on the anticipated ecological impact can be obtained by combining both chemical extraction with water and toxicity tests.

2.3 Techniques for Sediment Remediation.

Once the extent and toxicity of the *in-situ* sediment contaminants has been established, the appropriate remedial action should be selected. Development of an appropriate remedial plan depends on the collection of the following information:

o site of contamination,

o source of contamination,

o type and magnitude of contamination,

o ecological, recreational and/or commercial value of the contaminated area,

o cost of the remedial action,

o estimated success of the proposed clean-up,

o anticipated benefits, and

o existence, activity, and possible control of adjacent pollution sources.

There are two basic remedial options for treatment of sediment-related pollution. The first option is to remove the sediment from the contaminated site, and the second option is to contain and/or treat the sediment in place.

2.3.1 Removal of Contaminated Sediments. Sediment removal involves dredging and subsequent disposal of the spoil. Dredging is an expensive operation and is used primarily for maintenance of navigable waterways. Depending on the extent of the dredge/disposal operation the dredging cost ranges from $ 2 to $ 4 per cubic yard (U.S. Army Corp of Engineers, 1992). Besides its high operational cost, dredging may have an adverse near-field environmental impact. During dredging operations, the solubility, mobility, and bioavailability of sediment-adsorbed toxic metals are increased due to mechanical mixing, lowering of the pH, changing of the redox potential, introduction of organic compounds, and/or possible increase in salinity levels (Forstner, 1982, 1987).

2.3.1.1 Environmental Problems Related to Dredging. The pH and redox potential of the dredged material change as it mixes with the oxygenated water column. Change of the redox potential may greatly affect the mobility of toxic metals during dredging (Weber et al. 1982). Reduced sediments deposited at inland disposal sites, will become oxidized (Forstner, 1987). This change has an important effect on increasing the mobility of some metals (i.e., cadmium, Cd; cobalt, Co; copper, Cu; mercury, Hg; nickel, Ni; lead, Pb; and zinc, Zn) while decreasing the mobility of others (i.e., iron, Fe; and manganese, Mn). Oxidation has very little effect on the mobility of molybdenum, Mo; vanadium, V; uranium, U; and selenium, Se. The mobility of toxic metals in inland disposed spoil is also affected by the lowering of the pH (Gambrell et al. 1977). One possible reason for the lowering of the pH in inland disposal sites is oxidation of sulphide minerals. Increased acidity levels in surface waters (e.g., from acid precipitation) also contribute to increased mobility and toxicity of metals (Fagerstrom and Jernelov, 1972). However, naturally occurring organic acids, e.g., humic, fulvic, can bind certain metals and make them unavailable to biological systems. A probable order of binding strength of metal ions onto humic or fulvic acids is: $Hg^{2+} > Cu^{2+} > Pb^{2+} > Zn^{2+} > Ni^{+} > Co^{2+}$ (Jonasson, 1977). Salinity effects are particularly important for cadmium-contaminated, estuarine sediments (Forstner et al. 1986); chlorides can form complexes with available trace metals and thus inhibit uptake or precipitation by the suspended particulates.

The mixing of sediments during dredging may promote the growth of certain bacteria and cause biological transformation of certain compounds; e.g., biodegradation of PCBs, speciation of toxic metals by methylation (e.g., mercury), etc. It can also promote contaminant bioavailability and subsequent bioaccumulation by increasing concentrations of suspended, sediment-bound chemicals. Bioaccumulation occurs either within the animal gills or on tissue surfaces by direct exchange of chemicals or within the digestive system by uptake of the sediment-bound chemicals (Allan, 1984).

Adverse effects created by dredging are primarily limited to the vicinity of the dredging site, and usually do not have a long-term impact on the aquatic biota. However, additional problems may be introduced during the disposal of the dredged spoil (see Section 2.4). For detailed information on dredging operations and associated physicochemical and biological effects, extensive analyses and reviews can be found in the literature (Cable, 1969; Boyd et al. 1972; Windom, 1972; May, 1974; Lee and Plumb, 1974; Keeley and Egler, 1974; Sweeney et al. 1975; Brandema and Civoky, 1976; Sly, 1977; Engler, 1980; Hebert and Schwartz, 1983; Committee on Sediment Control, 1987; Herbich, 1992).

2.3.2 In-Situ Treatment of Contaminated Sediments.

Whenever dredging is not a feasible solution, contaminated sediments are either subjected to some *in-situ* inactivating method or left in place undisturbed. *In-situ* sediment inactivation can be achieved through application of one of the following four approaches (Rulkens et al. 1983; Thomas, 1987):

o mechanical encapsulation,
o chemical treatment,
o diminution of contaminant concentration, and
o bioremediation.

In some cases, contaminated sites have shown some improvement compared to the original contaminated condition even without treatment (Durham and Oliver, 1983). This *in-situ* "treatment" option (i.e., leaving the sediments undisturbed) is recommended whenever the contaminated area is very extensive and none of the other approaches is technically feasible or economically practical.

2.3.2.1 Mechanical Encapsulation. Mechanical encapsulation includes application of impervious screening and construction of containing barriers. Impervious screening consists of covering the contaminated sediment by some membrane of extremely low permeability (e.g., polymer films) or using a thick layer of fine-grain sediment, or applying waste "wool" blankets with sand and gravel overlays. The impervious screen will be covered eventually by natural deposits of uncontaminated sediment. The main disadvantage of this method is that the impervious screen can be disrupted by gas release from the organic-rich, covered sediments (e.g., methane, hydrogen sulfide, etc.) and/or by bioturbation activities. The practice of covering the contaminated sediments with materials of high permeability (e.g., gravel) is not recommended because, although these materials may reduce resuspension of the contaminated sediments, they will not prevent diffusive contaminant exchange between the sediments and the ambient water. Similar containment measures (e.g., geomembranes, clay layers, etc.) are also applied to land disposal sites for prevention of groundwater contamination and/or overland transport of the pollutants.

2.3.2.2 Chemical Treatment. Chemical treatment of contaminated sediments is more appropriate for small, isolated, aquatic areas (e.g., small lakes). Application of chemical remediation to areas subject to strong advective motion may not be very effective due to the short residence time of the remedial chemicals. Chemical treatment includes (Forstner, 1987):

o chemical immobilization (i.e., formation of insoluble compounds),

o fixation of water by adding chemicals, or

o reduction of certain chemical compounds by changing pH and redox conditions (Forstner, 1987).

As an example, sediment treatment with sodium carbonate has been suggested as a chemical remedial measure for acidified lakes (Lindmark, 1982). For mercury contamination, remedial steps include (Smith, 1972):

o anaerobic conversion of mercury to mercury sulphite,

o aerobic adsorption of mercury by clays and precipitation with hydrous iron and magnesium oxides,

o formation of volatile dimethyl mercury by increasing the pH followed by mechanical encapsulation.

Chemical immobilization techniques for persistent organic compounds are not yet available. Therefore, the methods which are presently being used for remediation of priority organic pollutants are limited to mechanical encapsulation, solidification, and/or biodegradation.

2.3.2.3 **Diminution-of-Concentration.** The main principle behind the diminution-of- concentration approach is to reduce the contaminant concentration using sediment dilution and mixing. The two main methods of diminution of concentration are the accelerated deposition and ploughing.

2.3.2.3.1 **Accelerated Deposition:** The accelerated deposition of contaminant-free sediment increases the rate of grain settling of the unpolluted sediments, so that the contaminated bottom sediments are covered and mixed with the new sediments. The higher the rate of deposition, the more rapidly the polluted layers are diluted. The method of accelerated deposition should be implemented very carefully to ensure that it does not disrupt the prevailing hydrodynamic patterns and sedimentological features of the ecosystem. Application of large amounts of sediment in a short period of time is not recommended. Whenever possible, sediment should be introduced into the system through its normal sources; i.e., rivers, tributaries, etc., and during the appropriate season of the year. A well-planned release of additional contaminant-free sediment loads is important for preserving the natural sediment yield and preventing any adverse effects on the biological cycle. Special attention should be given to minimizing, as much as possible, the turbidity resulting from dredging operations, especially during the seasons of high biological productivity (e.g., reproduction and photosynthesis). In certain cases, in order to enhance the rate of sediment deposition, a sediment slurry can be directly applied to the contaminated sites. Grain size and distribution of the applied sediment loads should match those of the *in-situ* sediments so that sedimentology patterns remain unchanged.

2.3.2.3.2 Ploughing: The method of ploughing is based on the fact that contaminated sediments in estuaries are usually restricted to the movable top 30 cm. Sediments below this contaminated layer are relatively unpolluted since they do not participate in the sediment transport cycle. Ploughing mixes the contaminated with the uncontaminated sediments and thus reduces the pollutant concentration. Theoretically, for a contaminated sediment layer of 30 cm, ploughing to a depth of 60 cm will produce a reduction in concentration by 50%. During ploughing, care should be taken to avoid excessive resuspension. However, ploughing destroys the surficial soil matrix, and bottom sediments will erode more easily. For effective and efficient ploughing, selection of the plough equipment should incorporate factors such as water depth, area extent, hydrodynamic field, sediment type, and benthic communities. These factors will define the design of the ploughing equipment (e.g., size, shape, and spacing of blades; weight; etc.), and the power requirement of the tow boat. For cost effectiveness, the sites proposed for remediation by ploughing should be precisely mapped and identified. The major cost for ploughing is related to boat chartering and is approximately $ 2 per cubic yard.

2.3.2.4 <u>Bioremediation.</u> Degradation of hazardous substances can also be achieved by means of biological treatment; i.e., by using either heterotrophic microbes, which require an external source of organic matter, or autotrophic microbes, which are capable of synthesizing the necessary organic compounds from inorganic sources (Tebbutt, 1971). Bacteria reproduce by fission; therefore, if there is an adequate supply of nutrients, the reproduction process will proceed rapidly and organic compounds will break down into less toxic or even non-toxic chemicals. However, bioremediation is very system-specific. Effective contaminant degradation requires development and growth of the proper microbial community. Although microbes are resistant to moderate amounts of heavy metals, their remedial activity can be substantially reduced by a high concentration of metals. Microbial bioremediation may be strongly inhibited if salt concentration is greater than 2%. For biodegradable organics, the microbial population can be maintained at appropriate levels so that the toxins are reduced below toxicity levels. For hard-to-degrade toxins, microbial strains that are resistant to these toxins must be enriched. Standardized test procedures for complete degradation of readily degradable compounds to carbon dioxide and water are given by Gerike and Fischer (1981). For water-insoluble substances, there is no one, standard method for determining the source of the inoculum, the composition of the growth medium, and the concentration of the test substances (Nielson, 1989).

In spite of the advanced state-of-the-art laboratory biotransformations, techniques for bioremediation of natural aquatic environments are not, as yet, readily available (Nielson, 1989). Recent research efforts toward this goal are focused on the following topics:

- identification and occurrence of substrates and their metabolites,
- quantification of the extent to which the initial xenobiotic - or its metabolites - are bound to soil particulates, and
- establishment of similarities between pathways observed in the laboratory and those occurring in nature.

All of the aforementioned methodologies (i.e., mechanical, chemical and biological) for remediation of contaminated aquatic sediments have been applied with varying degrees of success throughout the world. The cost associated with each methodology depends on the availability of the appropriate equipment, accessibility of the contaminated site, and the extent and type of contamination. In general, however, the most costly remedial operation is remediation by dredging and subsequent spoil disposal.

2.4 Disposal of Dredged Material.

If dredging is adopted as the remedial option, a disposal site needs to be selected. There are three options for dredge material disposal:

o ocean dumping,

o dumping into U.S. inland waters, and

o inland disposal.

2.4.1 Ocean Dumping. Ocean dumping must comply with the rules and regulations established by the EPA in consultation and conjunction with the U.S. Army Corps of Engineers (Marine Protection, Research and Sanctuaries Act - MPRSA; Federal Water Pollution Control Act - FWPCA). Generally, there is a ban on ocean dumping of certain contaminants that can cause significant environmental degradation (e.g., highly radioactive waste, chemical and biological warfare waste). It is, however, highly unlikely for estuarine-dredged materials to contain any of these contaminants. Before dumping into the ocean, thorough physical, chemical and biological investigations of the ecosystem are required.

2.4.2 Disposal Into U.S. Waters. Disposal into U.S. waters should be based on the same guidelines as those used for disposal into the ocean. However, due to their comparatively small volume, coastal or inland waters have much less assimilative capacity than the ocean. More specifically, for disposal of spoil materials into either the oceans or U.S. waters, the following studies should be conducted:

o dynamics of water/sediment mixing processes,

o quality of ambient water phase,

o amount and quality of suspended particulate phase, o quality and composition of bottom sediments,

o presence of trace metals and organic priority pollutants,

o presence of prohibited materials,

o bioavailability and bioaccumulation of pollutants, and

o ecological suitability of the disposal site.

Section 404 of the Federal Water Pollution Act (1977) is directed toward preventing environmental degradation of U.S. waters attributable to the disposal of polluted dredged or fill material. The Act provides guidelines for various pollutants such as: OPPs or toxic substances under Section 307 of the Act; hazardous substances under Section 307 of the Act; and PCBs under the Toxic Substances Control Act. The U.S. Army Corps of Engineers Dredge Material Research Program (DMRP) conducted an extensive field investigation at sixteen dredge disposal sites located along the east coast of the United States. The sites were monitored for short-term and long-term releases of eight trace metals under quiescent and stirred conditions (Brannon et al. 1980). The results indicated that no single short-term test is capable of predicting the long-term impacts of the dredged disposal.

The State of Florida has adopted programs for protection of surface-waters while permitting compatible human uses. These programs are described in the Florida Administrative Code (FAC), Chapter 17-3, and include: classified surface waters, outstanding Florida waters (OFW), and aquatic preserves.

The surface waters of the State of Florida have been classified, based on their designated uses, as follows:

Class I - Potable water supplies,

Class II - Shellfish propagation or harvesting,

Class III - Recreation; propagation, and maintenance of a healthy, well-balanced population,

Class IV - Agricultural water supplies,

Class V - Navigation, utility and industrial use.

All surface waters of the Lower St. Johns River Basin are presently Class III waters except for the shellfish harvesting area at the mouth of the St. Johns estuary (Duval County), which are classified as Class II. Surface water quality standards for trace metals and organic pollutants as established by the FAC (Chapter 17-3) are given in Table 2.2.

On July 13, 1978, the Environmental Regulation Commission adopted the Outstanding Florida Waters (OFW) Florida Department of Environmental Regulation (FDER) rule which directly affects activities that require a permit from the FDER. While old permits are not affected, any new permit application for discharging polluted water into an Outstanding Florida Water system must guarantee that it will not degrade the existing water quality of the OFW.

There are six areas in the Lower St. Johns River Basin designated as Outstanding Florida Waters:

o Haw Creek State Preserve, located in the southeast corner of Crescent Lake,
o Kingsley Lake/North Fork of Black Creek, located in west Clay County,
o Mike Roess/Gold Head Branch State Park, located in southwest Clay County,
o Nassau River/St. Johns River Aquatic Preserve, located at the borderline between Duval and Nassau Counties,

Table 2.2 Trace metal and OPP water quality criteria for the State of Florida (FAC, Chapter 17-3).

Parameter	Units	Class I	Class II	ClassIII	ClassIV	Class V	General Criteria
Trace Metals:							
Aluminum	(mg/l)		>1.5	>1.5			
Antimony	(mg/l)		>0.2	>0.2			
Cadmium	(µm/l)	>0.8	>3.0	>5.0			
Chromium	(mg/l)						>0.05
Copper	(µm/l)	>30	>0.015	>0.15			>0.5
Iron	(mg/l)	>0.3	>0.3	>0.3	>1.0		
Manganese	(mg/l)		>0.1				
Lead	(mg/l)	>0.03		>0.03			>0.05
Mercury	(µm/l)	>0.2	>0.1	>0.1	>0.2	>0.2	
Nickel	(mg/l)	>0.1	>0.1	>0.1	>01		
Selenium	(mg/l)	>0.01	>0.025	>0.025			
Silver	(µm/l)	>0.07	>0.05	>0.05			
Zinc	(mg/l)	>0.03		>0.03			
Pesticides and Herbicides:							
Aldrin/Dieldrin	(µm/l)	>0.003	>0.003	>0.003			
Chlordane	(µm/l)	>0.01	>0.004	>0.004			
2-4-D	(µm/l)	>100					
2,4,5-TP	(µm/l)	>10					
DDT	(µm/l)	>0.001	>0.001	>0.001			
Demeton	(µm/l)	>0.1	>0.1	>0.1			
Endosulfan	(µm/l)	>0.003	>0.001	>0.001			
Endrin	(µm/l)	>0.004	>0.004	>0.004			
Guthion	(µm/l)	>0.01	>0.01	>0.01			
Heptachlor	(µm/l)	>0.001	>0.001	>0.001			
Lindane	(µm/l)	>0.01	>0.004	>0.004			
Malathion	(µm/l)	>0.1	>0.1	>0.1			
Methoxychlor	(µm/l)	>0.03	>0.03	>0.03			
Mirex	(µm/l)	>0.001	>0.001	>0.001			
Parathion	(µm/l)	>0.04	>0.04	>0.04			
Toxaphene	(µm/l)	>0.005	>0.005	>0.005			
Phthalate Esters	(µm/l)	>3.0		>3.0			
PCBs	(µm/l)	>0.001	>0.001	>0.001			

o Ravine State Gardens, located in Palatka, and

o Volusia Water Recharge Area, located in the western part of Volusia County.

Ravine Gardens and the Volusia Water Recharge Area are not directly connected with the St. Johns River system. Nassau River is connected with the St. Johns River through the Intracoastal canal.

Aquatic preserves are state-owned lands protected under the Florida Aquatic Preserve Act (Florida Statutes, Chapter 258). These lands are managed by the Bureau of Land and Aquatic Resource Management, Division of Recreation and Parks, Florida Department of Natural Resources. The purpose of aquatic preserve management is to maintain the natural value of selected areas for future generations. Within the Lower St. Johns River Basin, the only aquatic preserve is the Nassau River/St. Johns River Marsh Aquatic Preserve.

2.4.3 Inland Disposal. Inland disposal is controlled by the Federal Water Pollution Control Act and requires a thorough field investigation of the disposal site, especially if it is characterized as wetland. Inland disposal includes containment and possible treatment of the spoil by chemical or biological methods. Containment can be achieved by means of geomembranes, clay layers, etc. Chemical and biological methods are similar to those applied for treatment of *in-situ* polluted sediments (Sections 2.3.2.2 and 2.3.2.4 of this chapter). In all cases, alternative disposal sites and treatment scenarios should be considered before making any final decision. Inland disposal and treatment must comply with EPA regulations for priority pollutants (Wentz, 1989).

Besides the traditional methods for upland treatment of heavily contaminated disposed sediments, i.e., containment, chemical treatment, and biodegradation, a new methodology has been suggested (Schouten and Rang, 1989). The new methodology, ceramic processing, involves ceramic sintering of the dredged mud. The success of the method depends strongly on the maximum temperature applied, the temperature duration curve and the oxygen supply in the oven. The processes that take place during the ceramic sintering can be generally subdivided according to three temperature ranges. In the first temperature range (100-250° C) hydroxides dehydrate, crystalline water evaporates and the material shrinks substantially; in the second range (250-700° C) combustion of the organic matter takes place while sulphides (e.g., pyrites) are oxidized, clay minerals are dehydroxylated and the crystalline structure of quartz changes; in the third range (700-1200° C) halogen salts, halogenated hydrocarbons, and sulphates decompose, a considerable amount of heavy metals is volatilized, and sintering of the mud occurs (i.e., formation of calcium silicate pellets). The conclusion of a feasibility study on ceramic processing of heavily polluted, dredged mud from estuarine sites in the Netherlands (Schouten and Rang, 1989), indicated several advantages of this method including:

o destruction of organic micro-pollutants (e.g., PCBs), and

o immobilization of heavy metals by ceramic bonding.

2.5 Strategic Management Planning for Contaminated Sediments.

A comprehensive management plan for sediments in an aquatic ecosystem involves a series of investigations, decisions, actions, and evaluations. Presently, there are no State regulations pertaining to management of contaminated sediments. In general, however, the components of such a comprehensive plan are:

o problem identification,

o initial evaluation of the level of contamination,

o source evaluation,

o assessment of the extent and importance of the problem,

o feasibility of remediation,

o determination of disposal or in-situ treatment alternatives,

o identification of potential problems associated with remediation procedures,

o establishment of testing protocol,

o assessment of potential solutions,

o design of implementation strategies,

o selection of remedial options,

o consideration of design

o determination of available control measures,

o monitoring of water/sediment system including physicochemical parameters and toxicity tests both *in-situ* and in the laboratory.

During implementation of the various phases of the management plan, equal consideration should be given to physical, chemical and biological factors, as well as to socio-economic issues (e.g., human health hazards, property devaluation, reduced aesthetics, loss of recreation areas, cost of remedial operations and monitoring, etc.).

Two detailed, sediment management strategy flow charts, developed by Lee et al. (1987) and Thomas (1987), respectively, are given in Figures 2.3 and 2.4. The strategic sediment management plan, shown in Figure 2.3 consists of nine tasks: initial evaluation, consideration of disposal alternatives, identification of potential problems, development of a testing protocol, assessment, implementation strategy, investigation of available options, design feasibility, and design of control measures.

The Lee et al. (1987) management plan starts with an initial evaluation of the site under consideration. If the sediments are found to be uncontaminated, disposal alternatives will be selected using the volume of the spoil as the only criterion.

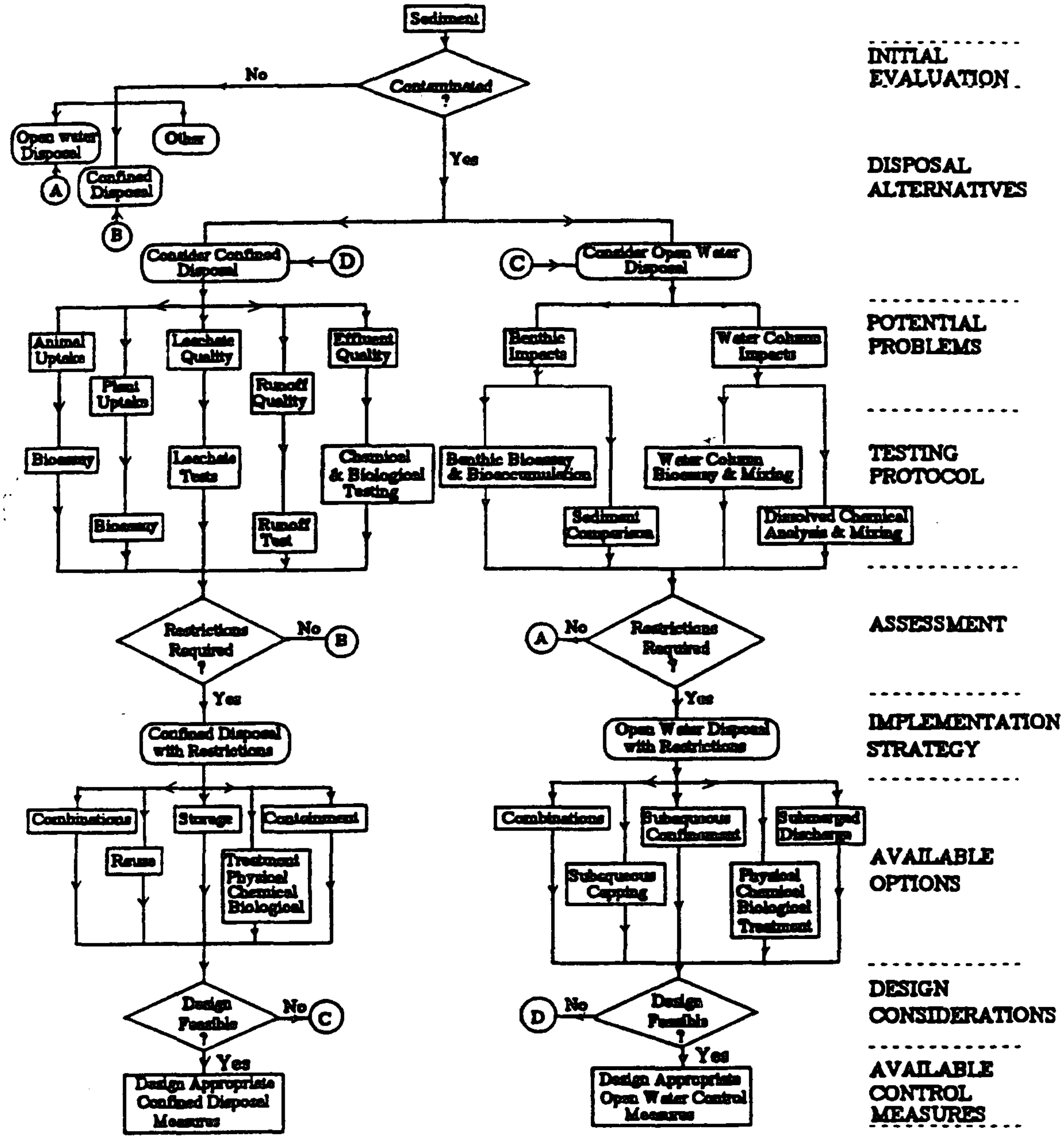


Figure 2.3 Strategic sediment management plan (Lee et al. 1987).

Otherwise, confined disposal (*in-situ* or upland) or open-water disposal should be considered based on anticipated problems such as animal and plant uptake, leachate, runoff or effluent quality, and benthic and water column impacts. Depending on the problem, a testing protocol will be developed. This protocol may incorporate physical (e.g., mixing, leachate, etc.); chemical (e.g., testing for metals, organic pollutants, etc); or biological analyses (e.g., toxicity assay and bioaccumulation tests). Application of the testing protocol will provide an assessment of whether the selected disposal alternative is subject to restrictions or not. If contamination levels are low enough (i.e., within the same order of magnitude but less than the allowable EPA limits), the spoil is treated as if it were uncontaminated. Otherwise, available remedial options (i.e., physical, chemical or biological) are examined, and the study proceeds on the design of the particular treatment method. If the design is successful, then the final task is to develop control measures and a monitoring program; otherwise, another disposal alternative should be considered.

The sediment remediation management plan in Figure 2.4 involves five tasks; i.e., problem identification, problem assessment, feasibility of remediation, development of remedial options, and development of a monitoring plan. The plan begins with problem identification, which includes a preliminary site assessment and selection of an action plan (e.g., dredging or leaving as-is in the case of unpolluted sediments). If the sediments are contaminated, any further action depends on whether the source can be controlled or not. No remedial action for sediment treatment is recommended until the source is controlled; otherwise, remedial plans are investigated based on data pertaining to pollutant mobility and the area extent of the contaminated site. Based on the

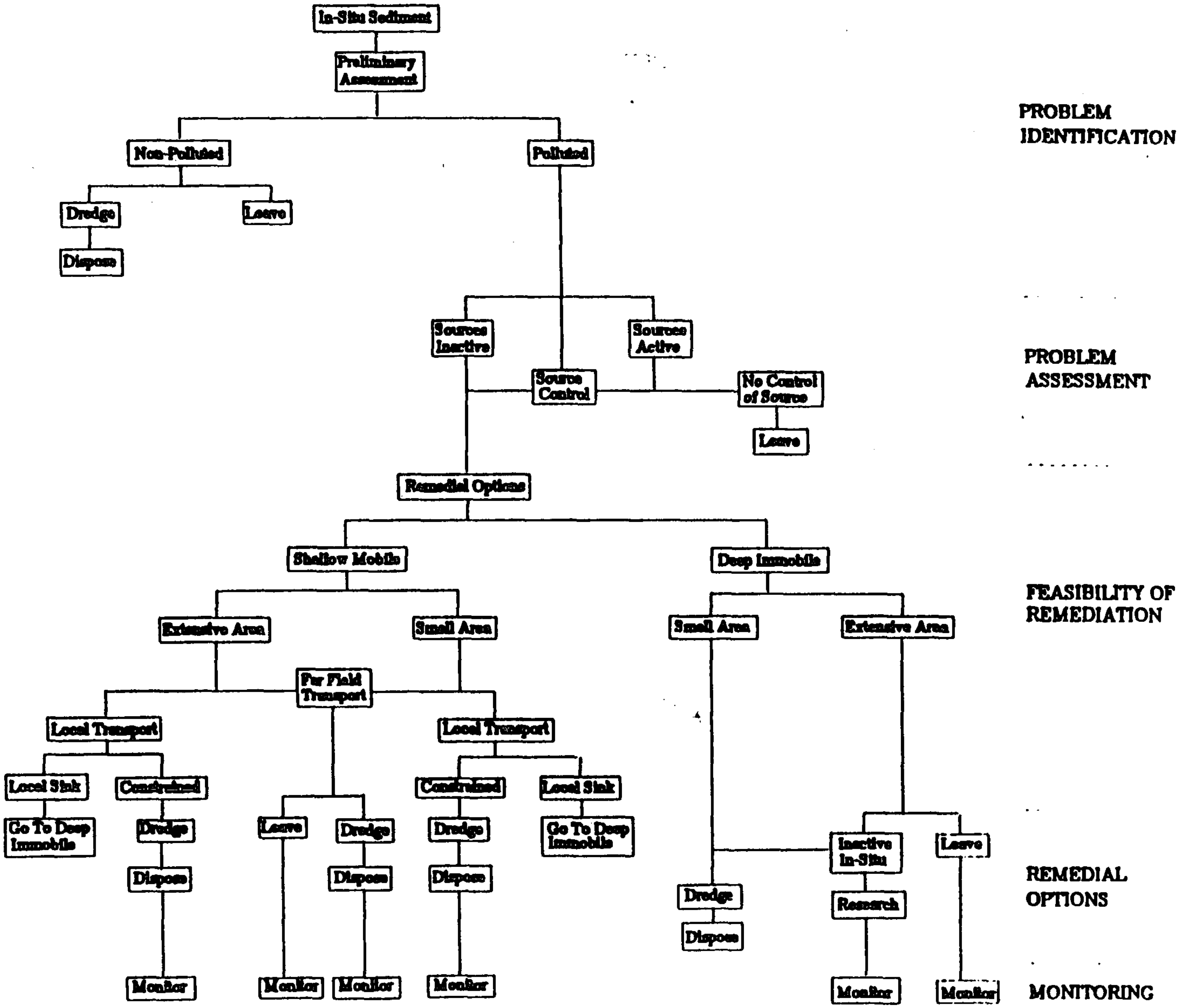


Figure 2.4 Flow chart for sediment remediation management plan (Thomas, 1987).

investigation results, remedial actions are selected (i.e., dredging or in-situ treatment) and a monitoring plan is developed.

F. ENVIRONMENTAL IMPACT ASSESSMENT

2.6 Environmental Impact Assessment (EIA) Issues and Methods.

The environmental impact assessment (EIA) is one of the most important tools used to decide how to manage contaminated sediments. This assessment compares the "benefits" and "costs" of leaving a contaminated area untreated versus taking remedial action for the same site. The "cost-benefit" analysis should incorporate both ecological and human factors. The ecological factors should include: geography-geomorphology, hydrology/hydraulics/sediments, physicochemistry, and biology. The human factors include: sociology and culture, economy, and human health.

2.6.1 EIA Components. The general steps required for an EIA are :

o identification of the location, size, special features, and degree of contamination of the polluted site,

o identification of pollution sources,

o qualitative description of the ecological components which are expected to be affected by the contaminated site, whether it is left untreated or remediated,

o establishment of baseline data and information,

o quantitative analysis and simulation of the anticipated pre-remediation and post-remediation environmental conditions (including the no-action option).

o analysis of the projected, possible changes in the prevailing ecological conditions,

o assessment of the trade-offs between beneficial and detrimental effects (e.g., health, cost, aesthetics, etc.),

- o statement of monitoring methodology and criteria for assessing attainment of environmental objectives, and
- o development of alternative plans.

2.6.2 EIA Methodologies. Since the National Environmental Policy Act became effective in the United States on January, 1, 1970, a number of EIA methodologies have been established. To facilitate the assessment procedures, a list of environmental indices has been developed. Most of the EIA projects incorporate 50 to 100 indices (Canter, 1985); but for major EIAs, the number can be increased up to 1,000. Of course, these indices are general and are not limited to sediments. Environmental indices can provide valuable information for baseline data collection. Four of the most extensively applied EIA methodologies are (Singh et al. 1985):

- o interaction matrix method,
- o checklist method,
- o network method, and
- o energetic method.

A general, conceptual approach to EIA is presented in Figure 2.5 (Beanlands, 1987).

2.6.2.1 Interaction Matrix Method. The interaction matrix method displays the action-response relationships in a tabular, matrix form (Figure 2.6). The matrix presents the existing environmental features and the anticipated environmental effects resulting from a particular action. Anticipated impacts are marked on the matrix and provide a synoptic and simplistic picture of the environmental assessment. Quantitative cause-effect investigations can later be conducted based on the results of the interaction matrix

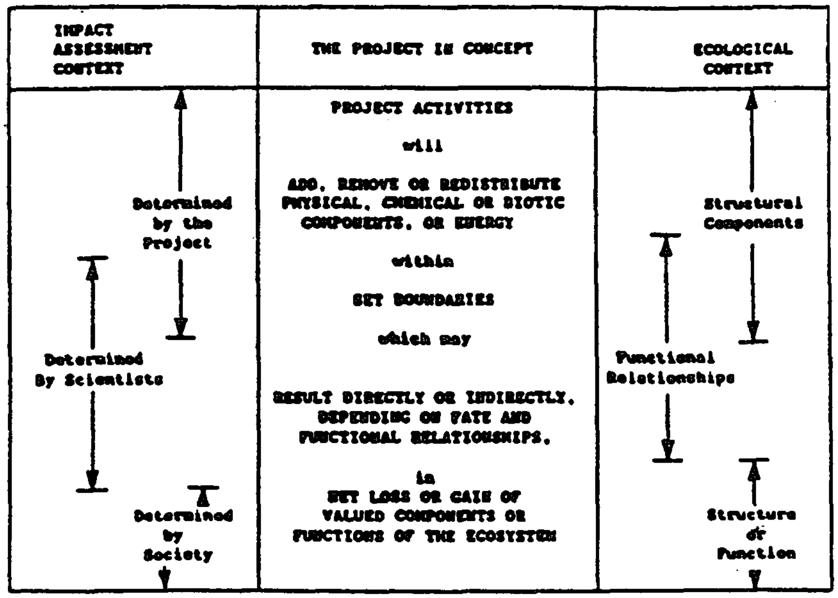


Figure 2.5 Conceptual approach to EIA (Beanlands, 1987).

Existing Conditions	Impact of Proposed Action						
	Aquatic biota	Turbidity	Aesthetics	Odor	Navigability	Recreation	Property Values
Land Uses			□			□	
Water Quality		○			○		
Sediments	○			○			
Fisheries		○			○		
Public Health						□	
Economy			□			□	□

○ During Implementation □ Post Project

Figure 2.6 Interaction matrix method for EIA.

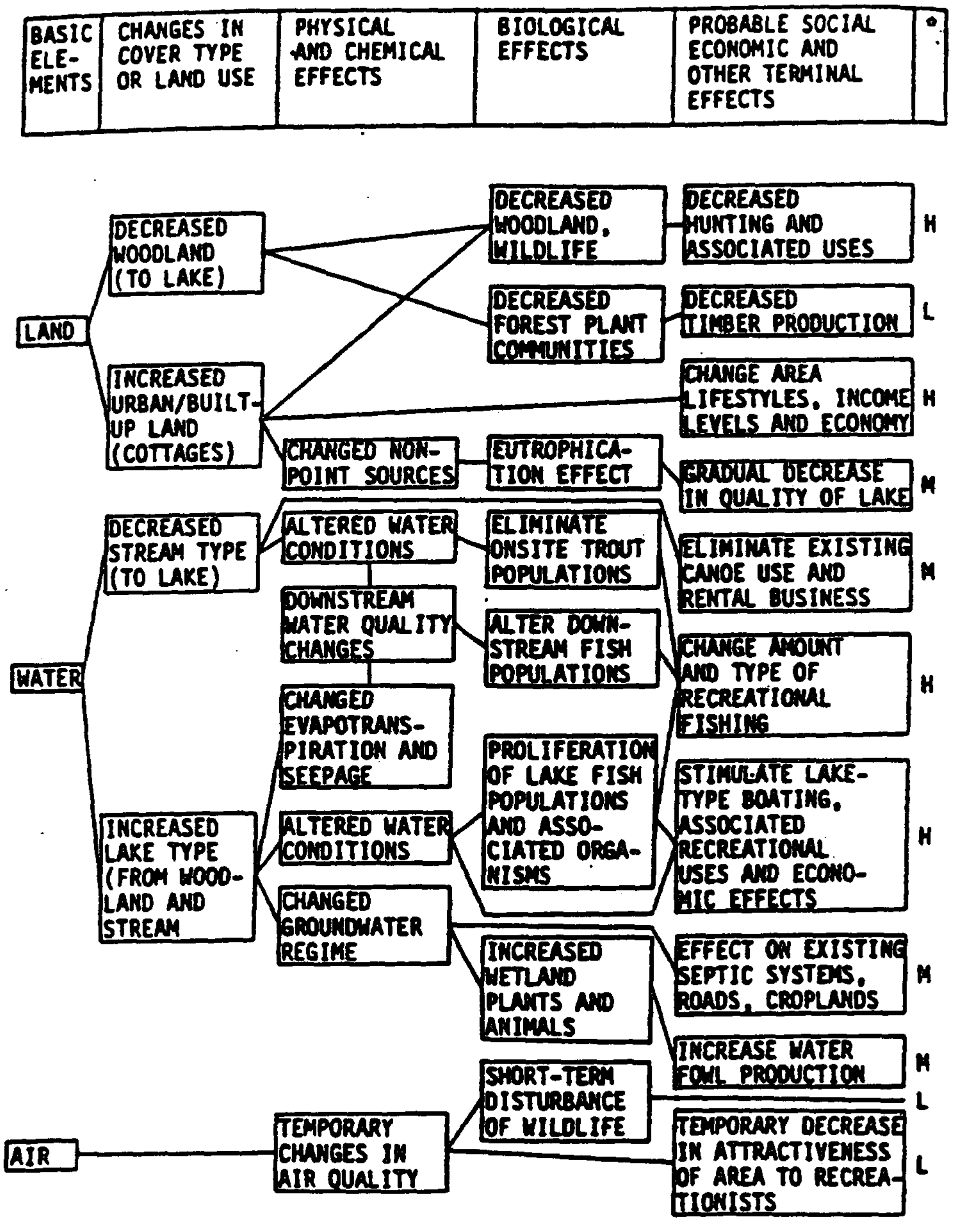
method. For sediment quality studies, an interaction matrix should relate existing environmental conditions to anticipated sediment-induced effects (i.e., shoaling, turbidity, release of contaminants, etc.) as a result of a particular action (e.g., dredging).

2.6.2.2 Checklist Methods. Checklist methods can vary from a simple checklist form to elaborate weighing factor schemes. A checklist describing 62 environmental quality indices is given in Table 2.3 (Canter and Hill, 1979). The parameters of this list are evaluated either quantitatively or qualitatively and compared to some predetermined standards. EIA checklist methods provide an inventory of the environmental components that would be affected by implementation of a proposed project. For sediment quality studies, the checklist in Table 2.3 can be used to assess the impact of a sediment management plan on the sixty two components listed in this table. A negative impact can be marked with a minus sign (-) next to the affected component and a positive impact with a positive sign (+). In this way, a synoptic view of the overall impact of a management action (e.g., dredging and island disposal) can be easily assessed.

2.6.2.3 Network Methods. Network methods represent action-response relations through a "node-link" network. The network represents the original action to all possible components and interactive scenarios and assesses the importance of final effects (U.S. Soil Conservation Service, 1977). Networks methods incorporate a wide variety of physicochemical, biological, and socioeconomic factors related to a sediment management plan. For example, if a network method is applied to assess the environmental impact of a dredging operation, then all possible effects should be examined and their impact qualitatively assessed. The more complete the network, the more accurate is the EIA. Illustrative examples of a network method are given in Figures 2.7a,b.

Table 2.3 Environmental quality indices (Canter and Hill, 1979).

Category	Component	Index #	Factor
Terrestrial	Population	1.	Crops
		2.	Natural vegetation
		3.	Herbivorous mammals
		4.	Carnivorous mammals
		5.	Upland game birds
		6.	Predatory birds
	Habitat/Land	7.	Bottomland forest
		8.	Upland forests
		9.	Open lands
		10.	Littoral zone
		11.	Land use
	Land Quality/ Soil erosion	12.	Soil erosion
		13.	Soil chemistry
		14.	Mineral extraction
	Communities	15.	Species diversity
Aquatic	Populations	16.	Natural Vegetation
		17.	Wetland vegetation
		18.	Zooplankton
		19.	Phytoplankton
		20.	Sport fish
		21.	Commercial fisheries
		22.	Intertidal organisms
		23.	Benthos/epibenthos
		24.	Waterfowl
	Habitats	25.	Stream
		26.	Freshwater lake
		27.	River swamp
		28.	Nonriver swamp
	Water quality	29.	pH
		30.	Turbidity
		31.	Suspended solids
		32.	Water temperature
		33.	Dissolved oxygen
		34.	BOD
		35.	Dissolved solids
		36.	Inorganic nitrogen
		37.	Inorganic phosphate
		38.	Salinity
		39.	Iron and manganese
		40.	Toxic substances
		41.	Pesticides
		42.	Fecal coliforms
		43.	Stream assimilative capacity
	Water quantity	44.	Stream flow variations
		45.	Basin hydrologic loss
	Communities	46.	Species diversity
Air	Air quality	47.	Carbon monoxide
		48.	Hydrocarbons
		49.	Oxides of nitrogen
		50.	Particulates
	Climatology	51.	Diffusion factor
Human Interface	Noise	52.	Noise
	Esthetics	53.	Geometric factors
		54.	Variety of vegetation type
		55.	Animals-domestic
		56.	Native fauna
		57.	Appearance of water
		58.	Odor and floating materials
		59.	Odor and visual quality
		60.	Sound
	Historical	61.	Historical factors
	Archeological	62.	Archeological factors



*Probable Importance of Terminal Effects (H - high; M - moderate; L - low).

Figure 2.7a Network diagram for EIA (U.S. Soil Conservation Service, 1977).

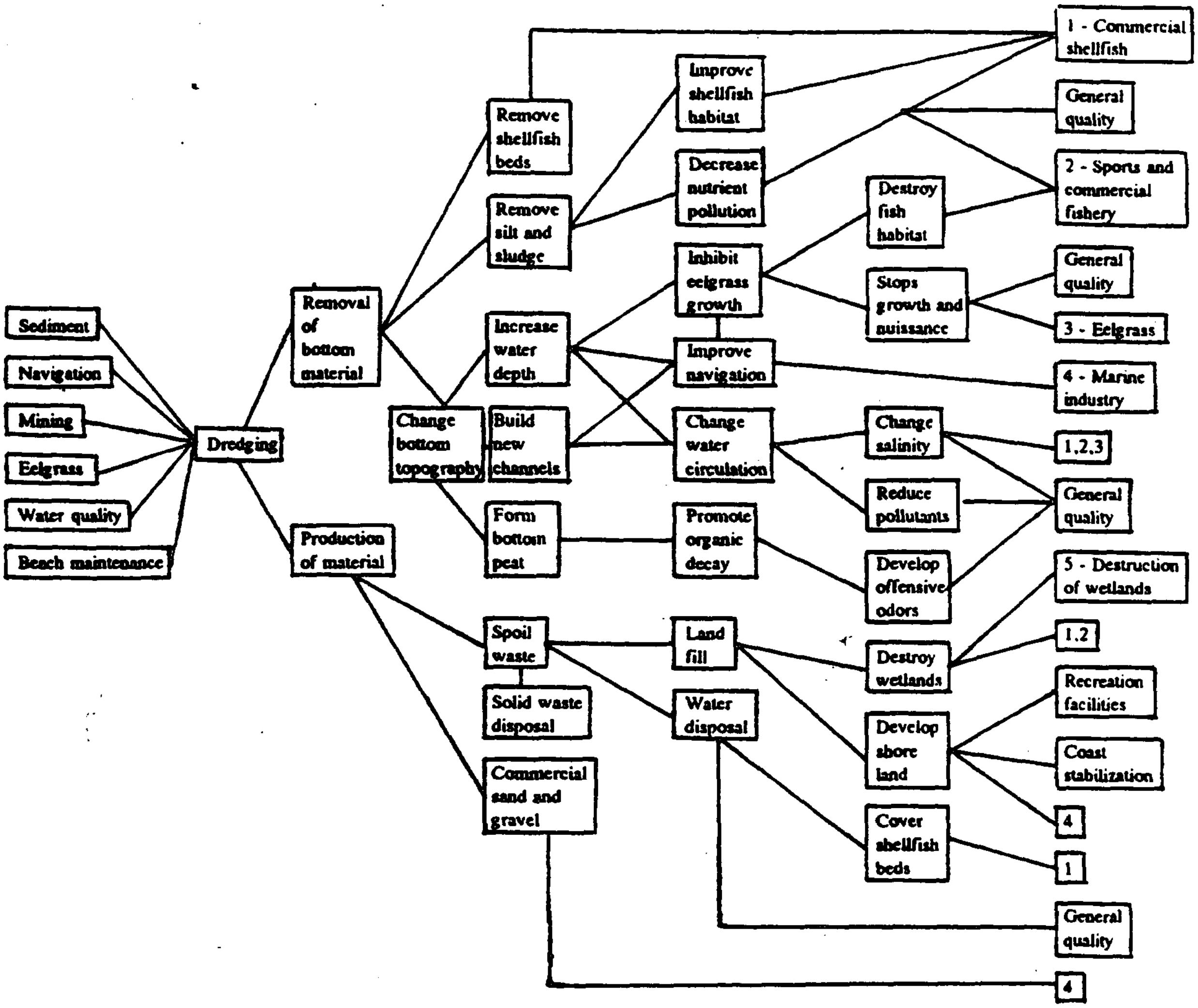


Figure 2.7b Network diagram for dredging project (Unknown source).

2.6.2.4 Energetic Methods. Energetic methods are based on the concept of "energy" flow and storage (Odum, 1971). Energetic methods describe the relationships between humans and the environment and require a high level of technical information and resources. An energetic method was applied to the assessment of the environmental, economic and hydrologic impacts of the Palm Bay coastal area in Florida (Barile, 1977). Hopkinson (1988) applied the energetic method for assessment of organic carbon exchange in estuarine/salt marsh ecosystems. Generally, energetic methods are hard to apply due to the detailed level of information required for quantification of the flow/storage processes. This method has not been applied for sediment contamination problems. However, it would be applicable if contamination sources, storage, and sinks can be quantified along with their interaction and transition mechanisms.

2.7 Indicators of Sediment Pollution.

Environmental quality indicators can be of great value in an EIA if they can be identified quantitatively. Traditionally, environmental indicators were limited to water quality. Recently, however, numerical indicators for aquatic sediment quality have been under development (MacDonald, 1992).

2.7.1 Water Quality Indicators. The Florida Administrative Code, Chapter 17-3, defines allowable limits for heavy metals, bacterial growth, DO, pH, turbidity, nutrients and organic compounds for the five classes of surface waters. A water quality index (WQI) for pristine waters developed by the National Sanitation Foundation (Horton, 1965) incorporates nine environmental factors and is defined as follows:

$$WQI = \prod_{i=1}^{i=9} (I_i w_i), \quad i = 1,\ldots,9 \qquad (2.1)$$

where Π is a multiplication operator, I_i is the environmental factor, and w_i is a weighing exponent. The environmental factors and their associated weighing exponents are given in Table 2.4. The correlations between index I_i and the actual values of the variables are given in Figures 2.8a,b. The three curves in Figures 2.8a,b correspond to the average value and the confidence limits between the parameters and the environmental factor I_i. Thus, based on these curves a sensitivity analysis can be conducted by comparing the

Table 2.4 Water Quality Index environmental factors and weights (Horton, 1965).

i	Factor I_i	Weighing exponent (w_i)
1	pH	0.12
2	Turbidity	0.08
3	Suspended solids	0.08
4	Water temperature	0.10
5	Dissolved oxygen	0.17
6	BOD	0.10
7	Inorganic nitrogen	0.10
8	Inorganic phosphate	0.10
9	Fecal coliforms	0.15

WQIs estimated from both the average and the extreme values of the I_i indices. The confidence limits are indicative of the ecological tolerance versus the variability of the parameters involved. From the same figures it is evident that the environmental effects of turbidity, total solids and total phosphates may vary drastically from one system to another. On the other hand, temperature, pH and DO allow only a narrow deviation from their average values. The value of the WQI ranges from 1 to 100. Classification of the environmental quality level of an aquatic ecosystem based on the water quality

Table 2.5 Water quality as defined by the WQI (Horton, 1965).

Water Quality	WQI
Excellent	91-100
Good	71-90
Medium	51-70
Poor	26-50
Very Bad	0-25

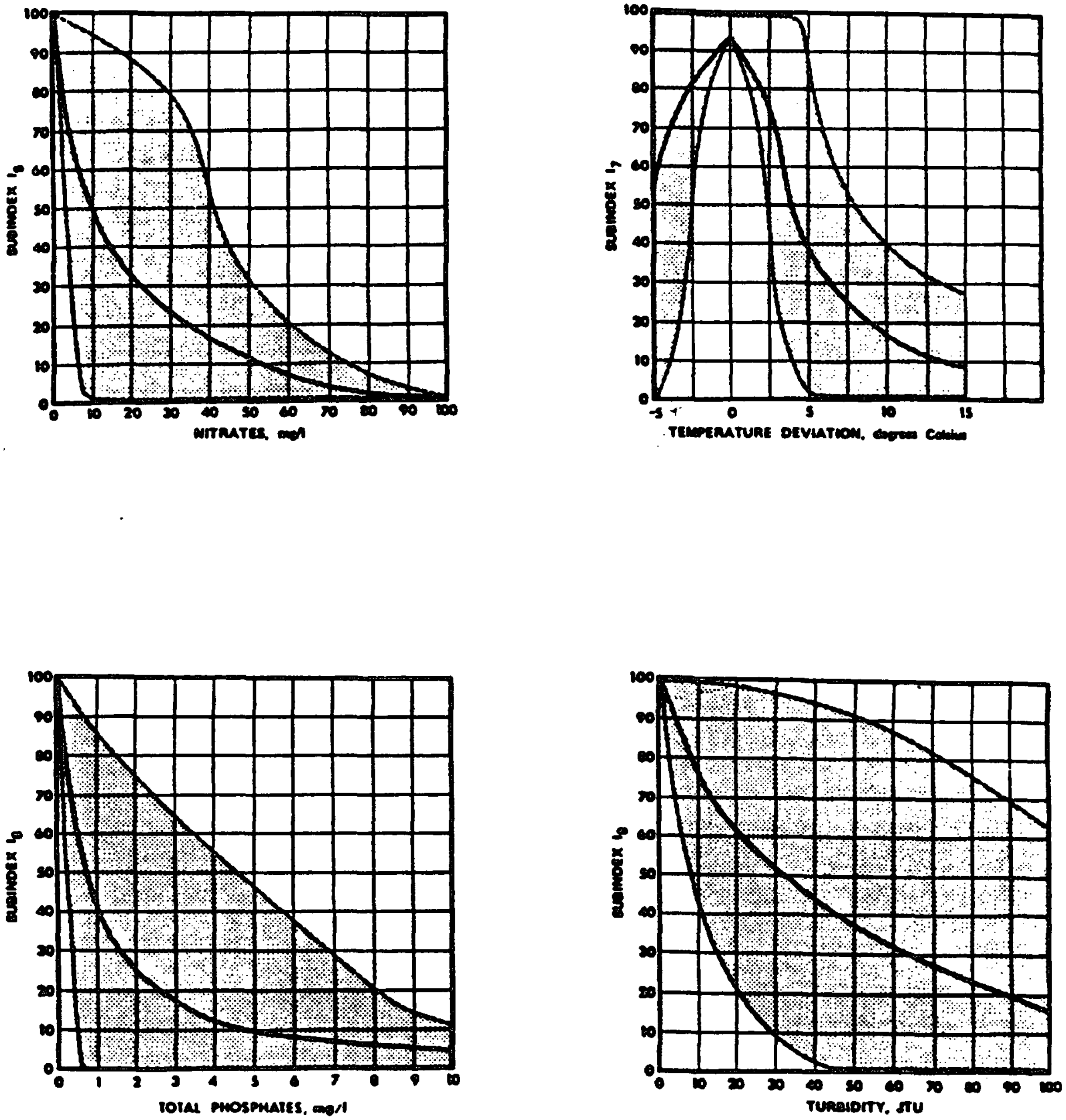


Figure 2.8a Water quality indices (WQIs) (Horton, 1965).

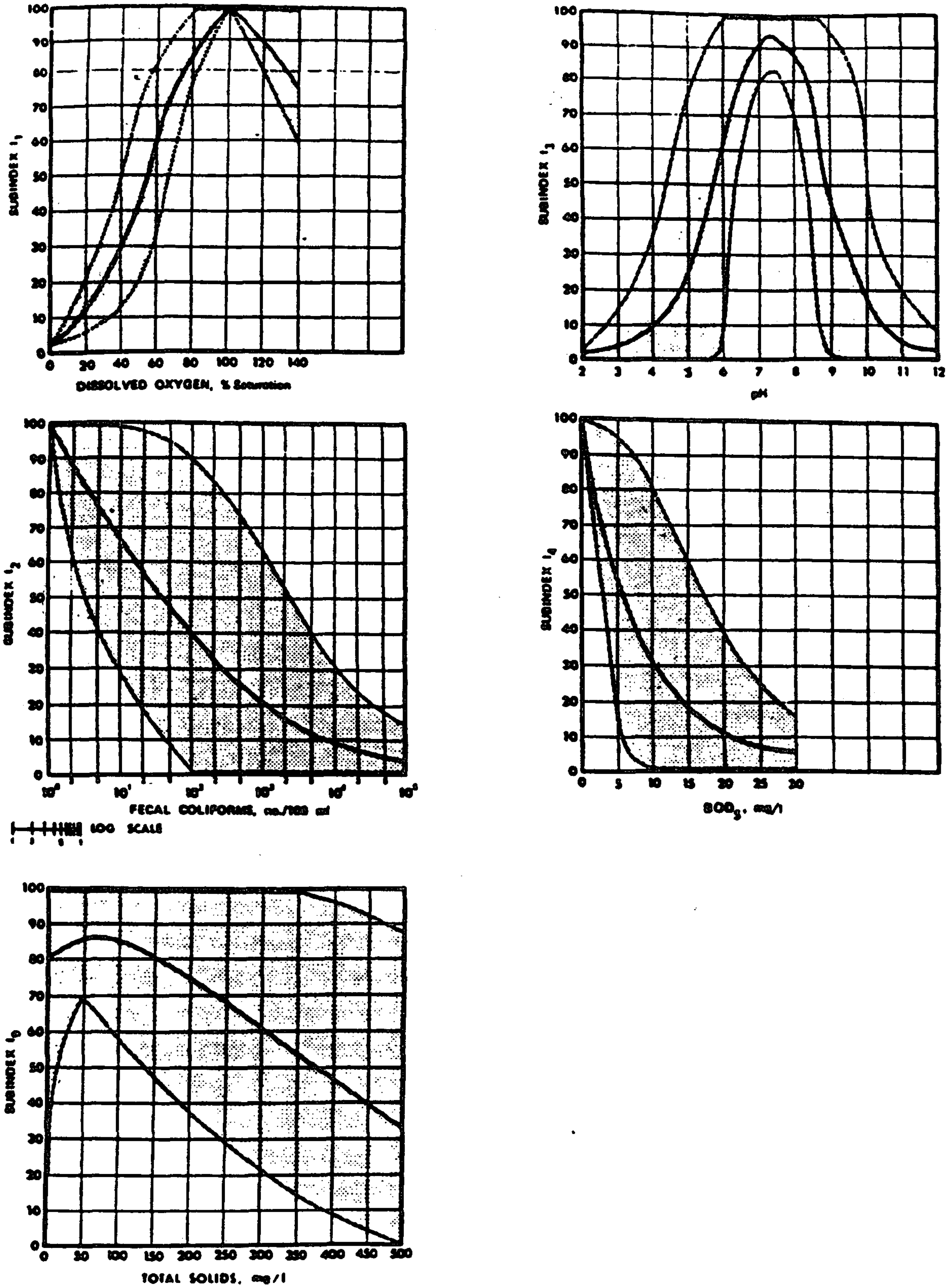


Figure 2.8b Water quality indices (WQIs) (Horton, 1965).

index is given in Table 2.5. WQIs are widely used for estimation of the environmental quality of aquatic ecosystems, since sediment quality standards, although available, they are not as yet formally adapted or standardized.

2.7.2 Sediment Quality Indices. The U.S. Environmental Protection Agency (EPA) has developed a range of contamination levels, which define pollution levels due to different elements and chemical compounds. The ranges for unpolluted, moderately polluted, and heavily polluted sediments are listed in Table 2.6.

A general list of ecological aquatic and terrestrial indicators was recently

Table 2.6 EPA sediment quality criteria (Masters, 1991).

Chemical	Unpolluted	Moderately polluted	Heavily polluted
Ammonia	<75	75-200	>200
Arsenic	<3	3-8	>8
Barium	<20	20-60	>60
Cadmium	-	-	>6
Chromium	<25	25-75	>75
COD	<40,000	40,000-80,000	>80,000
Copper	<25	25-50	>50
Cyanide	<0.10	0.10-0.25	>0.25
Hexane solubles	<1,000	1,000-2,000	>2,000
Iron	<17,000	17,000-25,000	>25,000
Lead	<40	40-60	>60
Manganese	<300	300-500	>500
Mercury	<1.0		>1.0
Nickel	<20	20-50	>50
Phosphorus	<420	420-650	>650
TKN	<1,000	1,000-2,000	>2,000
Volatile solids	<5%	5-8%	>8%
Zinc	<90	90-200	>200

Note: All units are in mg/kg dry weight except Volatile solids.

developed by the U.S. Environmental Protection Agency (EPA) (Hunsaker and Carpenter, 1990). For estuarine sediments, EPA suggested two quantitative indicators: a) acute sediment toxicity, and b) number of chemical contaminants in sediments.

2.7.2.1 Sediment Toxicity Assessment. The determination of sediment toxicity is based on laboratory toxicity tests of field-collected sediment samples. Toxicity tests are conducted under static laboratory conditions (i.e., constant salinity and temperature) for ten days using grab samples of 200 mL of sediment (Hunsaker and Carpenter, 1990). The American Society for Testing and Materials (ASTM) has recently developed standard methods for fresh water and estuarine amphipod sediment bioassays (ASTM, 1990; Ingersoll, 1991). For acute sediment toxicity bioassays in the laboratory, the following amphipods have been used:

o *Ampelisca abdita*,

o *Eohaustorius estuaries*,

o *Grandidierella japonica*,

o *Hyalella azteca*,

o *Rhepoxynius abronius*, or

o *Rhepoxynius hudsoni*.

For *in situ* toxicity tests different types of biota can be used, such as "earthworm" species, freshwater mussels, fish, etc.

2.7.2.2 Chemical Assessment of Sediments. To determine the number of chemicals in the sediment, samples of surficial sediments (top 2 cm) are collected and analyzed for toxic metals, major elements, Cl-pesticides, PCBs, PAHs, coprostanol, and *Clostridium* spores. These parameters were incorporated in sediment surveys conducted by the NOAA National Status and Trends Program in the S.E. Atlantic (including St. Johns River) and Gulf of Mexico coasts (Hanson and Evans, 1991). The number of chemicals present in the sediments indicates the degree of contamination.

2.7.3 Sedimentological Risk Index. Hakanson (1980) suggested that a sedimentological risk index (RI) for assessment of the effects of toxic substances should be based on the following four "requirements":

o pollutant concentration requirement,
o number of pollutants requirement,
o toxic factor requirement, and
o sensitivity requirement.

2.7.3.1 Pollutant Concentration Requirement. The concentration requirement involves comparison of historic uncontaminated sediments with recent contaminated deposits. Besides chemical analysis of the sediments, this requirement entails calculation of long-term erosion and sediment transport as well as pollutant sources and pathways, so that the sediment/pollutant accumulation patterns are established. Thus, the pollutant concentration requirement provides not only the degree but also the rate of sediment contamination.

2.7.3.2 Number of Pollutants Requirement. The number requirement emphasizes the fact that the environmental risk increases with the increasing number of pollutants. It also provides a basic list of the pollutants that should be included in estuarine ecological studies. This list includes the most commonly found metals, such as mercury, cadmium, lead, copper, zinc and selenium, as well as the major elements; i.e., silica, aluminum, potassium, magnesium, sodium, calcium, manganese, nitrogen, and phosphorus. Toxic compounds (such as PCBs, DDT, PAHs, oil and grease) are also included. In order to quantify the importance of the number requirement, Hakanson (1980) proposed that the degree of contamination is defined either by the individual, (C_i/C_{oi}), or the cumulative contamination factor, C_d, given as follows:

$$C_d = \sum_{i=1}^{i=8} (C_i/C_{oi}) \ , \ i = 1,\ldots,8 \qquad (2.2)$$

where C_i is the concentration of the pollutant i in the surficial sediment layer (top 1 cm) from the contaminated site, and C_{oi} is the concentration of the same pollutant at the standard pre-industrial concentration level. Hakanson's contaminant list includes PCBs, mercury, cadmium, arsenic, copper, lead, chromium and zinc. Pre-industrial standard values for these elements in lake sediments are given in Table 2.7.

Table 2.7 Preindustrial standard values for toxic chemicals in lakes (Hakanson, 1980).

Chemical	As	Cd	Cr	Cu	Hg	Pb	Zn	PCB
Concentration (ppm)	15	1.0	90	50	0.25	70	175	0.01

Although these values may vary from one aquatic ecosystem to another, they are representative of the order of magnitude of heavy metal and PCB concentrations in natural, "undisturbed" systems including estuaries. The degree of sediment contamination, based on the value of the individual and cumulative contamination factors (C_i/C_{oi} and C_d), is tabulated in Table 2.8.

2.7.3.3 Toxic Factor Requirement. The toxic factor requirement, expressed

Table 2.8 Degree of contamination according to the factor C_d (Hakanson, 1980).

Factor C_d	C_i/C_{oi}	Degree of contamination
$C_d < 8$	$C_i/C_{oi} < 1$	Low degree of contamination
$8 \le C_d < 16$	$1 \le C_i/C_{oi} < 3$	Moderate degree of contamination
$16 \le C_d < 32$	$3 \le C_i/C_{oi} < 6$	Considerable degree of contamination
$32 \le C_d$	$6 \le C_i/C_{oi}$	Very high degree of contamination

by the toxic factor (S_t), accounts for the relative natural background abundance of the various contaminants found in igneous rocks, soils, freshwater, land plants, and land animals, as well as their tendency to be deposited in sediments.

2.7.3.4 Sensitivity Requirement. The sensitivity requirement utilizes the toxic-response factor (T_i), which is defined as the product of the toxic factor (S_t) with the level of bioproductivity quantified by some function of the bioproduction index (BPI). The BPI-value is obtained by measuring the loss on ignition (IG) and the nitrogen content (N) of sediment samples. Quantitatively the BPI-value is defined as the N-content on the regression line for loss on ignition equal to 10% (IG=0.1) (Horie, 1969). The IG-value is taken as the loss on ignition of dry sediment samples subject to 550° C for 1 hour. The N-value is determined according to Kjeldahl standard method. The values for both the toxic factor (S_t) and the toxic-response factor (T_i), for all of the pollutants in Hakanson's contaminant list, are listed in Table 2.9. Numerical the toxic-response factor is defined as

$$T_i = S_t\,(5/\mathrm{BPI})^{\alpha} \qquad (2.3)$$

where α is a constant exponent.

Based on the four previously mentioned requirements, the sedimentological risk

Table 2.9 Toxic factor (S_t) and toxic-response factor (T_i) (Hakanson, 1980).

Chemical	S_t-value	T_i-value
As	10	$10\ (5/BPI)^{0}$
Cd	30	$30\ (5/BPI)^{1/2}$
Cr	2	$2\ (5/BPI)^{1/2}$
Cu	5	$5\ (5/BPI)^{1/2}$
Hg	40	$40\ (5/BPI)^{1}$
Pb	5	$5\ (5/BPI)^{1/2}$
Zn	1	$1\ (5/BPI)^{1/2}$
PCB	40	$40\ (5/BPI)^{-1}$

Note: BPI = bioproduction index.

factor, RI, is quantified as follows:

$$RI = \sum_{i=1}^{i=8} T_i\ (C_i/C_{oi}) = \sum_{i=1}^{i=8} S_{ti}(5/BPI)^{a}(C_i/C_{oi}), \quad i = 1,....,8 \qquad (2.4)$$

Characterization of the potential ecological risk of an aquatic ecosystem based on the risk factor RI is presented in Table 2.10.

Table 2.10 Assessment of ecological risk (Hakanson, 1980).

Risk factor	Environmental risk
$RI < 150$	Low ecological risk
$150 \leq RI < 300$	Moderate ecological risk
$300 \leq RI < 600$	Considerable ecological risk
$600 \leq RI$	Very high ecological risk

2.7.4 Metal-Aluminum Ratio. For assessment of metal enrichment attributable to anthropogenic activities, the Florida Department of Environmental Regulation uses the metal-aluminum ratio technique. This technique compares the metal-aluminum ratios in carbonate rocks of uncontaminated ("natural") ecosystems with those from areas suspected of being polluted (See Chapter III, Section 3.2.1.2). A deviation from the 95% prediction limits of "natural" metal-aluminum ratios is an indication of metal enrichment (Pierce et al. 1988).

2.7.5 Biotic Indices. Besides the sedimentological approach, aquatic pollution can be effectively assessed by means of biotic indices. Biotic indices are referred generally to species diversity, similarity and stability. Species diversity includes both the number of species in the community and the evenness with which the individuals are divided among the species.

Comparison of biotic indices derived from surveys of different areas require detailed information on: sampling method, sample size, depth of sampling (for benthic communities), duration of sampling, time of the year, and taxonomic level (Hughes, 1978).

2.7.5.1 Biodiversity Indices. One of the most widely applied biodiversity indices is the Shannon-Wiener index, H', (Krebs, 1989) defined as

$$H'' = -K \sum_{i=1}^{i=N} p_i \log_2 p_i \quad \ldots\ldots\ldots \quad (2.5)$$

where K is a constant which depends on the choice of the unit measure (usually K=1), N is the number of species, and p_i is the proportion of total sample belonging to i^{th} species (Krebs, 1989). The Shannon-Wiener index is recommended for random samples drawn from a large population for which the total number of species is known. If these conditions are not met then a more suitable approach is the Brillouin index, H,

$$H = (1/N) \log[N!/(n_1!n_2!n_3!\ldots)] \quad \ldots\ldots (2.6)$$

where n_i is the number of individuals belonging to species i. For efficient estimation of the Shannon-Wiener and Brillouin indices, computer programs such as the DIVERS are available (Krebs, 1989).

Biotic stability is defined as the ability of a community to remain almost similar to itself regardless of external variations, and is commonly associated to biodiversity (Macarthur, 1955).

2.7.5.2 Similarity Indices. Similarity indices are often applied for ordering samples according to overall similarities and then comparing the major deviations for possible correlations with environmental factors. One widely used similarity index is the percentage similarity (PSC) defined as:

$$PSC = 100 - \tfrac{1}{2} \sum_{i=1}^{i=N} |a-b| \quad \ldots\ldots (2.7)$$

where N is the number of species, and a, b are (for a given species) percentages of the total samples A and B which that species represent (Whittaker and Fairbanks, 1958).

2.7.5.3 Karr's Index of Biotic Integrity. Another approach for assessment of the integrity of aquatic biota utilizes information about the fish communities (Karr, 1981). This approach uses a classification system of six main categories (Table 2.11). These categories are assessed based on a number of parameters which are indicative of the health of fish communities (Table 2.12).

Application of Karr's approach presumes that:

o The fish sample is large enough to be representative of the fish community at the sampling site,

Table 2.11 Biotic integrity classes (Karr, 1981).

Class	IBI No.	Attributes
Excellent (E)	57-60	No anthropogenic influence; all regional species including the most intolerant are present with full array of sex and age classes; balanced trophic structure.
(E-G)	53-56	
Good (G)	48-52	Species richness slightly less than expected; some species with less than optimal abundance or size distributions; signs of stress on trophic structure.
(G-F)	45-47	
Fair (F)	39-44	Fewer intolerant species; older age classes of top predators may be rare; Skewed trophic structure (e.g., increasing frequency of omnivores.
(F-P)	36-38	
Poor (P)	28-35	Dominated by pollution tolerant forms, omnivores, and habitat generalists; growth rates and condition factors are depressed; presence of hybrids and deceased fish.
(P-VP)	24-27	
Very Poor (VP)	≤ 23	Few fish, either introduced or of very tolerant forms; hybrid species, deceased fish, parasites tumors and other abnormalities very common.
No Fish		Repetitive sampling failed to show any fish.

Table 2.12 Parameters used in assessment of fish communities (Karr, 1981).

A. Species Composition and Richness

- Number of Species
- Presence of Intolerant Species
- Species Richness and Composition of Darters
- Species Richness and Composition of Suckers
- Species Richness and Composition of Sunfish (except Green Sunfish)
- Proportion of Green Sunfish
- Proportion of Hybrid Individuals

B. Ecological Factors

- Number of Individuals in Sample
- Proportion of Omnivores (Individuals)
- Proportion of Insectivorous Cyprinids
- Proportion of Top Carnivores
- Proportion with Disease, Tumors, Fin Damage, and Other Anomalies

o The sampling site is representative of the larger area of interest, and

o The data collector is a trained biologist with strong familiarity with the local fisheries.

The classification of the system is done using an index of biotic integrity (IBI) number (Table 2.11). This number is calculated by assigning a value to each of the parameters listed in Table 2.12 and adding all of these values for each sampling site. More specifically, the assigned values are: 1 for degraded conditions, 3 for average conditions, and 5 for excellent conditions.

2.7.5.4 Rapid Bioassessment Protocols. Another interesting group of methods for assessment of the environmental conditions in rivers and streams are the rapid bioassessment protocols (RBPs) (Plafkin et al. 1989). These protocols advocate an integrated assessment, comparing existing habitat, physicochemical features and biological parameters with empirically defined reference conditions. The habitat assessment involves characterization of the conditions of the stream substrate and bed cover (i.e., bottom substrate and available cover, embeddedness, flow/velocity), channel morphology (i.e., channel alternation, bottom scouring/deposition, pool/riffle, run/bend ratio), and riparian and bank structure (i.e., bank stability, bank vegetation, streamside cover). The physical features include land use, watershed sediment budget, nonpoint-source pollution, stream characteristics (i.e., width, depth, high water mark, velocity), channel improvements (i.e., dam, channelization), canopy cover, and sediment characteristics (i.e., odor, oils, deposits, inorganic components, organic components). The chemical features include water characteristics (i.e., temperature, DO, pH, conductivity, odors, turbidity, oils) and stream designation according to State water quality standards. The biological parameters include sampling of benthic communities (a 100-count subsample is recommended) and fish sampling (preferably by the electrofishing method).

There are five rapid bioassessment protocols. Each of these RBIs requires a different level of data collection and analysis effort (Table 2.13).

Table 2.13 Comparison of rapid bioassessment protocols (Plafkin et al. 1989).

PROTOCOL I	
Objectives:	Determine whether biological impairment exists and if further investigation is needed.
Effort:	Field: 1-2 hrs, 1 biologist; Data analysis: 0.5-1 hr, 1 biologist; Laboratory: None
Conclusion:	Determine if impairment exists; Indicate generic cause of impairment (habitat, organic enrichment, toxicity)
PROTOCOL II	
Objectives:	Assess biological impairment; Provide information for ranking sites; Prioritize sites for further assessment and/or testing (chemical/toxicity).
Effort:	Field: 1.5-2.5 hrs/person, 1 biologist & 1 technician; Data analysis: 2-4 hrs, 1 biologist; Laboratory: None.
Conclusion:	Characterize conditions as no impairment, moderate impairment, severe impairment; Indicate generic cause of impairment (habitat, organic enrichment, toxicity).
PROTOCOL III	
Objectives:	Assess biological impairment; Establish basis for trend monitoring; Prioritize for further assessment and/or testing (chemical, toxicity).
Effort:	Field: 1-2 hrs/person, 1 biologist & 1 technician; Data analysis: 1-3 hrs, 1 biologist; Laboratory; 2-3 hrs, 1 technician.
Conclusion:	Evaluate site as no impairment, slight impairment, moderate impairment, severe impairment; Indicate generic cause of impairment (habitat, organic enrichment, toxicity).
PROTOCOL IV	
Objectives:	Determine whether biological impairment exists and if further investigation is needed.
Effort:	Field: None; Data analysis: 3 hrs, 1 biologist; Laboratory: None.
Conclusion:	Determine if impairment exists; Indicate generic cause of impairment (habitat, water quality).
PROTOCOL V	
Objectives:	Assess biological impairment; Establish basis for trend monitoring; Provide information for ranking sites; Prioritize for further assessment and/or testing (chemical, toxicity).
Effort:	Field: 1-5 hrs/person, 1 biologist & 1 technician; Data analysis: 1-2 hrs, 1 biologist; Laboratory: None.
Conclusion:	Evaluate biological integrity as excellent, good, fair, poor, very poor; Indicate generic cause of impairment (habitat, organic enrichment, toxicity).

Washington (1984) presented a comprehensive review of the diversity, similarity, and stability of biotic indices as applied to aquatic ecosystems. His conclusion is that the diversity-stability relationship cannot be presented in a simplistic form, and that diversity does not automatically guarantee similarity and/or stability, and vice versa. The EPA has recently developed a comprehensive set of environmental indices (Hunsaker and

Carpenter, 1990). Fish and macroinvertebrate indices are presently under review and development by the Florida Department of Environmental Regulation. Biotic indices are very important, but are not yet available, for pollution assessment of both sediments and ambient water.

2.8 Human Health Hazards from Contaminated Sediment Sites.

Many of the pollutants found in aquatic ecosystems are toxic and can cause acute or chronic health problems to humans through the food chain. It has been documented (Swain, 1983) that even low concentrations of lipid soluble pollutants in the water-sediment system can be magnified, passing through the aquatic food chain into edible fish. A study involving people who consume fish from Lake Michigan (Humphry, 1984) showed that lipid soluble compounds i.e, pesticides, PCBs, dioxins, etc. were detectable in nearly every individual. A comparison between PCB levels of "fisheaters" and "non-fisheaters" is given in Table 2.14.

Table 2.14 PCB levels in persons who consume fish from Lake Michigan (Humphrey, 1987).

Consumer category	Number of participants	Consumption (lb/yr)	Serum PCB (ppb) Range	Serum PCB (ppb) Median
Non-eaters	29	0	< 5-41	15
Occasional eaters	39	0-6	<-5-41	20
Regular eaters	90	> 24	25-366	56

Note: ppb ≡ parts per billion

When reviewing results of organic pollutant levels in humans, short-term effects should be considered very carefully. Indeed, concentration levels of PCBs in humans reach a peak after a meal of contaminated fish and slowly dissipate almost completely during a time period of approximately seven days. The effect of this short-term concentration spike on human health is unknown. Generally, the U.S. Environmental Protection Agency (USEPA) attempts to control human exposure to toxic compounds at a level that will pose lifetime risks of the order of 10^{-7} to 10^{-4}.

The National Academy of Sciences (1983) suggested that studies pertaining to human health risks from different toxic substances should follow four steps:

o hazard identification,
o dose-response assessment,
o exposure assessment, and
o risk characterization.

Hazard identification can be accomplished from epidemiologic studies, case studies and animal studies. For characterization of the hazard, the USEPA uses the letters A to E as follows:

o Group A: Human carcinogen,
o Group B: Probable human carcinogen,
o Group C: Possible human carcinogen,
o Group D: Not classified, and
o Group E: Evidence of noncarcinogenicity.

The dose-response assessment can be determined by using the lifetime risk, LR, defined as:

$$LR = LADD \times PF \quad \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots \quad (2.8)$$

where LADD is the lifetime average daily dose (mg/kg/day), and PF is the potency factor $(mg/kg/day)^{-1}$ (Table 2.15). The lifetime risk shows the probability of getting cancer assuming that an individual consumes the lifetime average daily dose for a 70-year time period.

For humans, the exposure assessment can be defined by means of different exposure pathway models (U.S. E.P.A., 1988). A relatively straightforward approach model involves human consumption of contaminated fish. The equilibrium concentration, C_{fc}, of a contaminant in fish tissue can be estimated as:

$$C_{fc} = C \times BCF \quad \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots \quad (2.9)$$

where C is the contaminant concentration in water, and BCF is the bioconcentration factor (Table 2.16).

The risk characterization is the last and most complicated step which incorporates the findings of the other steps in order to: assess the overall risk of some specific population, to estimate the benefit/cost ratio, and to develop regulatory and management procedures.

The toxicity level of a substance is determined by using a standardized laboratory test called extraction procedure, (EP), which estimates the amount of pollutants that can be mobilized during changes of the physicochemical conditions (Nemerow, 1984) (see also Sections 2.2.1 and 2.2.3). The EP test utilizes a representative sample of waste (minimum size of 100 g). The sample is separated into its component liquid and solid phases. The separation procedure involves a filtration device of nominal pore size of 0.45 μm where the sample is subject to a 75 psi (5.3 kg/m^2) pressure. If the solid residue is less than 0.5% of the sample then that residue can be discarded. Otherwise, the solid phase is placed in an extractor with sixteen times its weight of deionized water for 24 hours at 20 - 40°C. During this time period the pH of the solution is maintained at the level of 5.0 ± 0.2. If the pH rises above 5.2 then acetic acid is added. At the end of the 24 hours the extracted material is separated again into its solid and liquid phases. The liquid phase is then analyzed for the presence of any of the contaminants listed in Table 2.17. Based on the EP toxicity test, a substance is considered EP toxic if it contains one or more of the chemicals listed in Table 2.17 in concentrations that exceed those given in this table (Masters, 1991).

Table 2.15 Potency factor for selected potential carcinogens (Masters, 1991).

Chemical	Group	Potency factor oral route $(mg/kg/day)^{-1}$	Potency factor inhalation route $(mg/kg/day)^{-1}$
Arsenic	A	1.75	50
Benzene	A	2.9×10^{-2}	2.9×10^{-2}
Benzo(a)pyrene	B2	11.5	6.11
Cadmium	B1	-	6.1
Carbon tetrachloride	B2	0.13	-
Chloroform	B2	6.1×10^{-3}	8.1×10^{-2}
Chromium IV	A	-	41
DDT	B2	0.34	-
1,1-Dichloroethylene	C	0.58	1.16
Dieldrin	B2	30	-
Heptachlor	B2	3.4	-
Hexachloroethane	C	1.4×10^{-2}	-
Methylene chloride	B2	7.5×10^{-3}	1.4×10^{-2}
Nickel and compounds	A	-	1.19
PCBs	B2	7.7	-
2,3,7,8-TCDD (dioxin)	B2	1.56×10^{5}	-
Tetrachloroethylene	B2	5.1×10^{-2}	$1.0\text{-}3.3\times10^{-3}$
1,1,1-Trichloroethane (1,1,1-TCA)	D	-	-
Trichloroethylene (TCE)	B2	1.1×10^{-2}	1.3×10^{-2}
Vinyl chloride	A	2.3	0.295

Table 2.16 Bioconcentration factors for selected chemicals (Masters, 1991).

Parameter	BCF (L/kg)	Parameter	BCF (L/kg)
Aldrin	28	Dieldrin	4760
Arsenic & compounds	44	Formaldehyde	0
Benzene	5.2	Heptachlor	15700
Cadmium & compounds	81	Hexachloroethene	87
Carbon tetrachloride	19	Nickel & compounds	47
Chlordane	14000	PCBs	100000
Chloroform	3.75	2,3,7,8-TCDD (dioxin)	5000
Chromium & compounds	16	Tetrachloroethylene	31
Copper	200	1,1,1-Trichloroethane	5.6
DDE	51000	Trichloroethylene	10.6
DDT	54000	Vinyl chloride	1.17
1,1-Dichloroethylene	5.6		

Table 2.17 Maximum allowed concentrations of toxic substances for EP toxicity tests (Masters, 1991).

Toxic substance	EPA hazardous waste number	Max. concentration mg/L
Arsenic	D004	5.00
Barium	D005	100.00
Cadmium	D006	1.00
Chromium	D007	5.00
Lead	D008	5.00
Mercury	D009	0.20
Selenium	D010	1.00
Silver	D011	5.00
Endrin	D012	0.02
Lindane	D013	0.40
Methoxychlor	D014	10.00
Toxaphene	D015	0.50
2,4-D	D016	10.00
2,4,5-TP Silvex	D017	1.00

Jennett et al. (1980) summarized the toxicologic effects of mercury, lead, zinc, copper, and chromium. The toxic characteristics of some of the most common heavy metals found in estuarine sediments are discussed in detail in three reports published by the National Academy of Sciences (N.A.S. 1974, 1977, 1978).

In brief (Jennett et al. 1980), mercury is toxic to humans at levels of 8 ppm as intake and becomes lethal at consumption rates of 75-300 mg/day. Lead is considered toxic for human adults at a concentration of 80 mg per 100 g of blood; a concentration of 0.5-0.8 ppm is the threshold for acute lead poisoning. Zinc is toxic for some animals at levels of approximately 900 ppm. Cadmium is toxic at concentrations of 0.02-2.0 ppm; for humans, consumption of 4 ppm of cadmium is considered toxic. Copper causes toxic effects in biota at concentrations exceeding 1 ppm. Chromium effects on human health are attributed more to deficiency, rather than excess; for household use a level of 0.05 ppm is recommended. Selenium is the only element which is both essential and toxic in biota; concentrations greater than 4 μg/g are considered toxic.

2.8.1 <u>Animal Risks from Contaminated Sediments.</u> Aquatic animals can also be seriously affected by polluted sediments. Fish can suffer diseases such as tumors, fin damage and other anomalies. Pollutants are taken by the fish either from the sediment particulates (in adsorbed form) or from the ambient water (in dissolved form). Besides the effects on individual fish species, aquatic pollution affects the overall richness of the biotic populations in terms of species, ages and sexes. Polluted systems are characterized by skewed trophic structure as evidenced by the presence of more tolerant species, i.e., omnivores, hybrid species, and habitat generalists. Also, older age classes of top predators may be rare. Extreme degradation ecological conditions may lead to complete elimination of all aquatic plants and animals.

2.9 Socio-Economic Ramifications Related to Contaminated Sediments.

Currently, there is strong public awareness of the need for protection, remediation, and enhancement of the environment. Since humans are directly impacted by any scientific and/or technological plan for environmental protection or enhancement, socioeconomic concerns should be addressed as an integral part of plans for ecological management, monitoring, and research. As Desjardins (1987) indicated, the continuous cycle of pollution, industrial production, economics, and politics involves conflicts of power, social structures, lifestyles, and human values. Very often proposals for environmental remediation are in conflict with the NIMBY (not in my back yard) syndrome (Francis, 1987). In spite of the fact that social and economic benefits related to ecological improvement are well known (e.g., health protection, improved aesthetics, recreational facilities, increased property values, etc.) these factors are sometimes hard to quantify (Loucks et al. 1981). Whether or not the socioeconomic benefits can be quantified, there is no doubt that improvement of polluted waters and sediments will have a positive impact on the living standards of the affected communities.

Human health is undoubtedly the most crucial factor in environmental studies. However, improving living standards through technological advances is a human aspiration. It is a combination of needs, aspirations, fears, etc. that drives economic growth and technological progress (with its resulting environmental degradation) against social values for healthy and harmonious co-existence of humans with the natural environment.

A comprehensive socio-economic environmental study should include three major social groups (Francis, 1987):

- federal, state and local authorities responsible for environmental protection and rehabilitation,
- social groups (industrial, agricultural, sportsmen's, etc.) whose activities result in environmental stress, and
- institutional and legislative authorities that can allocate resources for environmental protection and remediation.

Cooperation between these groups can lead to the development of an operational plan that will combine: management consensus among all parties concerned, equitable rights to access and use of the natural resources, and collective self-constraints imposed by environmental criteria.

2.10 Recommendations.

The Lower St. Johns River Basin is a very diverse system with a variety of ecological and land-use characteristics. A broad attack on the problem of sediment contamination would be a very complex and possibly financially prohibitive task. Before pursuing any remedial action, the following steps are recommended:

- Select any highly contaminated area based on consideration of the surficial sediment composition. Since pollutants have an affinity for fine sediments, aquatic bottoms comprised of silt, clay, and organic matter generally have higher levels of contaminant concentration than those comprised of sandy soils. Therefore, in order to estimate contamination effects, a site with high percentages of silt-clay fractions should be selected.
- Identify all of the current and historical point and non-point pollution sources

(domestic, industrial, agricultural) by using GIS, permit information, etc.

o Once the sources have been identified, quantify their relative importance and determine whether each is presently active or inactive. This will include the types of chemicals used and/or produced, volumes of discharges and/or emissions, the degree and type of effluent treatment, etc.

o Survey the site(s) and determine the degree of contamination.

o Decide whether remediation is desirable/necessary based on level of contamination.

o Determine the objectives of the remedial action and consider the probability of successful control of sources.

o If the present (active) pollution sources cannot be controlled, any remedial action will ultimately fail; therefore, the only appropriate action in this case is to monitor and document contamination trends.

o Suggest approaches for control of the active pollution sources; i.e., compliance/ enforcement of EPA/DER regulations, permitting, increased level of effluent treatment, etc.

o If the pollution sources can be controlled, a remedial action plan can be developed according to the guidelines outlined in sections 1.2 to 1.4.

o Along with any remedial action, a monitoring plan should be established for the selected site that will include physical, chemical, and biological sampling and testing. Emphasis should be placed on bio-availability and bio-accumulation tests. The monitoring plan should include sampling of both sediments and ambient water for analysis of nutrients (P and N); toxic metals (As, Cd, Cr, Cu, Hg, Pb, Se, Zn); chlorinated phenols; PAHs; PCBs; chlorinated pesticides; and coprostanol.

o Short-term and long-term, temporal and spatial variations of contaminant concentrations should be documented and analyzed for identification of any pattern or trend of contamination changes.

o Set criteria for evaluating the success of the ecological improvement plan.

CHAPTER III

SEDIMENT QUALITY DATA

G. ENVIRONMENTAL CONDITIONS OF THE LOWER ST. JOHNS RIVER BASIN

3.1 Pollution Sources.

The riverine system of the lower St. Johns River and its tributaries is a very complex system whose pollution levels are extremely hard to describe. The St. Johns River is a "black water" estuary; i.e., the water has a dark brownish color due to high loads of tannin from decomposing forest leaves and suspended solid particulates. Because of the resulting reduction in light penetration, aquatic plants grow only in the shallows, and under natural conditions the system is a net consumer of oxygen.

The Lower St. Johns River Basin (LSJRB) system is the receiver of a variety of pollutants from point and non-point pollution sources. A number of these sources are discharging into the tributaries, where weak circulation patterns prevent efficient flushing of the pollutants. There are approximately 360 domestic and 49 industrial permitted sources (Table 3.1) discharging approximately 2009 million gallons per day (MGD) into the surface waters of the Lower St. Johns River Basin. The majority of these sources are located in the Jacksonville metropolitan area and discharge either directly into the main river or in some of its tributaries; e.g., Cedar River, Ribault River, Moncrief Creek, etc.

Table 3.1 Lower St. Johns River wastewater discharging facilities (SJRWMD, 1991).

Source	Type	Receiving Stream	Design Flow (MGD)
ARLINGTON RIVER SUB-BASIN			
Aetna Insurance	D	Arlington River	0.005
Fla. HRS (Arlington Exp.)	D	Arlington River	0.100
River Landing Club	D	Arlington River	0.040
Wendy's (University Blvd.)	D	Arlington River	0.007
Empress Garden Apartments	D	Pottsburg Creek	0.010
Expressway Motor Inn	D	Pottsburg Creek	0.003
Fleetwood MHP	D	Pottsburg Creek	0.015
Hogan Spring Glenn #64	D	Pottsburg Creek	0.015
Joe Motel	D	Pottsburg Creek	0.006
McGehee Realty, TV 47	D	Pottsburg Creek	0.002
Pineland Gardens	D	Pottsburg Creek	0.005
Royal Lakes STP	D	Pottsburg Creek	2.400
Shady Lane Trailer Park	D	Pottsburg Creek	0.010
Solar Office Building	D	Pottsburg Creek	0.078
SS Estates Elem. #75	D	Pottsburg Creek	0.010
State Farm Claims	D	Pottsburg Creek	0.002
Uncle Johns Pancake House	D	Pottsburg Creek	0.010
Villas Apartments	D	Pottsburg Creek	0.035
Weight Watchers	D	Pottsburg Creek	0.015
Dan's Sandwich Shop	D	Big Pottsburg Creek	0.003
Lil Champ Shopping	D	Big Pottsburg Creek	0.075
Publix Supermarket (Atlantic)	D	Big Pottsburg Creek	0.007
Royal Point Apartments	D	Big Pottsburg Creek	0.050
Secret Cove	D	Big Pottsburg Creek	0.125
Unilantic	D	Big Pottsburg Creek	0.005
Atlantic Oaks Apartments	D	Little Pottsburg Creek	0.025
Beach Blvd. Shopping Center	D	Little Pottsburg Creek	0.025
Cajeco, Inc.	D	Little Pottsburg Creek	0.003
Clearview Townhouse Apartments	D	Little Pottsburg Creek	0.025
Colonial Point Apartments	D	Little Pottsburg Creek	0.100
Demetree Builders	D	Little Pottsburg Creek	0.010
Emerson Assoc.	D	Little Pottsburg Creek	0.010
Harold House Apartments	D	Little Pottsburg Creek	0.025
Jax Liquor (Beach Blvd.)	D	Little Pottsburg Creek	0.005
Love Grove Elem. #82	D	Little Pottsburg Creek	0.008
Nemours Children's Hospital	D	Little Pottsburg Creek	0.020
Paradise Inn	D	Little Pottsburg Creek	0.003
Pottsburg Utilities, Inc.	D	Little Pottsburg Creek	0.150
Suntree MHP	D	Little Pottsburg Creek	0.030
Hull Chevrolet	D	Strawberry Creek	0.002
Mill Creek Manor Apartments	D	Strawberry Creek	0.007
Oaks Sewage Complex	D	Strawberry Creek	0.220
Rivermont Apartments	D	Strawberry Creek	0.050
Cimmaron Apartments	D	Strawberry Creek	0.025
Famous Amos (#1)	D	Silversmith Creek	0.005
Lawrence Pharmaceuticals	D	Bennett's Creek	0.005
		TOTAL	3.786
BLACK CREEK SUB-BASIN			
Camp Blanding	D	South Fork Black Creek	0.900
Grand Olde Shopping Center	D	South Fork Black Creek	0.020
Penney Farms Retirement Center	D	South Fork Black Creek	0.060
Clay Utilities	D	Little Black Creek	1.000
Doctor's Inlet Elem. School	D	Little Black Creek	0.0168
USN NAS Cecil Field	D	Rowell Creek	0.800
Schmitt Trailer Park	D	Taylor Creek	0.003
Dupont-Highland Plant	I	Boggy Branch	10.000
		TOTAL	12.7998

Continued:

Table 3.1 Lower St. Johns River wastewater discharging facilities (SJRWMD, 1991). (Continued:)

Source	Type	Receiving Stream		Design Flow (MGD)
BROWARD RIVER SUB-BASIN				
Oceanway Shopping Center	D	Broward River		0.014
Penny Pinchers Truck Stop	D	Broward River		0.007
Port Carriers, Inc.	D	Broward River		0.002
Simplex Industries	I	Broward River		0.003
Anheuser-Busch Turf Farm	I	Cedar Creek		1.000
CSX Transportation, West	I	Cedar Creek		0.200
Cleaners Hanger Company	I	Cedar Creek		0.008
Delta Airlines	I	Cedar Creek		0.006
East Coast Oil, Inc.	I	Cedar Creek		1.200
Airport Motor Inn	D	Little Cedar Creek		0.010
Oceanway Elem. #61 & #62	D	Little Cedar Creek		0.017
Ortega Airport System	D	Little Cedar Creek		0.175
Red Carpet Inn	D	Little Cedar Creek		0.030
JPA International Airport	D	Pickett Branch		0.500
			TOTAL	3.172
CRESCENT LAKE SUB-BASIN				
Crescent City Jr/Sr High	D	Crescent Lake		0.0015
City of Crescent City STP	D	Crescent Lake		0.250
City of Bunnel	D	Black Branch		0.300
St. Johns Riverside Estates	D	Dunn's Creek		0.005
			TOTAL	0.5565
DUNN CREEK SUB-BASIN				
Jax City Plt-Oceanway Manor	D	Dunn Creek		0.700
Sheffield Elementary	D	Caney Branch		0.010
			TOTAL	0.710
ETONIA/RICE CREEK SUB-BASIN				
Georgia Pacific Corp.	I	Rice Creek		50.000
			TOTAL	50.000
JULINGTON CREEK SUB-BASIN				
Baywood MHP	D	Julington Creek		0.080
Baywood Mobile Home Park	D	Julington Creek		0.080
Gateway Community Service	D	Julington Creek		0.0075
Mandarin Marina	D	Julington Creek		0.005
Mandarin Marina Seafood	D	Julington Creek		0.0033
Wesley Manor Retirement Village	D	Julington Creek		0.100
Bayard Raceways, Inc.	D	Durbin Creek		0.0135
Amoco Service Station	D	Sampson Creek		0.002
Exxon	D	Sampson Creek		0.0024
G & M Truck Plaza	D	Sampson Creek		0.0188
St. Aug./Jax South KOA	D	Sampson Creek		0.017
Stuckey's Pecan Shop	D	Sampson Creek		0.012
Lorreto Elem. School #30	D	Oldfield Creek		0.010
Sherrod Vans	D	Sweetwater Branch		0.005
Julington Creek School	D	Floral Branch		0.0075
			TOTAL	0.364
ORTEGA RIVER SUB-BASIN				
Airbase MHP	D	Ortega River		0.006
Americana MHP	D	Ortega River		0.010
Azalea MHP	D	Ortega River		0.025
Blanding MHP	D	Ortega River		0.003
Country Roads MHP	D	Ortega River		0.060
GT Properties	D	Ortega River		0.015
Justiss Trailer Park	D	Ortega River		0.015
Lake Forest Jr. High School	D	Ortega River		0.040
Malibu Garden Apartments	D	Ortega River		0.020
Ortega Forest S/D	D	Ortega River		0.150

Continued:

Table 3.1 Lower St. Johns River wastewater discharging facilities (SJRWMD, 1991). (Continued:)

Source	Type	Receiving Stream	Design Flow (MGD)
Ortega Hills STP	D	Ortega River	0.250
Riverside Village Shop. Center	D	Ortega River	0.010
Senior Citizen's MH Terrace	D	Ortega River	0.006
Stockton Elementary #88	D	Ortega river	0.008
Timuquana Elementary #98	D	Ortega River	0.009
Timuquana Village Condo.	D	Ortega River	0.016
Villa del Rio & Ortega	D	Ortega River	0.150
Westside MHP #1	D	Ortega River	0.001
Westside MHP #2	D	Ortega River	0.015
AT&T	D	Cedar River	0.003
All American Hot Dog	D	Cedar River	0.003
Baron's Amoco	D	Cedar River	0.002
Brown's Shopping Center	D	Cedar River	0.012
Buffalo Tank Company	D	Cedar River	0.002
Cedar Shores Apts. STP	D	Cedar River	0.035
Cross Creek Apartments	D	Cedar River	0.070
CSX Transportation (West End)	I	Cedar River	0.048
Denny Moran's Restaurant	D	Cedar River	0.007
Doctor's Office Building	D	Cedar River	0.005
Duplex Products, Inc.	D	Cedar River	0.002
Dura-Bond Protective Coat	D	Cedar River	0.005
Emerik Properties	D	Cedar River	0.003
Florida Wire and Cable Company	I	Cedar River	0.072
Florida Wire and Cable Company	D	Cedar River	0.002
Ford Motor Co. Parts Center	D	Cedar River	0.007
Four Seasons MHP	D	Cedar River	0.015
Hart Haven MHP	D	Cedar River	0.005
HKH Industries	D	Cedar River	0.003
Hyde Park Elem. School #77	D	Cedar River	0.008
JEA Westside Service Center	D	Cedar River	0.002
Kelly's MHP	D	Cedar River	0.009
Mike and Frank's Trailer Park	D	Cedar River	0.003
Murray Hill Lumber	D	Cedar River	0.007
Pat and Mike's Restaurant	D	Cedar River	0.007
Paxon Prof. Center	D	Cedar River	0.002
Ramona Elementary #79	D	Cedar River	0.009
Reichhold Chemicals, Inc.	I	Cedar River	0.084
Reichhold Chemicals, Inc.	D	Cedar River	0.010
River City Chrysler/Plymouth	D	Cedar River	0.002
Simmons Company	D	Cedar River	0.019
Sudler STP	D	Cedar River	0.003
Tara Manor Town Homes	D	Cedar River	0.002
U-Haul International of Fla.	D	Cedar River	0.005
US Bulk Post Office	D	Cedar River	0.050
United Parcel Service	D	Cedar River	0.004
Westside Dodge	D	Cedar River	0.003
Blair Road Apartments	D	McGirt's Creek	0.006
Bonanza Terrace MHP	D	McGirt's Creek	0.006
Colony MHP	D	McGirt's Creek	0.060
Duclay MHP	D	McGirt's Creek	0.003
DOT Area (eastbound)	D	McGirt's Creek	0.012
Holiday Estates MHP	D	McGirt's Creek	0.025
Lake Forest MHP	D	McGirt's Creek	0.060
Napoli Trailer Park	D	McGirt's Creek	0.015
Normandy Pines MHP	D	McGirt's Creek	0.012
Ortega-Blanding STP	D	McGirt's Creek	0.350
Parkwest MHP	D	McGirt's Creek	0.040
Taylor's MHP	D	McGirt's Creek	0.006
West Meadows MHP	D	McGirt's Creek	0.010
Whitehouse Elementary	D	McGirt's Creek	0.070
Jacksonville Heights STP	D	Fishing Creek	2.500

Continued:

Table 3.1 Lower St. Johns River wastewater discharging facilities (SJRWMD, 1991). (Continued:)

Source	Type	Receiving Stream	Design Flow (MGD)
Sacred Heart Church	D	Fishing Creek	0.008
Sault Plaza	D	Fishing Creek	0.002
Shops of Timuquana	D	Fishing Creek	0.010
Thunderbird Trailer Park	D	Fishing Creek	0.015
Wares MHP #1	D	Fishing Creek	0.015
Wares MHP #2	D	Fishing Creek	0.015
Westconnett Elementary #57	D	Fishing Creek	0.012
Wil-Mar Apartments	D	Fishing Creek	0.004
Hyde Grove S/D	D	Wills Branch	0.170
Londontowne Apartments	D	Wills Branch	0.150
Munchie Court Apartments	D	Wills Branch	0.007
Normandy Estates	D	Wills Branch	0.100
Normandy Village	D	Wills Branch	0.400
Ortega Utility (Herlong Syst.)	D	Wills Branch	0.065
Working Man's Friend	D	Wills Branch	0.002
Biltmore Elem. School #78	D	Little Sixmile Creek	0.007
Nimnicht Chevrolet	D	Big Fishweir Creek	0.002
		TOTAL	5.498
SIXMILE CREEK SUB-BASIN			
Days Inn	D	Six Mile Creek	0.025
Denny's Restaurant	D	Six Mile Creek	0.020
Red Carpet Inn/Candyland	D	Six Mile Creek	0.005
Mill Creek School	D	Mill Creek	0.004
		TOTAL	0.054
ST. JOHNS RIVER SUB-BASIN			
Amerada Hess Corporation	I	St. Johns River	0.00002
Amoco Oil Co.	I	St. Johns River	0.0024
Arlington STP	D	St. Johns River	10.000
Beakon Hills STP	D	St. Johns River	0.836
Boys Home Association	D	St. Johns River	0.013
Briarwood STP	D	St. Johns River	0.780
Browning Pierce Elem. #1	D	St. Johns River	0.005
Browning Pierce Elem. #2	D	St. Johns River	0.0043
Buckman Street STP	D	St. Johns River	52.500
Carriage House Apartments	D	St. Johns River	0.025
City of Green Cove Springs	D	St. Johns River	0.500
City of Palatka	D	St. Johns River	3.000
District 2 STP (City of Jax)	D	St. Johns River	10.000
Fleming Island System STP	D	St. Johns River	0.230
Fort Caroline Club Estates	D	St. Johns River	0.012
FPL Palatka Plant	I	St. Johns River	125.000
Jacksonville University	D	St. Johns River	0.200
Jax City Plt-Fort Caroline	D	St. Johns River	
Jax Episcopical High School	D	St. Johns River	0.020
JEA Kennedy, G.S. (Chem.)	I	St. Johns River	0.090
JEA Northside (Thermal)	I	St. Johns River	1246.000
JEA Southside (Thermal)	I	St. Johns River	358.100
JEA Southside (Emerson)	D	St. Johns River	0.003
Jefferson Smurfit Corp.	I	St. Johns River	6.000
Johns-Manville Mfg. Corp.	D	St. Johns River	0.005
JPA Blount Island (New)	I	St. Johns River	
JPA Blount Island (Old)	I	St. Johns River	
JPA Blount Island	D	St. Johns River	0.030
JPA Joysery (Talleyrand)	D	St. Johns River	0.010
JPA Talleyrand	I	St. Johns River	
Lakewood Apartments	D	St. Johns River	0.025
Magnolia Springs Apartments	D	St. Johns River	0.078
Mandarin STP	D	St. Johns River	3.250
Marine Science Center	D	St. Johns River	0.003
Miller Street STP	D	St. Johns River	4.000

Continued:

Table 3.1 Lower St. Johns River wastewater discharging facilities (SJRWMD, 1991). (Continued:)

Source	Type	Receiving Stream	Design Flow (MGD)
Miramar Shopping Center	D	St. Johns River	0.011
Monterey STP	D	St. Johns River	3.000
Naval Supply Center	I	St. Johns River	
Oak Lane Building	D	St. Johns River	0.002
OXCE Fuel Co. (COM Energy)	I	St. Johns River	0.0106
Parson's Restaurant	D	St. Johns River	0.0750
Petroleum Fuel and Terminal Co.	I	St. Johns River	
Phillips Petroleum	I	St. Johns River	
Port Buena Vista MHP	D	St. Johns River	0.015
Reynolds Industrial Park	I	St. Johns River	0.068
Riverside Plaza	I	St. Johns River	
San Jose S/D	D	St. Johns River	2.250
San Jose Shores	D	St. Johns River	0.028
Seminole Kraft Paper Co.	I	St. Johns River	63.000
Southwest District	D	St. Johns River	5.000
St. Johns Campground	D	St. Johns River	0.005
Strickland's Restaurant	D	St. Johns River	0.030
The Point Townhouses	D	St. Johns River	0.015
Town of Hastings	D	St. Johns River	0.100
Town of Orange Park	D	St. Johns River	2.500
University Park	D	St. Johns River	0.750
US Coast Guard	D	St. Johns River	0.005
USN Mayport Naval Station	I	St. Johns River	0.200
USN Mayport Naval Station	D	St. Johns River	1.800
USN NAS Mainside	D	St. Johns River	3.000
Witco Chemical	I	St. Johns River	0.075
CSX Transportation	I	McCoy's Creek	0.200
CSX Transportation, East End	I	McCoy's Creek	0.200
Beauclerc Bay Apartments	D	Goodby's Lake	0.020
Crazy Horse Saloon	D	Goodby's Lake	0.007
Mac Paper Converters	D	Goodby's Lake	0.002
Woodlake Apartments	D	Goodby's Lake	0.070
Bob's Super Sandwich Shop	D	Goodby's Creek	0.002
Ring Power Corporation	I	Goodby's Creek	0.001
Beach Road Chicken Dinners	D	Miller's Creek	0.005
Burger King #3 Beach Blvd.	D	Miller's Creek	0.004
Art Museum Garden Apartments	D	Miller's Cove	0.350
Camelot Village Apartments, STP	D	Christopher Creek	0.025
Derby House Restaurant	D	Christopher Creek	0.002
McDonalds (Old St. Augustine Rd.)	D	Christopher Creek	0.003
Mercury Luggage	D	Christopher Creek	0.007
Ponce De Leon Trailer Park	D	Jones Creek	0.015
Regency Plaza Shopping Center	D	Jones Creek	0.010
Regency Village Apartments	D	Jones Creek	0.100
Brookview Trailer Park	D	Ginhouse Creek	0.003
Lone Star Elementary #233	D	Ginhouse Creek	0.015
Duval Motors	D	Fishweir Creek	0.002
JEA Northside Domestic	D	San Carlos Creek	0.005
Ortega Executive Center	D	Pirate's Cove	0.015
Pine Forest Elementary #159	D	New Rose Creek	0.010
San Hose Elementary #83	D	New Rose Creek	0.008
Woodmere	D	Fairchild Branch	0.500
Holly Oaks	D	Cowhead Creek	1.000
Associated Minerals (USA) Inc.	I	Clark's Creek	8.000
Putnam Correctional Institute	D	Mill Branch	0.090
			TOTAL1913.332
INTRACOASTAL WATERWAY			
Bellinger Ship Yards	D	ICWW	0.020
City of Atlantic Beach	D	ICWW	2.000
Jacksonville Beach, STP	D	ICWW	3.000

Continued:

Table 3.1 Lower St. Johns River wastewater discharging facilities (SJRWMD, 1991). (Continued:)

Source	Type	Receiving Stream	Design Flow (MGD)
V-Edge Trailer Park	D	ICWW	0.005
College Square Apartments	D	Mill Dam Branch	0.150
Countryside Village of Jax	D	Mill Dam Branch	0.150
Zayre Department Store	D	Mill Dam Branch	0.020
Windy Hill Elementary	D	Mill Dam Creek	0.007
Buccaneer Services STP	D	Sherman Creek	0.400
Mayport MHP	D	Sherman Creek	0.007
Oak Harbor	D	Sherman Creek	0.192
Wonderwood Villas Homes	D	Sherman Creek	0.060
Jax County Day School	D	Pablo Creek	0.015
Isle of Palms	D	Pablo Creek	0.500
Neptune Beach	D	Hopkins Creek	0.750
Holiday Harbor	D	Hogpen Creek	1.250
Deerwood #1	D	Runcheon Branch	0.300
		TOTAL	8.826
TROUT RIVER SUB-BASIN			
Days Inn (Broward Road)	D	Trout River	0.040
Dinsmore Elementary School	D	Trout River	0.007
J. Strauss Utility	D	Trout River	0.253
Lake Forest Elementary #74	D	Trout River	0.010
Milligan's (Lem Turner)	D	Trout River	0.050
Northtown Square	D	Trout River	0.010
River Park Apartments	D	Trout River	0.035
Silver Dolphin Trailer Park	D	Trout River	0.006
Burger King #11 Lem Turner	D	Ribault River	0.002
Centurian Truck Plaza	D	Ribault River	0.020
Deluxe Strip Center	D	Ribault River	0.005
Famous Amos #3	D	Ribault River	0.004
George Washington Carver Elem.	D	Ribault River	0.011
Harborview	D	Ribault River	0.210
Henry Kite, School #37	D	Ribault River	0.012
Ideal Trailer Park	D	Ribault River	0.011
Krystal #9	D	Ribault River	0.002
Manna Provision Company	I	Ribault River	0.075
Monterey Hotel	D	Ribault River	0.015
Red Carpet Laundry and Cleaner	I	Ribault River	0.008
Riverview Baptist Church	D	Ribault River	0.002
Captain D's (Lem Turner)	D	Moncrief Creek	0.010
Janlotte, Inc.	I	Moncrief Creek	0.014
Lake Forest Garden Apartments	D	Moncrief Creek	0.007
Northwestern Jr. High School #155	D	Moncrief Creek	0.030
Save-A-Stop, Inc.	D	Moncrief Creek	0.010
SCM Organic Chemicals	I	Moncrief Creek	5.000
Briarwood Estates MHP	D	Nine Mile Creek	0.006
Mandalay Enterprises	I	Nine Mile Creek	0.009
Pinewood MHP	D	Nine Mile Creek	0.005
Shady Oaks MHP	D	Nine Mile Creek	0.015
Spacious Trailer Park	D	Nine Mile Creek	0.005
Trout River MHP	D	Nine Mile Creek	0.012
Jax Industries ZXT	D	Sixmile Creek	0.003
Picketville Elementary #205	D	Sixmile Creek	0.008
Ricky Villa Apartments	D	Sixmile Creek	0.020
Produce Terminal, Jax, Fla.	D	Little Sixmile Creek	0.004
Reynolds Lane Elementary #521	D	Little Sixmile Creek	0.009
Union Camp	I	Little Sixmile Creek	4.000
Pine Estates Elementary #250	D	Blockhouse Creek	0.040
Victory Christian Academy	D	Blockhouse Creek	0.010
		TOTAL	10.005

A major industrial complex within the lower St. Johns River basin is the Georgia-Pacific paper plant which discharges into Rice Creek, near Palatka. Of these effluents, 95.23% is released into the St. Johns River sub-basin while the remaining 4.77% is discharged into the other sub-basins as follows: Etonia/Rice Creek - 2.49%, Black Creek - 0.64%, Trout River - 0.50%, Intracoastal Waterway - 0.44%, Ortega River - 0.27%, Arlington River - 0.19%, Broward River - 0.16%, Dunn Creek - 0.04%, Crescent Lake - 0.03%, Julington Creek -0.02%, and Sixmile Creek 0.003%. Besides the open water effluent discharging, there are approximately 73 point sources discharging into the groundwater. These sources release 1.395 MGD (million gallons per day); i.e., 0.07% of the total amount of point source discharges (Table 3.2). It is believed, however, that the point sources account for only 10% of the pollution, with 90% attributed to non-point sources (SJRWMD, 1991). The non-point pollution is related mainly to urban growth and agricultural activities.

In spite of the removal of raw sewage outfalls and untreated paper mill discharges by 1978, the water quality in metropolitan Jacksonville is still poor (Wentzel and McVety, 1986). Violations of dissolved oxygen and coliform standards were often observed during the 1980's (City of Jacksonville, 1987). The FDER's 1990 305(b) report reiterated the fact that the main stem of the river near downtown Jacksonville is of poor water quality. In the same report, water quality was listed as "poor" in the Ribault River, Moncrief Creek, Wills Branch, Ortega River, Cedar River, Strawberry Creek, Doctors Lake, and the Rice Creek/Etonia Creek system. The rest of the riverine system was found to be in either fair or good condition. An earlier report by FDER (1975) suggested that the majority of tributaries in Duval County suffer from low

Table 3.2 Groundwater discharging facilities (SJRWMD, 1991).

Source	Type (MGD)	Design Flow	Source	Type	Design Flow (MGD)
Allied Corporation	I		Ambulatory Surgical Center	D	0.005
Anheuser Busch, Inc., Lannie	I		Anheuser Busch, Inc., Old	I	
Anheuser Busch, Inc., Spray	I		Arbor Pointe STP	D	0.070
Arlington Executive Park, Inc.	D	0.010	Bowl America, Inc.	D	0.010
BP Oil (Gulf Oil)	I		Cardinal Village MHP	D	0.004
Caroline Freight Carriers Co.	D	0.019	Celotex Corporation	D	0.005
Chimney Lakes STP	D	0.150	Circle Eleven MHP	D	0.010
Coggin Pontiac, Inc.	D	0.002	Coles MHP (overflow)	D	0.012
Colonial Village Apartments	D	0.030	Connie Jean MHP	D	0.015
Deluxe Check Printers	D	0.003	Dixie Oil	D	0.002
East Coast Oil, Inc.	I		Electromate Corporation	D	0.005
Ferrell's Elbow Room	D	0.005	Fla. Dept. of Correction	D	0.200
Fla. DOT Rest Area (W-bound)	D	0.020	Fla. HRS, Youth Center	D	0.005
Fla. HRS, Hodges Clusterhome	D	0.006	Floral Bluff Apts.	D	0.002
G&B of Jax., Inc.	D	0.002	Carden City Elementary	D	0.008
Genell's Hair Salon #1D-Geo	D	0.002	Genell's #2 Office	D	0.002
Golden Nugget Trailer Park	D	0.002	Hall's Trailer Park	D	0.003
Hanna Park	D	0.080	Hooter's	D	0.005
Jax. Assembly of God	D	0.025	JEA Northside (Chem.)	I	
JEA Southside G.S. (Chem.)	I		Krystal #11	D	0.003
Krystal #12	D	0.003	Lincoln Villas Comm. Center	D	0.005
Mardi Gras	D	0.010	Masters Apartments	D	0.002
Morse Avenue Elem. #116	D	0.007	Mustang MHP	D	0.003
National Merchandise Company	D	0.010	Northwest Jax Medical Center	D	0.007
Oakes Mobile Home Court	D	0.008	Owen Steel Co. of Fla.	D	0.002
Pappy's Wharf	D	0.012	Paradise Village MHP	D	0.050
Pine Villa Trailer Park	D	0.015	Pleasure Palace	D	0.005
Publix Superm. (W. Beaver)	D	0.006	Quincy's	D	0.008
Regency Dodge	D	0.003	Regency Trailer Park	D	0.006
Roux Laboratories Inc.	D	0.003	Salaam Club of Florida, Inc.	D	0.003
South Atlantic Utilities	D	0.080	Southside Nursing Center	D	0.015
Southwest Villas	D	0.090	Springtree Village	D	0.200
Stepp Avenue Complex	D	0.003	Sunny Acres MHP	D	0.015
Thorn's Country Homes	D	0.012	Town and Country	D	0.008
Turner Jones Specialty	D	0.005	University Lakes Apartments	D	0.075
Upper Florida Milk Producers	I		USN Mayport Naval Station	I	
Ward MHP #2	D	0.002			
		TOTAL 1.395			

dissolved oxygen, odor, high coliform count, and high nutrient concentrations.

Industrial activities in Duval County (e.g., Cedar River) have resulted in high accumulations of heavy metals and organic priority pollutants, particularly in sediments (St. Johns River Water Management District, 1991). Disposal and storage of hazardous industrial and military wastes increased the potential of surface water contamination and have had a significant impact on various sections of the LSJRB. Under the

Comprehensive Environmental Response, Compensation, and Liability Act of 1980, eight sites in the Lower St. Johns River Basin have been targeted for clean-up by the EPA's Superfund Program. Seven of these sites are in Duval County, and one is in Volusia County (Table 3.3). In addition to the Superfund sites, as of September 30, 1989, nine military installations in Duval and Putnam Counties, with a total of 106 contaminated sites, have been identified for remediation by the Defense Cleanup Programs (Table 3.4).

Table 3.3 EPA's Superfund cleanup sites in the LSJRB (U.S. Army C.O.E. 1986)

Site	County	NPL	Date	Status
Cecil Field NAS	Duval	Prop.	07/14/89	Site studies
Coleman-Evans Wood Preserving	Duval	Final	09/01/83	Remedy design
Hipps Road Landfill	Duval	Final	09/01/84	Cleanup ongoing
Jacksonville NAS	Duval	Final	11/21/89	Site studies
Pickettville Road Landfill	Duval	Final	09/01/83	Site studies
Sherwood Medical	Volusia	Prop.	09/01/83	Site studies
Whitehouse Waste Oil Pits	Duval	Final	09/01/83	Remedy design
Yellow Water Road Dump	Duval	Final	06/01/86	Site studies

Table 3.4 Department of Defense cleanup sites in the LSJRB (U.S. Army C.O.E. 1986)

	Site	County	Sites identified
ARMY:	Jax (Burpee) Army Reserve Center	Duval	3
	Jax (Milam) Army Reserve Center	Duval	5
	Jax (Phillips) Army Reserve Center	Duval	1
	Palatka Army Reserve Center	Putnam	8
	Palatka Army Reserve Center	Putnam	1
NAVY:	Cecil Field NAS	Duval	19
	Jacksonville NAS	Duval	45
	Mayport NAS	Duval	16
AIR FORCE:	Jax. Air National Guard	Duval	8

		TOTAL	106

To assess the environmental conditions of the Lower St. Johns River Basin, a number of water and sediment quality data have been collected in the past. The data involves analyses of water-sediment physicochemical characteristics and estimations of nutrients, heavy metals, and organic pollutants.

3.2 Sediment Quality Studies.

Although data describing specific levels of sediment chemical contamination are very limited, it is apparent that anthropogenic pollutants have accumulated in the surficial sediments in the LSJRB. The most comprehensive studies on sediment quality in the Lower St. Johns River Basin are the following six:

- o Dames and Moore Maintenance Dredging Study (Dames and Moore, 1983),
- o Mote Marine Laboratory Study (Pierce et al. 1988),
- o Jacksonville Port Authority Study (Savannah Lab. & Environmental Services, Inc. 1988a),
- o Coastal Zone Management, FDER, Study (Savannah Lab. & Environmental Services, Inc. 1988b),
- o University of Florida Study (Delfino et al. 1991), and
- o Bio-Environmental Services Division Laboratory and SJRWMD (1989-91).

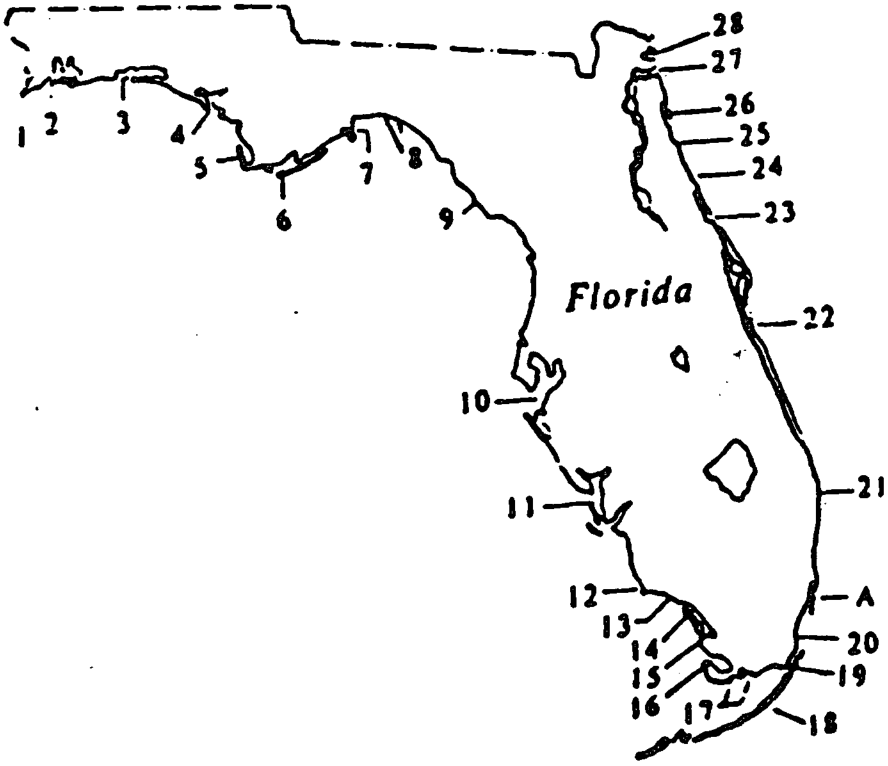
Within a two year period, the Mote Marine Laboratory and the FDER studies analyzed data from over 120 sites throughout the lower basin. The data included physical characteristics, nutrients, heavy metals, and organic priority pollutants. Whenever possible, the data were compared with existing standard limits for determination of the pollution level. Metal concentrations were compared using aluminum as a reference element.

3.2.1 Heavy Metals. Heavy metals were found in the bottom sediments primarily in the fresh/salt water mixing zone (Pierce et al. 1988; Savannah Lab. & Environmental Services, Inc. 1988b); i.e., from the Arlington River to Julington Creek. This metal accumulation is partially attributed to the enhanced flocculation of suspended particles that occurs in the mixing zone (Keller and Schell, 1993). The sediment quality studies involved analysis for aluminum (Al), silver (Ag), arsenic (As), cadmium (Cd), chromium (Cr), copper (Cu), iron (Fe), mercury (Hg), nickel (Ni), lead (Pb), selenium (Se), antimony (Sb), and zinc (Zn).

Since metal concentrations in natural sediments vary by orders of magnitude, normalization of metal concentrations using aluminum as a reference element has been suggested (Schropp et al. 1990) for effective interpretation of anthropogenic metal enrichment. Metal concentrations in sediments are usually much higher than in the ambient water (Trefry et al. 1983); therefore, metal analysis of sediment samples is more reliable and involves fewer contamination problems.

To account for the regional natural metal variability and to establish metal:aluminum ratios, from 1985 to 1987 FDER collected and statistically analyzed samples from 103 stations representing 28 estuaries in the State of Florida. The sites of these estuaries are given in Figure 3.1. The samples were analyzed for the following metals: Al, As, Cd, Cr, Cu, Hg, Ni, Pb, and Zn.

3.2.1.1 Metal Analyses. For determination of all metal concentrations except mercury, sediments were dried at 80°C, vigorously mixed, and 0.3 to 0.5 g was placed and weighed in a 100-ml polytetrafluroethylene vial. After adding 5 ml of HF and 10



Clean sediment sampling locations: 1) Perdido Bay, 2) Santa Rosa Sound, 3) Choctawhatchee Bay, 4) St. Andrew Bay, 5) St. Joe Bay, 6) Apalachicola Bay, 7) Ochlockonee Bay, 8)Apalachee Bay, 9) Suwannee River, 10) Lower Tampa Bay, 11) Charlotte Harbor, 12) Rookery Bay 13) Gullivan's Bay, 14) Chokoloskee Bay, 15) Ponce de Leon Bay, 16) Whitewater Bay, 17) Florida Bay, 18) Key Largo, 19) Barnes Sound, 20) Card Sound, 21) Loxahatchee River 22) Indian River Lagoon, 23) Mosquito Lagoon, 24) Halifax River, 25) Matanzas River, 26) Guana River, 27) Fort George River, 28) Nassau River.

Figure 3.1 Clean sediment sampling locations (Schropp et al. 1990).

ml concentrated HNO_3, the vials were capped. The samples were then digested by refluxing at 100°C for 48 hours. After digestion, the samples were dried and subsequently dissolved in 1 ml concentrated HNO_3 and 9 ml double distilled, de-ionized water. Mercury was analyzed by digesting the sediment samples with H_2SO_4 and HNO_3 in a water bath at 60°C, and then oxidizing the product with potassium permanganate (Plumb, 1981).

Metal concentrations were detected using flame atomic absorption spectroscopy (AAS) for Al and Zn; flameless AAS using a graphite furnace for Cd, Cr, Cu, Ni, and Pb; and flameless AAS for As (hydride) and Hg (cold vapor). The AAS methods were applied according to the methods described by the American Public Health Association (1985). For Quality Control/Quality Assurance (QC/QA), 10% of all samples was subjected to duplicate analyses and spikes. With each batch of sediment samples, a test was performed using Estuarine Sediment Standard Reference Material 1646 of the National Bureau of Standards. Metal recovery from the reference materials ranged from 94 to 105% with a relative standard deviation (RSD) of 2 - 7%. The only exception was As with recovery of 90% and RSD of 8.5% (Schropp et al. 1990).

3.2.1.2 Metal:Aluminum Ratio. Once the metal concentrations were established, the metal:aluminum ratios were calculated and then analyzed statistically. Aluminum concentrations ranged from 47 to 79,000 parts per million (ppm). Concentrations of other metals were one to two orders of magnitude less than aluminum. The entire analysis was based on the assumption that, for natural sediments, data for metal:aluminum ratios should be normally distributed and have constant variance. Applying log transformation, the plots of means versus standard deviation justified the constant variance assumption. Data points from reference locations that deviated from the normal distribution were removed under the hypothesis that the deviations were the result of metal enrichment by anthropogenic activities. Using simple regression analysis, the upper and lower 95% prediction limits for metal:aluminum ratios were estimated. A linear relationship between metals and aluminum was established according to the equation

$$\log(\mathrm{Metal}_{ppm}) = a + b[\log(\mathrm{Al}_{ppm}) \tag{3.1}$$

with the coefficients a and b given in Table 3.5. Since mercury did not correlate with aluminum ($r < 2.0$), this metal was excluded from further analysis. The metal:aluminum ratio data and their resulting linear correlations are presented in Figures 3.2a,b.

Table 3.5 Linear log-log relation coefficients for metals/aluminum (Schropp et al. 1990).

Metal		No. of Samples	Slope (b)	Intercept (a)
Arsenic	(As)	98	0.63	-1.8
Cadmium	(Cd)	102	0.29	-2.2
Chromium	(Cr)	102	0.55	-1.1
Copper	(Cu)	102	0.48	-1.2
Mercury	(Hg)	89	-0.10	-1.2
Nickel	(Ni)	72	0.40	-0.8
Lead	(Pb)	93	0.73	-2.1
Zinc	(Zn)	99	0.71	-1.8

3.2.2 Organic Pollutants. Intensive industrial, agricultural, and urban activities during the last four decades, combined with the existence of sediments containing high amounts of organic carbon in some areas of the Lower St. Johns River Basin, resulted in

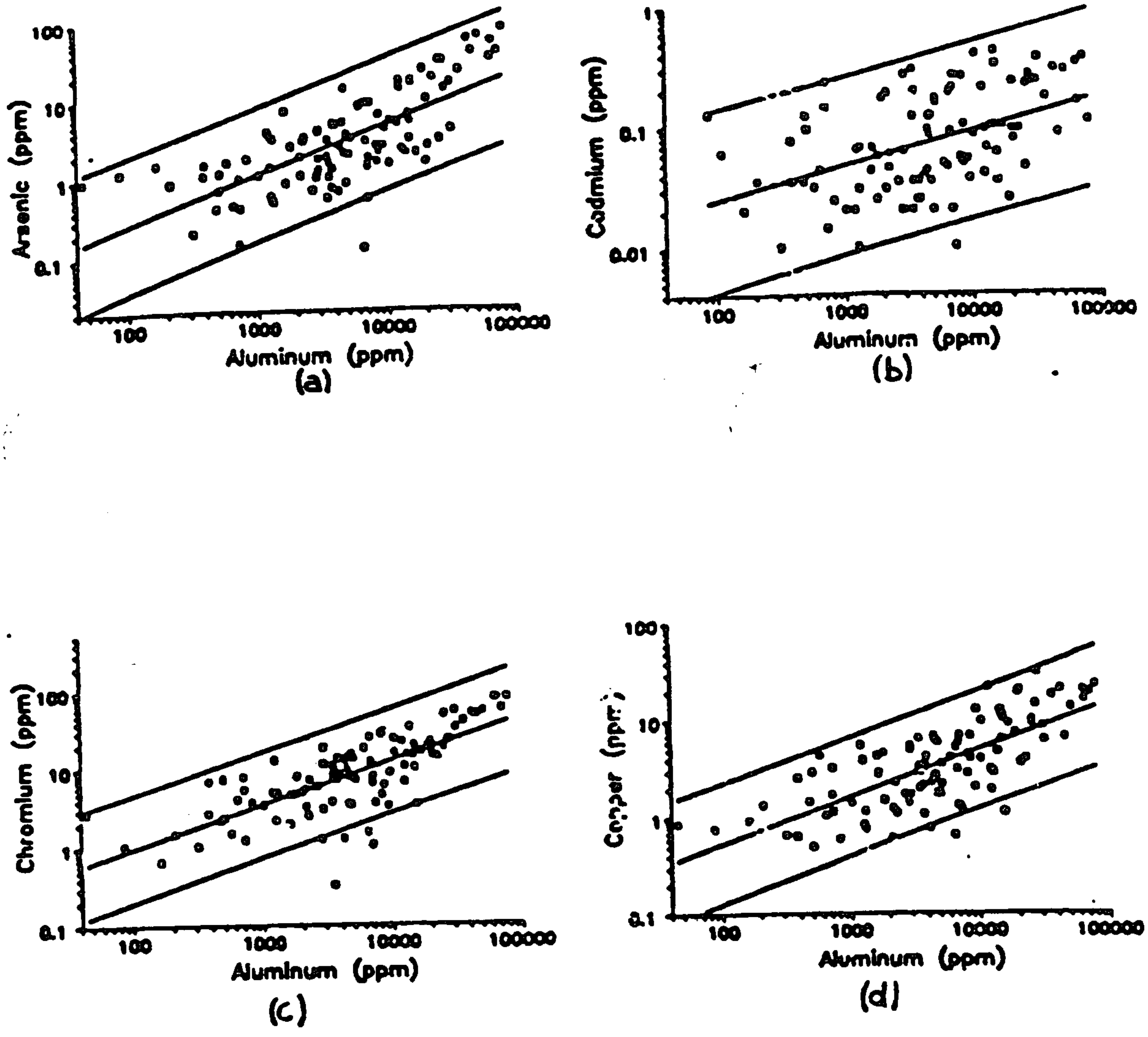
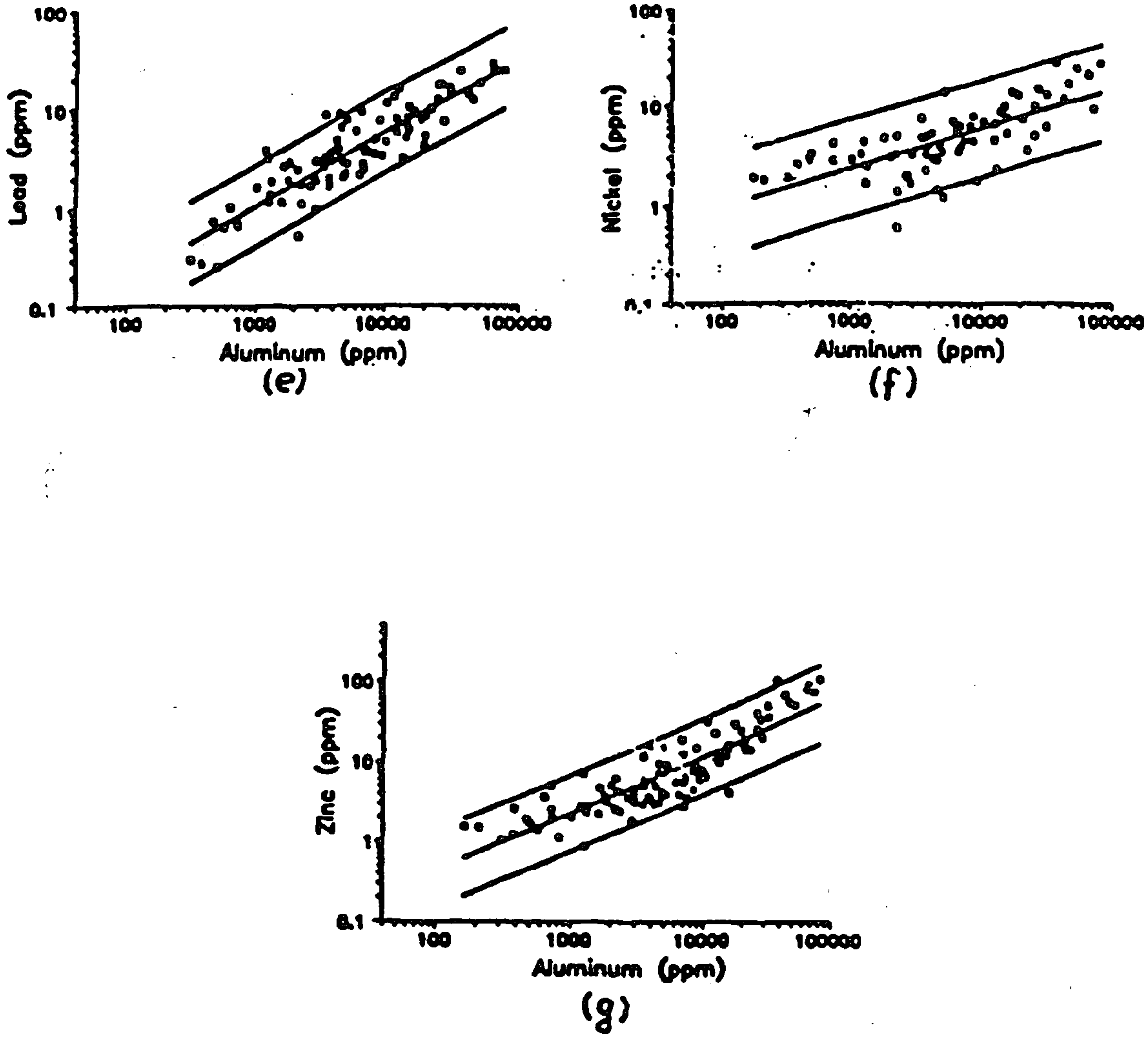


Figure 3.2a Metal:Aluminum ratios for natural sediments: As, Cd, Cr and Cu (Schropp et al. 1990).



(e)

(f)

(g)

Figure 3.2b Metal:Aluminum ratios for natural sediments: Pb, Ni and Zn (Schropp et al. 1990).

accumulations of organic priority pollutants (OPPs) within the riverine sediments (Schell, 1992). These OPPs included polynuclear aromatic hydrocarbons (PAHs), aliphatic hydrocarbons, polychlorinated biphenyls (PCBs), chlorinated pesticides, and chlorinated phenols.

Concentrations of aliphatic and aromatic hydrocarbons, excluding areas affected by petroleum spills or leaks, are directly related to anthropogenic activities. In areas remote from anthropogenic input, natural hydrocarbon concentrations may be two to three orders of magnitude lower (Wakeham and Farrington, 1980). Enrichment of aromatic hydrocarbons in aquatic sediments is attributed to compounds generated by fuel combustion. PCBs are complete or partially chlorinated biphenyl molecule mixtures exhibiting high chemical stability, high dielectric constant, and low flammability and volatility. PCBs are used as heat exchange and dielectric fluids as well as in plastics, caulking compounds, paints, adhesives, carbonless copy paper, and printing inks (Eisenreich et al. 1980). Chlorinated pesticide contamination in the LSJRB, especially in Duval County, is suspected to be from industrial and urban uses rather than from agriculture (Pierce et al. 1988).

Organic pollutants can be analyzed in the laboratory by using Gas Chromatography (GC) and Mass Spectrometry (GC-MS) (Teal et al. 1978). Detection limits can be as low as 0.001 mg/kg sediment. Extraction of the chemical compounds from the sediments can be achieved by using either neutral (e.g., for PAHs), or acid (e.g., for phenols) extraction standard techniques (American Public Health Association, 1985).

3.3 Dames and Moore Maintenance Dredging Study (1983).

3.3.1 Sampling Stations and Collected Parameters. The Dames and Moore study was focused on the Jacksonville metropolitan area, and all samples were collected from the main stem of the river. Water and sediment quality were obtained at eight sites in 1982 during both the spring (dry) and the fall (wet) seasons. The geographic coordinates and locations of the sampling sites are given in Figure 3.3 and Table 3.6. The study included analysis for nutrients, heavy metals, and organic pollutants. The data indicated high TOC (Total Organic Carbon) and elevated nutrient levels in the areas between the Trout River and Arlington River and in the vicinity of Blount Island. Sieve analyses showed that sediments in these sites, with the exception of station JAX 13-SFJ, had high percentages of silt and clay materials. The physicochemical characteristics of the sediments in the sampled sites are listed in Table 3.7. The samples were analyzed for

Table 3.6 Sampling stations of sediment quality (Dames and Moore, 1983).

No.	Station	Location	Latitude	Longitude	Date
1	JAX 1B	Winter Point	30 17 40 N	81 40 49 W	04/30/82
2	JAX 2A	Commodore Pt.	30 18 59 N	81 37 44 W	04/29/82
3	JAX 3A	Floral Bluff	30 20 57 N	81 37 15 W	04/30/82
4	JAX 4B	L. Branch Range	30 22 16 N	81 37 48 W	05/01/82
5	JAX 5B	SJR/Trout River	30 23 16 N	81 37 35 W	05/01/82
6	JAX 6A	Bartram Island	30 24 04 N	81 34 52 W	05/02/82
7	JAX 7A	Blount Island	30 23 10 N	81 33 16 W	05/04/82
9	JAX 8B	Short Cut Turn	30 23 37 N	81 28 44 W	05/03/82
10	JAX 3A-FWE	Floral Bluff	30 21 04 N	81 37 14 W	10/02/82
11	JAX 10-SFJ	Long Branch R.	30 19 00 N	81 38 24 W	10/02/82
12	JAX 12-SFJ	Blount Island	30 24 14 N	81 32 53 W	09/29/82
13	JAX 13-SFJ	Dames P.-Fulton	30 23 20 N	81 31 50 W	10/02/82
14	JAX 14-X	Drummond C.R.	30 24 11 N	81 36 37 W	09/28/82
15	JAX 15-WEJ	Arlington Chan.	30 20 18 N	81 37 21 W	09/26/82
16	JAX 16-S	Long Branch R.	30 21 52 N	81 37 19 W	09/30/82
17	JAX 17-S	Floral Bluff	30 21 06 N	81 37 08 W	10/01/82

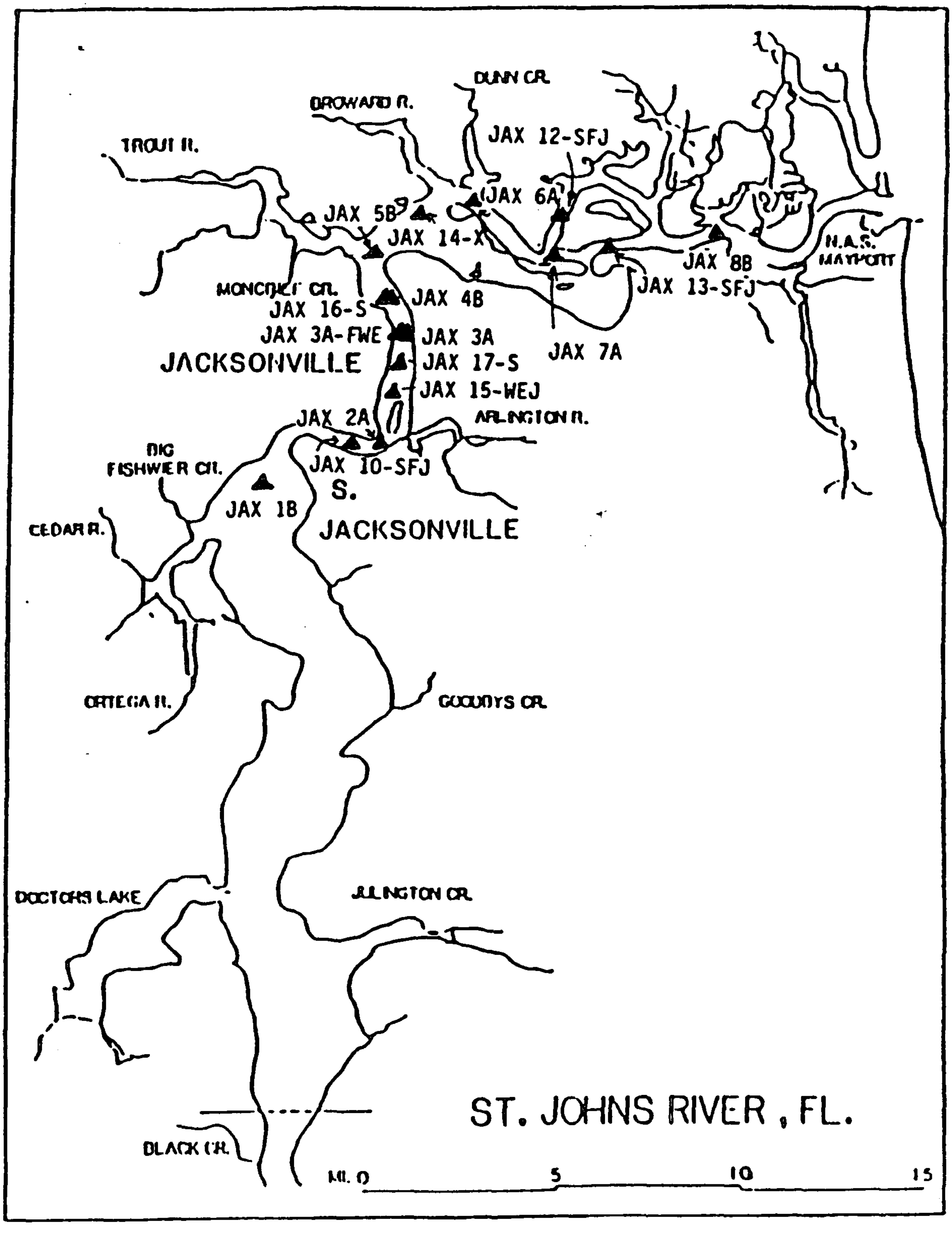


Figure 3.3 Sampling locations of sediment quality (Dames and Moore, 1983).

Table 3.7 General physical and chemical sediment data of the LSJRB (Dames and Moore, 1983).

		Parameter							
Station		Spec. Grav.	BOD mg O_2/kg	NO_3 mg/kg	PO_4 mg/kg	TKN mg/kg	TOC mg/kg	TP mg/kg	TS %
JAX 1B	Mean	2.60	350	5.20	31	240	4800	130	73.8
	S.D.	0.07	53	1.20	5	70	700	29	1.2
JAX 2A	Mean	2.69	440	0.61	37	320	3700	150	75.9
	S.D.	0.01	90	0.04	8	-	800	46	1.1
JAX 3A	Mean	2.46	4100	0.62	600	3200	11000	1600	26.9
	S.D.	0.04	1300	0.05	180	-	3300	330	3.8
JAX 4B	Mean	2.43	7000	1.00	900	9100	43000	2200	19.2
	S.D.	0.05	600	0.20	100	1900	1200	210	2.3
JAX 5B	Mean	2.48	160	0.32	15	650	1100	85	76.8
	S.D.	0.02	15	0.05	6	250	270	-	1.5
JAX 6A	Mean	2.67	200	1.30	56	89	960	240	76.2
	S.D.	0.02	60	0.10	14	-	230	-	1.8
JAX 7A	Mean	3.05	380	0.92	120	350	1100	670	76.1
	S.D.	0.13	70	0.09	10	90	57	21	3.2
JAX 8B	Mean	2.70	160	0.66	110	130	820	160	77.0
	S.D.	0.08	-	0.09	17	55	91	20	2.0

		Parameter			
Station		Spec. Grav.	Oil/Grease mg/kg	TKN mg/kg	TOC mg/kg
JAX 3A-FWE	Mean	2.92	-	4000	20000
	S.D.	0.05	-	760	1200
JAX 10-SFJ	Mean	2.55	90	105	790
	S.D.	0.06	49	-	310
JAX 12-SFJ	Mean	2.38	510	4100	24000
	S.D.	0.04	-	500	3600
JAX 13-SFJ	Mean	2.40	610	2700	19000
	S.D.	0.01	-	950	2100
JAX 15-WEJ	Mean	2.53	46	190	173
	S.D.	0.05	10	47	21
JAX 16-S	Mean	2.38	-	2800	13000
	S.D.	0.17	-	440	2600
JAX 17-S	Mean	2.30	1800	4900	25000
	S.D.	0.01	570	660	580

twelve metals; i.e., aluminum (Al), silver (Ag), arsenic (As), cadmium (Cd), chromium (Cr), copper (Cu), iron (Fe), mercury (Hg), nickel (Ni), lead (Pb), antimony (Sb), and zinc (Zn). Metal concentrations in mg/kg of dry weight are presented in Table 3.8.

Table 3.8 Metal concentrations in the LSJRB sediments (Dames and Moore, 1983).

	Parameter in μg/kg dry basis										
Station	**Al**	**Ag**	**As**	**Cd**	**Cr**	**Cu**	**Fe**	**Hg**	**Ni**	**Pb**	**Zn**
JAX 1B											
Mean	72	-	-	0.05	-	1.1	650	0.13	<1.0	2.4	6.9
S.D.	18	-	-	0.03	-	0.2	190	0.03	-	0.2	1.3
JAX 2A											
Mean	89	0.03	<1.0	0.05	6.3	2.3	1000	0.22	<1.0	11.0	11
S.D.	-	0.01	-	0.02	2.1	-	-	0.04	-	-	-
JAX 3A											
Mean	2500	0.25	6.6	0.33	40.0	25.0	20000	0.59	8.6	25.0	140
S.D.	350	0.01	2.0	-	-	3.0	3000	0.25	-	9.0	27
JAX 4B											
Mean	3000	-	-	0.47	-	24.0	26000	1.10	16.0	45.0	150
S.D.	400	-	-	0.11	-	5.0	3000	0.40	4.0	13.0	36
JAX 5B											
Mean	41	-	-	0.03	-	<1.0	495	0.15	<1.1	0.66	5.4
S.D.	-	-	-	-	-	-	-	0.02	-	0.14	1.7
JAX 6A											
Mean	74	0.06	<1.0	0.09	0.71	<1.5	670	0.12	<1.1	0.78	3.9
S.D.	-	-	-	-	-	-	-	0.03	-	0.12	-
JAX 7A											
Mean	320	0.04	<0.5	0.09	3.6	<1.1	2000	0.15	<1.0	2.2	8.8
S.D.	-	-	-	-	-	-	-	-	-	-	-
JAX 8B											
Mean	140	-	-	0.06	-	<1.1	1000	0.19	<1.0	0.87	2.9
S.D.	-	-	-	0.02	-	-	70	0.02	-	-	0.2

	Parameter in μm/kg dry basis									
Station	Al	Ag	Cd	Cr	Cu	Fe	Hg	Ni	Pb	Zn
				Mean and S.D. values						
JAX 3A-FWE	25000	0.90	1.00	27.0	33.0	27000	0.36	20.0	64.0	10
	2600	0.20	0.30	4.0	5.0	5000	0.03	4.0	15.0	1.0
JAX 10-SFJ	5700	0.10	0.12	6.7	3.9	-	0.16	5.3	6.9	11
	260	-	0.03	0.5	2.1	-	0.04	-	2.0	3.0
JAX 12-SFJ	23000	0.23	0.32	39.0	9.0	-	0.28	7.5	20.0	40
	2500	-	0.13	-	-	-	0.02	1.0	-	-
JAX 13-SFJ	800	0.62	0.12	3.1	1.4	-	0.11	21.0	9.7	3.4
	160	0.17	0.03	0.7	-	-	0.05	-	-	-
JAX 15-WEJ	20000	0.93	0.40	120.0	20.0	12000	0.39	18.0	45.0	125
	3000	0.21	-	-	4.0	2100	0.11	4.0	-	-
JAX 16-S	26000	0.30	0.70	132.0	19.0	-	0.43	34.0	53.0	265
	5200	-	-	-	3.0	-	0.04	-	-	-
JAX 17-S	32000	0.70	1.10	63.0	21.0	-	0.34	25.0	45.0	19
	3100	0.16	-	16.0	6.0	-	0.09	3.0	23.0	8.0

Note: Sb < 0.05 μg/Kg dry basis.

3.3.2 Data Analyses and Results. Based on the metal:aluminum ratios (Table 3.9), metal concentrations were above natural background levels in all of the data stations during the spring, while during the fall the only site that continued to have high Ag, Cd, Cr, Hg, and Ni levels was the one near Blount Island (Dames and Moore, 1983). Some possible explanations for the reduced metal concentrations during the wet season are the following: runoff may have caused higher water velocities resulting in sediment redistribution; runoff water may have concentrations low in metals but high in organic chemicals; or the chemistry of the runoff water resulted in metal mobilization. Traces of Cl-pesticides and PCBs were detected at all sampling sites and did not vary with season. However, concentrations of other organic compounds such as phenols and

Table 3.9 Metal:Aluminum ratios in the LSJRB sediments (Dames and Moore, 1983).

Station	Ag x10^{-4}	Cd x10^{-4}	Cr x10^{-2}	Cu x10^{-2}	Fe x10^{0}	Hg x10^{-3}	Ni x10^{-3}	Pb x10^{-2}	Zn x10^{-3}
JAX 1B	-	6.9	-	1.50	9.00	1.80	<13.9	3.30	95.8
JAX 2A	3.4	5.6	7.10	2.60	11.2	2.50	<11.2	12.40	123.6
JAX 3A	1.0	1.3	1.60	1.00	8.00	0.20	3.40	1.00	56.0
JAX 4B	-	1.6	-	0.80	8.70	0.40	5.30	1.50	50.0
JAX 5B	-	7.3	-	<2.40	12.1	3.70	<2.70	1.60	131.7
JAX 6A	8.1	12.2	1.00	<2.00	9.10	1.60	<14.9	1.10	52.7
JAX 7A	1.3	2.8	1.10	0.30	6.30	0.50	<3.10	0.70	27.5
JAX 8B	-	4.3	-	<0.80	7.10	1.40	<7.10	0.60	20.7
JAX 3A-WEJ	0.4	0.4	0.10	0.10	1.10	0.01	0.80	0.30	0.4
JAX 10-SFJ	0.1	0.1	0.07	0.04	-	0.02	0.55	0.07	1.1
JAX 12-SFJ	0.1	0.1	0.17	0.04	-	0.01	0.33	0.09	1.8
JAX 13-SFJ	7.8	1.5	0.39	0.18	-	0.14	26.3	1.21	4.3
JAX 15-WEJ	0.5	0.20	0.60	0.10	0.60	0.02	0.00	0.23	6.3
JAX 16-S	0.1	0.3	0.51	0.07	-	0.02	1.30	0.20	10.2
JAX 17-S	0.2	0.3	0.20	0.07	-	0.01	0.80	0.14	0.6
Crystal Ratio:	0.001	0.02	0.10	0.07	0.7	0.001	0.90	0.015	0.08
Carbonate Rocks:	0.05	0.09	0.30	10.00	0.9	0.01	4.80	21.000	48.00

Table 3.10 Pesticides, DDE, PCBs, oil/grease in the LSJRB sediments (Dames and Moore, 1983).

		Mirex	Toxaphene	DDT	Aldrin	Chlordane	DDE	PCBs	Oils
Station						(mg/kg dry basis)			
JAX 1B	Mean	<0.001	<0.02	<0.001	<0.001	<0.006	-	<0.002	-
JAX 2A	Mean	<0.001	<0.02	<0.001	<0.001	<0.006	-	<0.002	520
	S.D.	-	-	-	-	-	-	-	40
JAX 3A	Mean	<0.001	<0.02	<0.001	<0.001	<0.006	<0.001	<0.002	1600
	S.D.	-	-	-	-	-	-	-	440
JAX 4B	Mean	<0.001	<0.02	<0.001	<0.001	<0.007	-	<0.002	-
JAX 5B	Mean	<0.001	<0.02	<0.001	<0.001	<0.006	-	<0.002	-
JAX 6A	Mean	<0.001	<0.02	<0.001	<0.001	<0.006	-	<0.002	160
JAX 7A	Mean	<0.001	<0.02	<0.001	<0.001	<0.006	-	<0.002	160
	S.D.	-	-	-	-	-	-	-	20
JAX 8B	Mean	<0.001	<0.02	<0.001	<0.001	<0.006	-	<0.002	-
JAX 10-SFJ	Mean	<0.001	<0.015	<0.001	<0.001	<0.005			
JAX 12-SFJ	Mean	<0.001	<0.015	<0.001	<0.001	<0.005			
JAX 13-SFJ	Mean	<0.001	<0.015	<0.001	<0.001	<0.005			
JAX 16-S	Mean	<0.001	<0.015	<0.001	<0.001	<0.005			
Jax 17-S	Mean	<0.001	<0.015	<0.001	<0.001	<0.005			

Table 3.11 Mean value of phenols in the LSJRB sediments (Dames and Moore, 1983).

	Phenol	2-CP	2,4-DCP	2,4,6-TCP	4-C-m-Cre	2,4-DP	PentaCP
Station				(mg/kg dry basis)			
JAX 1B	<0.2	<0.01	<0.01	<0.01	<0.02	<0.30	<0.005
JAX 2A	<0.2	<0.01	<0.01	<0.01	<0.02	<0.30	<0.005
JAX 3A	<0.2	<0.02	<0.01	<0.01	<0.02	<0.30	<0.005
JAX 4B	<0.2	<0.01	<0.01	<0.01	<0.02	<0.10	<0.005
JAX 5B	<0.2	<0.01	<0.01	<0.01	<0.02	<0.30	<0.005
JAX 6A	<0.2	<0.01	<0.01	<0.01	<0.02	<0.30	<0.005
JAX 7A	<0.2	<0.01	<0.01	<0.01	<0.02	<0.30	<0.005
JAX 8B	<0.2	<0.01	<0.01	<0.01	<0.02	<0.30	<0.005
JAX 10-SFJ	<1.0	<0.05	<1.00	<0.01	<0.01	<0.07	<0.010
JAX 12-SFJ	<1.0	<0.05	<1.00	<0.01	<0.01	<0.07	<0.010
JAX 13-SFJ	<1.0	<0.05	<1.00	<0.01	<0.01	<0.07	<0.010
JAX 16-S	<1.0	<0.05	<1.00	<0.01	<0.01	<0.07	<0.010
JAX 17-S	<1.0	<0.05	<1.00	<0.01	<0.01	<0.07	<0.010

Note: 2-CP = 2-Chlorophenol; 2,4,-DCP = 2,4-Dichlorophenol; 2,4,6-TCP= 2,4,6 Trichlorophenol; 4-C-m-Cre = 4-Chloro-m-cresol; 2,4-DP = 2,4-Dinitrophenol; PentaCP = Pentachlorophenol.

chlorinated phenols did vary with season (Table 3.10). Concentrations of phenol, 2-

chlorophenol, 2,4-dichlorophenol, and pentachlorophenol increased during the wet season. The opposite was true for 4-chloro-m-cresol and 2,4-dinitrophenol (Table 3.11). Contamination of the sampled stations can be attributed to the high industrial and urban activities of the Jacksonville area and its proximity to Blount Island (dredge spoil site).

3.4 Mote Marine Laboratory Study (1987-88).

3.4.1 Sampling Stations and Collected Parameters. The Mote Marine Laboratory Study (Pierce et al. 1988) included three sampling periods. Two sediment and water samples were collected at each of 30 stations located between Mayport and Palatka (Figure 3.4a,b; Table 3.12), once during a dry month (May 1987) and once during a wet month (September 1987). During the third sampling period, 31 stations located between Mayport and Lake George were sampled in four days (March 21-24, 1988). Stations were located either near the mouths of the various tributaries where non-point sources were suspected of having a long-term effect on the system, or in the vicinity of known point sources. Field collected water quality data included temperature, conductivity, pH, dissolved oxygen (DO), and Secchi depths (turbidity). These data were collected at near-surface and near-bottom. For water depths exceeding 3 m, mid-depth measurements were included.

In every station, bottom sediments were collected by at least three petite ponar grabs. Sediment samples were placed in polyethylene trays, and the top 5 cm of the intact core were removed with a non-contaminating scoop and secured in polyethylene jars. Sediments were handled with acid-washed plastic utensils whenever they were to be analyzed for grain size, carbon, nutrients, and metals. Samples were frozen until

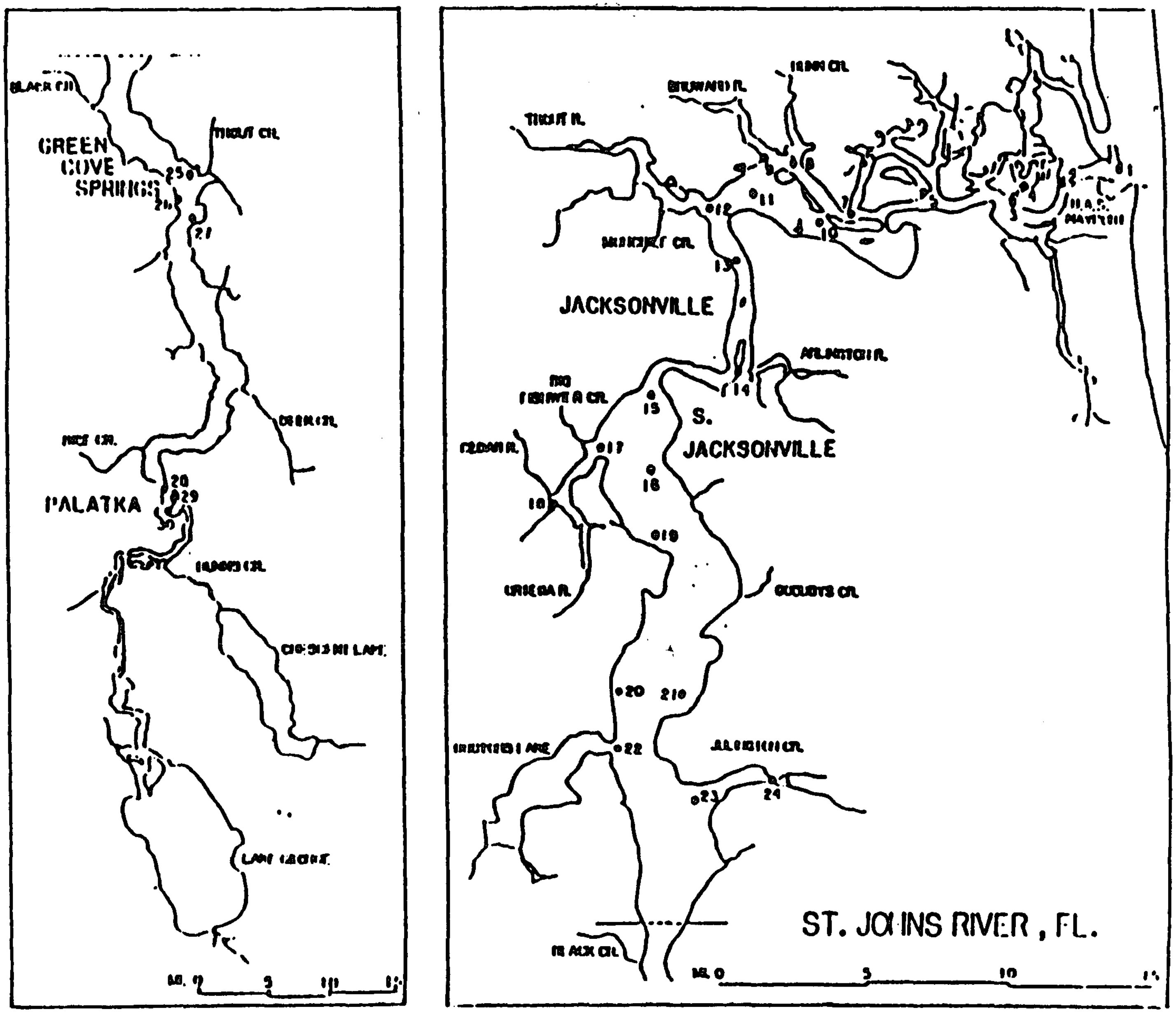


Figure 3.4a Sampling locations, Mote Marine Lab. study, May and September 1987 (Pierce et al. 1988)

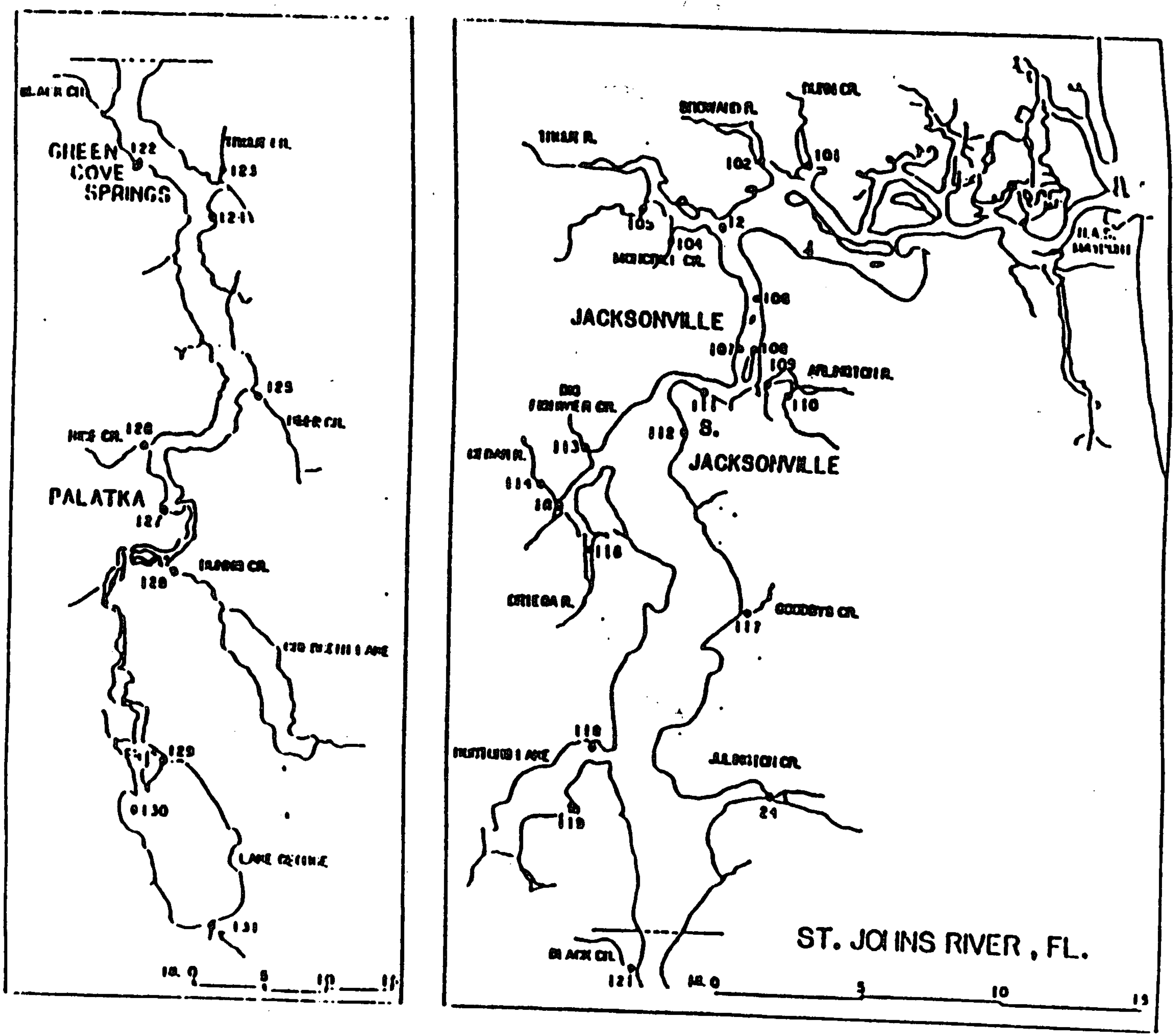


Figure 3.4b Sampling locations, Mote Marine Lab. study, March 1988 (Pierce et al. 1988).

Table 3.12 Sampling stations of sediment quality (metals and OPPs) (Pierce et al. 1988).

No.	Station	Location	Latitude	Longitude	Date
18	S 1	SJR/NAS Mayport	30 24 08 N	81 24 12 W	05,09/87
19	S 2	SJR Jax.	30 24 12 N	81 25 41 W	05,09/87
20	S 3	SJR Jax.	30 23 27 N	81 27 59 W	05,09/87
21	S 4	SJR Jax.	30 23 42 N	81 27 30 W	05,09/87
22	S 5	SJR Jax.	30 23 36 N	81 30 47 W	05,09/87
23	S 6	SJR Jax.	30 24 23 N	81 32 55 W	05,09/87
24	S 7	SJR Jax.	30 23 01 N	81 33 19 W	05,09/87
25	S 8	SJR Jax.	30 24 36 N	81 35 10 W	05,09/87
26	S 9	SJR/Broward R.	30 24 47 N	81 36 01 W	05,09/87
27	S 10	SJR Jax.	30 23 09 N	81 34 42 W	05,09/87
28	S 11	SJR Jax.	30 24 05 N	81 36 24 W	05,09/87
29	S 12	SJR/Trout River	30 23 17 N	81 37 59 W	05,09/87
30	S 13	St. Johns River	30 22 15 N	81 37 05 W	05,09/87
31	S 14	SJR/Arlington R.	30 19 12 N	81 37 10 W	05,09/87
32	S 15	St. Johns River	30 18 27 N	81 40 02 W	05,09/87
33	S 16	St. Johns River	30 16 25 N	81 40 22 W	05,09/87
34	S 17	St. Johns River	30 17 09 N	81 42 14 W	05,09/87
35	S 18	Cedar/Ortega R.	30 15 54 N	81 43 39 W	05,09/87 03/1988
36	S 19	St. Johns River	30 14 50 N	81 40 18 W	05,09/87
37	S 20	St. Johns River	30 10 32 N	81 41 34 W	05,09/87
38	S 21	St. Johns River	30 10 24 N	81 41 45 W	05,09/87
39	S 22	SJR/Doctors Lake	30 08 56 N	81 39 07 W	05,09/87
40	S 23	SJR/Julington Cr	30 07 17 N	81 36 12 W	05,09/87
41	S 24	Julington Creek	30 07 50 N	81 36 19 W	05,09/87
42	S 24A	Julington Creek	30 07 48 N	81 35 00 W	05,09/87
43	S 25	SJR/Trout Creek	29 58 18 N	81 35 00 W	05,09/87
44	S 26	St. Johns River	29 57 35 N	81 36 25 W	05,09/87
45	S 27	St. Johns River	29 56 21 N	81 35 08 W	05,09/87
46	S 28	SJR/Palatka	29 39 50 N	81 37 38 W	05,09/87
47	S 29	St. Johns River	29 39 12 N	81 36 44 W	05,09/87
48	S 30	St. Johns River	29 38 12 N	81 37 23 W	05,09/87
49	S 101	Dunn Creek	30 25 07 N	81 34 59 W	03/1988
50	S 102	Broward River	30 25 06 N	81 36 26 W	03/1988
51	S 104	Moncrief Creek	30 23 27 N	81 39 43 W	03/1988
52	S 105	Ribault River	30 24 12 N	81 40 36 W	03/1988
53	S 106	St. Johns River	30 21 48 N	81 36 48 W	03/1988
54	S 107	St. Johns River	30 20 05 N	81 37 36 W	03/1988
55	S 108	St. Johns River	30 20 10 N	81 36 50 W	03/1988
56	S 109	SJR/Arlington Cr	30 19 02 N	81 36 45 W	03/1988
57	S 110	Arlington River	30 18 48 N	81 35 43 W	03/1988
58	S 111	St. Johns River	30 18 51 N	81 38 26 W	03/1988
59	S 112	St. Johns River	30 17 43 N	81 39 17 W	03/1988
60	S 113	Big Fishwier Cr.	30 17 13 N	81 42 32 W	03/1988
61	S 114	Cedar River	30 16 27 N	81 43 58 W	03/1988
62	S 116	Ortega River	30 14 53 N	81 42 31 W	03/1988
63	S 117	Goodbys Creek	30 12 54 N	81 37 17 W	03/1988
64	S 118	Doctors Lake	30 09 20 N	81 42 34 W	03/1988
65	S 119	Doctors Lake	30 07 26 N	81 43 03 W	03/1988
66	S 121	Black Creek	30 02 35 N	81 42 37 W	03/1988
67	S 122	SJR/Green Cove	29 59 23 N	81 40 17 W	03/1988
68	S 123	Trout Creek	29 58 57 N	81 33 59 W	03/1988
69	S 124	St. Johns River	29 56 37 N	81 34 28 W	03/1988
70	S 125	Deep Creek	29 45 34 N	81 31 45 W	03/1988
71	S 126	Rice Creek	29 42 05 N	81 38 59 W	03/1988
72	S 127	SJR/Palatka	29 37 59 N	81 37 57 W	03/1988
73	S 128	Dunns Creek	29 34 37 N	81 37 24 W	03/1988
74	S 129	Lake George	29 22 17 N	82 37 29 W	03/1988
75	S 130	Lake George	29 19 37 N	82 37 29 W	03/1988
76	S 131	Lake George	29 12 08 N	82 34 17 W	03/1988

delivery to the Mote Marine Laboratory, where they were stored at 4°C.

3.4.2 Laboratory Methods. Percent moisture, total Kjeldahl nitrogen (TKN), and total phosphorus (TP) were analyzed according to the procedures described in the EPA/CE-81-1 report (Plumb, 1981). Concentrations were determined using a Technicon AutoAnalyzer II. Total organic carbon (TOC) was determined using a method developed specifically for marine sediments (Mills and Quinn, 1979). According to this method, dried ground samples weighing 0.01 - 0.02 g were placed in a precombusted ampule where organic carbonates were removed by treatment with phosphoric acid. Samples were then sparged with oxygen to remove dissolved and atmospheric CO_2. Finally, each sample was treated with potassium persulfate and heated in an autoclave. Analysis of the CO_2 produced from the resulting oxidation of the organic carbon was conducted with an OI TOC Analyzer equipped with an infrared detector.

The Mote Marine Laboratory analyzed sediments for six metals: Al, Cd, Cu, Hg, Pb, and Zn. All metals except Hg were determined by AAS analysis after 0.5-1.0 g of dried sediment were acid digested by $HF\text{-}HNO_3\text{-}HClO_4$. Mercury was determined by digesting the sample with acid-permanganate and then using a cold vapor AAS analysis (EPA/CE-81-1; FDER, Deepwater Ports Maintenance Dredging and Disposal Manual).

For analysis of organic compounds, the samples were brought to room temperature, and a homogenized portion of 10-20 g (wet weight) was placed in a 250 ml boiling flask. Internal standards and cyclohexane (C_6H_{12}) were added, and the flask was then covered with a Dean-Stark moisture trap. Once all of the water had been collected in the trap, the cyclohexane was recovered, and samples were extracted with dichloromethane (CH_2Cl_2). The extracts were concentrated by flash-evaporation, and the

C_6H_{12}/CH_2Cl_2 mixture was replaced with hexane (C_6H_{14}). An additional sediment sample was oven dried at 103°C for estimation of the moisture content.

Characteristic fractions of the organic compounds were estimated by using chromatographic separations through a column of alumina and silica gel. The extract was diluted with hexane, and the compounds were eluted from the column by using the following solvents: SA1-elute with C_6H_{14} for saturated hydrocarbons; SA2-elute with C_6H_{14}:CH_2Cl_2 (20:80) for PAHs; SA3-elute with methanol (CH_3OH) for coprostanol. Determination of organic pollutant concentrations was performed by high resolution gas chromatography (GC) employing 1000 series Varian gas chromatographs and 30 meter x 0.25 mm i.d. fused silica DB-5 columns. A flame ionization detector was employed for analysis of PAHs and coprostanol, and an electron capture detector was employed for analysis of chlorinated pesticides and PCBs.

3.4.3 <u>Data Analyses and Results.</u> Organic carbon in sediment samples ranged from 0.3 to 150 mg/g of dry weight with higher concentrations in the upstream sites (Table 3.13). In these sites, most of the fine material was comprised of organic matter. The only anomaly in the TOC data was observed at Rice Creek, where organic carbon concentrations exceeded the next highest observation by 50%. Stations with phosphorus concentrations higher than natural background levels were found at Rice Creek, Fishing Creek, the Cedar River, and at the mouth of Sisters Creek. Higher than average nitrogen concentrations were observed at Dunns Creek, and at the inlet and outlet of Lake George. The nitrogen:phosphorus ratio ranged from 0.05:1 to 38.1:1 on a weight:weight basis and increased from downstream to upstream (Figure 3.5). This increase was the result of increasing nitrogen concentrations in the upstream direction.

Table 3.13 Physicochemical characteristics of the LSJRB sediments (Pierce et al. 1988).

Station	TOC mg/g		TKN mg/g		Total P mg/g		Moisture Content		Silt/Clay %	
	Mean	S.D. (dry basis)	Mean	S.D.	Mean	S.D.	Mean	S.D.	Mean	S.D.
S 1	0.5	0.1	0.06	0.02	0.54	0.09	20.51	1.40	0.20	0.09
S 2	0.4	0.1	0.02	0.01	0.43	0.12	22.69	0.62	0.18	0.16
S 3	5.8	0.9	0.67	0.12	0.83	0.11	39.16	3.52	15.03	3.04
S 4	11.2	9.6	1.29	1.49	0.56	0.40	39.77	13.32	18.43	17.14
S 5	0.6	0.4	0.04	0.03	0.15	0.06	24.36	2.73	2.48	1.83
S 6	6.7	4.4	0.67	0.50	0.30	0.16	38.19	12.24	9.75	7.20
S 7	10.3	6.4	0.83	0.43	0.27	0.09	38.33	4.93	14.28	7.80
S 8	3.9	3.4	0.31	0.32	0.13	0.10	30.63	6.70	7.23	5.22
S 9	34.7	5.7	3.33	0.34	0.79	0.09	67.64	2.60	34.78	4.10
S 10	15.4	5.3	0.99	0.32	0.38	0.15	41.41	5.02	10.70	3.91
S 11	29.8	3.2	2.70	0.26	0.96	0.10	65.04	1.50	26.12	10.00
S 12	35.6	5.9	3.37	0.72	1.02	0.20	68.60	2.43	36.70	5.92
S 13	12.4	5.7	0.92	0.11	0.32	0.03	41.11	1.40	12.02	4.03
S 14	21.1	4.6	1.04	0.18	0.33	0.04	43.62	4.58	12.98	1.99
S 15	1.8	0.8	0.12	0.04	0.22	0.12	24.91	1.63	2.45	1.97
S 16	61.2	26.6	7.08	3.66	0.85	0.38	79.05	8.61	48.58	20.28
S 17	41.8	4.2	4.74	0.42	0.62	0.04	74.45	1.10	27.85	4.83
S 18	72.6	13.4	8.87	1.07	1.70	0.22	82.19	2.50	60.89	14.29
S 19	42.4	10.0	4.63	1.28	0.66	0.20	72.95	5.49	53.35	20.27
S 20	64.4	13.3	9.37	0.55	1.08	0.08	85.31	0.65	37.72	12.15
S 21	59.8	14.0	8.94	0.85	1.08	0.18	84.55	1.45	43.27	7.20
S 22	6.1	1.4	0.54	0.10	0.09	0.02	31.55	1.85	1.93	0.40
S 23	70.1	14.7	11.02	0.59	1.03	0.05	85.47	0.86	44.47	6.05
S 24	80.3	15.5	15.91	1.62	1.27	0.20	89.39	0.79	30.93	6.29
S 25	19.2	4.3	1.51	0.35	0.26	0.07	49.43	6.75	9.93	2.63
S 26	26.4	19.5	3.19	2.65	0.32	0.21	55.57	20.61	8.52	5.29
S 27	78.7	12.6	12.57	1.35	1.12	0.14	87.65	1.20	35.53	6.99
S 28	54.1	6.0	13.91	1.83	0.66	0.38	88.13	2.32	19.48	3.00
S 29	69.4	5.4	12.84	1.37	1.18	0.08	87.87	1.20	21.65	10.83
S 30	22.8	3.9	3.08	1.06	0.49	0.14	59.44	9.37	6.53	2.31
S 101	20.8	6.9	1.57	0.32	0.63	0.27	55.98	6.05	29.27	5.82
S 102	2.9	0.9	0.38	0.07	0.13	0.03	30.11	0.55	4.40	0.92
S 104	64.9	4.8	5.85	0.15	1.34	0.03	79.71	0.39	72.00	2.26
S 105	44.6	5.2	3.64	1.16	0.71	0.25	65.82	12.75	50.77	14.26
S 106	5.3	0.5	0.57	0.06	0.15	0.00	33.66	0.64	3.87	0.35
S 107	22.4	2.4	2.55	0.17	0.81	0.02	61.89	1.58	34.63	1.12
S 108	9.7	1.8	0.93	0.08	0.37	0.01	41.16	2.44	13.67	0.64
S 109	68.7	6.7	5.17	0.40	0.85	0.07	76.05	2.57	43.87	4.15
S 110	11.2	3.6	1.08	0.23	0.23	0.03	39.71	3.67	6.73	1.79
S 111	12.3	3.0	0.95	0.15	0.42	0.06	39.03	2.12	9.83	1.70
S 112	10.2	1.3	0.80	0.13	0.16	0.04	35.00	2.03	6.37	1.16
S 113	82.6	3.5	7.65	0.47	1.17	0.09	80.51	1.85	70.60	2.16
S 114	82.2	11.6	6.00	0.30	1.92	0.11	78.34	0.90	75.87	4.75
S 116	77.9	13.4	7.75	0.93	1.08	0.12	79.64	2.22	41.37	3.21
S 117	54.4	11.5	5.70	0.48	0.71	0.05	75.13	2.11	27.63	1.07
S 118	64.7	3.4	8.72	1.15	1.00	0.08	82.36	1.84	45.43	5.68
S 119	52.4	2.2	7.47	1.31	0.69	0.12	79.77	3.49	21.77	3.08
S 121	67.7	9.2	7.39	1.20	1.04	0.23	81.97	0.81	35.60	4.29
S 122	71.2	9.1	11.12	0.53	1.05	0.10	86.12	1.17	52.53	5.25
S 123	79.7	29.6	10.59	3.68	0.92	0.37	85.07	4.77	22.33	5.10
S 124	32.1	12.7	4.61	0.63	0.51	0.07	72.17	2.05	35.47	2.22
S 125	82.5	4.2	14.37	0.48	1.46	0.16	89.38	1.03	51.47	10.19
S 126	159.7	14.6	13.96	1.24	2.48	0.11	89.93	0.72	16.97	3.72
S 127	99.1	7.2	18.39	0.97	1.57	0.43	90.85	0.54	47.57	4.97
S 128	1.4	0.1	0.24	0.08	0.06	0.02	28.49	1.01	0.40	0.17
S 129	1.8	0.2	0.26	0.03	0.07	0.01	26.14	0.34	0.57	0.15
S 130	24.9	6.2	3.29	0.36	0.34	0.05	61.55	4.95	7.97	1.40
S 131	2.0	0.2	0.37	0.10	0.10	0.02	27.47	1.07	0.87	0.21

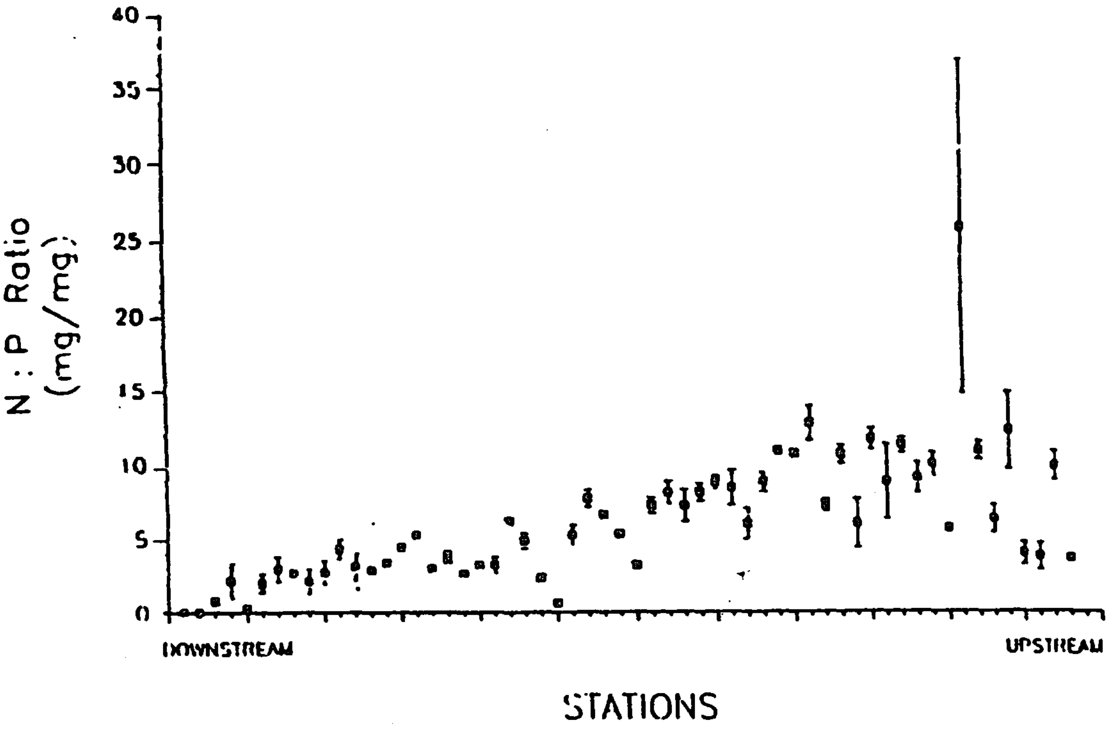


Figure 3.5 Nitrogen:Phosphorus ratios in the LSJRB sediments (Pierce et al. 1988).

Although some variation in the nitrogen:phosphorus ratio was observed across the three sampling periods, no pattern was established. A significant increase in the N:P ratio was observed at Julington Creek between May, 1987, and March, 1988.

Metal enrichment was determined based on the interpolated metal:aluminum ratio as presented in Tables 3.14, 3.15, and Figure 3.6. Locations with high metal contamination, compared to natural metal background levels, include: Moncrief Creek; Big Fishwier Creek; the Cedar River and Fishing Creek (enriched in five metals); the Ortega River and Julington Creek (four metals); Goodbys Creek (three metals); and the Arlington River, Broward River, and two sites in the main river near South Jacksonville (four metals). The only station that was sampled during all three periods was the one at the junction of the Cedar and Ortega Rivers (S 18). Although there were no substantial differences between metal:aluminum ratios for May and September 1987 at this station, all metal:aluminum ratios except mercury dropped significantly in the samples obtained in March of 1988 (Pierce et al. 1988). The reduction in the metal: aluminum ratios did not result from decreased anthropogenically-induced non-aluminum metal concentrations, but rather from high bulk aluminum natural background levels. In 1988, this site (S 18) was still highly contaminated with all metal concentrations exceeding 1.1 to 3.4 times that of the upper 95% metal:aluminum ratio confidence interval for all sites. Comparing the data of May and September, 1987, the only significant changes observed during this period in September were a decrease in all metals except cadmium in the main river off Point La Vista, and an increase in all metal content except cadmium at the mouth of the Ortega River in September. Table 3.16 summarizes the changes in metal concentrations in other stations that were observed from May to September, 1987.

Sediment contamination by organic priority pollutants was also quantified by the Mote Marine Laboratory Study (Table 3.17). PAH concentrations ranged from 0.04 to 25.25 mg/kg (dry weight) during May, 1987, and from 0.04 to 7.64 mg/kg during September, 1987. The station with the lowest concentration was the one near the mouth of the St. Johns River (S 3), while the most contaminated site was the one in the Ortega River (S 18). Data collected in March, 1988, revealed that tributaries in the central part of the LSJRB are the main source of PAH contamination. The analysis also indicated that the majority of the hydrocarbons were from combustion (pyrogenic) sources (i.e.,

Table 3.14 Metal concentrations in the LSJRB sediments (Pierce et al. 1988).

			Parameter in μm/kg dry basis					
Station	Samples		Al	Cd	Cu	Hg	Pb	Zn
S 3	6	Mean	17720	0.18	2.9	0.019	7.3	14.1
		S.D.	4820	0.10	0.5	0.005	0.7	3.3
S 7	6	Mean	13920	0.11	3.9	0.019	6.1	14.5
		S.D.	10280	0.09	3.4	0.012	1.4	7.7
S 9	6	Mean	27460	0.41	16.4	0.123	30.1	94.9
		S.D.	6160	0.22	2.7	0.040	3.9	9.6
S 12	6	Mean	28010	0.23	17.5	0.136	31.1	95.9
		S.D.	8100	0.12	1.8	0.047	5.7	9.2
S 14	6	Mean	10920	0.18	7.7	0.055	20.4	41.5
		S.D.	1160	0.12	0.8	0.010	2.3	1.5
S 16	6	Mean	32950	0.37	15.5	0.165	28.7	101.2
		S.D.	12040	0.16	9.2	0.135	25.0	70.7
S 17	6	Mean	20770	0.66	19.6	0.241	38.2	93.6
		S.D.	2280	0.25	3.8	0.072	8.9	14.4
S 18	9	Mean	50200	2.22	60.0	1.003	187.1	454.4
		S.D.	16030	0.48	3.7	0.331	22.9	35.1
S 19	6	Mean	35180	0.64	11.1	0.123	22.5	64.3
		S.D.	8290	0.29	1.0	0.067	6.3	11.5
S 22	6	Mean	2010	0.09	1.6	0.011	3.4	6.2
		S.D.	900	0.06	0.3	0.002	0.3	0.4
S 24	6	Mean	28010	0.76	15.7	0.301	43.9	104.1
		S.D.	5100	0.12	1.6	0.097	2.5	4.3
S 25	6	Mean	10820	0.10	1.7	0.030	5.8	10.9
		S.D.	2320	0.09	0.4	0.007	1.1	2.4
S 27	6	Mean	22620	0.37	12.5	0.235	26.0	64.1
		S.D.	3230	0.19	0.8	0.053	1.7	6.6
S 28	6	Mean	7380	0.70	4.2	0.095	6.9	10.0
		S.D.	1320	0.11	1.5	0.054	1.9	1.8
S 29	6	Mean	20590	0.30	10.3	0.043	23.3	33.7
		S.D.	1750	0.04	1.1	0.020	2.6	5.1
S 30	6	Mean	5150	0.22	2.5	0.048	8.6	8.0
		S.D.	810	0.03	0.3	0.018	0.9	0.8
S 101	3	Mean	29860	0.10	5.9	0.048	9.6	40.2
		S.D.	10920	0.07	1.2	0.005	1.8	3.4
S 104	3	Mean	63090	1.09	296.1	0.303	120.7	290.1
		S.D.	3370	0.09	23.9	0.032	13.2	21.7
S 105	3	Mean	45920	0.64	37.5	0.111	63.4	142.5
		S.D.	8140	0.08	2.9	0.019	0.8	27.3
S 106	3	Mean	6630	0.06	4.3	0.019	8.6	17.0
		S.D.	630	0.02	2.1	0.003	1.9	1.4
S 107	3	Mean	43490	0.15	29.1	0.091	34.4	108.1
		S.D.	6130	0.03	6.6	0.003	2.1	9.1
S 108	3	Mean	13160	0.30	10.8	0.030	14.6	24.7
		S.D.	2150	0.10	6.6	0.003	0.6	2.4
S 109	3	Mean	43840	0.51	25.8	0.191	50.2	141.8
		S.D.	4820	0.07	3.5	0.007	6.5	16.6

Continued:

Table 3.14 Metal concentrations in the LSJRB sediments (Pierce et al. 1988). (Continued:)

			Parameter in μm/kg dry basis					
Station	Samples		Al	Cd	Cu	Hg	Pb	Zn
S 111	3	Mean	12290	0.21	16.0	0.067	76.6	56.2
		S.D.	2960	0.06	1.1	0.039	9.0	1.8
S 112	3	Mean	6060	0.19	5.3	0.024	23.3	27.6
		S.D.	130	0.19	1.0	0.005	8.4	8.0
S 113	3	Mean	50300	2.00	57.4	0.367	425.4	388.3
		S.D.	2050	0.08	2.6	0.023	18.0	46.9
S 114	3	Mean	65870	2.80	88.2	0.534	358.5	702.5
		S.D.	2770	0.29	3.4	0.049	10.8	34.2
S 116	3	Mean	52410	1.06	30.5	0.308	111.0	190.9
		S.D.	7680	0.16	2.9	0.059	10.4	24.7
S 117	3	Mean	32460	0.49	28.4	0.252	69.5	113.2
		S.D.	3710	0.07	6.1	0.084	10.1	19.7
S 126	3	Mean	24650	0.58	18.7	0.264	17.1	93.5
		S.D.	2310	0.05	2.4	0.070	2.7	9.2
S 128	3	Mean	1790	<0.05	1.1	0.005	4.9	1.5
		S.D.	200	0.01	0.3	0.001	2.9	0.8
S 129	3	Mean	530	0.10	0.7	0.004	0.8	0.4
		S.D.	90	0.05	0.3	0.000	0.7	0.1

Table 3.15 Interpolated metal:aluminum enrichment ratios (Pierce et al. 1988).

Station	Samples		Cd	Cu	Hg	Pb	Zn
S 3	6	Mean	1.98	0.30	0.39	0.52	0.41
		S.D.	1.11	0.09	0.09	0.08	0.04
S 4	6	Mean	2.03	0.63	0.41	0.52	0.46
		S.D.	2.26	0.83	0.25	0.22	0.12
S 9	6	Mean	3.03*	1.18	2.30*	1.70*	2.17*
		S.D.	1.77	0.33	0.79	0.32	0.36
S 12	6	Mean	1.72	1.23*	2.52*	1.76*	2.19*
		S.D.	1.09	0.23	0.90	0.53	0.46
S 14	6	Mean	2.75	1.10	1.25*	1.76*	1.54*
		S.D.	1.78	0.13	0.23	0.22	0.08
S 16	6	Mean	2.13*	0.86	2.71	1.27	1.85
		S.D.	0.38	0.31	2.05	0.98	0.99
S 17	6	Mean	6.01*	1.74*	4.86*	2.51*	2.52*
		S.D.	2.76	0.45	1.51	0.71	0.48
S 18	9	Mean	9.79*	2.60*	15.51*	7.29*	7.03*
		S.D.	4.21	0.69	6.69	1.91	1.77
S 19	6	Mean	3.69*	0.65	2.10	1.10	1.26
		S.D.	1.56	0.17	1.11	0.39	0.33

Continued:

Table 3.15 Interpolated metal:aluminum enrichment ratios (Pierce et al. 1988). (Continued:)

Station	Samples		Cd	Cu	Hg	Pb	Zn
S 22	6	Mean	3.81*	0.54	0.28	0.42	0.35
		S.D.	2.50	0.14	0.05	0.05	0.04
S 24	6	Mean	5.36*	1.09	5.64*	2.43*	2.34*
		S.D.	1.79	0.20	2.16	0.35	0.24
S 25	6	Mean	1.63	0.25	0.69	0.51	0.40
		S.D.	1.59	0.06	0.16	0.11	0.08
S 27	6	Mean	2.91*	1.03	4.63*	1.61*	1.64*
		S.D.	1.33	0.19	1.18	0.08	0.15
S 28	6	Mean	1.70	0.80	2.26	0.68	0.43
		S.D.	2.64	0.33	1.28	0.21	0.09
S 29	6	Mean	2.67*	0.90	0.86	1.51*	0.91
		S.D.	0.41	0.05	0.38	0.11	0.14
S 30	6	Mean	5.50*	0.57	1.19	0.92	0.38
		S.D.	0.36	0.02	0.43	0.07	0.04
S 101	3	Mean	0.79	0.41	0.88	0.54	0.88
		S.D.	0.61	0.15	0.12	0.21	0.16
S 104	3	Mean	3.49*	9.72*	4.03*	3.80*	3.56*
		S.D.	0.34	0.51	0.32	0.57	0.42
S 105	3	Mean	2.80*	1.67*	1.71*	2.54*	2.23*
		S.D.	0.32	0.17	0.29	0.35	0.25
S 106	3	Mean	1.35	0.85	0.45	0.86	0.75
		S.D.	0.44	0.44	0.06	0.18	0.06
S 107	3	Mean	0.69	1.35*	1.43	1.42*	1.78*
		S.D.	0.18	0.28	0.14	0.08	0.23
S 108	3	Mean	3.94*	1.38	0.66	1.18	0.85
		S.D.	1.58	0.93	0.07	0.12	0.14
S 109	3	Mean	2.30*	1.18*	3.00*	2.05*	2.30*
		S.D.	0.11	0.05	0.18	0.12	0.09
S 111	3	Mean	2.83*	2.13*	1.48	6.39*	1.99*
		S.D.	0.40	0.26	0.80	1.29	0.24
S 112	3	Mean	4.42	1.09	0.59	2.39*	1.26
		S.D.	4.53	0.20	0.11	0.86	0.36
S 113	3	Mean	7.93*	2.32*	5.43*	15.79*	5.68*
		S.D.	0.28	0.09	0.41	0.65	0.57
S 114	3	Mean	8.59*	2.79*	6.96*	10.86*	8.31*
		S.D.	0.67	0.20	0.78	0.05	0.47
S 116	3	Mean	4.08*	1.20*	4.48*	4.00*	2.71*
		S.D.	0.63	0.16	0.96	0.08	0.28
S 117	3	Mean	2.94*	1.70*	4.41*	3.51*	2.27*
		S.D.	0.39	0.30	1.33	0.76	0.27
S 126	3	Mean	4.41*	1.42*	5.04*	1.02	2.25*
		S.D.	0.46	0.17	1.22	0.19	0.16
S 128	3	Mean	1.09	0.37	0.13	0.60	0.08
		S.D.	0.24	0.08	0.03	0.36	0.05
S 129	3	Mean	5.62*	0.31	0.10	0.10	0.03
		S.D.	2.72	0.14	0.00	0.09	0.00

Note: (*) denotes enriched metal concentrations.

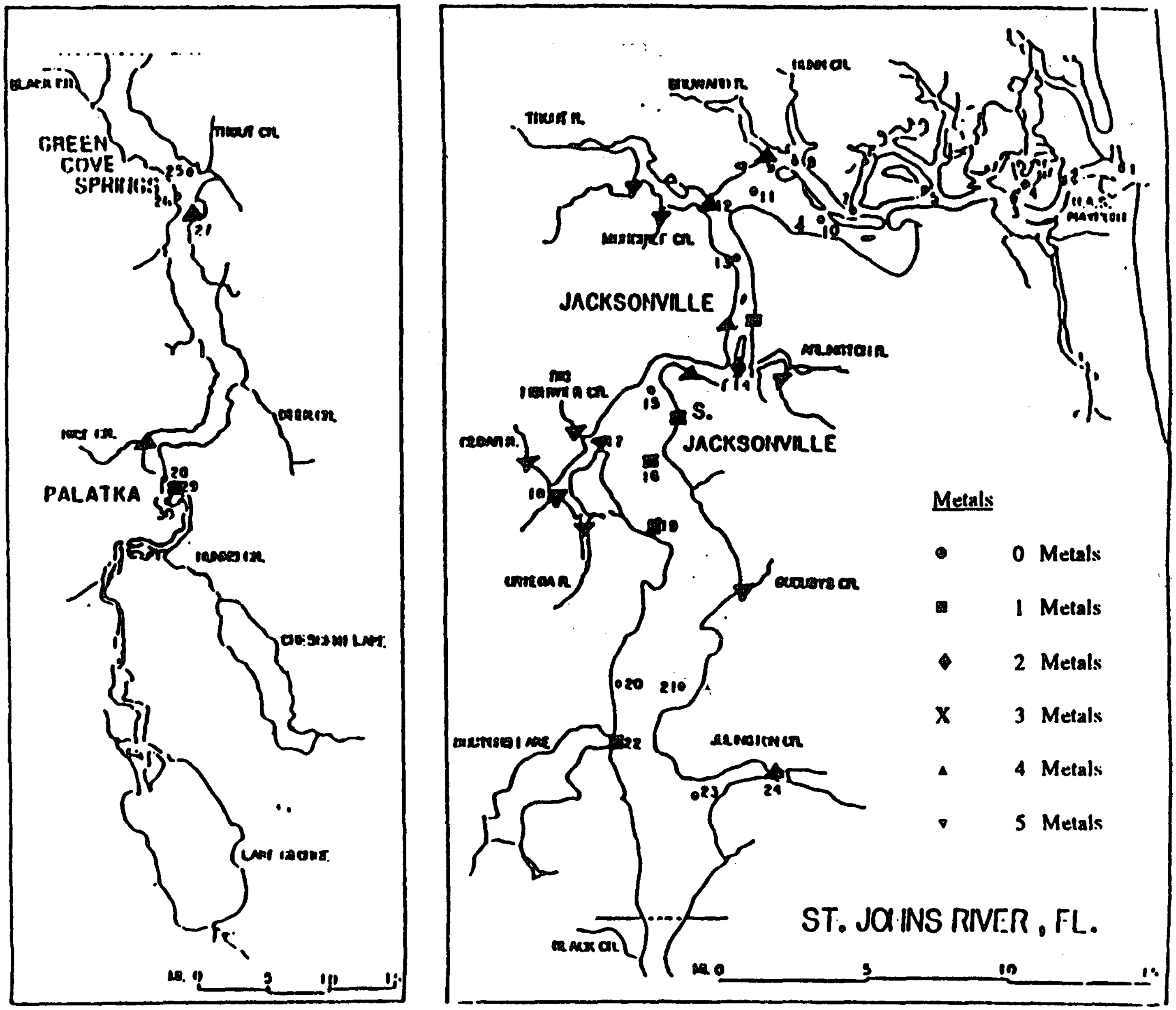


Figure 3.6 Stations with enriched metal concentrations in sediments (Pierce et al. 1988).

Table 3.16 Changes in metal:aluminum ratio from May to September, 1987 (Pierce et al. 1988).

Station	Ca	Cu	Hg	Pb	Zn	Station	Ca	Cu	Hg	Pb	Zn
S 3	+					S 19				-	-
S 7	+					S 25	+			+	
S 9			+			S 27			+		
S 14	+					S 28					+
S 16		-	-	-	-	S 30					-
S 17		+	+	+	+						

Note: (+) = increase; (-) = decrease

high temperature oxidation of primary organic compounds) rather than petroleum (petrogenic) sources (i.e., oil spill or leakage), which is indicative of highly industrialized environments.

In silt-clay, PAH concentrations of 10-20 mg/kg are defined as low, 20-30 mg/kg as moderate, and 30-70 mg/kg as high. Based on these categories, low PAH concentrations were detected in the Arlington River, Cedar/Ortega Rivers, Doctors Lake, and Trout River in May 1987 (Figures 3.7a-c). Samples selected in September, 1987, from the Arlington River and Point La Vista had low PAH concentrations. High PAH concentrations were measured in samples taken in March 1988 from Big Fishwier Creek, the Cedar/Ortega Rivers, Fishing Creek, Goodbys Creek, Ribault River, and off South Jacksonville, while moderate concentrations were found in Dunn Creek and low concentrations, in Moncrief Creek and Rice Creek.

Chlorinated pesticides (Cl-pesticides) and PCBs were also present in the LSJRB sediments presumably derived from industrial and agricultural activities. The most abundant Cl-pesticides included chlordane, trans-nanochlor, and DDT derivatives. The

Table 3.17 PAHs, Cl-pesticides, PCBs and coprostanol in the LSJRB sediments (Pierce et al. 1988).

		Total sediment				silt-clay	Only silt-clay (<10%)*			
Station	Date	PAHs mg/kg	Cl-Pest. µg/kg	PCBs µg/kg	Copr. mg/kg	%	PAHs mg/kg	Cl-Pest. µg/kg	PCBs µg/kg	Copr. mg/kg
S 3	05/87	0.04	4.0	6.0	-	16.0	0.25	25.0	37.5	0.01
S 7	05/87	0.16	<0.1	6.0	0.05	17.8	0.90	<0.1	33.7	0.28
S 9	05/87	9.94	37.0	86.0	0.73	31.5	31.52	117.5	273.0	2.31
S 12	05/87	2.71	60.0	126.0	0.51	33.2	8.16	180.7	379.5	1.53
S 14	05/87	1.91	56.0	216.0	0.16	11.2	16.99	500.0	1928.6	1.42
S 16	05/87	14.93	76.0	177.0	2.70	36.7	40.71	207.1	482.3	7.36
S 17	05/87	2.26	34.0	123.0	0.46	24.7	9.16	137.6	498.0	1.86
S 18	05/87	25.25	165.0	788.0	0.86	68.7	36.76	240.2	1147.0	1.25
S 19	05/87	2.66	50.0	124.0	1.48	24.2	11.01	206.6	512.4	6.13
S 22	05/87	0.09	5.0	80.0	0.05	1.8	(5.10	(277.8)	(444.4)	(2.82)
S 24	05/87	3.33	17.0	420.0	3.69	34.8	9.55	48.7	120.3	10.58
S 25	05/87	0.08	6.0	60.0	0.20	7.9	(1.00)	(75.9)	(75.9)	(2.53)
S 27	05/87	2.02	106.0	36.0	1.50	32.3	6.05	328.2	111.4	4.64
S 28	05/87	0.32	3.0	16.0	1.57	19.2	1.67	15.6	83.3	8.18
S 30	05/87	2.96	18.0	16.0	0.35	4.6	(65.00)	(391.3)	(374.8)	(7.67)
S 3	09/87	0.04	3.0	4.0	0.09	14.0	0.40	21.5	28.6	0.64
S 7	09/87	0.30	3.0	6.0	0.02	10.8	2.78	27.8	55.6	0.18
S 9	09/87	2.68	28.0	61.0	0.16	38.0	7.05	73.7	160.5	0.42
S 12	09/87	4.26	26.0	109.0	0.25	36.8	11.58	70.6	269.2	0.68
S 14	09/87	2.16	41.0	272.0	0.05	14.7	14.69	278.9	1850.3	0.34
S 16	09/87	3.36	85.0	214.0	1.96	65.7	5.11	129.4	325.7	2.99
S 17	09/87	2.54	24.0	107.0	0.40	31.0	8.19	77.4	345.2	1.29
S 18	09/87	7.64	310.0	1776.0	1.14	68.8	11.10	450.6	2581.4	1.66
S 19	09/87	2.08	167.0	250.0	1.02	70.6	2.95	236.5	354.2 1	1.44
S 22	09/87	0.31	6.0	7.0	0.04	2.1	(15.00)	(285.7)	(333.3)	(1.90)
S 24	09/87	3.47	115.0	193.0	2.18	57.7	6.01	199.3	334.5	3.78
S 25	09/87	0.22	6.0	15.0	0.16	29.5	0.75	20.3	50.8	0.54
S 27	09/87	2.07	45.0	111.0	1.61	38.7	5.81	116.3	286.8	4.16
S 28	09/87	0.82	12.0	10.0	0.84	19.8	4.14	60.6	50.5	4.24
S 30	09/87	2.83	26.0	11.0	0.38	8.5	(33.00)	(305.9)	(129.4)	(4.47)
S 18	03/88	22.38	84.8	119.1	1.38	45.2	49.52	187.7	263.5	3.05
S 101	03/88	8.18	60.7	70.0	0.69	29.3	27.96	207.2	238.9	2.36
S 104	03/88	8.62	25.7	247.1	1.63	72.0	11.97	36.7	343.2	2.26
S 105	03/88	15.45	24.6	6.0	1.82	50.8	30.44	48.4	11.8	3.59
S 106	03/88	2.31	27.9	9.4	0.06	3.9	(59.00)	(734.0)	(247.0)	(1.57)
S 108	03/88	2.19	29.9	47.3	0.09	13.7	16.01	218.2	345.3	0.66
S 109	03/88	2.69	17.1	5.5	0.30	43.9	6.11	39.0	12.5	0.68
S 111	03/88	5.69	55.3	7.2	0.10	9.8	57.82	553.0	72.7	(10.20)
S 112	03/88	2.58	13.5	6.3	0.09	6.4	(41.00)	-	(100.0)	(1.43)
S 113	03/88	21.96	113.0	31.5	1.04	70.6	31.12	160.3	44.6	1.47
S 114	03/88	32.45	31.6	243.6	1.49	75.8	42.79	41.7	320.5	1.96
S 116	03/88	16.33	63.5	55.5	2.17	41.3	39.50	153.8	134.4	5.25
S 117	03/88	17.89	69.1	31.0	0.89	27.6	64.72	33.0	112.3	3.22
S 126	03/88	2.51	<0.1	12.7	0.62	17.0	14.78	<1.0	74.7	3.65
S 128	03/88	0.54	<0.1	19.5	0.07	0.4	(186.00)	<1.0	(5000)	(17.90)
S 129	03/88	0.20	<0.1	2.1	0.15	0.6	(36.00)	<1.0	(318.0)	(27.30)

<u>Note</u>: Data include: PAHs: Total of 18 PAHs; Cl-Pesticides: Chlordane, DDT derivatives and Nanochlor; PCBs: Total of select PCB isomers.
() Analysis only of the silt-clay portion of the sample, i.e., approximately 10% of the total sample.

most abundant PCB compounds were the tri-, tetra-, penta-, and hexa-chlorinated isomers. In silt-clay, Cl-pesticide or PCB concentrations of 50-150 µg/kg were defined

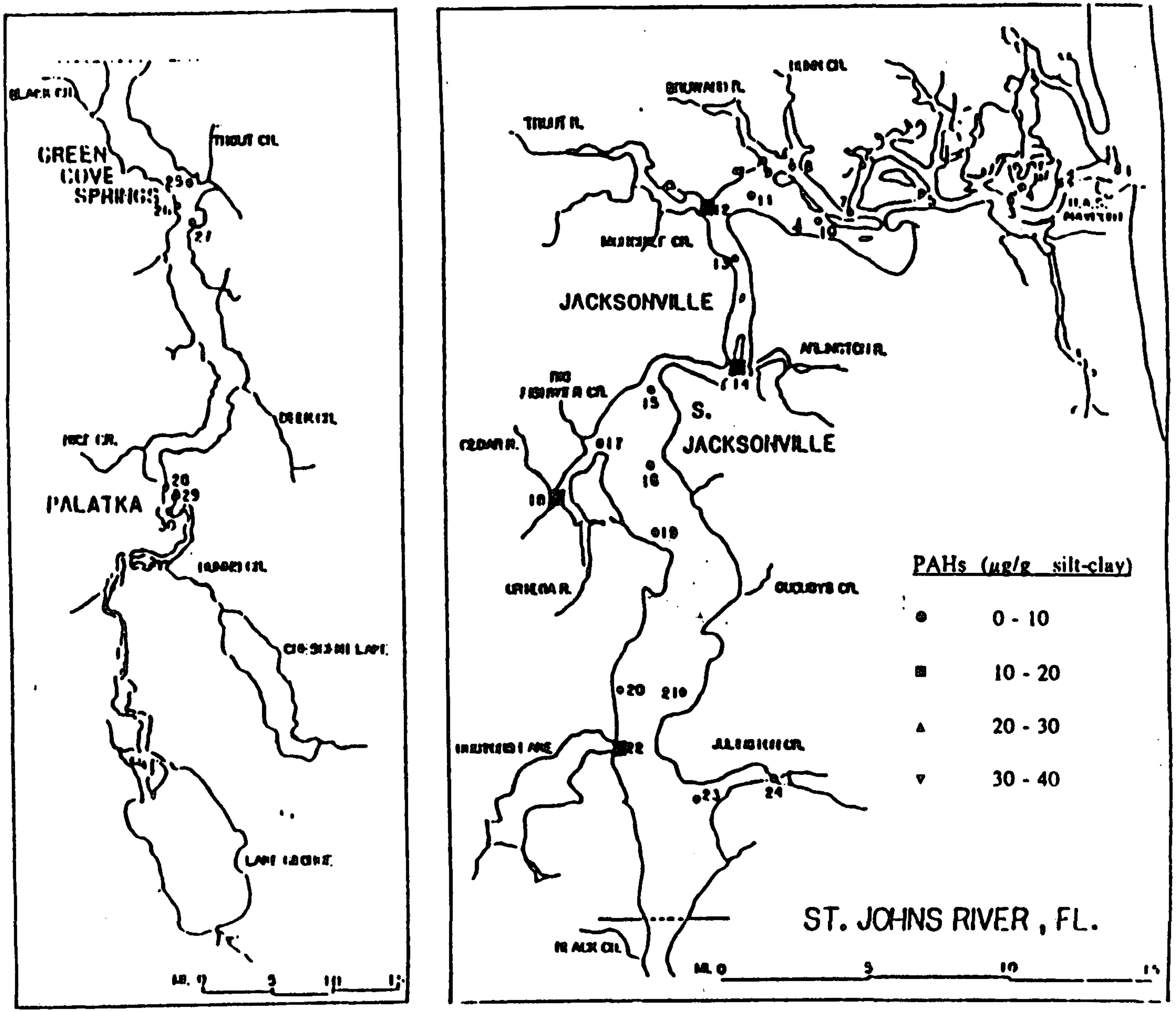


Figure 3.7a PAHs distribution in the LSJRB sediments, May 1987 (Pierce et al. 1988).

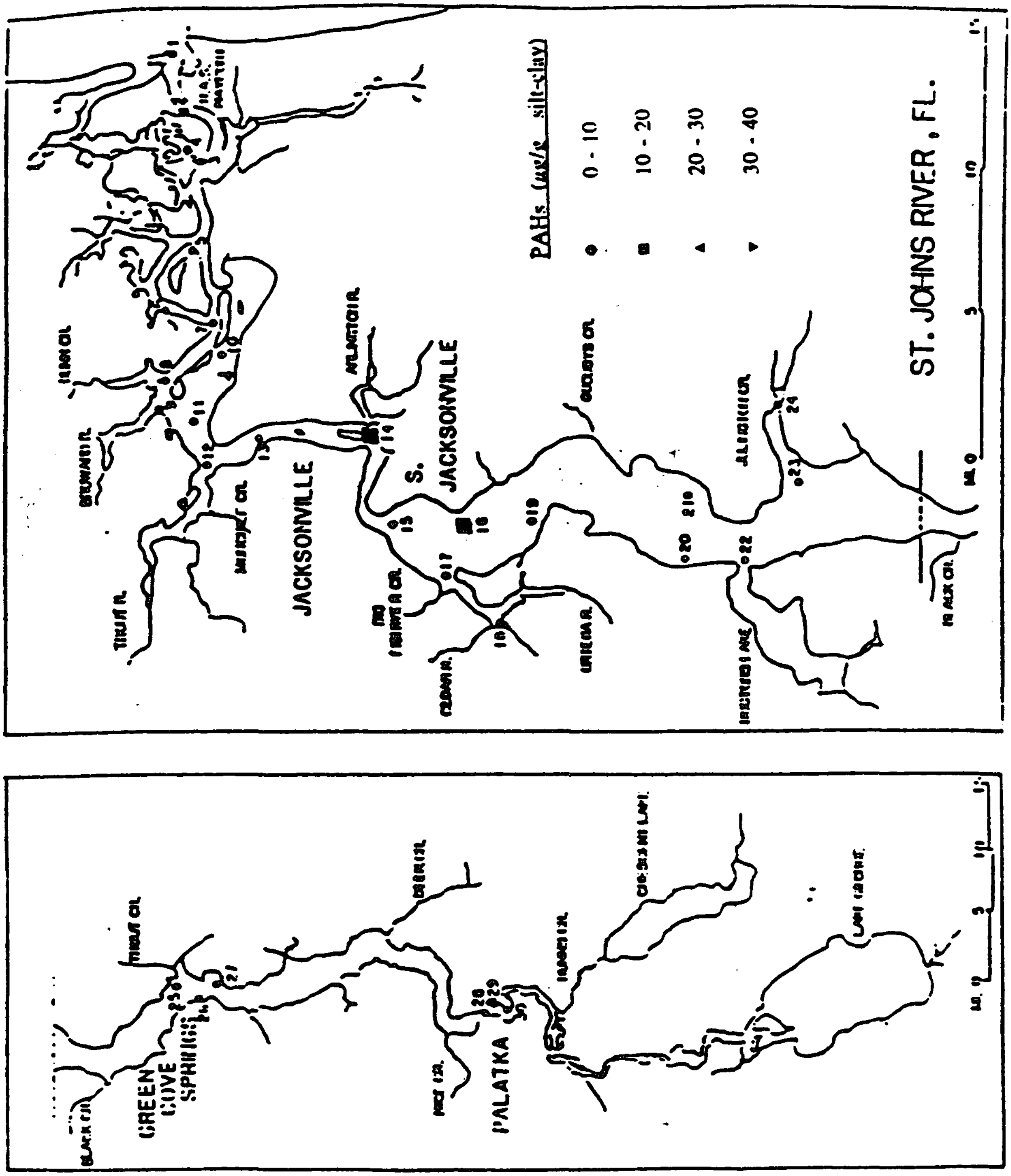


Figure 3.7b PAHs distribution in the LSJRB sediments, September 1987 (Pierce et al. 1988).

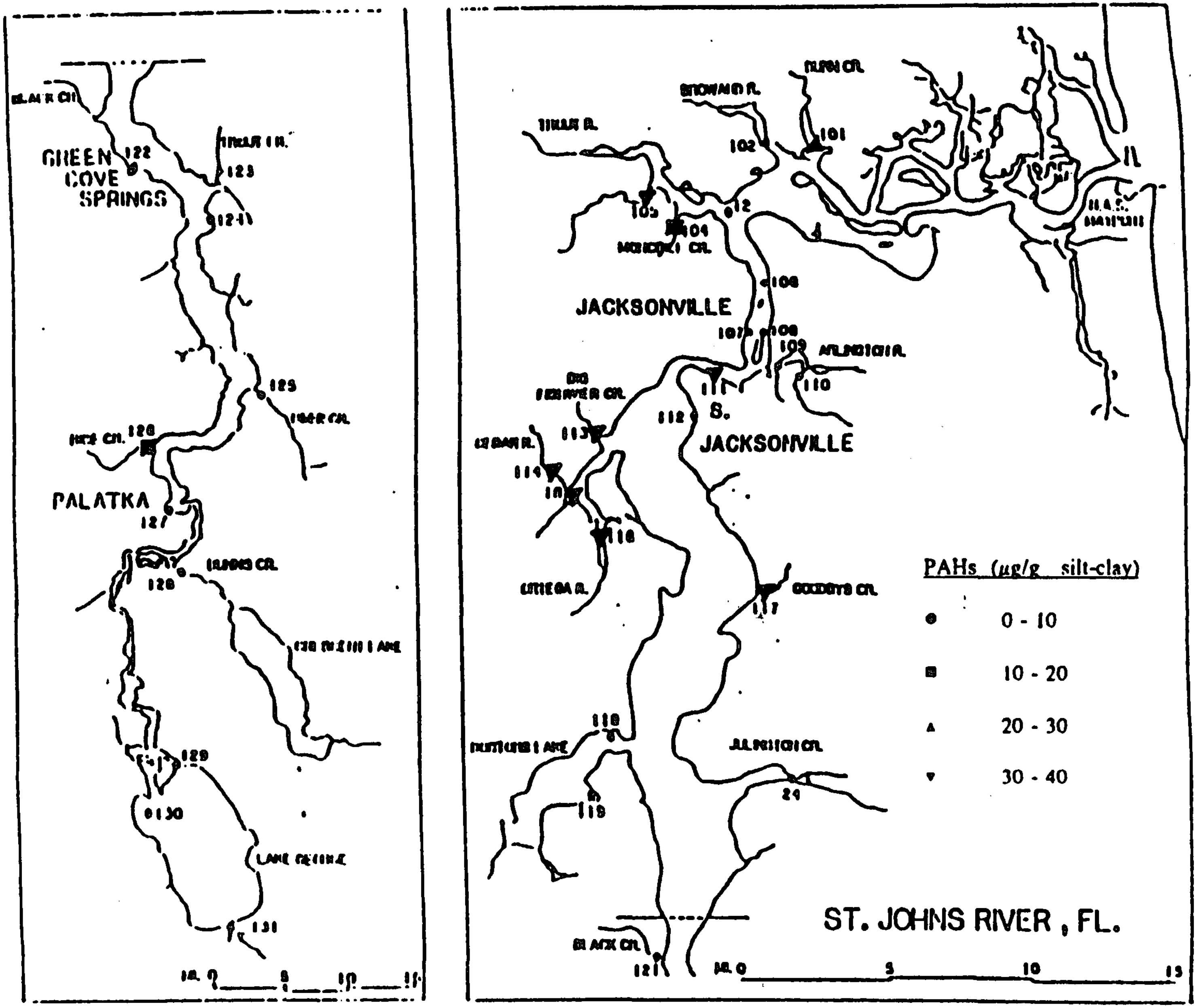


Figure 3.7c PAHs distribution in the LSJRB sediments, March 1988 (Pierce et al. 1988).

as low, 150-500 μg/kg as moderate, and >500 μg/kg as high.

Using these categories, Cl-pesticide concentrations were high in the Arlington River; moderate in Moncrief Creek, the Cedar/Ortega Rivers, off Point La Vista, and off Green Cove Springs; and low in the Broward River and Fishwier Creek during March 1987 (Figures 3.8a-c). Concentrations were reduced to the "low" category in samples collected in September 1987 at four of the stations (the Arlington River, Moncrief Creek, Point La Vista, and Green Cove Springs); they were unchanged from May 1987 in the Cedar/Ortega Rivers, the Broward River and Big Fishwier Creek; were moderate in the Julington River; and low near Palatka. In the March 1988 samples, lower Cl-pesticide levels were observed in seven of the stations (Moncrief Creek, Point La Vista, the Broward River, Green Cove Springs, Big Fishwier, the Julington River and near Palatka), as compared to May and September 1987; in the Arlington River, Cedar/Ortega Rivers and Big Fishwier Creek, concentrations remained constant; in Dunn Creek, they increased from low to moderate; and off South Jacksonville, they increased but still remained in the low category.

Concentrations of PCBs were high during the first sampling period in the Arlington River and the Cedar/Ortega Rivers; moderate in the Broward River, Trout River, Ortega River, and off Point La Vista; and low in the Julington River and Palatka. During the second sampling period, the conditions remained unchanged except for increases in the Julington River, Green Cove Springs and Trout Creek and a decrease in Palatka. The third sampling period documented medium concentrations in the Arlington River, Dunn Creek, Moncrief Creek, and the Cedar/Ortega River, and low concentrations in the Ortega River, Goodbys Creek, Rice Creek and off South

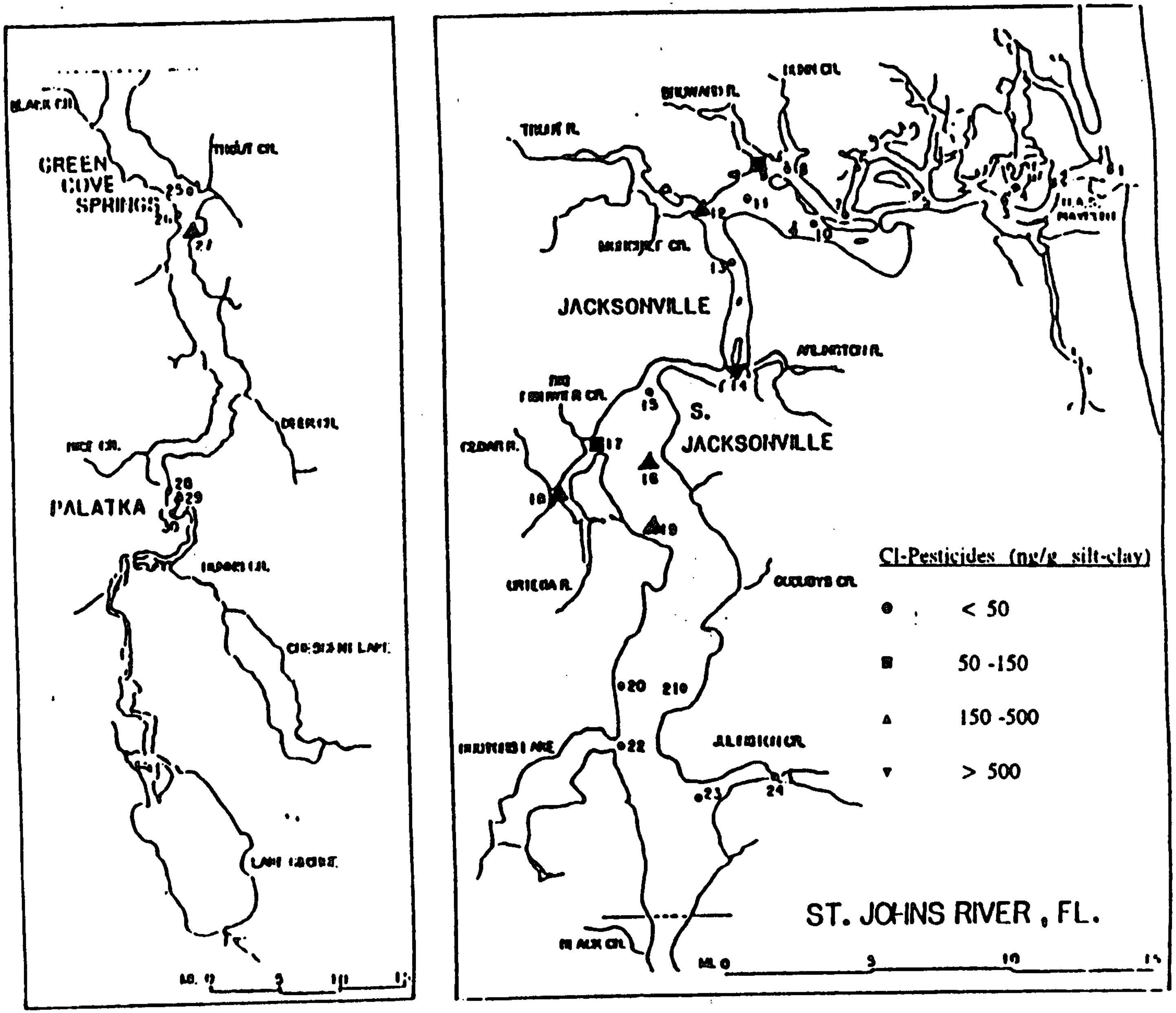


Figure 3.8a Cl-pesticides distribution in the LSJRB sediments, May 1987 (Pierce et al. 1988).

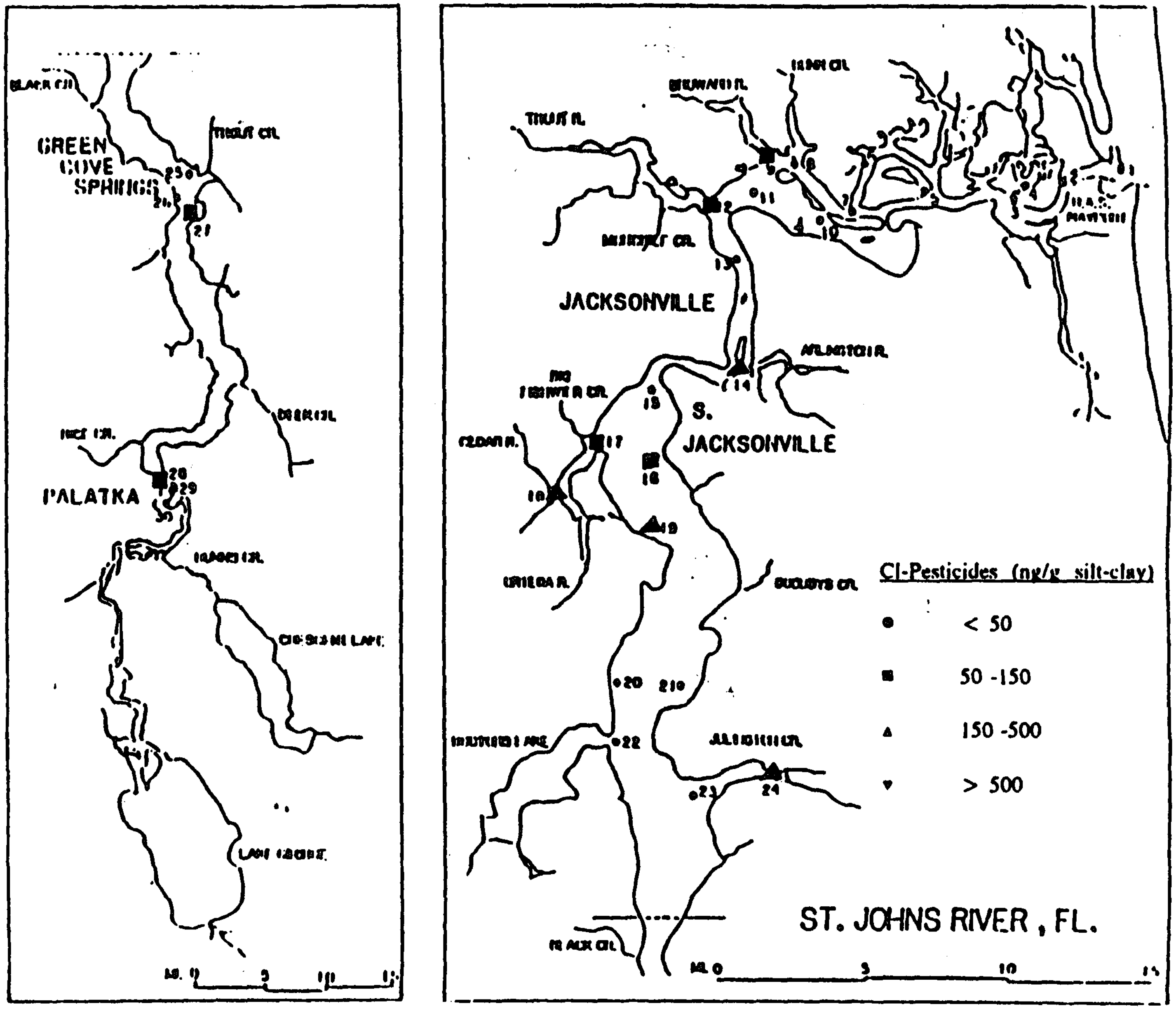


Figure 3.8b Cl-pesticides distribution in the LSJRB sediments, September 1987 (Pierce et al. 1988).

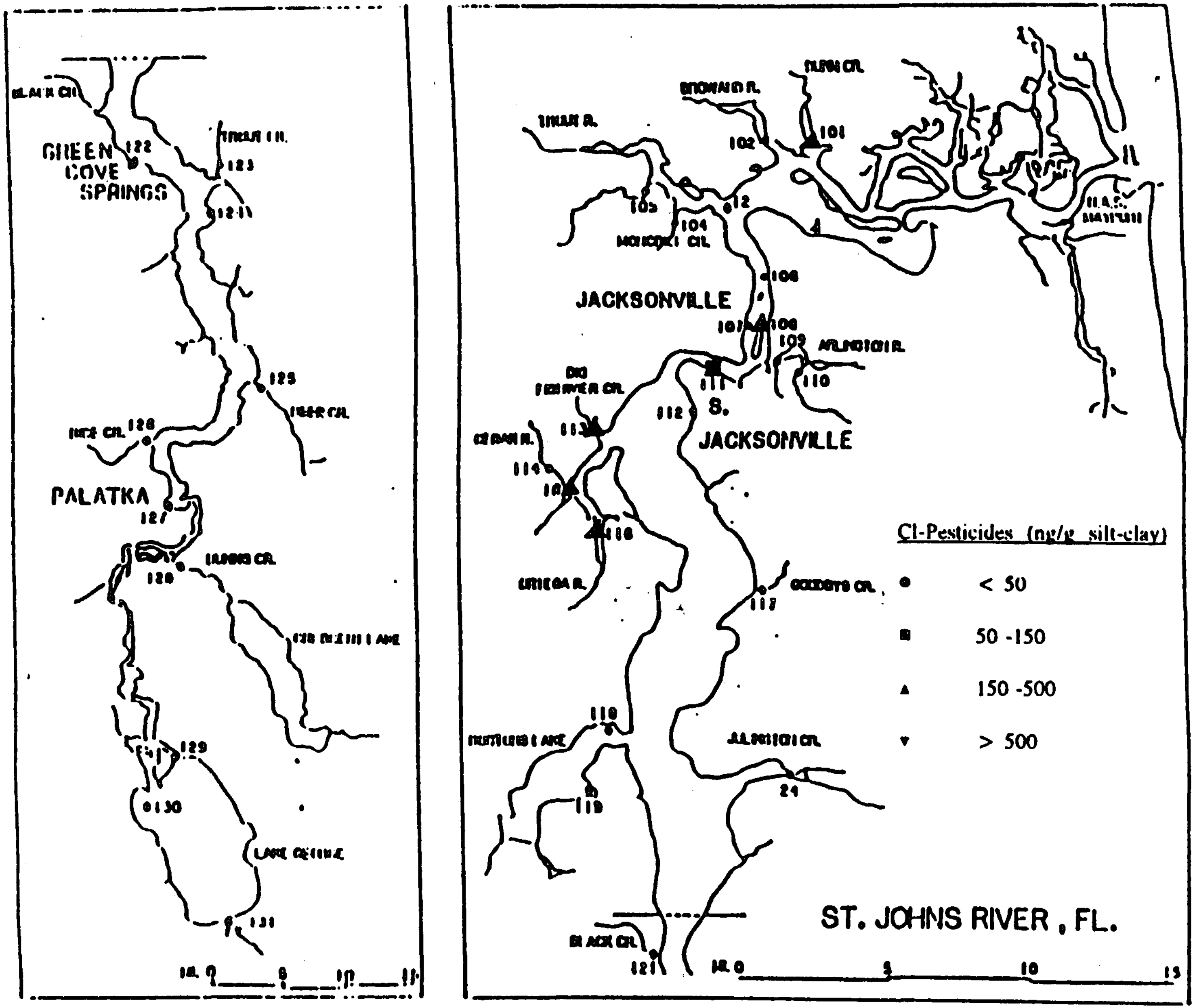


Figure 3.8c Cl-pesticides distribution in the LSJRB sediments, March 1988 (Pierce et al. 1988).

Jacksonville. The changes in PCB concentration during the three sampling events are illustrated in Figures 3.9a-c. Mote Marine Laboratory Study also analyzed coprostanol, another organic compound indicative of fecally-derived material (Table 3.17). Elevated coprostanol levels were detected not only in the tributaries but also within the main river. More specifically, high concentrations (>5 mg/kg silt-clay) were found off Point La Vista and off Piney Point, near Palatka, in Julington Creek and in the Ortega River (Figures 3.10a-c).

Since organic pollutants have an affinity for fine sediments, the concentrations of PAHs, Cl-pesticides, PCBs, and coprostanol in reference to the silt-clay portion of the sediments are also provided in Table 3.17.

In summary, the main pollution sources in the sampled stations were comprised of waste water and industrial discharges, septic tank and sewer system leachate, stormwater runoff, and eolian transport of contaminated particulates. Nutrients in the sampled locations were of organic nature (detritus and sewage) rather than inorganic fertilizers. Coprostanol levels were high compared to fecal coliform count. Since both coprostanol and fecal coliforms are indicators of domestic wastewater, this discrepancy is surprising. Coprostanol is the only sterol which is not of natural origin in sediments (Kanazawa and Teshima, 1978). Fossil fuels and fuel combustion products (e.g., benzene, fluoranthene, pyrene and toluene) were also present as a result of industrial and boating activities. The primary chlorinated pesticides that were found in sediment samples (e.g., chlordane and parent DDT isomers) were from residential-industrial rather than agricultural sources.

The most contaminated riverine section was found to be the mid-section of the estuary and its tributaries from Dunn Creek to the Ortega River. The most heavily contaminated sites were the Cedar River and near the Naval Air Station. This section was widely contaminated with heavy metals, PAHs, chlorinated pesticides, PCBs, and fecally-derived material. The highest PAH concentrations were documented during March 1988 following heavy rains, which probably flushed PAHs from the adjacent watersheds. Other contaminated sites included the Arlington River, Ortega River, Trout River and Ribault River. The main hazard to the aquatic biota appeared to stem from the presence of toxic metals (e.g., Cd, Cu, Hg, Pb, and Zn) and mutagenic/carcinogenic PAHs. A summary of the sites with "enriched" contamination levels as found by Pierce et al. (1988) is presented in Table 3.18.

Table 3.18 Sites with "enriched" contamination levels in the LSJRB (Pierce et al. 1988).

Location	Contaminant							
	TKN	ΣP	Metals Cd,Cu Pb,Zn	Hg	PAH	Coprost.	Cl-Pest.	PCB
Arlington R.			2		x	x	x	x
Big Fishwier Cr.			4		x	x	x	x
Blount Is.		x						
Broward R.			1			x	x	x
Cedar R.		x	4	x	x	x	x	x
Dunn Cr.					x		x	x
Dunns Cr.	x							
Goodbys Cr.			2	x	x			
Green Cove Spr.				x			x	x
Julington Cr.			3	x		x	x	x
Lake George	x							
Moncrief Cr.			4	x	x	x		x
Ortega R.		x	3	x	x	x	x	x
Palatka	x						x	x
Rice Cr.		x	1			x	x	x
Trout Cr.						x	x	x

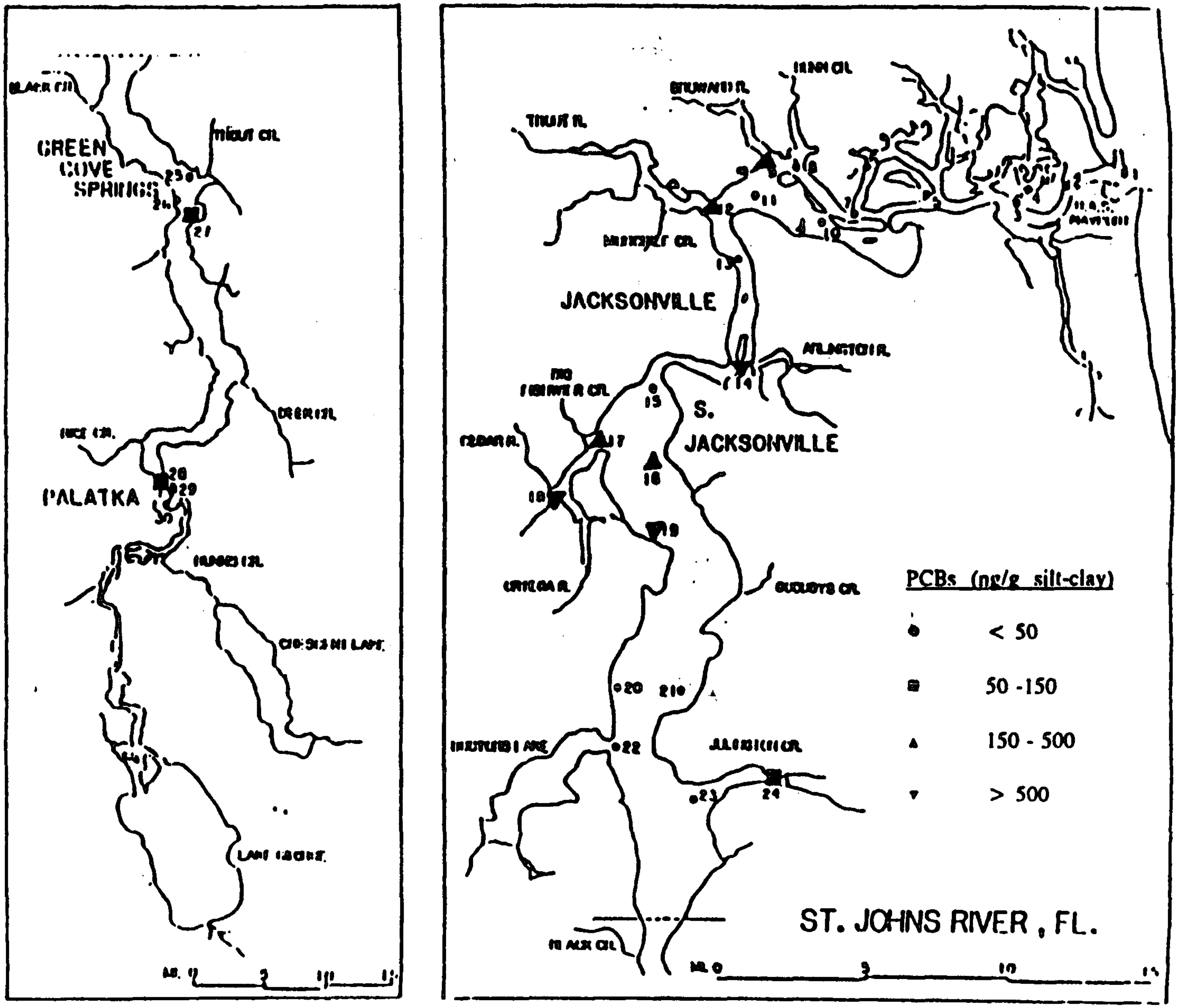


Figure 3.9a Total PCBs distribution in the LSJRB sediments, May 1987 (Pierce et al. 1988).

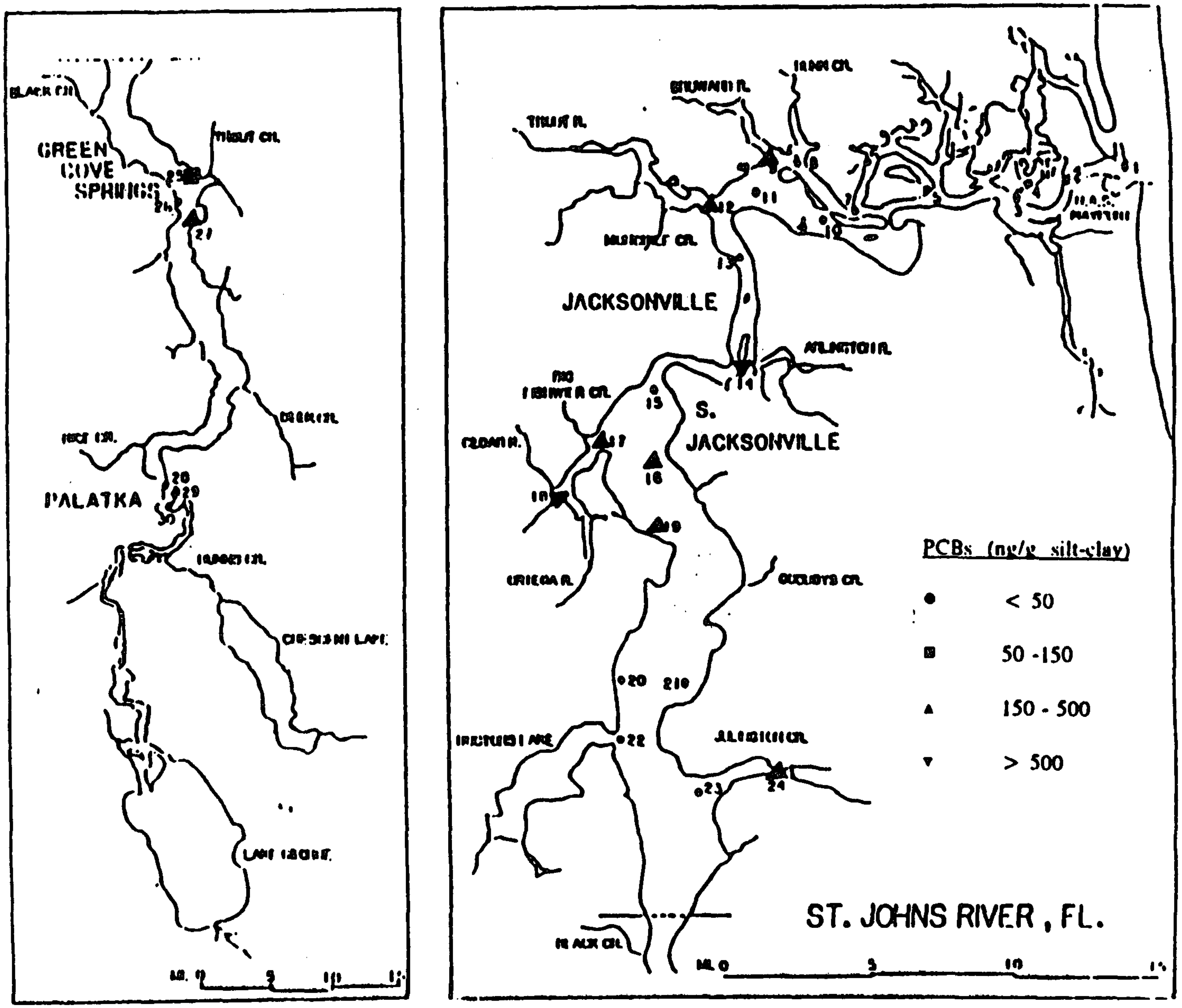


Figure 3.9b Total PCBs distribution in the LSJRB sediments, September 1987 (Pierce et al. 1988).

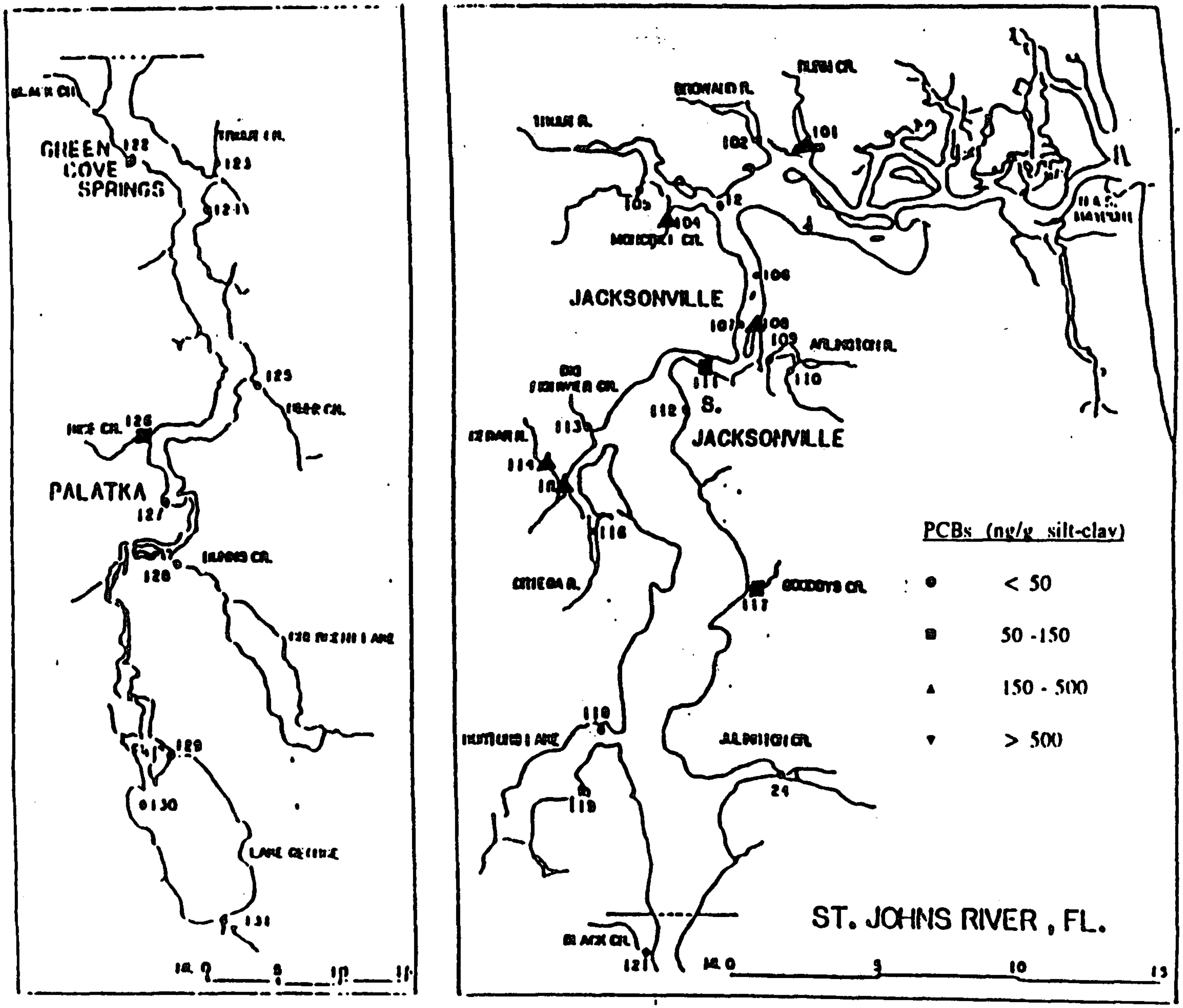


Figure 3.9c Total PCBs distribution in the LSJRB sediments, March 1988 (Pierce et al. 1988).

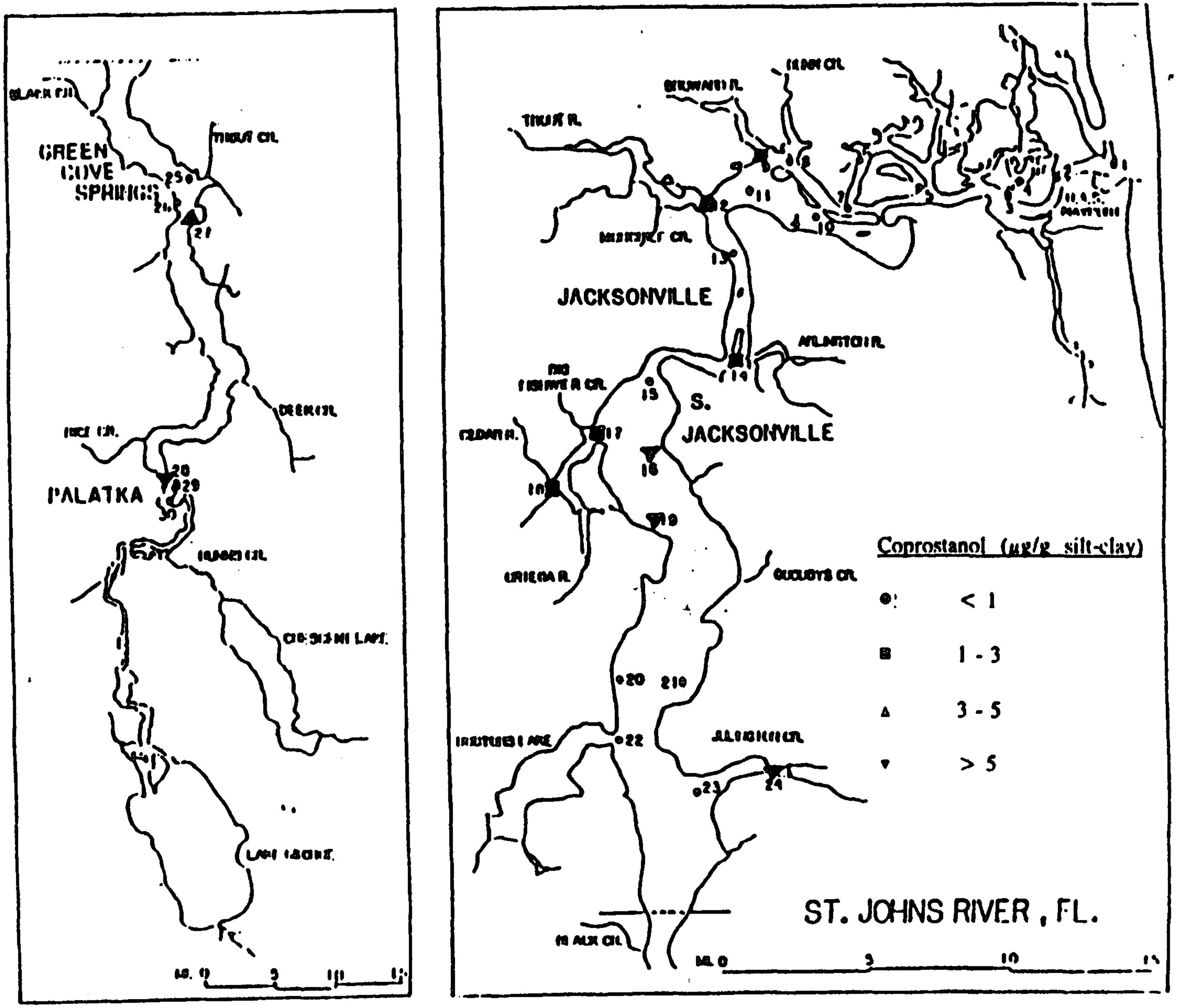


Figure 3.10a Coprostanol distribution in the LSJRB sediments, March 1987 (Pierce et al. 1987).

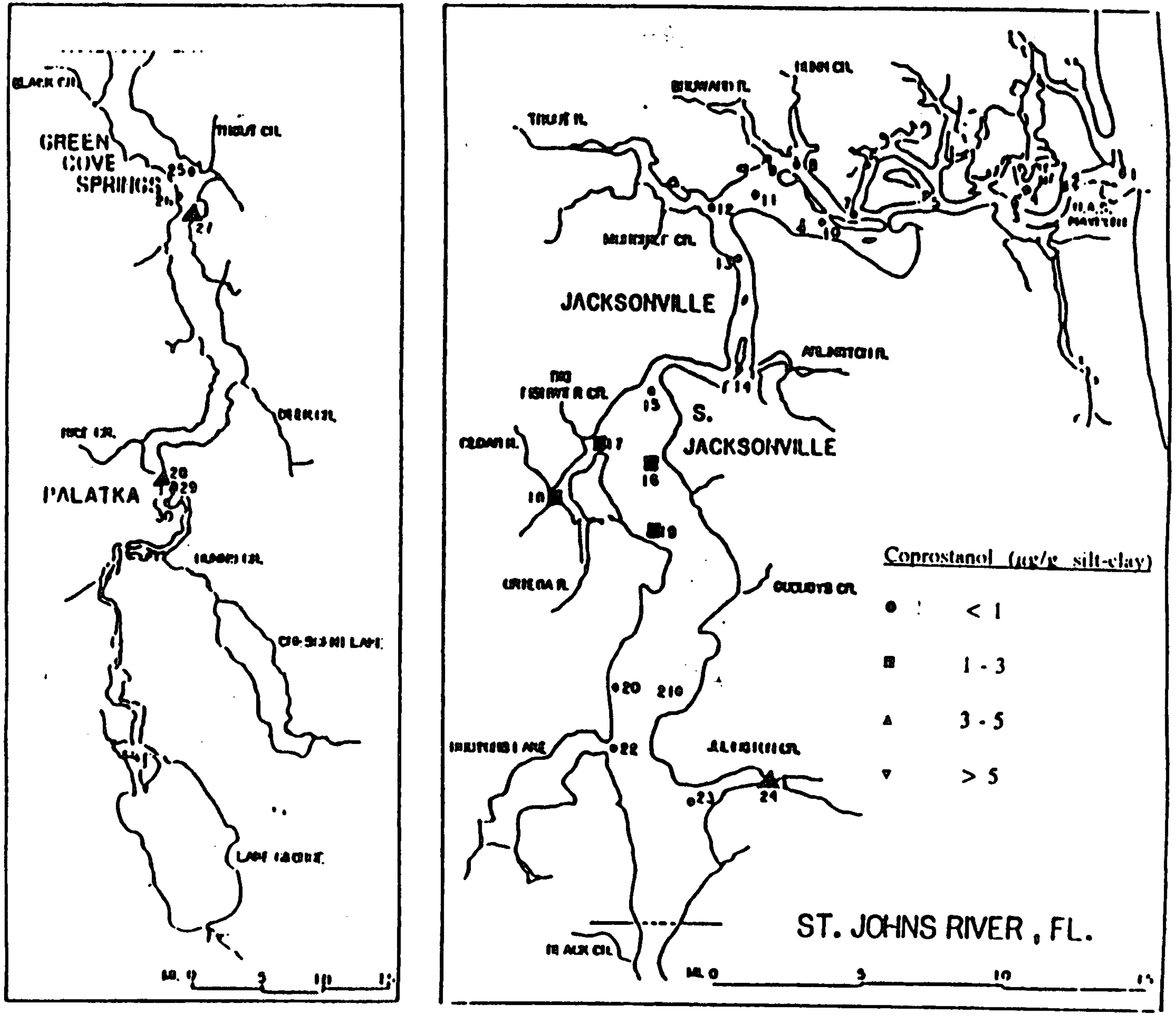


Figure 3.10b Coprostanol distribution in the LSJRB sediments, September 1987 (Pierce et al. 1987).

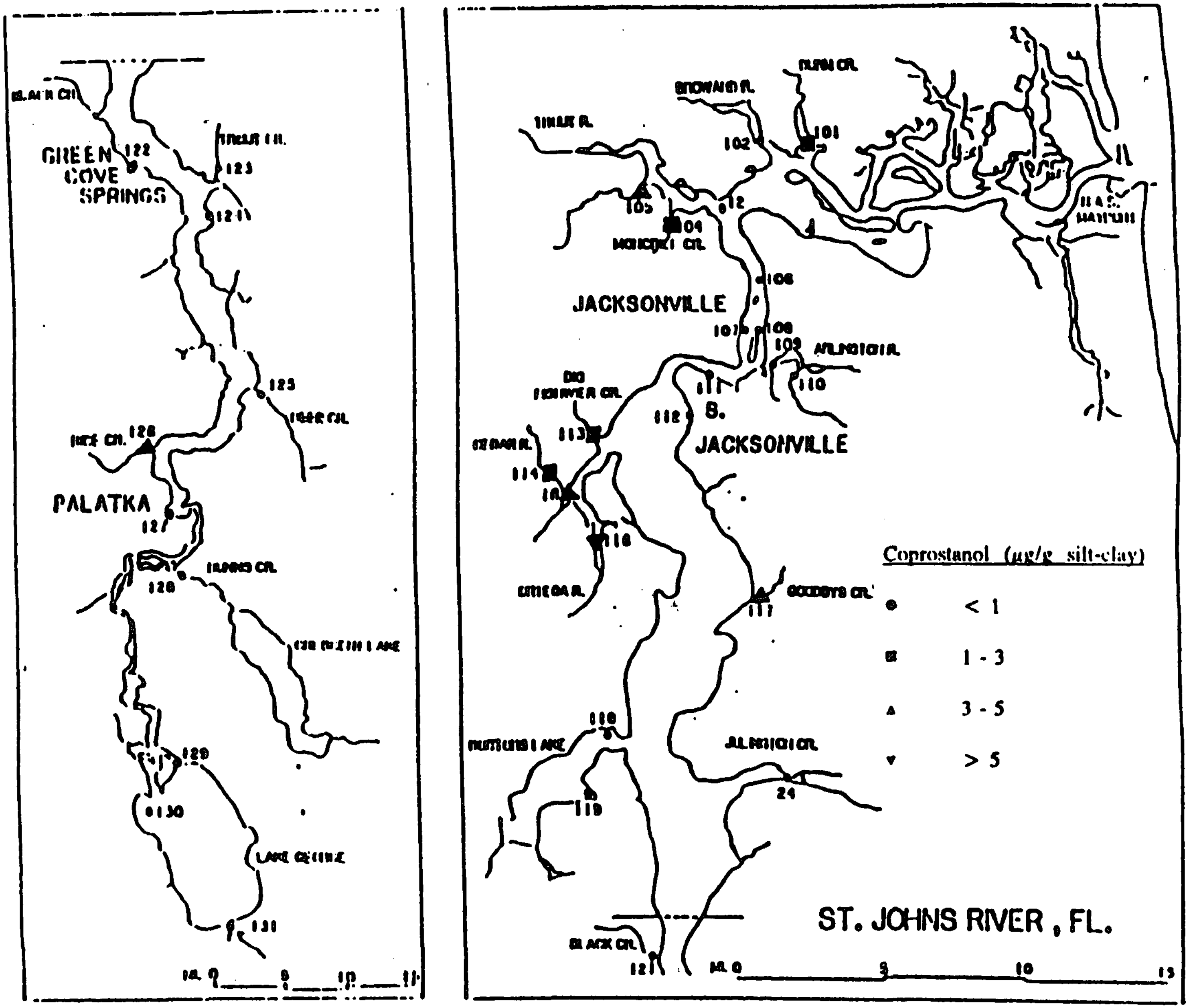


Figure 3.10c Coprostanol distribution in the LSJRB sediments, March 1988 (Pierce et al. 1988).

3.5 Jacksonville Port Authority Study (Savannah Lab. & Environ. Serv. 1988a).

3.5.1 Sampling Stations and Collected Parameters.

The Jacksonville Port Authority Study was conducted January 16-17, 1988 (Savannah Laboratory and Environmental Services, Inc. 1988a). All of the seven sampling stations were in the main stem of the St. Johns River near the Jacksonville Port Authority (Table 3.19). At each station, three ponar and three two-inch core samples were collected and homogenized in a teflon pan with teflon-coated utensils. Then the samples were secured in plastic bottles (if intended for grain size and metal analyses) or in glass containers with teflon-lined caps (for organics). All samples were stored in ice. The sampling procedures used were in accordance with the Deepwater Ports Maintenance Dredging and Disposal Manual, FDER. QC/QA provisions included duplicate analyses (10% of all samples), spiked samples (20% of all samples) and the use of reference materials. The study included analyses of sediment physicochemical characteristics (e.g., particle size, TOC, TKN, etc.), metals and organic pollutants.

Table 3.19 Sampling stations of sediment quality (Savannah Lab. & Environ. Serv. 1988a)

No.	Station	Location	Date
77	Site 1	SJR/Commodore Pt.	01/16-17/88
78	Site 2	JPA/Federal Ch.	01/16-17/88
79	Site 3	Trout R./Reddie Pt.	01/16-17/88
80	Site 4	Fulton-Dame Pt. Cut-Off	01/16-17/88
81	Site 5	Blount Island Channel	01/16-17/88
82	Site 6	SJR/Chaseville Turn	01/16-17/88
83	Site 7	SJR/Brills Cut Range	01/16-17/88

3.5.2 Data Analyses and Results.

Some physicochemical characteristics, i.e., TOC, TKN, total phosphorus, oil/grease, and percentage organics for the sampled stations, are listed in Table 3.20. From this Table, it is evident that high concentrations of organic carbon and nutrients were documented in sites 2, 6, and 7, while low concentrations of these parameters were found at sites 3 and 4.

Table 3.20 General sediment chemistry data (Savannah Lab. & Environ. Serv. 1998a).

Station	TOC mg/kg	TKN mg/kg	Total P mg/kg (dry basis)	Oil/Grease mg/kg	Organic mg/kg
Site 1	37000	3400	610	420	8.8
	41000	3800	560	360	9.6
Site 2	70000	6200	780	710	17.0
	76000	7200	960	680	21.0
Site 3	5500	610	10	260	1.3
	4300	360	83	290	2.1
Site 4	8400	630	68	230	1.9
	9200	490	96	360	2.0
Site 5	43000	3400	310	330	8.3
	47000	3200	270	280	9.7
Site 6	84000	6200	510	220	16.2
	72000	5300	430	230	12.7
Site 7	58000	4200	230	380	13.3
	43000	3200	310	320	12.2

Sediments were analyzed for Al, Ag, As (site 2 only), Cd, Cr, Cu, Hg, Pb, and Zn (Table 3.21). Analyses of the metal:aluminum ratios indicated metal enrichment primarily at site 1 (Cr, Cu, Pb and Zn); site 4 (Cr and Zn); and site 7 (Pb and Zn). All sites were found to have levels of zinc exceeding natural background levels. Metal:aluminum ratio data is given in Table 3.22 and Figures 3.11a-g.

Sediment samples were also analyzed for aliphatic hydrocarbons (Table 3.23), Cl-hydrocarbons (Table 3.24), and PAHs (Table 3.25). Concentrations of aliphatic hydrocarbons ranged from <0.01-13 mg/kg. The sites with the highest degree of

Table 3.21 Metal concentrations in the LSJRB sediments (Savannah Lab. & Environ. Serv. 1988a).

	Parameters in μm/kg dry basis								
Station	**Al**	**Ag**	**As**	**Cd**	**Cr**	**Cu**	**Hg**	**Pb**	**Zn**
Site 1	16000	-	-	0.24	29.0	79.0	0.05	42.0	220
	18000	-	-	0.20	32.0	83.0	0.05	50.0	210
Site 2	28000	0.03	17	0.08	49.0	20.0	0.12	7.8	90
	32000	0.02	23	0.12	56.0	23.0	0.10	6.2	97
Site 3	2800	-	-	0.03	7.4	2.8	0.06	1.3	10
	2300	-	-	0.05	9.3	3.1	0.08	2.1	10
Site 4	2300	-	-	0.05	8.9	2.9	0.03	1.3	12
	2600	-	-	0.08	7.2	3.0	0.03	2.6	17
Site 5	18000	-	-	0.11	28.0	12.0	0.09	16.0	33
	21000	-	-	0.13	36.0	16.0	0.12	21.0	40
Site 6	44000	-	-	0.29	55.0	20.0	0.12	24.0	120
	52000	-	-	0.21	58.0	19.0	0.15	25.0	110
Site 7	15000	-	-	0.07	21.0	6.7	0.07	17.0	40
	11000	-	-	0.05	20.0	5.2	0.05	14.0	32

Table 3.22 Metal:Aluminum ratios in the LSJRB sediments (Savannah Lab. & Environ. Serv. 1988a).

Station	Ag $\times 10^{-4}$	As $\times 10^{-4}$	Cd $\times 10^{-4}$	Cr $\times 10^{-2}$	Cu $\times 10^{-2}$	Hg $\times 10^{-3}$	Pb $\times 10^{-2}$	Zn $\times 10^{-3}$
Site 1	-	-	0.15	0.18*	0.49*	0.003	0.26*	0.14*
	-	-	0.11	0.18*	0.46*	0.003	0.28*	0.12*
Site 2	0.011	6.1	0.03	0.18	0.07	0.004	0.03	0.02*
	0.006	7.2	0.04	0.18	0.07	0.003	0.02	0.03*
Site 3	-	-	0.11	0.26	0.10	0.021	0.05	0.04*
	-	-	0.22	0.40	0.14	0.035	0.09	0.04*
Site 4	-	-	0.22	0.39*	0.13	0.013	0.06	0.05*
	-	-	0.31	0.28*	0.12	0.012	0.10	0.07*
Site 5	-	-	0.06	0.16	0.07	0.005	0.09	0.02*
	-	-	0.06	0.17	0.08	0.006	0.10	0.02*
Site 6	-	-	0.07	0.13	0.05	0.003	0.06	0.03*
	-	-	0.04	0.11	0.04	0.003	0.05	0.02*
Site 7	-	-	0.05	0.14	0.05	0.005	0.11*	0.03*
	-	-	0.05	0.18	0.05	0.005	0.13*	0.03*

Note: (*) denotes enriched metal concentrations.

contamination by aliphatic hydrocarbons were sites 1, 2, 6, and 7. At site 3, all compounds were in concentrations of less than 1.0 mg/kg.

PAHs present included anthracene, benzo(a)anthracene, benzo(b+k)fluoranthene, benzo(g,h,i)perylene, benzo(a)pyrene, chrysene, fluoranthene, phenanthrene, and pyrene, all of which are indicative of fuel combustion products. The most contaminated stations were sites 1, 2 and 6, followed by sites 4, 5 and 7. .Again, site 3 appeared to have the lowest pollutant concentration. Pollution by Cl-pesticides was detected only at site 2.

In conclusion, the sediments appeared to have elevated concentrations of toxic metals (Cr, Pb, and Zn) and organic pollutants due to intensive anthropogenic activities (i.e., industrial and agricultural) in the vicinity of the sampling stations during the last few decades.

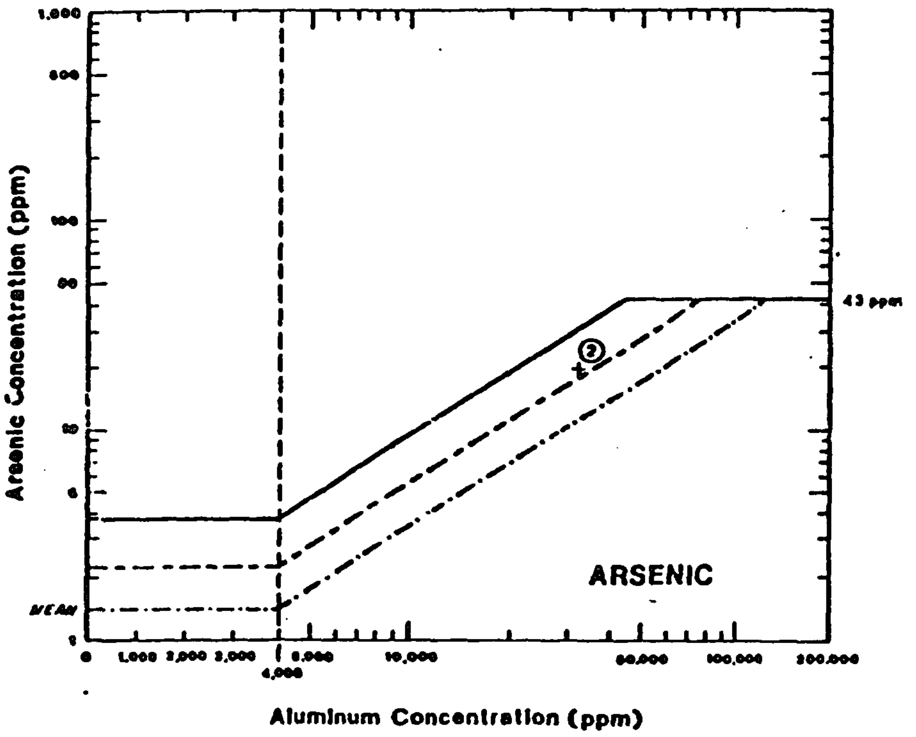


Figure 3.11a As:Al ratio in the LSJRB sediments (Savannah Lab. & Environ. Serv. 1988a).

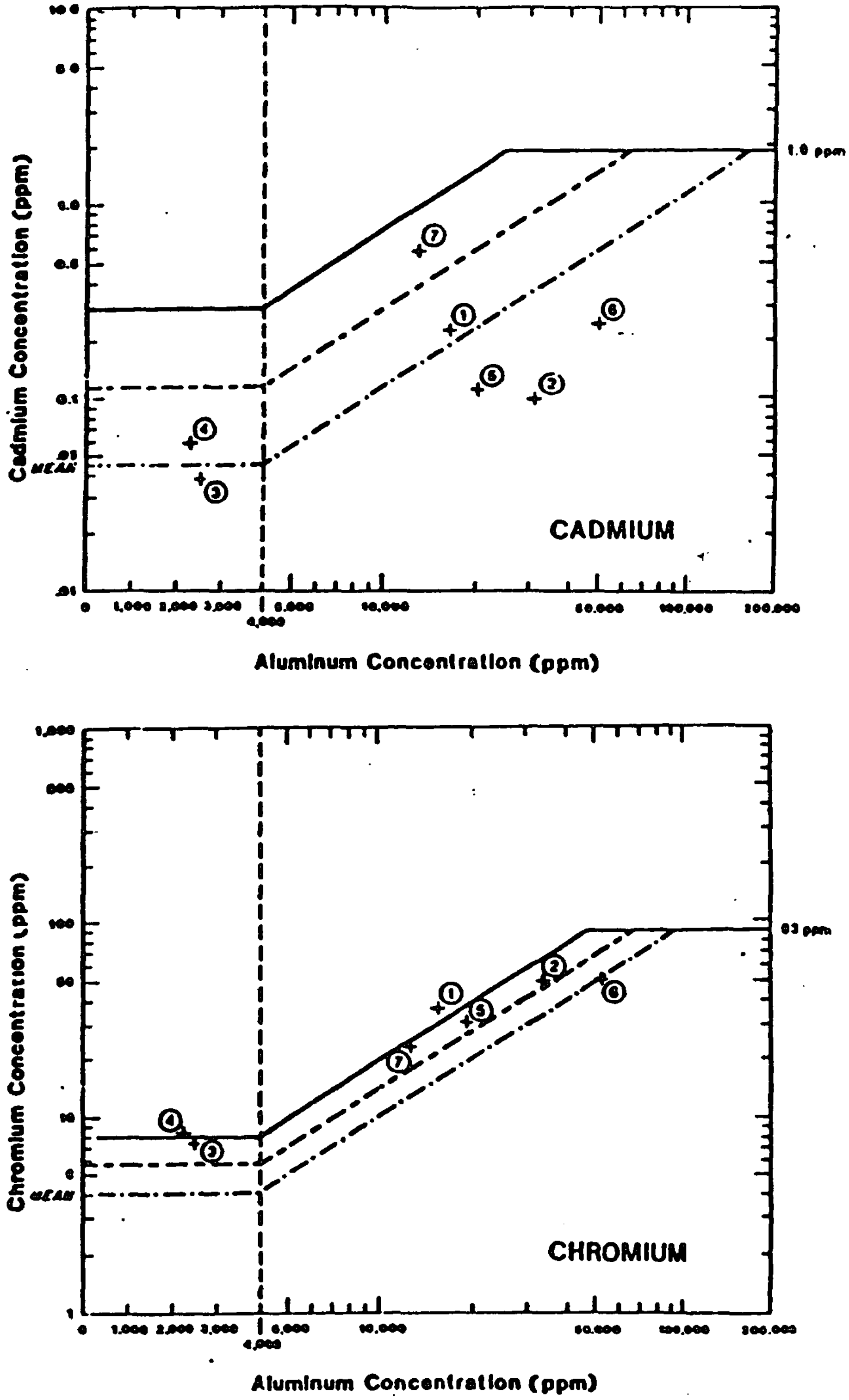


Figure 3.11b Cd:- & Cr:Al ratios in LSJRB sediments (Savannah Lab. & Environ. Serv. 1988a).

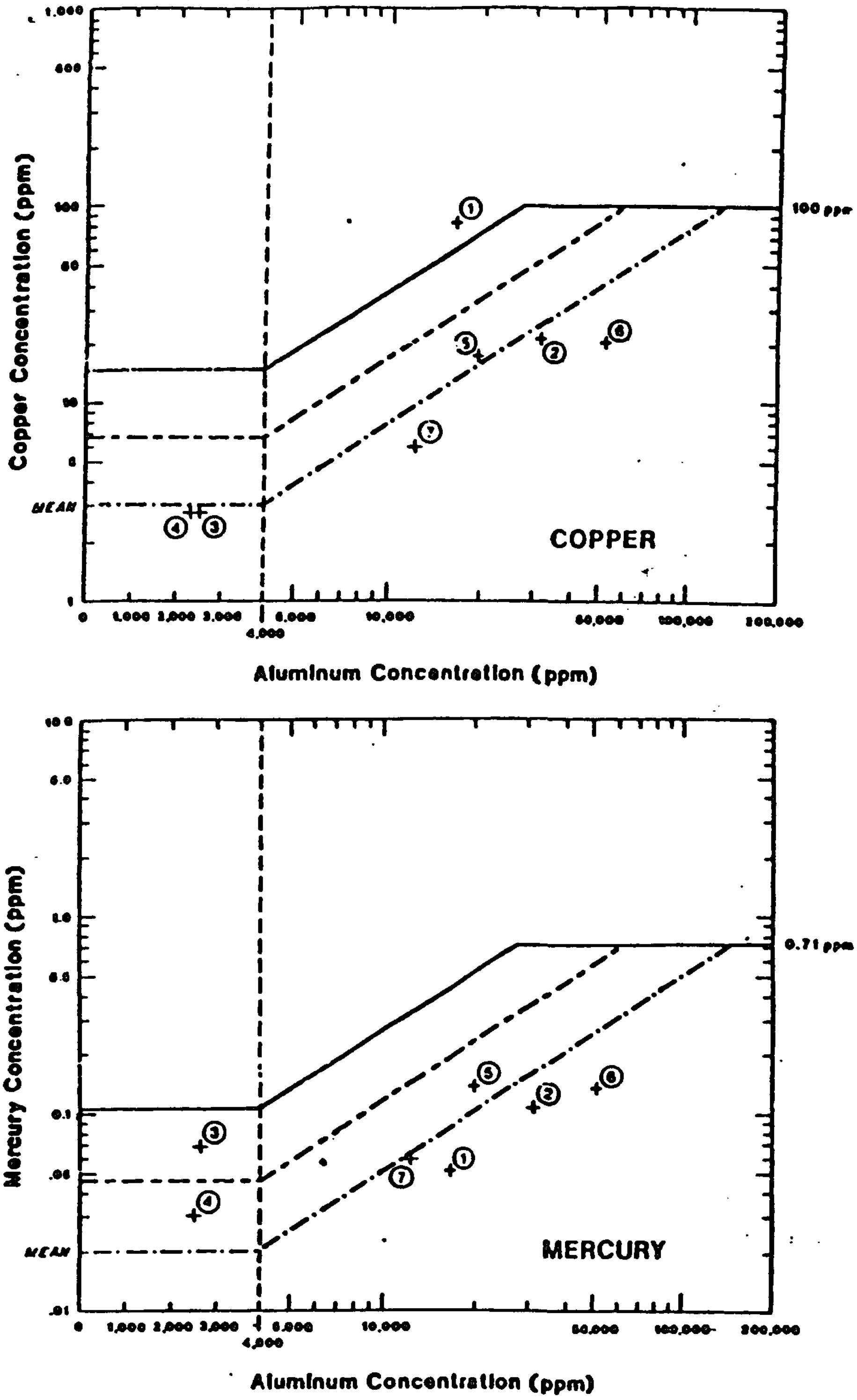


Figure 3.11c Cu:- & Hg:Al ratios in the LSJRB sediments (Savannah Lab. & Environ. Serv. 1988a).

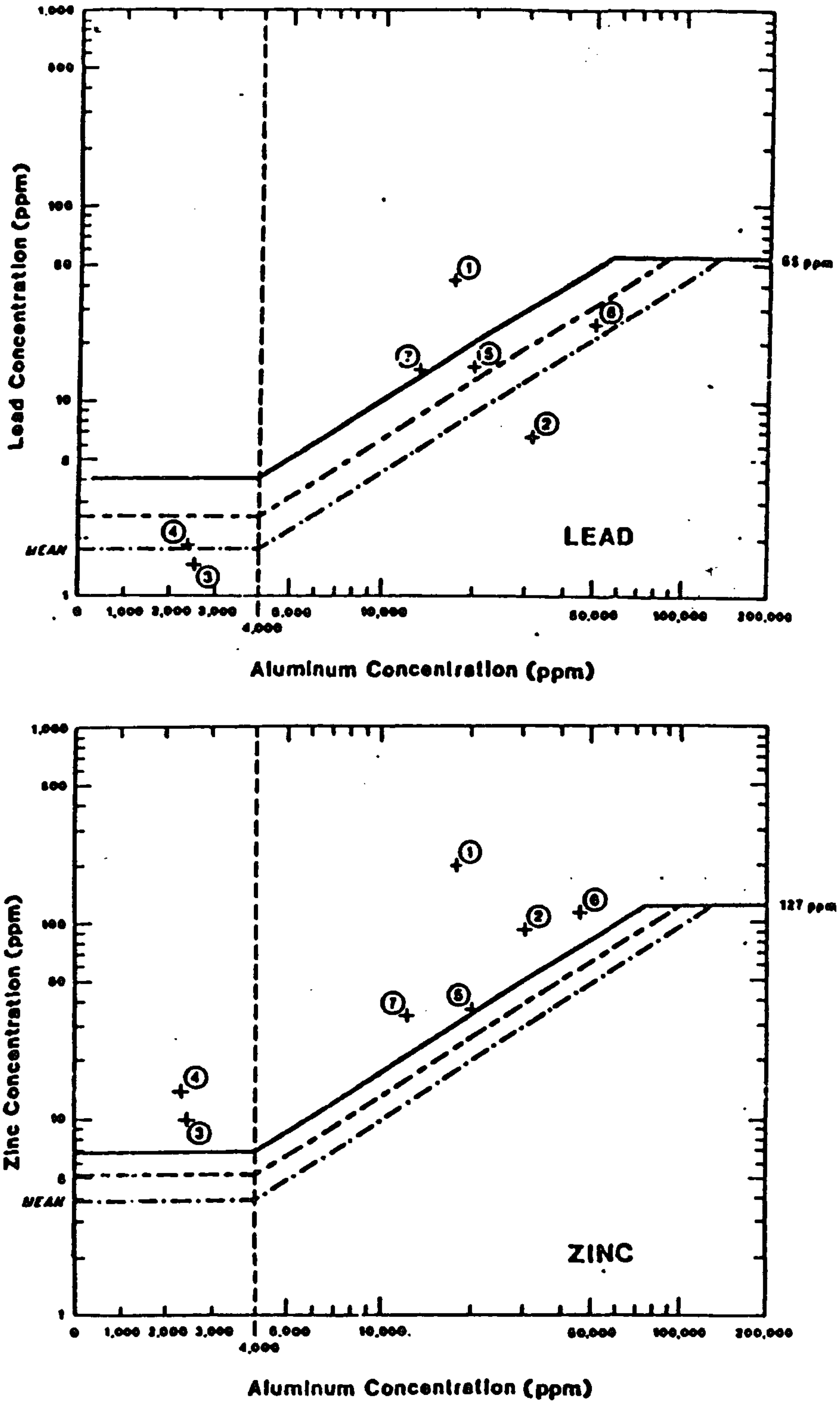


Figure 3.11d Pb:- & Zn:Al ratios in the LSJRB sediments (Savannah Lab. & Environ. Serv. 1988a).

Table 3.23 Aliphatic hydrocarbons in the LSJRB sediments (Savannah Lab. & Environ. Serv. 1988a).

Parameter (mg/kg dry basis)	Site 1	Site 2	Site 3	Station Site 4	Site 5	Site 6	Site 7
C12-C16	1.10	0.63	0.04	0.16	0.21	2.30	0.23
	0.96	0.78	0.07	0.12	0.26	2.10	0.33
C17	1.00	1.70	0.01	0.22	0.32	2.40	1.00
	0.72	2.10	0.01	0.12	0.41	2.70	1.30
Pristane	<0.01	<0.01	<0.01	<0.01	0.02	<0.10	<1.00
	<0.01	<0.01	<0.01	<0.01	0.03	<0.10	<1.00
C18	0.23	0.28	0.01	0.05	0.10	0.40	0.15
	0.29	0.29	0.01	0.07	0.10	0.35	0.12
Phytane	0.12	0.06	0.01	0.02	0.12	0.24	0.13
	0.09	0.01	0.01	0.03	0.13	0.23	0.17
C19-C26	1.80	3.60	0.05	1.60	0.55	3.30	3.50
	1.60	3.90	0.07	1.20	0.59	3.10	2.70
C26-C34	3.20	5.50	0.72	4.20	4.10	8.90	13.00
	2.70	6.30	0.60	3.70	4.00	9.00	10.00

Table 3.24 Chlorinated hydrocarbons in the LSJRB sediments (Savannah Lab. & Environ. Serv. 1988a).

			Station: Site 2		
			mg/kg dry basis		
Aldrin	<0.01	4,4'-DDT	<0.05	Toxaphene	<0.10
	<0.01		<0.05		<0.10
Alpha-BHC	<0.01	Dieldrin I	<0.02	PCB-1016	<0.50
	<0.01		<0.02		<0.50
Beta-BHC	<0.04	Endosulfan I	<0.02	PCB-1221	<0.50
	<0.04		<0.02		<0.50
Delta-BHC	<0.01	Endosulfan II	<0.05	PCB-1232	<0.50
	<0.01		<0.05		<0.50
Gamma-BHC	<0.01	Endosulfan Sulfate	<0.10	PCB-1242	<0.50
	<0.01		<0.10		<0.50
Chlordane	<0.20	Endrin	<0.02	PCB-1248	<0.50
	<0.20		<0.02		<0.50
4,4'-DDD	<0.02	Endrin Aldehyde	<0.10	PCB-1254	<0.50
	<0.02		<0.10		<0.50
4,4'-DDE	<0.02	Heptachlor	<0.10	PCB-1260	<0.50
	<0.02		<0.10		<0.50
		Heptachlor Epoxide	<0.02		
			<0.02		

Table 3.25 PAHs in the LSJRB sediments (Savannah Lab. & Environ. Serv. 1988a).

Parameter (mg/kg dry basis)				Station			
	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6	Site 7
Acenaphthene	<0.01	<0.10	<0.01	<0.01	<0.01	<0.10	<0.10
	<0.01	<0.10	<0.01	<0.01	<0.01	<0.10	<0.10
Acenaphthylene	<0.01	<0.10	<0.01	<0.01	<0.01	<0.10	<0.10
	<0.01	<0.10	<0.01	<0.01	<0.01	<0.10	<0.10
Anthracene	0.09	0.63	<0.01	0.06	0.06	0.23	<0.10
	0.11	0.78	<0.01	0.08	0.07	0.27	<0.10
Benzo(a)Anthracene	<0.01	0.60	<0.01	0.22	<0.01	0.42	0.16
	<0.01	0.72	<0.01	0.33	<0.01	0.37	0.18
Benzo(b+k)Fluoranthene	0.15	1.10	0.04	0.42	0.05	0.21	0.23
	0.19	0.96	0.04	0.61	0.03	0.35	0.27
Benzo(g,h,i)Perylene	0.01	0.16	<0.01	<0.10	<0.01	<0.10	<0.10
	0.02	0.23	<0.01	<0.10	<0.01	<0.10	<0.10
Benzo(a)Pyrene	0.26	0.50	0.01	0.12	0.05	0.09	0.12
	0.11	0.48	0.01	0.13	0.03	0.05	0.15
Chrysene	0.46	0.72	<0.01	0.23	0.36	0.40	0.13
	0.61	0.78	<0.01	0.32	0.30	0.32	0.11
Dibenzo(a,h)Anthracene	0.04	<0.10	<0.01	<0.10	<0.01	0.12	0.10
	0.09	<0.10	<0.01	<0.10	<0.01	0.10	0.17
Dibenzothiophene	0.02	0.19	<0.01	<0.10	<0.01	<0.10	<0.10
	0.02	0.26	<0.01	<0.10	<0.01	<0.10	<0.10
Fluoranthene	0.56	2.10	0.04	0.40	0.18	0.83	0.34
	0.42	2.30	0.05	0.55	0.19	0.97	0.39
Fluorene	0.08	0.59	0.02	0.03	0.03	0.08	0.13
	0.09	0.76	0.03	0.04	0.05	0.12	0.17
Indedo(1,2,3-cd)Pyrene	<0.01	0.15	<0.01	0.13	<0.01	0.16	0.13
	<0.01	0.19	<0.01	0.12	<0.01	0.05	0.18
2-Methylnaphthalene	0.01	<0.10	<0.01	<0.10	-	<0.01	<0.01
	0.01	<0.10	<0.01	<0.10	-	<0.01	<0.10
Naphthalene	0.01	<0.10	<0.01	<0.01	<0.01	<0.10	<0.10
	0.01	<0.10	<0.01	<0.01	<0.01	<0.10	<0.10
Phenanthrene	0.43	2.60	0.02	0.16	0.18	0.32	0.19
	0.41	2.74	0.03	0.18	0.10	0.46	0.17
Pyrene	0.51	1.50	0.03	0.33	0.21	1.20	0.27
	0.43	2.10	0.04	0.42	0.13	1.30	0.38

3.6 Coastal Zone Management, FDER, Study (Savannah Lab. & Environ. Serv. 1988b).

3.6.1 Sampling Locations and Collected Parameters. Another major sediment quality study was conducted by the Coastal Zone Management Section, FDER (Savannah Laboratory and Environmental Services, Inc. 1988b). This study extended from the

Amelia River downstream to Cedar Creek upstream and included thirty-eight sampling locations. It used six of the stations sampled by the Dames and Moore Study in 1982 (Table 3.26, Figures 3.12a-c). Savannah Laboratories and Environmental Services, Inc., collected and analyzed the sediment samples following the guidelines set by the Deepwater Ports Maintenance Dredging and Disposal Manual, FDER. The samples were analyzed for toxic metals (Table 3.28); Cl-pesticides and arochlor (Table 3.30); and PAHs, aliphatics and other organics (Table 3.31).

3.6.2 Data Analyses and Results. Qualitative observations of the sediments revealed that the bottom was comprised of black or brown silty clay at most of the stations (Table 3.27). The only stations with predominantly sandy sediments were in Goodbys Lake, off North Buck Island, in Clapboard Creek, and in the main river (SJR 29). Enrichment in toxic metals was determined using the metal:aluminum ratios, and the polluted sites are shown in Table 3.29. From the metal:aluminum ratios, it is evident that high levels of Cd, Hg, Pb, Zn, and Cu to a lesser extent, were present at twenty sampling sites (Calder and Sloan, 1992). Enrichment in four toxic metals was documented at the mouth of the Arlington (Cd, Hg, Pb, Zn) and Julington Rivers (Cu, Hg, Pb, Zn), in Black Creek (Cd, Hg, Pb, Zn), Cedar Creek (Cd, Hg, Pb, Zn), Bluff Branch Creek (Cd, Hg, Pb, Zn), at the Jacksonville shipyards (Cu, Hg, Pb, Zn), off the Naval Air Station (Cu, Hg, Pb, Zn), and near Hendricks Point (Cd, Hg, Pb, Zn). Enrichment in three metals was observed at McCoys Creek (Cu, Pb, Zn), the Ortega River (Cd, Hg, Zn), Doctors Lake (Cd, Hg, Pb), Popo Point (Hg, Pb, Zn), and off Green Cove Springs (Hg, Pb, Zn).

In analyses for organic pollutants, traces of Cl-pesticides (especially chlordane), arochlors, and toxaphene were detected at fifteen stations (Table 3.30). Elevated

Table 3.26 Sampling stations of sediment quality (Savannah Lab. & Environ. Serv., 1988b).

No.	Station	Location	Latitude	Longitude	Date
84	AMR 1	Amelia River	30 40 51 N	81 27 43 W	12/06/84
85	AMR 2	Amelia River	30 39 49 N	81 28 25 W	12/06/84
86	BWC 1	Broward Creek	30 27 27 N	81 27 11 W	09/11/86
87	FGR 1	Ft. George River	30 26 34 N	81 26 14 W	09/11/86
88	MCV 1	Mill Cove	30 22 05 N	81 31 44 W	09/11/86 12/07/88
89	MCV 2	Mill Cove	30 23 11 N	81 35 26 W	09/11/86 12/07/88
90	NSR 1	Nassau River	30 32 32 N	81 28 39 W	09/11/86
91	PCB 1	Pablo Creek	30 19 27 N	81 26 25 W	12/07/84
92	PCB 2	Pablo Creek	30 19 17 N	81 26 17 W	12/07/84
93	PCB 3	Pablo Creek	30 17 19 N	81 25 12 W	12/07/84
94	PCB 4	Pablo Creek	30 14 40 N	81 25 37 W	12/07/84
95	SJR 1	Goodbys Lake	30 17 34 N	81 40 29 W	12/06/88
96	SJR 3	SJR/N. Mathews B.	30 20 52 N	81 37 08 W	12/06/88
97	SJR 4	SJR/Long Branch	30 22 15 N	81 38 00 W	12/06/88
98	SJR 8	SJR/N. Buck Is.	30 23 17 N	81 28 42 W	12/07/88
99	SJR 11	SJR/Talleyrand	30 21 04 N	81 37 13 W	12/06/88
100	SJR 14	SJR/Blount Is.	30 24 16 N	81 32 55 W	12/07/88
101	SJR 16	SJR/Cedar Creek	29 48 41 N	81 35 01 W	12/08/88
102	SJR 17	SJR/Green Cove S.	29 59 19 N	81 39 00 W	12/08/88
103	SJR 18	SJR/Popo Point	30 02 08 N	81 40 15 W	12/08/88
104	SJR 19	SJR/Black Creek	30 02 32 N	81 42 27 W	12/08/88
105	SJR 20	Julington Creek	30 07 48 N	81 37 55 W	12/07/88
106	SJR 21	SJR/Julington C.	30 08 01 N	81 39 50 W	12/07/88
107	SJR 22	Doctor's Lake	30 09 01 N	81 42 35 W	12/07/88
108	SJR 23	SJR/S.E. of NAS	30 12 56 N	81 39 55 W	12/05/88
109	SJR 24	SJR/E. of NAS	30 14 11 N	81 39 37 W	12/05/88
110	SJR 25	SJR/Bluff Branch	29 45 20 N	81 33 49 W	12/08/88
111	SJR 26	SJR/Ortega River	30 17 23 N	81 41 29 W	12/06/88
112	SJR 27	Ortega River	30 17 47 N	81 41 43 W	12/06/88
113	SJR 28	SJR/McCoys Cr.	30 19 20 N	81 40 03 W	12/06/88
114	SJR 29	St. Johns River	30 19 04 N	81 39 05 W	12/06/88
115	SJR 30	SJR/Jax. S'yards	30 19 22 N	81 39 08 W	12/06/88
116	SJR 31	Arlington River	30 19 04 N	81 36 47 W	12/06/88
117	SJR 32	Trout River	30 23 34 N	81 38 28 W	12/06/88
118	SJR 33	Broward River	30 25 02 N	81 36 19 W	12/07/88
119	SJR 34	SJR/Clapboard C.	30 24 25 N	81 30 48 W	12/07/88
120	SJR 35	Clapboard Cr.	30 24 54 N	81 30 08 W	12/07/88
121	SJR 36	SJR/S.E. Ortega	30 16 41 N	81 41 30 W	12/06/88

concentrations of PAHs and aliphatics were found at Cedar Creek, Green Cove Springs, Popo Point, Black Creek, Julington Creek, and Bluff Branch. PAHs and aliphatics were present to a lesser degree in Mill Cove, Doctors Lake, and near the Naval Air Station

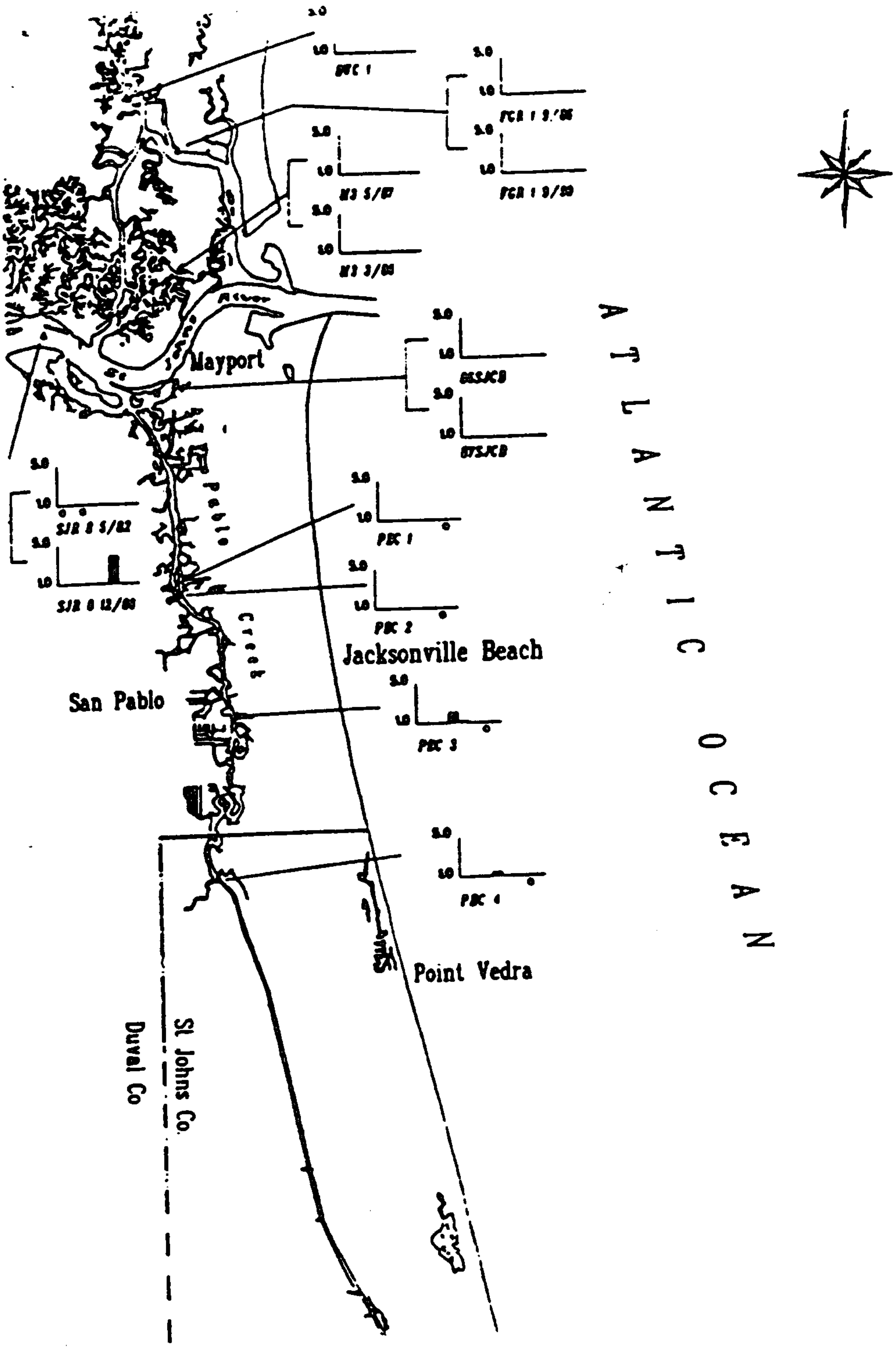


Figure 3.12a Sampling stations of sediment quality (Savannah Lab. & Environ. Serv. 1988b).

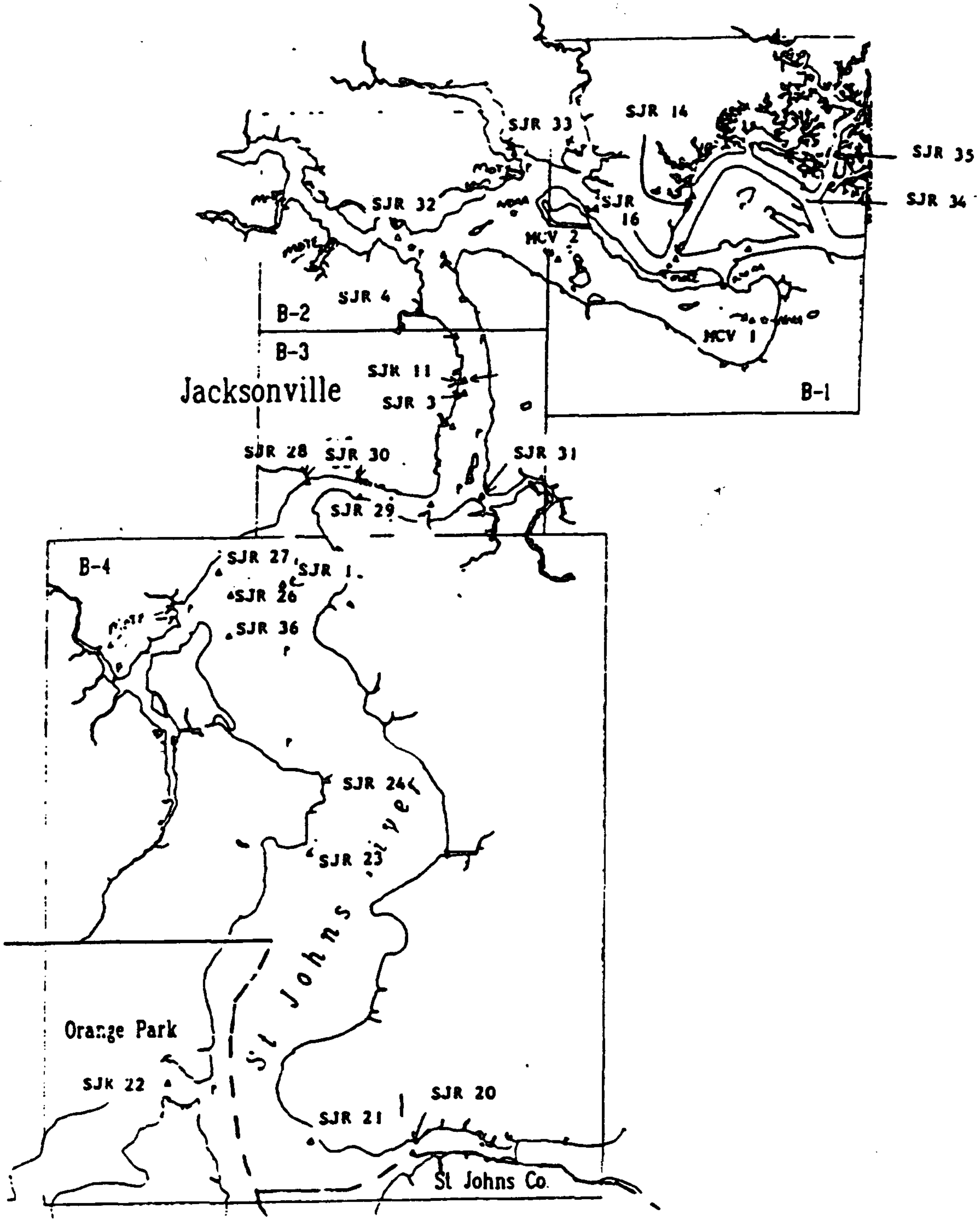


Figure 3.12b Sampling stations of sediment quality (Savannah Lab. & Environ. Serv. 1988b).

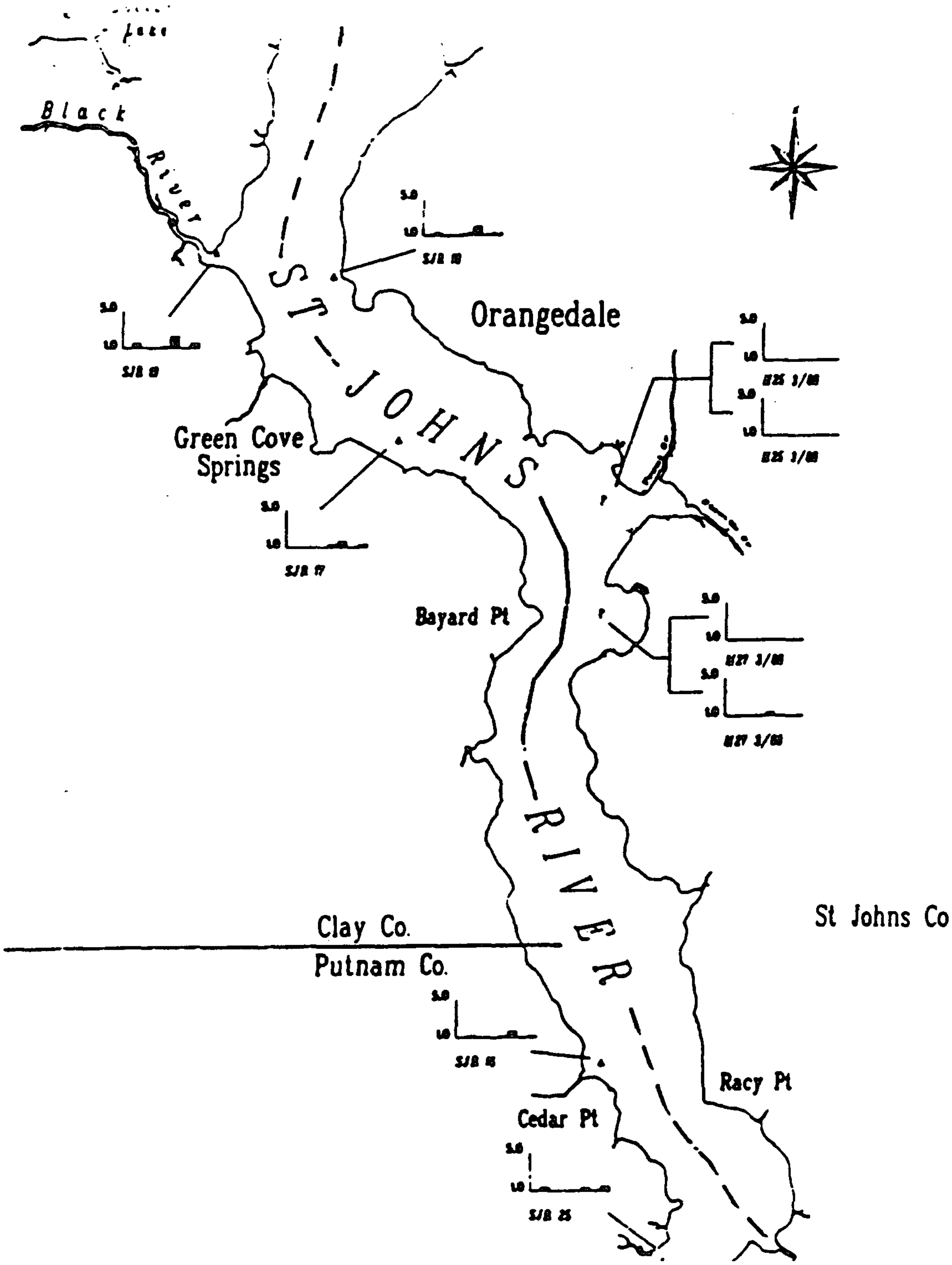


Figure 3.12c Sampling stations of sediment quality (Savannah Lab. & Environ. Serv. 1988b).

Table 3.27 Physical description of the LSJRB sediments (Savannah Lab. & Environ. Serv. 1988b).

Station	Qualitative sediment description
MCV 1	Black/brown silt with few shell fragments; H_2S odor
MCV 2	Brown/black silt with sand and shell fragments; live snails
SJR 1	Black and brown coarse sand
SJR 3	Black silty clay with layers of brown sand
SJR 4	Silt (brown) top, black clay below
SJR 8	Medium brown sand with shell fragments
SJR 14	Brown silt top, grey silty clay below; few fine shell fragments
SJR 16	Black silty clay
SJR 17	Black silt with live clams
SJR 18	Brown/black silt with shell fragments and live clams
SJR 19	Brown/black silty clay with peat layers and chunks
SJR 20	Brown silty clay with few shell fragments; few live clams
SJR 21	Black/brown silty clay
SJR 22	Black silty clay
SJR 23	Soft black ooze mud
SJR 24	Black ooze mud-silty sand
SJR 25	Black silty clay
SJR 26	Black silty clay with numerous live clams and shell fragments
SJR 27	Black silty clay with few live clams and clam shells
SJR 29	Black silty sand
SJR 31	Consolidated brown silt
SJR 32	Consolidated silt/clay with shell fragments and sand
SJR 33	Brown silty sand over blackish green clay; few shells; slightly H_2S
SJR 34	Brown silt on top with black silty sand below
SJR 35	Silty sand with black/brown macro algae and roots
SJR 36	Black/grey silty clay with many dead and several live clams

(Tables 3.31, 3.31a).

In summary, Cedar Creek (SJR 16), Green Cove Springs (SJR 17), Popo Point (SJR 18), Black Creek (SJR 19), Julington Creek (SJR 20 and SJR 21), Doctors Lake (SJR 22), the Naval Air Station (SJR 24) and Bluff Branch (SJR 25) were the sites most contaminated by both toxic metals and organic pollutants. All of these sites are located in the middle part of the Lower St. Johns River Basin, where human activities are the most concentrated.

Table 3.28 Metal concentrations in the LSJRB sediments (Savannah Lab. & Environ. Serv. 1988b).

		Parameter									
Station		Al	As	Cd	Cr	Cu	Hg	Ni	Pb	Zn	TOC
SJR 1	Mean	2130	-	0.070	3.00	6.70	0.043	2.20	3.05	5.50	1450
	S.D.	382	-	0.005	0.28	6.08	-	0.71	0.21	0.85	71
SJR 3	Mean	40000	7.150	0.430	38.00	23.50	0.210	8.95	23.00	85.50	25500
	S.D.	8485	1.202	0	7.07	4.95	0.028	1.48	8.48	17.68	17678
SJR 4	Mean	51000	9.150	0.445	48.00	25.00	0.225	12.00	26.00	110.00	15000
	S.D.	1414	0.778	0.049	0	1.41	0.007	0	1.41	0	2828
SJR 8	Mean	450	-	0.056	1.35	0.72	-	-	5.70	1.60	300
	S.D.	14	-	0.020	0.21	0.11	-	-	0.14	0.14	0
SJR 11	Mean	41000	7.150	0.400	39.50	26.00	0.130	9.60	28.00	98.00	16000
	S.D.	12728	2.333	0.990	7.78	2.83	0.014	0.57	0	16.98	0
SJR 14	Mean	20500	6.650	0.290	20.50	11.35	-	5.45	15.50	38.50	6800
	S.D.	4950	1.909	0.014	0.71	2.33	-	0.07	0.71	2.12	3677
SJR 16	Mean	13500	6.400	0.645	29.00	11.50	0.245	9.05	28.50	49.50	23500
	S.D.	707	0.849	0.007	2.83	2.12	0.035	1.34	2.12	7.78	2121
SJR 17	Mean	18000	7.250	0.495	36.50	17.00	0.275	9.45	35.00	35.00	31000
	S.D.	1414	0.071	0.007	3.54	1.41	0.007	0.35	1.41	4.24	9899
SJR 18	Mean	24500	9.050	0.815	48.50	18.00	0.260	12.00	51.50	82.50	28500
	S.D.	707	1.344	0.035	0.71	1.41	0.057	1.41	0.71	10.61	12021
SJR 19	Mean	18500	5.250	0.890	32.50	15.00	0.240	9.20	49.50	84.50	30000
	S.D.	2121	0.071	0.297	3.54	0	0	0	4.95	0.71	0
SJR 20	Mean	36000	4.200	0.955	35.50	17.50	0.380	8.15	44.00	75.50	26000
	S.D.	0	0.990	0.205	0.71	2.12	0.028	0.64	2.83	0.71	1414
SJR 21	Mean	41000	8.500	0.870	48.50	22.50	0.340	10.00	62.00	104.00	25500
	S.D.	9899	1.414	0.042	3.54	2.12	0.014	0	5.66	8.49	707
SJR 22	Mean	31000	5.950	0.375	29.00	11.25	0.140	6.80	39.50	45.50	21000
	S.D.	4243	2.192	0.163	1.41	2.47	0.014	0.57	7.78	10.61	0
SJR 23	Mean	31000	6.600	0.092	44.50	21.00	0.295	9.80	50.00	115.00	22000
	S.D.	15556	0.283	0.113	6.36	1.41	0.064	1.70	8.49	7.07	4243
SJR 24	Mean	25000	8.800	1.100	56.00	23.00	0.270	13.50	28.50	107.00	26500
	S.D.	1414	0.849	0	0	1.41	0	0.71	2.12	4.24	2121
SJR 25	Mean	9350	7.900	0.675	27.00	16.00	0.220	11.00	17.70	49.50	25000
	S.D.	919	0.566	0.148	0	1.41	0.014	0.20	0.22	2.12	1414
SJR 26	Mean	48000	8.050	0.935	49.00	34.00	0.275	10.75	39.50	140.00	23500
	S.D.	8485	0.778	0.233	8.49	2.83	0.092	1.77	4.95	14.14	9192
SJR 27	Mean	42000	8.750	0.725	48.50	32.50	0.305	12.00	36.50	130.00	25500
	S.D.	5657	1.202	0.021	2.12	0.71	0.021	1.41	4.95	0	3536
SJR 28	Mean	56000	9.350	0.730	51.00	54.50	0.710	13.00	81.50	150.00	16500
	S.D.	14142	0.212	0.156	1.41	33.23	0.014	0	0.71	14.40	707
SJR 29	Mean	47000	6.150	0.435	42.50	44.50	0.265	10.80	54.00	120.00	18500
	S.D.	7071	1.626	0.064	9.19	6.36	0.064	3.11	14.14	0	707
SJR 30	Mean	49000	8.500	0.565	51.00	85.00	0.250	27.00	69.00	165.00	17000
	S.D.	7071	0.283	0.064	1.41	49.50	0.014	19.80	1.41	7.07	1414
SJR 31	Mean	40000	7.250	1.000	45.50	31.00	0.380	11.10	47.00	145.00	25500
	S.D.	8485	0.636	0	9.19	7.07	0.014	2.69	46.67	49.50	6364
SJR 32	Mean	28000	3.300	0.220	25.50	4.70	0.062	5.20	6.10	27.00	14500
	S.D.	8485	0.424	0.057	4.95	0.14	0.002	0.42	0.99	1.41	707
SJR 33	Mean	21500	5.200	0.240	24.50	4.65	-	4.65	10.15	26.50	4150
	S.D.	6364	0.566	0.170	13.44	0.21	-	2.33	2.62	0.71	2051
SJR 34	Mean	14000	4.700	0.195	12.50	5.20	-	2.65	8.80	20.00	8000
	S.D.	2828	1.273	0.007	0.71	0.57	-	0.21	0.57	0	1980
SJR 35	Mean	20300	-	0.057	4.30	2.05	-	0.69	7.70	5.50	4300
	S.D.	2203	-	0.008	0.57	0.35	-	0.09	0.28	0.99	707
SJR 36	Mean	39000	8.000	1.750	53.00	27.50	0.485	10.30	44.00	108.50	20000
	S.D.	1414	0.849	0.636	1.41	2.12	0.134	0.99	15.56	16.26	4243

Table 3.29 Metal:Aluminum ratios in the LSJRB sediments (Savannah Lab. & Environ. Serv. 1988b).

Station	As $x10^{-4}$	Cd $x10^{-4}$	Cr $x10^{2}$	Cu $x10^{2}$	Hg $x10^{3}$	Ni $x10^{3}$	Pb $x10^{2}$	Zn $x10^{3}$
SJR 1	-	0.33	0.14	0.31	0.020	1.03	0.14	2.58
SJR 3	1.79	0.11	0.10	0.06	0.005*	0.22	0.06	2.14
SJR 4	1.79	0.09	0.09	0.04	0.004*	0.24	0.05	2.16
SJR 8	-	1.24	0.30	0.16	-	-	1.27*	3.56
SJR 11	1.74	0.10	0.10	0.06	0.003	0.23	0.07	2.39
SJR 14	3.24	0.14	0.10	0.06	-	0.27	0.08	1.88
SJR 16	4.74	0.48*	0.21	0.09	0.018*	0.67	0.21*	3.67*
SJR 17	4.03	0.28	0.20	0.09	0.015*	0.53	0.19*	3.83*
SJR 18	3.69	0.33	0.20	0.07	0.011*	0.49	0.21*	3.37*
SJR 19	2.84	0.48*	0.18	0.08	0.013*	0.50	0.27*	4.57*
SJR 20	1.17	0.27*	0.10	0.05	0.011*	0.23	0.12*	2.10
SJR 21	2.07	0.21*	0.12	0.05	0.008*	0.24	0.15*	2.54*
SJR 22	1.92	0.12*	0.09	0.04	0.005*	0.22	0.13*	1.47
SJR 23	2.13	0.03*	0.14	0.07	0.010*	0.32	0.16*	3.71*
SJR 24	3.52	0.44*	0.22	0.09	0.011*	0.54	0.11*	4.28*
SJR 25	8.45	0.73*	0.29	0.17	0.024*	1.18	0.19*	5.29*
SJR 26	1.68	0.19*	0.10	0.10	0.006*	0.67	0.08	2.92*
SJR 27	2.08	0.17	0.12	0.08	0.007*	0.29	0.09	3.10*
SJR 28	1.67	0.13	0.09	0.10*	0.003	0.23	0.15*	2.68*
SJR 29	1.31	0.09	0.09	0.09*	0.006*	0.23	0.11*	2.55*
SJR 30	1.73	0.12	0.10	0.17*	0.005*	0.55	0.14*	3.37*
SJR 31	1.81	0.25*	0.11	0.08	0.010*	0.28	0.12*	3.63*
SJR 32	1.18	0.08	0.09	0.02	0.002	0.19	0.02	0.96
SJR 33	2.42	0.11	0.11	0.02	-	0.22	0.05	1.23
SJR 34	3.36	0.14	0.08	0.02	-	0.19	0.06	1.43
SJR 35	-	0.03	0.02	0.01	-	0.03	0.04	0.27
SJR 36	2.05	0.45*	0.14	0.07	0.012*	0.26	0.11*	2.78*

Note: (*) denotes metal enrichment.

Table 3.30 Cl-pesticides, toxaphene, and arochlor in sediments (Savannah Lab. & Environ. Serv. 1988b).

Stations
MCV-1,-2; SJR-8, -14, -16, -17, -18, -19, -20, -21, -22, -25, -33, -34, -35

Parameter
(μg/kg dry basis)

Aldrin	< 1	Endrin	< 2
alpha-BHC	< 1	Endrin aldehyde	< 5
beta-BHC	< 1	Heptachlor	< 1
gamma-BHC	< 1	Heptachlor epoxide	< 2
delta-BHC	< 1	Toxaphene	<20
Chlordane	<10	Arochlor-1016	< 5
4,4'-DDD	< 2	Arochlor-1221	< 5
4,4'-DDE	< 2	Arochlor-1232	< 5
4,4'-DDT	< 5	Arochlor-1242	< 5
Dieldrin	< 2	Arochlor-1248	< 5
Endosulfan I	< 2	Arochlor-1254	< 5
Endosulfan II	< 5	Arochlor-1260	< 5
Endosulfan sulfate	< 5		

Table 3.31 PAHs, aliphatics and other OPPs in sediments (Savannah Lab. & Environ. Serv. 1988b).

Parameter				Station				
(μg/kg dry basis)	MCV 1	MCV 2	SJR 8	SJR 14	SJR 16	SJR 17	SJR 18	SJR 19
Acenaphthene	71	<20	<20	33	<400	620	<400	<400
	56	<20	<20	29	<400	<400	<400	<400
Acenaphthylene	<15	<15	<15	65	<100	<100	<100	<100
	<15	<15	<15	57	<100	<100	<100	<100
Anthracene	10	10	<2	49	<50	<50	<50	<50
	71	5	<2	62	<50	<50	<50	58
Benzo(a)Anthracene	<50	<50	<50	<50	<50	<50	<50	<50
	<50	<50	<50	<50	<50	<50	<50	<50
Benzo(a)Pyrene	<20	<20	<20	<20	430	750	210	890
	<20	<20	<20	<20	310	940	520	710
Benzo(b)Fluoranthene	56	<50	<50	<50	<50	100	120	130
	53	<50	<50	<50	<50	150	220	110
Benzo(g,h,i)Perylene	<50	<50	<50	<50	<100	<100	<100	<100
	56	<50	<50	<50	<100	<100	<100	<100
Benzo(k)Fluoranthene	160	<50	<50	<50	<50	<50	<50	<50
	150	<50	<50	<50	<50	<50	<50	110
Chrysene	36	30	<2	10	<50	<50	<50	<50
	65	14	<2	25	<50	<50	<50	150
Dibenzo (a,h)Anthracene	<150	<150	<150	<150	<550	<550	<550	<550
	<150	<150	<150	<150	<550	<550	<550	<550
Fluoranthene	60	42	<5	38	<50	180	62	<50
	120	50	<5	34	<50	<50	62	610
Fluorene	<1.5	<1.5	<1.5	<1.5	160	<150	<150	<150
	<1.5	<1.5	<1.5	<1.5	<150	<150	<150	<150
Indeno(1,2,3-cd)Pyrene	<50	<50	<50	<50	<50	<50	<50	52
	43	<50	<50	<50	<50	<50	<50	77
Naphthalene	32	63	<10	11	<500	<500	<500	<500
	49	12	<10	25	<500	<500	<500	<500
Pyrene	81	76	<5	36	<50	<50	<50	710
	160	60	<5	30	<50	<50	<50	630
Phenanthrene	11	18	<5	8.2	<50	<50	<50	<50
	26	18	<5	6	<50	75	<50	<50
1-MethylNaphthalene	<150	<150	<150	<150	<400	<400	<400	<400
	<150	<150	<150	<150	<400	<400	<400	<400
2-MethylNaphthalene	<150	<150	<150	<150	<400	<400	<400	<400
	<150	<150	<150	<150	<400	<400	<400	<400
Benzonitrile	<500	<500	<500	<500	<400	<400	<400	<400
	<500	<500	<500	<500	<400	<400	<400	<400
Quinoline	<250	<250	<250	<250	<1500	<1500	710	<1500
	<250	<250	<250	<250	<1500	<1500	820	<1500
Quinaldine	<350	<350	<350	<350	<150	<150	<150	<150
	<350	<350	<350	<350	<150	<150	<150	<150
8-Methylquinaline	<650	<650	<650	<650	<100	<100	<100	<100
	<650	<650	<650	<650	<100	<100	<100	<100
7,8-Benzoquinoline	<650	<650	<650	<650	<50	<50	<50	<50
	<650	<650	<650	<650	<50	<50	<50	<400
2,4-Dimethylquinoline	<110	<110	<110	560	<600	<600	650	<600
	<110	<110	390	500	<600	<600	710	<600

Continued:

Table 30. PAHs, aliphatics and other OPPs in sediments (Savannah Lab. & Environ. Serv. 1988b)..
(Continued:)

Parameter	Station							
(µg/kg dry basis)	MCV 1	MCV 2	SJR 8	SJR 14	SJR 16	SJR 17	SJR 18	SJR 19
Acridine	<2.5	<2.5	<2.5	<2.5	<50	<50	<50	<50
	<2.5	<2.5	<2.5	<2.5	<50	<50	<50	<50
Carbazole	<25	<25	<25	<25	<75	<75	<75	<75
	<25	<25	<25	<25	<75	<75	<75	<75
C10 Aliphatics	<50	<50	<50	<50	<50	<50	<50	<50
	<50	<50	<50	<50	<50	<50	<50	<50
C11 Aliphatics	<50	<50	<50	<50	<50	<50	<50	<50
	<50	<50	<50	<50	<50	<50	<50	<50
C12 Aliphatics	<50	<50	<50	<50	<50	<50	<50	<50
	<50	<50	<50	<50	<50	<50	<50	<50
C13 Aliphatics	<50	<50	<50	<50	<50	<50	<50	<50
	<50	<50	<50	<50	<50	<50	<50	<50
C14 Aliphatics	<50	<50	<50	<50	<50	<50	<50	<50
	<50	<50	<50	<50	<50	<50	<50	<50
C15 Aliphatics	<50	<50	<50	<50	<50	<50	<50	<50
	<50	<50	<50	<50	<50	<50	<50	<50
C16 Aliphatics	<50	<50	<50	<50	<50	<50	<50	<50
	<50	<50	<50	<50	<50	<50	<50	<50
C17 Aliphatics	<50	<50	<50	<50	<50	<50	<50	<50
	<50	<50	<50	<50	<50	<50	<50	<50
C18 Aliphatics	<50	<50	<50	<50	<50	<50	<50	<50
	<50	<50	<50	<50	<50	<50	<50	<50
C19 Aliphatics	<50	<50	<50	<50	<50	<50	<50	<50
	<50	<50	<50	<50	<50	<50	<50	<50
C20 Aliphatics	<50	<50	<50	<50	<50	<50	<50	<50
	<50	<50	<50	<50	<50	<50	<50	<50
C21 Aliphatics	<50	<50	<50	<50	<50	<50	<50	<50
	<50	<50	<50	<50	<50	<50	<50	<50
C22 Aliphatics	<50	<50	<50	<50	<50	<50	<50	<50
	<50	<50	<50	<50	<50	<50	<50	<50
C23 Aliphatics	<50	<50	<50	<50	<50	<50	<50	81
	<50	<50	<50	<50	<50	<50	<50	<50
C24 Aliphatics	<50	<50	<50	<50	130	140	96	88
	<50	63	<50	<50	160	140	110	<50
C25 Aliphatics	140	430	<100	<100	490	810	600	840
	<100	590	<100	<100	460	730	800	510
C26 Aliphatics	<100	370	<100	340	870	810	470	900
	<100	470	<100	310	760	750	560	360
C27 Aliphatics	560	<100	<100	<100	<100	<100	<100	<100
	530	<100	<100	<100	<100	<100	<100	<100
C28 Aliphatics	<100	560	<100	<100	750	690	500	830
	<100	530	<100	<100	620	800	<100	330
C29 Aliphatics	<100	<100	<100	<100	<100	<100	<100	<100
	<100	<100	<100	<100	<100	<100	<100	<100
C30 Aliphatics	<100	270	<100	<100	530	590	500	400
	<100	300	<100	<100	470	730	610	300

Table 3.31a PAHs, aliphatics and other OPPs in sediments (Savannah Lab. & Environ. Serv. 1988b)

Parameter (µg/kg dry basis)	Station SJR 20	SJR 21	SJR 22	SJR 25	SJR 33	SJR 34	SJR 35
Acenaphthene	110	910	<20	<400	<20	27	<20
	150	230	<20	680	<20	32	<20
Acenaphthylene	<15	220	<15	<100	<15	<15	<15
	<15	810	<15	<100	<15	<15	<15
Anthracene	10	<2	20	59	<2	<2	2.1
	10	<2	60	<50	4	3	3
Benzo(a)Anthracene	<50	<50	<50	<50	<50	<50	<50
	<50	<50	<50	<50	<50	<50	<50
Benzo(a)Pyrene	<20	<20	<20	<200	<20	<20	<20
	<20	<20	<20	<200	<20	<20	<20
Benzo(b)Fluoranthene	<50	<50	<50	460	<50	<50	<50
	<50	<50	<50	560	<50	<50	<50
Benzo(g,h,i)Perylene	<50	<50	<50	<100	<50	<50	<50
	<50	<50	<50	<100	<50	<50	<50
Benzo(k)Fluoranthene	<50	230	<50	62	<50	<50	<50
	<50	200	<50	<50	<50	<50	<50
Chrysene	70	<10	7	<50	7	4	<2
	90	54	48	<50	<2	<2	<2
Dibenzo(a,h)Anthracene	<150	<150	<150	<550	<150	<150	<150
	<150	<150	<150	<550	<150	<150	<50
Fluoranthene	200	150	170	<50	19	5.3	6.7
	200	160	220	100	8	28	7.1
Fluorene	20	50	<1.5	<150	<1.5	14	<1.5
	20	<1.5	<1.5	<150	<1.5	<1.5	<1.5
Indeno(1,2,3-cd)Pyrene	<50	<50	<50	<50	<50	<50	<50
	<50	<50	<50	<50	<50	<50	<50
Naphthalene	120	110	73	<500	<10	67	<10
	110	53	62	<500	<10	<10	<10
Pyrene	400	200	20	370	17	<5	<5
	320	<20	120	670	<5	28	<5
Phenanthrene	10	<5	<5	<50	<5	<5	<5
	11	<5	<5	<50	<5	5	<5
1-Methylnaphthalene	<150	<150	<150	<400	<150	<150	<150
	<150	<150	<150	<400	<150	<150	<150
2-Methylnaphthalene	<150	<150	<150	<400	<150	<150	<150
	<150	<150	<150	<400	<150	<150	<150
Benzonitrite	<500	<500	<500	<400	<500	<500	<500
	<500	<500	<500	<400	<500	<500	<500
Quinoline	<250	<250	<250	<1500	<250	<250	<250
	<250	<250	<250	<1500	<250	<250	<250
Quinaldine	<350	<350	<350	<150	<350	<350	<350
	<350	<350	<350	<150	<350	<350	<350
8-Methylquinoline	<650	<650	<650	130	<650	<650	<650
	<650	<650	<650	110	<650	<650	<650
7,8-Benzoquinoline	<650	<650	<650	<50	<650	<650	<650
	<650	<650	<650	<50	<650	<650	<650
2,4-Dimethylquinoline	1100	1300	<200	580	500	420	<200
	100	1400	210	<600	170	220	<200

Continued:

Table 3.31a PAHs, aliphatics and other OPPs in sediments (Savannah Lab. & Environ. Serv. 1988b).
(Continued:)

Parameter				Station			
(µg/kg dry basis)	SJR 20	SJR 21	SJR 22	SJR 25	SJR 33	SJR 34	SJR 35
Acridine	<2.5	<2.5	<2.5	<50	<2.5	<2.5	<2.5
	<2.5	<2.5	<2.5	<50	<2.5	<2.5	<2.5
Carbazole	<25	<25	<25	<75	<25	<25	<25
	<25	<25	<25	<75	<25	<25	<25
C10 Aliphatics	<50	<50	<50	<50	<50	<50	<50
	<50	<50	<50	<50	<50	<50	<50
C11 Aliphatics	<50	<50	<50	<50	<50	<50	<50
	<50	<50	<50	<50	<50	<50	<50
C12 Aliphatics	<50	<50	<50	<50	<50	<50	<50
	<50	<50	<50	<50	<50	<50	<50
C13 Aliphatics	<50	<50	<50	<50	<50	<50	<50
	<50	<50	<50	<50	<50	<50	<50
C14 Aliphatics	<50	<50	<50	<50	<50	<50	<50
	<50	<50	<50	<50	<50	<50	<50
C15 Aliphatics	<50	<50	<50	<50	<50	<50	<50
	<50	<50	<50	<50	<50	<50	<50
C16 Aliphatics	<50	<50	<50	<50	<50	<50	<50
	<50	<50	<50	<50	<50	<50	<50
C17 Aliphatics	<50	<50	<50	<50	<50	<50	<50
	<50	64	<50	<50	<50	<50	<50
C18 Aliphatics	<50	<50	<50	<50	<50	<50	<50
	<50	<50	<50	<50	<50	<50	<50
C19 Aliphatics	<50	<50	<50	<50	<50	<50	<50
	<50	<50	<50	<50	<50	<50	<50
C20 Aliphatics	<50	<50	<50	<50	<50	<50	<50
	<50	<50	<50	<50	<50	<50	<50
C21 Aliphatics	<50	<50	<50	<50	<50	<50	<50
	<50	<50	<50	<50	<50	<50	<50
C22 Aliphatics	<50	<50	<50	<50	<50	<50	<50
	<50	<50	<50	<50	<50	<50	<50
C23 Aliphatics	<50	<50	<50	110	<50	<50	<50
	<50	<50	<50	100	<50	<50	<50
C24 Aliphatics	<50	<50	72	160	<50	<50	<50
	54	<50	83	110	<50	<50	<50
C25 Aliphatics	660	240	500	910	<100	<100	310
	510	600	510	970	<100	<100	290
C26 Aliphatics	270	110	460	970	<100	<100	200
	380	360	530	950	<100	<100	180
C27 Aliphatics	<100	<100	<100	<100	<100	<100	<100
	<100	<100	<100	<100	<100	<100	<100
C28 Aliphatics	<100	120	520	1100	<100	<100	<100
	350	360	610	1300	<100	<100	120
C29 Aliphatics	<100	<100	<100	<100	<100	<100	<100
	<100	<100	<100	<100	<100	<100	<100
C30 Aliphatics	260	360	400	1100	<100	<100	<100
	300	350	350	1000	<100	<100	<100

3.7 University of Florida Study (Delfino et al. 1991).

3.7.1 Sampling Locations and Collected Parameters. The University of Florida Study (Delfino et al. 1991) was conducted during August, 1989, and involved three sites near the downtown Jacksonville area (Deer Creek, Little Sixmile Creek, and near the Naval Air Station), and another at Rice Creek (Figures 3.13-3.16). Eleven samples were taken from each site. The geographic coordinates and descriptions of the locations of the sampling sites are given in Table 3.32. The study analyzed only for organic pollutants, including both water and sediment samples. No metal analyses were performed.

3.7.2 Data Analyses and Results. All four sample stations were contaminated with PAHs (>100 μg/kg). In addition, Cl-pesticides were also present in Deer Creek and Little Sixmile Creek. Pollution sources were identified as fuel combustion products leaking from old creosote tanks, landfill leachates, and pesticides from industrial and urban uses.

Data from Deer Creek (Figures 3.13a,b) indicated the presence of organic priority pollutants (OPPs) (Wentz, 1989) in the water column, and twelve PAHs along with pesticide residues in the surficial sediments. Creosote concentrations in sediments were also high. A gradient of decreasing contaminant concentration was evident from the upstream station 16-07 to the mouth, station 16-04. This gradient indicates the effects of mixing and reduction of pollutant concentration by natural processes. Contamination in Deer Creek is attributed mainly to the creosote tanks owned by Pepper Industries. These tanks were removed almost ten years ago, and the site is currently inactive. However, the area is heavily industrialized and Deer Creek may still receive pollutant loads from stormwater drainage, groundwater seepage, and/or atmospheric transport from

Table 3.32 Sampling stations of sediment quality (Delfino et al. 1991).

No.	Station	Location	Latitude	Longitude	Date
159	14-01	Lt. Sixmile Cr.	30 22 15 N	81 44 20 W	08/17/89
160	14-02	Lt. Sixmile Cr.	30 22 15 N	81 44 20 W	08/17/89
161	14-03	Lt. Sixmile Cr.	30 22 15 N	81 44 20 W	08/17/89
162	14-04	Lt. Sixmile Cr.	30 22 15 N	81 44 20 W	08/17/89
162	14-05	Lt. Sixmile Cr.	30 22 15 N	81 44 20 W	08/17/89
163	14-06	Lt. Sixmile Cr.	30 22 15 N	81 44 20 W	08/17/89
164	14-07	Lt. Sixmile Cr.	30 22 15 N	81 44 20 W	08/17/89
165	14-08	Lt. Sixmile Cr.	30 22 15 N	81 44 20 W	08/17/89
166	14-09	Lt. Sixmile Cr.	30 22 15 N	81 44 20 W	08/18/89
167	14-10	Lt. Sixmile Cr.	30 22 15 N	81 44 20 W	08/17/89
168	14-11	Lt. Sixmile Cr.	30 22 15 N	81 44 20 W	08/17/89
169	15-01	NAS/Jax.	30 13 50 N	81 48 30 W	08/18/89
170	15-02	NAS/Jax.	30 13 50 N	81 48 30 W	08/18/89
171	15-03	NAS/Jax.	30 13 50 N	81 48 30 W	08/18/89
172	15-04	NAS/Jax.	30 13 50 N	81 48 30 W	08/18/89
173	15-05	NAS/Jax.	30 13 50 N	81 48 30 W	08/18/89
174	15-06	NAS/Jax.	30 13 50 N	81 48 30 W	08/18/89
175	15-07	NAS/Jax.	30 13 50 N	81 48 30 W	08/18/89
176	15-08	NAS/Jax.	30 13 50 N	81 48 30 W	08/18/89
177	15-09	NAS/Jax.	30 13 50 N	81 48 30 W	08/18/89
178	15-10	NAS/Jax.	30 13 50 N	81 48 30 W	08/18/89
179	15-11	NAS/Jax.	30 13 50 N	81 48 30 W	08/18/89
180	16-01	SJR/Deer Creek	30 20 30 N	81 37 40 W	08/18/89
181	16-02	SJR/Deer Creek	30 20 30 N	81 37 40 W	08/18/89
182	16-03	SJR/Deer Creek	30 20 30 N	81 37 40 W	08/18/89
183	16-04	SJR/Deer Creek	30 20 30 N	81 37 40 W	08/18/89
184	16-05	SJR/Deer Creek	30 20 30 N	81 37 40 W	08/18/89
185	16-06	SJR/Deer Creek	30 20 30 N	81 37 40 W	08/18/89
186	16-07	SJR/Deer Creek	30 20 30 N	81 37 40 W	08/18/89
187	16-08	SJR/Deer Creek	30 20 30 N	81 37 40 W	08/18/89
188	16-09	SJR/Deer Creek	30 20 30 N	81 37 40 W	08/18/89
189	16-10	SJR/Deer Creek	30 20 30 N	81 37 40 W	08/18/89
190	16-11	SJR/Deer Creek	30 20 30 N	81 37 40 W	08/18/89
191	17-01	SJR/Rice Creek	-	-	08/22/89
192	17-02	SJR/Rice Creek	-	-	08/22/89
193	17-03	SJR/Rice Creek	-	-	08/22/89
194	17-04	SJR/Rice Creek	-	-	08/22/89
195	17-05	SJR/Rice Creek	-	-	08/22/89
196	17-06	SJR/Rice Creek	-	-	08/22/89
197	17-07	SJR/Rice Creek	-	-	08/22/89
198	17-08	SJR/Rice Creek	-	-	08/22/89
199	17-09	SJR/Rice Creek	-	-	08/22/89
200	17-10	SJR/Rice Creek	-	-	08/22/89
201	17-11	SJR/Rice Creek	-	-	08/22/89

Figure 3.13a Data collection sites in Deer Creek: Creosote Tanks - Talleyrand Rd. (Delfino et al. 1991).

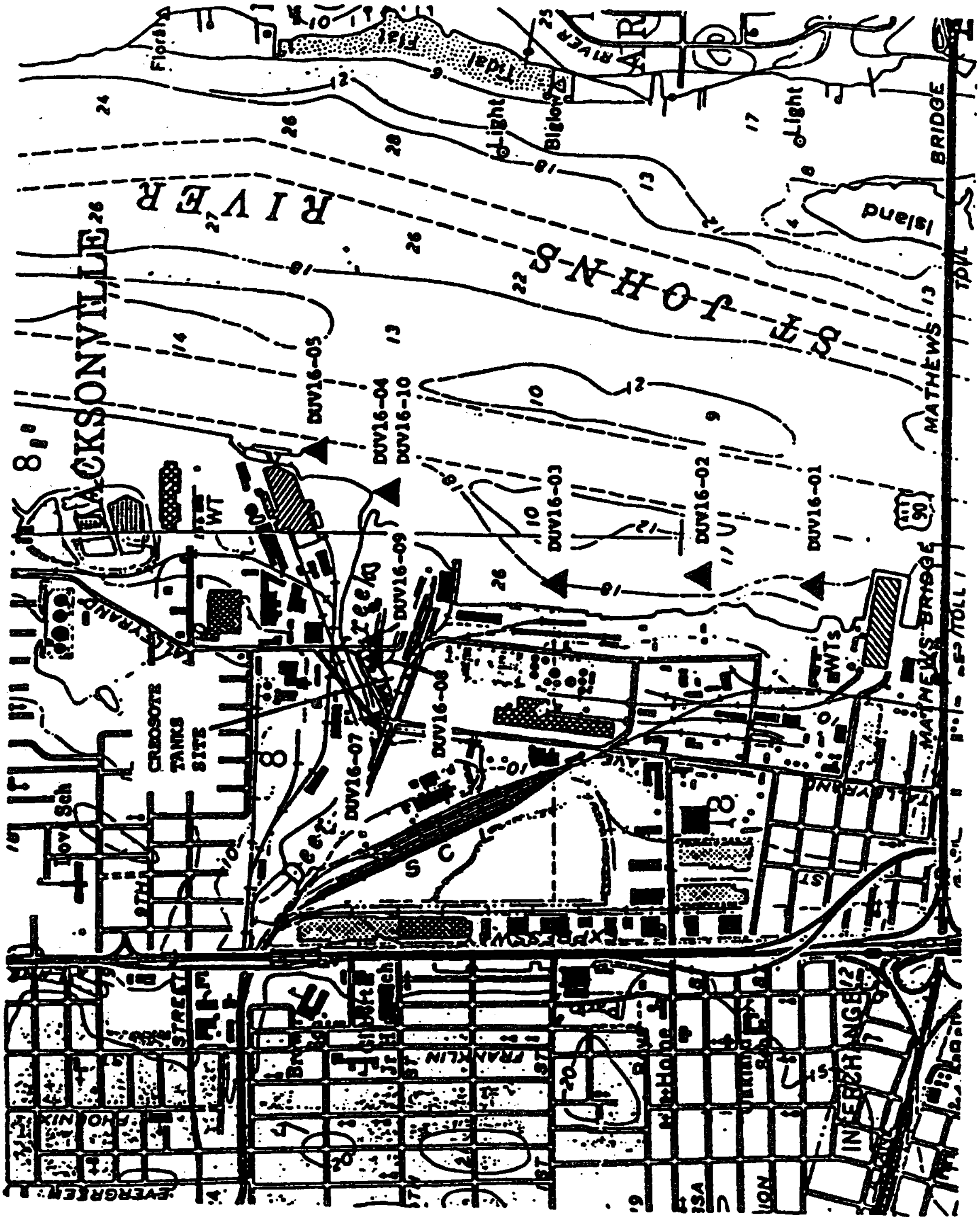


Figure 3.13b Data collection sites in Deer Creek: Creosote Tanks - Talleyrand Rd. (Delfino et al. 1991).

other sources. Release of creosote from the sediments may be a continuous source of contamination.

Data from Rice Creek (Figures 3.14a,b) shows three organic priority pollutants in the water column and ten PAHs in the surficial sediments. Anthracene, fluoranthene, and phenanthrene (i.e., fuel combustion products) were also present. The most contaminated sites were near the mouth of the tributary, station 17-07, and close to the Georgia-Pacific industrial complex, stations 17-03 and 17-04. Contamination is mainly attributed to by-products of fossil fuel combustion. Data are missing for TCDD-dioxin, organic acids and resin acids; these chemicals may be present due to the proximity of the paper mill industrial complex.

In Sixmile Creek (Figures 3.15a,b), phenanthrene was detected but not quantified in the surface water. Sediments in the same tributary contained thirteen PAHs along with 4,4'-DDD, fluoranthene and pyrene. The most contaminated sites were stations 14-01 and 14-02 near the Southern and Florida railroad tracks. Contamination in Sixmile Creek is attributed to a nearby landfill as well as to heavy, industry-related traffic.

Data from the St. Johns River estuary near the Jacksonville Naval Air Station (Figure 3.16) shows three OPPs in the water column and four PAHs in the sediments. The most contaminated sites were stations 15-01, 15-08, and 15-05. Contamination in this area is attributed to boat traffic, transport of industrial chemicals, and storage of chemical containers at the Naval Air Station. The water/sediment data obtained by Delfino et al. (1991) are summarized for the water quality in Table 3.33 and for the sediment quality in Tables 3.33a,b.

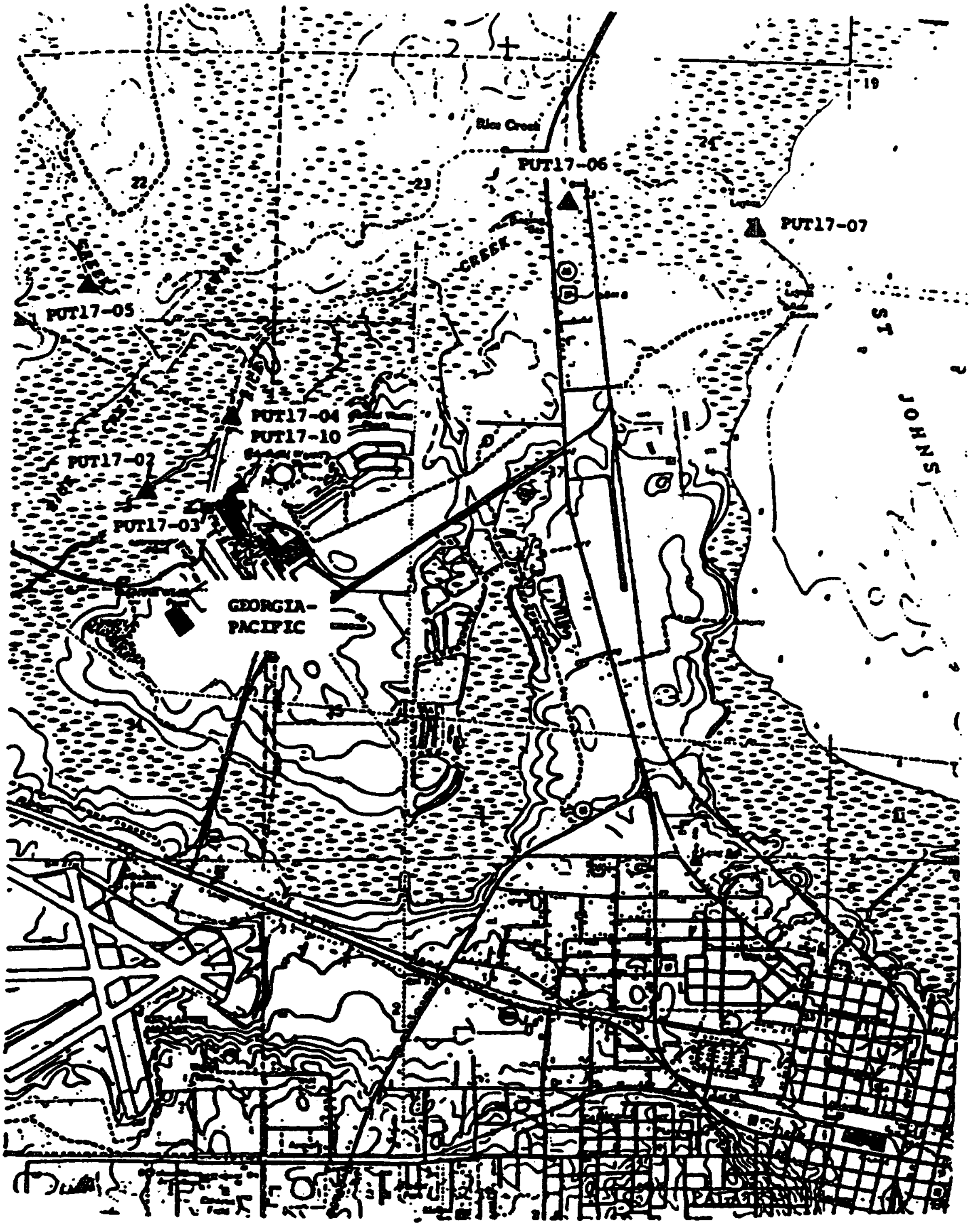


Figure 3.14a Data collection sites in Rice Creek: Georgia-Pacific (Delfino et al. 1991).

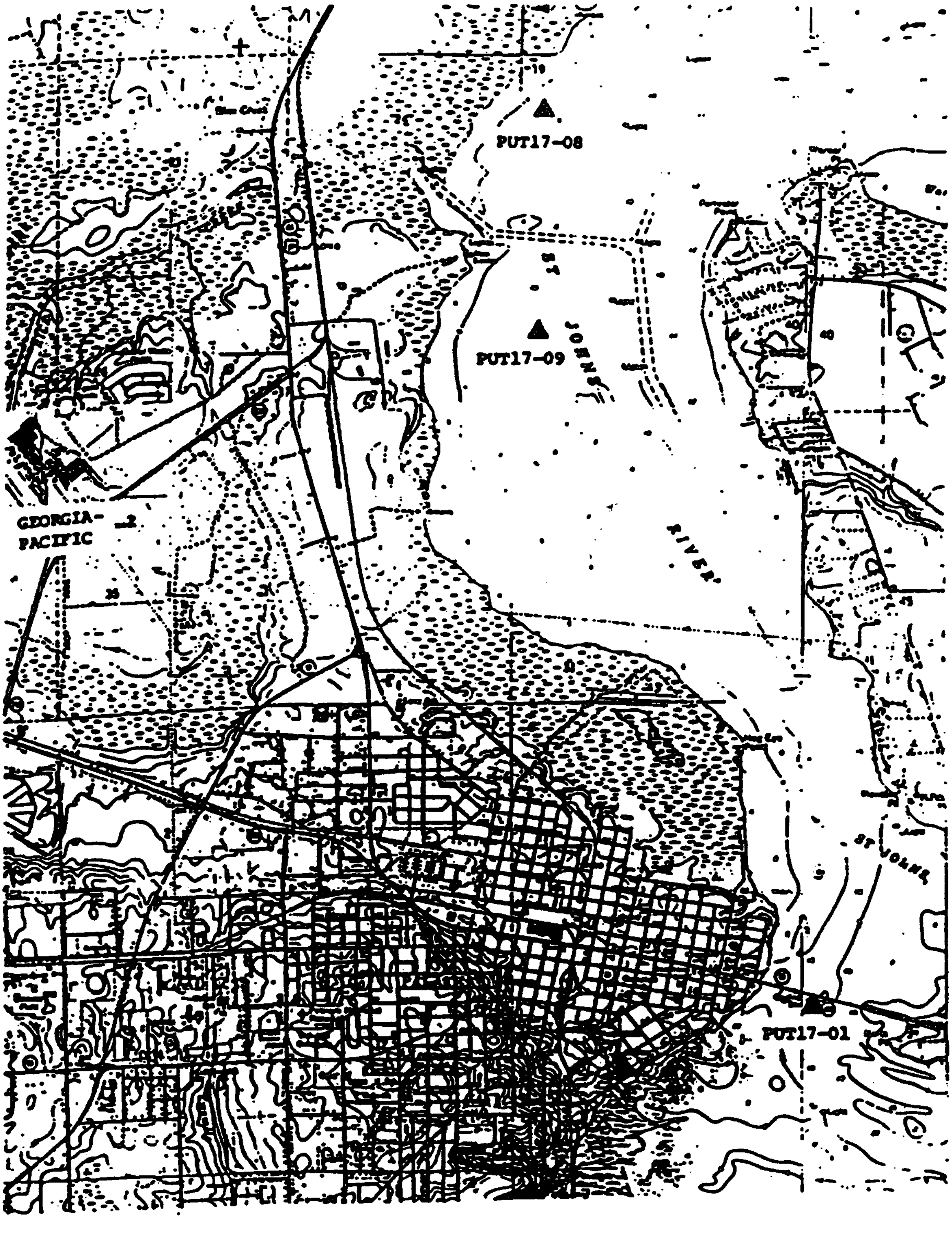


Figure 3.14b Data collection sites in Rice Creek: Georgia-Pacific (Delfino et al. 1991).

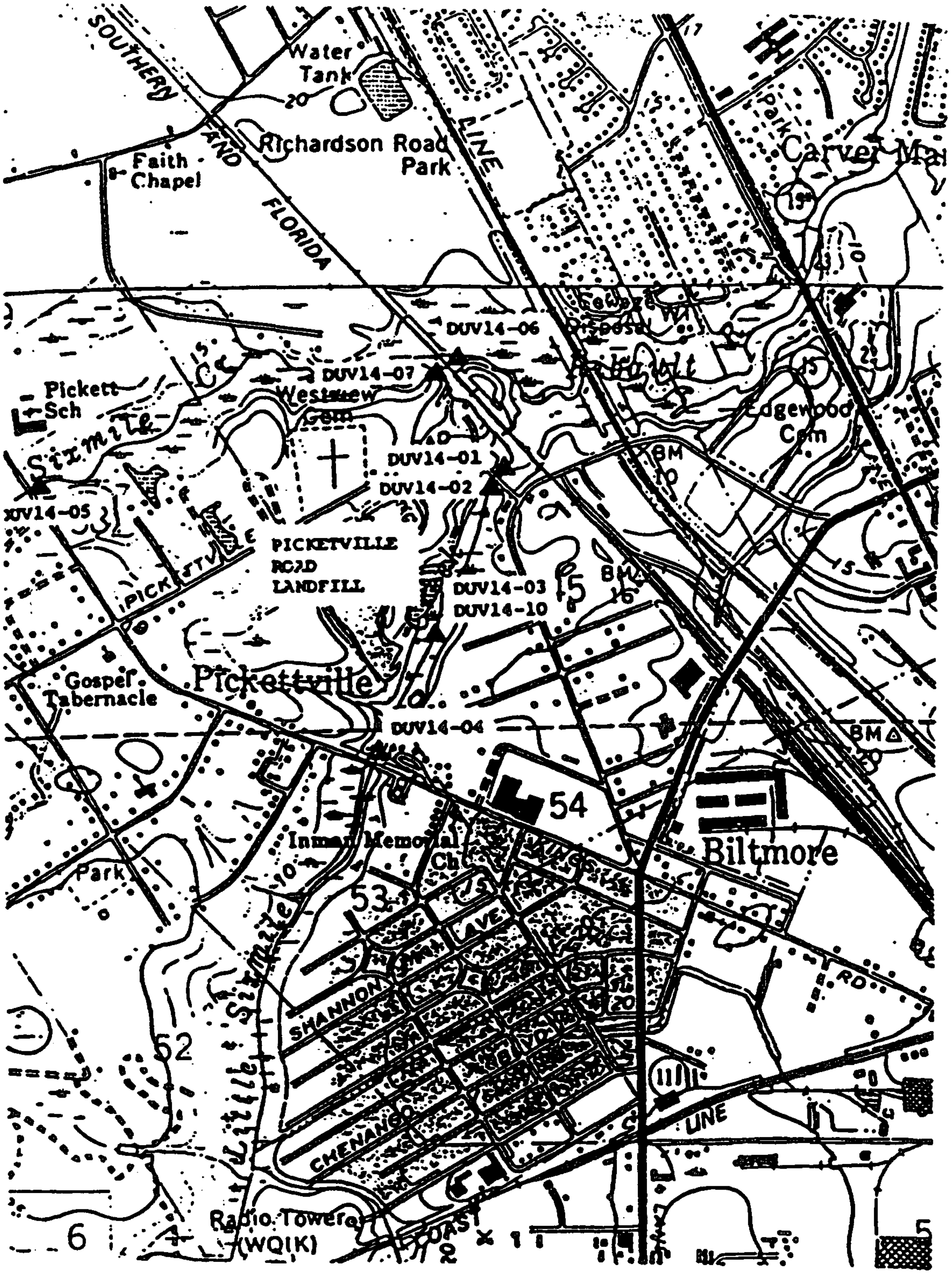


Figure 3.15a Data collection sites in Sixmile Creek: Picketville Rd. Landfill (Delfino et al. 1991).

Figure 3.15b Data collection sites in Sixmile Creek: Picketville Rd. Landfill (Delfino et al. 1991).

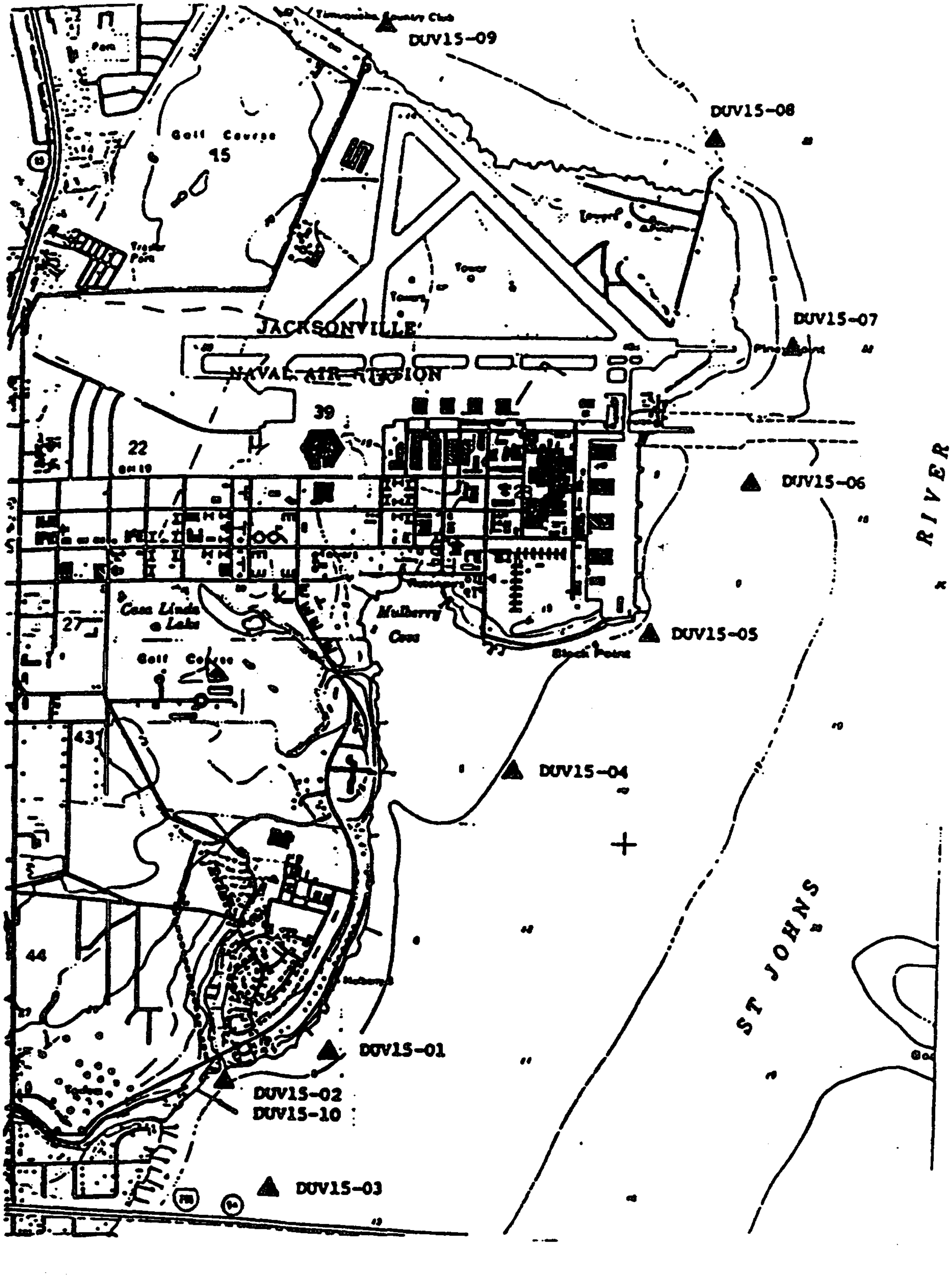


Figure 3.16 Data collection sites in St. Johns River estuary, NAS Jax (Delfino et al. 1991).

Table 3.33 Water quality data of the LSJRB (Delfino et al. 1991).

Location & Site Number	Contaminant	Max. concentration in mg/kg dry weight
Deer Creek		
16-07	1,1-dichloroethylene	9.53
16-07	Tetrachloroethylene	2.10
16-07	Trichloroethylene	6.57
16-09	Fluoranthene	*
16-07	Phenanthrene	0.44
16-07	alpha-BHC	9.44
16-10	Benzo(b+k)fluoranthene	0.21
Rice Creek		
17-05	Benzene	0.12
17-05	Ethyl benzene	*
17-05	Toluene	0.49
Sixmile Creek		
14-02	Phenanthrene	*
St. Johns River		
15-01	Benzene	0.03

Note: (*) denotes that chemical presence was verified but not quantified.

Table 3.33a Organic priority pollutants in the LSJRB sediments (Delfino et al. 1991).

Parameter	Station						
(μg/kg dry basis)	14-01	14-02	14-09	15-04	15-05	15-06	15-07
Acenaphthene	7.55	-	-	-	-	-	-
Acenaphthylene	0.11	-	-	-	-	-	-
Anthracene	14.80	-	-	-	-	-	-
Benzo(a)Anthracene	14.24	-	-	-	-	-	-
Benzo(b+k)Fluoranthene	0.37	-	-	-	-	-	-
Benzo(a)Pyrene	1.46	-	-	-	0.27	-	-
Chrysene	10.33	-	-	-	-	-	-
Dibenzo(a,h)Anthracene	0.13	-	-	-	-	-	-
Dimethyl Phthalate	-	-	-	-	-	-	0.10
Fluoranthene	23.27	-	-	0.41	-	-	-
Fluorene	13.45	-	-	-	-	-	-
Naphthalene	1.17	-	-	-	-	-	-
Phenanthrene	5.01	-	-	-	-	-	-
Pyrene	-	-	1.95	-	-	0.29	-
4,4'-DDE	-	*	-	-	-	-	-

Note: (*) denotes that chemical presence was verified but not quantified.

Table 3.33b Organic priority pollutants in the LSJRB sediments (Delfino et al. 1991).

Parameter (µg/kg dry basis)	Station 16-05	16-09	16-10	17-04	17-05	17-07
Acenaphthene	-	347.88	-	*	-	-
Acenaphthylene	-	-	0.24	-	-	0.07
Anthracene	-	139.49	-	-	-	3.73
Benzo(a)Anthracene	-	3.96	-	0.31	-	-
Benzo(b+k)Fluoranthene	-	-	0.21	-	-	-
Benzo(a)Pyrene	-	-	0.97	1.29	-	-
Butyl Benzyl Phthalate	-	-	-	-	*	-
Chrysene	-	13.53	-	0.97	-	-
Fluoranthene	-	156.96	-	2.76	-	-
Fluorene	-	270.48	-	0.14	-	-
Naphthalene	-	347.77	-	-	-	-
Phenanthrene	-	16.11	-	-	-	0.56
Pyrene	1.67	-	-	2.15	-	-
4,4'-DDE	-	1.23	-	-	-	-
4,4'-DDD	-	5.01	-	-	-	-

Note: (*) denotes that chemical presence was verified but not quantified.

3.8 Bio-Environmental Services Division Laboratory and SJRWMD Studies (1989-1991).

3.8.1 Sampling Locations and Collected Parameters. A study of sediment quality in the LSJRB was carried out from September 1989 to September 1991 by the City of Jacksonville's Bio-Environmental Services Division Laboratory. Sediment samples from the top 2-5 cm were collected using a stainless steel petite ponar, placed in glass jars with teflon-lined lids, and frozen until analyzed. The samples were collected in coordination with the SJRWMD and analyzed for their physicochemical properties by the Bio-Environmental Services (BES) Laboratory. Chemical analyses of sediment quality included eleven heavy metals, PAHs, phenols, chlorinated pesticides, and PCBs. The metals analyzed in this study include aluminum (Al), silver (Ag), arsenic (As), cadmium (Cd), chromium (Cr), copper (Cu), iron (Fe), mercury (Hg), nickel (Ni), lead (Pb), selenium (Se), and zinc (Zn). The physical analysis was limited to estimation of the sediment bulk density. The sampling stations were spread from the downtown Jacksonville area to Rice Creek (Figure 3.17, Table 3.34). Ten stations were sampled at least twice to get an estimate of accuracy and sample variability.

3.8.2 <u>**Data Analyses and Results.**</u> Bulk density was low for the Ortega River, Julington Creek and Durbin Creek which were 87-93% water. The lowest moisture content (21-50 %) was measured in samples from Cedar River (Table 3.35). High moisture content is indicative of fine sediments however particle sizes were not measured. Metal concentrations in sediments measured during the various sampling efforts are listed in Table 3.36. Metal enrichment information is provided in the same table, from which it is evident that all stations were contaminated by at least one toxic metal. The greatest contamination was by Ag, Cd, Pb and Zn, followed by Hg, Ni, and Cu; only a few stations have shown high concentrations of As, Cr, or Se. The metal:aluminum ratios are given in Table 3.37. The sites with the highest metal enrichment were the Cedar River, Ortega River, Trout River, and one near the Naval Air Station. The sites least contaminated by metals were Julington Creek, the Ft. George River, Hallowes Cove, and Durbin Creek.

Sediment samples were also tested for organic pollutants (i.e., PAHs, phenols, Cl-pesticides, and PCBs). The detection limits for the various chemical compounds analyzed during 1989-90 are listed in Table 3.38. The most commonly found compounds during 1989 were bi(2-ethylhexyl) phthalate, chrysene, fluoranthene, and pyrene (Table 3.39). The site with the most elevated organic pollutants concentrations was the Cedar River, followed by the Arlington and Trout Rivers. The predominant organic chemicals

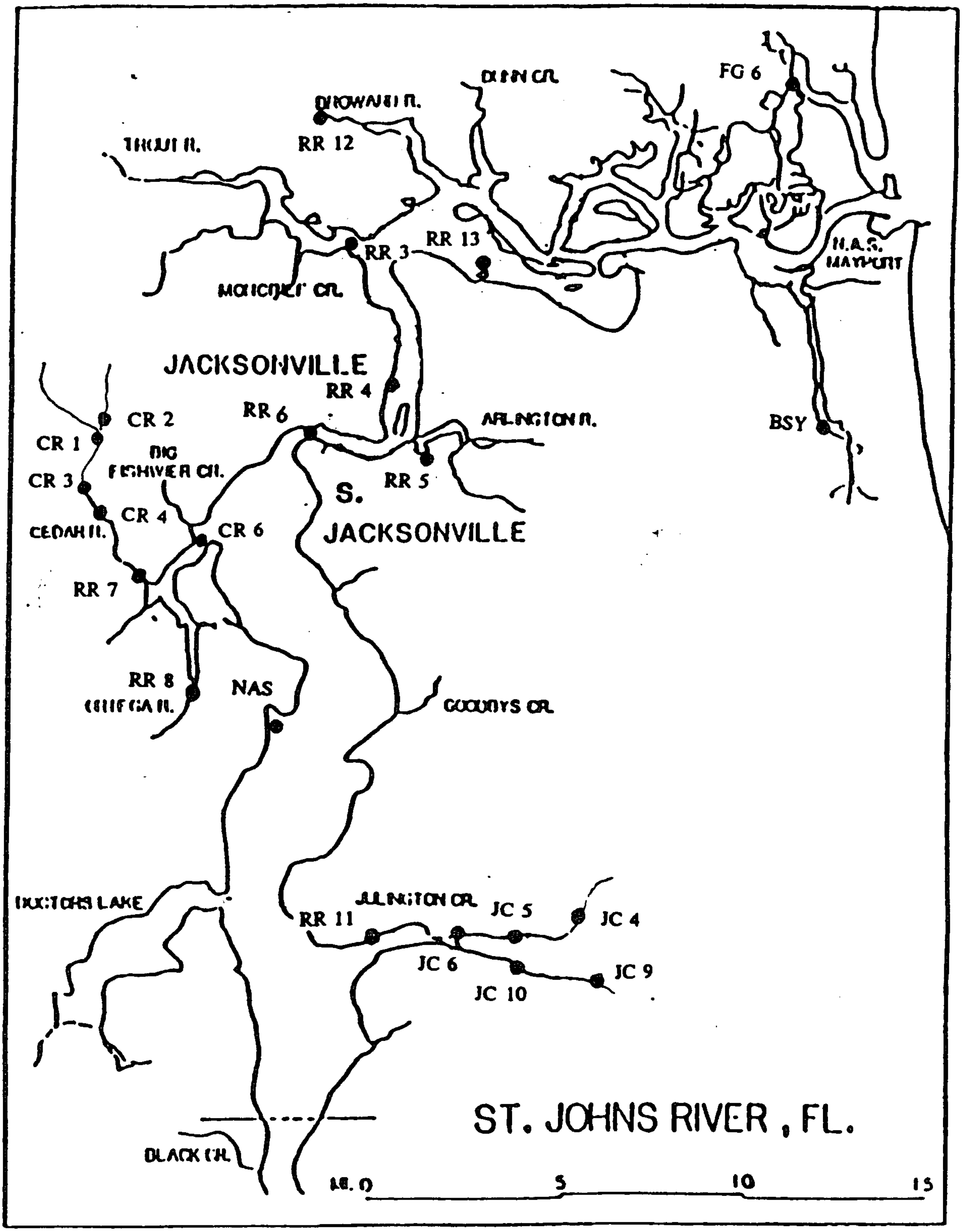


Figure 3.17 Sampling stations of sediment quality (BES Division Lab. 1992).

Table 3.34 Sampling stations of sediment quality (BES Division Lab. 1992).

No.	Station	Location	Date
122	RR 3	SJR/Trout River	10/25/89
		SJR/Trout River	03/27-28/90
123	RR 4	St. Johns River	03/27-28/90
124	RR 5	SJR/Arlington R.	10/25/89
		SJR/Arlington R.	03/27-28/90
125	RR 7	Cedar/Ortega R.	03/27-28/90
126	RR 8	Ortega River	03/27-28/90
127	RR 11	SJR/Julington C.	03/27-28/90
		SJR/Julington C.	06/12/91
128	RR 12	Broward River	03/27-28/90
128	RR 13	SJR/Mill Cove	10/25/89
		SJR/Mill Cove	03/27-28/90
129	FG 6	Ft. George River	03/27-28/90
130	NAS	Jax./NAS	03/27-28/90
131	BSY	Bellinger S'yards	03/27-28/90
132	CR 1	Cedar River	10/09/89
		Cedar River	03/27-28/90
		Cedar River	09/19/91
133	CR 2	Cedar River	10/09/89
		Cedar River	03/27-28/90
		Cedar River	09/19/91
134	CR 3	Cedar River	10/09/89
		Cedar River	03/27-28/90
135	CR 4	Cedar River	10/09/89
		Cedar River	03/27-28/90
136	CR 5	Cedar River	10/09/89
137	CR 6	SJR/Cedar River	03/27-28/90
138	JC 4	Julington Creek	09/12/89
139	JC 5	Julington Creek	09/12/89
		Julington Creek	06/12/91
140	JC 6	Julington Creek	09/12/89
141	JC 9	Durbin Creek	09/12/89
142	JC 10	Durbin Creek	09/12/89
		Durbin Creek	06/12/91
143	DI	Doctors Inlet	09/90
144	HC	Hallowes Cove	09/90
145	JUL	Julington Creek	09/90
146	RB 1	Red Bay	07/16/91
147	RB 2	Red Bay	07/16/91
148	NAS 1	Jax./NAS	07/16/91
149	NAS 2	Jax./NAS	07/16/91
150	NAS 3	Jax./NAS	07/16/91
151	MC 1	Moncrief Creek	06/13/91
152	MC 2	Moncrief Creek	06/13/91
153	R 1	Ribault River	06/13/91
154	R 2	Ribault River	06/13/91
155	R 3	Ribault River	06/13/91
156	RC 1	Rice Creek	09/90
157	RC 2	Rice Creek	09/90
158	RC 3	Rice Creek	09/90

Table 3.35 Moisture content of sediment samples (BES Division Lab. 1992).

Station	% Moisture	Station	% Moisture	Station	% Moisture
RR 3	73 78	BSY	70 76	JC 5	92 92
RR 4	77 78	CR 1	23 28	JC 10	93 90
RR 5	54 52		24 22	RB 1	89 88
RR 7	67 83	CR 2	22 21	RB 2	57 62
RR 8	90 87		39 23	MC 1	80 76
RR 11	82 86	CR 3	41 48	MC 2	76 77
	86 87	CR 4	65 38	R 1	83 82
RR 12	86 80	CR 6	76 81	R 2	82 78
RR 13	70 73	DI	86 85	R 3	82 81
FG 6	68 69	HC	80 78	RC 1	81 -
NAS	85 81	JUL	69 82	RC 2	86 -
				RC 3	89 -

detected in 1990 were benzo(b)fluoranthene, fluoranthene, naphthalene and pyrene (i.e., fuel combustion products) (Table 3.39a, 3.39b). The stations with the highest organic pollution levels were the Arlington River, Cedar River, Ortega River, and the main river off the Naval Air Station. Sites contaminated by only naphthalene were Julington Creek, the Broward River, Mill Cove, and near the Bellinger Shipyards. During the 1991 sampling event, (Tables 3.39c,d), areas with elevated PAHs and chlorinated pesticides contamination included the Cedar River, Ribault River, Mill Cove, the Naval Air Station, and Julington Creek.

In summary, the sediment samples collected from 1989 to 1991 by the BES Division Laboratory revealed high concentrations of heavy metals, PAHs, and Cl-pesticides in various sites along the middle section of the LSJRB. The most heavily contaminated sites were the Cedar River and near the Naval Air Station. Other contaminated sites included the Arlington River, Ortega River, Trout River and Ribault River.

Table 3.36 Trace metals in the LSJRB sediments (BES Division Lab. 1992).

		Parameter in μm/kg dry basis										
Station	Date	Al	Ag	As	Cd	Cr	Cu	Hg	Ni	Pb	Se	Zn
RR 3	10/25/89	20	<4.4•	3.2	4.20•	71	68.0•	0.09	11.0	42•	<1.00	140•
		21	<5.0•	-	5.90•	102•	120.0•	-	14.0	57•	-	200•
RR 5	10/25/89	36	4.2•	2.8	5.50•	76	38.0	0.02	15.0	90•	1.80	220•
		39	3.8•	-	5.20•	72	35.0	-	16.0	91•	-	210•
RR 13	10/25/89	16	<4.1•	-	0.61•	21	5.2	-	6.1	20•	-	32
		17	<3.9•	2.6	0	23	5.9	0.44•	11.0	19•	<1.00	37
CR 1	10/09/89	7	<2.2•	-	0.26	26	15.0•	-	2.8	39•	-	120•
		6	<3.0•	-	0.18	24	14.0	-	3.4	35•	-	110•
CR 2	10/09/89	20	6.9•	-	3.40•	71	240.0•	-	28.0•	360•	-	1200•
		18	7.4•	-	2.00•	81•	230.0•	-	25.0•	350•	-	1030•
CR 3	10/09/89	7	<3.7•	-	1.30•	34	26.0•	-	8.4	73•	-	220•
		8	<3.5•	-	1.80•	21	21.0•	-	5.9	68•	-	190•
CR 4	10/09/89	9	<4.5•	-	2.00•	12	15.0	-	2.8	55•	-	110•
		5	<2.3•	-	0.73•	13	16.0	-	5.9	61•	-	130•
CR 5	10/09/89	24	<12.0•	-	4.60•	97•	86.0•	-	32.0•	210•	-	670•
		28	<10.0•	-	4.20•	93	87.0•	-	24.0	180•	-	610•
JC 4	09/12/89	12	<2.9•	-	1.00•	11	6.1	-	8.4	25•	-	85•
		11	<5.2•	-	0.62•	10	7.6	-	6.6	18•	-	62•
JC 5	09/12/89	7	<4.0•	-	0.80•	5	4.6	-	4.2	19•	-	35
		11	<6.0•	-	1.20•	6	5.4	-	3.9	22•	-	44
JC 6	09/12/89	28	<13.0•	-	3.80•	23	23.0	-	21.0	31	-	130•
		21	<10.0•	-	2.70•	26	19.0	-	18.0	32	-	110•
JC 9	09/12/89	9	<4.0•	-	1.10•	6	4.6	-	4.9	20•	-	30
JC 10	09/12/89	16	<11.0•	-	2.30•	10	8.5	-	12.0	37•	-	99•
		14	<6.1•	-	1.80•	8	9.1	-	8.2	47•	-	100•
RR 3	03/27/90	29	<7.8•	6.7	3.00•	110•	170.0•	0.73•	53.0•	120•	0.93	300•
	03/28/90	26	<8.2•	7.7	2.80•	110•	250.0•	0.38•	44.0•	100•	0.98	290•
RR 4	03/27/90	42	<9.9•	8.5	3.60•	74	45.0	0.50	41.0•	45	1.10	190•
	03/28/90	45	<9.7•	8.7	3.50•	88	54.0	0.36	46.0•	55•	1.30	210•
RR 5	03/27/90	10	<4.9•	2.0	1.40•	19	14.0	0.15•	17.0	32•	0.48	93•
	03/28/90	8	<4.7•	2.2	1.20•	16	20.0	0.14•	26.0•	230•	0.39	230•
RR 7	03/27/90	13	<5.5•	3.3	2.50•	34	55.0•	0.29•	25.0•	160•	0.73	405•
	03/28/90	30	<12.0•	5.9	5.80•	66	120.0•	0.75•	59.0•	370•	1.70	900•
RR 8	03/27/90	38	<22.0•	4.7	5.30•	65	74.0•	1.00•	67.0•	160•	2.10	470•
	03/28/90	26	<18.0•	3.3	3.80•	50	55.0•	0.98•	49.0•	160•	1.80	340•
RR 11	03/27/90	12	<12.0•	2.2	3.10•	33	31.0•	0.38•	53.0•	26•	1.40•	140•
	03/28/90	22	<15.0•	3.3	3.30•	42	20.0	0.56•	38.0•	31	2.20•	160•
RR 12	03/27/90	40	<13.0•	8.2	3.90•	110•	47.0	0.54	53.0•	73•	1.70	290•
	03/28/90	31	<12.0•	5.6	3.00•	70	37.0	0.30	45.0•	51•	1.00	220•
RR 13	03/27/90	30	<7.8•	5.9	2.50•	58	26.0	0.28	30.0	39	0.67	150•
	03/28/90	31	<6.6•	6.0	2.50•	60	24.0	0.28	30.0	37	0.72	130•
FG 6	03/27/90	37	<6.5•	7.8	2.30•	50	6.9	0.12	28.0	12	0.48	83
	03/28/90	30	<6.8•	8.0	2.00•	56	10.0	0.09	30.0	16	0.45	83•
NAS	03/27/90	27	<14.0•	5.2	6.50•	66	36.0	0.42•	60.0•	240•	1.30	250•
	03/28/90	21	<12.0•	5.7	2.90•	47	30.0	0.29•	44.0•	40•	1.60	170•
BSY	03/27/90	32	<7.7•	6.4	2.60•	73	68.0•	0.16	46.0•	190•	0.74	250•
	03/28/90	29	<8.6•	7.3	2.40•	63	55.0•	0.15	33.0•	29	0.75	150•
CR 1	03/27/90	6	<3.1•	1.6	0.68•	11	7.0	0.08•	0.5	14•	0.06	60•
	03/28/90	5	<3.2•	2.7	0.64•	15	4.8	0.06•	9.5	19•	<0.06	61•
CR 2	03/27/90	2	<3.1•	0.4	0.61•	8	18.0•	0.05•	6.8	33•	<0.06	130•
	03/28/90	3	<2.9•	0.4	0.59•	6	64.0•	0.06•	5.3	22•	<0.06	220•
CR 3	03/27/90	12	<4.1•	2.3	1.90•	30	35.0•	0.17	14.0	120•	0.23	290•
	03/28/90	19	<3.9•	3.1	2.30•	38	49.0•	0.13	18.0	150•	0.23	380•
CR 4	03/27/90	16	<7.0•	15.0•	6.00•	87	85.0•	1.80•	40.0•	200•	0.90	480•
	03/28/90	5	<3.3•	1.9	1.10•	13	47.0•	1.50•	9.3	50•	0.04	200•
CR 6	03/27/90	24	<9.6•	4.6	3.10•	54	94.0•	0.59	4.3	47•	1.90	180•
	03/28/90	31	<13.0•	4.2	3.80•	47	40.0	1.30	38.0•	51•	1.70	190•

Continued:

Table 3.36 Trace metals in the LSJRB sediments (BES Division Lab. 1992). (Continued:)

		Parameter in µm/kg dry basis										
Station	**Date**	**Al**	**Fe**	**As**	**Cd**	**Cr**	**Cu**	**Hg**	**Ni**	**Pb**	**Se**	**Zn**
DI	09/90	40	2650	3.2	6.50*	75	25.6*	1.02*	10.0	44	<1.49	140*
		40	2730	2.9	6.30*	68	24.2*	0.87*	11.1	45	<1.55	124*
HC	09/90	23	1920	1.3	4.50*	41	6.6	0.32*	7.3	12	<1.23	26
		25	2003	1.6	4.70*	45	8.1	0.44*	5.1	34	<1.00	53
JUL	09/90	16	1103	1.6	4.50*	26	9.3	<0.10	3.6	10	<1.00	48
		28	2110	1.8	5.30*	46	16.9	0.52	12.4	26	1.05	87*
RC 1	09/90	24	817	1.2	3.30*	45	23.8	0.59*	29.2	20	<1.13	166*
RC 2	09/90	21	760	<1.5	2.50*	36	28.5	0.70*	29.8	18	<1.58	177*
RC 3	09/90	22	774	<1.5	2.30*	40	91.4*	0.90*	86.1*	12	<1.54	192*
JC 5	06/12/91	34	16900	2.5	1.05*	29	20.5	0.12	30.6	47*	<1.00	131*
		36	17300	1.9	1.13*	30	24.9	0.32	10.4	51*	<1.00	150*
JC 10	06/12/91	20	12900	1.5	0.99*	17	13.3	0.15	7.2	43*	<1.00	82*
		19	11200	2.2	0.72*	14	9.7	0.01	6.0	33*	<1.00	57
RR 11	06/12/91	41	24100	3.7	1.54*	55	29.6	0.46*	17.3	51*	<1.00	136*
		42	24800	2.9	1.20*	57	27.7	0.56*	14.5	43	<1.00	110*
MC 1	06/13/91	54	27700	15.8*	1.87*	8	258.0*	0.47	27.9	178*	<1.00	398*
		41	23100	10.2	1.51*	106*	196.0*	0.45	25.7	140*	<1.00	309*
MC 2	06/13/91	51	27500	7.1	1.37*	128*	292.0*	0.52*	31.1	156*	<1.00	256*
		48	26700	7.2	1.56*	140*	278.0*	0.52*	31.1	178*	<1.00	304*
R 1	06/13/91	53	28152	3.8	2.91*	74	73.0	0.49	21.9	150*	<1.00	400*
		48	26041	3.7	2.72*	71	60.6	0.40	19.9	166*	<1.00	358*
R 2	06/13/91	46	29158	4.6	2.74*	75	64.6	0.49	18.9	114*	<1.00	383*
		34	22072	3.2	2.16*	60	48.1	0.29	17.4	124*	<1.00	312*
R 3	06/13/91	55	29657	5.6	2.25*	78	53.4	0.41	19.3	94*	<1.00	206*
		57	26493	5.1	1.93*	83	57.6	0.43	20.2	122*	<1.00	224*
RB 1	07/16/91	34	22100	4.5	0.84*	54	19.2	0.32*	13.4	32	1.96	85*
		35	22500	3.8	1.02*	54	21.5	0.38*	15.0	37	1.59	98*
RB 2	07/16/91	15	10800	3.1	0.69*	27	29.3*	0.17*	7.0	41*	0.81	152*
		10	4710	2.2	0.36	17	17.4	0.16*	5.0	26*	0.28	84*
NAS 1	07/16/91	34	23600	5.7	2.36*	59	19.3	0.31	14.0	46*	1.08	93*
		54	32700	5.4	1.49*	82	29.7	0.36	17.9	59*	1.45	145*
NAS 2	07/16/91	49	34400	3.8	3.89*	89	33.7	0.28	20.2	85*	1.42	160*
		48	37400	2.2	3.68*	96*	31.9	0.33	19.7	80*	1.56	160*
NAS 3	07/16/91	54	34300	5.2	1.47*	88	24.6	0.40	16.8	60*	0.90	123*
		58	36100	5.4	1.38*	100*	27.4	0.43	18.0	59*	1.18	130*
CR 1	09/19/91	44	23900	8.7	3.59*	81	134.0*	0.24	19.8	329*	0.88	862*
		52	27100	10.1	4.23*	94*	148.0*	0.28	22.1	373*	0.50	989*
CR 2	09/19/91	27	12900	5.7	1.64*	41	44.8	0.13	11.2	116*	<0.49	347*
		22	10800	3.6	1.38*	32	40.1	0.28	9.5	107*	<0.49	338*

Note: (*) denotes enriched metal concentrations.

Table 3.37 Metal:Aluminum ratios in the LSJRB sediments (BES Division Lab. 1992)

Station	Ag $x10^{-4}$	As $x10^{-4}$	Cd $x10^{-4}$	Cr $x10^{2}$	Cu $x10^{2}$	Hg $x10^{3}$	Ni $x10^{3}$	Pb $x10^{2}$	Se $x10^{3}$	Zn $x10^{3}$
RR3	<2.2	1.6	2.1	0.36	0.34	0.005	0.57	0.21	<0.05	7.0
	<2.4	-	2.5	0.51	0.60	-	0.70	0.29	-	10.0
RR 5	1.2	0.8	1.2	0.21	0.11	0.001	0.42	0.25	0.05	6.1
	1.0	-	1.3	0.18	0.09	-	0.41	0.23	-	5.4
RR 13	<2.6	-	0.4	0.13	0.03	-	0.38	0.13	-	2.0
	<2.3	1.5	0	0.14	0.04	0.026	0.65	0.11	<0.06	2.2
CR 1	<4.7	-	0.6	0.55	0.32	-	0.60	0.83	-	25.5
	<4.9	-	0.3	0.39	0.23	-	0.56	0.57	-	18.0
CR 2	3.5	-	1.7	0.36	1.20	-	1.40	1.80	-	60.0
	4.1	-	1.0	0.45	1.28	-	1.40	1.94	-	57.2
CR 3	<5.1	-	1.7	0.47	0.36	-	1.15	1.00	-	30.1
	<4.5	-	2.3	0.27	0.27	-	0.75	0.87	-	24.4
CR 4	<5.3	-	2.4	0.14	0.18	-	0.33	0.65	-	12.9
	<4.8	-	1.5	0.27	0.33	-	1.20	1.27	-	27.1
CR 5	<5.0	-	1.9	0.40	0.36	-	1.30	0.88	-	27.9
	<3.6	-	1.5	0.33	0.31	-	0.86	0.64	-	21.8
JC 4	<2.4	-	0.8	0.09	0.05	-	0.70	0.21	-	7.1
	<4.7	-	0.6	0.09	0.07	-	0.60	0.16	-	5.6
JC 5	<5.6	-	1.0	0.07	0.06	-	0.58	0.26	-	4.9
	<5.5	-	1.1	0.05	0.05	-	0.31	0.20	-	4.0
JC 6	<4.6	-	1.4	0.08	0.08	-	0.75	0.11	-	4.6
	<4.8	-	1.3	0.12	0.09	-	0.86	0.15	-	5.2
JC 9	<4.5	-	1.2	0.07	0.05	-	0.55	0.22	-	3.4
JC 10	<6.9	-	1.4	0.06	0.05	-	0.75	0.23	-	6.2
	<4.4	-	1.2	0.06	0.07	-	0.59	0.34	-	7.1
RR 3	<2.7	2.3	2.7	0.38	0.59	0.025	1.80	0.41	0.03	10.3
	<3.2	3.0	1.1	0.42	0.96	0.015	1.70	0.38	0.04	11.2
RR 4	<2.4	2.0	0.9	0.18	0.11	0.012	1.00	0.11	0.03	4.5
	<2.2	1.9	0.8	0.20	0.12	0.008	1.00	0.12	0.03	4.7
RR 5	<5.0	2.0	1.4	0.19	0.14	0.015	1.70	0.33	0.05	9.5
	<5.7	2.7	1.4	0.19	0.24	0.017	3.10	2.77	0.05	27.7
RR 7	<4.2	2.5	1.9	0.26	0.42	0.022	1.90	1.23	0.06	31.2
	<4.0	2.0	1.9	0.22	0.40	0.025	2.00	1.23	0.06	30.0
RR 8	<5.8	1.2	1.4	0.17	0.19	0.026	1.80	0.42	0.06	12.4
	<6.9	1.3	1.5	0.19	0.21	0.038	1.90	0.62	0.07	13.1
RR 11	<10.0	1.8	2.5	0.28	0.26	0.032	4.40	0.22	0.11	11.7
	<6.8	1.5	1.5	0.19	0.09	0.025	1.70	0.14	0.10	7.3
RR 12	<3.3	2.1	1.0	0.28	0.12	0.014	1.30	0.18	0.04	7.3
	<3.8	1.8	1.0	0.23	0.12	0.010	1.50	0.16	0.03	7.1
RR 13	<2.6	2.0	0.8	0.19	0.09	0.009	1.00	0.13	0.02	5.0
	<2.1	1.9	0.8	0.19	0.08	0.009	1.00	0.12	0.02	4.2
FG 6	<1.8	2.1	0.6	0.14	0.02	0.003	0.80	0.03	0.01	2.2
	<2.3	2.7	0.7	0.19	0.03	0.003	1.00	0.05	0.02	2.8
NAS	<5.2	1.9	2.4	0.24	0.13	0.016	2.20	0.89	0.05	9.3
	<5.7	2.7	1.4	0.22	0.14	0.014	2.10	0.19	0.07	8.1
BSY	<2.4	2.0	0.8	0.23	0.21	0.005	1.40	0.59	0.02	7.8
	<2.9	2.5	0.8	0.22	0.19	0.005	1.10	0.10	0.03	5.2
CR 1	<5.6	2.9	1.2	0.20	0.13	0.015	0.09	0.25	0.01	10.9
	<6.7	5.7	1.3	0.31	0.10	0.013	1.97	0.40	<0.01	12.7
CR 2	<21.0	3.0	4.0	0.53	1.20	0.033	4.50	2.20	<0.04	86.7
	<11.6	1.6	2.4	0.24	2.56	0.024	2.10	0.88	<0.02	88.0
CR 3	<3.4	1.9	1.6	0.25	0.29	0.014	1.20	1.00	0.02	24.2
	<2.1	1.6	1.2	0.20	0.26	0.007	0.95	0.79	0.01	20.0
CR 4	<4.4	9.4	3.8	0.54	0.53	0.113	2.50	1.25	0.06	30.0
	<6.9	4.0	2.3	0.27	0.98	0.312	1.90	1.04	0.01	41.7
CR 6	<4.0	1.9	1.3	0.23	0.39	0.025	0.18	0.20	0.08	7.5
	<4.2	1.4	1.2	0.15	0.13	0.042	1.20	0.16	0.05	6.1

Continued:

Table 3.37 Metal:Aluminum ratios in the LSJRB sediments (BES Division Lab. 1992). (Continued:)

Station	Fe $x10^{2}$	As $x10^{4}$	Cd $x10^{4}$	Cr $x10^{2}$	Cu $x10^{2}$	Hg $x10^{3}$	Ni $x10^{3}$	Pb $x10^{2}$	Se $x10^{3}$	Zn $x10^{3}$
DI	6.61	0.8	1.6	0.19	0.06	0.025	0.25	0.11	<0.04	3.5
	6.81	0.7	1.6	0.17	0.06	0.022	0.28	0.11	<0.04	3.1
HC	8.45	0.6	2.0	0.18	0.03	0.014	0.32	0.05	<0.05	1.1
	7.92	0.6	1.9	0.18	0.03	0.017	0.20	0.13	<0.04	2.1
JUL	7.10	1.0	2.9	0.17	0.06	<0.006	0.23	0.06	<0.10	3.1
	7.56	0.7	1.9	0.16	0.06	0.019	0.44	0.09	0.04	3.1
RC 1	3.42	0.5	1.4	0.19	0.10	0.025	1.20	0.08	<0.05	6.9
RC 2	3.60	<0.7	1.2	0.17	0.14	0.033	1.41	0.09	<0.07	8.4
RC 3	3.49	<0.7	1.0	0.18	0.41	0.041	3.87	0.05	<0.07	8.6
JC 5	50.15	0.7	0.3	0.09	0.06	0.003	0.91	0.14	<0.03	3.9
	47.92	0.5	0.3	0.08	0.07	0.009	0.29	0.14	<0.03	4.2
JC 10	64.82	0.8	0.5	0.09	0.07	0.008	0.86	0.22	<0.05	4.1
	59.57	1.2	0.4	0.07	0.05	0.001	0.32	0.18	<0.05	3.0
RR 11	58.50	0.9	0.4	0.13	0.07	0.011	0.41	0.12	<0.02	3.3
	59.47	0.7	0.3	0.14	0.07	0.013	0.35	0.10	<0.02	2.6
MC 1	51.78	3.0	0.3	0.01	0.48	0.009	0.52	0.33	<0.02	7.4
	56.76	2.5	0.4	0.26	0.48	0.011	0.63	0.34	<0.02	7.6
MC 2	54.13	1.4	0.3	0.25	0.57	0.010	0.61	0.31	<0.02	5.0
	56.21	1.5	0.3	0.29	0.59	0.011	0.65	0.37	<0.02	6.4
R 1	52.92	0.7	0.5	0.14	0.14	0.009	0.41	0.28	<0.02	7.5
	53.80	0.8	0.6	0.15	0.13	0.008	0.41	0.34	<0.02	7.4
R 2	62.98	1.0	0.6	0.16	0.14	0.010	0.41	0.25	<0.02	8.3
	65.69	1.0	0.6	0.18	0.14	0.009	0.52	0.37	<0.03	9.3
R 3	53.53	1.0	0.4	0.14	0.10	0.007	0.35	0.17	<0.02	3.7
	46.16	0.9	0.3	0.14	0.10	0.007	0.35	0.21	<0.02	3.9
RB 1	65.97	1.3	0.3	0.16	0.06	0.010	0.40	0.10	0.06	2.5
	64.29	1.1	0.3	0.15	0.06	0.011	0.43	0.11	0.05	2.8
RB 2	73.97	2.1	0.5	0.18	0.20	0.012	0.48	0.28	0.06	10.4
	49.22	2.3	0.4	0.18	0.18	0.017	0.52	0.27	0.03	8.8
NAS 1	70.24	1.7	0.7	0.18	0.06	0.009	0.42	0.14	0.03	2.8
	60.44	1.0	0.3	0.15	0.05	0.007	0.33	0.11	0.03	2.7
NAS 2	70.78	0.8	0.8	0.18	0.69	0.006	0.42	0.17	0.03	3.3
	78.41	0.5	0.8	0.20	0.67	0.007	0.41	0.17	0.03	3.4
NAS 3	63.40	1.0	0.3	0.16	0.45	0.007	0.31	0.11	0.02	2.3
	62.56	0.9	0.2	0.17	0.47	0.007	0.31	0.10	0.05	2.3
CR 1	54.57	2.0	0.8	0.18	0.31	0.005	0.45	0.75	0.02	19.7
	52.02	1.9	0.8	0.18	0.28	0.005	0.42	0.72	0.01	19.0
CR 2	47.08	2.1	0.6	0.15	0.16	0.005	0.41	0.42	<0.02	12.7
	49.54	1.7	0.6	0.30	0.18	0.013	0.44	0.49	<0.02	15.5

Table 3.38 Detection limits: PAHs, phenols, pesticides, & PCBs in sediments (BES Division Lab. 1992).

Parameter (mg/kg dry basis)	Det. Limit	Parameter (mg/kg dry basis)	Det. Limit
1,2,4-Trichlorobenzene	0.5	Butyl Benzyl Phthalate	0.5
1,2-Dichlorobenzene	0.5	Chrysene	0.5
1,3-Dichlorobenzene	0.5	Dibenzo(a,h)Anthracene	0.5
1,4-Dichlorobenzene	0.5	Dibutyl Phthalate	0.5
2,4-Dinitrotoluene	0.5	Diethyl Phthalate	0.5
2,6-Dinitrotoluene	0.5	Dimethyl Phthalate	0.5
2-Chloronaphthalene	0.5	Diphenylhydrazine (Azobenzene)	0.5
3,3'-Dichlorobenzidine	0.5	Di-N-Octyl Phthalate	0.5
4-Bromophenyl Phenyl Ether	0.5	Fluoranthene	0.5
4-Chlorophenyl Phenyl Ether	0.5	Fluorene	0.5
Acenaphthene	0.5	Hexachlorobenzene	0.5
Acenaphthylene	0.5	Hexachlorobutadiene	0.5
Anthracene	0.5	Hexachlorocyclopentadiene	0.5
Benzidine	0.5	Hexachloroethane	0.5
Benzo(a)Anthracene	0.5	Indeno(1,2,3,c,d)Pyrene	0.5
Benzo(a)Pyrene	0.5	Isophorone	0.5
Benzo(b)Fluoranthene	0.5	Naphthalene	0.5
Benzo(g,h,i)Perylene	0.5	Nitrobenzene	0.5
Benzo(k)Fluoranthene	0.5	N-Nitrosodimethylamine	0.5
Bis(2-Chloroethoxy)Methane	0.5	N-Nitrosodiphenylamine (Diphenylamine)	0.5
Bis(2-Chloroethyl)Ether	0.5	N-Nitrosodi-N-Propylamine	0.5
Bis(2-Chloroisopropyl)Ether	0.5	Phenanthrene	0.5
Bis(2-Ethylhexyl)Phthalate	0.5	Pyrene	0.5
Phenol	0.5	2,4,6-Trichlorophenol	0.5
2-Chlorophenol	0.5	2,4-Dinitrophenol	1.0
2-Nitrophenol	0.5	4-Nitrophenol	0.5
2,4-Dimethylphenol	0.5	2-Methyl-4,6-Dinitrophenol	1.0
2,4-Dichlorophenol	0.5	Pentachlorophenol	1.0
4-Chloro-3-Methylphenol	0.5		
4,4'-DDD	0.02	Endrin Aldehyde	0.02
4,4'-DDE	0.02	g-BHC	0.02
4,4;-DDT	0.08	Heptachlor	0.02
Aldrin	0.02	Heptachlor Epoxide	0.02
a-BHC	0.02	PCB 1016	0.08
b-BHC	0.02	PCB 1221	0.08
Chlordane	0.20	PCB 1232	0.08
Dieldrin	0.02	PCB 1242	0.08
d-BHC	0.02	PCB 1248	0.08
Endosulfan I	0.02	PCB 1254	0.08
Endosulfan II	0.02	PCB 1260	0.20
Endosulfan Sulfate	0.02	Toxaphene	0.20
Eldrin	0.02		

Table 3.39 OPPs in the LSJRB sediments (BES Division Lab. 1992).

	Data Collected During 1989						
Station: Parameter (mg/kg dry basis)	RR 3	RR 5	CR 1	CR 2	CR 3	CR 4	CR 5
Benzo(a)Anthracene	-	-	-	0.16	-	-	-
	-	-	-	0.35	-	-	-
Benzo(a)Pyrene	-	-	0.32	0.43	-	-	-
Benzo(b)Fluoranthene	-	-	0.76	-	-	-	-
Benzo(g,h,i)Perylene	-	-	0.43	0.32	-	-	-
Bis(2-Ethylhexyl)Phthalate	0.16	0.21	0.20	0.19	0.30	0.37	0.35
	-	-	-	0.21	-	0.15	0.15
Chrysene	-	-	0.28	0.22	0.12	-	-
	-	-	0.10	0.41	-	-	-
Dibutyl Phthalate	-	-	-	-	0.16	0.14	-
Diethyl Phthalate	-	-	-	-	-	0.12	-
Fluoranthene	-	0.13	0.37	0.34	0.20	0.12	0.15
	-	-	0.12	0.83	0.17	-	0.12
Indeno(1,2,3,c,d)Pyrene	-	-	0.33	0.20	-	-	-
Pyrene	-	0.14	0.32	0.30	0.11	-	0.13
	-	-	0.11	0.57	0.16	-	0.10
PCB 1260	-	-	-	4.35	1.10	-	-
	-	-	-	3.66	1.20	-	-

Note: OPPs were not detected in Stations RR 13, JC 4, JC 5, JC 6, JC 9 and JC 10.

Table 3.39a OPPs in the LSJRB sediments (BES Division Lab. 1992).

	Data Collected During 1990									
Station: Parameter (mg/kg dry basis)	RR 3	RR 4	RR 5	RR 7	RR 8	RR 11	RR 12	RR 13	NAS	BSY
Benzo(a)Anthracene	-	0.12	-	-	-	-	-	-	-	-
Benzo(a)Pyrene	-	0.11	-	-	-	-	-	-	-	-
Benzo(b)Fluoranthene	-	0.23	0.13	0.14	-	-	-	-	-	-
Benzo(g,h,i)Perylene	-	0.18	-	-	-	-	-	-	-	-
Bis(2-Ethylhexyl)Phthalate	0.40	-	-	0.27	0.14	-	-	-	-	-
Chrysene	-	0.18	-	-	-	-	-	-	-	-
Fluoranthene	-	0.13	0.18	0.14	-	-	-	-	-	-
	-	0.35	-	-	-	-	-	-	-	-
Indeno(1,2,3,c,d)Pyrene	-	0.18	-	-	-	-	-	-	-	-
Naphthalene	0.56	0.62	0.54	0.41	0.60	0.57	0.60	0.60	0.60	0.57
Pyrene	-	0.22	0.14	0.14	-	-	-	-	-	-
	-	-	-	0.14	-	-	-	-	-	-
PCB 1260	1.22	-	-	1.61	-	-	-	-	1.10	-
	-	-	-	0.98	-	-	-	-	-	-

Note: OPPs were not detected in Station FG 6.

Table 3.39b OPPs in the LSJRB sediments (BES Division Lab. 1992).

	Data Collected During 1990				
Station: Parameter (mg/kg dry basis)	CR 1	CR 2	CR 3	CR 4	CR 6
Benzo(a)Anthracene	-	-	0.35	-	-
Benzo(a)Pyrene	-	-	0.33	-	-
Benzo(b)Fluoranthene	-	-	0.62	-	-
Bis(2-Ethylhexyl)Phthalate	0.16	0.12	0.79	0.47	-
Chrysene	-	-	0.56	-	-
Dibutyl Phthalate	-	0.93	-	-	-
Fluoranthene	-	-	0.44	0.28	-
	-	-	0.87	0.29	-
Naphthalene	0.54	0.54	0.54	0.53	0.56
Phenathrene	-	0.40	0.23	0.24	-
Pyrene	-	-	0.25	0.16	-
	-	-	0.56	0.19	-
PCB 1260	-	-	0.84	0.98	-

Note: OPPs were not detected in Stations DI, HC, JUL, RC 1, RC 2 and RC 3.

Table 3.39c OPPs in the LSJRB sediments (BES Division Lab. 1992).

	Data Collected During 1991							
Station: Parameter (mg/kg dry basis)	JC 5	JC 10	RR 11	MC 1	MC 2	R 1	R 2	R 3
1-Methylnaphthalene	<0.14	-	<0.14	<0.98	<0.76	<1.00	<1.10	<1.00
	<0.14	-	<0.14	<0.75	<0.82	<1.00	<0.72	<0.92
1-Methylphenathrene	<0.14	-	<0.14	0.55	<0.76	<1.00	<1.10	<1.00
	<0.14	-	<0.14	3.80	<0.82	<1.00	<0.72	<0.92
2-Methylnaphthalene	<0.14	-	<0.14	<0.98	<0.76	<1.00	<1.10	<1.00
	<0.14	-	<0.14	<0.75	<0.82	<1.00	<0.72	<0.92
2-Chloronaphthalene	<0.14	-	<0.14	<0.98	<0.76	<1.00	<1.10	<1.00
	<0.14	-	<0.14	<0.75	<0.82	<1.00	<0.72	<0.92
2,6-Dimethylnaphthalene	<0.14	-	<0.14	<0.98	<0.76	<1.00	<1.10	<1.00
	<0.14	-	<0.14	<0.75	<0.82	<1.00	<0.72	<0.92
Acenaphthene	<0.14	<0.24	<0.14	<0.98	<0.76	<1.00	<1.10	<1.00
	<0.14	<0.24	<0.14	<0.75	<0.82	<1.00	<0.72	<0.92

Continued:

Table 3.39c OPPs in the LSJRB sediments (BES Division Lab. 1992).
(Continued:)

	Data Collected During 1991							
Station: Parameter (mg/kg dry basis)	JC 5	JC 10	RR 11	MC 1	MC 2	R 1	R 2	R 3
Acenaphthylene	<0.14	<0.24	<0.14	<0.98	<0.76	<1.00	<1.10	<1.00
	<0.14	<0.24	<0.14	<0.75	<0.82	<1.00	<0.72	<0.92
Anthracene	<0.14	<0.24	<0.14	<0.98	<0.76	<1.00	<1.10	<1.00
	<0.14	<0.24	<0.14	<0.75	<0.82	<1.00	<0.72	<0.92
Benzo(a)Anthracene	<0.14	<0.24	<0.14	0.72	0.30	0.65	0.74	0.38
	<0.14	<0.24	<0.14	0.53	<0.82	0.32	1.80	0.18
Benzo(a)Pyrene	<0.14	<0.24	<0.14	0.88	0.34	0.80	1.00	0.40
	<0.14	<0.24	<0.14	0.40	0.42	0.42	2.30	0.15
Benzo(e)Pyrene	<0.14	-	<0.14	0.75	0.26	0.64	0.72	0.27
	<0.14	-	<0.14	0.41	0.26	0.64	1.60	-
Benzo(b)Fluoranthene	<0.14	<0.24	<0.14	0.97	0.67	0.89	1.00	0.43
	<0.14	<0.24	<0.14	0.65	0.46	0.85	2.30	0.24
Benzo(k)Fluoranthene	<0.14	<0.24	<0.14	1.00	<0.76	0.89	1.10	0.43
	<0.14	<0.24	<0.14	0.47	0.40	0.88	2.40	0.24
Benzo(g,h,i)Perylene	<0.14	<0.24	<0.14	0.73	<0.76	0.81	0.82	0.25
	<0.14	<0.24	<0.14	0.41	<0.82	0.91	1.80	-
Biphenyl	<0.14	-	<0.14	<0.98	<0.76	<1.00	<1.10	<1.00
	<0.14	-	<0.14	<0.75	<0.82	<1.00	<0.72	<0.92
Chrysene	<0.14	<0.24	<0.14	1.50	0.40	1.10	1.30	0.64
	<0.14	<0.24	0.09	0.83	0.41	1.00	2.90	0.22
Dibenzo(a,h)Anthracene	<0.14	<0.24	<0.14	<0.98	<0.76	<1.00	<1.10	<1.00
	<0.14	<0.24	<0.14	<0.75	<0.82	<1.00	<0.72	<0.92
Fluoranthene	<0.14	<0.24	<0.14	1.70	0.51	1.20	1.60	0.48
	<0.14	<0.24	<0.14	1.20	0.59	1.30	3.40	0.56
Fluorene	<0.14	<0.24	<0.14	<0.98	<0.76	<1.00	<1.10	<1.00
	<0.14	<0.24	<0.14	<0.75	<0.82	<1.00	<0.72	<0.92
Indeno(1,2,3,c,d)Pyrene	<0.14	<0.24	<0.14	0.82	<0.76	0.63	1.20	0.32
	<0.14	<0.24	<0.14	0.41	<0.82	0.49	2.00	0.13
Naphthalene	<0.14	<0.24	<0.14	<0.98	<0.76	<1.00	<1.10	<1.00
	<0.14	<0.24	<0.14	<0.75	<0.82	<1.00	<0.72	<0.92
Perylene	0.51	<0.24	0.07	0.30	<0.76	0.26	0.27	<1.00
	0.25	<0.24	0.13	<0.75	<0.82	0.30	0.50	<0.92
Phenathrene	<0.14	<0.24	<0.14	0.63	0.12	0.36	0.50	0.08
	<0.14	<0.24	<0.14	0.33	0.15	0.44	0.89	0.08
Pyrene	<0.14	<0.24	<0.14	1.30	0.50	1.30	1.60	0.67
	0.13	<0.24	0.13	0.81	0.63	1.20	3.00	0.28
4,4'-DDD	<0.04	<0.04	<0.06	<0.24	<0.19	<0.25	<0.27	<0.26
	<0.04	<0.04	<0.04	<0.19	<0.20	<0.25	<0.18	<0.23
4,4'-DDE	<0.04	<0.04	<0.06	<0.24	<0.19	<0.25	<0.27	<0.26
	<0.04	<0.04	<0.04	<0.19	<0.20	<0.25	<0.18	<0.23
4,4'-DDT	<0.04	<0.04	<0.06	<0.24	<0.19	<0.25	<0.27	<0.26
	<0.04	<0.04	<0.04	<0.19	<0.20	<0.25	<0.18	<0.23
Aldrin	<0.04	<0.04	<0.06	<0.24	<0.19	<0.25	<0.27	<0.26
	<0.04	<0.04	<0.04	<0.19	<0.20	<0.25	<0.18	<0.23

Continued:

Table 3.39c OPPs in the LSJRB sediments (BES Division Lab. 1992).
(Continued:)

	Data Collected During 1991							
Station: Parameter (mg/kg dry basis)	JC 5	JC 10	RR 11	MC 1	MC 2	R 1	R 2	R 3
a-BHC	<0.04	<0.04	<0.06	<0.24	<0.19	<0.25	<0.27	<0.26
	<0.04	<0.04	<0.04	<0.19	<0.20	<0.25	<0.18	<0.23
b-BHC	<0.04	<0.04	<0.06	<0.24	<0.19	<0.25	<0.27	<0.26
	<0.04	<0.04	<0.04	<0.19	<0.20	<0.25	<0.18	<0.23
d-BHC	<0.04	<0.04	<0.06	<0.24	<0.19	<0.25	<0.27	<0.26
	<0.04	<0.04	<0.04	<0.19	<0.20	<0.25	<0.18	<0.23
g-BHC	<0.04	<0.04	<0.06	<0.24	<0.19	<0.25	<0.27	<0.26
	<0.04	<0.04	<0.04	<0.19	<0.20	<0.25	<0.18	<0.23
Chlordane(alpha)	<0.04	<0.04	<0.06	0.14	<0.19	<0.25	<0.27	<0.26
	<0.04	<0.04	<0.04	<0.19	<0.20	<0.25	<0.18	<0.23
Dieldrin	<0.04	<0.04	<0.06	<0.24	<0.19	<0.25	<0.27	<0.26
	<0.04	<0.04	<0.04	<0.19	<0.20	<0.25	<0.18	<0.23
Dursban	<0.04	<0.04	<0.06	<0.24	<0.19	<0.25	<0.27	<0.26
	<0.04	<0.04	<0.04	<0.19	<0.20	<0.25	<0.18	<0.23
Endosulfan I	<0.04	<0.04	<0.06	<0.24	<0.19	<0.25	<0.27	<0.26
	<0.04	<0.04	<0.04	<0.19	<0.20	<0.25	<0.18	<0.23
Endosulfan II	<0.04	<0.04	<0.06	<0.24	<0.19	<0.25	<0.27	<0.26
	<0.04	<0.04	<0.04	<0.19	<0.20	<0.25	<0.18	<0.23
Endosulfan Sulfate	<0.04	<0.04	<0.06	<0.24	<0.19	<0.25	<0.27	<0.26
	<0.04	<0.04	<0.04	<0.19	<0.20	<0.25	<0.18	<0.23
Endrin	<0.04	<0.04	<0.06	<0.24	<0.19	<0.25	<0.27	<0.26
	<0.04	<0.04	<0.04	<0.19	<0.20	<0.25	<0.18	<0.23
Endrin Aldehyde	<0.04	<0.04	<0.06	<0.24	<0.19	<0.25	<0.27	<0.26
	<0.04	<0.04	<0.04	<0.19	<0.20	<0.25	<0.18	<0.23
Heptachlor	<0.04	<0.04	<0.06	<0.24	<0.19	<0.25	<0.27	<0.26
	<0.04	<0.04	<0.04	<0.19	<0.20	<0.25	<0.18	<0.23
Heptachlor Epoxide	<0.04	<0.04	<0.06	<0.24	<0.19	<0.25	<0.27	<0.26
	<0.04	<0.04	<0.04	<0.19	<0.20	<0.25	<0.18	<0.23
2,3',5-TC Biphenyl	<0.04	<0.04	<0.06	<0.24	<0.19	<0.25	<0.27	<0.26
	<0.04	<0.04	<0.04	<0.19	<0.20	<0.25	<0.18	<0.23
2,3',4,4',5-PC Biphenyl	<0.04	<0.04	<0.06	<0.24	<0.19	<0.25	<0.27	<0.26
	<0.04	<0.04	<0.04	<0.19	0.009	<0.25	<0.18	<0.23
2,2',4,4',5,5'-HC Biphenyl	<0.04	<0.04	<0.06	<0.24	<0.19	<0.25	<0.27	<0.26
	<0.04	<0.04	<0.04	<0.19	0.008	<0.25	<0.18	<0.23
2,2',3,4,4',5'-HC Biphenyl	<0.04	<0.04	<0.06	0.02	0.02	<0.25	<0.27	<0.26
	<0.04	<0.04	<0.04	0.03	<0.20	<0.25	<0.18	<0.23
3,3',4,4',5-PC Biphenyl	<0.04	<0.04	<0.06	<0.24	<0.19	<0.25	<0.27	<0.26
	<0.04	<0.04	<0.04	<0.19	<0.20	<0.25	<0.18	<0.23
3,3',4,4',5,5'-HP Biphenyl	<0.04	<0.04	<0.06	<0.24	<0.19	<0.25	<0.27	<0.26
	<0.04	<0.04	<0.04	<0.19	<0.20	<0.25	<0.18	<0.23
Toxaphene	<0.04	<0.04	<0.06	<0.24	<0.19	<0.25	<0.27	<0.26
	<0.04	<0.04	<0.04	<0.19	<0.20	<0.25	<0.18	<0.23
Dioxins	<0.04	<0.04	<0.06	-	-	-	-	<0.26
	<0.04	<0.04	<0.04	-	-	-	-	<0.23

Table 3.39d OPPs in the LSJRB sediments (BES Division Lab. 1992).

	Data Collected During 1991						
Station: Parameter (mg/kg dry basis)	RB 1	RB 2	NAS 1	NAS 2	NAS 3	CR 1	CR 2
1-Methylphenathrene	-	*	-	*	-	0.25	-
Anthracene	-	*	-	-	-	-	-
Benzo(a)Anthracene	-	*	-	*	-	0.28	0.21
	-	*	-	*	*	0.23	0.14
Benzo(a)Pyrene	-	*	-	*	*	0.35	0.28
	-	*	*	*	*	0.29	-
Benzo(e)Pyrene	-	*	-	*	*	0.29	0.24
	-	*	*	*	*	0.22	0.18
Benzo(b)Fluoranthene	-	0.06	*	-	-	0.45	0.39
	-	*	*	-	-	0.38	0.34
Benzo(k)Fluoranthene	-	0.08	-	-	*	0.45	0.39
	-	*	-	-	-	0.38	0.34
Chrysene	-	0.10	-	*	*	0.50	0.42
	-	*	*	*	*	0.40	0.30
Fluoranthene	*	0.15	*	*	*	0.84	0.77
	-	*	*	*	*	0.67	0.53
Indeno(1,2,3,c,d)Pyrene	-	-	-	-	-	0.13	-
Phenathrene	-	*	-	*	*	0.14	0.13
	-	-	*	*	*	0.14	-
Pyrene	-	-	-	-	-	0.71	0.45
	-	-	-	-	-	0.65	0.40
Chlordane(alpha)	-	-	-	-	-	0.08	-
	-	-	-	-	-	0.07	-
2,3',5-TC Biphenyl	-	-	-	-	-	-	0.024
	-	-	-	-	-	0.055	0.022
2,3',4,4',5-PC Biphenyl	-	-	-	-	-	0.022	-
2,2',4,4',5,5'-HC Biphenyl	-	-	-	-	-	0.022	-
	-	-	-	-	-	0.017	-
2,2',3,4,4',5'-HC Biphenyl	-	-	-	-	-	0.023	-
	-	-	-	-	-	0.018	-

Note: (*) denotes that chemical presence was likely present but unconfirmed.

3.9 Comparison of Sediment Quality Data and Trend Analysis.

Although the existing sediment quality data for the LSJRB is limited, it provides a preliminary basis for planning future sediment management. Starting with the Dames and Moore Study in 1983, the sediment quality database extends to 1991 with the majority of the data collected during 1988. In spite of the fact that a nine-year period for data is relatively short to fully assess the dynamics of sediment contamination or to establish a detailed trend analysis, general conclusions can be made.

The majority of available data are for sediment samples taken from the SJR in the Jacksonville metropolitan area. This region has experienced anthropogenic impacts resulting from intensive urbanization and industrialization (see Tables 3.1 and 3.2). Data were also collected in several tributaries, where low flow conditions (relative to the river) promote sedimentation and subsequent accumulation of toxic substances in the bottom sediments. Based on the six independent sediment quality studies, it is clear that the most contaminated section of the LSJRB extends from Jacksonville upstream to Julington Creek.

3.9.1 Contamination by Toxic Metals. One of the areas with the highest metal enrichment is downtown Jacksonville. Comparing the data collected by the various studies with the sediment quality criteria given by EPA (Chapter II, Table 2.3) it is evident that many sites are moderately to heavily polluted by trace metals. Enrichment by Cr, Cd, Hg, Ni, and Zn were found at Commodore Point, Talleyrand, and the mouth of the Trout River, (Dames and Moore, 1982; Pierce et al. 1988; Savannah Lab. and Environ. Serv. 1989a). The study by the Coastal Zone Management Section, (Savannah Laboratories and Environmental Services, Inc. 1988b), indicated enrichment from only mercury. A year later, the BES Division found elevated levels of Cd, Cr, Pb, and Zn in the mouth of the Trout River. In the Blount Island area, Dames and Moore found enrichment in Cd, Cr, and Hg; the Jacksonville Port Authority noted enrichment in Cr and Zn, while the Coastal Zone Management Section study showed increased Hg levels (Table 3.40). In spite of the slight differences in the type or number of toxic metals, the

Table 3.40 Metal enrichment in LSJRB tributaries based on all sediment quality studies.

Study	Site	Cd	Cr	Cu	Hg	Ni	Pb	Zn
MML	Arlington R.				+		+	+
CZM		+			+		+	+
BES		+					+	+
MML	Broward R.	+			+		+	+
CZM								
BES		+		+		+	+	+
MML	Cedar R.	+	+	+	+		+	+
CZM		+			+		+	+
BES		+	+	+		+	+	+
MML	Cedar/Ortega	+		+	+		+	+
BES		+		+		+	+	+
MML	Moncrief Cr.	+	+		+		+	+
BES		+	+	+	+		+	+
MML	Ortega R.	+	+	+	+		+	+
CZM		+			+			+
BES		+		+		+	+	+
MML	Ribault R.	+	+		+		+	+
BES		+					+	+
MML	Rice Cr.	+		+	+			+
BES		+		+	+		+	+

three studies of tributaries consistently found high metal enrichment in all stations except the Broward River. The Coastal Zone Management Section (Savannah Laboratories and Environmental Services, Inc. 1988b) found no metal contamination, at that location. These differences indicate that contaminant level at the same site can vary. However, all of the studies conducted between 1982-1991 found that the sediments have been contaminated by toxic metals. This indicates that either pollution sources are still active and contributing heavy metals to the system or that deposited sediments are rarely subject to resuspension events or both of the above.

3.9.2 <u>Contamination by Organic Pollutants.</u> All of the studies support the fact that the area from the Trout River to Julington Creek is extensively contaminated by PAHs and chlorinated pesticides. Contamination by PCBs is present, but to a lesser extent than by other organic compounds. Negligible levels of Cl-pesticides and PCBs were found by Dames and Moore (1983) and Coastal Zone Management, FDER (Savannah Laboratories and Environmental Services, Inc. 1988b) in this part of the system. However, both studies were concentrated in the main channel instead of the tributaries or other protected site.

In order to assess the environmental state of the system, the individual and cumulative contamination factors (C_i/C_{oi} and C_d) as suggested by Hakanson (1980) (see Chapter II) were applied for the most recent data (1991) of the Bio-Environmental Services Division Laboratory (Table 3.41). The cumulative contamination factor which is based on the heavy metal and PCB concentrations is indicative of the environmental conditions. From these data it is evident that out of fifteen sampling stations, five (MC 1, MC 2, R 1, R 2, R 3) were highly contaminated, i.e., $C_d \geq 32$, one (Cr 1) had a considerable degree of contamination, i.e., $16 \leq C_d < 32$, three were moderately polluted (RR 11, NAS 2, CR 2) and six (JC 5, JC 10, RB 1, RB 2, NAS 1, NAS 3) had a low degree of contamination, i.e., $8 \leq C_d < 16$. Data for the rest of the sampled stations showed negligible contamination, i.e., $C_d < 8$. Exceedence of the EPA sediment quality criteria, high Hakanson's contamination factors, elevated metal:aluminum ratios, etc. were indicative of moderate to high anthropogenic contamination at various sites of the LSJRB. Although the exact environmental impact cannot be assessed based on the existing data, it is reasonable to believe that violation of these environmental criteria, regardless of their objectivity, may have serious negative effects on the ecosystem.

Table 3.41 Hakanson's contamination factors based on the BES Div. Lab. data.

	C_i/C_{oi}								C_d
Station	**As**	**Cd**	**Cr**	**Cu**	**Hg**	**Pb**	**Zn**	**PCB**	
JC 5	0.167	1.050*	0.322	0.410	0.480	0.671	0.749	4.00+	7.85
JC 10	0.100	0.990	0.189	0.266	0.600	0.614	0.469	4.00+	7.22
RR 11	0.247	1.540*	0.611	0.592	1.840*	0.728	0.777	6.00+	12.34*
MC 1	1.053*	1.870*	0.089	5.160+	1.880*	2.530*	2.274*	24.00★	38.86★
MC 2	0.480	1.560*	1.556*	5.560+	2.080*	2.540*	1.737*	20.00★	35.51★
R 1	0.253	2.910*	0.822	1.460*	1.960*	2.143*	2.286*	25.00★	36.83★
R 2	0.307	2.740*	0.833	1.290*	1.960*	1.628*	2.189*	27.00★	37.95★
R 3	0.340	1.930*	0.922	1.152*	1.720*	1.743*	1.280*	-	35.09★
RB 1	0.300	0.840	0.600	0.384	1.280*	0.457	0.486	-	4.35
RB 2	0.207	0.690	0.300	0.586	0.680	0.586	0.869	-	3.91
NAS 1	0.360	1.490*	0.911	0.594	1.440*	0.843	0.828	-	6.47
NAS 2	0.253	3.890+	0.989	0.674	1.120*	1.214*	0.914	-	9.05*
NAS 3	0.347	1.470	0.978	0.492	1.600*	0.857	0.703	-	6.45
CR 1	0.580	3.590+	0.900	0.268	0.960	4.700+	4.930+	5.5+	23.84+
CR 2	0.380	1.640*	0.456	0.896	0.520	1.667*	1.983*	2.2+	9.73*

Note:
* : moderate degree of contamination
+ : considerable degree of contamination
★ : very high degree of contamination

H. FUTURE NEEDS OF SEDIMENT QUALITY DATA.

3.10 Sediment Quality Data.

To determine the status and health of an aquatic ecosystem, it is important to collect and analyze both long-term water quality and sediment quality data. Water quality data is useful because it provides information about short-term conditions. Indeed, water circulation and mixing caused by tides and winds can rapidly change water quality conditions; thus, only short-term effects can be detected. Interpreting long-term water quality data is difficult because water quality changes quickly as a result of numerous factors, e.g., rain storms, tides, seasons. On the other hand, sediment quality is indicative of long-term, cumulative impacts and conditions. Indeed, analysis of undisturbed bottom sediment layers can provide a detailed short-term (surficial samples) or long-term (core samples) history of pollution. By estimating particle deposition rates and analyzing sediment quality characteristics layer-by-layer, a chronological relationship between anthropogenic events and sediment contamination can be developed (Figure 3.18). In Figure 3.18, the concentration of aliphatic and aromatic hydrocarbons in the bottom sediments of Lake Zurich is given along with an approximate age of deposition (Wakeham et al. 1980). Examination of the concentrations revealed that the reduction in aromatic hydrocarbons around 1955 coincided with the closing of a major coal-carrying railway. Of course, this type of sediment analysis is meaningful only for sites where there is net sediment deposition.

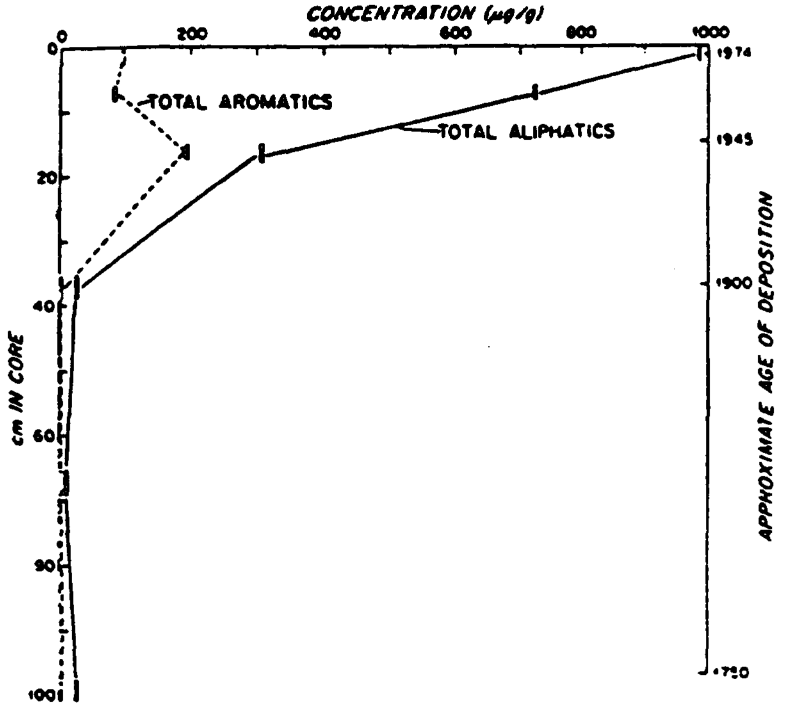


Figure 3.18 PAHs distribution in a sediment core from Lake Zurich (Wakeham et al. 1980).

Examination of the top few centimeters of sediments provides information about the recent history of contamination. This type of data is particularly useful for active (movable) bottoms, since it can establish spatial sediment-contaminant distribution. All of the studies conducted in the LSJRB were focused on the top layer and not on sediment core profiles. Sediment quality data can be combined with data pertaining to documented or suspected point and non-point pollution sources so that a cause/effect relationship can be developed. This method relates changes in chemical components (e.g., toxic metals, organic pollutants) to changes in effluent discharges or use of adjacent watersheds. In case a pollution source ceases to exist, periodic or continuous monitoring of sediment quality will document the rate of sediment clean-up by natural processes. Based on such information, decisions can be made with respect to management actions that should be taken to remediate contamination, permit point sources, or control non-point sources.

A database containing all sediment data should be developed to facilitate the analysis. This database should include the exact location of the sampling station, the date the sample was taken, the prevailing environmental conditions during the sampling event, qualitative and quantitative contaminant analyses, and, if possible, identification of the source. Based on the spatial distribution of the sampling stations evaluation of the impact of point and non-point pollution sources should be assessed. Comparing data on pollution sources to historical sediment quality records will permit a determination of environmental trends. Cross-correlations of the sediment quality data with other pertinent data (e.g., anthropogenic activities, weather pattern, episodic events, etc.) should be used to develop cause/effect relations. The establishment of site-specific relations for each contaminant or group of contaminants (e.g., nutrients, trace metals and organic compounds), can lead to the development of management decisions pertaining to control of pollution sources or to remediation of sediment quality. The database should be user-friendly, have a quality control/quality assurance system for data input, and provide quantitative sediment quality indices for easy comparison and evaluation of the data.

3.11 Sub-basin Prioritization Based on Sediment Pollution Data.

Based on the number of contaminated sites in each sub-basin, a preliminary prioritization for protection or remediation of LSJRB can be made (Table 3.42). The Cedar River sub-basin, the Trout River sub-basin, and sites near the Jacksonville downtown area of the SJR sub-basin are the most seriously contaminated areas and may require remediation.

Based on the existing data, application of trends analysis is not feasible since sediment quality information is very sparse and most stations have only been sampled once, sample collection and analysis was not uniform for all of the studies, and detailed information on the prevailing hydrodologic conditions is missing. Since so little data is available there may be other contaminated locations that have not been identified yet.

For a comprehensive understanding of sediment quality dynamics a continuous monitoring program must be established. The program should emphasize establishment of trends, parameter correlations, and cause/effects relationships rather than simply collect sediment samples and estimate their contaminant content. Sampling stations can be added or deleted from the monitoring program depending on documented or suspected environmental degradation or improvement respectively.

Table 3.42 Prioritization of remediation/protection of LSJRB sub-basins based on sediment quality data.

Sub-basin	Tributary or Location with Sediment Quality Problems
Ortega River	Ortega River Cedar River Fishing Creek Butcher Pen Creek
Trout River	Trout River Moncrief Creek Ribault River Sixmile Creek
St. Johns River	NAS Blount Island Talleyrand McCoy Creek Big Fishwier Creek Goodbys Creek Cedar Creek
Arlington River	Arlington River Pottsburg Creek
Broward River	Broward River Cedar Creek Little Cedar Creek
Sixmile Creek	Sixmile Creek Trout Creek
Julington Creek	Julington Creek Durbin Creek
Rice Creek	Rice Creek
Crescent Lake	-
Black Creek	-
Dunn Creek	-
McCullough Creek	-

Note: The ranking is from highest priority to lowest priority.

3.11.1 Optimal Sampling Locations. Sampling locations for future studies should preferably be selected using the following three criteria: a) existence of available sediment quality data and information, b) high levels of pollutant concentrations in water or sediments, and c) areas with documented or suspected contaminant sources. Data should be collected twice per year (dry and wet seasons) every other year for each group of stations. For effective interpretation of the data, hydrologic conditions during the previous ten to fifteen days should be also collected. The hydrologic data will help to assess pollution inputs from watershed drainage. Based on the existing sediment quality data, the following sampling schedule is suggested:

Year 1:	Station 1A - Confluence of Cedar/Ortega Rivers (1)
	Station 2A - Mouth of Ortega River (2)
	Station 3A - Upstream in Cedar River (2)
	Station 4A - Mouth of Julington Creek (1)
	Station 5A - Upstream in Durbin Creek (2)
	Station 6A - Rice Creek (1)
Year 2:	Station 1B - Mouth of Broward River (2)
	Station 2B - Mouth of Trout River (1)
	Station 3B - Confluence of Trout and Ribault River (2)
	Station 4B - Main River off NAS (1)
	Station 5B - Mouth of Arlington River (1)
	Station 6B - Upstream in Pottsburg Creek (2)

Such a schedule capitalizes on existing information and will provide an effective picture of the sediment quality conditions in the LSJRB. Numbers in parentheses next to each sampling location indicate the first and the second choices if the number of stations must be reduced by half due to budgetary constraints.

For each site three samples should be collected from the surficial layer (top 5 cm) and analyzed for nutrients, heavy metals, and organic compounds according to EPA/FDER approved methods. During the first sampling period a core sample should also be obtained (top 50 cm) in order to establish the history of sediment pollution.

A monitoring plan like the one described above, will provide an assessment of the effects of remediation or permit changes and would help to develop effective management decisions for the Lower St. Johns River Basin.

CHAPTER IV

SEDIMENT CHEMISTRY AND QUALITY INDICES

I. SEDIMENT QUALITY MODELING, SAMPLING, AND ANALYSIS

4.1 Modeling of Sediment-Induced Contamination.

While considerable scientific inquiry has centered on dissolved contaminants and their impacts on aquatic ecosystems, less is known about sediment-bound pollutants. However, polluted sediments can be a major source of aquatic contamination. Assessment of the potential for environmental impact caused by contaminated sediments requires simulation of dynamics and chemical behavior of sediment-bound pollutants specifically because sediment-bound pollutants behave different from dissolved pollutants (Uchrin and Weber, 1980). Mathematical models can be used to describe sediment transport and distribution (Chapter I, Section 1.12). Since the transport of particulate-bound pollutants is ultimately related to the transport of solid particles, both physical and chemical properties must be examined. Some of the physicochemical properties, e.g., diffusion coefficients, partitioning coefficients, etc., must be measured for each sediment type. How well the simulated system predicts actual conditions depends on the validity of the assumptions made and the similarity between the natural and the laboratory conditions under which the physicochemical coefficients were estimated. For example, experiments for estimation of the rates of erosion have shown that results can vary substantially by using either remolded or naturally deposited beds. The former bed erodes with a linear rate while the latter erodes with an exponential rate. Increased or decreased salinity in the water used for the experiments can also vary the erosional rates.

4.1.1 **Modeling of Physical Processes.** The first step for any sediment contamination study is to quantify the sediment budget. Various methods and approaches are used to prepare a sediment budget. All of these methods are based on the simple concept of sediment mass balance, i.e., (sediment inflow) - (sediment outflow) = (change in sediment volume). Application of the mass balance principle requires definition of the natural boundaries of the system under consideration. For large systems, since the areal distribution of the sediment mass is not uniform, the system is subdivided into a number of subareas where sediment distribution is assumed to be constant. Then, the mass conservation principle is applied for each individual subarea. The major physical processes required for quantification of sediment movement include advection by the moving water, dispersion and molecular diffusion, and erosion/deposition. Assuming a control volume (single-area system) the time-dependent, three-dimensional mass balance equation that incorporates all of the sediment movement processes reads

$$C_{,t}+(uC)_{,x}+(vC)_{,y}+(wC)_{,z}-(E_xC_{,x})_{,x}-(E_yC_{,y})_{,y}-(E_zC_{,z})_{,z}\pm KC^n = 0 \quad \ldots\ldots \quad (4.1)$$

where C is the concentration of the suspended sediment, U = {u,v,w} are the velocity components, E_x, E_y, E_z are the turbulent diffusion coefficients, K, n are constants defining the rate of sediment erosion or deposition, and x, y, z are the Cartesian coordinates. The first term in Eq. (4.1) shows the temporal change in sediment concentration, the next three terms show the sediment transported by the moving water, the following three terms show the diffusion effects, while the last term defines the erosion/deposition processes. Solution of Eq. (4.1) quantifies the sediment concentration at every point both in space and time. Subdividing the solution domain into a grid of hexahedral elements, the mass balance equation can be simplified considerably. Assuming the case of a two-dimensional, horizontal rectangular-grid system where an element (subarea) i is surrounded by elements (subareas) j, k, l and m (Figure 4.1), the total advective sediment mass flux F_{TA} in element i is the sum of the individual fluxes through each of its four sides,. This can be formulated mathematically as:

$$F_{TA} = Q_{ji}(a_{ji}C_j + b_{ji}C_i) - Q_{il}(a_{il}C_i + b_{il}C_l) + Q_{mi}(a_{mi}C_m + b_{mi}C_i) - Q_{ik}(a_{ik}C_i + b_{ik}C_k) \qquad (4.2)$$

where Q_{ji} is the flow across the interface of elements j and i, C_i is the average sediment concentration within the element i, and a_{ji} and $b_{ji} = 1 - a_{ji}$ are weighing factors derived

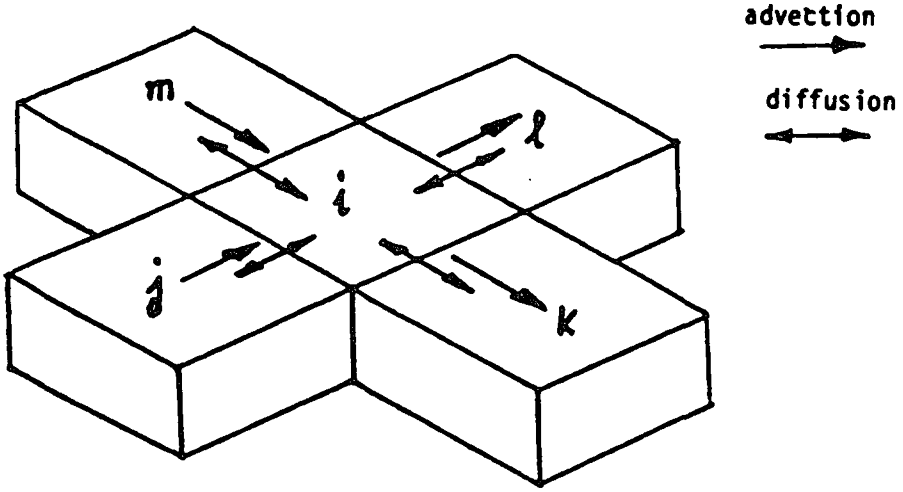


Figure 4.1 Finite-element configuration for sediment-pollutant mass balance.

empirically at the interface of the subscribed elements. For the same single-component configuration, the total diffusive sediment transport F_{TD} through element i can be similarly written as:

$$F_{TD} = E_{ji}A_{ji}(C_j-C_i)/L_{ji}+E_{li}A_{li}(C_l-C_i)/L_{li}+E_{mi}A_{mi}(C_m-C_i)/L_{mi}+E_{ki}A_{ki}(C_k-C_i)/L_{ki} \quad (4.3)$$

where A_{ji} is the interface area between elements j and i, and L_{ji} is a characteristic mixing length (usually of the order of the average length of elements j and i). Combining Eqs. (4.2) and (4.3), the total sediment mass balance can be estimated as

$$V_iC_{,t} = F_{TA} + F_{TD} \pm V_iK_iC_i^n \pm S_{iE} \quad \ldots\ldots\ldots\ldots \quad (4.4)$$

where V_i is the volume of element i, and S_{iE} is any external source/sink that affects element i directly. This way, quantification of the sediment mass transport is substantially simplified since the original partial differential equation (Eq. (4.1)) reduces to a system of ordinary differential equations (ODE) (Eq. (4.4)) that can be solved numerically by using a Runge-Kutta integrating method. Under steady-state conditions, the ODE system reduces to a system of algebraic equations that can be solved by using the Gauss-Seidel algorithm.

4.1.2 Modeling of Chemical Processes. For sediment-bound pollutants, the fate of a pollutant P can be assessed by using a coupled approach, i.e., by coupling the sediment mass balance with the pollutant mass balance equations :

$$VC_{,t} = [\text{Sediment Flux}] \pm K_CC^n \quad \ldots\ldots\ldots\ldots \quad (4.5)$$

$$VP_{,t} = [\text{Pollutant Flux}] \pm K_PP^m \pm A_C \quad \ldots\ldots\ldots\ldots \quad (4.6)$$

where K_C, n, and K_P, m are rate law parameters for the sediment and the pollutant, and A_{CP} is a function which quantifies the effects of the sediment dynamics on the pollutant. The function A_{CP} and the rate law parameters are unique for every polluted sediment system and must be determined experimentally. Eqs. (4.5) and (4.6) state that considering a control volume V, the change of sediment/pollutant mass in time equals the net sediment/pollutant flux (entering/exiting) through the sides of the control volume plus the production/reduction of sediments/pollutants within the control volume.

To model the chemical reactions which occur in an aquatic ecosystem, the following information is usually required:

- o quantitative data related to the physicochemical parameters of the water-sediment system,
- o thermodynamic and stoichiometric data referred to the equilibrium chemical state of the system, and
- o Analytical expressions for the kinetics of the system; i.e., definition of the instantaneous rate of change given its present state.

Complete kinetic data is rarely available. However, by using whatever information is available and comparing the time scale (minutes, days, years) of the process under consideration, a pseudo-equilibrium steady-state model can be built whenever the occurring reactions are faster than the time scale of interest (Morel and Yeasted, 1977). Chemical equilibrium models for aquatic systems, expressed in terms of constant temperature and constant pressure conditions, are based on either the minimum Gibbs free energy or the mass law equations (Deland, 1967; Morel and Yeasted, 1977). In both cases the model constraints are: a) the concentrations of the complexes must be non-negative, and b) the mole balances must be satisfied according to the given analytic expressions.

Assessing the potential impact of pollution on an aquatic ecosystem depends on the measurement of pollutants such as heavy metals, oils, chlorinated hydrocarbons, pesticides, other organic priority pollutants, and radionuclides. Modeling of the chemical behavior of these toxic substances can be divided into three categories (Jorgensen, 1983):

o physical processes, including: adsorption and volatization;

o chemical processes, including: oxidation, photolysis, hydrolysis, ionization, and complexation; and

o biological processes, including: biodegradation, uptake, and excretion.

4.1.2.1 Adsorption. Adsorption of chemical elements or compounds on solid particulates is a very important process for sediment quality studies. Thermodynamically, adsorption is defined as the state at which the pollutant fugacities ("escaping tendencies") are the same in both the adsorbed and the dissolved phases; i.e.,

$$f_s = f_w \tag{4.7}$$

where s and w denote the adsorbed and dissolved phases respectively. Sediments are composed of a mixture of different sorbents so that the adsorbed pollutant concentration S can be defined as

$$S = f_s \, \Sigma(\epsilon_i/\phi_{si}) \tag{4.8}$$

where ϵ_i is the weight fraction of the various components (i), and ϕ_s is the sediment fugacity coefficient. The fugacity of the dissolved phase ϕ_w is related to the dissolved pollutant concentration C by

$$f_w = \phi_w C \tag{4.9}$$

Using Eqs (4.7) to (4.9), the dissolved and adsorbed pollutant concentrations can be combined as

$$S = \phi_w \Sigma(\epsilon_i/\phi_{si})C = K_p C \tag{4.10}$$

where K_p is the partitioning coefficient. Since the fugacity coefficients depend on the pollutant concentration C, Eq. (4.10) is nonlinear. For sufficiently low concentration systems, Eq. (4.10) approaches linearity and K_p is independent of C.

Estimation of pollutant exchange rates between sediment and water through the processes of adsorption/desorption is essential for management of estuarine contaminated sites. This estimation assists in assessing the significance of bottom sediments during resuspension events. Quantification of the partitioning coefficients can be done by conducting either *in-situ* or laboratory experiments. These experiments require monitoring the changes in pollutant concentration in the water column, which results from pollutant release (or capture) by the suspended sediment particles. Laboratory experiments for estimation of partitioning coefficients can be conducted by using different apparatus such as cylindrical columns, oscillating-grid tanks, etc. In all of these experiments contaminated sediments are mixed with uncontaminated water and then the sediments are allowed to settle. The partitioning coefficient is calculated by a comparison of the contaminant concentration in the water before and after mixing.

Adsorption data usually fit different adsorption isotherm equations given in a form similar to that of Eq. (4.10). A list of the most well-known adsorption isotherms is given in Table 4.1. The Langmuir and Freundlich adsorption isotherms are the most commonly used, not only because they adequately represent the partitioning process but also because their adjustable parameters i.e., partitioning coefficient (K), adsorption maximum (M), and heterogeneity coefficient (b) (see Table 4.1) can be easily estimated either graphically or by linear regression. All other isotherms given in Table 4.1 require nonlinear regression for the estimation of their adjustable parameters which usually

Table 4.1 Adsorption isotherm equations (Kinniburgh, 1986)

Isotherm	Equation	No. of adjustable parameters
Langmuir:	$S = KCM/(1+KC)$	2 (K,M)
Freundlich:	$S = (KC)^b$	2 (K,b)
Langmuir-Freundlich:	$S = (KC)^b M/[1+(KC)^b]$	3 (K,M,b)
Redlich-Peterson:	$S = KCM/[1+(KC)^b]$	3 (K,M,b)
Toth:	$S = KCM/\{[1+(KC)^b]^{1/b}\}$	3 (K,M,b)
Multisite Langmuir:	$S = \Sigma_{i=1,m} [K_iCM_i/(1+K_iC)]$	2m (K_i,M_i)
Sum of Freundlichs:	$S = \Sigma_{i=1,m} (K_iC)_b i$	2m (K_i,b_i)
Dubinin-Radushkevich:	$\log S = -b[\log^2(KC)+\log M$ $\log S^* - \log C^* + \log C \; ; C < C^*$	3 (K,M,b)
Modified Dubinin-Radushkevich:	$\log S =$ $-b\log^2[KC/(1+KC)]+\log M \; ; C > C^*$	3 (K,M,b)

Notations: S = amount adsorbed; C = equilibrium solution concentration; K = partitioning coefficient; M = adsorption maximum; m = affinity parameter; b = parameter depending on the degree of heterogeneity; (S^*,C^*) is the point where $\log S_{,\log C} = 1$.

involves minimization of the residual sums of squares. Most of the commercially available statistical computer software packages contain subroutines for nonlinear regression analysis. Kinniburgh (1986) suggested that, for environmental applications, the multisite Langmuir, modified Dubinin-Radushkevich, and Toth isotherms provide more flexibility and accuracy than the Langmuir or Freundlich isotherms and that they are particularly suitable for adsorption on heterogeneous surfaces. Chemical equilibrium adsorption data can be effectively analyzed with existing models such as FITEQL (Westall, 1982) or ISOTHERM (Kinniburgh, 1985). For simplicity, in water-sediment quality models adsorption is usually treated by first-order kinetics; e.g., HSPF (Johanson et al. 1984); SERATRA (Onishi and Wise, 1982).

Information on chemical partitioning to mineral soils and biota can be found in Ambrose et al. (WASP3, 1986) and Brownwell et al. (1990). Data on chemical partitioning to natural aggregates and colloids was given by Gschwend and Wu (1985), and Gschwend et al. (1986). Dzombak and Morel (1987), and DiToro et al. (1986) presented comprehensive studies of adsorption of inorganic pollutants. Adsorption-desorption of various organic pollutants has also been studied extensively; e.g., PCBs (DiToro and Horzempa, 1982); polyaromatic hydrocarbons (Vowels and Mantoura, 1987); pesticides (Drexter, 1979) and other organic chemicals (Karickhoff, 1984).

Besides the electrochemical mechanisms controlling adsorption, it has been found that adsorption depends also on the suspended particle concentration (O'Connor and Connolly, 1980; Mackay and Powers, 1987), the particle interaction (DiToro, 1985), and the particle size and shape (Mathews and Zayas, 1986).

4.1.2.2 Volatilization. Volatilization occurs along a fluid-gas interface. In an aquatic ecosystem, volatilization across the interface of the water surface and the air is quantified as

$$K_v = A/\{V[1/K_l+RT/(H_cK_g)]\} \quad \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots \quad (4.11)$$

where K_v is the overall transfer coefficient, A is the area of the water-air interface, V is the water volume, R is the water vapor gas constant, T is the temperature, K_l is the liquid film mass transfer coefficient, K_g is the gas film mass transfer coefficient, and H_c is Henry's law constant. Generally, volatilization is not a particularly important process for contaminated sediments so that it can be neglected.

4.1.2.3 Oxidation and Reduction. Oxidation and reduction are processes which affect the behavior of major ions and trace metals (particularly iron and manganese). Whenever there is chemical exchange between suspended particles and bed sediments, oxidation and reduction become very important processes (Lu and Chen, 1977).

Precipitates forming during oxidation of iron and manganese provide a substratum that is suitable for further sorption of toxic metals. Reduction of sulfate to sulfide can result in the formation of metal sulfides (FeS, CdS, ZnS) that subsequently can affect dissolved toxic metal concentrations. While iron oxidation depends on a number of parameters, such as Fe^{2+} concentrations, pH, and the partial pressure of oxygen, manganese oxidation can be effectively described by simple first-order kinetics. Environments with highly active oxidative or reductive processes cannot be simulated accurately due to the complexity of the chemical reactions involved (Medine and McCutcheon, 1989). Oxidation and mobility of toxic metals are included in some complex chemical equilibrium models such as MINTEQA1 (Brown et al. 1987).

Under certain environmental conditions (e.g. aerobic) oxidation can also be an important process for toxic organic pollutants. Simulation of oxidation of organics is generally accomplished by using a second-order rate equation as

$$P_{,t} = k_{ox}[O_x]P \quad \ldots\ldots\ldots\ldots\ldots\ldots \quad (4.12)$$

where P is the concentration of the organic compound, $[O_x]$ is the concentration of the oxidant, and k_{ox} is the second-order rate constant. The kinetics of redox processes for organic compounds are still not well-understood.

Generally, redox can be measured by means of electroanalytical procedures which are based upon the potential-current behavior of a polarizable electrode placed in the solution being analyzed. In order to ensure polarization the dimensions of the electrode are made very small (Skoog and West, 1971). Techniques for measuring the redox potential in aquatic environments include: polarography, linear sweep voltammetry, or cyclic voltammetry. The redox potential depends primarily on the concentration of the contaminant. However, temperature, pH, and the presence of other chemicals can affect its magnitude. Therefore, seasonal variations in the redox potential can occur. However these variations may be hard to predict because of the complexity of the components involved. Standard values of the redox potential are given in Handbook of Chemistry and Physics (1975-76).

4.1.2.4 <u>Photolysis.</u> Photolysis is an important chemical process for many toxic organic compounds. The rate of direct photochemical transformation is given as:

$$S_{,t} = - eI_l\phi \qquad (4.13)$$

where S is the pollutant concentration, e is the molecular light extinction coefficient, I_l is the intensity of the incident light, and ϕ is the quantum yield. The quantum yield is indicative of the efficiency by which absorbed light is converted into chemical energy. Photolytic effects are more pronounced in shallow waters. In estuarine systems, sediment resuspension or inflow from sediment laden inland runoff reduces the intensity of the incident light I_l and eventually decreases the rate of photochemical transformation.

4.1.2.5 <u>Hydrolysis.</u> Hydrolysis is the process by which a chemical compound reacts with water. A hydroxyl group (OH^-) is generally introduced into the compound as a result of this process. Hydrolysis of metals results primarily in the formation of inorganic complexes (see section 4.1.2.7). For organic compounds containing a halogen or other leaving group X, hydrolysis may be exhibited by the following equation:

$$RX + H_2O = ROH + HX \qquad (4.14)$$

The kinetics of hydrolysis for acids and bases can be described as

$$K_h[S] = K_B[OH^-][S] + K_A[H^+][S] + K_N[H_2O][S] \qquad (4.15)$$

where K_h, K_A, K_B, K_N are rate constants. The applicability of Eq. (4.15) is hindered by the fact that very often data is insufficient to establish the behavior of K_h as a function of the pH and temperature for a wide variety of organic pollutants.

Hydrolysis along with ionization and complexation control the molecular or ionic form of the contaminant in the water column and interstitial water. This subsequently affects the dissolved, interstitial, and sediment-bound contaminant concentrations by changing either the equilibrium conditions or the kinetics of chemical precipitation/dissolution or adsorption/desorption processes. The reactivity or the toxicity of the pollutant may be substantially altered by means of these relatively simple processes. Therefore, in estuarine contamination studies it is important to quantify as many as possible of these chemical processes.

4.1.2.6 **Ionization.** Ionization is a very fast reaction where a molecule is separated or dissociated into particles of the opposite charge (Lassiter, 1978). Since the reaction is rapid, an equilibrium expression can be used effectively for modeling purposes. For weak acids in water, the relationship between the concentrations of hydrogen ions $[H^+]$, dissociated anions $[A^-]$, and associated acid molecules $[HA]$ is described by the acid dissociation constant K_A:

$$K_A = [H^+][A^-]/[HA] \quad (4.16)$$

The ratio of the concentration of hydroxyl ions $[OH^-]$ and a weak base $[B^+]$ to the neutral species $[BOH]$ is described by the base dissociation constant K_B as:

$$K_B = [OH^-][B^+]/[BOH] \quad (4.17)$$

Ionization is very important for trace metals, especially in aquatic zones of vigorous mixing or when there is a rapid change in the pH. Ionization is also important for volatile or positively charged organic pollutants (Medine and McCutcheon, 1989).

4.1.2.7 Complexation. Complexation is basically the process where two species react chemically to form a third species. Metal complexation can be described in general as:

$$mM + nL + hH \leftrightarrow M_mL_nH_h \quad \ldots\ldots \quad (4.18)$$

where M is the metal (e.g., Ag, Cd, Cu, Hg, Pb, Zn, etc); L is the ligand (e.g., OH^-, Cl^-, F^-, S^{2-}, CO^{23-}, SO^{24-}, NH^3, amino acids, humates, fulvates, etc); H is the proton; and m, n, h are stoichiometric coefficients. For natural systems, especially those which involve dispersive type flows, the equilibrium assumption for chemical complexation is very realistic. For short-time simulations (e.g., minutes to hours), complexation kinetics may involve a pseudo-equilibrium expression. Two comprehensive models that simulate metal complexation in natural aquatic systems are the MINTEQ (Felmy et al. 1984) and the MINEQL (Westall et al. 1976).

4.1.2.8 Biodegradation. Biodegradation of a toxic compound can be described by the Monod analogy to the Michaelis-Menton enzyme kinetics as:

$$S_{,t} = -(\mu/Y)/X = -(\mu_{max}/Y)[SX/(K_s+S)] \quad \ldots\ldots \quad (4.19)$$

$$X_{,t} = \mu X = \mu_{max}[SX/(K_s+S)] \quad \ldots\ldots \quad (4.20)$$

where S is the concentration of the substrate, X is the biomass per unit volume, Y is the cell yield, μ is the specific growth rate, and K_s is the half-saturation constant. If $K_s >> S$, then the substrate decay process is of the first-order for both the substrate and the biomass. Prediction of biodegradation is limited by the inability to reproduce natural bacterial population growth and the lack of accurate field data on pH, nutrients, and geochemistry.

4.1.2.9 Biological Uptake and Clearance. Biological uptake for an organism can be quantified in terms of the following energy balance equation:

$$e_f = \dot{M}_r + W_{,t} = aW^b + W_{,t} \tag{4.21}$$

where e_f is the energy of the ingested ration including fecal and other losses, M_r is the metabolic rate, W is the weight of the organism, and a, b are empirical coefficients. The value of the exponent b ranges between 0.75-0.80 and is independent of the level of metabolism (Norstrom et al. 1976). In addition to the uptake, two other processes of major importance to biota are pollutant bioaccumulation and pollutant clearance.

Bioaccumulation of toxic substances T_c is proportional to the uptake from both water (respiration) and food. This is shown by the equation

$$T_{c,t} = e_{pf}C_{pf}R + e_{pw}C_{pw}V \tag{4.22}$$

where R is the ingestion rate, V is the respired volume of water, C_{pf}, C_{pw} are the toxic substance concentrations in the food and water, and e_{pf}, e_{pw} are the efficiencies of toxic substance uptake from food and water respectively. The respired volume of water is proportional to the metabolic rate and inversely proportional to the oxygen concentration.

Body clearance of toxic compounds follows a first-order relation; i.e.,

$$T_{c,t} = k_c T_c W^a \tag{4.23}$$

where k_c is the clearance coefficient, and a is a constant ($-0.2 \leq a \leq -0.8$). The clearance coefficient depends on the type of toxic compound.

Many of the aforementioned physicochemical processes have been successfully incorporated into a number of available computer models. These models mainly utilize three interacting compartments; i.e., water, sediment, and suspended particulates, e.g., SERATRA, (Onishi and Wise, 1982). In addition to these three compartments, some models include simulation of contaminant transport within the food chain (invertebrates, plants, and fish). Examples of compartment models that include food chain behavior are: Miller's model (1979) for mercury contamination, Gillett's et al. model (1974) for pesticide contamination, Lassiter's model (1978) for methyl parathion and benzothiophene contamination, etc.

4.1.3 Statistical Modeling.

Besides the deterministic methods for assessment of pollutant transport and exchange between water, sediments and biota, there are also some statistical approaches that are very useful for environmental impact assessment. These approaches provide data regarding the frequency of occurrence and duration of pollutant concentrations associated with certain hydrological events (e.g., surface runoff, high winds, etc.) or human actions (e.g., dredging, water treatment operations, etc). These data which are mostly site-specific are subsequently used to determine the potential and chronic impact of pollutants on aquatic biota and eventually on humans. This is accomplished by correlating and comparing a sequence of actions and events that had a negative impact on humans and the environment. The inference of the correlations is based on field and laboratory observations, past experiences, theoretical principles, and any other data that support a cause/effect relation. One such model that evaluates the risk of exposure to pesticides is the FRANCO model (Olsen and Wise, 1979).

In spite of the usefulness and simplicity of the statistical models, they must be used with caution. Unless all of the conditions under which the correlations of these models were established are satisfied, the results may be misleading and inaccurate. For this reason, development of a good statistical model requires a wide breadth of data and a good documentation of its assumptions and applicability conditions.

4.1.3.1 Geostatistics.

A very powerful statistical tool for analysis of contamination data is the method of geostatistics (Clark, 1979), which can be defined as a set of statistical procedures which help describe the correlation of spatially distributed random variables, and perform interpolation (kriging) and areal estimations for these variables (Cooper and Istok, 1988). Geostatistics are based on the application of regionalized variables (i.e., variables that spread out in space and time and demonstrate a certain structure) which can provide the best (i.e., minimal error), unbiased estimates of contaminant concentrations at arbitrary, unmeasured sites by using a limited number of sediment samples. These estimates can then be used to map sediment contamination throughout the polluted area. The information obtained from geostatistic analysis can be used to develop sampling and monitoring programs and to design efficient remediation techniques. Although geostatistics have been applied mainly to groundwater contamination, their suitability to studies of aquatic sediment contamination was successfully tested by Zirschky et al. (1985).

4.1.3.2 Time Series Analysis. Environmental data are usually collected in the form of time series. It is not uncommon to observe patterns within these series associated with hydrological phenomena or human-induced activities. Such associations are major factors for high serial correlations (autocorrelations) in the observed time series. One traditional approach is to apply conventional regression analysis to assess the effects of some relevant variables (e.g., dredging, runoff, etc.) and to estimate the potential impact of a policy or program under study. The major limitation of such an approach is that is does not model the dynamic relationship between the various variables involved.

Most time series models help to characterize the homogeneous memory pattern (i.e., relationship between present and past observations). However, in addition to the homogeneous memory time series data include the effects of exogenous factors and nonrepetitive events. In order to separate the homogeneous memory from the other effects a combination of a transfer function model (Box and Jenkins, 1970) and an intervention model (t-test with serial correlations) (Hipel and McLeod, 1989) is used. The former model is employed to estimate the effects of important parameters on the variable of primary interest. The latter is designed to estimate the effects of some intervention (e.g., change of an environmental regulation, implementation of new management policy, etc). The intervention model requires information on the occurrence time and the impact form of the unusual event. If this information is not available then the outlier detection procedure is applied (Tsay, 1988).

A combination of an autoregressive-integrated moving averaging (ARIMA), intervention, and transfer function models for analysis of environmental time series data was successfully applied by Liu and Chen (1991). Application of this methodology for the LSJRB environmental quality studies would require continuous monitoring of one or more sampling stations.

4.1.3.3 __Empirical Models.__ Another category of models applied to sediment quality studies are empirical models. These models are derived from a large amount of data and define "normal" pollutant concentrations based on a limited number of "control" parameters. These "control" parameters are usually chosen according to their relative importance regarding sediment-pollutant trace chemistry. Horowitz and Elrick (1987) identified a number of physicochemical "control" parameters and defined their relative significance. In order of their importance they are:

o concentration of amorphous iron oxides in the sediment,

o particle surface area,

o loss on ignition,

o total extractable iron,

o other organic compounds (e.g., PAHs, pesticides, etc.),

o percent of sediment particles less than 63μm in diameter,

o concentration of reactive iron in the sediment,

o total organic carbon,

o percent of sediment particles less than 125μm in diameter,

o mean grain size,

o percent clay minerals,

o percent of sediment particles less than 16 μm in diameter,

o percent of sediment particles less than 2 μm in diameter,

o concentration of total extractable manganese in the sediments,

o concentration of manganese oxides in the sediments.

These parameters can be used to identify which sediment/water pollutant exchange processes (such as adsorption/desorption, precipitation, complexation, etc.) are controlling the release of pollutants from the sediment.

By comparing the existing environmental conditions to those described by the empirical models, anthropogenic impact can be easily assessed. A set of empirical models for heavy metals developed from nationwide data is presented in Table 4.2. The models were derived for sediments with grain sizes either less or greater than 125μm. The models were tested for a small coastal basin in Louisiana and were able to identify

Table 4.2 Empirical models for heavy metals (Horowitz, 1991).

Element	Particle Size	Model
As	$<125\mu m$	$As = 1.282Fe + \% < 125\mu + 0.690$
	$>125\mu m$	$As = 0.498SA + 0.563Fe + 0.157$
Co	$<125\mu m$	$Co = 1.661Ti + 0.89\% < 2\mu m + 1.76$
	$>125\mu m$	$Co = 0.658Fe + 0.132\Sigma Ex.Mn + 0.290$
Cr	$<125\mu m$	$Cr = 0.580OOM + 0.442Fe - 0.151SA + 1.347$
	$>125\mu m$	$Cr = 0.422Fe.Fe_2O_3 + 0.419Ti + 0.166\,Al + 0.259$
Cu	$<125\mu m$	$Cu = 0.674Al + 0.367LOI + 0.205Fe.Fe_2O_3 - 1.87$
	$>125\mu m$	$Cu = 0.520LOI + 0.351Al + 0.395Ti + 0.875$
Hg	$<125\mu m$	$Hg = 0.605Fe.Fe_2O_3 + 0.091Mn.MnO_2 - 3.613$
	$>125\mu m$	$Hg = 0.493Fe.Fe_2O_3 + 0.224Al - 3.145$
Ni	$<125\mu m$	$Ni = 0.962Al + 0.248TOC + 0.648$
	$>125\mu m$	$Ni = 0.387Fe.Fe_2O_3 + 0.327Al - 0.266$
Pb	$<125\mu m$	$Pb = 0.351SA + 0.383OOM + 0.603Ti + 0.329$
	$>125\mu m$	$Pb = 0.512Al - 0.316Ti + 0.156Mn + 0.039\% < 125\mu m + 0.756$
Sb	$<125\mu m$	$Sb = 1.013Al + 0.465\% < 63\mu m + 0.145Mn.MnO_2 - 2.062$
	$>125\mu m$	$Sb = 0.362SA = 0.346Fe + 0.518D_m - 1.932$
Se	$<125\mu m$	$Se = 0.920OOM + 0.485SA + 0.616Ti - 1.255$
	$>125\mu m$	$Se = 0.918LOI - 0.352Al - 0.962$
Zn	$<125\mu m$	$Zn = 0.983Al + 0.263Fe.MnO_2 + 0.255LOI + 0.329$
	$>125\mu m$	$Zn = 0.548Fe.Fe_2O_3 + 0.396Al - 0.300$

Note: Al = total Al; Fe = total Fe; Mn = total Mn; Ti = total Mi; $Fe.MnO_2$ = reactive Fe; $Fe.fe_2O_3$ = Fe oxide; $Mn.MnO_2$ = Mn oxide; LOI = loss on ignition; OOM = other organic matter; TOC = total organic carbon; D_m = mean grain size; SA = surface area; $\Sigma Ex.Mn$ = total extract. Mn.

locations with anthropogenically elevated As and Hg concentrations (Horowitz, 1991). These models can be applied to any ecological system as long as there is enough data for comparison and validation.

4.2 Sediment Sampling and Analysis.

4.2.1 Sampling Designs. Data collection procedures are very important for the ultimate success of an environmental project. It is believed that more errors are introduced by improper sampling techniques and handling of the samples than by the actual analysis of the samples in the laboratory (Gy, 1979). Among the typical data collection designs are random sampling, weighted random sampling, systematic sampling, and fixed location sampling.

4.2.1.1 **Random Sampling.** In a random sampling design, sites are selected on a random basis and equal importance is assigned to each site. This design is very effective whenever the physicochemical or biological parameters under study are homogeneously distributed throughout the contaminated area. The method is commonly used for reconnaissance surveys in areas where little or no data are available so that random sampling is the only available sampling choice.

4.2.1.2 **Weighted Random Sampling.** This design of data sampling is an improvement on the effectiveness of the previous method but requires some preliminary data pertaining to the physicochemical and biological conditions of the area under study. Based on this preliminary data, the area is divided into homogeneous sub-areas, and sampling sites are distributed according to the significance of each sub-area. In this way, only sites within the sub-areas are selected on a random basis.

4.2.1.3 **Systematic Sampling.** In the systematic sampling design, sites are evenly distributed through the project area. This method is simple to implement but may produce biased information if the sediment distribution is non-homogeneous.

4.2.1.4 **Fixed Location Sampling.** Sampling according to this design is restricted to fixed, preselected sites. Selection criteria for these sites include: accessibility, point-sources, etc. Thus, information is limited to only the sampled sites, and no statistical inference can be made for the areal distribution of the data.

4.2.2 Preliminary Analysis of Sediment Data. After the sampling sites are selected, and data collection is initiated, the following steps of action are recommended (Green, 1979):

- examination of initial data to identify any spatial distribution of contaminants,
- examination of the effectiveness of sampling devices over the range of expected natural conditions,
- collection of replicate samples,
- quality control of collected data, and
- statistical analysis of the data for assessing the effectiveness of the selected sampling locations, number of sites, frequency of sampling, etc.

4.2.3 Sediment Sampling Devices. Sediment sampling is commonly divided into suspended sediment sampling, surficial sampling, and deep-bed sampling. Suspended sediment samples are indicative of the present conditions of the estuarine system. Surficial sampling is limited to the upper 1 to 3 cm of the sediment bed and is indicative of recent environmental conditions. Deep-bed sampling can be on the order of a few meters and is useful for the establishment of historical changes in pollutant concentrations.

4.2.4 Sampling of Suspended Sediments. Sampling of suspended matter is a more difficult task than sampling of bottom sediments. A typical suspended sediment trap consists of an anchor, a buoy, a support line, and cylinder acting as traps (Figure 4.2). A critical review of suspended sediment traps was given by Bloesch and Burns (1980) and Blomqvist and Hakanson (1981). From their studies it was concluded that:

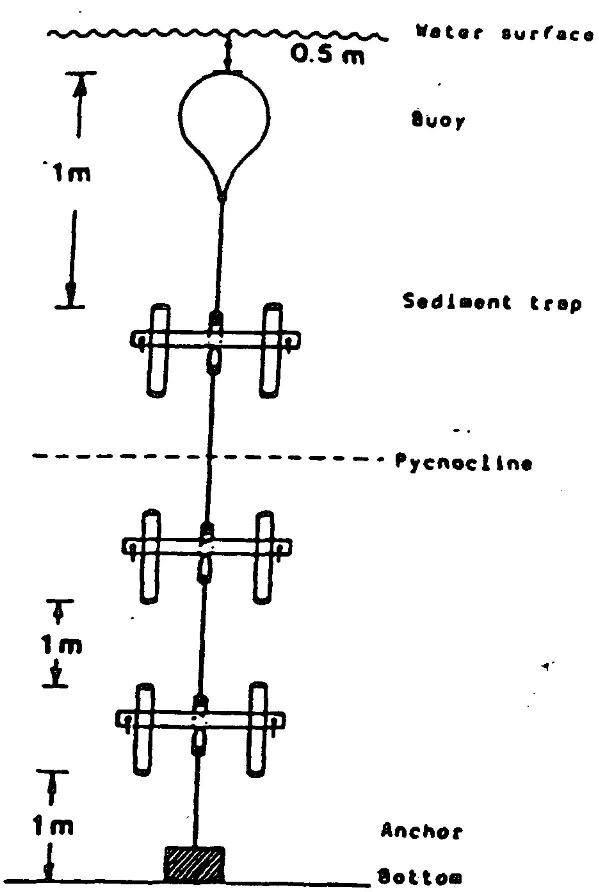


Figure 4.2 Suspended sediment trap device (Hakanson et al. 1989).

o Cylindrical traps provide the best shape for suspended sediment collection in all types of aquatic environments. Narrow-necked containers overtrap sediments while funnel-shape containers undertrap suspended material.

o Cylindrical traps should generally have a height:diameter ratio greater than 3. This ratio should be increased up to 6 for highly turbulent waters. The diameter of the cylinder must be larger than 4 cm.

Bloesch and Burns (1980) suggested that samples should be recovered on a weekly to bi-weekly basis. Longer periods of sample recovery may result in an approximate 10% loss of dry weight, particulate carbon, nitrogen and phosphorus due to grazing zooplankton or bacterial degradation.

Suspended particulate matter (SPM) can be identified by using different instruments that measure the attenuation of an optical, acoustical, or radioactive signal (Bukata et al. 1985). In addition, with proper calibration, turbidity measurements can be used to estimate the amount of suspended particles. However, the most commonly used suspended sediment trap devices are the same with water samplers (Table 4.3).

Table 4.3 Water samplers (Mudroch and MacKnight, 1991).

Type	Volume (liters)	Construction Material	Special Features
Bacti-Bulb	0.8-1	Glass	Depth sampling up to 366 meters
Depth Integrator	1-8	Al & PVC	Portrays WQ without the need of multiple samplers
Friedinger	1-3	Light metal or PVC	Lids open vertically
Go-Flo	1.7-100	PVC	Close-open-close design prevents surface contamination
Kemmerer	0.5-8	Cu, Ni-plated or PVC	Good for general water sampling
Knudsen	1.2	Ni-platted brass	Holds three reversing thermometers
Nansen	1.5	Tin-platted brass	
Niskin	1.2-30	Teflon-lined or PVC	
Rosette	1.2-30		Multiple Go-Flo or Niskin bottles
Ruttner	1-3	Plexiglass	Lids open horizontically
Submersible Pump		Brass or Stainless Steel	Unlimited volume of sample
Van Dorn-Horizontal	2,3,6	PVC	Good for sediment-water interface samples
Van Dorn-Vertical	2,3,6	PVC	Good for general water sampling

Hakanson et al. (1989) completed a comprehensive analysis of sediment trap records to determine the relationship between gross versus net sediment deposition. For that purpose, they defined three possible scenarios of gross sedimentation rates that may occur in estuarine waters (Figure 4.4). These sedimentation scenarios depend on the type

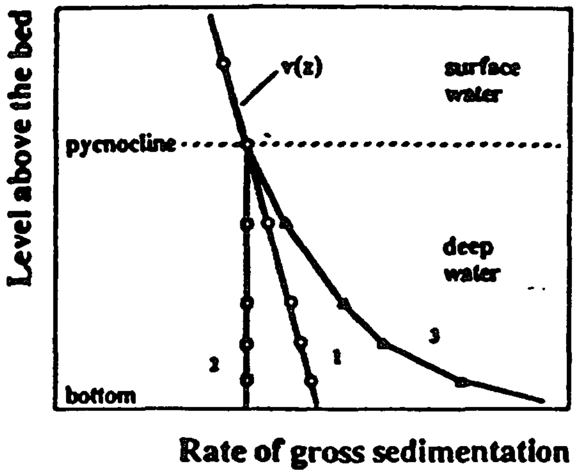


Figure 4.3 Rates of gross sedimentation (Hakanson et al. 1989).

of the suspended sediment, i.e., primary material, resuspended material, or combination of both. Accordingly the sedimentation rates are defined as:

o Constant rate: resulting from complete mixing conditions of the waters above the pycnocline (steep density gradient) only; i.e., settling of primary material (curve 2, Figure 4.3),

o Linear rate: resulting from complete mixing conditions of the entire water column (curve 1, Figure 4.3),

o Exponential rate: resulting from resuspended (secondary) matter (curve 3, Figure 4.3).

A full assessment of sedimentation rates in a coastal/estuarine environment, will require:

o data describing the allochthonous and anthropogenic external sediment loads, o estimation of the sediment trapping capacity of the adjacent drainage basins,

o estimation of the autochthonous production, or internal bioproduction,

o quantification of the sediment loss induced by metabolic processes such as biological re-working and transformations, and

o assessment of the capacity of a given area to act as a sediment trap.

4.2.5 Bottom Sediment Sampling Devices. Sediment sampling is accomplished by certain devices that can be separated generally into three categories: grab samplers, corers, and dredges. Grab samplers collect surficial sediments; corers collect both surficial and deep-sediment samples; and dredges collect large amounts of well-mixed, near-surface sediments. The advantage of corers over the other two devices is that corers obtain the least disturbed samples. A partial listing of grab samples and corers is given in Tables 4.4 and 4.5 respectively (Horowitz, 1991).

Table 4.4 Most commonly used grab sampling devices (Horowitz, 1991).

Name	Amount of Sample	Recommended Type of Sediments
Birge-Ekman	Small/Bulk	Soft clay, mud, silt, silty-sand
Dietz-Lafond	Small	Soft clay, mud, silt, sand
Franklin-Anderson	Bulk	Soft clay, mud, silt, sand, rarely gravel
Orange Peel Bucket	Bulk	Indurated or soft clay, mud sand
Peterson	Bulk	Soft clay, mud, silt, sand, gravel
Ponar	Bulk	Indurated or soft clay, mud, silt, sand, gravel
Shipek	Bulk	Indurated or soft clay, mud, silt, sand, gravel
Smith-McIntyre	Bulk	Indurated or soft clay, mud, silt, sand, gravel
US BM-54	Small	Soft clay, mud, silt, silty-sand
US BMH-60	Small	Soft clay, mud, silt, silty-sand
Van Veen	Bulk	Indurated or soft clay, mud, silt, sand, gravel

Table 4.5 Most commonly used core samplers (Horowitz, 1991).

Name	Core Depth	Recommended Type of Sediments
Box Corers	Short	Soft fine-grained clay to coarse-grained sand
Free-Fall Gravity Corers	Short	Very soft fine-grained
Kastenlots	Medium	Soft fine-grained clay to medium-grained sand
Piston Corers	Medium/long	Soft fine-grained clay to medium-grained sand
Standard Gravity Corers	Short/medium	Soft fine-grained clay to fine-grained sand
Vibrocorers	Medium/long	Soft fine-grained clay to indurated coarse-grained sand

4.2.5.1 Grab Samplers and Dredges. Grab samplers are used widely for surficial sediment data collection in shallow waters (Figure 4.4). The quality of a grab sample is evaluated according to four criteria:

o degree of physical disturbance of the sample,

o washout of fine materials through the sampler,

o ability to collect heterogeneous sediment samples (i.e., particle diameter, degree of induration, etc.), and

o potential for sample contamination.

Since few grab samplers can satisfy all four criteria, it has been suggested that coring devices are the best alternative to sediment sampling (Norris, 1988). Dredge samplers differ from grab samplers primarily on the amount of the collected material. Therefore, the main advantage of dredges over grab samplers is that they provide a larger, and thus more representative well-mixed sample of the near-the-surface sediments.

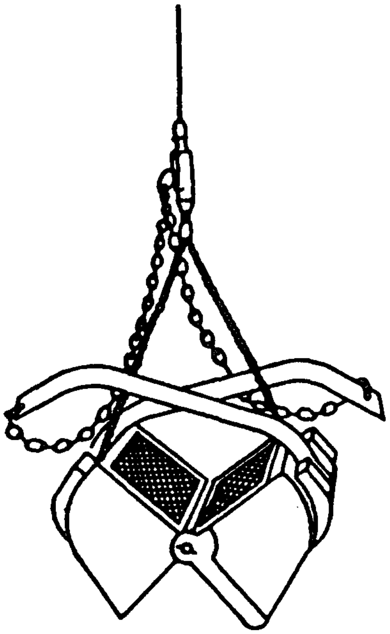


Figure 4.4 Van Veen type of sediment grab sampler (Mudroch and MacKnight, 1991).

4.2.6 Deep-Bed Coring.

4.2.6.1 Coring Samplers. Depending on their operational mode, coring devices are categorized as gravity corers, piston corers, and vibrocorers (Figures 4.5, 4.6). Gravity corers penetrate into the bottom sediments under the action of gravity. Box corers and Kastenlots are gravity type devices, and both are extremely heavy. Typical gravity corers have a maximum length of 6 feet; however, Kastenlots of about 20 feet long have been applied successfully. Piston corers are heavy devices used for collection of deep cores (up to 100 feet) of soft material. Penetration is enhanced by the creation of vacuum conditions by the piston. Vibrocores are mainly used for deep samples (up

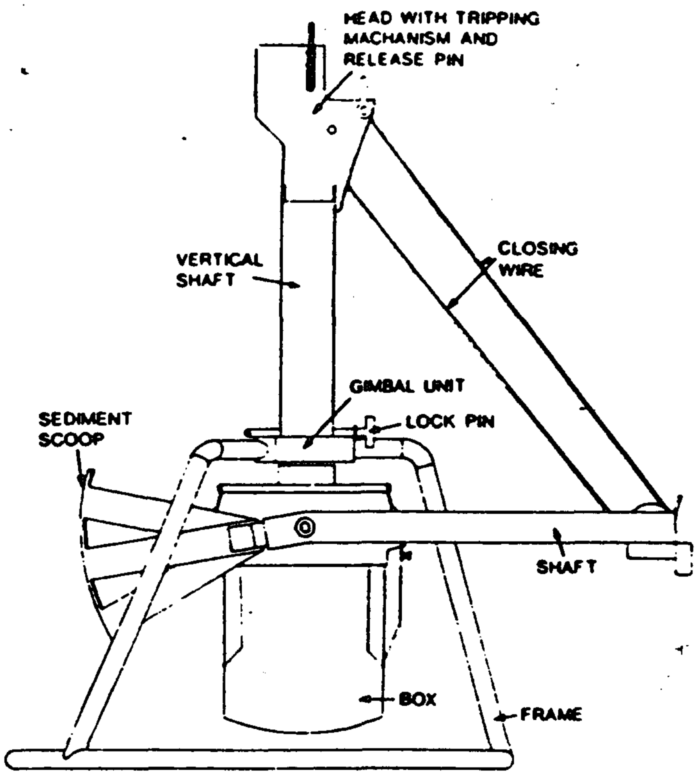


Figure 4.5 Sediment box corer (Mudroch and MacKnight, 1991).

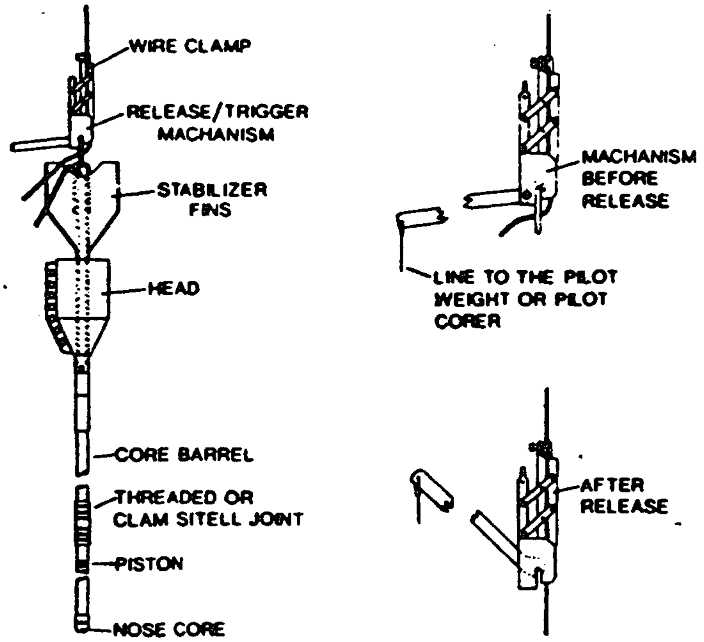


Figure 4.6 Sediment piston corer (Mudroch and MacKnight, 1991).

to 40 feet) in indurated (hardened) sediments. Vibrocorers facilitate penetration by using vibrations to fluidize the soil. Therefore, vibrocorer samples are more disturbed than samples taken by gravity or piston devices.

4.2.7 **Sampling of Interstitial Water.** Besides the collection of sediment samples, sometimes it is important to collect interstitial water from coarse-particle sediment beds (sand and gravel). Investigation of the interstitial water quality is useful for the study of the biological communities which are living in these substrates. Devices that can be used for this purpose include the Makemson sampler (1972), the Williams and Hynes sampler (1974), and the Whitman sampler (1989). A schematic representation of the Whitman sampler is given in Figure 4.7. Interstitial water samplers are used for sample

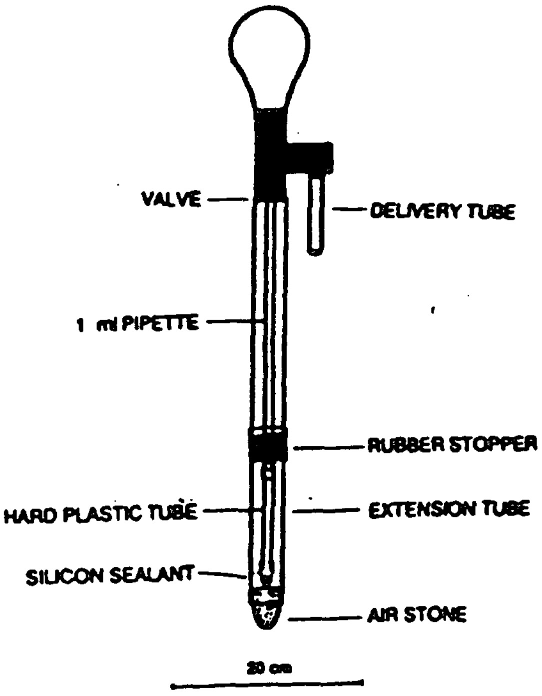


Figure 4.7 Interstitial water sampling device (Whitman, 1989).

depths ranging from 10 to 40 cm. Their performance is tested by measuring dissolved oxygen (DO) data which are sensitive to sampling errors and very important to benthic organisms.

Another device for *in-situ* sampling of pore water is the dialyzer (Hesslein, 1976). This device is based on the principal that given enough time, a contained quantity of water in the sampler will diffuse and equilibrate through a dialysis membrane (e.g., 0.2 μm pore-size polysulfone, 0.2 μm pore-size polycarbonate membrane, or 0.45 μm pore-size biologically inert PVC) with the surrounding water and its dissolved matter (Mudroch and MacKnight, 1991). The membrane can be substituted by other materials such as porous 3 μm Teflon (Figure 4.8).

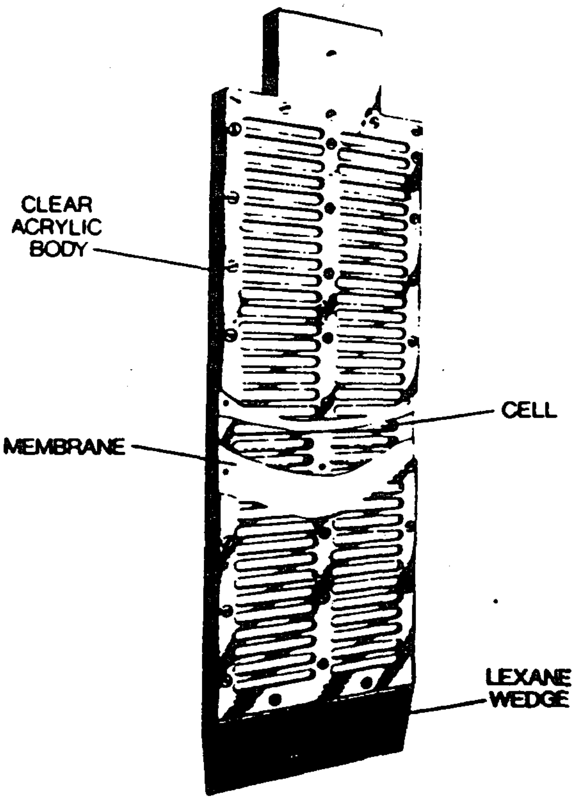


Figure 4.8 Dialyzer samplers (Mudroch and MacKnight, 1991).

4.2.8 <u>Containers for Sediment Samples.</u> Containers and implements used to store and handle sediment samples must be selected with care because they can either cause significant contamination or reduce the amount of constituents under investigation. Sediment containers are usually made of wide-mount screw-capped glass or polyethylene bottles with sizes ranging from 30 ml to 4.0 liters. The caps can be either from polyethylene or any other appropriate Teflon-lined material. Sediment samples to be analyzed for inorganic compounds can be stored also in polyethylene, polypropylene or any other plastic bag. Before use, all containers and implements should be washed with detergent, de-ionized water and dilute nitric acid.

4.2.8.1 <u>Container Cleaning Procedures for Inorganic Contaminants.</u> For analysis of inorganic pollutants the least contaminating containers, if properly cleaned, are Teflon and polyethylene. Moody and Lindstrom (1977) recommended the following procedure for container cleaning:

o fill container with reagent grade HCl (1:1),

o allow to remain for a week at room temperature or at 80°C for Teflon containers,

o empty and rinse with distilled water,

o fill container with reagent grade HNO_3,

o allow to remain for a week at room temperature or at 80°C for Teflon containers,

o empty and rinse with distilled water,

o fill with very pure water and allow to stay for weeks or until needed,

o change the water periodically,

o rinse with very pure water and allow to dry in a fume-free particle-free environment.

For better results some times in addition to HCl and HNO_3, sulfuric and nitric acids are used (Water Quality National Laboratory, Environment Canada, 1985).

4.2.8.2 Container Cleaning Procedures for Organic Contaminants.

Whenever sediments are to be analyzed for organic pollutants it is recommended to avoid storage in plastic containers. Instead, containers made of glass, porcelain, stainless steel, Teflon or Teflon-coated are recommended. However, certain care should always be used since certain compounds (e.g., phenols) can be adsorbed to Teflon or aluminum surfaces. The Water Quality National Laboratory, Environment Canada (1985) recommends the following steps for cleaning containers to be used for analysis of organic compounds:

- wash with high pressure tap water,
- wash with chromic acid and water,
- wash with soap water,
- rinse well with organic-free water,
- rinse twice with washing acetone and once with special-grade acetone,
- rinse twice with pesticide-grade hexane,
- dry containers in a hot air oven at 360°C for at least 6 hours.

4.2.9 Analysis of Sediment Data. Once the sediments samples have been collected, a complete analysis should follow, including: physical analysis (i.e., manual and instrumental selection of water and sediment phases), chemical extractions and analyses of pertinent parameters, statistical manipulation of data, and mathematical modeling of the water/sediment dynamics and pollutant exchange.

4.2.9.1 Field Handling of Sediment Samples. Once collected the wet sediment samples are subject to some preliminary analysis and preparation including:

- sample separation for biological testing,

- o measuring of pH and Eh,
- o subsampling into prepared containers (for later analysis of wet and dry sediments),
- o subsampling for determination of sediment cation exchange capacity (CEC),
- o subsampling for oxygen-free tests (for later analysis of pore water and leaching tests).

The pH, Eh and CEC measurements should be conducted on wet, untreated bulk sediments otherwise the estimated values may differ drastically from the "natural" values. Laboratory analysis of the samples employs several procedures to determine the physicochemical characteristics of the sediment including sample preparation, measurement of grain size distribution, estimation of surface area, determination of substrate geochemistry, and pollutant extraction. Analysis under oxygen-free conditions should be applied to sediment samples taken from bottom depths greater than 3 cm. Surficial sediments (top 1-3 cm) and suspended sediments are primarily oxidized. Preservation of anoxic conditions for bottom sediment samples is very important especially for analysis of certain trace metals such as copper, iron and zinc (Mudroch and MacKnight, 1991).

4.2.9.2 Sample Preparation. Depending on the type of the studies to be performed sediment samples can be prepared for wet sediment or dry sediment analysis.

4.2.9.2.1 Wet Sediment Analysis. The preparation of samples for analysis and tests on wet sediments involves the following steps (Mudroch and MacKnight, 1991):

- o analysis of particle size distribution,
- o geotechnical tests,
- o sediment stratigraphy,
- o freezing (-20°C) for wet sediment bank,

- o separation into different particle fractions,
  - wet sieving, elutriation,
  - freeze-drying of different particle fractions,
  - grinding of dried fractions,
  - determination of inorganic constituents,
  - determination of organic constituents,
- o biological analysis and bioassays,
- o determination of volatile organic contaminants (VOC).

4.2.9.2.2 Dry Sediment Analysis. Dry sediment samples can be prepared by air, oven or freeze drying. The various steps taken for each of the drying techniques are as follows (Mudroch and MacKnight, 1991):

Air Drying:

- o breaking dry sediment aggregates,
- o plant bioassays and other tests.

Oven Drying:

- o breaking dry sediment aggregates,
- o sieving through 0.841 mm (20 mesh) sieve,
- o further sieving to separate small particles,
- o grinding and homogenizing,
- o sediment analysis for inorganic constituents.

Freeze Drying:

- o breaking dry sediment aggregates,
- o sieving through 0.841 mm (20 mesh) sieve,

- wet sieving,
    - separation into different granular fractions,
    - freeze-drying of separated particle fractions,
    - grinding and homogenizing,
    - determination of volatile organic constituents,
    - determination of inorganic constituents,
    - determination of non-volatile organic constituents,
- dry sieving,
    - separation into different granular fractions,
    - grinding and homogenizing,
    - determination of inorganic constituents,
    - determination of organic constituents,

o further sieving to separate small particles,

o grinding and homogenizing,

o determination of inorganic constituents,

o determination of organic constituents.

4.2.9.3 Grain Size Distribution. Physicochemical analyses of sediments require either separation of certain particle fractions or measurement of their grain size distribution. Estimation of the grain size distribution is accomplished by using a number of laboratory techniques (Table 4.6). For particle size greater than $64\mu m$ direct methods such as sieving or visual accumulation can be employed. For particle fractions with $D_s < 64\ \mu m$, measurement of physical properties requires specialized devices and indirect methods (Table 4.6).

Table 4.6 Analysis of sediment grain size (Horowitz, 1991).

Particle Size	Physical Separation	Property Measurements
> 64μm	Sieving Visual Accumulation	Direct measurements
< 64μm	Air Elutriation Bottom Withdrawal Centrifugation Decantation Pipet Analysis Filtration/Hyperfiltration	Electrozone Counters Hydrometers Image Analyzers Microscopy Sedigraph^R Different size filters

Droppo et al. (1992) conducted a study on the accuracy of the estimation of primary grain-size distribution of cohesive sediments using the procedure of filtering and resuspending sediment off two types of filters (i.e., Nuclepore polycarbonate plastic membrane filters, 0.4 μm, and Millipose cellulose membrane filters, 0.45 μm). This procedure is necessary whenever suspended sediment concentrations are low so that sediments must be concentrated for laboratory analysis. Their conclusions indicated that although some particle disintegration and pore clogging occurred, the primary grain-size distribution did not significantly alter with this procedure.

4.2.9.4 Particle Specific Surface Area. Estimation of the particle specific surface area is important to assess the electrochemical behavior of adsorption-desorption processes. The specific surface area (i.e., surface area per unit mass) is directly related to the free energies and geometry of the particle which control its electrochemical properties (Adamson, 1990). The grain surface area is estimated implicitly by using gas or liquid adsorption techniques and the Brunauer-Emmett-Teller (BET) equations. Gas adsorption techniques may vary from single-point to multiple-point adsorption of static or flowing gas. The most commonly used gases involve mixtures of helium with nitrogen, argon, krypton or n-butane. Liquid adsorption techniques employ media such as water, glycerol, ethylene glycol, and aqueous ammonium chloride (Horowitz, 1991). Liquid adsorption techniques provide a better measure of the grain surface area, since they include the surface of the particle interstices while gas adsorption provides only the external surface of the particle.

4.2.9.5 Substrate Geochemistry and Sediment Quality. Sediment samples are comprised of a mixture of inorganic and organic materials. Inorganic materials are detected through extraction procedures that involve a number of steps. For example, estimation of iron and manganese involves analysis for: adsorbate/carbonate Fe and Mn; Mn oxides and reactive Fe; amorphous Fe oxides and Mn; organically-bound Fe and Mn; total extractable Fe; total extractable Mn; and, percent of clay minerals (Horowitz and Elrick, 1987). Whenever metal concentrations are too low to be detected directly by inductively coupled plasma atomic emission spectrometry (ICP-AES), some metal concentration enhancement methods have been developed, such as liquid-liquid extraction, ion exchange osmosis and coprecipitation. These methods are however suitable for brackish or seawater and do not apply to field conditions. Coprecipitation of metals in brackish water were developed by using different carriers, such as indium hydroxide (Hiraide et al. 1980), borhydride reduction (Skogerboe et al. 1985) or $Mg(OH)_2$ (Bostrom and Bostrom, 1991). A simple and inexpensive metal coprecipitation field method for fresh waters is $Mg(OH)_2$ and $MgSO_4$ and adjusting the pH to about 10-11 by adding NaOH (Anderson and Ingri, 1991).

Since trace metals in natural aquatic systems are present in various physicochemical forms (i.e., ions, ion pairs, organic or inorganic complexes, colloids and suspended particles), knowledge of their particular form is important for assessment of their bioavailability and the selection of remedial techniques. In recent years various methods have been developed for physicochemical speciation of trace metals such as, centrifugation, gel filtration, dialysis, filtration and ultrafiltration. Tanizaki et al. (1992) successfully applied a metal speciation technique using size fractionation (filtration, 0.45 μm, and ultrafiltration, 1 nm) and a neutron activation analysis (NAA). Their fractionation results included major components (e.g., nutrients, TOC, Cl^-), alkali and alkaline earth elements (Ba, Ca, Cs, K, Mg, Na, Rb and Sr), transition elements (Ag, Co, Cr, Fe, Mn, Ni, Sc, V and Zn), rare earth elements (Ce, Eu, Ho, La, Lu, Sm, Tm and Yb) and other elements (Al, As, Au, Br, Hf, Ir, Mo, Sb, Se, Ta, Th, U and W).

Organic content is generally reported as: total organic content (TOC), total organic matter, or other organic matter. Total organic carbon is estimated by infrared detection of the CO_2 evolving from combustion of a sample pretreated with 10% hydrochloric acid. Total organic matter is defined as the loss on ignition and can be estimated by combustion of a dried sample in a muffle furnace at 500°C for 30 minutes. Finally, the other organic matter is defined as the difference between total organic matter and total organic carbon (Horowitz, 1991).

Waters acidified by the presence of humid substances may affect the ionic charge-balance and control the pH of the system (Kramer and Davies, 1988; Tipping et al. 1988). These weak acids are very important in binding metal cations, especially Al^{3+} and $AlOH^{2+}$. A comprehensive model for estimation of aluminum binding by organic substances was developed recently by Tipping et al. (1991). The model utilizes different inputs such as temperature, pH, ionic strength, total concentration of fluoride, sulphate, humid substances, calcium, magnesium, sodium, potassium and ammonium. Changes in the pH are important factors for the mobility of nutrients and metals (Boers, 1991).

Besides inorganic and organic constituents the water/sediment samples should be analyzed for bacteria resulting from leakage of septic tanks. In marine waters F-specific and *B. fragalis* bacteriphages are usually found in smaller concentrations than *E. coli*. Thus, it is suggested that fecal spectrococci and *E. coli* are better indicators of sewage pollution in estuarine systems than F-specific or *B. fragalis* bacteria (Cornax et al. 1991). The bacteria counts and the phage counts can be conducted according to the methods suggested by the American Public Health Association (1985).

4.2.9.6 Protection of Organic Matter in Sediment Samples. Entrapped sediment samples usually suffer from two major problems. The first is related to zooplankton activities such as grazing and extraction, and the second is related to the bacterial degradation of organic matter. In order to overcome these problems, it is suggested to add certain preservatives and poisons into the sediment samples (Table 4.7). However, since the actual effects of the preservatives/poisons are not well established it is recommended to have both treated and untreated sediment samples for comparison.

4.2.9.7 Pollutant Extraction. One of the most widely used methods for chemical partitioning is the partial extraction method. The basic concept behind this method is that a certain reagent is usually phase-specific and/or process-specific (Horowitz, 1991). Therefore, if reagents with increasing strength are applied sequentially to a sediment sample, they would chemically remove a series of sediment-

Table 4.7 Preservatives and poisons used in sediment traps (Mudroch and MacKnight, 1991).

Preservatives and Poisons	Effects
Formaldehyde	Changes of organic bindings of organic matter
Para-formaldehyde	Chemical reaction with POM (dissolution)
Glutaraldehyde	Artificial increase of POM flux; decreasing particle size; increasing dead zooplankton
Lugol	General bias
I_2 crystals	-
Chloroform	Dissolved POM, pigments, lipids, amino acids, etc.; disintegrates zooplankton; does not kill all the bacteria
Na-N_3 (sodium azide)	Oxidizes PN to NO_3; bias of N analysis; does not kill all the bacteria
$HgCl_2$ (mercurium)	Bias of P-analysis; bias of Hg and other metals analysis
$KMnO_4$	MnO_2 is formed and precipitated
Polyacrylamide	May affect settling velocity of small particles
Phenol	Does not kill all bacteria
Pentachlorophenol (PCP)	Increases P and N flux by increased sulfur bacteria
Antibiotics (thymol)	May not kill all bacteria
Solid copper	-
Tributyl tin (TBT)	-
NaCl	Does not kill all bacteria; establishment of a density gradient

bound pollutants (Table 4.8). This procedure helps to identify not only the type and amount of a particular pollutant but also the strength of the binding forces between sediment and pollutants. However, it must kept in mind that extraction reagents may not be as selective as they are expected to be, and extraction efficiencies may vary according to the length of treatment and the sample:extractant ratio.

The success of chemical partitioning of sediment-bound chemicals in laboratory studies is greatly enhanced by a number of instruments and techniques such as: electron microprobe analysis; Scanning Electron Microscopy/Energy Dispersive X-ray Analysis (SEM/EDAX); Electron Spectroscopy for Chemical Analysis/X-ray Photoelectron Spectroscopy (ESCA/XPS); ultraviolet photoelectron spectroscopy; Auger Electron Spectroscopy (AES); Secondary Ion Mass Spectroscopy (SIMS) and Ion Scattering Spectroscopy (ISS) (Johnson and Maxwell, 1981).

Table 4.8 Reagents employed in partial extraction methods (Horowitz, 1991).

Substance Classification	Reagent
Adsorbates and Exchangeables	0.2M $BaCl_2$-triethanolamine, pH 8.1
	1M NH_4OAc, pH 7
	Distilled Deionized Water
	1M $MgCl_2$, pH 7
	1M NaOAc, pH 8.2
Carbonates	1M HOAc (25% v/v HOAc)
	1M NaOAc w/HOAc, pH 5
	CO_2 treatment
	Exchange columns
Detrital/Authigenic	EDTA treatment
Hydrogenous/Lithogenous	0.1M HCl
	0.3M HCl
Reducible	1M $NH_2OH.HCl$ w/25% v/v HOAc
Moderately Reducible	Oxalate buffer
	Dithionate/citrate buffer
Easily Reducible	0.1M $NH_2OH.HCl$ w/0.01M HNO_3
Organics	Na hypochlorite w/dithionate/citrate
	30% H_2O_2 at 95°C, pH 2
	30% H_2O_2 w/0.02N HNO_3, pH 2, extracted w/1M NH_4OAc in 6% HNO_3
HNO_3	30% H_2O_2 w/0.02N HNO_3, pH 2, extracted w/ 0.01 M 1:1 Methanol:Benzene
	0.1N NaOH
NH_4OAc	0.02M HNO_3 w/H_2O_2, pH 2, w/HNO_3 at 85°C, w/3.2M in 20% HNO_3
	30% H_2O_2 in 0.5N HCl, heat
Sulfides	30% H_2O_2 at 95°C, extracted w/1N NH_4OAc
	0.1N HCl w/air
Detrital Silicates	$HF/HClO_4/HNO_3$
	Borate fusion, extracted w/ HNO_3

Note: M = molarity; N = normality

4.2.9.10 Sediment Age Determination. Among the most important methods for estimation of the age (history) of sediments are the radiometric and chemical dating methods (Geyh and Schleicher, 1990). These methods are based on measuring some time-dependent properties of the sediments. There is a wide variety of techniques for sediment dating which use different geochronometers such as the cosmogenic radionuclides, uranium series, radiation-damage, chemical degradation, etc. (Table 4.9). The most commonly used methods for measuring the age of new sediments (10^0-10^2 years) are the carbon-14 (^{14}C), lead-210 (^{210}Pb), the stable oxygen isotope and the artificial radionuclides methods. All of these methods are based on measuring the decay of the isotope (radionuclide) of a particular element. The radionuclei decay is characterized by the emitted radiation, the particle energies and the half-life. The laboratory analysis of the isotopes is accomplished by means of different instruments such as the Geiger-Muller counter, Liquid Scintillation Counter (LSC), semiconductor detectors, Mass Spectrometry (MS), Accelerator Mass Spectrometry (AMS), Resonance-Ionization Spectrometry (RIS), Isotope Dilution Analysis (IDA), Neutron Activation Analysis (NAA), Atomic Absorption Spectroscopy (AA), Inductive Couple Plasma Analysis (ICP), Ion-Microprobe (IPM), Laser Microprobe Mass Analysis (LAMMA) and the X-Ray Fluorescence Analysis (XRF) (Geyh and Schleicher, 1990).

Pollutant (e.g., PCBs, PAHs, etc.) concentration profiles of a sediment core may differ from the core dating profile measured by some radionuclide method (e.g., ^{14}C, ^{210}Pb). The reason for such a discrepancy can be attributed to the differences in the physicochemical behavior of the pollutants and radionuclides within the bottom sediments, i.e., input rates, diffusion processes, chemical remobilization, half-life, extraction by benthic organisms, etc. (Eisenreich et al. 1980).

4.2.10 Differences Between Fresh and Saltwater Sediments. Since aquatic sediments

Table 4.9 Methodologies for aquatic sediment dating (Geyh and Schleicher, 1990).

Method	Application	Suitability Unconsolidated marine sediments	 Lake and river sediments	Time Period in years (Approximately)	
Cosmogenic Radionuclides:					
^{14}C	Well-developed	Suited	Well-suited	10^{1}-$0.5x10^{5}$	(*)
^{10}Be	In development	Suited	-	10^{5}-10^{7}	
^{26}Al	In development	Used	-	10^{5}-$0.5x10^{7}$	
^{32}Si	In development	Suited	-	10^{2}-10^{3}	
^{53}Mn	Well-developed	Suited	-	$0.5x10^{6}$-10^{7}	
^{129}I	Limited	Questionable	-	$0.5x10^{7}$-10^{8}	
$^{26}Al/^{10}Be$	Limited	Used	Used	10^{5}-10^{7}	
Uranium Series:					
$^{230}Th/^{234}U$	Well-developed	Well-suited	Suited	10^{3}-$0.5x10^{6}$	
$^{231}Pa/^{235}U$	Limited	Suited	-	$0.5x10^{4}$-10^{5}	
$^{234}U/^{238}U$	Limited	Suited	Used	$0.5x10^{5}$-10^{6}	
^{230}Th excess	Well-developed	Well-suited	-	10^{3}-$0.5x10^{6}$	
^{231}Pa excess	Limited	Suited	-	$0.5x10^{4}$-10^{5}	
^{234}Th excess	Limited	Used	-	$0.5x10^{0}$-10^{0}	(*)
^{228}Th excess/^{232}Th	Limited	Suited	Suited	10^{0}-10^{2}	(*)
^{226}Ra	Limited	Used	-	10^{3}-$0.5x10^{5}$	
^{224}Ra & ^{228}Ra	Limited	Suited	Suited	10^{0}-10^{2}	(*)
^{210}Pb	Well-developed	Well-suited	Well-suited	10^{0}-10^{2}	(*)
Radiation-Damage:					
Thermoluminescence	Well-developed	Questionable	Questionable	$0.5x10^{3}$-10^{5}	
Electron Spin Resonance	In development	Suited	Used	10^{3}-10^{6}	
Global Time Markers:					
Paleomagnetism	Well-developed	Suited	Suited	10^{2}-$0.5x10^{9}$	
Stable oxygen isotopes	Well-developed	Well-suited	Used	10^{0}-10^{6}	(*)
$\delta^{34}S$, $\delta^{13}C$, $^{87}Sr/^{86}Sr$	Limited	Well-suited	-	10^{6}-$0.5x10^{9}$	
Artificial radionuclides	Well-developed	Well-suited	Well-suited	10^{0}-10^{2}	(*)
Chemical pollution markers	Limited	Well-suited	Well-suited	10^{0}-10^{2}	(*)
Chemical Dating:					
Amino-acid racemization	Limited	Well-suited	Used	10^{2}-10^{6}	
Amino-acid degradation	Limited	Suited	-	10^{5}-$0.5x10^{7}$	
Obsidian hydration	Well-established	Suited	-	10^{2}-10^{6}	

NOTE: (*) denotes methods applicable to sediment quality studies.

are constantly in contact with the ambient water, the physicochemical properties of the water may drastically affect the behavior of particle-bound contaminants. The main properties of the water along with their environmental significance are listed in Table 4.10. Besides the general properties of the water, its physicochemical features are

Table 4.10 Properties of water (Horne, 1978).

Property	Environmental significance
State	Provides a media for life
Heat capacity	Good heat sink
Latent heat of fusion	Moderates temperature effects
Latent heat of vaporization	Moderates temperature effects
Density	Top water freezes, so that aquatic life can survive winters
Surface tension	Important for surface effects
Dielectric constant	Good solvent for ionic and polar compounds
Hydration	Good solvent; mobilization of ionic pollutants
Dissociation	Provides a near neutral medium; good medium for acid/base properties
Heat conduction	Important heat transfer material

strongly affected by the various dissolved and suspended constituents which may differ drastically from one aquatic ecosystem to another. For practical reasons, surface water/sediments are separated into two main categories: fresh water/sediments and saline water/sediments.

4.2.10.1 Characteristics of Marine Water/Sediments. Oceanic waters are relatively constant in their composition. Coastal and estuarine waters may show localized differences due to freshwater discharges of rivers and overland runoff. The salinity of sea water ranges between 32-37 ppt with a mean of 35 ppt. The major elements found in seawater are listed in Table 4.11. The pH of seawater is relatively constant between 7.5-8.3. The pH is controlled primarily by the bicarbonate ion which is the major inorganic compound present in water.

The contaminant level in estuarine waters may vary from site to site depending on the inputs and the nature of the estuary. There has been controversy about the fate of contaminants entering an estuary. For example, it appears that for many estuaries the level of heavy metals entering the estuary decreases when mixed with saline water,

Table 4.11 Composition of seawater (Fergusson, 1990)

Cations	Concentration g/kg	Anions	Concentration g/kg
Na^+	10.77	Cl^-	19.354
Mg^{2+}	1.29	SO_4^-	2.712
Ca^{2+}	0.412	HCO_3^-	0.142
K^+	0.399	Br^-	0.067
H_3BO_3	0.026	F^-	0.0013
Sr^{2+}	0.008		
SiO_2	0.003		

Major Cations: $Na^+ >> Mg^{2+} > Ca^{2+} > K^+$
Major Anions: $Cl^- >> SO^{2-} > HCO_3^- > Br^-$

while on other estuaries metals are mobilized from the sediments into solution. There is a number of explanations given for these different results (Fergusson, 1990):

- decrease in pollutant concentration as fluvial sediments mix with less contaminated marine sediments,
- metal mobilization due to increasing concentration of cations in saline water (i.e.
  $M^{n+}\text{-Clay} + nNa^+(aq) \leftrightarrows nNa^+\text{-Clay} + M^{n+}(aq)$),
- metal mobilization from ligands produced by the decay of organic compounds,
- metal mobilization caused by changes in solid particle sizes during mixing, and
- tailing off of concentrations with distance for the pollutant source.

4.2.10.2 Characteristics of Fresh Water/Sediments. The composition of fresh water varies according to the geochemical composition of the surrounding soil. For example, riverine water passing over limestone terrain will be high in dissolved calcium and carbonate. The median values of fresh water constituents are given in Table 4.12.

Table 4.12 Composition of freshwater - Median values (Fergusson, 1990).

Cations	Concentration g/kg	Anions	Concentration g/kg
Ca^{2+}	15.0	HCO_3^-	58.5
SiO_2	13.1	SO_4^{2-}	11.2
Na^+	6.3	Cl^-	7.8
Mg^{2+}	4.4	F^-	0.10
K^+	2.3	Br^-	0.014
Sr^{2+}	0.07		
H_3BO_3	0.015		

Major Cations: $Ca^{2+} >> Na^+ > Mg^{2+} > K^+$
Major Anions: $HCO_3^- >> SO_4^{2-} > Cl^- > Br^-$

It has been documented that particle-bound pollutants tend to decrease with increasing water pollution (Forstner, 1980). A summary of trace metals concentrations found in different fresh water environments was compiled by Fergusson (1990) (Table 4.13).

Table 4.13 Trace metal concentration in freshwaters (Fergusson, 1990).

Trace element (μg/l)	Fresh water	Rivers unpolluted	Rivers polluted	Lakes areas	Mining waters	Geothermal waters
As	< 1-5	< 1-10	10-1000	1-70	100-5000	1000-5000
Bi	-	0.02	-	-	-	-
Cd	< 1	0.01-1	1-> 10	0.01-20	100-700	0.01-0.5
Hg	0.02-0.1	0.0001-1	> 1	0.02	-	0.05-60
Pb	< 10	< 1-10	20-100	0.1-30	100-1000	1-10
Sb	< 0.3	0.3-5	-		- -	-
Se	< 0.1	0.1-0.3	-		- -	-
Tl	0.004	-	-		- -	-

Generally, in order to assess the significance of the amount of pollutant concentrations in an aquatic system various parameters should be considered. A function that summarizes the diversity of factors affecting pollutant concentrations can be given as (Salomons and Forstner, 1984)

$$C = f(L,H,G,A,V,M,E) \quad \ldots\ldots\ldots\ldots\ldots\ldots \quad (4.24)$$

where C is the contaminant concentration, L is the lithogenous effects, H is the hydrologic effects, G is the geologic features, A is the anthropogenic influences, V is the vegetation effects, M is the mineralized zones effects, and E is the error involved.

4.2.10.3 Analysis of Marine Versus Fresh Water Sediments. Laboratory analysis of aquatic sediments for nutrients and organic constituents does not depend on the salinity content of the water. Thus, the analytical methods discussed earlier apply both to marine and fresh water sediments. However, the same is not true for analysis of trace metals particularly if the samples are analyzed with the Atomic Absorption Spectroscopy or the Inductive Coupled Plasma Emissions Spectroscopy. In these cases the dissolved salt ions may interfere with the tests and produce misleading results. Thus, in order to avoid any interference by dissolved matter it is suggested to chemically precipitate the salts before the analysis for metal detection.

For sediment toxicity tests different type of organism groups are recommended for marine and freshwaters. Burton and Scott (1992) summarized and presented selected species for toxicity tests of freshwater (Table 4.14) and marine sediments (Table 4.15). The same authors suggested that research should be focused on the effects of sample handling, relationships between acute and chronic effects, and the significance of changes in the community structure and function to ecosystem processes.

Table 4.14 Selected freshwater sediment toxicity tests (Burton and Scott, 1992).

Organism Group	Response Measures	Test Species
Amphibians	Embryo-larval survival, terata	*Xenopus laevis*
Fish	Embryo-larval survival, length, weight, terata	*Pimephales promelas* *Oncornynchus mykiss* *Oryzius latipes*
Zooplankton	Survival, reproduction	*Daphnia magna* *Ceriodaphnia dubia* *Brachionus sp.* *Colpidium campylum*
Benthic invertebrates	Survival, size, reproduction, molting, emergence, avoidance	*Panagrellus redivivus* *Caenorhabditis elegans* *Tubifex tubifex* *Stylodrilus heringianus* *Pristina leidyi* *Lumbriculus variegatus* *Hyalella azteca* *Diporeia sp.* *Gammarus pulex* *Gammarus fasciatus* *Corbicula fluminea* *Anodonta imbecillis* *Chironomus tentans* *Chironomus riparius* *Hexagenia limbata* *Hexagenia bilieata*
Microbial	Luminescence	
Phytoplankton	Cell number, ^{14}C uptake	*Selenastrum capricornutum*
Macrophytes	Frond number, chlorophyll, biomass, root and shoot length, peroxidase	*Lemna sp.* *Hydrilla verticillata*
Benthic indigenous communities	Structure indices, functional indices, chlorophyll, respiration, enzyme activities	Bacteria Protozoan Periphyton Phytoplankton Macroinvertebrates

Table 4.15 Selected estuarine and marine sediment toxicity tests (Burton and Scott, 1992).

Organism Group	Response Measures	Test Species
Fish	Survival, abnormalities	Silversides Shiner perch Top smelt Speckled sanddab Spot Arrow goby Grunion
Zooplankton	Survival	*Acartia sp.* *Nannopus palustris*
Mysids	Survival, growth, fecundity	*Mysidopsis bahia* *Holmesimysis costata* *Neomysis sp.*
Shrimp	Survival	Ridgeback prawn Grass shrimp Commercial shrimp
Bivalves	Survival, abnornal growth	Littleneck calm Bentnose clam Blue mussel Eastern oyster Pacific oyster *Ostrea sp.*
Echinoderms	Survival, abnormal growth	Purple sea urchin Green sea urchin *Strongylocentrotus fransciscanus* *Arbacia punctulata* *Lytechinus pictus*
Amphipods	Survival, reburial, growth	*Rhepoxynius abronius* *Eohaustorius estuarius* *Ampelisca abdita* *Leptocheirus plumulosus* *Grandidierella japonica* *Corophium insidiosum*
Polychaetes	Survival, reburial, growth	*Dinophilus gyrociliatus* *Neanthes arenaceodentata* *Hediste sp.* *Capitella capitata* *Streblospio benedicti*
Macroalgae	Survival, reproduction	*Champia parvula*
Macrophyte	Survival, germination, growth	*Echinochloa crusgalli*

J. SEDIMENT QUALITY CRITERIA

4.3 Assessment of Sediment Quality.

Florida's coastal and estuarine waters are very important to the natural plant and animal communities of the State. This brackish water ecosystem sustains a large number of marine fishes and invertebrates, numerous species of wading birds, terrestrial animals, and a myriad of mangrove forests and salt marshes. Protection, maintenance, and enhancement of such an invaluable environment require an integrated approach to ecological management (MacDonald, 1992).

Increased anthropogenic activities such as urbanization, agriculture, and industry (pulp and paper, organic chemicals, pesticides, plastics, etc.) have created concerns for potential ecological hazards resulting from point and non-point pollution sources. Since many pollutants have an affinity for sediments, effective management of coastal ecosystem dynamics requires understanding not only of the ambient water but also of the sediments. It is well known that contaminated sediments could be a serious cause of adverse effects on humans and aquatic biota.

Information on sediment quality may be used as an indicator for short-term and long-term pollution histories of estuarine aquatic ecosystems. Since the importance of sediment quality to ecological studies has been fully recognized, a number of sediment quality criteria (SQC) have been developed recently. These sediment quality criteria are based on different approaches such as:

o Apparent Effects Threshhold Approach (AETA),

o Equilibrium Partitioning Approach (EqPA),

o National Status and Trends Program Approach (NSTPA), o Sediment Background Approach (SBA),

o Screening Level Concentration Approach (SLCA),

o Sediment Quality Triad Approach (SQTA),

o Spiked Sediment Bioassay Approach (SSBA),

o Tissue Residue Approach (TRA), and

o Weight-of-Evidence Approach (WEA).

A summary and evaluation of the various SQC approaches can be found in Chapman (1981); EPA (1989a,b); Sediment Criteria Subcommittee - EPA (1989, 1990); MacDonald (1992), Burton (1992), Adams et al. (1992), etc.

4.3.1 **Apparent Effects Threshold Approach (AETA).** Tetra Tech Inc. (1986) was the first to develop sediment quality criteria (SQC) based on the AETA. This approach defines sediment pollutant concentrations which always produce significant effects on benthic biota. These effects are documented through changes observed in the benthic and water column species; i.e., abundance, benthic community structure, toxicity, etc. For SQC values, AETA uses dry-weight normalized concentrations for metals and either dry-weight or TOC normalized concentrations for organic matter.

The advantage of the AETA is that it can match sediment chemistry data and biological effects for every contaminant and biological species for which laboratory analysis is available. The main disadvantage of the approach is that the SQC cause-response relationships are usually site specific.

The AETA has been applied successfully in the Puget Sound Dredged Disposal Analysis Program. Recently the State of Washington adapted the AETA for developing marine sediment quality management guidelines.

4.3.2 Equilibrium Partitioning Approach (EqPA). The EqPA is probably one of the most well known SQC approaches, especially for non-polar hydrophobic organic pollutants (DiToro et al. 1991). The basic concept for this approach is that there is a continuous equilibrium exchange of chemicals between sediment particles and interstitial water. Numerical values of SQC (μm/kg) for non-ionic organic compounds (e.g., PAHs, biphenyls, dioxins, furans, polychlorinated benzenes, Cl-pesticides, etc.) using the EqPA can be estimated as:

$$SQC = K_p \, FCV \quad (4.25)$$

where K_p (L/kg) is the sediment/water partitioning coefficient, and FCV (μm/L) is the final chronic value (i.e., effects that persist over a long period) as defined by water quality criteria (DiToro et al. 1991). For organic compounds TOC-normalized values provide a good estimation of toxicity effects. Bioavailability of metals may be established by determining the acid volatile sulfide (AVS) (DiToro et al. 1992).

The advantage of the EqPA is that it utilizes existing water quality criteria and water/sediment partitioning coefficients (see section 4.1.2.1), which are applicable to a wide variety of aquatic ecosystems. The disadvantages of the method are: a) SQCs have been established only for non-ionic chemicals with significant organic carbon contents, b) the existing water quality criteria are not very comprehensive, and c) possible additive, antagonistic, or synergistic contaminant effects are not addressed.

The EqPA has been used widely in the US and other countries (i.e., Canada, the Netherlands). Currently, the EqPA is under review for possible adaptation by the U.S. Environmental Protection Agency.

4.3.3 National Status and Trends Program Approach (NSTPA). The NSTPA was developed to assess the negative effects on aquatic biota of sediment-bound contaminants tested in the National Status and Trends Program. This program developed SQCs by using other approaches such as EqPA, SSBA, and methods that related biological and chemical data, i.e., AETA, SLCA, and SQTA (Long and Morgan, 1990). Biological effects as related to contaminant concentrations were identified, and concentration data was sorted in ascending order. The lower 10% of the data was defined as the effects range-low (ER-L) and was considered as the threshold value above which adverse effects on sensitive species or life stages should be expected. The 50% value of the data was defined as the effects range-median (ER-M) where adverse effects on most species were frequently or always observed.

The main advantages of the NSTPA are that: a) it uses a weighted methodology approach, and b) it can be applied directly by using existing data. The main disadvantage of the approach is that its database contains a wide variety of information which was derived by different analytical procedures and pertains to different species and locations throughout the nation.

Long and Morgan (1990) suggested that the degree of confidence for the values of their sediment quality criteria (ER-L, ER-M) can be considered as moderate for trace metals and PCBs, and low for PAHs and Cl-pesticides.

4.3.4 Sediment Background Approach (SBA). The SBA compares pollutant concentrations at contaminated sites with concentrations at sites with "natural" background. Concentrations are considered elevated if they exceed natural levels by a significant margin. Using the SBA, Schropp et al. (1990) developed SQC for metals in Florida's estuaries by comparing metal:alumimum ratios. Empirical relations for "natural" metal levels based on nationwide data were discussed in section 4.1.3.2 (see Table 4.2).

The main advantage of the SBA is its simplicity. Its main disadvantage is the lack of reference to contaminant bioavailability or biological effects.

This approach has been extensively applied throughout the United States for establishment of SQC by various agencies such as the U.S. Environmental Protection Agency (USEPA) (Regions V, VI), the Illinois Environmental Protection Agency, the Texas Water Quality Board, the Florida Department of Natural Resources, and the Virginia Water Control Board.

4.3.5 Screening Level Concentration Approach (SLCA). The SLCA is an approach based on "cause-effect" relationships which define the highest contaminant concentration that can be tolerated by a certain percentage of the benthic infaunal community (Neff et al. 1986). This is accomplished by estimating the species screening level concentration (SSLC), defined as the upper 95% of the contaminant concentration data that species can tolerate without suffering any significant harm.

The advantage of the SLCA is its versatility and reliance on data which is commonly available. Disadvantages of this approach include: a) the inability to provide SQC for single contaminants, b) detailed physicochemical conditions of the aquatic ecosystem are ignored, and c) dose/response relationships are not considered to any extent.

The SLCA has not been applied for the development of SQC in the United States. However, SQC for ten metals, nine Cl-pesticides, and PCBs were developed for the Ontario Province in Canada (Jaagumagi, 1990a,b).

4.3.6 Sediment Quality Triad Approach (SQTA). The SQTA is based on the relation between three components: sediment chemistry, sediment bioassays, and *in-situ* biological characteristics (Long and Chapman, 1985; Chapman, 1989; Long, 1989). Sediment chemistry measures contaminant levels; sediment bioassays assess toxicity effects; and *in-situ* biological characteristics provide information pertaining to abnormalities and community structure alterations, as related to the contaminants, for both benthic and water column species.

The main advantage of the SQTA is that SQC are developed based on an integrated database including three different data sources. The disadvantages of this approach are: a) requirement of a large amount of data, b) no statistical inference from the data, and c) no rigorous criteria available for estimation of each of the separate parameters involved.

The SQTA has been applied in various locations through the nation; e.g., the Great Lakes, San Francisco Bay, the Gulf of Mexico, Puget Sound, Vancouver Harbor, etc.

4.3.7 Spiked Sediment Bioassay Approach (SSBA). The SSBA investigates the sediment quality effects on aquatic biota by using data from well- controlled laboratory studies (Smith and MacDonald, 1992). In this approach, uncontaminated sediments are spiked with known amounts of contaminants, and the pollution effects on certain biological communities are monitored and quantified.

The main advantages of the SSBA are: a) its applicability to almost any sediment type, chemical compound, or biological species, and b) its accountability for bioavailiability factors such as the TOC and the AVS. The main disadvantage of this approach is the fact that spiked-sediment bioassays have been conducted for only a limited number of chemical contaminants (i.e., Cd, Cu, Hg, a few PAHs, and a few Cl-pesticides).

The SSBA has been successfully utilized for the development of SQC by the USEPA. Environment Canada has developed a method to establish SQC based on sediment-spiked bioassays.

4.3.8 Tissue Residue Approach (TRA). The TRA defines "safe" sediment contamination levels which are expected to result in acceptable tissue residues (MacDonald, 1991). This approach relies on the relationship between sediment contaminant levels and tissue residue levels. A relation between tissue residue and biological impacts on secondary consumers must also be developed. A quantitative criterion for estimation of these effects is the acceptable bioaccumulation factor (BAF). The bioaccumulation factor is established from human or wildlife tissue residue data of persistent toxic substances such as PCBs, DDT and Hg.

The advantages of the TRA are its simplicity and the explicit consideration of the bioaccumulation effects. The disadvantages of this approach are that tissue residue criteria are not available for most wildlife species, and dose-response relationships for the majority of the pollutants are non-existent.

The TRA has been successfully applied to the establishment of SQC for several chemical compounds (e.g., Hg, DDT and PCBs).

4.3.9 Weight-of-Evidence Approach (WEA). The WEA was developed for the State of Florida from the National Status and Trends Program Approach. The WEA develops sediment quality criteria based on data from several sediment toxicity analyses such as the AETA, EqPA, SQTA and SSBA (MacDonald, 1992). In this approach, existing databases are divided into two parts: "biological effects data set" (BEDS) and "no biological effects data set" (NBEDS). Based on these data sets, a threshold effects level (TEL) is estimated as

$$TEL = [(BEDS\text{-}L).(NBEDS\text{-}M)]^{1/2} \quad (4.26)$$

where BEDS-L is the 15th percentile concentration in the biological effects data, and NBEDS-M is the 50th percentile concentration in the no biological effects data. By using a safety factor of 2, a no observed effect value (NOEL) is defined as

$$NOEL = TEL/2 \quad (4.27)$$

In addition to the TEL, a probable effects level (PEL) is proposed for addressing contaminant concentrations that may usually or always cause adverse effects on the biota. The PEL is defined as:

$$PEL = [(BEDS\text{-}M)\text{-}(NBEDS\text{-}H)]^{1/2} \quad (4.28)$$

where BEDS-M is the 50th percentile concentration in the biological effects data, and NBEDS-H is the 85th percentile concentration in the no biological effects data.

The advantages of the WEA are its practicality and applicability. Indeed, the method is supported by a comprehensive database on the biological effects of various contaminant concentrations. This method also recognizes the uncertainty associated with the cause-effect relations between chemical concentrations and biota. The main disadvantage of this approach is the existing limitations on the data that describe bioavailability of the sediment-bound contaminants.

The WEA is under review by the Florida Department of Environmental Regulation for possible use as a tool for establishment of sediment quality criteria for the

State of Florida. A list of sediment quality criteria, as derived by the WEA, is given in Table 4.16 (MacDonald, 1992). Either the WEA or a combination or the other approaches can be used for assessment of sediment quality in the Lower St. Johns River Basin.

4.3.10 Comparison of the Sediment Quality Assessment Approaches. The nine approaches that were discussed in the previous sections were critically analyzed by MacDonald (1992) in terms of practicality, cost effectiveness, scientific defensibility, applicability, and overall assessment. The two approaches that scored highest were the EqPA and NSTPA, while the one with the lowest score was the SBA (Table 4.17) (MacDonald, 1992). The conclusions of this comparison are general and apply to any estuarine system including the lower St. Johns River.

4.4 Summary of Sediment Quality Assessment and Modeling.

Management of sediment quality requires a comprehensive integrated approach to the physical, chemical, and biological aspects. Modeling of the various processes involved in an aquatic ecosystem is feasible by means of the mass balance equation, applied for each individual phase (e.g., water, sediments, microbiota), coupled with appropriate relations to account for chemical reaction effects (Sections 4.1.1 and 4.1.2). The applicability of these models is usually restricted by the limited data available for parameter calibration, the lack of understanding of many of the processes involved, and the large number of components that are present in a real-life system. These difficulties are often bypassed by using empirical relations derived from statistical manipulations of the databases (Section 4.1.3).

Table 4.16 Sediment quality criteria, Weight-of-Evidence approach (MacDonald, 1992).

Substance	No. of Records	NOEL	PEL	Degree of Confidence NOEL	PEL
Metals (mg/kg)					
Arsenic	140	8	64	H	M
Cadmium	261	1	7.5	H	H
Chromium	191	33	240	H	H
Copper	218	28	170	H	H
Lead	210	21	160	H	H
Mercury	169	0.1	1.4	H	L
Nickel	161	-	-	-	-
Silver	87	0.5	2.5	H	H
Tributyl Tin	21	-	-	-	-
Zinc	219	68	300	H	H
PAHs (µm/kg)					
Acenaphthene	69	22	450	L	H
Acenaphthylene	45	-	-	-	-
Anthracene	87	85	740	M	H
Fluorene	94	18	460	L	H
2-methyl naphthalene	46	-	-	-	-
Naphthalene	91	130	1100	M	H
Phenanthrene	98	140	1200	M	H
Sum LMW-PAHs	66	250	2400	H	H
Benzo(a)anthracene	79	160	1300	L	H
Benzo(a)pyrene	87	230	1700	H	H
Chrysene	87	220	1700	M	H
Dibenzo(a,h)anthracene	73	31	320	H	M
Fluoranthene	116	380	3200	H	H
Pyrene	93	290	1900	H	H
Sum HMW-PAHs	60	870	8500	M	H
Total PAHs	77	2900	28000	H	H
Pesticides (µm/kg)					
Aldrin	40	-	-	-	-
Azinophosmethyl	0	-	-	-	-
Chlordane	42	-	-	-	-
Chlorthalonil	0	-	-	-	-
Chlorpyrifos	1	-	-	-	-
p,p'-DDD	46	-	-	-	-
p,p'-DDE	64	1.7	130	H	M
p,p'-DDT	45	-	-	-	-
Total DDT	54	4.5	270	L	M
Dieldrin	47	-	-	-	-
Disulfoton	0	-	-	-	-
Endosulfan	9	-	-	-	-
Endrin	19	-	-	-	-
Heptachlor	27	-	-	-	-
Heptachlor epoxide	7	-	-	-	-
Lindane	42	-	-	-	-
Mirex	7	-	-	-	-
Phorate	0	-	-	-	-
Quintozene	0	-	-	-	-
Toxaphene	25	-	-	-	-
Trifluralin	0	-	-	-	-
PCBs (µm/Kg)					
Total PCBs	125	24	260	M	M
Cl-Organics (µm/kg)					
Pentachlorophenol	18	3	15	-	-
Phthalates (µm/kg)					
Bis(2-ethylhexyl)phthalate	31	-	-	-	-
Dimethyl phthalate	15	-	-	-	-
Di-n-butyl phthalate	16	-	-	-	-

Table 4.17 Comparison of sediment quality assessment approaches (MacDonald, 1992).

Evaluation Criteria	AETA	EqPA	NSTPA	SBA	SLCA	SQTA	SSTA	TRA	WEA
Practicality									
Supports development of numerical SQCs?	Y	Y	Y	Y	Y	Y	Y	Y	Y
Feasible to implement in the near future?	N	Y/N	Y	Y	Y/N	N	N	N	Y
Cost Effectiveness									
Expensive to implement?	Y	N	N	N	Y	Y	Y	Y	N
Requires generation of new data?	Y	N	N	N	Y	Y	Y	Y	N
Scientific Defensibility									
Considers bioavailability?	Y/N	Y	Y/N	N	N	N	Y	Y	Y/N
Provides cause and effect relationships?	Y/N	Y	Y/N	N	Y/N	Y/N	Y	N	Y/N
Based on biological effects data?	Y	Y	Y	N	Y	Y	Y	Y/N	Y
Considers data from S.E.?	N	N	Y/N	Y	N	N	N	N	Y/N
Provides weight of evidence?	Y	Y/N	Y	N	N	Y	Y/N	N	Y
Supports definition of ranges of concentration rather than absolute absolute assessment values?	N	N	Y	N	N	N	N	N	Y
Considers mixtures of contaminants?	Y	N	Y	N	Y	Y	N	N	Y
Requires field validation?	Y	Y	Y	Y	Y	Y	Y	Y	Y
Considers site-specification conditions?	Y	N	N	Y/N	N	Y	Y/N	N	Y/N
Applicable to all classes of chemicals?	Y	Y	Y	Y	Y	Y	N	Y	Y
Applicability									
Supports monitoring programs?	Y	Y	Y	Y/N	Y	Y	Y	Y	Y
Supports program identification?	Y	Y	Y	Y/N	Y	Y	Y	Y/N	Y
Supports regulatory programs?	Y	Y/N	Y/N	N	Y/N	Y/N	Y	N	Y
Overall Assessment	G	E	E	P	F	G	G	F	E

Note: E = excellent; G = good; F = fair; P = poor.

The two major components of any sediment quality study are data collection and data analysis. There are a wide variety of sampling designs and sample collection techniques and devices (Sections 4.2.1 to 4.2.7). The degree, extent, and history of sediment contamination can be effectively assessed from a carefully designed monitoring program. Collected sediment samples should be analyzed for estimation of their physicochemical properties, such as grain size characteristics, chemical partitioning, etc. A number of methodologies and procedures are available for this purpose (Section 4.2.8).

Once a sediment quality database has been established, development of a set of sediment quality criteria (SQC) is feasible. These SQC must relate sediment-bound pollutant concentrations to adverse biological effects. There are nine available approaches that can be utilized for the establishment of SQC (Sections 4.3.1 to 4.3.9). The most effective of these approaches are the ones that utilize a wide range of physicochemical factors for different contaminants to assess bioavailiability and bioaccumulation of benthic and water column communities.

For effective management, focused on environmental protection, remediation, or enhancement of an estuarine system, operational decisions should be made based upon a variety of information sources including but not limited to mathematical modeling, field and laboratory data, statistical analyses of the data, sediment quality criteria, etc. This information should incorporate physicochemistry of the water/sediment system and biological effects as related to the prevailing physicochemical conditions.

CHAPTER V

DREDGING ACTIVITIES

K. DREDGING ACTIVITIES AND ASSOCIATED ISSUES

5.1 Major Waterways of the St. Johns River Estuary.

The St. Johns River estuary is associated with four main waterway systems. These are: the main river waterway from Sanford to Jacksonville, the unfinished Cross Florida Barge Canal, the Jacksonville Harbor which extends from downtown Jacksonville to the Atlantic Ocean, and the Atlantic Intracoastal Waterway. The main river waterway has channel improvements between river mile 169.5 above Lake Monroe, and river mile 24.9 at downtown Jacksonville. The Cross Florida Barge Canal involves channel improvements near Palatka, Lake Ocklawaha, and the Henry H. Buckman Lock. Improvements for navigation purposes in the St. Johns River estuary were initiated in 1898 by the Secretary of War. In 1899 he requested and authorized a 13 ft deep channel from Palatka to Jacksonville. Additional improvements were added as far upstream as Lake Harney in 1908 and 1927. Currently the channel is 34 ft deep and 200 ft wide from the ocean to Jacksonville, 13 ft deep and 200 ft wide between Jacksonville and Palatka, 12 ft deep and 100 ft wide between Palatka and Sanford, and 5 ft deep and 100 ft wide between Sanford and Osteen bridge upstream of Lake Monroe. The main river waterway is connected to the Jacksonville Harbor waterway at the Florida East Coast

(FEC) Railroad Bridge at river mile 24.9. The Jacksonville Harbor waterway is 30 ft deep and 300-600 ft wide between FEC Railroad Bridge and Commodore Point. Then the waterway increases to 34 ft deep and 590 ft wide via Terminal Channel to river mile 20. Downstream from that point the waterway continues at 38 ft deep and 400-1,200 ft wide to the Atlantic Ocean. A brief history of the Rivers and Harbors Acts for the St. Johns River estuarine system is given in Table 5.1.

5.1.1 **Jacksonville Port Authority Terminals.** Out of the four waterways of the St. Johns River estuarine system, the most heavily used is the Jacksonville Harbor Waterway. In the vicinity of this waterway, the Jacksonville Port Authority operates four major container and general cargo terminals, i.e., the Blount Island, the Talleyrand Terminal, the Sea-Land Terminal and the Eighth Street Terminal (US Army COE, 1986a).

The Blount Island Terminal handles containers, recreational trailers, scrapmetal, imported autos, crated auto parts, and paper products. The terminal consists of a 3,550 ft linear bulkhead without any piers or slips. At the far western edge of the bulkhead the wharf extends to a T-shaped platform. Future plans extend the eastern side of the wharf for another 1,150 ft, bringing the total bulkhead length to 4,700 ft.

The Talleyrand Terminal is near the upper limit of the 38 ft deep section of the navigable waterway at a distance of approximately 20 miles from the ocean. This terminal handles automobiles, lumber, steel, tires, and petroleum. Talleyrand Terminal comprises 2,838 ft of continuous wharf, over 13.8 acres of covered storage facilities, and 111 acres of paved, lighted, and fenced open storage area.

The Sea-Land Terminal consists of 1,200 ft of wharf and 16 acres of open

Table 5.1 Rivers and harbors Acts for the St. Johns River estuary (US Army COE, 1992).

Act	Authorized Work	Document
06/14/1880	Channel over Volusia Bar with a 15 ft depth at mean low water; Jetties at entrance.	Annual Report, 1979
07/14/1880	Improvements on the Volusia Bar.	
07/05/1884	Improvement between Lakes George and Monroe	
07/13/1892	15 ft depth from Dames Point to Mile Point	
06/03/1896	300 ft wide and 24 ft deep channel from Jax to the ocean; Extended jetties.	House Document No. 346, 53rd Congress, 3rd Session
03/03/1899	200 ft wide and 13 ft deep channel from Jax to to Palatka.	House Document No. 523, 55th Congress, 2nd Session
06/13/1902	5 ft deep channel from Palatka to Sanford; Improvements at the Volusia Bar.	
03/02/1907	24 feet depth at mean low water at the Jax pierhead line.	House Document No. 663, 59th Congress, 1st Session
06/25/1910	100 ft wide and 8 ft deep channel from Palatka to Sanford; Side channel to Enterprise.	House Document No. 1111, 61st Congress, 2nd Session
06/25/1910	Main channel 300 ft wide and 30 ft deep at mean low water.	House Document No. 611, 61st Congress, 2nd Session
03/04/1913	100 ft wide and 8 ft deep channel from St. Johns to Crescent City.	House Document No. 1320, 62nd Congress, 3rd Session
03/02/1919	Improvements to Deep Creek	House Document No. 699, 63rd Congress, 2nd Session
06/03/1930	Cutoffs at Snake Creek, Starks Creek, and Butcher Bend	House Document No. 691, 69th Congress, 2nd Session
06/03/1930	Widening the bend at Dames Point	House Document No. 483 70th Congress, 2nd Session
08/30/1935	400 ft wide-30 ft deep channel along Jax terminals; Widening Dummond Creek, Trout Creek, and Six Mile Creek.	Senate Committee, 74th Congress, 1st Session
03/02/1945	100 ft wide and 10 ft deep channel from Palatka to Sanford; Side channel to Enterprise.	House Document No. 603, 76th Congress, 3rd Session
03/02/1945	Maintenance of existing widths; Widening Terminal Channel to 590 ft.	House Document No. 322, 77th Congress, 1st Session
03/02/1945	Main channel 34 ft deep via Terminal Channel	Senate Document No. 230, 78th Congress, 2nd Session
03/02/1945	Combining two previous Acts (3/2/45 and 7/3/30) into a single project.	House Document No. 445., 78th Congress, 2nd Session
03/02/1945	500 ft wide and 34 ft deep channel from Dame Point to Fulton Cutoff.	Senate Document No. 179, 79th Congress, 2nd Session
07/24/1946	100 ft wide and 12 ft deep channel from Palatka to Sanford.	Senate Document No. 208, 79th Congress
10/27/1965	Maintenance of existing depths at the entrance; Deepen main channel to 38 ft depth to mile 20; Widen main channel near Mile 5 and Mile 7.	House Document No. 214, 89th Congress

storage. This terminal handles cargo destined for foreign ports. Approximately 12,000 containers pass through Sea-Land Terminal annually.

At the Eighth Street Terminal, imported automobiles are landed and processed before being delivered to the dealers. This terminal consists of a 707 ft long wharf, and has 1.5 acres of covered facilities for transit storage and 31 acres of open storage facilities.

Besides the Jacksonville Port Authority terminals, there are many private bulk terminals for handling petroleum on the northern and western shore of the river, between miles 15 and 19. At mile 18 there is a large liquid and dry bulk facility for exporting phosphate rock and superphosphoric acid. This facility includes six concrete silos with a total design storage capacity of 30,000 tons of rock. In addition there are six rubber-lined steel tanks with a total design capacity of 90,000 tons of acid.

There are several shipyards in the vicinity of the metropolitan Jacksonville area. The largest of these shipyards is located in downtown Jacksonville and includes three dry docks and nine wet berths. The dry dock facilities can accommodate vessels with a total length of more than 900 feet and dead weight exceeding 125,000 tons.

5.1.2 Commercial Tonnages. All waterways of the St. Johns River system are subject to heavy commercial traffic, except the Cross Florida Barge Canal (Table 5.2). The Cross Florida Barge Canal is incomplete and commercial traffic is restricted to charter fishing and other recreational activities. In the St. Johns River Waterway the transported freight is comprised primarily of residual fuel oil used by various power plants along the river (Table 5.3). However, the new "coal-by-weir" policy initiated by the Florida Power and Light plants at Sanford and Palatka, is anticipated to reduce the demand for

Table 5.2 Commercial tonnages in the St. Johns River waterways (US Army COE, 1986b).

Year	Jax Harbor	Tons Intracoastal Waterway	St. Johns River
1973	15,513,590	1,182,475	1,058,297
1974	14,794,938	1,505,534	972,954
1975	13,397,951	1,228,098	1,154,664
1976	14,397,951	1,393,370	1,231,882
1977	15,108,032	1,269,218	1,160,926
1978	15,712,893	1,187,362	1,558,423
1979	15,278,008	1,475,654	1,516,374
1980	15,644,000	1,214,075	1,476,207
1981	15,843,690	1,021,788	1,849,213
1982	12,892,163	1,103,955	1,118,148
1983	11,760,221	2,536,585	482,253
1984	11,849,955	2,285,751	248,317
1985	11,332,178	2,606,224	202,110
1986	12,441,812	2,374,127	379,396

Table 3. Freight traffic during 1982 in the St. Johns River waterway (US Army COE, 1986).

Commodity	Tons
Residual fuel oil	1,063,688
Jet fuel	50,772
Naptha, petroleum solvents	3,668
Fresh fish excluding shellfish	20
Total	1,118,148

fuel oil and consequently the freight traffic. According to the "coal-by-weir" policy, power plants receive electrical power from overhead lines for retransmission. This reduces the utility's dependency upon fuel oil.

5.1.3 **Dredging Activities.** In order to maintain safe navigable depths the U.S. Army Corps of Engineers oversees an active dredging program in the estuarine portion of the St. Johns River estuarine system. Dredging operations are mainly conducted between Mayport and downtown Jacksonville. The dredged spoil, is either used for beach nourishment, or disposed of at nearby spoil sites (i.e., Buck Island, Quarantine Island, Blount Island) or in the ocean. Dredging operations follow an approximate 18-month cycle.

Maintenance dredging in the St. Johns River estuary started in early 1950's. As of May 1990, 43,504,169 yd^3 had been dredged at a total cost of $ 67,616,715 (Table 5.4).

5.1.4 **Submerged Land Marinas.** Besides the Jacksonville harbor terminals there are a number of other minor commercial facilities in the estuarine portion of the St. Johns River. These facilities are mainly comprised of ship/boat repair shops, docking sites, marinas, and other enterprises for commercial or sports fishing. The Florida Department of Natural Resources (FDNR) leases the submerged lands adjacent to these enterprises for commercial use. The leases are classified into different categories based on their intended use (Table 5.5). Any leasing of submerged lands by the FDNR involves special consideration for the protection of manatees or any other special environmental condition of the waterbody. There are approximately fifty leases in the St. Johns River and its tributaries which include commercial marinas, wetslips, docks, and shipyards (Table 5.6).

Table 5.4 History of dredging operations in the Jacksonville Harbor (US Army COE, 1992).

Month/ Year	Cuts	Dredged Material Type	Dredged Volume	Cost in US $
11/53	Pilot Town	-	236,230	236,230
05/55	Terminal Channel	-	91,777	25,839
01/56	Bar Cut and Pilot Town	-	714,159	-
-		-	775,772	-
-	Trout River to Mayport	-	122,062	27,618
01/58	Terminal Channel	Silt/Clay	15,000	6,150
12/58	Terminal Channel	Silt/Clay	7,118,998	2,372,882
09/59-11/59	Terminal Channel	Silt/Clay	389,063	102,425
07/60-12/60	Trout River to Mayport	-	771,092	555,546
12/60-02/61	Terminal Channel	-	614,926	174,669
09/61-10/61	Terminal Channel	Silt/Clay	187,920	77,099
09/61	Harbor at Dunn Creek	-	15,680	6,050
05/62-06/62		-	-	151,500
07/62-09/62	Bar Cut and Pilot Town	-	475,787	146,322
11/62-05/63	Trout River to Mayport	-	1,048,943	644,333
3-12/63,11/64	Terminal Channel	-	161,290	137,121
11/64-12/64	Bar Cut	-	371,306	246,033
07/64-04/65		-	1,171,206	528,480
04/65	Dredging Depot	-	8,618	7,333
01/66-02/66	Terminal Channel	-	151,999	133,805
04/66-05/66	Pilot Town	-	226,321	349,006
06/66-10/66	Harbor	-	1,333,074	825,544
09/66-11/66	Blount Island Channel	Sand/Silt	1,157,291	179,427
01/67-02/67	Bar Cut and Pilot Town	Sand	239,511	169,467
06/67-07/67	Bar Cut	Silt/Sand	454,571	222,817
07/68-12/68		-	486,693	344,250
08/68	Terminal Channel	Silt/Clay	127,509	108,254
04/70-09/70	Blount Isl. & Main River	-	1,098,156	1,052,649
08/72-12/72	N.W. Section 1	-	6,235,768	12,804,964
05/72-11/72	Harbor Channels	-	375,766	462,590
03/74-05/74	Cuts 3-6 & Pilot Town	-	347,283	627,925
11/74-03/76	Section 2	-	1,308,531	5,980,290
12/74-08/75	Blount Island Channel	-	781,922	1,165,365
04/75-06/76		-	660,862	4,931,365
04/76-12/77		-	1,541,694	6,572,961
06/80-11/80	Harbor	Sand/Silt	2,242,794	3,019,770
06/80-11/80	Ocean to Mayport	Sand	822,806	3,144,210
07/80-8/80	Bar Cut	Clayey Sand, Shells	265,153	864,501
11/80-03/81	Term. Chan. & Blount Is.	Silt/Sand/Clay	529,380	2,181,250
10/81-04/82	NAS Pier 139	Black & Grey Clay	617,051	2,468,034
03/83		Ocean Borrow	8,005	60,000
1983		-	237,644	-
02/93-09/83	Blount Isl. & Main Riv.	Silt/Sand	1,113,226	2,303,175
10/83-08/84	Ocean to Trout River	Sand	1,629,500	4,179,263
1984		-	436,729	-
1985		-	218,760	-

Continued:

Table 5.4 History of dredging operations in the Jacksonville Harbor (US Army COE, 1992).
(Continued)

Month/ Year	Cuts	Dredged Material Type	Dredged Volume	Cost in US $
04/85-6/85		-	1,284,425	765,550
10/86-05/87	Term. Chan. to Blount Is.	-	959,850	3,230,381
10/86-05/87	Blount Island	-	50,550	118,671
10/86-05/87	Dredging Depot	-	2,242	35,043
10/86-05/87	JEA/JPA Term., Blount Is.	-	52,435	169,365
01/88	Bar Cut	-	210,521	875,000
01/88-10/88	Mill Cove East Channel	-	830,112	1,604,186
01/88-10/88	Mile Cove West Channel	-	616,227	1,122,007
11/89-04/90	Cut 3-6, D/A S. Jetty	Sand	559,979	-
05/90	Cut 7-12, D/A Buck Isl.	Sand	392,000	-

5.2 Environmental Concerns Related to Navigation, Marinas and Dredging.

Activities associated with the operation of shipyards and marinas along with the heavy volume of ship/boat traffic in the St. Johns River estuary may be the source of potential or existing environmental problems. These problems relate to fuel oil leaking, dissolving of ship paint and other protective coatings, residuals of petroleum solvents, or accidental spills/leaking of toxic cargo (e.g., superphosphoric acid, naptha, fuel oil), etc. Once introduced into the water column, these pollutants are mainly adsorbed by suspended particulates which are eventually deposited on the bottom sediments. Stirring of the sediments during maintenance dredging of the waterways re-introduces the pollutants into the water column. This pollution cycle creates potential environmental hazards for the aquatic biota even long after the pollution source has ceased to exist. Inland or marine disposal of the contaminated dredge spoil can cause additional environmental problems through groundwater seepage, surface runoff, mixing, etc.

Table 5.5 Types of submerged land leases (Barber, 1992).

Code	Activity description
01	Commercial marina with renting wetslips. Include condos if 50% of their wetslips are available to the general public.
02	Public/Local Government.
03	Yacht Clubs/Country Clubs with required membership.
04	Multi family/but upland revenue-generating (i.e., housing developments, trailer parks, apartments).
05	Condominiums (requires upland ownership)
06	Commercial upland activity for use of the customer at no charge (i.e., temporary docking and/or fishing associated with upland revenue generating activities such as restaurants, hotels, and motels).
07	Miscellaneous commercial upland enterprises where there is a charge associated with the use of their overwater structure (i.e., charter boats, tour boats and fishing piers).
08	Ship building/boat repair service facilities.
09	Commercial fishing related (i.e. off loading of seafood, seafood processing).
10	Aquaculture
26	Private single-family residential docking facilities; townhome docking facilities; single-family subdivision/single-family homeowners docking facilities (leases requires upland ownership)

Any major anthropogenic activity (including dredging) in the Lower St. Johns River Basin is subject to a number of environmental regulations. These regulations were established by federal statutes, executive orders and state policies (Table 5.7). Before initiation of any action it must be ascertained that all of the applicable environmental

Table 5.6 Submerged lands marinas in the LSJRB (Barber, 1992).

NAME	ADDRESS	LOCATION	FACILITY	WATERBODY	TYPE
Ms. Candis L. Whitney	3027 Highway 17	Orange Park	Commercial marina	SJR & Doctor's Inlet	01
Villas Continental & Yacht Club	2223 Astor Street	Orange Park	Commercial docking	St. Johns River	01
Florida Water Sports	1492 River Lane	Green Cove Springs	Commercial marina	Black River	01
Inlet Marine Sales & Service	P.O. Box 1207	Orange Park	Commercial marina	Doctor's Lake	01
Moody Bros. of Jacksonville	State Road 16	Green Cove Springs	Dry dock & boat lift	St. Johns River	08
Edwards Marina	451-B Trout River Dr.	Jacksonville	Commercial marina	Trout River	01
Julington Creek Partnership	12807 San Jose Blvd.	Jacksonville	Commercial marina	Julington Creek	01
Jackie's Seafood	531 Trout River Dr.	Jacksonville	Commercial marina	Trout River	01
Pier 66 Marine, Inc.	8137 North Main St.	Jacksonville	Commercial marina	Trout River	01
Sisters Creek Marina	8203 Heckaher Drive	Jacksonville	Comm. mar. w/ boat repair	Sisters Creek	01
Ortega Yacht Club Marina, Inc.	4585 Lakeside Drive	Jacksonville	Commercial marina	Ortega River	01
Seafarers Marina	455 Trout River Dr.	Jacksonville	Commercial marina	Trout River	01
Mandarin Holiday Marina	12796 San Hose Blvd.	Jacksonville	Commercial marina	Julington Creek	01
Sea-Jays Boat Storage	3436 Lakeshore Blvd.	Jacksonville	Comm. mar. w/ upland boat stor.	Ortega River	01
Ortega River Boatyard/Marina	4451 Herschel Str.	Jacksonville	Commercial marina	Ortega River	01
Clapboard Creek Marina, Inc.	6220 Heckaher Dr.	Jacksonville	Com. Mar. w/ upland boat repair	St. Johns River	01
Register Enterprises	308 Raleigh Road	Jacksonville	Commercial marina	St. Johns River	01
Lamb's Yacht Center	P.O. Box 7039	Jacksonville	Com. mar. w/ dry dock	Ortega River	01
Nelson Holding, Inc.	8065 Buffalo Ave.	Jacksonville	Commercial marina	Trout River	01
John H. Owens	4252 Lakeside Ave.	Jacksonville	Commercial marina	Ortega River	01
John H. Owens	2374 Lakeshore Blvd.	Jacksonville	Commercial marina	Cedar River	01
Wiley E. Andreu	14983 Mandarin Road	Jacksonville	Commercial marina	Julington Creek	01
Mayport Marine	4852 N. Ocean Blvd.	Mayport	Commercial marina docking	St. Johns River	01
S & H Land Corporation	P.O. Box 9852	Ft. George Island	Marina & commercial docking	St. Johns River	08
Moody Bros. of Jacksonville	5137 Arlington Road	Jacksonville	Commercial dock w/ marine lift	St. Johns River	08
Atlantic Marine, Inc.	8503 McKenna Road	Ft. George Island	Commercial shipbuilding/docking	Sisters Creek	08
Jacksonville Shipyards, Inc.	1000 E. Bay Street	Jacksonville	Commercial docking (dolphin)	St. Johns River	08
Cross State Towing Service, Inc.	5138 Arlington Road	Jacksonville	Marine repair	St. Johns River	08
St. Johns Bar Pilots, Inc.	4910 Ocean Street	Mayport	Commercial docking	St. Johns River	08
Trailer Marine Transport	1045 Bond Avenue	Jacksonville	Commercial deepwater shipping	St. Johns River	08
Atlantic Marine, Inc.	8503 McKenna Road	Ft. George Island	Commercial ship repair	Sisters Creek (SJR)	08
Atlantic Dry Dock Corp.	8500 Heckscher Drive	Jacksonville	Com. shipyard w/ upl. repair	St. Johns River	08
Andy's Mini Marina	Hwy 309	Georgetown	Commercial marina	SJR-Lake George	01
Wolfe's Camp	P.O. Box 118	Welaka	Commercial docking	St. Johns River	01
River Bass Resort	Roberts Blvd	San Mateo	Docking w/ upl. rest. & cottag.	St. Johns River	01
J.L.O.G., Inc.	309 Georgetown Road	Georgetown	Com. marina w/ upl. restaurant	St. Johns River	01
Port Cove Mainland Marina	Merks Landing Rd.	Georgetown	Commercial marina	St. Johns River	01
Calkins Development, Inc.	Route 5, Box 1826	Palatka	Commercial marina & boat ramp	St. Johns River	01
Acosta Creek Marina & Boatworks	Hwy 309/Acosta Rd.	Welaka	Commercial marina	St. Johns River	01
Mission Inn Golf & Tennis	P.O. Box 441	Howey-In-The-Hills	Commercial marina	Little Lake Harris	01
Holland Marine, Inc.	25934 Holmer Drive	Astor	Commercial marine railway	St. Johns River	08
Six Mile Marina	State Road 13	Green Cove Springs	Commercial marina & boat ramp	Six Mile Creek	01
Toroi Fish Camp, Inc.	4780 A1A South	St. Augustine	Com. mar. w/ boatlifts & camp	St. Johns River	01
Amity Anchorage Yacht/Pool Club	1106 Friendship Dr.	Switzerland	Commercial marina	St. Johns River	01
H.P. Demery	312 N. Main Street	Jacksonville	Commercial marina	Tolomato River	01
Conch House Marina Restaurant	57 Comares Ave.	St. Augustine	Com. docking w/ upland resort	Salt Run	01
Harol Hendry Jr. & Stephanie	39 Magnolia Ave.	St. Augustine	Commercial marina	Matanzas River	01
Hontoon Landing Marina	2317 River Ridge Rd.	Deland	Commercial marina	St. Johns River	01
Three Buoys Houseboat	2280 Hontoon	Deland	Commercial marina	St. Johns River	01

requirements are satisfied.

5.2.1 Critical Issues in Dredging Operations. Dredging is considered an operation of major environmental concern. This is particularly true if it involves dredging of contaminated sediments (see Sections 2.3.1.1 and 2.4).

Table 5.7 Statutes for environmental requirements (US EPA, 1992).

Federal Statutes

Archeological and Historic Preservation Act,	as amended, 16 USC 469, *et seq.* PL 93-291
Clean Air Act,	as amended , 42 USC 1857h-7, *et seq.* PL 91-604
Clean Water Act,	as amended, FWPCA 33 USC 1251, *et seq.* PL 92-500
Coastal Barrier Resources Act,	as amended, 16 USC 3501, *et seq.* PL 97-348
Coastal Zone Management Act,	as amended, 16 USC 1451, *et seq.* PL 92-583
Endangered Species Act,	as amended, 16 USC 1531, *et seq.* PL 93-205
Estuary Protection Act,	as amended, 16 USC 1221, *et seq.* PL 90-454
Federal Water Project Recreation Act,	as amended, 16 USC 460-1(12), *et seq.* PL 89-72
Fish and Wildlife Coordination Act,	as amended, 16 USC 661, *et seq.* PL 85-624
Land and Water Conservation Fund Act,	as amended, 16 USC 4601-11, *et seq.* PL 88-578
Marine Mammal Act,	as amended, 16 USC 1361, *et seq.* PL 92-522
Marine Protection, Research and Sanctuaries Act,	as amended, 33 USC 1401, *et seq.* PL 92-532
National Historic Preservation Act,	as amended, 16 USC 470a, *et seq.* PL 89-655
National Environmental Policy Act,	as amended, 42 USC 4321, *et seq.* PL 91-190
River and Harbor Act,	as amended, 33 USC 401, *et seq.*
Watershed Protection and Flood Prevention Act,	as amended, 16 USC 1001, *et seq.* PL 83-566
Wild and Scenic River Act,	as amended, 16 USC 1271, *et. seq.* PL 90-542

Executive Orders

Floodplain Management,	EO 11988
Protection of Wetlands,	EO 11990
Protection and Enhancement of Environmental Quality,	EO 11514, as amended EO 11991
Protection and Enhancement of the Cultural Environment,	EO 11593
Federal Compliance with Pollution Control Standards,	EO 11654

State Policies

Florida Coastal Management Program

A comprehensive study on the environmental effects of dredging operations was conducted by the Commission of the Permanent International Association of Navigation Congresses (PIANC) (Per Bruun, 1981). In their report the PIANC Commission made the following concluding remarks:

o Assessment of the environmental effects of dredge operations should emphasize specific quantitative rather than general qualitative standards pertaining to expanding lists of toxic substances which are subject to regulations and control.

- Water pollution, which leads to bottom sediment pollution, should be eliminated at its source.
- Contamination of the dredged spoil rarely originates from the dredging activity itself but rather from pre-existing degraded conditions.
- It is more cost effective to terminate pollution at its source rather than to remediate polluted sediments.
- Adverse effects from dredging/disposal activities should be minimized or eliminated based on a careful cost-benefit analysis.
- The existing criteria to evaluate the degree of contamination of dredged materials, although very sophisticated, have not been as yet sufficiently standardized to allow precise and consistent assessment.
- There is not a general agreement on the applicability of elutriate tests for spoil disposal in ocean waters.
- Maintenance dredging is more likely to cause environmentally adverse effects than other marine related dredging activities because the former usually involves contaminated sediments from anthropogenic activities.
- Other dredging operations mainly involve changes of the waterway cross-section that affect drastically the biological communities.
- The biological oxygen demand (BOD) can be reduced by covering polluted sediments with biologically clean sand.
- Certain adverse environmental effects can be reduced by application of alternate methods of dredging.

- Disposal areas should be selected after a careful cost-benefit analysis which emphasize minimization of undesirable environmental effects and maximization of beneficial use.
- Because of the growing lack of suitable disposal areas in the proximity of dredging projects the need for reuse of the removed material is expected to be stressed.
- Any general guidelines pertaining to the water depth and the distance from the shore for disposal of polluted dredged materials must be used with care because each ecosystem has its own special features that control its behavior.
- Inland disposal of the spoil, especially in contained areas, has fewer adverse environmental effects than open water disposal. This is due to the fact that open water disposal is more difficult to control and contain due to the water dynamics. However, attention should be given to agricultural uses of reclaimed disposal areas.
- Inland disposal of polluted sediments may cause groundwater contamination.
- From all of the various dredging methods, plain suction is the one that has the least environmentally adverse impact to the dredging site since turbidity effects are minimum.
- Excessive turbidity from hopper dredges should be controlled by careful operation in critical areas.
- Borrow areas should be located downstream of seagrass beds so that the turbidity plume is driven past them.
- Adverse environmental effects can be reduced by proper selection of dredging equipment.

- Noise pollution during dredging can be reduced by proper lubrication, muffling, and modernization of the equipment.

In summary, the PIANC Commission re-iterated the fact that planning for a dredging operation, including selection of dredge equipment, operational procedures, disposal methods, and scheduling, should be based on *ad hoc* studies incorporating not only sound economics and engineering but also a detailed environmental analysis.

5.2.2 Marine Protection, Research and Sanctuaries Act. Major projects such as dredging and spoil dumping must comply with the requirements of the National Environmental Policy Act (NEPA) of 1969. NEPA requires preparation of an Environmental Impact Statement (EIS) for projects that may significantly affect the quality of the environment.

Dumping of any type of materials in the oceans is regulated by the Marine Protection, Research and Sanctuaries Act (MPRSA) of 1972. Section 102 of this Act authorizes the Environmental Protection Agency (EPA) to designate sites and establish criteria for selection and management of ocean dumping sites (EPA Ocean Dumping Regulations and Criteria, 40 CFR 220-229). Section 103 of the Act authorizes the U.S. Army Corps of Engineers to issue permits for the transportation of dredge materials for ocean dumping.

Regarding ocean dumping, items for consideration as established in 40 CFR, Sections 228.5 and 228.6 are as follows:

- 40 CFR 228.5(a): The dumping of materials into the ocean will be permitted only at sites or in areas selected to minimize the interference of disposal activities with other activities in the marine environment, particularly avoiding areas of existing fisheries or shellfisheries, and regions of heavy commercial or recreational navigation.

- o 40 CFR 228.5(b)1: Locations and boundaries of disposal sites will be chosen so that temporary perturbations in water quality or other environmental conditions during initial mixing caused by disposal operations anywhere within the site can be expected to be reduced to normal ambient seawater levels or to undetectable contaminant concentrations or effects before reaching any beach, shoreline, marine sanctuary or known geographically limited fishery or shellfishery.
- o 40 CFR 228.5(c): If, any time during or after disposal site evaluation studies, it is determined that existing disposal sites presently approved on an interim basis for ocean dumping do not meet the criteria for site selection set forth in 228.5 and 228.6, the use of such sites will be terminated as soon as alternative disposal sites can be determined.
- o 40 CFR 228.5(d): The size of ocean disposal sites will be limited in order to localize the identification and control of any immediate adverse impacts and permit the implementation of effective monitoring and surveillance programs to prevent adverse long-range impacts. The size, configuration, and location of any disposal site will be determined by part of the disposal site evaluation or designation study.
- o 40 CFR 228.5(e): EPA will, wherever feasible, designate ocean dumping sites beyond the edge of the Continental Shelf and other such sites that have been historically used.
- o 40 CFR 228.6(a)1: Geographical position, depth of water, bottom topography and distance from the coast.

- o 40 CFR 228.6(a)2: Location in relation to breeding, spawning, nursery, feeding or passage areas of living resources in adult or juvenile phases.
- o 40 CFR 228.6(a)3: Location in relation to beaches, and other recreational, cultural, and protected areas.
- o 40 CFR 228.6(a)4: Types and quantities of waste to be disposed of and proposed methods of release, including methods of packing the waste, if any.
- o 40 CFR 228.6(a)5: Feasibility and accessibility for surveillance and monitoring.
- o 40 CFR 228.6(a)6: Dispersal, horizontal transport and vertical mixing characteristics of the area, including prevailing current direction and velocity, if any.
- o 40 CFR 228.6(a)7: Existence and effects of current and previous discharges and dumping in the area, including cumulative effects.
- o 40 CFR 228.6(a)8: Interference with shipping, fishing, recreation, mineral extraction, desalination, aquaculture, areas of special scientific importance, and other legitimate uses of the ocean.
- o 40 CFR 228.6(a)9: Existing water quality and ecology of the site as determined by available data or by trend assessment or baseline surveys.
- o 40 CFR 228.6(a)10: Potential for the development or recruitment of nuisance species in the disposal site.
- o 40 CFR 228.6(a)11: Existence at or in close proximity to the site of any significant natural or cultural features of historical importance.

Any sediment management action in the St. Johns River estuary must considered all of the above items. In the environmental impact assessment it should be clearly stated that all of the proposed management actions were developed after detailed consideration of these items.

5.2.3 Management and Monitoring of Disposal Sites. Once selected and activated, Ocean Dredged Material Disposal Sites (ODMDSs) should be carefully monitored and managed. The main management objectives of an ODMDS are the following (US EPA, 1992):

- Protection of the marine environment,
- Beneficial use of dredged spoil whenever possible, and
- Record-keeping of all activities at the ODMDS.

Before any disposal at the selected site, samples must be collected from new construction and/or maintenance dredging material for verification of its suitability for ocean disposal. This verification is valid for three years and includes:

- A case-specific evaluation for the exclusion criteria (i.e., radioactive and military chemical and biological warfare materials) (40 CFR 227.139(b),2),
- Determination of the need for toxicity and bioaccumulation tests of the non-excluded materials, and
- Laboratory analysis and confirmation that the non-excluded materials are suitable for ocean disposal.

These efforts are coordinated and reviewed by the US EPA and the US Army COE. Details on methods of disposal and monitoring are also developed and issued by the same agencies.

In order to assess the extent of any environmental impact due to dredging a monitoring program should be planned and implemented. Part of the monitoring should incorporate recording of all of the dredging activities including (US EPA, 1992):

- Vessel name and type,
- Captain of the vessel,
- Date/Time,
- Number of scows in tow and distance from vessel, or any other vessel used,
- Horizontal position of the vessel and the scows with recording frequency of: 5 minutes within the channel limits, 2 minutes from the dredging area to disposal site and return, 30 seconds within the disposal site during actual disposal,
- Location of points of disposal initiation and completion,
- Draft conditions of the vessel and the scows recorded simultaneously with the horizontal positions,
- Volume of disposed material, and
- Disposal technique used.

Based on these records and on follow-up surveys of the disposal site any environmental impacts on the general area can be estimated. The follow-up surveys must be taken along the same lines/points of the pre-project surveys. Data from the pre-project surveys are used as baseline for the monitoring program. The extent of the impact is assessed based on the following information:

- Movement of materials into estuaries, marine sanctuaries, aquaculture areas, or onto beaches or shorelines.
- Absence from the disposal site of pollution-sensitive biota which are otherwise present in the general area.
- Progressive degradation but no seasonal changes in water or sediment quality at the disposal site attributable to the spoil material.

- o Progressive degradation but no seasonal changes in composition and abundance of pelagic, demersal or benthic biota at the disposal site attributable to the spoil material.
- o Bioaccumulation of toxic pollutants in marine biota at or near the disposal site.

A schematic representation (flow-chart) of all actions pertinent to an ODMDS monitoring is presented in Figure 5.1 (US EPA, 1992).

5.2.4 **Estimation of Chemical Contamination Levels.** Development of a sediment management plan for the LSJRB requires knowledge of the type and amount of pollutants, their spatial distribution and if possible, their temporal changes. This data must be compared to information pertaining to anthropogenic activities in the basin so that cause/effects relationships are established.

Estimation of the degree of estuarine contamination is a laborious procedure that requires collection of water and sediment field samples and laboratory analyses of the samples. Depending on the type of chemical element or compound there are certain pollutant concentration limits that can be detected by using currently available laboratory techniques. These detection limits vary between water and sediment samples (Florida DER, 1992) (Table 5.8). Pollutant extraction from water and sediment samples should be conducted according to analytical methods developed by the US EPA and adopted by the Florida Department of Environmental Regulation (Tables 5.9, 5.10). These methods ensure best results and consistency of chemical analyses that allows comparison between results taken from different laboratories.

5.2.5 **Summary and Conclusions.** The St. Johns River estuary and its tributaries are subject to intensive anthropogenic activities including main harbor facilities, marinas,

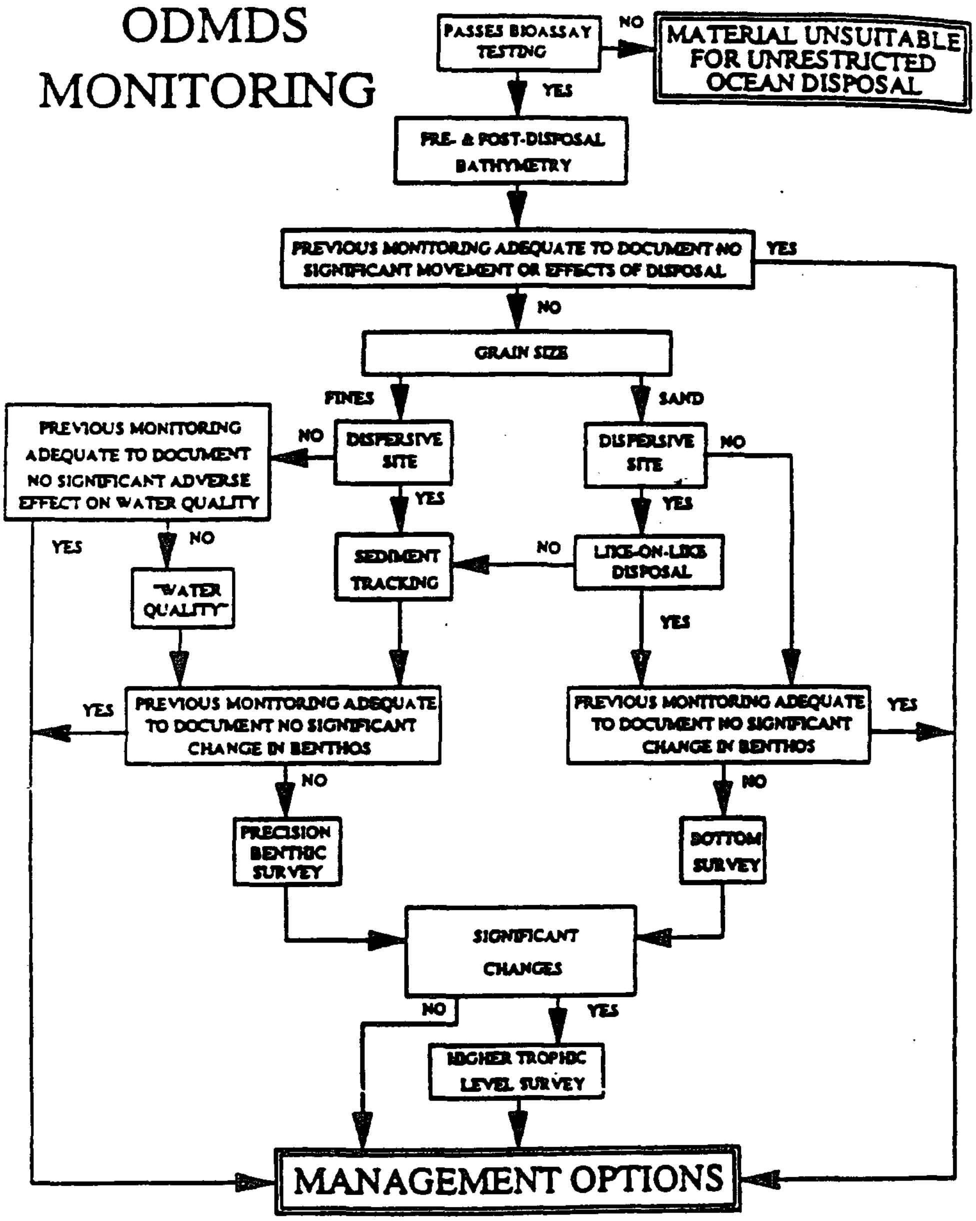


Figure 5.1 ODMDS monitoring (US EPA, 1992).

Table 5.8 Detection limits for water and sediment samples (Florida DER, 1992).

Parameter	Water or elutriate (μg/l)	Sediment (dry weight) (ppm)
Aluminum	N/A	10
Antimony	1.0	1.0
Arsenic	1.0	1.0
Barium	N/A	10
Cadmium	0.1	0.05
Calcium	N/A	10
Chromium	0.5	1.0
Copper	0.5	1.0
Cyanide	20	1.0
Fluoride	100	1.0
Iron	1.0	1.0
Lead	0.5	1.0
Magnesium	N/A	10
Mercury	0.2	0.01
Nickel	0.5	1.0
Selenium	1.0	1.0
Silver	0.05	0.05
Strontium	N/A	10
Tin	10	10
Zinc	0.5	1.0
Chlorinated pesticides	0.05-1.0	0.001-0.1
Oil & grease	100	1.0
Aldicarb	1.0	0.01
Aliphatic hydrocarbons	10	0.5
Dioxin, chlorinated classes	0.002-0.1	0.001-0.1
Dibenzo furan, chlorinated classes	0.002-0.1	0.001-0.1
Volatile organics	10	0.05
Phosphorated pesticides	1-5	0.1-0.5
Nitrogen containing aromatics	10	0.05-0.2
Herbicides chlorinated	0.1-10	0.1-10
Butyltin	0.05	0.01
PAHs	5-50	0.05-0.2
PCBs	0.5-1.0	0.1
Phenols	10-50	0.05-0.2
Ammonia	2	0.6
BOD	1000	500
Nitrate	1	0.4
Phosphate	2	0.8
Total phosphorus	10	1.0
TKN	20	1.0
TOC	1000	100

Table 5.9 Procedures for brackish water chemical analysis (Florida DER, 1992).

Parameter	Analytical method		Special Procedures
Antimony	EPA 206.3*	AA-hydride	
Arsenic	EPA 206.3*	AA-hydride	
Cadmium	EPA 213.2*	AA-furnace	APDC-DDDC chelation, pH 5 (200 ml sample)
Chromium	EPA 218.3*	AA-furnace	APDC-DDDC chelation, pH 2.4 (100ml sample)
Copper	EPA 220.2*	AA-furnace	APDC-DDDC chelation, pH 5 (200 ml sample)
Cyanide	EPA 335.2*	distillation	
Fluoride	EPA 340.2*	AA-potentiometric	
Iron	EPA 236.2*	AA-furnace	APDC-DDDC chelation, pH 5 (200 ml sample)
Lead	EPA 239.2*	AA-furnace	APDC-DDDC chelation, pH 5 (200 ml sample)
Mercury	EPA 245.1*	Cold vapor	
Nickel	EPA 249.2	AA-furnace	APDC-DDDC chelation, pH 5 (200 ml sample)
Selenium	EPA 270.3*	AA-hydride	
Silver	EPA 272.2*	AA-furnace	APDC-DDDC chelation, pH 5 (200 ml sample)
Tin	EPA 282.2*	AA-furnace	
Zinc	EPA 289.1*	AA-flame	APDC-DDDC chelation, pH 5 (200 ml sample)
Cl-pesticides	EPA 608**	GC-EC	Extraction w/methylene chloride (750 ml sample)
Oil and grease	EPA 413.2*	IR	
Aldicarb	EPA 531**	HPLC	
Aliphatic HC	EPA 610**	GC-FID	
Dioxins (Cl-)	EPA 8280***	GC-MS	
Dibenzo furans	EPA 8280***	GS-MS	
Volatile organics	EPA 601/602**	GC	
	EPA 624**	GC-MS	
P-pesticides	EPA 8140***	GC	
N- aromatics	EPA 625**	GC-MS	
Cl-Herbicides	EPA 615**	GC-EC	
Butyltin		GC-MS-SIM	
		GC-PID	
		AA-hydride	
PAH	EPA 610**	GC-FID	
PCBs	EPA 608**	GC-EC	
Phenols	EPA 604**	GC-EC or FID	
	EPA 625**	GC-MS	
Ammonia	EPA 350.3*	Potentiometric	
	EPA 350.2*	Automated	
BOD	EPA 405.1*	5 day - 20°C	
Nitrate	EPA 353.1*	Hydrazine	
Phosphate	EPA 365.2*	Ascorbic acid	
	EPA 365.4*	Colorimetric automated	
Total phosphorus	EPA 365.2*	Persulfate digestion, ascorbate	
	EPA 365.4*	Colorimetric automated	
TKN	EPA 351.3*	Digestion, potentiometric	
Total solids	EPA 160.2*	Gravimetric - 150°C	
TOC	EPA 415.1*	Oxidation	

Source: (*): EPA-600/4-79-020 & EPA-600/4-82-055, "Methods for Chemical Analysis of Water and Wastes";

(**): 40 CFR Part 136, "Guidelines for Establishing Test Procedures for the Analysis of Pollutants";

(***): EPA SW 846, "Test Methods for Evaluating Solid Wastes", 2nd Edition, July, 1982.

Table 5.10 Laboratory procedures for sediment chemical analysis (Florida DER, 1992).

Parameter	Analytical method		Special procedures
Aluminum	EPA 202.1*	AA-flame	
	EPA 200.7*	ICP	
Antimony	EPA 204.2*	AA-furnace	
	EPA 206.3*	AA-hydride	
Arsenic	EPA 206.2*	AA-furnace	
	EPA 206.3*	AA-hydride	
Barium	EPA 208.1*	AA-flame	
	EPA 200.7*	ICP	
BOD	EPA 405.1*	5 day - 20°C	
Cadmium	EPA 213.2*	AA-furnace	
Calcium	EPA 215.1*	AA-flame	
	EPA 200.7*	ICP	
Chromium	EPA 218.3*	AA-furnace	
	EPA 200.7*	ICP	
Copper	EPA 220.2*	AA-furnace	
	EPA 200.7*	ICP	
Cyanide	EPA 9010**	Distillation-colorimetric	
Fluoride	EPA 340.2*	Potentiometric	Extraction 1:20 (dry sediment:distilled water)
Iron	EPA 236.1*	AA-flame	
	EPA 200.7*	ICP	
Lead	EPA 239.2*	AA-furnace	
Mercury	EPA 245.5*	Cold vapor	Digest w/sulf.+nitric acid+potass.permanganate
Magnesium	EPA 200.7*	ICP	
	EPA 242.1*	AA-flame	
Nickel	EPA 249.2*	AA-furnace	
	EPA 200.7*	ICP	
Selenium	EPA 270.2*	AA-furnace	
Silver	EPA 272.2*	AA-furnace	
Strontium	EPA 6010*	ICP	
Zinc	EPA 289.1*	AA-flame	
	EPA 200.7*	ICP	
TNK	EPA 351.3*	Digestion, potentiometric	
	EPA 351.1*	Automated	
Oil and grease	EPA 413.2*	IR	
Cl-pesticides	EPA 8080**	GC-EC	Extraction by EPA 3550** or EPA 3540**
PAHs	EPA 8010**	GC-EC	Extraction by EPA 3550** or EPA 3540**
	EPA 8250**	GC-MS	Extraction by EPA 3550** or EPA 3540**
	EPA 8270**	GC-MS	Extraction by EPA 3550** or EPA 3540**
PCBs	EPA 8080**	GC-EC	Extraction by EPA 3550** or EPA 3540**
Phenols	EPA 8040**	GC-EC and FID	Extraction by EPA 3550** or EPA 3540**
	EPA 8250**	GC-MS	Extraction by EPA 3550** or EPA 3540**
	EPA 8270**	GS-MS	Extraction by EPA 3550** or EPA 3540**
Aldicarb	EPA 531**	HPLC	
Aliphatic HC	EPA 8100*	GC-FID	
Dioxins (CL-)	EPA 8280**	GC-MS	
Dibenzo furans	EPA 8280**	GC/MS	
Volatile organics	EPA 8010/8020**	GC	
P-pesticides	EPA 8140**	GC	Extraction by EPA 3350** or EPA 3540**

Continued:

Table 5.10 Laboratory procedures for sediment chemical analysis (Florida DER, 1992). (Continued:)

Parameter	Analytical method		Special procedures
N-aromatics	EPA 8270**	GC-MS	
	EPA 8250**	GC-MS	
Cl-Herbicides	EPA 8150**	GC-EC	
Butyltin		GC-MS-SIM	
		GC-PID	
		AA-hydride	
Ammonia	EPA 350.3*	Potentiometric	Extraction 1:20 (dry sediment:distilled water)
	EPA 350.2*	Automated	Extraction 1:20 (dry sediment:distilled water)
Phosphate	EPA 365.2*	Ascorbic acid	Extraction 1:20 (dry sediment:distilled water)
	EPA 365.1*	Colorimetric automated	Extraction 1:20 (dry sediment:distilled water)
Nitrate	EPA 353.3*	Cadmium reduction	Extraction 1:20 (dry sediment:distilled water)
	EPA 353.1*	Automated hydrazine	Extraction 1:20 (dry sediment:distilled water)
Total phosphorus	EPA 365.2*	Persulfate digestion ascorbate, colorimetric	
	EPA 365.1*	Colorimetric automated	
Total solids	EPA 160.2*	Gravimetric - 150°C	
TOC	EPA 415.1*	Oxidation	Analysis by ampule technique (e.g., OI instrument)
Grain size	ASTM-19-D422		
Specific gravity	ASTM-19-D854		
Atterberg limits	ASTM-19-D423 and D424		

Source: (*): EPA-600/4-79-020 & EPA-600/4/82-055 "Methods for Chemical Analysis of Water and Wastes";

(**): EPA SW 846 "Test Methods for Evaluating Solid Wastes", 2nd Edition, July, 1982.

contaminant source. The monitoring program should include both sites with known and heavy ship/boat traffic. In addition there is an active dredging program for navigation purposes. Generally, these activities can pose potential hazards to the environment through negligence or accidents.

In order to evaluate the impact of dredging on the river's ecosystem a monitoring program should be established. Past data on the environmental conditions of the system must be compiled and used as baseline information. Since the major pollutants resulting from these marine activities include fuel byproducts, solvents and paints, it is relatively easy to identify the contamination problems and sites suspected of potential contamination. In order to evaluate seasonal effects, monitoring should be conducted on a seasonal basis (i.e., twice per year). However, if this is cost-prohibitive, monitoring can be limited to an annual basis. However, Depending on the data evaluation new stations may be added (if new contamination problems arise) or monitoring of old stations may be reduced or deleted if the pollution source ceases to exist and the conditions of the area under surveillance are back to normal.

In the Lower St. Johns River Basin emphasis should be based on the metropolitan Jacksonville area where most of the ship/boat industry related activities are concentrated. Past data collection studies (see Chapter III) already documented the presence of trace metals and organic compounds (PAHs, PCBs and coprostanol) in this general area.

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