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Volume I

Ecological Risk Assessment of the Marsh Area of the LCP Chemical Site in Brunswick, Georgia

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CONTENTS

	Page
LIST OF FIGURES AND PLATES	vii
LIST OF TABLES	ix
ACRONYMS AND ABBREVIATIONS	xii
LIST OF SPECIES	xiii
1. INTRODUCTION	1-1
2. SITE BACKGROUND	2-1
2.1 PHYSICAL SETTING	2-1
2.2 SITE HISTORY	2-2
3. PROBLEM FORMULATION	3-1
3.1 CONCEPTUAL SITE MODEL	3-1
3.1.1 Operable Units	3-1
3.1.2 Chemicals of Potential Concern	3-1
3.1.3 Potential Ecological Receptors	3-1
3.1.4 Potential Exposure Routes	3-3
3.2 SELECTION OF ECOLOGICAL ENDPOINTS AND RECEPTORS	3-3
3.2.1 Ecological Endpoints	3-3
3.2.1.1 Assessment Endpoints	3-3
3.2.1.2 Measurement Endpoints	3-5
3.2.2 Ecological Receptors	3-6
3.3 EXPERIMENTAL PLAN	3-6
3.3.1 Objectives	3-6
3.3.2 Study Design	3-7
3.3.3 Assessment Approach	3-7
3.3.3.1 Surface Water Investigation	3-8
3.3.3.2 Sediment Investigation	3-8
3.3.3.3 Mercury Methylation Investigation	3-9

Page

3.3.3.4 Wildlife Food-Web Investigation 3-9
3.3.3.5 Threatened and Endangered Species Investigation 3-9

4. METHODS 4-1

4.1 MODIFICATIONS TO PROPOSED SCOPE OF WORK 4-1

4.2 SAMPLING DESIGN 4-5

4.2.1 Surface Water Investigation 4-5
4.2.2 Sediment Investigation 4-7
4.2.2.1 Channel Sediment Chemistry Study 4-7
4.2.2.2 Marsh Sediment Chemistry Study 4-8
4.2.2.3 PCB Congener Study 4-9
4.2.2.4 Field Macroinvertebrate Study 4-12
4.2.3 Wildlife Food-Web Investigation 4-12
4.2.4 Threatened and Endangered Species Investigation 4-14

4.3 ANALYSIS METHODS 4-14

4.4 DATA MANAGEMENT 4-18

4.5 QUALITY ASSURANCE AND QUALITY CONTROL 4-18

4.5.1 Field Quality Control 4-18
4.5.2 Laboratory Quality Assurance 4-19

4.6 DATA ANALYSIS 4-20

4.6.1 Chemical Analyses 4-20
4.6.2 Wildlife Food-Web Exposure Models 4-20

5. EXPOSURE ASSESSMENT 5-1

5.1 ECOSYSTEM CHARACTERIZATION 5-1

5.2 STRESSOR CHARACTERIZATION 5-2

5.2.1 Natural Physical and Chemical Stressors 5-2
5.2.2 Spatial Distribution of CoPCs in Water 5-2
5.2.3 Spatial Distribution of CoPCs in Sediment 5-5
5.2.3.1 Channel Sediments 5-7
5.2.3.2 Marsh Sediments 5-11

Page

5.2.4 Congener Analysis of Environmental Samples 5-22
5.2.4.1 Homolog Distribution in Sediment, Vegetation, and Invertebrate Samples 5-22
5.2.4.2 Congener Distribution in Sediment, Vegetation, and Invertebrate Samples 5-22

5.3 EXPOSURE CHARACTERIZATION FOR AQUATIC LIFE AND WILDLIFE 5-26

5.3.1 Body Burdens of Prey of Wildlife 5-26
5.3.2 Wildlife Food-Web Exposure Modeling Approach 5-27
5.3.2.1 Area Use Factor 5-27
5.3.2.2 Time Use Factor 5-28
5.3.2.3 Food Ingestion Rate 5-28
5.3.2.4 Sediment Ingestion Rate 5-28
5.3.2.5 Water Ingestion Rate 5-29
5.3.2.6 Receptor Body Weight 5-29
5.3.2.7 Chemical Bioavailability 5-29
5.3.2.8 Mercury Speciation 5-31
5.3.2.9 Life History Characteristics of Receptor Species 5-33

5.4 WILDLIFE FOOD-WEB EXPOSURE ESTIMATES 5-38

5.5 PREVIOUS INVESTIGATIONS OF BODY BURDENS IN WILDLIFE AT THE LCP SITE 5-38

5.5.1 Clapper Rail 5-38
5.5.2 Diamondback Terrapin 5-46

5.6 AERIAL SURVEYS OF WADING BIRD ABUNDANCE 5-46

6. EFFECTS ASSESSMENT 6-1

6.1 EVALUATION OF DIRECT TOXICITY OF CoPCs TO WATER COLUMN ORGANISMS 6-1

6.1.1 Laboratory Toxicity Study 6-1
6.1.1.1 Mysids 6-1
6.1.1.2 Sheepshead Minnows 6-6
6.1.2 Field Bioaccumulation Study 6-6

6.2 EVALUATION OF DIRECT TOXICITY OF CoPCs TO BENTHIC ORGANISMS 6-6

Page

6.3 EVALUATION OF FOOD-WEB RELATED TOXICITY OF MERCURY AND PCBs TO WILDLIFE 6-12

6.3.1 Toxic Effects of Mercury and PCBs 6-12
6.3.1.1 Mercury 6-15
6.3.1.2 PCBs 6-15
6.3.2 Derivation of Toxicity Reference Values 6-16
6.3.2.1 Derivation of Toxicity Reference Values for Birds 6-17
6.3.2.2 Derivation of Toxicity Reference Values for Mammals 6-18

6.4 EVALUATION OF THREATENED AND ENDANGERED SPECIES 6-18

7. RISK CHARACTERIZATION 7-1

7.1 RISK ESTIMATION 7-1

7.1.1 Maintenance of Ecological Health of the Salt Marsh Community 7-1
7.1.2 Protection of Long-Term Health and Reproductive Capacity of Aquatic Reptiles 7-4
7.1.3 Protection of Long-Term Health and Reproductive Capacity of Omnivorous Mammals 7-4
7.1.4 Protection of Long-Term Health and Reproductive Capacity of Piscivorous Mammals 7-4
7.1.5 Protection of Long-Term Health and Reproductive Capacity of Birds 7-5
7.1.6 Protection of Long-Term Health and Reproductive Capacity of Fishery Resources 7-5
7.1.7 Protection of Fishery Nursery Function 7-6
7.1.8 Protection of Individual Threatened or Endangered Marine Turtles 7-6
7.1.9 Protection of Individual Wood Storks 7-7
7.1.10 Protection of Individual West Indian Manatees 7-7
7.1.11 Protection of Individual Shortnose Sturgeons 7-7

7.2 COMPARISON WITH ESTIMATION OF RISK PRESENTED BY SPRENGER ET AL. (1997) 7-8

7.2.1 Raccoon 7-10
7.2.2 Clapper Rail 7-17
7.2.3 Marsh Wren 7-18
7.2.4 Conclusion 7-19

Page

7.3 UNCERTAINTY ANALYSIS 7-19

7.3.1 Representativeness of Sampling Locations 7-20
7.3.2 Representativeness of Sample Sizes of Prey 7-20
7.3.3 Characterization of Body Burdens in Prey 7-20
7.3.4 Exposure Estimates for Receptors 7-21
7.3.5 Toxicity Reference Values for Receptors 7-22
7.3.6 Choice of Receptors 7-23

7.4 CONCLUSIONS 7-23

8 REFERENCES 8-1

APPENDIX A - Wading Bird Utilization of Two Saltmarsh Systems in Glynn County, Georgia, in 1996
APPENDIX B - Laboratory Study of Direct Toxicity of CoPCs to Benthic Organisms
APPENDIX C - Mercury Methylation Investigation
APPENDIX D - Area-weighted Average Sediment Concentrations for Mercury and Aroclor® 1268 in Channel and Non-channel Areas at the LCP Site

LIST OF FIGURES AND PLATES

		Page
Figure 1-1.	Location of the LCP Site and reference stations, Brunswick, Georgia	1-2
Figure 4-1.	Location of sampling stations used in the surface water investigation (water chemistry study, laboratory toxicity study, and field bioaccumulation study) and the sediment investigation (channel sediment chemistry study only) for the LCP Site and reference stations	4-2
Figure 4-2.	Concentrations of lead, total mercury, and Aroclor® 1268 and percent total organic carbon (TOC) in surface channel sediment from the LCP Site	4-3
Figure 4-3.	Locations of marsh grid sediment sampling stations and two stations (LM-8 and LM-13) used in the sediment toxicity dilution series at the LCP Site	4-4
Figure 4-4.	Vertical chemical profiles of marsh sediment at the LCP Site and reference stations	4-10
Figure 4-5.	Locations of January sediment sampling at the LCP Site	4-11
Figure 4-6.	Concentrations of lead, total mercury, methylmercury, and Aroclor® 1268 in prey species from the LCP Site and reference stations	4-13
Figure 5-1.	Overview of total mercury and methylmercury concentrations in unfiltered surface water from the LCP Site and reference stations	5-6
Figure 5-2.	Overview of lead, total mercury, and Aroclor® 1268 concentrations in channel sediment from the LCP Site and reference stations	5-10
Figure 5-3.	Vertical chemical profiles of channel sediment from the LCP Site and reference stations	5-12
Figure 5-4.	Concentrations of Aroclor® 1268 in marsh sediment from creek transects A and B (CRTA and CRTB) at the LCP Site	5-21
Figure 5-5.	Homolog concentrations in sediment, vegetation, and invertebrates from the LCP Site	5-24

Plate 1. Lead concentrations (mg/kg) in marsh sediment grid

Plate 2. Mercury concentrations (mg/kg) in marsh sediment grid

Plate 3. Aroclor® 1268 concentrations (mg/kg) in marsh sediment grid

LIST OF TABLES

		Page
Table 4-1.	Rationale for selection of sampling locations for the LCP Site and reference stations	4-6
Table 4-2.	Summary of analytical methods and data quality objectives for the LCP Site and reference stations as part of this investigation	4-15
Table 5-1.	Physical and chemical characteristics of surface water from the LCP Site and reference stations and comparisons to Georgia marine water quality standards	5-3
Table 5-2.	Chemical characteristics of channel sediment from the LCP Site and reference stations	5-8
Table 5-3.	Summary of chemistry data for creek channel surface sediment samples from the LCP Site	5-13
Table 5-4.	Summary of chemistry data for marsh grid sediment samples from the LCP Site	5-14
Table 5-5.	Summary of chemistry data for sediment samples from reference stations	5-15
Table 5-6.	Summary of chemistry data for fine-scale marsh grid sediment samples from the LCP Site	5-19
Table 5-7.	Polychlorinated biphenyl homolog analyses for surface (0–5 cm) sediment and tissue samples from the LCP Site	5-23
Table 5-8.	Polychlorinated biphenyl congener analyses for surface (0–5 cm) sediment and tissue samples from the LCP Site	5-25
Table 5-9.	Summary of area-weighted average sediment concentrations for mercury and Aroclor® 1268 in channel and non-channel samples at the LCP Site	5-30
Table 5-10.	Concentrations of chemicals of potential concern in prey species used in wildlife food-web exposure modeling for the LCP Site	5-32
Table 5-11.	Exposure values for food-web modeling of potential for unacceptable risk to West Indian manatee in the marsh at the LCP Site	5-39

Table 5-12. Exposure values for food-web modeling of potential for unacceptable risk to river otter in the marsh at the LCP Site 5-40

Table 5-13. Exposure values for food-web modeling of potential for unacceptable risk to raccoon in the marsh at the LCP Site 5-41

Table 5-14. Exposure values for food-web modeling of potential for unacceptable risk to wood stork in the marsh at the LCP Site 5-42

Table 5-15. Exposure values for food-web modeling of potential for unacceptable risk to clapper rail in the marsh at the LCP Site 5-43

Table 5-16. Exposure values for food-web modeling of potential for unacceptable risk to marsh wren in the marsh at the LCP Site 5-44

Table 5-17. Exposure values for food-web modeling of potential for unacceptable risk to diamondback terrapin in the marsh at the LCP Site 5-45

Table 6-1. Statistical analysis of survival of mysids exposed to surface water from the LCP Site and reference stations 6-2

Table 6-2. Statistical analysis of growth (weight) of mysids exposed to surface water from the LCP Site and reference stations 6-4

Table 6-3. Statistical analysis of post-hatch survival of embryonic/larval sheepshead minnows exposed to surface water from the LCP Site and reference stations 6-7

Table 6-4. Survival and growth of hatchery oysters deployed at the LCP Site and reference stations 6-10

Table 6-5. Summary of taxa identified during the benthic macroinvertebrate study for the LCP Site and reference stations 6-13

Table 6-6. Summary of quantitative results of the benthic macroinvertebrate study for the LCP Site and reference stations 6-14

Table 7-1. Hazard quotients of potential for unacceptable risk to wildlife receptors in the marsh at the LCP Site 7-2

Table 7-2. Comparison of assumptions made for food-web exposure modeling calculations for assessment of potential for unacceptable risk to selected wildlife receptors potentially inhabiting the area of marsh subject to removal actions 7-11

Table 7-3. Exposure values for food-web modeling of potential for unacceptable risk to raccoons potentially inhabiting the area of marsh subject to removal actions 7-12

Table 7-4. Exposure values for food-web modeling of potential for unacceptable risk to clapper rails potentially inhabiting the area of marsh subject to removal actions 7-13

Table 7-5. Exposure values for food-web modeling of potential for unacceptable risk to marsh wrens potentially inhabiting the area of marsh subject to removal actions 7-14

Table 7-6. Hazard quotient comparison of this ecological risk assessment and the assessment by Sprenger et al. (1997) for selected wildlife species potentially inhabiting the area of marsh subject to removal actions 7-15

ACRONYMS AND ABBREVIATIONS

AlliedSignal	AlliedSignal Inc.
AOC	Administrative Order by Consent
ARCO	Atlantic Richfield Company
CoPC	chemical of potential concern
EPA	U.S. Environmental Protection Agency
ERA	ecological risk assessment
GaDNR	Georgia Department of Natural Resources
Georgia Power	Georgia Power Company
HPAH	high molecular weight PAH
LCP Site	Linden Chemicals and Plastics Site
LCS	laboratory control sample
LOAEL	lowest-observed-adverse-effect level
$LOAEL_T$	test species LOAEL
LPAH	low molecular weight PAH
mg/kg	milligrams per kilogram
mg/L	milligrams per liter
μg/kg	micrograms per kilogram
ng/L	nanograms per liter
NOAEL	no-observed-adverse-effect level
$NOAEL_T$	test species NOAEL
$NOAEL_W$	wildlife species NOAEL
PAH	polycyclic aromatic hydrocarbon
PCB	polychlorinated biphenyl
ppt	parts per thousand
PTI	PTI Environmental Services
QAPP	quality assurance project plan
QA/QC	quality assurance and quality control
RI/FS	remedial investigation and feasibility study
TCB	tetrachlorobiphenyl
TOC	total organic carbon
TRV	toxicity reference value
SOW	scope of work
SVOC	semivolatile organic compound

LIST OF SPECIES

American oyster	*Crassostrea virginica*
Atlantic bottle-nosed dolphin	*Tursiops truncatus*
Atlantic croaker	*Micropogon undulatus*
bald eagle	*Haliaeetus leucocephalus*
black drum	*Pogonias cromis*
black needlerush	*Juncus roemerianus*
blue crab	*Callinectes sapidus*
clapper rail	*Rallys longirostris*
diamondback terrapin	*Malaclemys terrapin*
eastern painted turtle	*Chrysemys picta*
fiddler crab	*Uca* spp.
grass shrimp	*Palaemonetes pugio*
great blue heron	*Ardea herodia*
great egret	*Casmerodius albus*
green turtle	*Chelonia mydas*
hawksbill turtle	*Eretmochelys imbricata*
Kemp's ridley turtle	*Lepidochelys kempi*
killifish	*Fundulus* spp.
leatherback turtle	*Dermochelys coriacea*
little blue heron	*Egretta caerulea*
loggerhead turtle	*Caretta caretta*
marsh rabbit	*Sylvilagus palustris*
marsh rice rat	*Oryzomys palustris*
marsh wren	*Cistothorus palustris*
mink	*Mustela vison*
mummichog	*Fundulus heteroclitus*
mysids	*Mysidopsis bahia*
penaeid shrimp	*Penaeus* spp.
periwinkle	*Littorina* sp.
raccoon	*Procyon lotor*
red drum/channel bass	*Sciaenops ocellata*
river otter	*Lutra canadensis*
sheepshead	*Archosargus probatocephalus*
sheepshead minnow	*Cyprinodon variegatus*
shortnose sturgeon	*Acipenser brevirostrum*
smooth cordgrass	*Spartina alterniflora*
snowy egret	*Egretta thula*
spot	*Leiostomus xanthurus*
spotted sea trout	*Cynoscion nebulosus*
West Indian manatee	*Trichechus manatus*
white ibis	*Eudocimous albus*
wood stork	*Mycteria americana*

1. INTRODUCTION

This ecological risk assessment (ERA) addresses the marsh area of the Linden Chemicals and Plastics Site (LCP Site) in Brunswick, Georgia (Figure 1-1). The ERA is mandated by an Administrative Order by Consent (AOC) dated July 6, 1995 (U.S. EPA 1995a), between Region IV of the U.S. Environmental Protection Agency (EPA) and AlliedSignal Inc. (AlliedSignal), Atlantic Richfield Company (ARCO), and Georgia Power Company (Georgia Power). The AOC was issued in response to the Comprehensive Environmental Response, Compensation and Liability Act of 1980, as amended. The ERA was conducted to assess potential risk to ecological receptors in the marsh operable unit following completion of removal actions at the LCP Site. Removal actions have been in progress within the industrialized part of the LCP Site and immediately adjacent marsh (Operable Unit 1) and groundwater (Operable Unit 2) since 1994.

The AOC for the LCP Site specifies that AlliedSignal is required to conduct studies in the marsh and mandates that an ERA (a baseline assessment) be conducted for the Site according to protocols in *Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments* (U.S. EPA 1996). The AOC further references the use of a food-web model to assist EPA in determining cleanup goals. The general scope of work (SOW) (U.S. EPA 1995b) attached to the AOC also states the need to evaluate both flora and fauna at the Site and requires the use of a food-web model to evaluate potential risks of bioaccumulative substances. Finally, the SOW identifies *Guidance for Conducting Remedial Investigations and Feasibility Studies under CERCLA* (U.S. EPA 1988a) as the main guidance manual for conducting the remedial investigation and feasibility study (RI/FS).

This ERA identifies the marsh area referenced in the AOC and SOW for the LCP Site as the "estuarine operable unit." The rationale for this change in terminology is that a coastal marsh typically consists of low-lying, tidally influenced "soft" land; whereas, an estuary consists of marsh and open water (Odum 1961). Both of these habitats are addressed in the risk assessment. However, these estuarine studies have been integrated with studies being conducted for the upland part of the Site (GeoSyntec 1996).

The Environmental Response Team (ERT) of the EPA previously produced an ERA for the marsh in support of the EPA Removal Program objectives (Sprenger et al. 1997). The overall approach of that ERA was to establish exposure-response relationships between sediment concentrations and biological responses along a contaminant concentration gradient. The ERT ERA produced to support the Removal Program was conservative in its assumptions when Site data did not exist. Based on the conclusions of the ERT ERA, the Superfund Removal Branch of EPA determined that the presence of acute toxicity warranted a removal action in the marsh. The EPA then determined the removal area using a

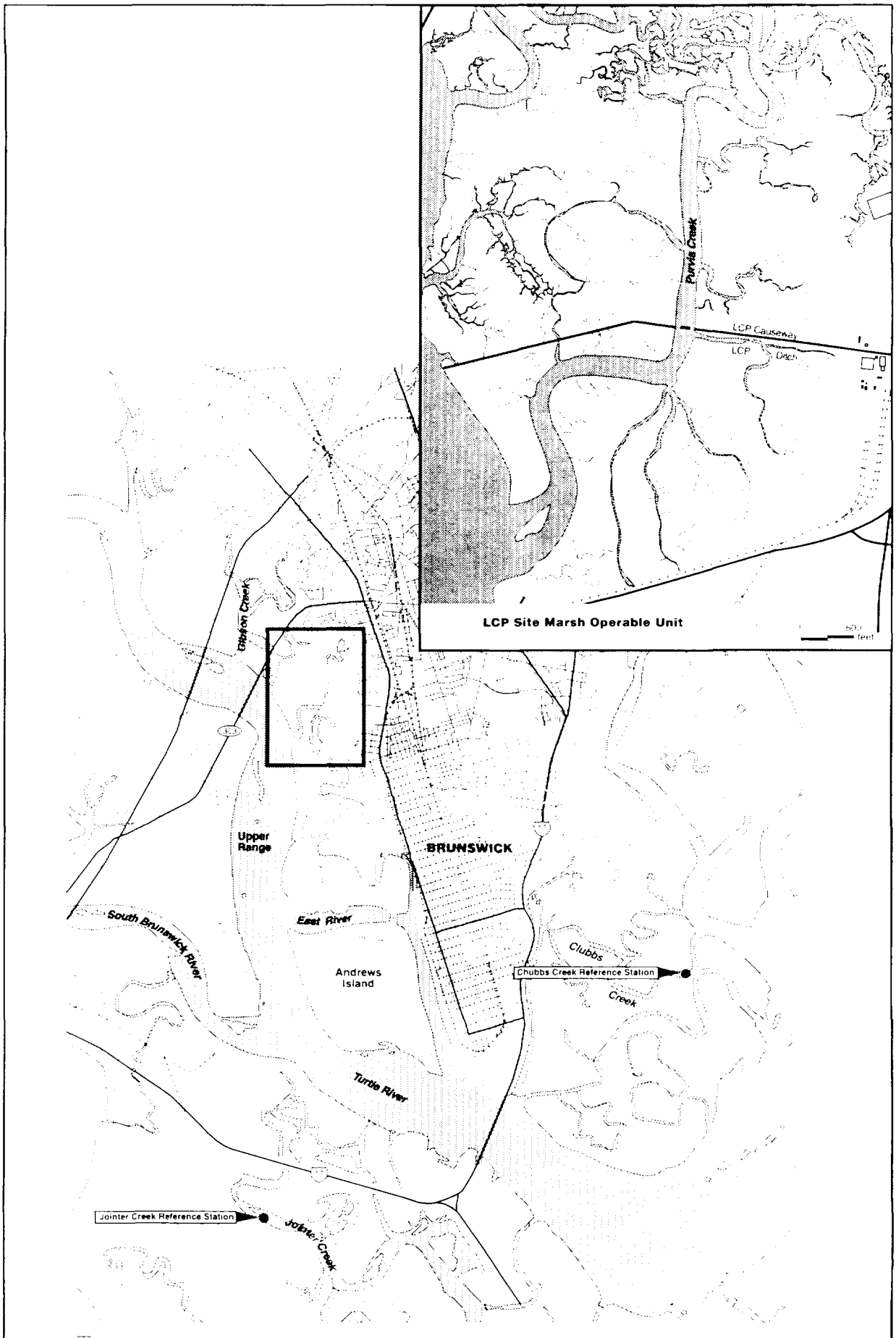


Figure 1-1. Location of the LCP Site and reference stations, Brunswick, Georgia.

geographic information system to optimize mass removal while minimizing impacts to the marsh.

The objective of this ERA is different from the ERT ERA. This ERA uses food-web models that incorporate site-specific data when available to evaluate the potential ecological effects of contaminants in the marsh under post-removal conditions. Consequently, this ERA addresses marsh areas outside the removal area. Despite having differing objectives, the two risk assessments used similar methods to assess ecological risks, except that the ERT ERA used more conservative assumptions than this ERA. However, both risk assessments are valid evaluations of risk and reflect a range of conservative exposure or effects assumptions used in the calculation of hazard quotients. A comparison of the risk assessment conclusions for this ERA and the ERT ERA is presented in this report, and explanations for the alternative assumptions and resultant risk estimates are presented in an uncertainty analysis.

The fundamental objectives of this ERA for the estuarine operable unit at the LCP Site are to:

- Determine the distribution of chemicals of potential concern (CoPCs) throughout the estuary
- Quantify the extent of potential hazard of CoPCs to ecological receptors (aquatic life and wildlife) in the estuary
- Provide a basis for determining the need for appropriate remedial alternatives for the LCP Site.

The analyses used in this ERA are intended to comply with protocols contained in the two previously referenced EPA documents (U.S. EPA 1988a, 1996), as supplemented by other relevant documents and guidelines published by EPA and the State of Georgia.

This ERA, which follows the framework recommended by U.S. EPA (1992a), is organized into the following major sections:

- Site Background
- Problem Formulation
- Methods
- Exposure Assessment
- Effects Assessment
- Risk Characterization
- Conclusions.

This ERA contains four appendices: Appendix A is the complete report submitted by A. Lawrence Bryan, Jr. of the Savannah River Ecology Laboratory on wading bird utilization of the estuary at the LCP Site and a reference site. Appendix B presents the methods and results of sediment toxicity tests of marsh sediments subject to removal actions. Appendix C presents the results of the mercury methylation investigation, and Appendix D presents a description and results of the approach to derive area-weighted averages for mercury and Aroclor® 1268 in channel and non-channel marsh sediments at the LCP Site.

Volume II, 1996 Field Investigation Report, contains data and associated quality assurance reports for the data collected by PTI Environmental Services (PTI) and CDR Environmental Specialists on behalf of AlliedSignal. These data were used as the primary basis for this ERA. Supplemental data from other investigations (e.g., EPA studies) were also used in the ERA.

2. SITE BACKGROUND

This section includes a discussion of the geographic location and the physical features of the LCP Site and a brief discussion of the operational history of the industrial part of the Site.

2.1 PHYSICAL SETTING

The LCP Site is located immediately northwest of the city of Brunswick, in Glynn County, Georgia (see Figure 1-1). The Site, which has an area of about 550 acres, consists of approximately 70 acres of largely developed (industrialized) upland and 480 acres of estuary. The estuary, situated west of the industrialized area, drains into Purvis Creek, which, in turn, discharges to the Turtle River. A ditch, termed the LCP Ditch, runs from the industrialized upland part of the Site to Purvis Creek. A secondary road parallels the ditch along its northern bank and, at one time, connected with a boardwalk (now in ruins) that crossed Purvis Creek and the most western marsh to the Turtle River. The Turtle River/Purvis Creek estuarine system is tidally influenced, with tidal range being about 6 ft in the vicinity of the LCP Site.

The LCP Site is bordered by a county landfill and police firing range on the north, Ross Road on the east, and the Georgia-Pacific Pulp and Paper Company on the south. The pulp and paper company discharges effluent to the Turtle River, as does the City of Brunswick Academy Creek Wastewater Treatment Plant (via Academy Creek), which is located south of the pulp and paper company.

The surface geology at the LCP Site consists of sandy beach and dune deposits in the upland area and organic-rich silty clays in the tidal marsh (GeoSyntec 1996). These surface sediments are about 15 m thick. Underlying the surface sediments is a layer of coarse sand, silty clay, and sandstone (deposited during the later Miocene Epoch), which extends to a depth of approximately 55 m. These later Miocene sediments are underlain by a sequence of silt, clay, phosphatic sand, and limestone of the Hawthorne group (an early Miocene formation) that extends to a depth of about 150 m.

Storm water runoff from the industrial part of the Site, which historically discharged to the estuary, is now mostly contained by storm water diversion structures. Potentiometric surface measurements indicate that shallow-aquifer groundwater (0–15 m in depth) discharges to the estuary (GeoSyntec 1996).

2.2 SITE HISTORY

Industrial activities began at the LCP Site in 1836 when construction began on an approximately 4,000-ft segment of the Brunswick-Altamaha Canal that ran in a north-south direction along the interface between the upland and marsh at the Site (GeoSyntec 1996). The canal eventually extended approximately 12 miles from Academy Creek (Brunswick Harbor) north to the Altamaha River. The canal opened in 1854 and was operated only until 1855. Waste-disposal and soil-filling activities appear to have occurred along parts of the canal that traversed the Site (i.e., in the north and south disposal areas).

The Atlantic Refining Company, ARCO's predecessor, used the Site as a petroleum refinery from 1919 through 1935. The refinery processed Gulf Coast and Mexican crude oil into finished products that included light asphalt, fuel oil, lubricating oil, gas oil, kerosene, and gasoline. The plant boiler was fueled by coal until 1922, after which oil was used as fuel.

Georgia Power purchased part of the Site from ARCO and operated an oil-fired power-generating facility during the period of 1937 through 1950 that, in 1941, reached a generating capacity of 5,500 kW (GeoSyntec 1996). The Dixie Paint and Varnish Company (which eventually became the Dixie O'Brien Corporation and, subsequently, a subsidiary of the O'Brien Corporation) purchased another part of the Site from ARCO in 1941, where it operated a paint and varnish manufacturing facility until 1955 (GeoSyntec 1996).

Allied Chemical and Dye Company (now AlliedSignal) purchased all of the land that constitutes the Site (except a 2.9-acre parcel still owned by Georgia Power) in 1955 (GeoSyntec 1996). AlliedSignal constructed and operated a chlor-alkali facility at the Site, using the Solvay (mercury-cell) process. Primary products of the chlor-alkali operation were chlorine gas, hydrogen gas, and sodium hydroxide solution.

LCP Chemical-Georgia Inc. (a division of the Hanlin Group, Inc.) purchased all of the Site (except the 2.9-acre parcel owned by Georgia Power) from AlliedSignal in 1979 (GeoSyntec 1996) and continued to operate the chlor-alkali facility. LCP currently owns all but the 2.9-acre part of the Site and operated the facility until 1994, when production was discontinued.

3. PROBLEM FORMULATION

3.1 CONCEPTUAL SITE MODEL

Issues relevant to the conceptual site model, including identification of operable units, CoPCs, potential ecological receptors, and potential exposure routes are addressed below.

3.1.1 Operable Units

The estuarine investigation focuses on the marsh portion of Operable Unit 1. The unit, identified here as the estuarine operable unit, includes both the marsh proper and open-water systems, as described in Section 2.1 *Physical Setting*.

3.1.2 Chemicals of Potential Concern

Major CoPCs in the estuarine ecosystem at the LCP Site have been identified by U.S. EPA (1995a) and the Natural Resource Trustees for the LCP Site (1995). These major CoPCs are lead, mercury, polychlorinated biphenyls (PCBs) (specifically Aroclor® 1268), and polycyclic aromatic hydrocarbons (PAHs).

3.1.3 Potential Ecological Receptors

Aquatic life and wildlife are the general categories of estuarine receptors potentially exposed to CoPCs. Specific receptors selected for this ERA and the rationale for their selection are discussed later in this section.

Aquatic life consists of two major components: water-column organisms and benthic organisms. Aquatic life inhabiting the water column includes phytoplankton, zooplankton, nektonic macroinvertebrates, and various trophic levels of fishes. These types of receptors are characterized by a range of mobility that occurs either passively (in the case of most plankton) or actively (for fishes).

Although phytoplankton occur in the estuary, they are less important than detritus as a food source that supports the base of the food web (Pomeroy and Wiegert 1981). Two basic types of zooplankton inhabit the estuary: holoplankton (organisms characterized by a passively floating life form throughout their lives) and meroplankton (organisms with a passively floating life form during only early life stages). One category of meroplankton, ichthyoplankton (early life stages of fishes), is of critical ecological importance because of

the use of the estuary by a wide variety of fishes as a breeding, nursery, and/or growing area. These fishes include various members of the families Cyprinodontidae (mummichog, *Fundulus heteroclitus*; sheepshead minnow, *Cyprinodon variegatus*), Sparidae (sheepshead, *Archosargus probatocephalus*), Sciaenidae (spotted sea trout, *Cynoscion nebulosus*; spot, *Leiostomus xanthurus*; Atlantic croaker, *Micropogon undulatus*; black drum, *Pogonias cromis*; red drum or channel bass, *Sciaenops ocellata*), and Bothidae (lefteye flounders). In addition, the shortnose sturgeon (*Acipenser brevirostrum*), an endangered fish species, may pass through the estuary but is not known to frequent the Turtle River or Purvis Creek.

Macroinvertebrates that inhabit the water column of the estuary as meroplankton and/or adults include several types of penaeid shrimp (*Penaeus* spp.), grass shrimp (*Palaemonetes pugio*), mysids (*Mysidopsis bahia*), and the American oyster (*Crassostrea virginica*). These receptors, like demersal fishes, also frequent the benthic environment at various times during their lives but are commonly considered to be water-column organisms during at least a part of their life cycle. These organisms are routinely used in toxicity testing of aqueous media (U.S. EPA 1985a, 1988b).

Aquatic life indigenous to the benthic environment of the estuary include various species of marine worms (polychaetes), univalve molluscs (periwinkles [*Littorina* sp.]), and numerous crustaceans (e.g., cirripeds [little grey barnacles], amphipods, and two types of decapods [fiddler crabs, *Uca* spp. and blue crabs, *Callinectes sapidus*]). As in the case of some of the "water-column" receptors, benthic receptors do not always exhibit complete fidelity to their environment. Many benthic receptors are characterized by early life stages (larvae) that are planktonic. Moreover, juvenile and adult benthic animals sometimes forage in the water column. In addition to benthic animals, plants are important benthic receptors in the estuary because, when they die, they form most of the detritus that directly supports the lower levels of the estuarine food web. These plants are primarily emergent smooth cordgrass (*Spartina alterniflora*) and black needlerush (*Juncus roemerianus*).

Wildlife receptors indigenous to the estuary include a variety of reptiles, birds, and mammals. The most common reptile in Atlantic coast salt marshes is the diamondback terrapin (*Malaclemys terrapin*). In addition, several species of threatened or endangered Atlantic sea turtles, the green turtle (*Chelonia mydas*), Kemp's ridley turtle (*Lepidochelys kempi*), hawksbill turtle (*Eretmochelys imbricata*), loggerhead turtle (*Caretta caretta*), and leatherback turtle (*Dermochelys coriacea*), may visit the Site but do not use the area for extended periods. Birds include a variety of grebes, cormorants, herons and bitterns, ibises, geese, marsh ducks, mergansers, vultures, hawks, ospreys, falcons, rails (including the clapper rail, *Rallys longirostris*), stilts, plovers, sandpipers, gulls and terns, pelicans, skimmers, kingfishers, and passeriform birds (including the marsh wren, *Cistothorus palustris*). Threatened or endangered avian species that may frequent the estuary are the wood stork (*Mycteria americana*) and bald eagle (*Haliaeetus leucocephalus*). The wood stork, an endangered species, has been observed foraging in tidal creeks of the LCP saltmarsh and breeds at several colonies in the vicinity of Brunswick (see Appendix A).

Mammals indigenous to the estuary include various shrews, bats, raccoons (*Procyon lotor*), mink (*Mustela vison*), river otters (*Lutra canadensis*), marsh rice rats (*Oryzomys palustris*), and marsh rabbits (*Sylvilagus palustris*). The West Indian manatee (*Trichechus manatus*), an endangered species, and the Atlantic bottle-nosed dolphin (*Tursiops truncatus*) are mammalian species protected under the Marine Mammal Protection Act that occur in the Brunswick estuary and have been observed in Purvis Creek. Zoodsma (1996b, pers. comm.) reports that West Indian manatees have been observed feeding on smooth cordgrass on the banks of the Turtle River. Other reports indicate that a manatee has been seen near the LCP Site.

3.1.4 Potential Exposure Routes

Exposure routes for water-column and benthic organisms depend on microhabitat and diet. Benthic macroinvertebrates may be exposed to CoPCs via contact with surface water or sediments, direct or ancillary ingestion of sediment (e.g., direct contact by detritus feeders and incidental ingestion by predatory species), and ingestion of prey. Fish and other water-column organisms may be exposed to CoPCs via direct contact with water and sediment and direct ingestion of sediment and food items. Exposure of estuarine wildlife to CoPCs at the LCP Site may involve all of the environmental pathways described above. Routes of terrestrial wildlife exposure for all CoPCs include direct contact with surface water and surface sediment, ingestion of water and sediment, and uptake from food.

3.2 SELECTION OF ECOLOGICAL ENDPOINTS AND RECEPTORS

Assessment endpoints and measurement endpoints for both aquatic and terrestrial ecosystems are discussed along with receptor species in the following sections. Ecological effects in both aquatic and terrestrial ecosystem components may result from direct exposure of organisms to CoPCs in specific media or from transfer through food webs, resulting, in some cases, in biomagnification of CoPCs. Ecological effects may be expressed as a result of effects on organisms and can be categorized as 1) acute mortality of individuals and 2) chronic sublethal effects, such as reduced growth and reproductive inhibition.

3.2.1 Ecological Endpoints

3.2.1.1 Assessment Endpoints

Assessment endpoints are defined as explicit expressions of the environmental values that are to be protected (U.S. EPA 1996). Eleven basic assessment endpoints were identified for the estuary at the LCP Site (Sprenger et al. 1997) as follows:

1. Maintenance of ecological health of the salt marsh community, specifically in terms of the structure and function

2. Protection of long-term health and reproductive capacity of aquatic reptiles utilizing the marsh and Purvis Creek

3. Protection of long-term health and reproductive capacity of omnivorous mammal species utilizing the marsh

4. Protection of long-term health and reproductive capacity of piscivorous mammal species (either marine or terrestrial mammals) utilizing the system

5. Protection of long-term health and reproductive capacity of avian species utilizing the marsh and Purvis Creek

6. Protection of long-term health and reproductive capacity of fishery resources that utilize the system

7. Protection of the fish nursery function of the marsh system

8. Protection of individual threatened or endangered sea turtles that feed in the marsh and/or adjacent areas

9. Protection of individual wood storks that feed in the marsh and/or adjacent areas

10. Protection of individual West Indian manatees that feed in the marsh and/or adjacent areas

11. Protection of individual shortnose sturgeons that feed in the marsh and/or adjacent areas.

Assessment endpoints were selected to focus primarily on upper trophic level species in recognition that some CoPCs at the Site (mercury and Aroclor® 1268) are bioaccumulative and have the potential to elicit adverse effects on ecologically significant life-history traits, such as survival and fecundity, in these species. Terrestrial and aquatic higher trophic level receptors may be exposed to these CoPCs through ingestion of prey that have accumulated the chemicals in their tissues, or through incidental ingestion of contaminated sediment. The specific risk question for higher trophic level species is whether levels of CoPCs in site media and biota are sufficient to result in an exposure that could lead to potential adverse effects on receptor populations, or individuals in the case of threatened and endangered species.

This list of assessment endpoints distinguishes between the need to protect individuals of threatened or endangered species and higher organizational levels of other biota. For example, four of the assessment endpoints describe the same standard of protection for four threatened or endangered species and differ only in the name of the species that the endpoints are designed to protect. In addition, five other assessment endpoints

independently describe the same basic standard of protection for aquatic reptiles, birds, omnivorous and piscivorous mammals, and fishery resources. For clarity in comparing risk assessment conclusions in this baseline ERA with the risk assessment conclusions reported by Sprenger et al. (1997) in support of removal actions at the marsh, their 11 basic assessment endpoints have been retained in this evaluation.

3.2.1.2 Measurement Endpoints

Measurement endpoints are defined as measurable ecological characteristics (typically arithmetic or statistical summaries) that are related to assessment endpoints (U.S. EPA 1996). For the estuarine investigation at the LCP Site, measurement endpoints have been selected to reflect ecological receptors likely to be exposed to CoPCs at the Site and relevant toxic effects of major CoPCs. Consequently, emphasis has been placed on measurement endpoints that address sublethal toxic effects on pertinent life stages (i.e., embryos, larvae, and/or young) of biota (in particular, crustaceans and vulnerable wildlife species) indigenous to the LCP Site.

For assessment endpoint 1 (ecological health of the salt marsh community), measurement endpoints are CoPC body burdens in benthic and water-column organisms; chronic toxicity tests with surface water and representative laboratory-reared, water-column organisms (mysids and sheepshead minnows); chronic toxicity tests with surface sediment and representative benthic organisms (*Leptocheirus plumulosus*); and macrobenthic community characteristics. Long-term health and reproductive capacity of aquatic reptiles (assessment endpoint 2) and threatened and endangered marine turtles (assessment endpoint 8) are evaluated using body burdens and histopathology biomarkers in diamondback terrapins, and food-web accumulation models as measurement endpoints. Long-term health and reproductive capacity of omnivorous and piscivorous mammals (assessment endpoints 3 and 4, respectively) and individual manatees (assessment endpoint 10) are evaluated using food-web accumulation models. Protection of health and reproductive capacity of avian species (assessment endpoint 5) and individual wood storks (assessment endpoint 9) are evaluated using body burdens and histopathology biomarkers in clapper rails and food-web accumulation models as measurement endpoints. Assessment endpoints for protected fish populations (assessment endpoints 6 and 7) and shortnose sturgeons (assessment endpoint 11) are evaluated using laboratory toxicity testing with various species of aquatic organisms and surface water from the Site, as well as body-burden evaluations, as the measurement endpoints.

To evaluate chronic ecological risks, this assessment uses a weight-of-evidence approach to evaluate these measurement endpoints. In this approach, bulk media concentrations receive the lowest weight, because effects are generic and even with site-specific data, effects cannot be reliably attributed to specific compounds. Body burdens relative to effects levels are also assigned a low weight, because ecologically meaningful population-level effects cannot be derived from this line of evidence. Food-web exposure models are considerably more useful because dose estimates can be compared to site-specific findings,

and are thus assigned the next-to-highest weight in the evaluation scheme for this ERA. Because toxicity test findings and benthic invertebrate community structure provide direct evidence of exposure and quantitative measures of effects, they receive the highest weight in supporting chronic risk conclusions.

3.2.2 Ecological Receptors

Aquatic receptors relevant to this ERA are the water-column and benthic receptors present in Purvis Creek, the tidal channels and drainage creeks connected to Purvis Creek, and the Turtle River. Water-column organisms are represented by two species of invertebrates (i.e., American oysters and mysids) and one species of fish (i.e., sheepshead minnow). These species were selected to represent the major taxonomic groups of water-column organisms present in the estuary and because they are routinely used in toxicity testing of aquatic media. Sheepshead minnows are also used as a surrogate for the shortnose sturgeon, an endangered fish species. Benthic organisms selected as receptors were macrofaunal assemblages and one amphipod species (*Leptocheirus plumulosus*).

Wildlife receptors selected for the ERA were West Indian manatee, river otter, raccoon, wood stork, clapper rail, marsh wren, and diamondback terrapin. Species were selected because they are common salt marsh inhabitants (i.e., raccoon, clapper rail, marsh wren, diamondback terrapin), because they are upper-trophic-level species most likely to be indicators of biomagnification of CoPCs (river otter, wood stork, diamondback terrapin), because they are federally listed threatened or endangered species (wood stork, West Indian manatee), or because they are surrogates for federally listed species (diamondback terrapin as a surrogate receptor for Kemp's ridley turtle, loggerhead turtle, and green turtle).

3.3 EXPERIMENTAL PLAN

The following section outlines the objectives, study design, and assessment approach followed in this ERA.

3.3.1 Objectives

The fundamental objectives of the studies reported in this ERA for the estuarine operable unit at the LCP Site are to:

- Determine the distribution of CoPCs throughout the estuary
- Quantify the extent of potential hazard of CoPCs to ecological receptors (aquatic life and wildlife) in the estuary

- Provide a basis for determining the need for appropriate remedial alternatives for the LCP Site.

3.3.2 Study Design

The study design for the ecological investigation at the estuarine operable unit was outlined in the draft RI/FS work plan (PTI and CDR 1996c). The draft RI/FS work plan was organized in a phased approach to allow data collected early in the investigation to be used to define later sampling methods (e.g., level of replication) and to refine the initial layout of sampling stations. The draft work plan to conduct the RI/FS and ecological assessment for the marsh (estuarine) operable unit was provided to EPA, the Georgia Department of Natural Resources (GaDNR), the National Oceanic and Atmospheric Administration, and the U.S. Fish and Wildlife Service in March 1996. The draft work plan was developed as quickly as possible to provide input to both EPA's removal and remedial program site activities. Proposed field efforts were designed to provide data to evaluate alternatives and select a long-term remedy and to provide contaminant distribution data needed for the design of interim remedial measures in the marsh for EPA's removal program. To meet the accelerated schedule of the removal program, which includes cleanup and closure of many upland portions of the LCP Site, AlliedSignal initiated several field studies described in the draft work plan prior to receipt of written comments from EPA or work plan approval. Otherwise, spring and summer biological data collection would have been delayed until 1997, and sediment data needed to support the site project objectives would not have been available when requested by EPA. This rapid initiation of field data collection was consistent with EPA's use of an accelerated cleanup process for the LCP Site. A description of the ecological investigations conducted at the marsh operable unit in 1996 and major modifications from the proposed SOW are provided in Section 4 *Methods*.

3.3.3 Assessment Approach

The ecological assessment of the marsh operable unit at the LCP Site included the following five investigations conducted during 1996:

- Surface water investigation
- Sediment investigation
- Mercury methylation investigation
- Wildlife food-web investigation
- Threatened and endangered species investigation.

The components and objectives of each of these five investigations are presented in the following sections.

3.3.3.1 Surface Water Investigation

The surface water investigation consisted of three basic studies: 1) a water chemistry study, 2) a laboratory toxicity study, and 3) a field bioaccumulation study. The objectives of these studies were to:

- Identify water-borne CoPCs in surface waters within the study area (i.e., target analyte list and target compound list chemicals)
- Evaluate bioaccumulation potential and toxicity to aquatic organisms directly exposed to water-borne CoPCs
- Provide data to determine aquatic transport pathways
- Develop data for scoping and evaluating remedial approaches.

3.3.3.2 Sediment Investigation

The sediment investigation consisted of five studies: 1) a channel sediment chemistry study, 2) a marsh sediment chemistry study, 3) a PCB congener study, 4) a laboratory toxicity study, and 5) a benthic macroinvertebrate study. The objectives of these studies were to:

- Identify CoPCs in sediments within the study area (an objective complementary to the identification of water-borne CoPCs in the surface water investigation)
- Characterize the horizontal and vertical distribution of CoPCs in the study area (including reference areas)
- Evaluate the effects of exposure of benthic organisms to sediment-borne CoPCs
- Develop data for scoping and evaluating remedial approaches.

The approach and results of the laboratory toxicity study, which was used to evaluate effects to benthic organisms areas of the marsh subject to removal actions, are presented in Appendix B.

3.3.3.3 Mercury Methylation Investigation

The mercury methylation investigation consisted of collecting surface water samples at selected marsh drainage sites during two field events: January and June 1996. Samples were analyzed for total mercury and methylmercury. The objectives of the investigation were to:

- Characterize the concentrations of total mercury and methylmercury in the water and particles onsite and offsite to identify potentially impacted and non-impacted drainages
- Determine if the drainages (marsh sediment and channels) are sources or sinks for total mercury and methylmercury
- Characterize temporal patterns in total mercury and methylmercury concentrations that may reveal transport patterns.

Results of the mercury methylation investigation are presented in Appendix C.

3.3.3.4 Wildlife Food-Web Investigation

Samples of dominant prey species in the aquatic and terrestrial salt marsh food webs were collected from various locations in the LCP Site marsh and from the Jointer Creek and Clubbs Creek reference areas. The objective of this investigation was to provide baseline data on CoPC concentrations in prey species that can be used in food-web exposure models to evaluate potential risk to the various wildlife receptors through their diet.

3.3.3.5 Threatened and Endangered Species Investigation

Field studies conducted during 1996 that were pertinent to the threatened and endangered species investigation were an aerial survey of wading bird abundance and distribution in the marsh at the LCP Site and a reference area at Hawkins Creek and a preliminary evaluation of diamondback terrapin reproduction. The objective of the wading bird aerial survey was to qualitatively evaluate use of the LCP Site marsh by foraging wading birds, particularly wood storks. The objective of the diamondback terrapin study was to use site-specific data on reproductive success to develop toxicity reference values (TRVs) for terrapins that could be used in a food-web exposure model to evaluate potential risk to endangered sea turtles.

4. METHODS

Descriptions of sampling methods were provided in the draft field sampling plan (PTI and CDR 1996b) and corresponding quality assurance project plan (QAPP; PTI and CDR 1996a). The draft RI/FS work plan (PTI and CDR 1996c) presented a list of the media to be sampled, station locations, samples per station, and analyses performed. Changes to the schedule for conducting the RI/FS investigations resulted in substantial modifications being made to the proposed SOW. These modifications are discussed in the following section. Brief descriptions of sampling methods for all investigations conducted in 1996 are also provided in subsequent sections.

4.1 MODIFICATIONS TO PROPOSED SCOPE OF WORK

A major modification of the initially proposed SOW for the surface water, sediment, mercury methylation, and wildlife food-web investigations was that reference stations were relocated from Dunbar Creek and the Back River to Jointer Creek and Clubbs Creek, as recommended by EPA's onsite coordinators and GaDNR. This change was made in response to concerns raised by GaDNR regarding representativeness of the sites initially selected and sources of contamination that could affect the sites. The rationale for selecting the Jointer Creek location was based, in part, on its previous use as a shellfish hatchery and, consequently, its absence of chemical contamination.

The SOW for the surface water investigation was modified by the addition of a sampling station in Gibson Creek, bringing the total number of stations evaluated (including the two relocated reference stations) from 10 to 11 (Figure 4-1). The field bioaccumulation study was expanded to include the use of native American oysters in addition to the originally planned hatchery-produced oysters. Hatchery oysters were deployed in the study area at three different times, rather than once as initially planned. Maximum field exposure time for transplanted hatchery oysters was extended from 30 to 90 days.

For the sediment investigation, the SOW for the channel sediment chemistry study was modified by the addition of a sampling station in Gibson Creek, three sampling stations in tributaries east of Purvis Creek (Figure 4-1), and seven onsite channel stations for analysis of lead, total mercury, and total PCBs in surface sediment (Figure 4-2). The scope of the marsh sediment chemistry study was modified based on data provided by EPA (Sprenger and Finley 1996) to focus field sampling in the area of the marsh south of the causeway and immediately west of the upland portion of the Site (Figure 4-3), instead of at stations distributed throughout the marsh. Prior EPA sampling indicated that CoPC concentrations varied widely in this part of the marsh, and some locations had very elevated concentrations. Therefore, it was considered warranted to sample intensively here to define spatial variation in CoPC concentrations to fully characterize the extent of potential

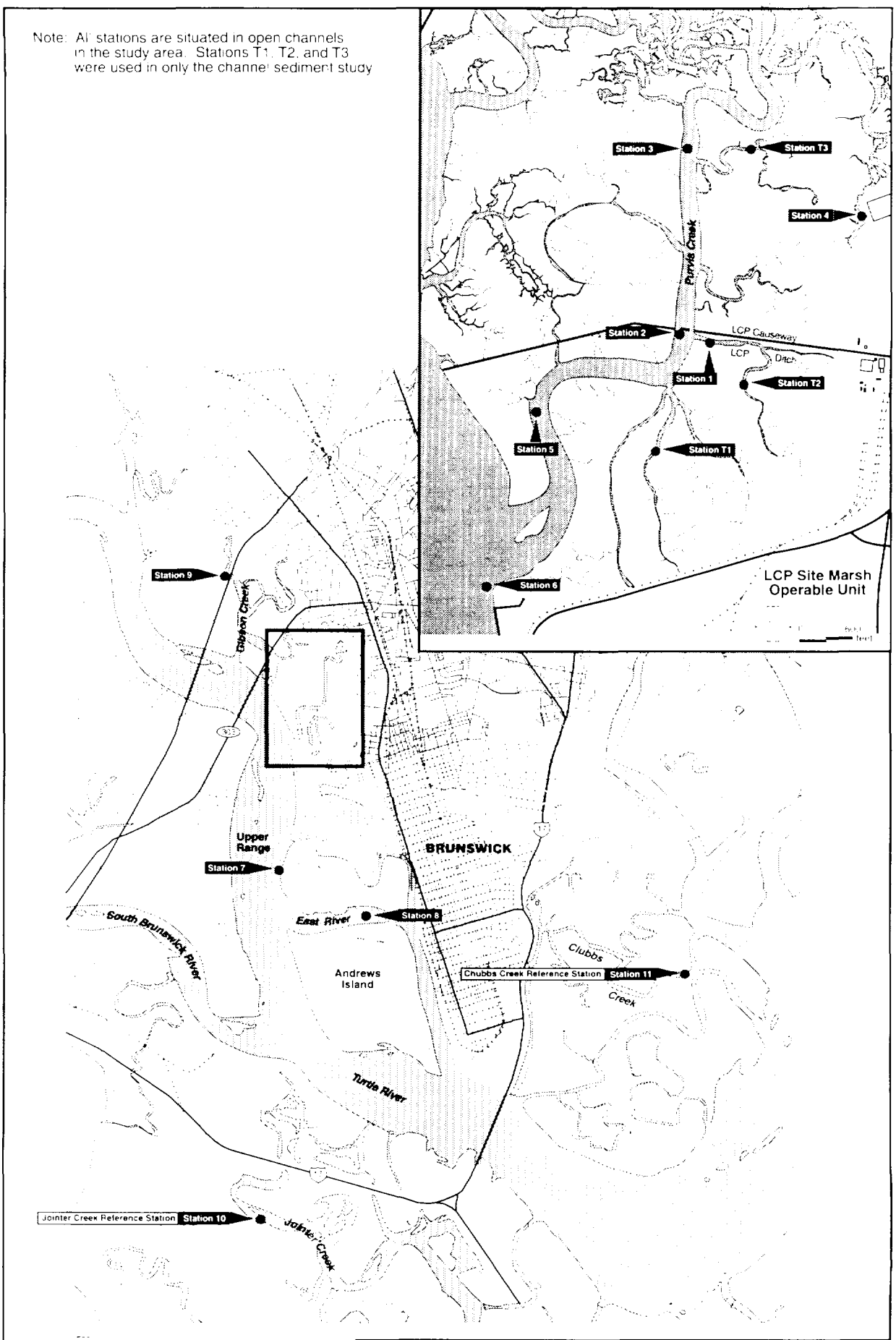
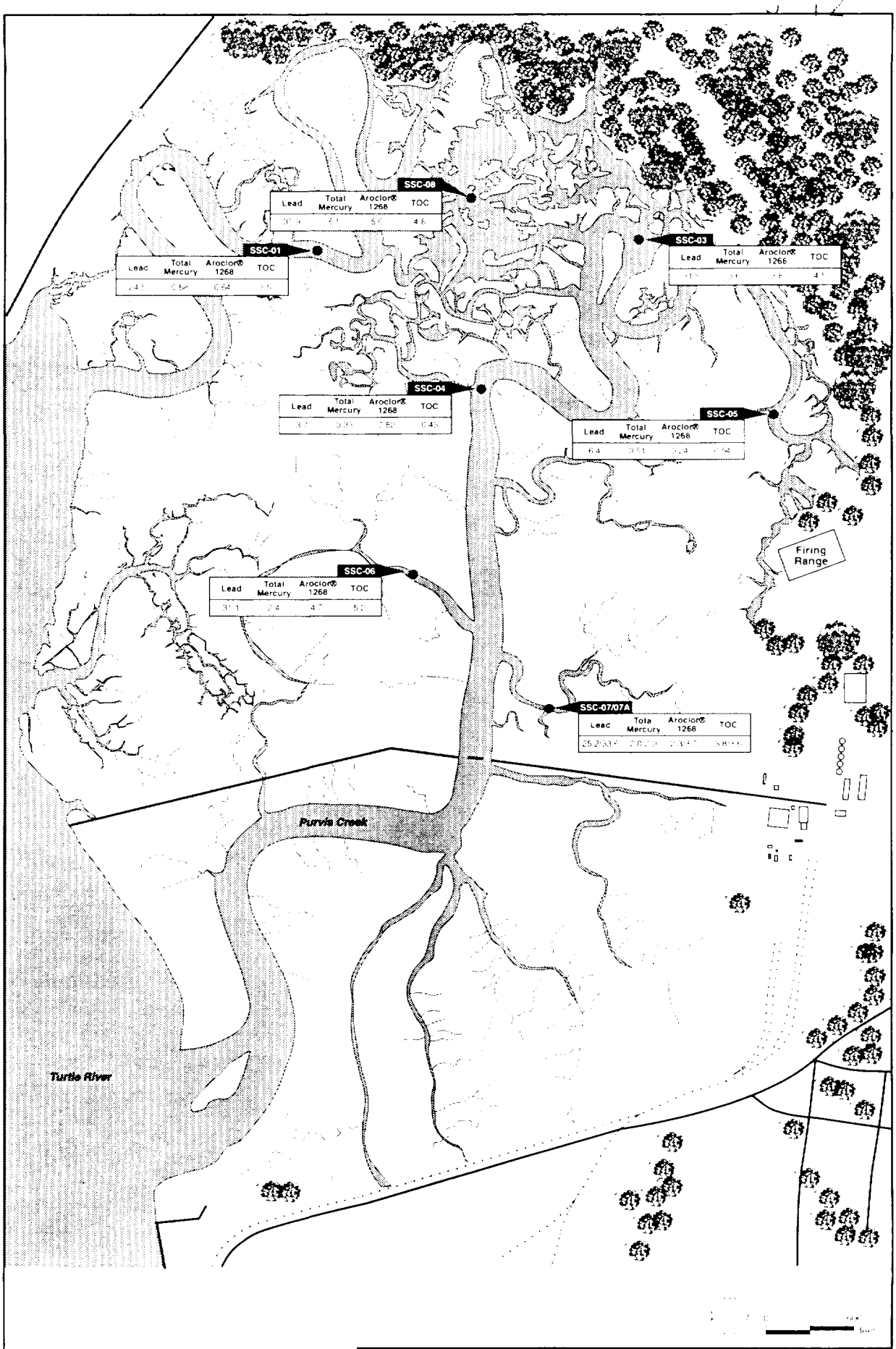


Figure 4-1. Location of sampling stations used in the surface water investigation (water chemistry study, laboratory toxicity study, and field bioaccumulation study) and the sediment investigation (channel sediment chemistry study only) for the LCP Site and reference stations



Note: Concentrations in mg/kg dry weight

Figure 4-2. Concentrations of lead, total mercury, and Aroclor® 1268 and percent total organic carbon (TOC) in surface channel sediment from the LCP Site

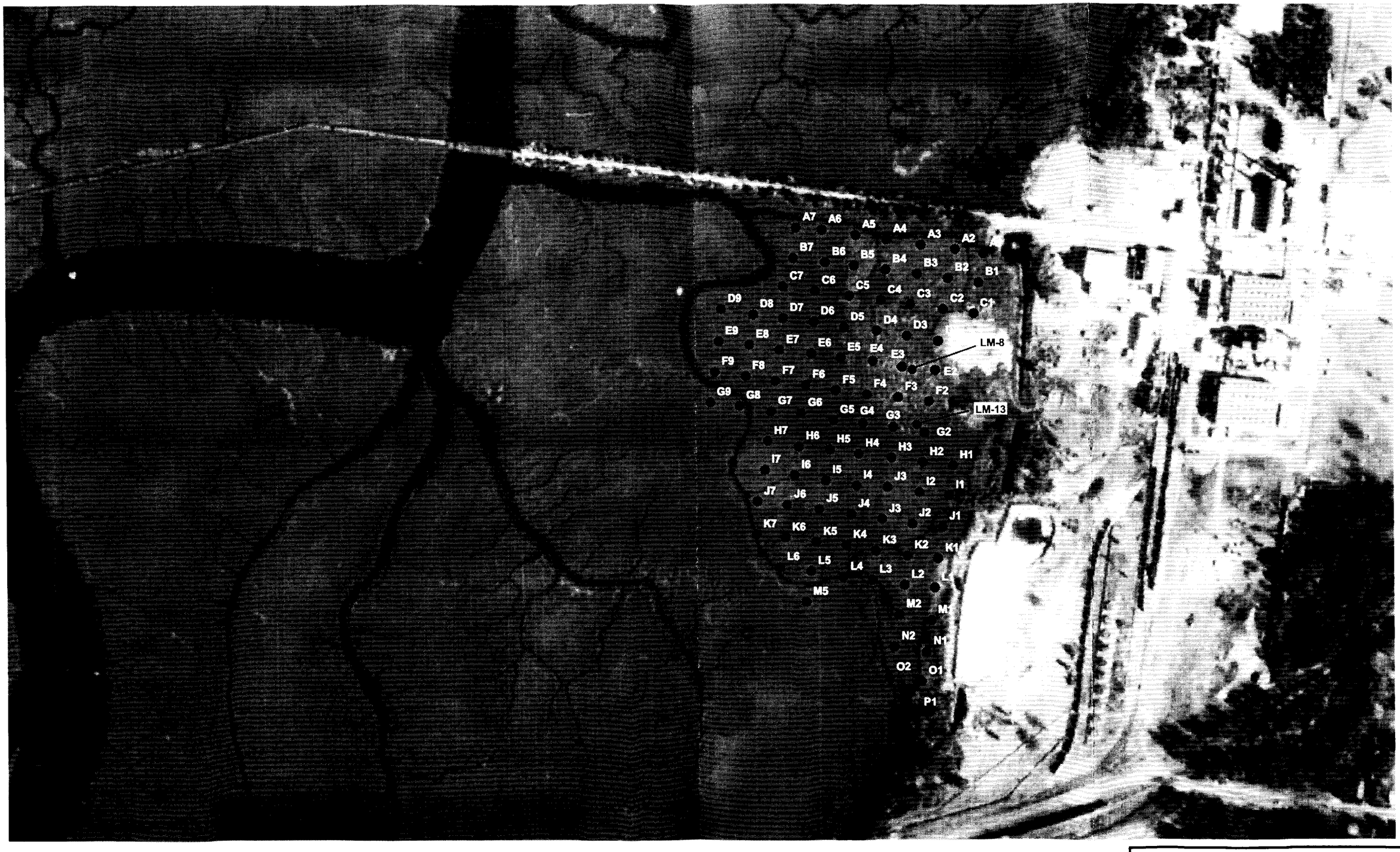


Figure 4-3. Locations of marsh grid sediment sampling stations and two stations (LM-8 and LM-13) used in the sediment toxicity dilution series at the LCP site.

ecological risk and to assist in design of the removal action. Elsewhere in the marsh, concentrations of mercury and PCBs were lower and less variable (typically between 0–10 milligrams per kilogram [mg/kg]). Therefore, less intensive sampling was considered appropriate to confirm this more homogeneous, broad-scale pattern of CoPC distribution.

Characterization of the macrobenthic community at channel stations and censuses of fiddler crabs and periwinkles at marsh stations, which were proposed in the draft work plan as components of a triad investigation, were not performed, with the exception of a preliminary evaluation of macrobenthos. The censuses were not undertaken because of EPA's concern that the distribution of organisms was inherently too patchy to reasonably evaluate. The fiddler crab body-burden study was not conducted because of difficulties in obtaining sufficient biomass for comprehensive chemical analyses. Although crabs were commonly observed, suitable methods for capturing large numbers of individuals (about 100 crabs of the same sex and species at each of 10 sampling stations) could not be devised.

4.2 SAMPLING DESIGN

Sampling locations and methods for the five investigations are discussed in the following sections. All collected samples were documented and handled in a manner that ensured sample integrity from the time of field collection to the time of laboratory analysis. These procedures required that each sample be 1) placed in an appropriate pre-cleaned container, 2) properly preserved, 3) transported and stored in a secure manner, 4) analyzed within the specified holding time, and 5) properly documented.

4.2.1 Surface Water Investigation

The surface water investigation consisted of three studies performed during the summer of 1996: 1) a water chemistry study, 2) a laboratory toxicity study, and 3) a field bioaccumulation study. Water for the chemistry study (channel surface water) was collected from nine sampling stations in the study area and from two reference stations (Stations 1–11; Figure 4-1). Reference stations were located at Jointer Creek and Clubbs Creek. The primary rationale used for selecting sampling station locations is presented in Table 4-1. Water for the chemistry study was usually collected from a motor boat. However, if the water depth in the channel was too shallow, field personnel would walk to the desired station and collect the sample. Samples bottles were held underwater until filled and then capped.

Water for the mysid and sheepshead minnow toxicity tests was collected from the same 11 stations that were used for the water chemistry study (Figure 4-1), using the same collection methods. These tests were conducted to supplement the more important studies of potential bioaccumulation-related risks to wildlife with at least limited studies of direct toxicity to aquatic life.

TABLE 4-1. RATIONALE FOR SELECTION OF SAMPLING LOCATIONS FOR THE LCP SITE AND REFERENCE STATIONS

Sampling Station	Primary Rationale for Selection
1. LCP Ditch (N. Lat. 31° 11.34'; W. Long. 81° 30.84')	Located at LCP marsh site near former discharge point
2. Middle Purvis Creek (N. Lat. 31° 11.21'; W. Long. 81° 31.01')	Located in middle of LCP marsh site
3. Upper Purvis Creek (N. Lat. 31° 11.51'; W. Long. 81° 31.01')	Located near northern part of LCP marsh site
4. Purvis Creek headwaters (N. Lat. 31° 11.24'; W. Long. 81° 30.57')	Located in the marsh about 0.5 km north of LCP Site near the Glynn County Landfill
5. Lower Purvis Creek (N. Lat. 31° 11.50'; W. Long. 81° 31.27')	Located in southern part of LCP marsh site
6. Turtle River—near mouth of Purvis Creek (N. Lat. 31° 10.79'; W. Long. 81° 31.40')	Located near southern part of LCP marsh site
7. Turtle River—upper range (N. Lat. 31° 09.76'; W. Long. 81° 31.44')	Located about 1.8 km south of Station 6 near Georgia-Pacific Pulp and Paper Company
8. East River (N. Lat. 31° 09.11'; W. Long. 81° 30.35')	Located about 3.0 km (river distance) southeast of Station 7 near Brunswick Academy Creek Wastewater Treatment Plant
9. Gibson Creek (N. Lat. 31° 12.47'; W. Long. 81° 32.14')	Located about 4.0 km (river distance) north of Station 6 near outfall of Georgia Power Company generating facility
10. Jointer Creek (N. Lat. 31° 06.11'; W. Long. 81° 31.61')	Reference station located in a different estuarine system (Satilla River estuary) south of Brunswick estuary
11. Clubbs Creek (N. Lat. 31° 08.54'; W. Long. 81° 26.86')	Reference station located in Brunswick estuary about 17 km southeast (river distance) of LCP marsh site

American oysters were deployed at the same 11 stations that were used for the water chemistry study (Figure 4-1). Hatchery oysters obtained from Chesapeake Bay were deployed at the nine study area and two reference stations on June 13–14, 1996. A second batch of oysters was placed in the LCP Ditch on June 25, shortly after an accidental release to the LCP Ditch of surface water runoff from the onsite treatment facility on June 19. On July 3, a third batch of oysters was deployed at two onsite stations and at the two reference stations. This third oyster deployment occurred because inspection of previous oyster deployments indicated relatively high mortality at all sampling stations. The high mortality is believed to have been caused by the high water temperature and high salinity that occurred in the study area.

All deployments of hatchery oysters were retrieved on September 11–12, 1996, resulting in a maximum field exposure period of 90 days (Deployment 1 oysters). Native oysters also were collected from the nine study area and two reference stations on those dates. In addition, subsamples of Deployment 2 hatchery oysters (originally placed in the LCP Ditch) and native oysters from middle Purvis Creek were transplanted at the Jointer Creek reference station to evaluate depuration of chemicals as a function of time.

4.2.2 Sediment Investigation

The sediment investigation consisted of five studies: 1) a channel sediment chemistry study, 2) a marsh sediment chemistry study, 3) a PCB congener study, 4) a laboratory toxicity study, and 5) a benthic macroinvertebrate study. Sampling design for the laboratory toxicity study is presented in Appendix B.

Prior to sediment sampling for any given study, all sampling equipment was scrubbed with Alconox®; rinsed with site water; solvent-rinsed with acetone (excluding plastic sampling equipment); and then hexane-dried, air-dried, and rinsed with site water.

All sediment samples were placed in precleaned sample jars received from the analytical laboratory. Depending on the type of analysis, either a glass or plastic container was used for sediment samples. Sample containers for other media sampled during this investigation are specified below.

After sediment sample collection, all sample containers were immediately placed on ice in a cooler. Samples were stored and transported at 4°C. Upon return from the field, all samples were transferred to refrigerators or freezers, where samples were maintained at the proper temperature until shipped to the analytical laboratory by overnight courier.

4.2.2.1 Channel Sediment Chemistry Study

Sediment cores for the channel sediment chemistry study were collected from the same 11 sampling stations used in the surface water investigation, plus an additional 3 stations (T1,

T2, and T3) in unnamed tributaries located east of Purvis Creek (Figure 4-1). Samples were collected during two field events from May 14 to May 22, 1996, and from July 13 to July 15, 1996. An additional seven channel stations located north of the LCP causeway (Figure 4-2) were sampled for surface sediments from June 13 to June 17, 1996. Sediment cores were collected using titanium coring tubes (10 cm diameter). Sediment cores were collected to a depth of 50 cm, and each core was divided into 2-cm intervals for the upper 20 cm and into 5-cm intervals for the lower 30 cm. Eight intervals were submitted for analysis (0–2, 2–4, 4–6, 6–8, 8–10, 10–12, 14–16, and 18–20 cm). All other intervals were archived for future analysis, if necessary. At the seven channel stations north of the LCP causeway, a 0.06-m^2, stainless-steel, modified Ponar grab sampler was used to collect surface sediment from the upper 0–5 cm. Each sample was evaluated for acceptability and was accepted if the surface sediment was relatively undisturbed and the planned penetration depth was achieved.

Additional channel sediment samples were collected during other Site investigations performed by EPA and by GeoSyntec Consultants. These data are considered in the data interpretation discussed in Section 5.3.2.4.

4.2.2.2 Marsh Sediment Chemistry Study

Surface sediments for the marsh chemistry study were collected from 95 sampling stations south of the causeway and immediately west of the upland portion of the LCP Site from June 17 to June 21, 1996. This area of the marsh is bordered by the industrialized part of the LCP Site to the east, the LCP causeway to the north, and the first major tidal channel to the west and south. Within this area, a 100 ft × 100 ft sampling grid was established (Figure 4-3). The grid was arranged with a slight deviation from a true north-south orientation to correspond approximately with sampling transects previously established by EPA.

To evaluate the heterogeneity of CoPC distributions over short distances, replicate sediment sampling was performed at two randomly selected stations within the marsh grid (Station H4 and Station J7). At each of these stations, a 20 ft × 20 ft grid was established and subdivided into 2 ft × 2 ft sections to produce 100 compartments (fine-scale grid sampling). Sediment was collected separately from 10 randomly selected compartments for analysis of total mercury and total PCBs (including Aroclor® 1268).

Surface sediments were also collected from June 18 to June 25, 1996, at three marsh stations at each of the Jointer Creek and Clubbs Creek reference areas (Figure 4-1). At both areas, one station was located on the exposed, non-vegetated bank of a tidal creek channel that was comparable in size to the tidal channel that constitutes the southwestern boundary of the sampling grid at the LCP Site. The other two stations were located at distances of approximately 100 and 300 ft from the tidal channel.

Because the cordgrass root mat impaired penetration by the coring tubes or Ponar grab sampler, sediment samples were collected with stainless-steel spoons. Surface sediment was collected from the upper 0–5 cm. Each sample was evaluated for acceptability and was accepted if the surface sediment was relatively undisturbed.

Sediment core samples were collected from three locations in the marsh (Stations SCM-01, SCM-02, SCM-03) and from one location at each of the reference areas (Figure 4-4) from May 14 to May 18, 1996, using collection methods previously described for the channel sediment chemistry study.

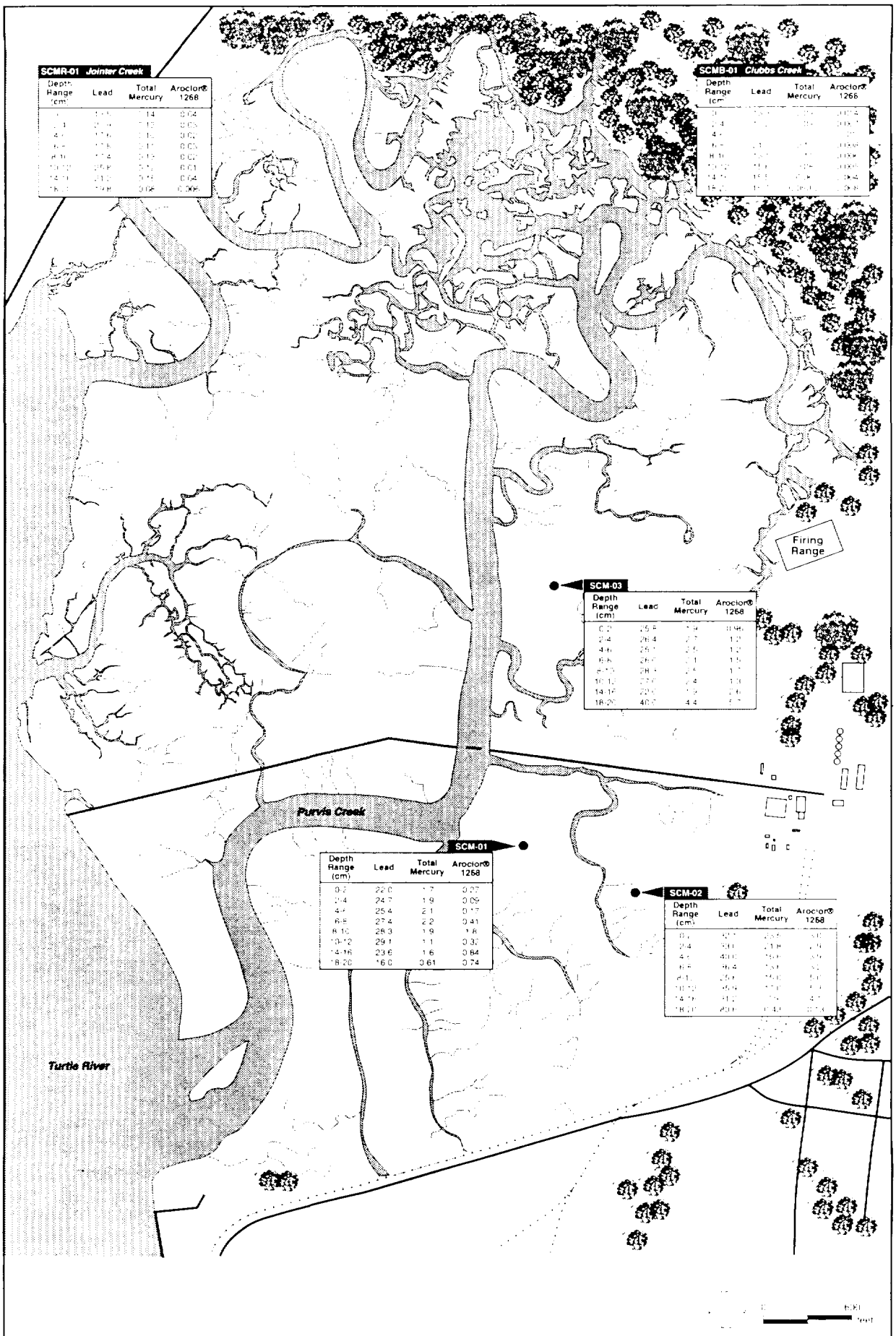
Additional marsh sediment samples were collected during other Site investigations performed by EPA and by GeoSyntec Consultants. These data are considered in the data interpretation discussed in Section 5.3.2.4.

4.2.2.3 PCB Congener Study

As part of the preliminary field sampling event in January 1996, PCB congener analysis was performed on four sediment and three tissue samples. The objective of this study was to provide a preliminary characterization of PCB congener composition in abiotic and biotic media at the LCP Site. Two sediment samples were collected from Stations 7 and 8 at creek Transect CRTB, and two additional sediment samples were collected at distinct areas in the marsh (Stations CONA and CONB; Figure 4-5). Two tissue samples, a vegetation sample (smooth cordgrass), and an invertebrate sample (periwinkle) were collected in the vicinity of Station CONA. A second invertebrate sample (a shrimp species) was collected in the LCP Ditch. Taxonomic identification of the shrimp species was not done. However, based on the small body size of the organism, it was likely a grass shrimp.

Sediment samples were collected using either a 10-cm-diameter, stainless-steel coring tube or stainless-steel spoons. If the cordgrass root mat impaired penetration by the coring tube, sediment samples were collected with stainless-steel spoons. Surface sediment was collected from the upper 0–5 cm. Each sediment sample was evaluated for acceptability and was accepted if the surface sediment was relatively undisturbed.

Smooth cordgrass was collected by cutting the plant stem just above the sediment interface and placing the sample in an uncontaminated Ziploc® bag. Periwinkles (approximately 80 individuals) were collected by hand from the cordgrass stems and also placed in an uncontaminated Ziploc® bag. The shrimp species was collected using a dip net in a small pool of water during low tide in the center of the LCP Ditch channel (10 individuals). This second invertebrate sample was placed in a precleaned glass sample jar.



Note: Concentrations in mg/kg dry weight.

Figure 4-4 Vertical chemical profiles of marsh sediment at the LCP Site and reference stations.

Figure 4-5. Locations of January sediment sampling at the LCP Site.

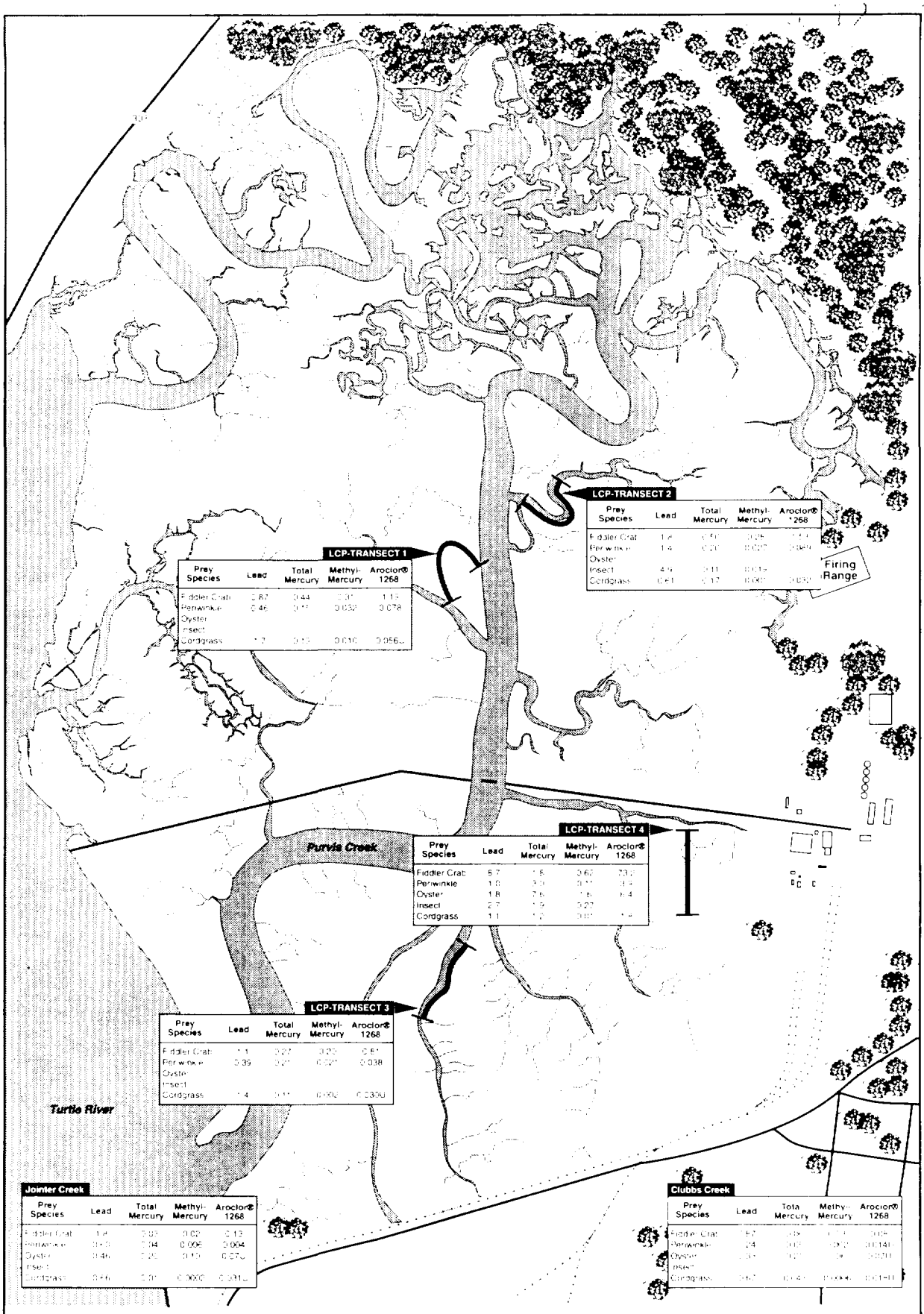
4.2.2.4 Field Macroinvertebrate Study

Field macroinvertebrate assemblages were evaluated at two stations at the LCP Site and two stations at the Jointer Creek reference area in January 1996. One station was located in mid-channel, and the other station was located on the channel bank at each of the sampling areas. Sediment was collected using 10-cm-diameter Plexiglas® core tubes. Five sample replicates were collected at each station. Sediment was collected to a depth of 15 cm. Using gentle streams of water, the sediment was washed through a 1.0-mm screen and then a 0.5-mm screen. The invertebrates were collected from the screen, placed in precleaned sample bottles, and preserved with formalin.

4.2.3 Wildlife Food-Web Investigation

Prey tissue sampling was conducted at four transects in the LCP Site marsh, at one transect in the Jointer Creek reference area, and at one transect in the Clubbs Creek reference area (Figure 4-6) from June 25 to June 28, 1996. Transect locations at the LCP Site were selected to characterize body burdens in individuals of prey species inhabiting regions at differing distances from the primary sources of CoPCs and regions where different transport mechanisms were expected to influence CoPC distribution. As such, locations were considered to be representative of the specific geographic position of the transect as well as representative of the larger geographic area within which they occurred. One transect (Transect 4) was located in the area of the marsh subject to removal actions. Prior sampling had indicated that sediment CoPC concentrations were most elevated in this region of the marsh, and elevated body burdens were anticipated in resident prey species. One transect (Transect 3) was located alongside one of the tidal channels south of the LCP causeway in a portion of the marsh outside the immediate area influenced by activities on the adjacent upland part of the Site. This transect was selected to reflect body burdens in prey species from most of the marsh south of the causeway that was not subject to removal actions. Two transects (Transects 1 and 2) were established north of the LCP causeway in areas where tidal movement was believed to be the most likely mechanism influencing dispersion of CoPCs. These latter two transects were located on either side of Purvis Creek to determine if distribution of CoPCs was influenced by the creek.

Several prey species were collected along each transect. Large fiddler crabs were captured by hand and placed in clean Ziploc® bags. Periwinkles were collected by hand from the cordgrass stems and placed in clean Ziploc® bags. Oysters were also collected by hand and placed in large plastic bags. The whole oyster, including shell, was sent to the analytical laboratory. Terrestrial insects were collected using sweep nets and were placed in clean Ziploc® bags. Smooth cordgrass stems were cut just above the sediment interface and placed in clean Ziploc® bags.



Note Cordgrass concentrations from Transect 4 may be elevated due to affected sediment. All values are expressed as mg/kg dry weight.

U = Analyte undetected at detection limit shown.

Figure 4-6. Concentrations of lead, total mercury, methylmercury, and Aroclor® 1268 in prey species from the LCP Site and reference stations

4.2.4 Threatened and Endangered Species Investigation

Field studies of threatened and endangered species conducted during 1996 were aerial surveys of wading bird abundance and distribution in the marsh at the LCP Site and a reference site at Hawkins Creek and an evaluation of chemical concentrations in diamondback terrapin eggs and hatchlings. The wading bird aerial surveys were performed by Larry Bryan, coordinator of the wood stork program at the Savannah River Ecology Laboratory, Aiken, SC. Complete details of survey locations and census methods are provided in Appendix A.

The diamondback terrapin study was restricted by limited success in locating nesting females or recently laid clutches. One clutch of nine eggs (Clutch A) was collected on May 24, 1996, from a natural, intact nest at a reference area. One gravid female was caught by hand on May 27, 1996, at a reference area on Cedar Creek. A second gravid female was caught by hand on May 29, 1996, from the marsh at the LCP Site. Eggs were obtained from both gravid females by injection of synthetic oxytocin. Clutch D (five eggs) was obtained from the female collected at the reference site, and Clutch E (five eggs) was obtained from the female collected at the LCP Site. The eggs from all three clutches were artificially incubated in the laboratory. Two additional clutches (Clutches B and C, seven eggs each) were collected on May 25 and May 27, 1996, from road-killed females that were not in the reference site. Viability of these eggs may have been affected by the interval between death of the female and collection of the carcass; therefore, these eggs were not included in the analysis. Because of small sample sizes, results of the terrapin study are presented only as empirical observations.

4.3 ANALYSIS METHODS

All samples for the LCP investigation were analyzed using EPA methods (e.g., SW-846 [U.S. EPA 1994c]) or methods approved or recommended by EPA, when available. These methods included all associated quality assurance and quality control (QA/QC) procedures as specified. A summary of methods used during this investigation for chemical analyses and toxicity testing are provided in Table 4-2. All analyses were performed by laboratories that have established protocols and quality assurance procedures that meet or exceed applicable EPA guidelines. Summary descriptions of the analytical procedures are provided in the QAPP (PTI and CDR 1996a).

Analytical methods were selected that usually yield detection limits that are sufficiently low to support interpretive activities for this investigation (e.g., establish conformance with sediment and water quality standards or assess ecological risk). However, the standard target analyte list and target compound list reporting limits for some metals and some organic compounds are greater than associated marine water quality criteria. For example, the detection limit for PCBs in surface water ranged from 0.5 to 1.0 μg/L, compared to Georgia's ambient water quality criterion of 0.014 μg/L for most PCBs. The actual detection limits attained during this site investigation were, in some cases, elevated with

TABLE 4-2. SUMMARY OF ANALYTICAL METHODS AND DATA QUALITY OBJECTIVES FOR SAMPLES COLLECTED AT THE LCP SITE AND REFERENCE STATIONS AS PART OF THIS INVESTIGATION

Analysis	Method Reference	Units	Method Reporting Limit[a,b]	Bias (percent)	Precision (RPD)	Completeness (percent)
Water						
CoPCs						
Mercury (total and dissolved)	EPA 1631 (U.S. EPA 1995c)	ng/L	0.1	75–125	±25	95
Methylmercury (total and dissolved)	Bloom (1989)	ng/L	0.05	65–135	±35	95
Lead (total and dissolved)	EPA 7421 (U.S EPA 1994a)	mg/L	1	75–125	±25	95
PCBs (including Aroclor® 1268)	EPA 8080 (U.S. EPA 1994b)	mg/L	0.2	50–150	±50	95
PCB congeners	EPA 680 (U.S. EPA 1985b)	ng/L	5	25–150[c]	±50	95
PAHs	EPA 8270 with SIM (U.S. EPA 1994b)	mg/L	0.5	50–150	±50	95
TAL/TCL Analyses						
TAL metals (total and dissolved)	EPA 6010 and 6020 (U.S. EPA 1994b)	mg/L	0.2–45	75–125	±25	95
TCL SVOCs	EPA 8270 (U.S. EPA 1994b)	mg/L	10–50	50–150	±50	95
TCL pesticides/PCBs (including Aroclor® 1268)	EPA 8080 (U.S. EPA 1994b)	mg/L	0.05–0.25[d]	50–150	±50	95
Conventional Analyses						
Total suspended solids	EPA 160.2 I (U.S. EPA 1983)	mg/L	2	85–115	±20	95
Total organic carbon	EPA 415.1 (U.S. EPA 1983)	mg/L	1	85–115	±20	95
Dissolved organic carbon	EPA 415.1 (U.S. EPA 1983)	mg/L	1	85–115	±20	95
Water Toxicity Tests						
Mysid - survival, growth, and fecundity	EPA 1007 (U.S. EPA 1988b)	percent survival, growth, egg development	--	--	--	100
Sheepshead minnow - survival and teratogenicity	EPA 1005 (U.S. EPA 1988b)	percent hatched, percent larval survival without debilitating abnormalities	--	--	--	100

TABLE 4-2. (cont.)

Analysis	Method Reference	Units	Method Reporting Limit[a,b]	Bias (percent)	Precision (RPD)	Completeness (percent)
Soil/Sediment						
CoPCs						
Total mercury	Bloom and Fitzgerald (1988)	mg/kg	1	75–125	±35	95
Methylmercury	Bloom (1989)	mg/kg	1	65–135	±35	95
Lead	EPA 7421 (U.S. EPA 1994a)	mg/kg	0.1	75–125	±35	95
PCBs (including Aroclor® 1268)	EPA 8080 (U.S. EPA 1994b)	mg/kg	10[e]	50–150	±50	95
PCB congeners	EPA 680 (U.S. EPA 1985b)	mg/kg	1	25–150[c]	±50	95
PAHs	EPA 8270 with SIM (U.S. EPA 1994b)	mg/kg	10[e]	50–150	±50	95
TAL/TCL Analyses						
TAL metals	EPA 6010 and 6020 (U.S. EPA 1994a)	mg/kg	0.02–4.5	75–125	±35	95
TCL SVOCs	EPA 8270 (U.S. EPA 1994b)	mg/kg	40–200[e]	50–150	±50	95
TCL pesticides/PCBs (including Aroclor® 1268)	EPA 8080 (U.S. EPA 1994b)	mg/kg	2–30[d,e]	50–150	±50	95
Conventional Analytes						
Total organic carbon	Standard Method 5310B (Franson 1992)	percent	0.1	75–125	±35	95
Total sulfide	EPA 9030 (U.S. EPA 1992b)	mg/kg	1	65–135	±35	95
Grain size	PSEP (1986)	percent	0.1	NA	±35	95
Sediment Toxicity Tests						
Amphipod - mortality and growth (*L. plumulosus*)	DeWitt et al. (1992); Emery (in prep.)	percent survival, biomass	--	--		100
Plant and Animal Tissue						
CoPCs						
Total mercury	Bloom and Fitzgerald (1988)	mg/kg	1	75–125	±35	95
Methylmercury	Liang et al. (in press)	mg/kg	0.02	65–135	±35	95
Lead	EPA 7421 (U.S. EPA 1994a)	mg/kg	0.1	75–125	±35	95
PCBs (including Aroclor® 1268)	EPA 8080 (U.S. EPA 1994b)	mg/kg	20	50–150	±50	95

TABLE 4-2. (cont.)

Analysis	Method Reference	Units	Method Reporting Limit[a,b]	Bias (percent)	Precision (RPD)	Completeness (percent)
PCB congeners	EPA 680 (U.S. EPA 1985b)	mg/kg	0.5	25–150c	+50	95
PAHs	EPA 8270 with SIM (U.S. EPA 1985a)	mg/kg	20	50–150	±50	95
TAL/TCL Analyses						
TAL metals (total and dissolved)	EPA 6010, 6020, 7000 series (U.S. EPA 1994a)	mg/kg	0.2–45	75–125	±25	95
TCL SVOCs	EPA 8270 (U.S. EPA 1994b)	mg/kg	60–330	50–150	±50	95
TCL pesticides/PCBs (including Aroclor® 1268)	EPA 8080 (U.S. EPA 1994b)	mg/kg	4–60	50–150	+50	95
Conventional Analyses						
Total lipids	AOAC 17.016	percent	0.1	NA	±35	95

Note:

CoPC	-	chemical of potential concern	SIM	-	selective ion monitoring
NA	-	not applicable	SVOC	-	semivolatile organic compound
PAH	-	polycyclic aromatic hydrocarbon	TAL	-	target analyte list
PCB	-	polychlorinated biphenyl	TCL	-	target compound list
RPD	-	relative percent difference			

[a] For inorganic analytes, the method reporting limit is the instrument detection limit adjusted for sample size and dilution during sample preparation. For organic analytes, the practical quantification limit is given. Quantification limit goals will be within the calibration range of the analysis. Expected detection limits are 2–10 times lower than the quantification limit.

[b] Method reporting limit goals for sediment samples are given on a dry weight basis assuming 50 percent moisture. For tissue samples, method reporting limit goals are given on a wet weight basis.

[c] The control limits for recovery of isotopically labeled internal standards are 25–150 percent. The effective accuracy after recovery correction will be much closer to 100 percent than the actual recovery of the internal standards.

[d] Quantification limit goal for toxaphene is 3–5 times higher than the other pesticides.

[e] Quantification limit goals are based on a larger initial sample weight (typically 50–100 g) and a smaller extract volume than those specified in the SW-846 method (U.S. EPA 1994b).

respect to theoretical detection limits when interferences were encountered because of the sample matrices.

4.4 DATA MANAGEMENT

Computerized systems were used to record, store, and sort the data that were generated for the investigation. The data record included a unique sample code, station ID, sample type (matrix), analyte, analyte concentration, and concentration units. Automated data handling was used to the fullest practical extent to increase data integrity by reducing errors, omissions, and ambiguities that can be introduced by manual procedures. In addition, automated procedures were used by all of the contract laboratories to capture and summarize analytical results. Electronic data files were imported directly from the laboratory to PTI's database, minimizing both data entry effort and opportunities for error.

4.5 QUALITY ASSURANCE AND QUALITY CONTROL

The data generated and analyzed during the 1996 investigation of the marsh operable unit at the LCP Site were subjected to QA/QC reviews. A discussion of these procedures is presented in the following sections.

4.5.1 Field Quality Control

Field quality control samples analyzed with the environmental samples included three types of field blanks (i.e., equipment rinse blanks, bottle blanks, and environmental blanks), replicate field samples, and reference material samples. Equipment blanks were analyzed to determine if contamination may have been introduced during sampling. Bottle blanks were analyzed to determine if contamination may have been introduced from the bottles themselves as a result of improper preparation, handling, or cleaning techniques. Environmental blanks were analyzed to determine if contamination may have been introduced from atmospheric sources during the collection of the samples. Replicate field samples were collected to assess small-scale field variability in analyte concentration in addition to laboratory and sampling variability. Reference material samples were analyzed to provide an additional check on overall laboratory performance.

No target analytes were detected at concentrations above the method reporting limits in the equipment rinsate blanks associated with the sediment samples. Total mercury and methylmercury were detected at low levels in some of the bottle and environmental blanks collected with the water samples, but no sample results required qualification because contaminant levels were very low with respect to analyte levels in the sample.

Field replicate samples were collected at a frequency of 5 percent of environmental samples for all sediment and surface water samples. Replicate samples could not be collected for the mercury methylation study because limitations to the number of samples that could

be collected were imposed by the configuration of the automatic samplers that were used for this study. The results reported for the field duplicates were generally similar.

Reference material samples were submitted for metals, semivolatile organic compounds (SVOCs), pesticides, and PCBs. Reference material samples were not submitted for methylmercury analyses because reference materials are routinely analyzed by the laboratories as laboratory control samples (LCSs) with each sample batch. The concentrations reported for the target analytes in each reference material sample generally fell within the 95 percent confidence interval or within the reporting limits established by the manufacturer of the reference material, with several exceptions. No data were qualified on the basis of results reported for reference material samples.

A complete summary of the results for all field quality control samples is presented in quality assurance review summary presented in Volume II of this report.

4.5.2 Laboratory Quality Assurance

All data were subjected to a quality assurance review and audit to verify that the laboratory QA/QC procedures were completed and documented as required and that the quality of the data was sufficiently high to support its use in the ERA. Data validation procedures and qualifier assignments were completed according to EPA's functional guidelines for evaluating inorganic and organic analyses (U.S. EPA 1994a,b), as applicable, with modifications made as appropriate to accommodate quality control requirements for methods not specifically addressed by the functional guidelines (i.e., modified methods, conventional analyses, PCB congener analyses, and low-level mercury and methylmercury analyses).

Data qualifiers were assigned during the quality assurance reviews if control limits were not met, in accordance with U.S. EPA (1994a,b), quality control requirements stated in the methods, and the data quality objectives established for the project (PTI and CDR 1996a), as applicable. All data qualified as estimated (*J*) have an acceptable degree of uncertainty and represent data of good quality and reasonable confidence (U.S. EPA 1989). Rejected data are considered unusable for their intended purposes. Only four results were rejected for this study, and the rejected values have been removed from the database.

Data validation procedures were completed according to EPA Level 3 requirements (U.S. EPA 1995c). Data validation procedures included reviewing the laboratory case narratives and the results of analytical quality control measurements for each sample delivery group. Results for all method blanks, initial calibration blanks, and continuing calibration blanks were reviewed to evaluate laboratory contamination and instrument baseline drift. To determine that all instruments were calibrated correctly, results were reviewed for instrument tuning, initial calibrations, and continuing calibration verifications. Analytical accuracy was quantified as the recovery of applicable matrix spikes, LCSs, and surrogate compounds and by the acceptability of internal standard responses. Analytical

precision was quantified as the relative percent difference between duplicate samples, matrix spikes, or LCS analyses, as applicable. In addition, a 100-percent comparison was made of data reported in the laboratory data packages with electronic data imported or hand-entered into the PTI database, and all discrepancies were resolved. Validation procedures generally did not include verification of laboratory calculations and transcriptions or review of instrument printouts and bench sheets, except for the first data package from each laboratory and additionally, as needed, to explain or verify unexpected QA/QC results or sample data. Details regarding the data validation procedures used to assess data quality are provided in the QAPP (PTI and CDR 1996a). Results of the quality assurance review summaries for the chemical and toxicity test data are presented in Volume II of this report.

The data for the 1996 sampling effort were generally of good quality. Only one result (for one grain size fraction) was of unacceptable quality and was rejected (i.e., labeled with an *R* in the data package and removed from the database). Three results for total solids for the food-web study were additionally rejected as outliers. Overall, 9.4 percent of all results were qualified as estimated (*J*) because control limits were exceeded for one or more laboratory quality control samples or procedures. These qualified data may have a larger associated bias or may be less precise than unqualified data. However, the qualified data are of acceptable quality for use in the ERA.

4.6 DATA ANALYSIS

4.6.1 Chemical Analyses

To evaluate the distribution of CoPCs in the marsh grid area, detailed maps were generated depicting the horizontal distribution of chemicals in the surface sediment. In the field, coordinates from the 100 × 100 ft sampling grid were read into Arc/INFO to establish the base file for creating a map of the grid. Chemical data were added to Arc/INFO using a spreadsheet file generated from the project database. The chemical concentrations were plotted on a base map that shows an aerial photo depicting the physical features of the marsh. Data collected by the EPA Environmental Response Team during 1995 were mapped in a similar manner to permit direct visual comparison of results for the two studies.

4.6.2 Wildlife Food-Web Exposure Models

Exposure of wildlife to CoPCs at the LCP Site marsh was evaluated using a food-web modeling approach. Food-web exposure models for the receptor species of West Indian manatee, river otter, raccoon, wood stork, clapper rail, marsh wren, and diamondback terrapin were used to provide a basis for interpreting projected doses of CoPCs received through the diet. Food-web exposure models estimate exposure as a total daily dose for each receptor species. Because most TRVs are typically reported as the threshold daily

dose to an individual, estimation of a site-specific dose allows direct comparison of exposure estimates to TRVs.

The general structure of the model to estimate average daily ingestion of a given chemical by a wildlife receptor is as follows:

$$IR_{ingestion} = \sum_h (T_h)(IR_h) = \left[\sum_h (T_h)\right]\left[\frac{\sum_i (C_{ih} \times M_i \times A_i \times FR)}{BW}\right]$$

where:

$IR_{ingestion}$ = species-specific total rate of chemical intake by ingestion (mg/kg-day dry weight)

T_h = fraction of year that species occurs in the geographic region of habitat h

IR_h = species-specific rate of chemical intake by ingestion in habitat h (mg/kg-day dry weight)

C_{ih} = concentration of the chemical in medium i of habitat h (mg/kg dry weight for solids, milligrams per liter [mg/L] for water)

M_i = rate of ingestion of medium i (kg/day dry weight for solids, L/day for water)

A_i = relative gastrointestinal absorption efficiency for the chemical in medium i (proportion)

FR = proportion of LCP Site area relative to receptor foraging range

BW = body weight of receptor species (kg).

Time use factors (T_h in the equation) and area use factors (FR in the equation) are included to account for two different aspects of a species' biology. Inclusion of a time use factor is appropriate for migratory species that do not occur in the geographic region of a site on a permanent, year-round basis. An area use factor is incorporated into the model to account for differences between the area of the site and the foraging range of a receptor. Fractional area use factors (i.e., values less than 1) are applied for species with foraging ranges larger than the size of the site. A combination of time use and area use factors can be applied for a species that is migratory, and that forages over a range larger than the site during its period of residency in the geographic region of the site.

For all food-web exposure models, A_i was considered to equal 1.0.

Site-specific data on chemical concentrations in abiotic media and key food species of the receptor of concern are incorporated directly into the models to estimate chemical intake.

5. EXPOSURE ASSESSMENT

Potential exposure of aquatic and terrestrial ecological receptors to CoPCs at the LCP Site is evaluated in the following sections. The estuary and naturally occurring physical and chemical stressors are characterized. Site-specific chemistry data are presented to illustrate the spatial distribution of CoPCs in environmental media and prey species at the Site. Food-web models are developed to estimate exposure of wildlife receptors to CoPCs. Chemical characteristics and life history characteristics of receptor species that modify potential exposure also are evaluated. Previous investigations of body burdens in clapper rails and diamondback terrapins at the LCP Site are summarized. Finally, the aerial surveys of wading bird abundance are discussed.

5.1 ECOSYSTEM CHARACTERIZATION

The estuary at the LCP Site is bisected by Purvis Creek, a salt water, tidally influenced system with a tidal fluctuation of about 6.5 ft. Purvis Creek has a maximum width of approximately 500 ft and a maximum depth approximately 16 ft and is about 2.5 miles in length. Tributaries to Purvis Creek occur throughout the marsh, forming a complex and extensive hydrological system. Purvis Creek discharges into Turtle River, a larger salt water, tidally influenced system with a maximum width of about 3,900 ft and depth of about 39 ft.

The marsh at the Site is bisected by an earthen causeway running from the industrialized part of the LCP Site to Purvis Creek. The causeway separates the northern portion of the marsh from the southern portion, although the two areas are indirectly connected by tidal cycling of Purvis Creek. A drainage ditch (LCP Ditch) running along the southern edge of the causeway connected a plant outfall with Purvis Creek.

Emergent vegetation on the marsh consists almost entirely of smooth cordgrass, which occurs in a taller form along the banks of tidal creeks and in a lower form on higher portions of the marsh. At the highest portions of the marsh near the upland areas where flooding does not occur during every tide, cordgrass is replaced in some locations by patches of black needlerush. For coastal marshes in Georgia, it is believed that black needlerush stands develop in shoreward areas where irregular flooding and rainfall runoff from adjacent land lower interstitial salinity (Wiegert and Freeman 1990).

Atlantic coast salt marshes support a diverse assemblage of aquatic and terrestrial wildlife species (Wiegert and Freeman 1990). Species either actually or potentially occurring at the LCP Site are described in Section 3.1.3 *Potential Ecological Receptors*.

5.2 STRESSOR CHARACTERIZATION

As defined by U.S. EPA (1992a), potential stressors can include physical and chemical factors of both natural and anthropogenic origin that can have adverse effects on the ecosystem being evaluated. Natural stressors are discussed briefly in this section, although this ERA focuses on site-related chemical stressors.

5.2.1 Natural Physical and Chemical Stressors

Naturally-occurring stressors in the estuary at the LCP Site are largely related to the tidal movement of water, which produces a repeating cycle of inundation of salt marshes on incoming tides and draining of marshes and tidal creeks on ebbing tides. Tidal amplitude on the Georgia coast is greater than other regions of southeastern North America. The large daily fluctuation in tide height affects marsh development, primarily through formation of natural levees, which are more prominent in Georgia than elsewhere in the southeastern United States (Wiegert and Freeman 1990). The levees act as natural barriers that prevent tidal waters from reaching the central areas of extensive marshes except during fortnightly spring tides, resulting in increased salinity at the central areas of the marsh surface, which influences the size and distribution of dominant plants (Wiegert and Freeman 1990).

Despite the protective nature of the levees, storm-induced tidal surges can cause erosion and slumping of the levees and destabilization of marsh surfaces. Storm surges can also deposit rafts of dead cordgrass stems ("wrack") on the marsh surface. Vegetation on areas covered by the wrack is killed, and the area remains bare until revegetated by seedlings (Wiegert and Freeman 1990). Storm surges may also cause disturbance of benthic communities in tidal channels. The tidal regime may also have indirect effects on marsh productivity through its influence on salinity, pH, temperature, light density, and nutrient loads (Pomeroy and Wiegert 1981). Seasonal variation in temperature affects cordgrass and algal productivity. Elevated summer temperatures may cause physiological stress for aquatic and terrestrial organisms. Other natural stressors also occur. Elevated concentrations of sulfides and ammonia in marsh sediments can affect benthic organisms and may affect sediment toxicity test results.

5.2.2 Spatial Distribution of CoPCs in Water

The analytical results for whole water and dissolved samples from the study area and reference stations are presented in Table 5-1. Salinity of channel surface water in the study area generally ranged from 19 to 25 parts per thousand (ppt). However, both reference stations exhibited higher salinity (30 ppt in Clubbs Creek and 32 ppt in Jointer Creek). The higher salinities at these reference stations are attributable to their proximity to open ocean water and might confound interpretation of the results of the toxicity and bioaccumulation studies conducted at these stations.

TABLE 5-1. PHYSICAL AND CHEMICAL CHARACTERISTICS OF SURFACE WATER FROM THE LCP SITE AND REFERENCE STATIONS AND COMPARISONS TO GEORGIA MARINE WATER QUALITY STANDARDS

	Concentration in Channel Water											
	Study Area Stations											Georgia Marine Water Quality Standards (ecology/ human-health)
		Purvis Creek					Turtle River			Reference Stations		
Physical/Chemical Variable[a]	LCP Ditch (No. 1)	Head-waters (No. 4)	Upper (No. 3)	Middle (No. 2)	Lower (No. 5)	Gibson Creek (No. 9)	Mouth of Purvis Creek (No. 6)	Upper Range (No. 7)	East River (No. 8)	Clubbs Creek (No. 11)	Jointer Creek (No. 10)	
General Water Quality Variables												
Temperature (°C)	31	30	32	32	32	35	31	32	31	31	30	--
Salinity (ppt)	19	25	21	20	--	21	23	21	20	30	32	--
pH (pH units)	7.3	6.9	7.5	7.5	7.6	7.4	7.6	7.5	7	7.5	6.6	6.0–8.5
Dissolved oxygen (% sat)	60	44	94	93	100	68	100	93	54	66	49	--
Total suspended solids (mg/L)	68	59	49	61	43	200	43	28	60	33	92	Allow legitimate water uses
Metals (mg/L) (24 metals evaluated)												
Aluminum												
Total	0.8	0.8	0.7	0.5	0.6	4.7	0.3	0.3	0.7	0.5	1.3	--
Dissolved	<0.1	<0.1	<0.1	<0.1	<0.1	0.5	<0.1	<0.1	<0.1	<0.1	0.1	--
Arsenic												
Total	<0.004	0.005	0.005	<0.004	<0.004	0.007	<0.004	<0.004	0.005	<0.004	0.006	0.036
Dissolved	<0.08	<0.04	<0.04	<0.04	<0.08	<0.04	<0.04	<0.04	<0.040	<0.08	<0.04	--
Barium												
Total	0.03	0.03	0.03	0.03	0.03	0.04	0.03	0.03	0.03	0.02	0.02	--
Dissolved	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.02	0.02	--
Calcium												
Total	240	220	220	220	230	220	240	230	210	260	270	--
Dissolved	240	220	190	220	240	210	240	240	210	270	290	--
Iron												
Total	0.51	0.52	0.46	0.35	0.34	4.2	0.12	0.13	0.55	0.26	0.9	--
Dissolved	<0.04	<0.04	<0.04	<0.04	<0.04	0.21	<0.04	<0.04	<0.04	<0.04	0.1	--
Magnesium												
Total	880	740	750	730	790	720	830	770	660	890	930	--
Dissolved	820	730	630	720	820	710	820	790	660	930	1,000	--
Manganese												
Total	0.05	0.25	0.26	0.13	0.04	0.09	<0.01	<0.01	0.23	<0.01	0.13	--
Dissolved	<0.01	0.2	0.16	0.08	<0.01	0.02	<0.01	<0.01	0.18	<0.01	0.07	--
Mercury (ng/L)												
Total	**170**	**110**	**97**	**39**	22	**130**	9.4	10	12	3.6	5.8	25
Dissolved	10	24	**27**	17	7.2	4.5	2.9	5.2	3.6	1.3	1.0	25
Methylmercury (ng/L)												
Total	0.68	2.6	2.7	1.3	0.6	0.66	0.23	0.37	0.56	0.25	0.35	
Dissolved	0.5	1.1	1.3	0.83	0.41	0.32	0.15	0.33	0.46	0.16	0.16	--
Potassium												
Total	270	250	250	250	260	240	280	260	230	300	320	--
Dissolved	280	250	210	260	280	240	280	270	230	320	350	
Sodium												
Total	6,700	6,600	6,900	6,100	6,900	6,400	6,800	7,000	5,700	7,100	8,300	--
Dissolved	7,400	6,800	5,000	6,100	7,400	6,100	7,500	7,600	5,400	8,600	8,400	

TABLE 5-1. (cont.)

Physical/Chemical Variable[a]	Concentration in Channel Water											Georgia Marine Water Quality Standard (ecology/ human-health)
	Study Area Stations											
		Purvis Creek					Turtle River			Reference Stations		
	LCP Ditch (No. 1)	Head waters (No. 4)	Upper (No. 3)	Middle (No. 2)	Lower (No. 5)	Gibson Creek (No. 9)	Mouth of Purvis Creek (No. 6)	Upper Range (No. 7)	East River (No. 8)	Clubbs Creek (No. 11)	Jointer Creek (No. 10)	
Vanadium												
Total	<0.01	<0.01	<0.01	<0.01	<0.01	0.1	<0.01	<0.01	0.1	<0.01	<0.01	--
Dissolved	<0.10	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	--
Zinc												
Total	0.01	<0.01	<0.01	0.01	<0.01	0.02	<0.01	0.03	<0.01	<0.01	<0.01	0.086
Dissolved	<0.01	<0.01	<0.01	<0.01	<0.01	0.03	<0.01	<0.01	<0.01	<0.01	<0.01	--
Semivolatile Organic Chemicals (65 semivolatiles evaluated)	none detected											
Pesticides (19 pesticides evaluated)	none detected											
Polychlorinated Biphenyls (8 Aroclors' evaluated)	none detected											

Note: Physical and chemical data are abstracted from Volume II, Section1.1 and pertain to channel water collected from 11 sampling stations distributed throughout the study area.

Channel water was collected on June 26, 1996. Grab samples of water were collected about 0.5 m below the air/water interface from approximately 2 hours on either side of slack low water. The water column at sampling stations was well mixed, as evidenced by differences in subsurface vs. bottom salinity of less than 5 ppt (typically less than 1–2 ppt).

-- - information not available

[a] Only those physical/chemical variables detected at at least one sampling station are addressed in this table. Exceedances of Georgia's marine water quality standards by variables are indicated by bold print.

Only 13 of 24 evaluated metals were detected in any of the surface water samples collected. Total mercury concentrations in bulk water samples from the LCP Ditch; headwaters, as well as upper and middle reaches, of Purvis Creek; and Gibson Creek (Figure 5-1) were greater than the 25 nanograms per liter (ng/L) standard established by the State of Georgia for coastal and marine water quality. (Concentrations above Georgia's standard were also observed during the mercury methylation investigation at stations in the marsh and at the Clubbs Creek reference station.) Some reported total mercury concentrations, however, may be biased high (see Appendix C) because of an accidental release of untreated storm water runoff from the onsite treatment plant (Riner 1996).

Dissolved total mercury concentrations, measured in filtered samples, were near the 25 ng/L State standard in the headwaters and upper reach of Purvis Creek. It is not stated in the standards whether the total or dissolved mercury concentration is the relevant metric. To consider total mercury concentrations on a dissolved water basis is more relevant to toxicity and bioaccumulation concerns than on a whole water sample basis (U.S. EPA 1993b). Dissolved mercury is typically available, depending on the chemical form, to be methylated or to contribute directly to toxicity. Furthermore, mercury concentrations in whole water samples are strongly influenced by mercury associated with particles in the water sample.

Methylmercury constituted approximately 2–5 percent of total mercury in whole water samples at most sampling stations (Figure 5-1). However, this ratio was lower in the LCP Ditch (0.40 percent) and Gibson Creek (0.51 percent) and higher at the two reference stations (6.9 percent at Clubbs Creek and 6.0 percent at Jointer Creek). The low ratio of methylmercury to total mercury in surface water reflects the fact that most of the mercury in water occurs in the divalent inorganic form (Wren et al. 1995). The percentage of total mercury present in dissolved form was highly variable, ranging from 3.5 percent in Gibson Creek (and 5.9 percent in the LCP Ditch) to 52 percent in the upper range of the Turtle River. In the case of methylmercury, 42 percent (headwaters of Purvis Creek) to 89 percent (upper range of Turtle River) was present in dissolved form.

The relatively high concentrations of two common earth metals, aluminum (4.7 mg/L, or parts per million) and iron (4.2 mg/L), in whole water samples from Gibson Creek were consistent with a high concentration of total suspended solids (200 mg/L) measured in the creek and, thus, appear attributable to naturally-occurring conditions.

SVOCs, pesticides, and PCBs were not detected in surface water at any of the 11 sampling stations.

5.2.3 Spatial Distribution of CoPCs in Sediment

The distribution of CoPCs in tidal channel and marsh sediments is discussed in the following sections.

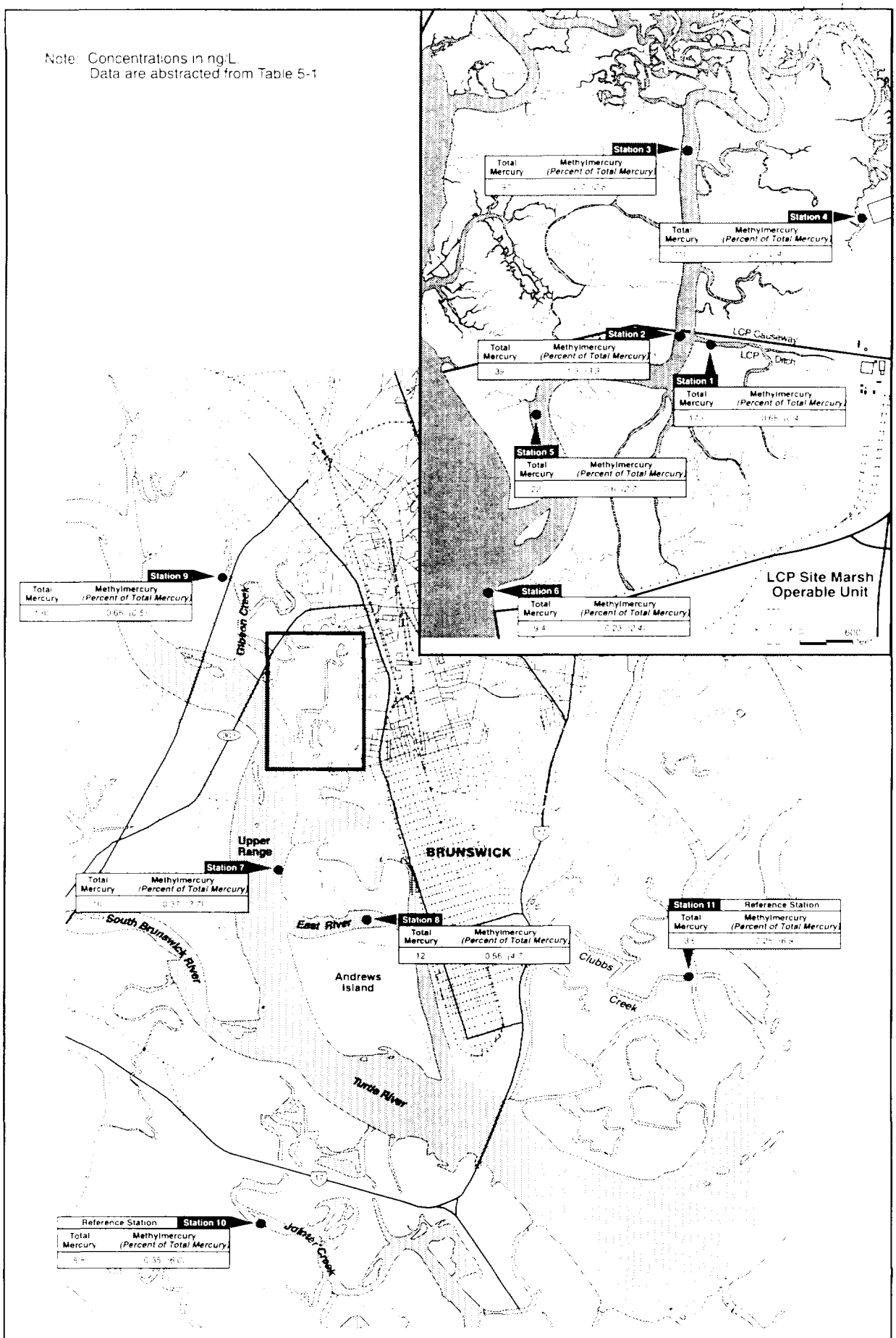


Figure 5-1. Overview of total mercury and methylmercury concentrations in unfiltered surface water from the LCP Site and reference stations.

5.2.3.1 Channel Sediments

Total organic carbon (TOC) concentrations of channel sediment in the study area and reference areas ranged from 0.49 to 9.9 percent (Table 5-2). Concentrations of chemicals in sediment usually bear a positive relationship with TOC (and a negative relationship with sediment particle size) because chemicals bind to organic matter in sediment, thus rendering the chemicals less bioavailable (less toxic) to benthic organisms.

Seven metals (antimony, arsenic, chromium, copper, lead, mercury, and nickel) of the 24 evaluated metals were detected in samples from at least 1 of the 12 study area sampling stations. The two reference stations were generally indistinguishable from each other in terms of presence of metals (and most other chemicals evaluated).

Twelve of the 67 SVOCs evaluated, 2 of the 19 pesticides evaluated, and 1 of the 8 PCBs evaluated were detected. Of the 12 SVOCs (acenaphthene, anthracene, benz[a]anthracene, benzo[a]pyrene, bis[2-ethylhexyl]phthalate, chrysene, dibenz[a,h]anthracene, fluoranthene, fluorene, 2-methylnaphthalene, phenanthrene, and pyrene), 4 compounds did not appear to be uniquely associated with the LCP Site. Acenaphthene and fluorene were never detected in the LCP Ditch or the three site tributaries. Bis[2ethylhexyl]phthalate (which is a common laboratory contaminant) and fluoranthene also occurred at elevated concentrations at one or both reference stations. The pesticides 4,4′-DDT and dieldrin were each detected at only one sampling station within the study area, and both were present at the Jointer Creek reference station. Aroclor® 1268 was present at all nine stations in the study area and the two reference stations.

An overview of CoPC (i.e., lead, total mercury, and Aroclor® 1268) concentrations in channel sediment in the study area (Figure 5-2) shows that lead concentrations were highest in sediment at the LCP Ditch (48–120 mg/kg dry weight) and the headwaters of Purvis Creek (100–180 mg/kg). All other sampling stations, including the two reference stations, were usually characterized by lower and similar sedimentary lead concentrations (2.5–38 mg/kg).

Concentrations of total mercury were highest in the LCP Ditch (33–43 mg/kg dry weight) and associated tributary T2 (2.4–38 mg/kg). Progressively lower concentrations generally occurred in middle Purvis Creek (3.0–18 mg/kg); the headwaters of Purvis Creek (2.3–7.7 mg/kg); the other Purvis Creek stations, the Turtle River stations and the T1 and T3 tributaries (0.06–3.3 mg/kg); Gibson Creek and the East River (<0.05–0.76 mg/kg); and the two reference stations (<0.05–0.10 mg/kg for Clubbs Creek and 0.09–0.13 mg/kg for Jointer Creek).

Aroclor® 1268 concentrations in sediment were highest at the LCP Ditch (140–200 mg/kg dry weight), associated tributary T2 (18–96 mg/kg), and middle Purvis Creek (14–100 mg/kg). Progressively lower concentrations characterized Turtle River near the mouth of Purvis Creek, the East River, and tributary T1 (0.007–35 mg/kg); all remaining

TABLE 5-2. CHEMICAL CHARACTERISTICS OF CHANNEL SEDIMENT FROM THE LCP SITE AND REFERENCE STATIONS

	Concentration in Channel Sediment[a,b]						
	Study Area Stations						
		Site Tributaries			Purvis Creek		
Chemical Variable	LCP Ditch (No. 1)	Northern (No. T3)	Middle (No. T2)	Southern (No. T1)	Headwaters (No. 4)	Upper (No. 3)	Middle (No. 2)
General Sediment Quality Variables							
Total organic carbon (% dry weight)	4.8–6.6	1.9–6.2	4.6–9.9	4.3–6.2	3.4–6.8	0.50–0.84	3.0–5.2
Metals (mg/kg dry weight) (24 metals evaluated)							
Antimony	0.08–0.11	0.03–0.06	0.07–0.12	0.04–0.14	0.06–3.2	<0.02–0.05	0.05–0.09
Arsenic	11–12	5.2–11	11–26	11–15	4.8–10	1.8–3.2	7.4–14
Chromium	61–75	23–100	43–57	38–69	26–86	12–24	46–55
Copper	18–30	3.8–12	6.7–11	6.8–9.6	11–23	1.6–10	9.8–17
Lead	48–120	8.4–27	20–45	19–25	100–190	5.1–6.8	21–38
Mercury							
Total mercury	33–43	0.48–3.3	2.4–38	1.4–2.6	2.3–7.7	0.77–1.5	3.0–18
Methylmercury (μg/kg)	--	0.15–20	0.24–2.0	0.24–2.9	--	--	--
Nickel	14–18	3.7–12	8.1–12	6.2–9.8	5.4–12	1.2–2.4	7.4–15
Semivolatile Organic Chemicals (mg/kg dry weight) (67 semivolatiles evaluated)							
Acenaphthene	<0.50–<0.60	<0.15–<0.60	<0.40–<0.60	<0.40–<1.0	<0.30–<0.60	<0.30–<0.50	<0.30–<0.90
Anthracene	0.05 (<0.60)	<0.20–<0.60	<0.40–<0.60	<0.40–<1.0	0.04–1.5	<0.30–<0.50	0.04–0.05 (<0.90)
Benz[a]anthracene	0.20–0.30 (<0.50)	0.04–0.06 (<0.30)	0.08–0.10 (<0.60)	0.20 (<1.0)	0.20–0.50	<0.30–<0.50	0.06–0.10
Benzo[a]pyrene	0.20–0.40	0.05 (<0.30)	0.04–0.10 (<0.45)	0.08 (<1.0)	0.09–0.40	<0.30–<0.50	0.04–0.07 (<0.90)
Bis[2-ethylhexyl]phthalate	0.50–13	0.10–3.5	0.10 (<0.60)	0.50–2.3	0.30–1.4	0.05–0.20 (<.40)	0.05–18
Chrysene	0.20–0.40 (<0.50)	0.05–0.06 (<0.30)	0.07–0.10 (<0.60)	0.20 (<1.0)	0.20–0.50	<0.30–<0.50	0.06–0.20
Dibenz[a,h]anthracene	0.05–0.07 (<0.50)	<0.20–<0.60	<0.40–<0.60	<0.40–<1.0	<0.30–<0.60	<0.30–<0.50	<0.30–<0.90
Fluoranthene	0.20–0.60	0.04–0.10 (<0.30)	0.10–0.17	0.10–0.20	0.30–0.70	0.05 (<0.50)	0.10–0.30
Fluorene	<0.50–<0.60	<0.20–<0.60	<0.40–<0.60	<0.40–<1.0	0.10 (<0.60)	<0.30–<0.50	<0.30–<0.90
2-Methylnapthalene	0.05 (<0.60)	<0.20–<0.60	<0.40–<0.60	<0.40–<1.0	0.05 (<0.60)	<0.30–<0.50	<0.30–<0.90
Phenanthrene	0.05–0.30 (<0.50)	<0.20–<0.60	<0.40–<0.60	<0.40–<1.0	0.04–0.30	<0.30–<0.50	0.04–0.08 (<0.60)
Pyrene	0.40–1.8	0.04–0.09 (<0.30)	0.10–0.20	0.10–0.30	0.30–0.80	0.04–0.05 (<0.50)	0.10–0.30
Pesticides (mg/kg dry weight) (19 pesticides evaluated)							
4,4'-DDT	<0.04–<0.20	<0.0004–<0.005	<0.01–<0.02	<0.01	<0.004–<0.02	<0.002	<0.01–<0.04
Dieldrin	<0.07–<0.20	<0.002-0.002 (<0.005)	<0.01–<0.02	<0.01	<0.008–<0.02	<0.002	<0.01–<0.05
Polychlorinated Biphenyls (mg/kg dry weight) (8 Aroclors evaluated)							
Aroclor® 1268	140–200	0.14–1.9	18–96	9.0–19	1.4–5.0	1.5–3.4	14–100

TABLE 5-2. (cont.)

Chemical Variable	Concentration in Channel Sediment[a,b]						
	Study Area Stations						
			Turtle River				
	Purvis Creek					Reference Stations	
	Lower (No. 5)	Gibson Creek (No. 9)	Mouth of Purvis Creek (No. 6)	Upper Range (No. 7)	East River (No. 8)	Clubbs Creek (No. 11)	Jointer Creek (No. 10)
General Sediment Quality Variables							
Total organic carbon (% dry weight)	0.49–1.1	0.75–3.2	1.9–4.3	2.8–3.8	0.58–3.9	0.92–4.6	2.5–3.1
Metals (mg/kg dry weight) (24 metals evaluated)							
Antimony	0.20	<0.02–0.04	0.04–0.56	0.03–0.22	<0.02–0.03	<0.02–0.04	0.03–0.04
Arsenic	1.8–3.2	2.4–7.5	6.3–10	10–13	1.4–11	4.7–16	8.5–10
Chromium	7.0–18	17–96	23–44	22–35	9.0–74	12–37	33–47
Copper	1.8–12	2.3–10	6.7–17	7.9–12	1.4–11	2.8–9.2	6.1–8.8
Lead	4.6–26	4.5–15	16–25	15–23	2.5–19	5.0–17	14–20
Mercury							
Total mercury	0.06–1.3	<0.05–0.70	0.20–3.0	0.10–1.9	0.05–0.76	<0.05–0.10	0.09–0.13
Methylmercury (μg/kg)	--	0.079–1.2	1.5	0.005–0.82	0.081–2.5	0.34–2.1	0.06–0.28
Nickel	1.6–3.1	2.0–7.6	5.6–11	6.0–9.5	1.5–11	2.9–9.6	6.6–11
Semivolatile Organic Chemicals (mg/kg dry weight) (67 semivolatiles evaluated)							
Acenaphthene	<0.30–<0.40	<0.15–<0.26	0.06 (<2.2)	<0.15–<0.62	<0.15–<0.38	<0.15–<0.36	<0.40–<0.80
Anthracene	<0.30–<0.40	<0.15–<0.26	0.07–0.20 (<1.0)	0.04–0.05 (<0.62)	<0.15–<0.38	<0.15–<0.36	<0.40–<0.80
Benz[a]anthracene	0.04–1.0	0.06 (<0.26)	0.20–0.50	0.10–0.40	<0.15–<0.38	0.04 (<0.35)	<0.40–<0.80
Benzo[a]pyrene	<0.30–1.4	0.04 (<0.26)	0.20–0.30	0.05–0.30	<0.15–<0.38	0.05 (<0.35)	<0.40–<0.80
Bis[2-ethylhexyl]phthalate	0.05–0.20 (<0.40)	<0.15–<0.43	0.30–3.1	0.10–0.30	<0.10–<2.0	0.10–0.60	<0.40–4.0
Chrysene	0.05–0.80	0.05 (<0.26)	0.30–0.50	0.09–0.30	<0.15–<0.38	0.04 (<0.35)	<0.40–<0.80
Dibenz[a,h]anthracene	<0.30-0.50	<0.15–<0.26	<0.50–<2.2	<0.15–<0.62	<0.15–<0.38	<0.15–<0.36	<0.40–<0.80
Fluoranthene	0.20 (<0.40)	0.10 (<0.26)	0.60–1.4	0.40–1.8	0.07 (<0.38)	0.04–0.70	<0.40–<0.80
Fluorene	<0.30–<0.40	<0.15–<0.26	0.04–0.06 (<2.2)	<0.15–<0.62	<0.15–<0.38	<0.15–<0.36	<0.40–<0.80
2-Methylnapthalene	<0.30–<0.40	<0.15–<0.26	<0.50–<2.2	<0.15–<0.62	<0.15–<0.38	<0.15–<0.36	<0.40–<0.80
Phenanthrene	0.10 (<0.40)	<0.15–<0.26	0.10–0.30 (<1.0)	0.04–0.06 (<0.62)	<0.15–<0.38	<0.15–<0.36	<0.40–<0.80
Pyrene	0.04–9.6	0.10 (<0.26)	0.40–0.90	0.30–1.6	0.06 (<0.38)	0.05–0.06 (<0.35)	<0.40–<0.80
Pesticides (mg/kg dry weight) (19 pesticides evaluated)							
4,4'-DDT	<0.002–<0.02	<0.002–<0.013	<0.005–<0.01	<0.0004–0.002	<0.0003–<0.01	<0.0007–<0.002	<0.001-0.005
Dieldrin	<0.002–<0.02	<0.002	<0.005–<0.01	<0.0005–<0.002	<0.002–<0.01	<0.002	<0.0006-0.0009 (<0.002)
Polychlorinated Biphenyls (mg/kg dry weight) (8 Aroclors evaluated)							
Aroclor® 1268	0.11–6.2	0.05–0.29	0.09–25	0.003–0.01	0.007–35	0.006–0.06	0.01–0.05

Note: Chemical data are abstracted from Volume II, Section 1.3 and pertain to sediment collected on May 15–July 15, 1996, from channels at 14 sampling stations distributed throughout the study area.

-- information not available

[a] The highest detection limit for a nondetected sample is included in parentheses when it exceeds a detected concentration.

[b] Each set of chemical concentrations for each sampling station typically pertains to the concentration range of eight 2-cm-thick samples taken at different depths (0–2, 2–4, 4–6, 6–8, 8–10, 10–12, 14–16, and 18–20 cm) from a single sediment core with total length of 20 cm. Detection limits are presented only if they clearly represent minimum or maximum chemical values.

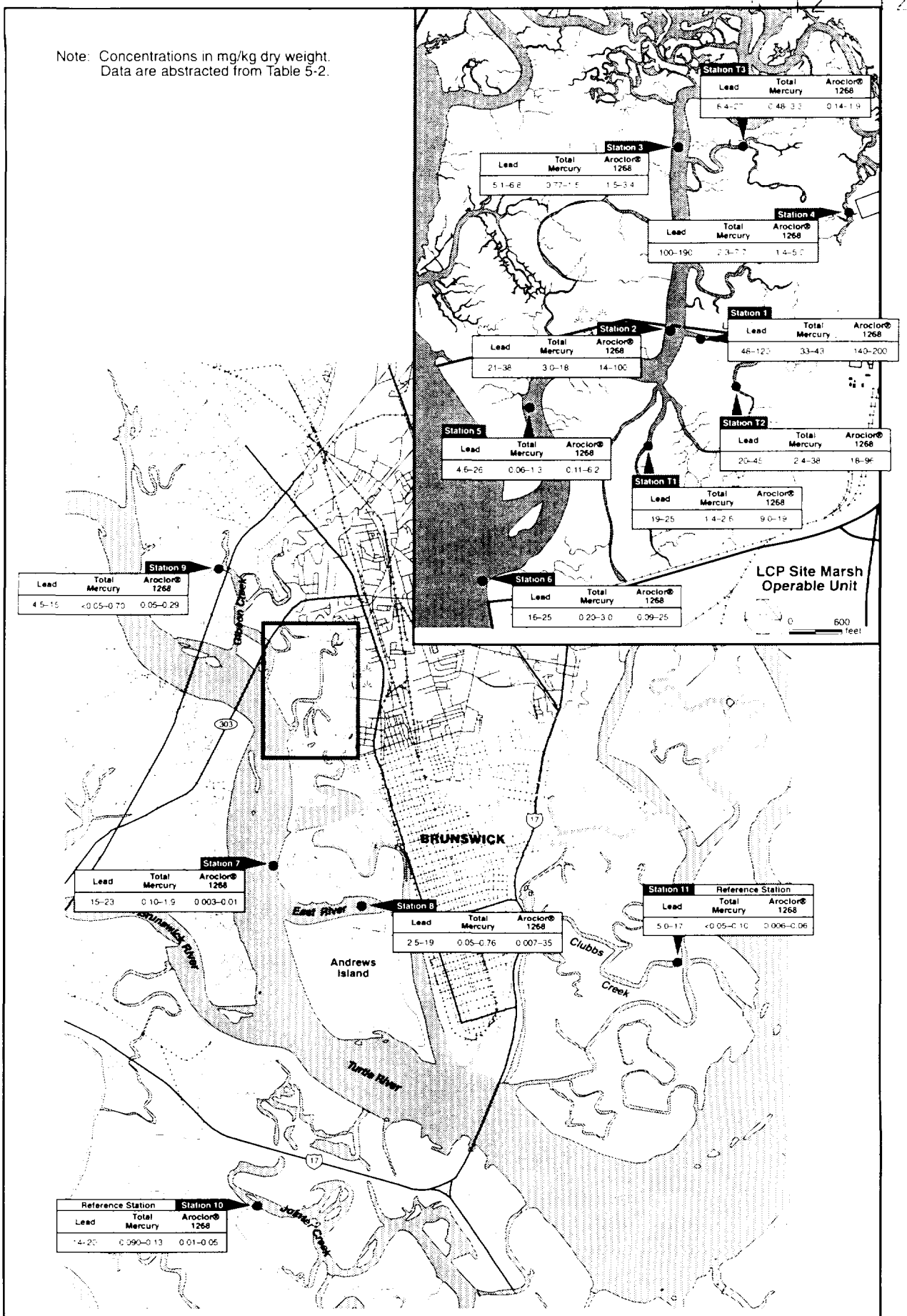


Figure 5-2. Overview of lead, total mercury, and Aroclor® 1268 concentrations in channel sediment from the LCP Site and reference stations.

non-reference stations (0.003–6.2 mg/kg); and the two reference stations (0.006–0.06 mg/kg for Clubbs Creek and 0.01–0.05 mg/kg for Jointer Creek).

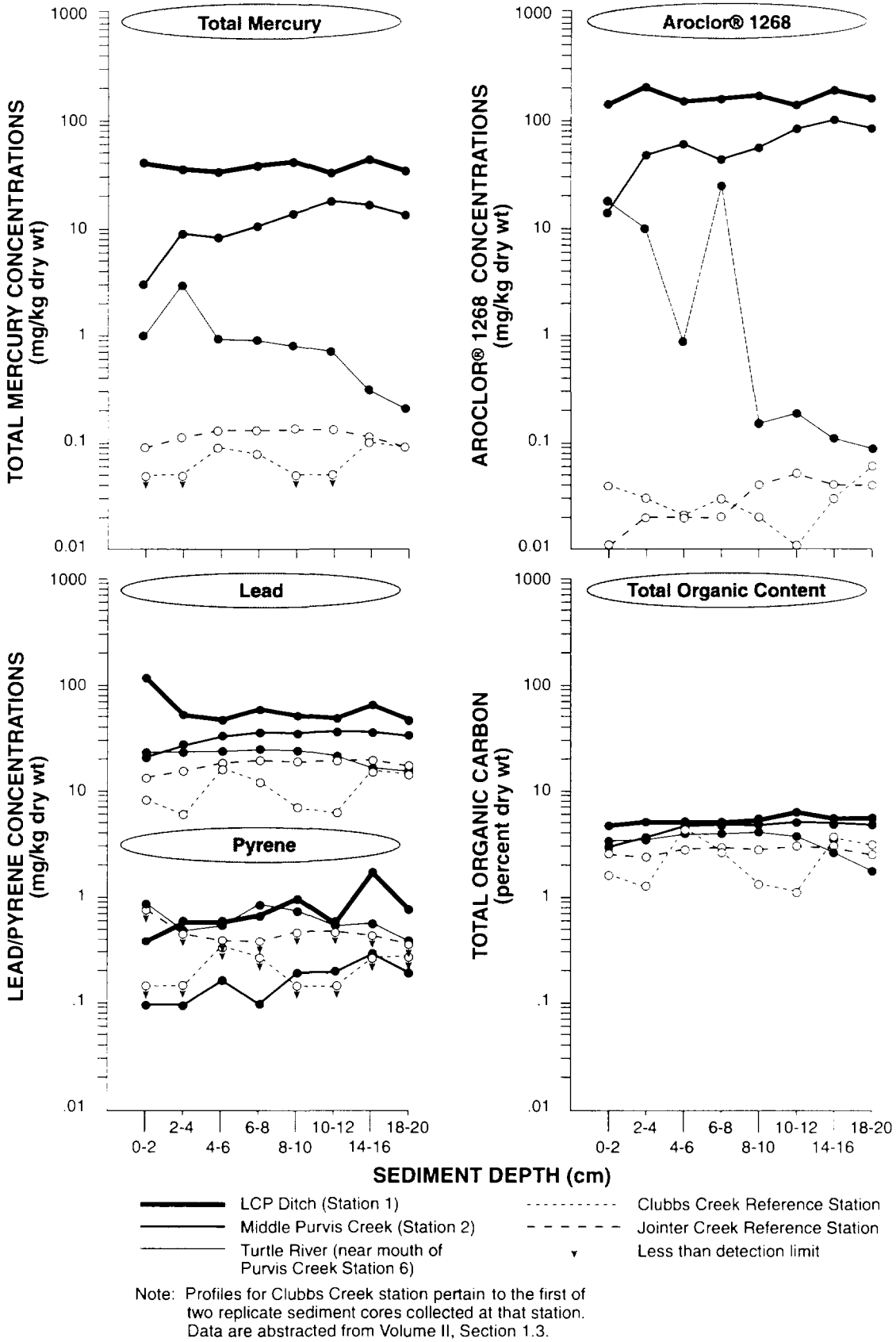
Vertical profiles of four (lead, mercury, Aroclor® 1268, and pyrene) of the more common chemicals detected in channel sediment from five stations in the study area are shown in Figure 5-3. Mercury at middle Purvis Creek and Turtle River (near mouth of Purvis Creek) and Aroclor® 1268 at middle Purvis Creek show lower concentrations in the top 2 cm (relative to the 2–4 cm depth) that may reflect water transport effects and decreased loading of Purvis Creek sediments; other trends in these data were not identified.

Surface sediment samples were collected from seven creek channel stations north of the LCP causeway, including two stations in Purvis Creek, to characterize distribution of CoPCs in tidal channel sediments in the northern part of the marsh (see Figure 4-2). Lead concentrations ranged from 3.7 to 35.9 mg/kg dry weight, and mercury concentrations ranged from 0.33 to 6.5 mg/kg (Table 5-3). For Aroclor® 1268, the only Aroclor detected in the samples, concentrations ranged from 0.24 to 5.0 mg/kg. Concentrations of all CoPCs were lowest at Stations SSC-04 and SSC-05; these stations also had the lowest levels of TOC and the highest sand fractions. Mercury and Aroclor® 1268 concentrations were also low at Station SSC-01, a station with higher lead concentration, TOC, and silt and clay fractions. Lead, mercury, and Aroclor® 1268 concentrations were highest at Station SSC-08, the most northern station sampled. CoPC concentrations at this location were substantially lower than in the LCP ditch and associated tributary, but were high compared to other stations sampled in the northern part of the marsh. This localized increase in concentrations may reflect the influence of limited tidal flushing and lack of significant fresh water input in this part of the marsh. Additional analytical results are presented in Volume II, Section 1.2.

5.2.3.2 Marsh Sediments

Surface Sediments—Samples were collected from the top 5 cm of sediment at 95 sampling stations on the grid established in the marsh south of the causeway and immediately west of the upland portion of the Site (see Figure 4-3). Three marsh stations were also sampled at each of the reference areas along transects running perpendicular to the stream channel. Analytical data for the marsh grid sediment samples are summarized in Table 5-4. Analytical data for the reference area sediment samples are summarized in Table 5-5. Complete data tables for the grid area and the reference area samples are provided in Volume II, Section 1.2. Results for each CoPC are discussed briefly in the remainder of this section.

Similar distribution patterns were found for all of the CoPCs, which include lead, mercury, PCBs, and PAHs (total low molecular weight PAHs [LPAHs] and total high molecular weight PAHs [HPAHs]) in the marsh grid area. Concentrations were highest near the east edge of the marsh, in the area between east-west Transects E and L, and decreased



Note: Profiles for Clubbs Creek station pertain to the first of two replicate sediment cores collected at that station. Data are abstracted from Volume II, Section 1.3.

Figure 5-3. Vertical chemical profiles of channel sediment from the LCP Site and reference stations.

TABLE 5-3. SUMMARY OF CHEMISTRY DATA FOR CREEK CHANNEL SURFACE SEDIMENT SAMPLES FROM THE LCP SITE

Analyte	Frequency of Detection	Range	Mean of Detected Values	Standard Deviation of Detected Values
Conventional Analytes				
Total organic carbon (% dry weight)	7/7	0.43–5.2	3.3	1.8
Total solids (% wet weight)	7/7	23–69.5	38.9	18.1
Metals (mg/kg dry weight)				
Lead	7/7	3.7–35.9	24.0	12.3
Total mercury	7/7	0.33–6.5	2.3	2.0
PCBs (mg/kg dry weight)				
Aroclor® 1016	0/7	<0.04–<0.4	--	--
Aroclor® 1221	0/7	<0.06–<0.6	--	--
Aroclor® 1232	0/7	<0.08–<0.8	--	--
Aroclor® 1242	0/7	<0.03–<0.3	--	--
Aroclor® 1248	0/7	<0.03–<0.3	--	--
Aroclor® 1254	0/7	<0.03–<0.3	--	--
Aroclor® 1260	0/7	<0.03–<0.3	--	--
Aroclor® 1268	7/7	0.24–5.0	2.7	1.9

Note: -- - not applicable

TABLE 5-4. SUMMARY OF CHEMISTRY DATA FOR MARSH GRID SEDIMENT SAMPLES FROM THE LCP SITE

Analyte	Frequency of Detection	Range	Mean of Detected Values	Standard Deviation of Detected Values	Station of Maximum Value
Conventional Analytes					
Ammonia-nitrogen (mg/kg dry weight)	50/50	8.4–270	48.7	40.6	FFDA-E4
Sulfides (mg/kg dry weight)	42/42	20–3300	807	834	FFDA-H2
Total organic carbon (% dry weight)	100/100	1.2–19.1	8.8	4.0	FFDA-I1
Total solids (% wet weight)	100/100	12.0–73.8	25.3	8.4	FFDA-M1
Total Metals (mg/kg dry weight)					
Lead	100/100	14.4–370	67.6	64.4	FFDA-K1
Total mercury	100/100	3.5–476	66.6	100.8	FFDA-F2
Polychlorinated Biphenyls (mg/kg dry weight)					
Aroclor® 1016	0/100	<0.04–<8	--	--	--
Aroclor® 1221	0/100	<0.06–<20	--	--	--
Aroclor® 1232	0/100	<0.08–<20	--	--	--
Aroclor® 1242	0/100	<0.03–<6	--	--	--
Aroclor® 1248	0/100	<0.03–<6	--	--	--
Aroclor® 1254	0/100	<0.03–<31	--	--	--
Aroclor® 1260	0/100	<0.1–<70	--	--	--
Aroclor® 1268	100/100	1.1–890	51	136	FFDA-E3
Semivolatile Organic Compounds (µg/kg dry weight)					
Low Molecular Weight Polycyclic Aromatic Hydrocarbons (LPAH)					
Naphthalene	30/42	2–19	6.9	3.9	FFDA-L1
2-Methyl-naphthalene	39/42	3–52	12	11	FFDA-L1
Acenaphthylene	42/42	2–18	5.4	3.0	FFDA-C1
Acenaphthene	30/42	0.7–8	3.0	1.8	FFDA-L1
Fluorene	33/42	1–10	4.3	2.2	FFDA-A1 & FFDA-L1
Phenanthrene	42/42	4–81	26	19	FFDA-L1
Anthracene	42/42	3–33	14	8.1	FFDA-A1 & FFDA-E9
Total LPAH	42/42	13–196	67	45	FFDA-L1
High Molecular Weight Polycyclic Aromatic Hydrocarbons (HPAH)					
Fluoranthene	42/42	20–183	69	35	FFDA-A1
Pyrene	42/42	19–411	90	72	FFDA-C1
Benz[a]anthracene	42/42	10–114	42	28	FFDA-C1
Chrysene	42/42	14–252	57	45	FFDA-C1
Benzo[b]fluoranthene	41/42	14–319[a]	66	54	FFDA-C1
Benzo[k]fluoranthene	41/42	14–153[a]	45	27	FFDA-C1
Benzo[a]pyrene	42/42	17–640	90	116	FFDA-E3
Indeno[1,2,3-cd]pyrene	42/42	12–313	59	63	FFDA-E9
Dibenz[a,h]anthracene	37/42	4–180	30	41	FFDA-E9
Benzo[ghi]perylene	37/42	14–669	90	126	FFDA-E9
Total HPAH	42/42	144–2,420	620	500	FFDA-C1
Miscellaneous Semivolatile Organic Compounds					
Dibenzofuran	27/42	1–5	2.3	0.9	FFDA-A1

Note: Fine grid samples are not included in this summary.

-- - not applicable

[a] One sample detection limit of 500*U* at Station FFDA-E3 exceeds the maximum detected concentration for this analyte.

TABLE 5-5. SUMMARY OF CHEMISTRY DATA FOR SEDIMENT SAMPLES FROM REFERENCE STATIONS

Analyte	Clubbs Creek				Jointer Creek			
	Frequency of Detection	Range	Mean of Detected Values	Standard Deviation of Detected Values	Frequency of Detection	Range	Mean of Detected Values	Standard Deviation of Detected Values
Conventional Analytes								
Ammonia-nitrogen (mg/kg dry weight)	6/6	5.1–22	13.5	6.0	6/6	22–45	28.3	9.0
Total organic carbon (% dry weight)	6/6	3.3–5.5	4.2	0.7	6/6	5.6–7.4	6.3	0.7
Sulfides (mg/kg dry weight)	3/6	<15–170	121	49	3/6	<15–380	173	187
Total solids (% wet weight)	6/6	37.2–45.3	39.8	3.1	6/6	21.5–33.7	28.7	4.5
Metals (mg/kg dry weight)								
Lead	6/6	16.4–23.3	20.5	2.7	6/6	19.8–27.3	23.3	2.5
Total mercury	6/6	0.06–0.2	0.1	0.05	6/6	0.08–0.1	0.09	0.007
Polychlorinated Biphenyls (mg/kg dry weight)								
Aroclor® 1016	0/6	<0.01–<0.01	--	--	0/6	<0.01–<0.01	--	--
Aroclor® 1221	0/6	<0.01–<0.01	--	--	0/6	<0.01–<0.01	--	--
Aroclor® 1232	0/6	<0.01–<0.01	--	--	0/6	<0.01–<0.01	--	--
Aroclor® 1242	0/6	<0.01–<0.01	--	--	0/6	<0.01–<0.01	--	--
Aroclor® 1248	0/6	0.004–<0.0	--	--	0/6	0.005–<0.0	--	--
Aroclor® 1254	0/6	0.005–<0.0	--	--	0/6	0.004–<0.0	--	--
Aroclor® 1260	0/6	0.004–<0.0	--	--	0/6	0.004–<0.0	--	--
Aroclor® 1268	6/6	0.006–0.05	0.03	0.01	6/6	0.02–0.05	0.03	0.006
Semivolatile Organic Compounds (μg/kg dry weight)								
Low Molecular Weight Polycyclic Aromatic Hydrocarbons (LPAH)								
Naphthalene	4/6	0.4–1[a]	0.8	0.3	3/6	0.7–1[b]	0.9	0.2
2-Methyl-naphthalene	0/6	<5–<5	--	--	0/6	<5–<5	--	--
Acenaphthylene	2/6	0.9–2[c]	1.5	--	5/6	0.6–1[d]	0.84	0.22
Acenaphthene	1/6	1–1[e]	1	--	1/6	1–1[e]	1	--
Fluorene	0/6	<5–<5	--	--	0/6	<5–<5	--	--
Phenanthrene	1/6	<5–5	5	--	2/6	3–22	12.5	--
Anthracene	4/6	0.3–2[a]	0.88	0.76	3/6	0.5–3[b]	1.4	1.4
Total LPAH	5/6	<5–9.9	3.1	4.0	5/6	<5–28	7.4	11.4
High Molecular Weight Polycyclic Aromatic Hydrocarbons (HPAH)								
Fluoranthene	6/6	2–17	9.2	6.3	6/6	5–9	6.5	1.8
Pyrene	6/6	2–12	7.2	4.4	6/6	4–7	5.2	1.3
Benz[a]anthracene	3/6	1–4[f]	2.3	1.5	3/6	2–2[b]	2	0
Chrysene	4/6	2–10	6.3	3.3	4/6	3–6	3.8	1.5
Benzo[b]fluoranthene	4/6	<5–10	8	1.8	3/6	<5–8	7.3	1.2
Benzo[k]fluoranthene	4/6	<5–8	7.3	1.0	3/6	<5–7	6.3	0.6
Benzo[a]pyrene	6/6	2–8	5	2.2	6/6	4–6	4.7	0.8
Indeno[1,2,3-cd]pyrene	3/6	3–6[g]	4	1.7	3/6	4–5[h]	4.3	0.6

TABLE 5-5. (cont.)

Analyte	Clubbs Creek Frequency of Detection	Clubbs Creek Range	Clubbs Creek Mean of Detected Values	Clubbs Creek Standard Deviation of Detected Values	Jointer Creek Frequency of Detection	Jointer Creek Range	Jointer Creek Mean of Detected Values	Jointer Creek Standard Deviation of Detected Values
Dibenz[a,h]anthracene	3/6	0.8–1[b]	0.93	0.12	3/6	0.9–1[b]	0.97	0.06
Benzo[ghi]perylene	3/6	2–5[f]	3.3	1.5	3/6	4–5[f]	4.3	0.6
Total HPAH	6/6	13–68	41	2.3	6/6	27–37	32	4.7
Miscellaneous Semivolatile Organic Compounds								
Dibenzofuran	0/6	<5–<5	--	--	1/6	0.8–0.8[e]	0.8	--

Note: -- - not applicable

[a] Two sample detection limits of 5 *U* exceed the maximum detected concentraton for this analyte.

[b] Three sample detection limits of 5 *U* exceed the maximum detected concentraton for this analyte.

[c] Four sample detection limits of 5 *U* exceed the maximum detected concentraton for this analyte.

[d] One sample detection limit of 5 *U* exceeds the maximum detected concentraton for this analyte.

[e] Five sample detection limits of 5 *U* exceed the maximum detected concentraton for this analyte.

[f] One sample detection limit of 6 *U* exceeds the maximum detected concentraton for this analyte.

[g] One sample detection limit of 7 *U* exceeds the maximum detected concentraton for this analyte.

[h] Two sample detection limits of 6 *U* exceed the maximum detected concentraton for this analyte.

outward to the west, north, and south of this area. The distributions of lead, mercury, and Aroclor® 1268 are depicted in Plates 1, 2, and 3, respectively (included at the end of the report).

No concentration gradients were evident along the tidal channel transects in the reference areas, although small decreases in total mercury and Aroclor® 1268 concentrations were seen with distance from the channel at the Clubbs Creek reference area. Lead concentrations in samples from the marsh grid area ranged from 14.4 to 370 mg/kg dry weight and averaged 68 mg/kg. Lead concentrations in sediment samples from the two reference areas ranged from 16.4 to 23.3 mg/kg at Clubbs Creek and from 19.8 to 27.3 mg/kg at Jointer Creek. The arithmetic average for both reference areas combined was 21.9 mg/kg. The minimum concentration found in the grid area, 14.4 mg/kg at Station D2, was slightly lower than the reference area concentrations.

Total mercury concentrations in samples from the marsh grid area ranged from 3.5 to 476 mg/kg dry weight and averaged 67 mg/kg. Mercury concentrations in sediment samples from the two reference areas were similar to each other, with concentrations ranging from 0.06 to 0.2 mg/kg at Clubbs Creek and from 0.08 to 0.1 mg/kg at Jointer Creek. The arithmetic average for both reference areas combined was 0.09 mg/kg.

Aroclor® 1268, the only Aroclor identified in the marsh grid area during this study, was detected at every station. Aroclor® 1268 concentrations in the marsh grid area ranged from 1.1 to 890 mg/kg dry weight and averaged 51 mg/kg. Aroclor® 1268 concentrations in sediment samples from the reference areas ranged from 0.006 to 0.05 mg/kg at Clubbs Creek and from 0.02 to 0.05 mg/kg at Jointer Creek. The arithmetic average for both reference areas combined was 0.030 mg/kg.

Target PAHs were detected at most stations in the grid area (Table 5-4). The frequency of detection for individual PAHs ranged from 30 to 42 for 42 samples. Total LPAH concentrations (only detected values were included in the sum) ranged from 13 to 196 micrograms per kilogram (μg/kg), with an average of 67 μg/kg. Total HPAH concentrations (only detected values were included in the sum) ranged from 144 to 2,420 μg/kg, with an average of 620 μg/kg.

Low concentrations of LPAHs and/or HPAHs were found at every station in both reference areas (Table 5-5). Individual LPAHs were detected at concentrations equal to or less than the method reporting limit of 5 μg/kg in all samples, with one exception: the concentration of phenanthrene in the sample collected from Jointer Creek Station FCR-02 (100 ft from the tidal channel) on June 18, 1996, was 22 μg/kg. Phenanthrene was undetected in the second sample from this station, which was collected 5 days later on June 23, 1996. The concentrations of individual HPAHs detected at the Clubbs Creek reference area ranged from 0.8 μg/kg for dibenz[a,h]anthracene to 17 μg/kg for fluoranthene. At the Jointer Creek reference area, concentrations of individual HPAHs ranged from 0.9 μg/kg for dibenz[a,h]anthracene to 9 μg/kg for fluoranthene.

Concentrations of mercury and Aroclor® 1268 showed good reproducibility for all samples collected at each of the fine-scale grid sampling stations (Table 5-6). The percent relative standard deviation for mercury was similar between the two sampling stations. The percent standard deviation varied about 2-fold between stations for Aroclor® 1268 and was larger than the values for mercury. For all analytes at each station, the variability of results was similar to or only slightly higher than the analytical precision that might be expected for replicate analyses of a single sample at the laboratory.

Subsurface Sediments—Sediment core samples were collected from three locations in the marsh (Stations SCM-01, SCM-02, SCM-03) and from one location at each of the reference areas in May 1996 (see Figure 4-4). Sediment cores were collected to a depth of 20 cm and divided into 2-cm intervals, with selected intervals submitted for analysis. Complete data tables for the core samples are provided in Volume II, Section 1.3.

Lead concentrations in the surface layer (0–2 cm) of core samples from the marsh ranged from 22 to 32 mg/kg dry weight and were approximately 2-fold higher than concentrations in samples from the Jointer Creek reference area. The Clubbs Creek reference area had slightly lower concentrations than the Jointer Creek reference area. There was little change in concentration with depth for all samples, except that Stations SCM-02 and SCM-03 showed an approximate 2-fold increase in lead concentrations at 18–20 cm.

Mercury concentrations in the surface layer of core samples from the marsh ranged from approximately 2 mg/kg dry weight at Stations SCM-01 and SCM-03 to 24 mg/kg dry weight at Station SCM-02. The Clubbs Creek reference area had slightly lower concentrations than the Jointer Creek reference area. There was little change in concentration with depth for all samples, except that concentrations at Stations SCM-01 and SCM-02 decreased substantially below 16 cm and 12 cm, respectively.

Aroclor® 1268 concentrations in the surface layer of core samples from the marsh ranged from 0.27 mg/kg dry weight at Station SCM-01 to 3.0 mg/kg dry weight at Station SCM-02. The reference area samples had Aroclor® 1268 concentrations approximately an order of magnitude lower than the marsh samples. There was little change in concentration with depth for samples from either reference area, but there was an indication of relatively higher concentrations at mid-depth for the three LCP marsh stations.

The concentrations of LPAHs and HPAHs at every marsh station were at or below the detection limit for all compounds, except for pyrene, benzo[a]pyrene, and benzo[ghi]perylene. These compounds were only detected at low concentrations in the 18–20 cm sediment layer from Station SCM-02.

Comparison With Previously Collected Sediment Data—The EPA Environmental Response Team collected sediment samples from the marsh grid area during the

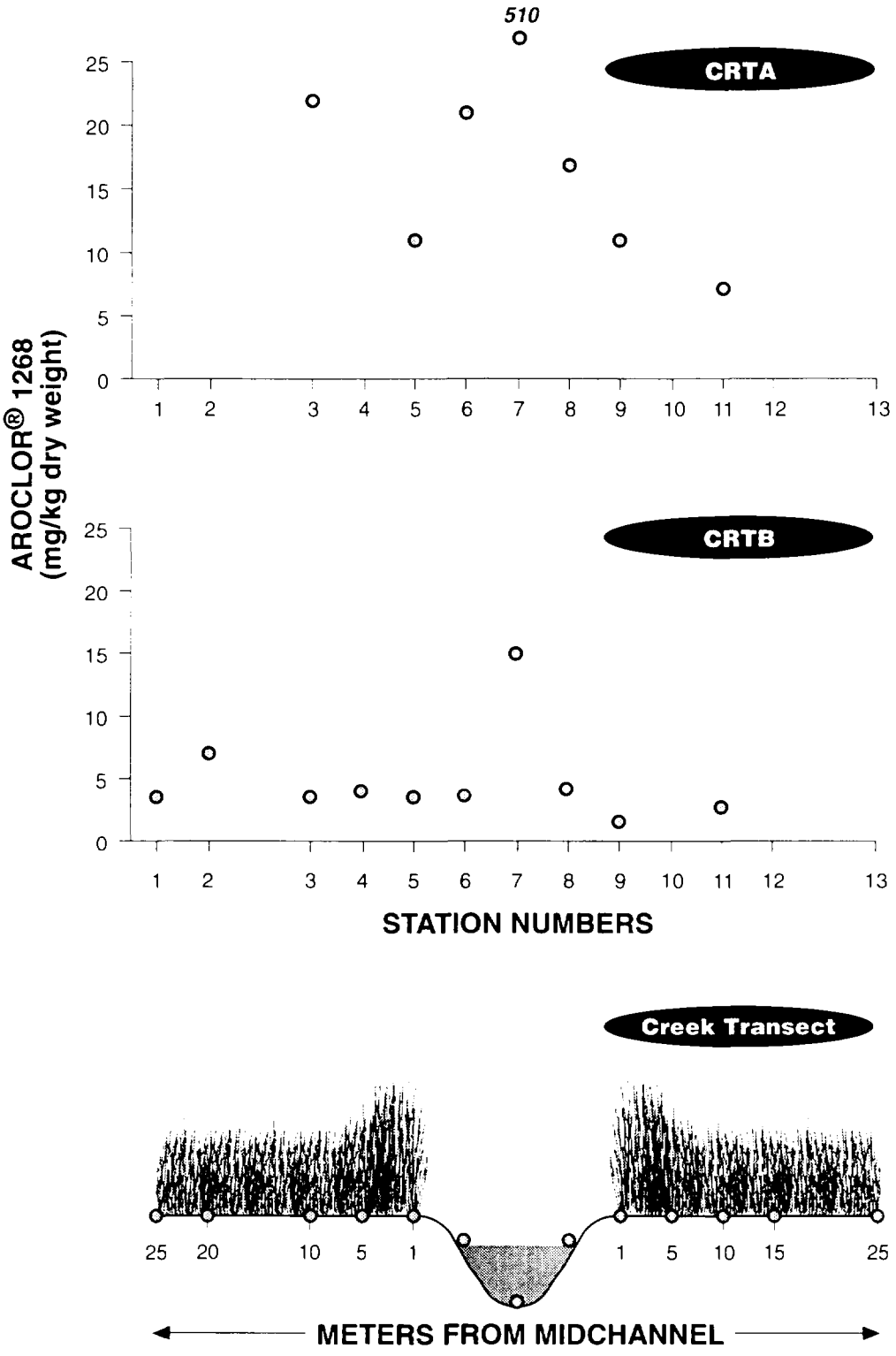
TABLE 5-6. SUMMARY OF CHEMISTRY DATA FOR FINE-SCALE MARSH GRID SEDIMENT SAMPLES FROM THE LCP SITE

Analyte	Number of Samples	Concentration Range	Average Concentration	Standard Deviation	Relative Standard Deviation (percent)
Station H4					
Total solids (percent)	10	18.6–21.2	19.6	0.8	4.1
Total mercury (mg/kg dry weight)	10	55.1–67.6	61.0	3.9	6.4
Aroclor® 1268 (mg/kg dry weight)	10	8.3–23	14.5	5.1	35.2
Station J7					
Total solids (percent)	10	28.2–31.6	29.6	1.2	4.0
Total mercury (mg/kg dry weight)	10	7.8–10.4	8.8	0.8	9.1
Aroclor® 1268 (mg/kg dry weight)	10	2.7–5.5	4.3	0.9	20.9

summer and fall of 1995 as part of a study at the LCP Site (Sprenger et al. 1997). Results of this investigation fall within a factor of approximately 2 to 3 of EPA's results for lead and mercury concentrations. No consistent bias for either of the two data sets was evident. Differences in results between the data sets for samples from similar locations can reasonably be attributed to a combination of small-scale environmental variability, analytical variability, and the transport of wind-blown clean fill from the soil cap adjacent to the marsh grid area.

Both data sets showed similar concentration and distribution patterns for lead, mercury, and total PCBs. The concentrations of lead and mercury were highest near the east edge of the marsh grid area and decreased toward the west, north, and south of the grid area. One notable difference in EPA's study was the detection of low concentrations of Aroclor® 1260, in addition to Aroclor® 1268. Aroclor® 1260 was not detected in the marsh grid area during the 1996 investigation. Another difference between the data sets was the magnitude of the maximum Aroclor® 1268 concentrations found near Stations E3 and F3 of this investigation: EPA found a maximum concentration of 3,800 mg/kg for Aroclor® 1268 at Station E3, and this study found a maximum concentration of 890 mg/kg. In addition, EPA's data showed a localized maximum of 942 mg/kg for Aroclor® 1268 at EPA Station 51 that was not confirmed by the 1996 investigation data set. Finally, concentrations of total PCBs varied from 8.8 to 28 mg/kg at the 1996 stations (Stations K2, K3, L2, and L3) located adjacent to EPA's stations, which showed the highest concentrations of PCBs. The differences in reported values between data sets suggest that a large degree of heterogeneity exists over short distances in areas of the marsh with highly elevated Aroclor® 1268 concentrations. In contrast, results summarized in Table 5-6 for the fine-scale marsh grid samples indicate that spatial variability is less pronounced in areas of the marsh with moderately elevated Aroclor® 1268 concentrations.

Elevated concentrations of lead, mercury, and PCBs were found by EPA along the banks of the drainage channel to the west of the grid site. Comparable samples were not collected during the 1996 AlliedSignal investigation. However, as part of a preliminary field sampling event in January 1996, two sediment sampling transects (CRTA and CRTB) were arranged perpendicular to drainage creeks in the marsh at the LCP Site (see Figure 4-5). Thirteen sampling stations were established along each transect to determine the pattern of sediment PCB concentrations with respect to the centerline of the drainage creek, the banks and adjacent overbank (1 m from the edge of the creek), and the surrounding upper marsh area up to a distance of 25 m from the tidal channel (Figure 5-4). Sediment samples were analyzed by gas chromatography for Aroclors® 1016, 1221, 1232, 1242, 1248, 1254, 1260, and 1268; Aroclor® 1268 was the only PCB mixture detected. Concentrations of Aroclor® 1268 in sediment samples from creek Transect CRTB were lower than values observed at creek Transect CRTA (Figure 5-4). Aroclor® 1268 concentrations in Transects CRTA and CRTB were highest in the creek (510 and 15 mg/kg dry weight, respectively), with concentrations generally decreasing with distance from the centerline of the creek. These data support the conceptual model that channel transport processes are responsible for the transport of PCBs in the marsh ecosystem.



Note: Station numbers align with distances indicated on creek

Figure 5-4. Concentrations of Aroclor® 1268 in marsh sediment from creek transects A and B (CRTA and CRTB) at the LCP Site.

5.2.4 Congener Analysis of Environmental Samples

As part of the preliminary field sampling event in January 1996, PCB congener analysis was performed on four sediment and three tissue samples. Two sediment samples were collected from Stations 7 and 8 at creek Transect CRTB, and two additional sediment samples were collected at distinct areas in the marsh (Stations CONA and CONB; see Figure 4-5). Two tissue samples, a vegetation sample (smooth cordgrass) and an invertebrate sample (periwinkle), were collected in the vicinity of Station CONA. A second invertebrate sample (a shrimp species) was collected in the LCP Ditch. Sediment and tissue samples were analyzed for PCB homologs and a total of 31 individual PCB congeners.

5.2.4.1 Homolog Distribution in Sediment, Vegetation, and Invertebrate Samples

Hepta-, octa-, nona-, and deca-chlorinated biphenyls (i.e., PCBs with 7, 8, 9, and 10 chlorine atoms) were the primary homologs detected in all sediment and tissue samples (Table 5-7). Total PCB concentrations, as determined by homolog analysis for sediments from Stations CONA and CONB, were 1.9 and 0.5 mg/kg dry weight, respectively. These results are consistent with previous EPA and AlliedSignal data (Plate 3; included at the end of the report) that indicate a concentration gradient of total PCBs in marsh sediments from east (high) to west (low). These homolog concentration trends are also consistent with trends observed for congener analysis and Aroclor® 1268 concentrations from gas chromatography analyses of the same sediment samples.

The one vegetation sample (cordgrass) contained a lower concentration of total PCB homologs (9.5 μg/kg dry weight) of all invertebrate samples analyzed. This is consistent with our knowledge of uptake of polychlorinated dibenzo-*p*-dioxins and polychlorinated dibenzofurans, namely that root uptake/translocation is not relevant, and that volatilization/deposition predominates (McLachlan 1997). As for the sediment samples, only the higher chlorinated congeners (hexa and higher) were dominant (Table 5-7 and Figure 5-5). The higher chlorinated PCBs (hexa and higher) were also the primary homologs detected in the periwinkle and shrimp tissue samples. Total PCB concentrations in the invertebrate samples were intermediate between the cordgrass and sediment samples.

5.2.4.2 Congener Distribution in Sediment, Vegetation, and Invertebrate Samples

Sediment and tissue samples were analyzed for a total of 31 PCB congeners (Table 5-8). PCB congeners detected in sediment and tissue samples were primarily the higher chlorinated congeners (more than five chlorine atoms). PCB congener profiles of each of the sediment samples are nearly identical. For example, PCB congeners 28, 101, 138, 153, 170, 180, 187, 195, 202, 206, 207, and 209 were detectable in the low μg/kg dry weight

TABLE 5-7. POLYCHLORINATED BIPHENYL HOMOLOG ANALYSES FOR SURFACE (0–5 cm) SEDIMENT AND TISSUE SAMPLES FROM THE LCP SITE

	Sediment				Tissue		
	Station CONA	Station CONB	CRTB07	CRTB08	Cordgrass	Periwinkle	Shrimp[a]
Mono	0	0	0	0	0	0	0
Di	0	0	0	0	0	0	0
Tri	1	0	0	0	0	0	0
Tetra	0	0	0	0	0	0	0
Penta	1	0	0	0	0	0	0
Hexa	19	0	6	4	0	4	22
Hepta	150	30	62	49	1	21	52
Octa	640	170	370	180	1	56	87
Nona	870	210	530	260	8	71	61
Deca	210	45	100	57	1	8	6
Total	**1,891**	**455**	**1,068**	**550**	**11**	**160**	**228**

Note: Concentrations reported as μg/kg dry weight for sediment and as μg/kg wet weight for cordgrass, periwinkle, and shrimp.

[a] Shrimp sample was not taxonomically identified.

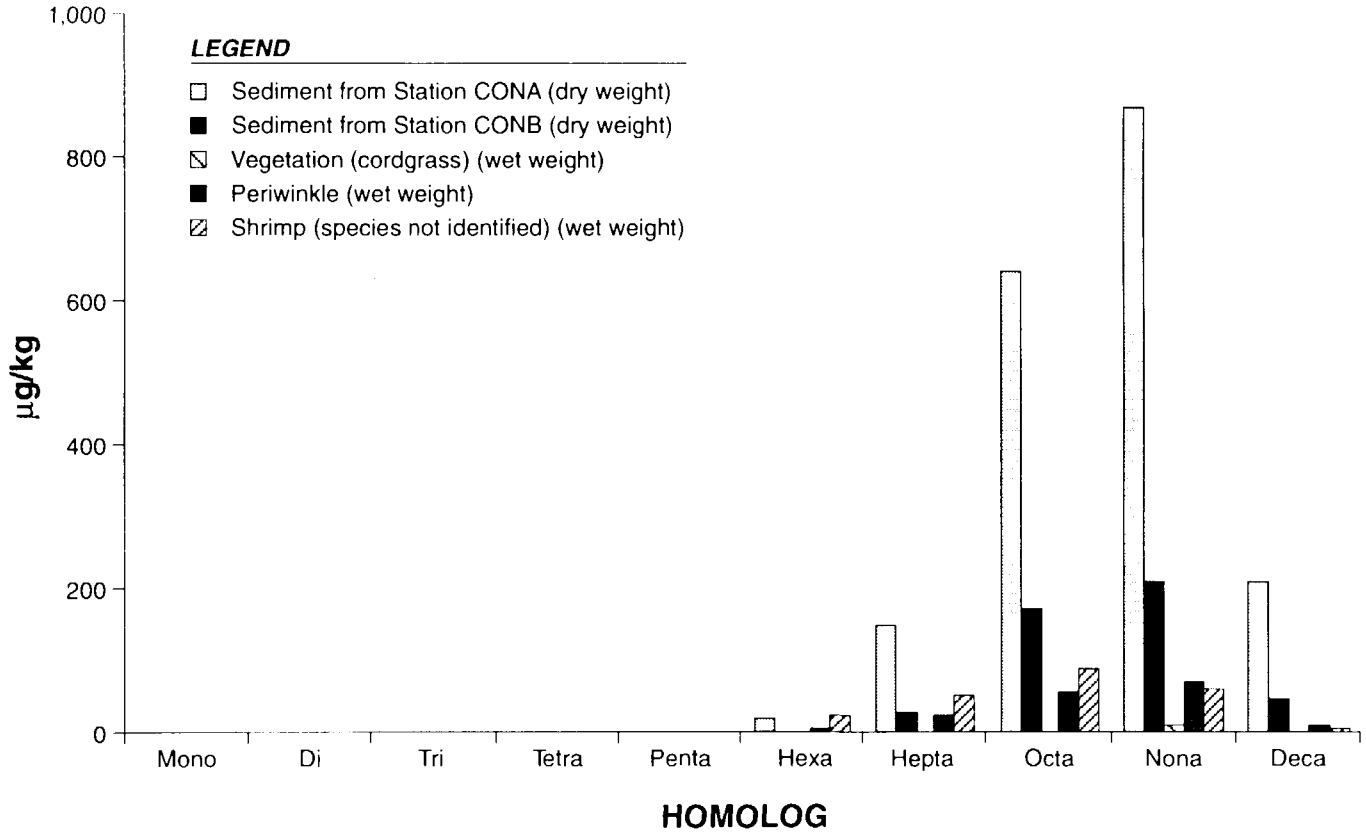


Figure 5-5. Homolog concentrations in sediment, vegetation, and invertebrates from the LCP Site.

TABLE 5-8. POLYCHLORINATED BIPHENYL CONGENER ANALYSES FOR SURFACE (0–5 cm) SEDIMENT AND TISSUE SAMPLES FROM THE LCP SITE

			Sediment								Tissue					
			Station CONA		Station CONB		CRTB-7		CRTB-8		Cordgrass		Periwinkle		Shrimp[a]	
PCB	Homolog	PCB Structure	µg/kg DW	Percent of total	µg/kg DW	Percent of total	µg/kg DW	Percent of total	µg/kg DW	Percent of total	µg/kg DW	Percent of total	µg/kg DW	Percent of total	µg/kg DW	Percent of total
3	Mono	4														
15	Di	4,4'														
18	Tri	2,2',5														
28		2,4,4'	0.58	<0.1												
44	Tetra	2,2',3,5'														
52		2,2',5,5'														
66		2,3',4,4'														
77		3,3',4,4'														
81		3,4,4',5														
101	Penta	2,2',4,5,5'	0.56	<0.1												
105		2,3,3',4,4'														
114		2,3,4,4',5														
118		2,3',4,4',5														
123		2',3,4,4',5														
126		3,3',4,4',5														
128	Hexa	2,2',3,3',4,4'														
138		2,2',3,4,4',5'	1.4	0.1												
153		2,2',4,4',5,5'	3.3	0.3			1.2	0.2					0.85	1.3	5.6	5
156		2,3,3',4,4',5														
157		2,3,3',4,4',5'														
167		2,3',4,4',5,5'														
169		3,3',4,4',5,5'														
170	Hepta	2,2',3,3',4,4',5	0.62	0.1												
180		2,2',3,4,4',5,5'	20	2	4.5	2	11	1.8	5.6	1.8			2.3	3.5	5.9	5.3
187		2,2',3,4',5,5',6	68	6.6	14	6	41	6.8	25	7.9	0.96	10.1	1.8	2.8	33	30
189		2,3,3',4,4',5,5'														
195	Octa	2,2',3,3',4,4',5,6	1.7	0.2			0.85	0.1								
202		2,2',3,3',5,5',6,6'	110	11	26	11	66	11	33	10	0.78	8.2			19	17
206	Nona	2,2',3,3',4,4',5,5',6	550	54	130	56	350	58	180	57	6.3	67	46	71	38	34
207		2,2',3,3',4,4',5,6,6'	59	5.8	12	5.2	35	5.8	17	5.3	0.71	7.5	5.4	8.3	3.8	3.4
209	Deca	2,2',3,3',4,4',5,5',6,6'	210	21	45	19	100	17	57	18	0.73	7.7	8.4	13	5.6	5
TOTAL			**1,025**	**101.1**	**232**	**99.2**	**605**	**100.7**	**318**	**100.0**	**9.5**	**100.5**	**65**	**99.9**	**111**	**99.7**

Note: No entry indicates that the compound was not detected.

DW - dry weight

[a] Shrimp species was not taxonomically identified.

range in sediment from Station CONA. The congener profile of the sediment sample from Station CONB strongly resembles the sample from Station CONA; however, the values are considerably lower and congeners 28, 101, 138, 153, 170, and 195 were not detected. PCB congener profiles of sediment samples from Stations 7 and 8 at Transect CRTB are similar to profiles for Stations CONA and CONB sediment samples, and concentrations of individual and total PCB congeners of creek sediment are generally intermediate between the two marsh stations (Table 5-8). Moreover, none of the coplanar substituted congeners (i.e., IUPAC #s 77, 81, 126, and 169) or mono-ortho substituted congeners (i.e., IUPAC #s 105, 114, 118, 123, 156, 157, 167, and 189) were detected in any of the sediment samples. These are considered to be the "dioxin-like" PCB congeners (Ahlborg et al. 1994).

PCB congener profiles for tissue (vegetation and invertebrate) are similar to profiles for sediment samples; however, concentrations of individual congeners in tissue samples were consistently lower than concentrations observed in sediment samples (Table 5-8). Only 5 of 31 congeners were detectable in the vegetation sample, and concentrations of individual congeners were dramatically lower than concentrations observed for all other samples. Concentrations of individual PCB congeners appear to be similar in the two invertebrate tissue samples.

Results and conclusions of the PCB congener analysis are based on small sample sizes. In particular, congener analysis in tissue samples is based on one sample per species; although for periwinkles and cordgrass, this sample represents a composite of multiple individual organisms. Because of the very limited sample sizes, it is uncertain if these results are representative of biota in the marsh, and conclusions should not be over-interpreted. This investigation was used to provide some initial insight into congener patterns in environmental media. Results of this study are not used in the food-web evaluations or in the weight-of-evidence approach to evaluate risk to the estuarine ecosystem at the LCP Site.

5.3 EXPOSURE CHARACTERIZATION FOR AQUATIC LIFE AND WILDLIFE

This section addresses data on CoPC concentrations in prey species at the LCP Site and food-web model issues or species-specific factors that influence the extent of exposure for the wildlife receptors.

5.3.1 Body Burdens of Prey of Wildlife

Prey tissue sampling was conducted at three transects in the LCP Site marsh, at one transect in the Jointer Creek reference area, and at one transect in the Clubbs Creek reference area to provide baseline data on CoPC concentrations in prey species (see Figure 4-6). Prey species collected were fiddler crabs, periwinkles, American oysters, insects, and smooth cordgrass.

Fiddler crabs and periwinkles showed similar spatial trends in body burdens of CoPCs among transects. Fiddler crabs collected from Transects 1, 2, and 3 had similar tissue burdens of lead, total mercury, methylmercury, and Aroclor® 1268. Lead concentrations in crabs from these three transects were less than or equal to concentrations in crabs from the Jointer Creek reference area, total mercury and methylmercury concentrations were approximately 9- to 15-fold higher, and Aroclor® 1268 concentrations were 6- to 12-fold higher. Tissue burdens in crabs from the Clubbs Creek reference area were similar to the burdens in crabs from the Jointer Creek reference area, except for lead, which was twice as high in crabs at Jointer Creek than crabs at Clubbs Creek.

Periwinkles collected from Transects 1, 2, and 3 also had similar tissue burdens of total mercury and Aroclor® 1268. Lead concentrations in periwinkles from Transects 1 and 3 were similar to periwinkles from reference areas, but concentrations in periwinkles at Transect 2 were 2- to 6-fold higher. Total mercury and methylmercury concentrations were approximately 3- to 16-fold higher, and Aroclor® 1268 concentrations were 10- to 22-fold higher in periwinkles from Transects 1, 2, and 3 than in periwinkles from the reference areas. Tissue burdens in periwinkles from the Clubbs Creek reference area were generally the same or less than burdens in periwinkles from the Jointer Creek reference area.

Chemical concentrations in smooth cordgrass (stems only) from Transects 1, 2, and 3 were relatively similar, with lead concentrations approximately 1- to 3-fold higher than at the reference areas, total mercury concentrations 3- to 19-fold higher, and Aroclor® 1268 concentrations 1- to 3-fold higher.

5.3.2 Wildlife Food-Web Exposure Modeling Approach

Exposure assumptions for the wildlife species selected for this ERA are based on CoPC characteristics and natural history information compiled from the literature for each receptor species. Estimates of exposure based on the food-web model are presented later in this section.

The initial part of this section is a general discussion of several issues related to exposure that are pertinent to all species for which exposure is being modeled. These issues include assumptions about area use factors, time use factors, food ingestion rates, sediment ingestion rates, receptor body weights, chemical bioavailability, and mercury speciation. The latter part of the section presents species-specific exposure parameters.

5.3.2.1 Area Use Factor

An area use factor is used to account for differences in the size of an organism's foraging range in the Brunswick area and the size of the LCP Site. Because of the large areal extent of the marsh at the LCP Site (480 acres), the foraging range of some receptors may be confined entirely within the boundaries of the marsh, indicating that an area use factor

of 1 is appropriate. However, for species with a home range substantially smaller in area than the area of the marsh, individuals likely use only a small section of the total marsh. In this case, the area use factor of 1 should not be assumed to indicate that an individual of the receptor species uses all parts of the marsh equivalently. Larger-bodied, higher-trophic-level species may require larger foraging ranges to meet nutritional demands, and an area use factor of 1 represents an overly conservative estimate of site use by these species. For estimated risk calculations, site-specific home range information more accurately estimates the percentage of time a receptor would use a site (U.S. EPA 1996). Appropriate area use factor values for each species are discussed below in the section on life histories of individual receptor species.

5.3.2.2 Time Use Factor

Some receptors may only occur in the Brunswick area seasonally and migrate elsewhere for the remainder of the year. For these species, the assumption that exposure to CoPCs at the LCP Site is continuous represents an overly conservative estimate of site use. Scaling the exposure estimate by a time use factor to account for the proportion of the year a species occurs in the Brunswick area more accurately estimates chronic exposure. Time use factors are discussed in Section 5.3.2.8 in the description for West Indian manatee, the only species for which a time use factor is applied.

5.3.2.3 Food Ingestion Rate

A key component of the exposure assessment is the food ingestion rate for receptors. The ingestion rate of an organism is a function of the energy requirements, energy density (caloric content) of the diet, and the efficiency of energy assimilation from the diet. Food ingestion rates are rarely measured in wild animals, and estimates of ingestion rates based on studies of animals fed in captivity may not be representative of food requirements of free-ranging individuals. Nagy (1987) developed allometric equations for birds, mammals, and lizards that estimate food ingestion rates as a function of body weight, free-living metabolic rate, and dietary composition. For exposure models, food ingestion rates, in grams of dry weight per day (although food is consumed in a wet weight state) consumed by the receptor, were modeled based on Nagy's allometric equations.

5.3.2.4 Sediment Ingestion Rate

Receptors may be exposed to CoPCs in sediment as a result of incidental ingestion of sediment when consuming prey items. To quantify exposure through the sediment ingestion route, methylmercury and Aroclor® 1268 concentrations in sediment are included in food web exposure models for receptor species.

A geostatistical analysis was used to estimate the area-weighted average sediment concentrations for mercury and Aroclor® 1268 in marsh areas at the Site, based on data

collected as part of this ERA and by GeoSyntec Consultants (both on behalf of AlliedSignal) and data from EPA (Appendix D). Data were analyzed separately for channel and non-channel areas, and in combination. Data were analyzed for the entire marsh, or for different areas of the marsh using four kriging facies based on physical characteristics that limit spatial continuity among data points. The four facies were Zone 1, the marsh area between the removal area and the unnamed channel immediately to the west; Zone 2, the generally southerly zone between Purvis Creek and the aforementioned channel; Zone 3, the marsh north of the causeway and east of Purvis Creek; and Zone 4, the zone west of Purvis Creek. Average concentrations calculated by these kriging analyses are presented in Table 5-9. Complete details on computational procedures, raw data, and derivation of results are presented in Appendix D.

5.3.2.5 Water Ingestion Rate

Daily water requirements depend on the rate at which animals lose water to the environment by evaporation and excretion. Ambient environmental conditions and physiological adaptations regulate the rate of water loss. Animals can meet daily water requirements through dietary water content and metabolic water production, as well as by drinking water. Calder and Braun (1983) developed allometric equations for drinking water ingestion rates for birds and mammals, and these equations are used for food web exposure models. This risk assessment follows the assumption of Sprenger et al. (1997) that birds and mammals can use saline water at the Site as a source of drinking water unless information to the contrary is available from scientific literature. Allometric equations are not available for drinking water rates for reptiles, and it is assumed for this risk assessment that diamondback terrapins (the selected reptilian receptor) do not ingest site water.

5.3.2.6 Receptor Body Weight

Body weights used in the exposure models presented below are based on mean body weights of females derived from populations in the closest reported geographic proximity to the LCP Site, thereby minimizing bias caused by clinal (latitudinal) variation. Body weights of females are used because studies used in this ERA report to derive TRVs in test species measured female-specific endpoints (e.g., decreased fecundity and decreased egg hatching success).

5.3.2.7 Chemical Bioavailability

Chemical analyses of sediments and prey tissue measure the total concentration of chemicals but not necessarily the amount of chemicals bioavailable to biota, which may be much smaller. Sediments tend to bind both mercury and PCBs and may retain them even in the digestive system of receptors, so that some of the ingested chemical dose is excreted without prior absorption from the digestive tract (McKim 1994). Bioavailability is limited by binding effects, and risks estimated from sediment doses without accounting for

TABLE 5-9. SUMMARY OF AREA-WEIGHTED AVERAGE SEDIMENT CONCENTRATIONS FOR MERCURY AND AROCLOR® 1268 IN CHANNEL AND NON-CHANNEL SAMPLES AT THE LCP SITE.

Area	Mercury (mg/kg, dry weight)	Aroclor® 1268 (mg/kg, dry weight)
Non-channel only	2.8	2.9
Channel only	1.5	2.5
Non-channel and channel, combined	2.5	2.9
Zone 1[a]	11.5	5.3
Zone 2 (location of LCP Transect 3)[a]	2.8	3.4
Zone 3 (location of LCP Transect 2)[a]	2.2	1.9
Zone 4 (location of LCP Transect 1)[a]	1.7	3.8

[a] Geographic extent of zones is defined in Figure 3 of Appendix D.

incremental bioavailability are highly uncertain (Alexander 1995), although the Agency for Toxic Substances and Disease Registry believes ". . . it is apparent that the default assumption of 100% relative bioavailability for mercury-contaminated soils is excessively conservative" (Canady et al. 1997). The conservative assumption that chemicals in the field are as equally bioavailable as chemicals in laboratory studies is retained in this ERA in the absence of data on relative bioavailability.

5.3.2.8 Mercury Speciation

Organomercurial species, such as methylmercury, are more toxic to organisms than inorganic forms of mercury. For a number of receptors, mercury exposure is largely associated with ingestion of sediments and lower trophic-level prey, in which less toxic and less available forms of mercury predominate. Using generic toxicity thresholds (i.e., TRVs) that are based on methylmercury overestimates risk if total mercury in sediment is used in quantifying the exposure dose. A more accurate approach to deriving and applying toxicity thresholds is to use methylmercury-based thresholds for all wildlife receptors and use the measured methylmercury fraction in sediments and prey tissue as the exposure dose. In this way, site-specific data on mercury speciation is matched with its corresponding toxicity thresholds and combined to quantify risks. This approach is used in food-web exposure models presented below.

A subset of sediment samples collected during this investigation was analyzed for total mercury and methylmercury concentrations. The maximum concentration of methylmercury as a percentage of total mercury in any marsh or channel surface sediment sample from the Site with an elevated total mercury concentration was 0.59 percent. Therefore, for food-web exposure model calculations, the sediment methylmercury concentration is estimated as 0.59 percent of the measured total mercury concentration. This is a conservative assumption because the fraction of total mercury in marsh and channel surface (0–2 cm) sediment represented by methylmercury is inversely related to the concentration of total mercury. Site marsh samples with the highest total mercury concentrations (551 and 972 mg/kg) contained the lowest percentage of methylmercury (0.01 percent). Conversely, sediment from reference areas with the lowest total mercury concentrations (0.08–0.09 mg/kg) exhibited the highest percentage of methylmercury (2.6–3.1 percent) observed in this study. A similar inverse relationship of percent methylmercury to total mercury has also been observed by Bloom (pers. comm.) in an evaluation of approximately 400 estuarine sediment samples.

Whenever measurements of methylmercury were available for prey species, those values were used in the exposure model (Table 5-10). If methylmercury concentrations were not available, as in the case of grasshoppers, total mercury concentrations were used as a conservative estimate of the methylated form because the relative proportion of methylmercury generally increases through a food web (Becker and Bigham 1995).

TABLE 5-10. CONCENTRATIONS OF CHEMICALS OF POTENTIAL CONCERN IN PREY SPECIES USED IN WILDLIFE FOOD-WEB EXPOSURE MODELING FOR THE LCP SITE

Prey Species	Station	Aroclor® 1268	Total Mercury	Methyl-mercury
Cordgrass	T-1	0.056 [a]	0.19	0.010
	T-2	0.032	0.17	0.001
	T-3	0.030 [a]	0.11	0.002
	Station mean	0.039	0.16	0.004
Periwinkle	T-1	0.078	0.15	0.032
	T-2	0.089	0.20	0.027
	T-3	0.038	0.21	0.021
	Site mean	0.068	0.19	0.027
Fiddler crab	T-1	1.19	0.44	0.31
	T-2	1.53	0.50	0.28
	T-3	0.81	0.27	0.20
	Site mean	1.18	0.40	0.26
Blue crab[b] (edible tissue)	Downstream Purvis Creek (EPA)	1.63	10.09	9.08
	Upstream Purvis Creek (EPA)	2.03	7.15	6.44
	Site mean	1.83	8.62	7.76
Oyster	SW-01	0.20	3.48	1.80
	SW-02	0.30	2.03	0.83
	SW-03	0.30	2.14	1.31
	SW-04	0.10	2.30	1.39
	SW-05	0.09	0.95	0.56
	SW-06	0.06	0.58	0.49
	Site mean	0.18	1.91	1.06
Brown shrimp[c] (edible tissue)	Downstream Purvis Creek (EPA)	0.64	0.40	0.36
	Upstream Purvis Creek (EPA)	0.97	0.52	0.47
	Site mean	0.81	0.46	0.41
Killifish[d]	LCP 43 (EPA)	3.80	0.99	0.79
Spot[e] (fillets)	Downstream Purvis Creek (EPA)	1.34	1.13	0.90
	Upstream Purvis Creek (EPA)	2.77	1.50	1.20
	Site mean	2.06	1.31	1.05
Insect	T-2	--	0.11	0.019
Grasshopper	M-1 (EPA)	0.76	--	--
	M-2 (EPA)	0.52	--	--
	Site mean	0.64	--	--

Note: Concentrations reported as mg/kg dry weight.

[a] Undetected, concentration estimated as one-half the detection limit.

[b] Assume methylmercury comprises 90 percent of total mercury for blue crabs from Purvis Creek based on two samples reported by Sprenger et al. (1997) that were analyzed for total mercury and methylmercury.

[c] Assume methylmercury comprises 90 percent of total mercury for brown shrimp from Purvis Creek based on one sample reported by Sprenger et al. (1997) that was analyzed for total mercury and methylmercury.

[d] Assume methylmercury comprises 80 percent of total mercury for killifish from Station LCP 43 based on average of 2 samples reported by Sprenger et al. (1997) that were analyzed for total mercury and methylmercury.

[e] Assume methylmercury comprises 80 percent of total mercury for spot from Purvis Creek based on one sample reported by Sprenger et al. (1997) that was analyzed for total mercury and methylmercury.

5.3.2.9 Life History Characteristics of Receptor Species

Species-specific life history characteristics that are used in assessing chemical exposure through the food web are discussed in the following sections.

West Indian Manatee—Manatees are herbivores. In coastal southeastern Georgia, their primary food is smooth cordgrass (Baugh et al. 1989). Average food ingestion rate is 7.42 kg/day dry weight, as estimated from the allometric equation presented by Nagy (1987) for herbivorous mammals and an average female body weight of 449 kg (Silva and Downing 1995). Baugh et al. (1989) indicate that manatees grazing on cordgrass bite off only the tips of blades, leaving one-third to two-thirds of the original length of the blade. Therefore, incidental sediment ingestion is unlikely to be a major exposure route. A sediment ingestion rate estimated at 2 percent of the food ingestion rate, or 0.15 kg/day, is used in the exposure model as a highly conservative estimate of the amount of sediment that may adhere to cordgrass blades eaten by manatees. Manatees move into regions of fresh water to drink, so ingestion of water is not expected at the Site because of the brackish water present.

Individual West Indian manatees may wander widely (>100 km) during the breeding season. Individual manatees are known to undertake multiple migrations along the Georgia coast and into Florida during a single year (Head, no date). Therefore, an area use factor of 1 for the manatee is an unrealistic overestimate of site use. As a conservative overestimate, an area use factor of 0.1 is used in exposure model calculations. Manatees are extremely sensitive to water temperatures lower than 20°C, and they typically migrate south from Georgia to Florida for the winter. Most manatee sightings in Camden, Glynn, and McIntosh counties, Georgia, occur from May to October (Head, no date). Therefore, a conservative time use factor of 0.5 is applied to the food-web exposure model to account for the lack of year-round presence of manatees at Brunswick. Manatees are believed to actually visit the Site only rarely.

Manatee foraging is restricted to the banks of tidal creeks, preferably with water depths of 1–3 m (Head, no date). Therefore, the exposure model uses mean concentrations of methylmercury and Aroclor® 1268 in cordgrass collected from transect sampling stations T-1, T-2, and T-3 to reflect the average chemical concentrations to which an individual foraging manatee could potentially be exposed (Table 5-10). Methylmercury and Aroclor® 1268 concentrations in sediments potentially ingested by manatees are determined as the average of values for channel locations only (Table 5-9). The mercury concentration reported in Table 5-9 is multiplied by 0.0059 to estimate the methylmercury concentration. Inclusion of sediment ingestion in the exposure model adds conservatism because the manatee behavior (i.e., eating cordgrass tops) minimizes sediment ingestion.

River Otter—River otters in coastal regions primarily consume fishes, however, they also consume smaller amounts of crustaceans and aquatic insects (U.S. EPA 1993c).

The diet of the river otter at the LCP Site marsh is assumed to be 35 percent spot, 35 percent killifish (*Fundulus* sp.), 15 percent fiddler crab, and 15 percent blue crab. Average food ingestion rate is 0.33 kg/day dry weight, as estimated from the allometric equation presented by Nagy (1987) for all mammals and an average female body weight of 6.7 kg for female otters in Alabama and Georgia (Lauhachinda 1978). The sediment ingestion rate is estimated at 2 percent of the food ingestion rate, or 0.0066 kg/day. Water ingestion is estimated to be 0.55 L/day, based on allometric scaling of body size (U.S. EPA 1993c).

Foy (1984) reported that female river otters in coastal marshes in southeast Texas had home ranges of 295 ha. Based on this value, the LCP Site marsh (195 ha) equals 66 percent of the total foraging range, and an area use factor of 0.66 is appropriate for female otters. Otters are presumed to forage over the entire Site, including banks of tidal channels and the upper parts of the marsh. Therefore, body burdens of chemicals in prey species are modeled as a site mean to reasonably reflect site exposure (Table 5-10). Methylmercury and Aroclor® 1268 concentrations in sediments potentially ingested by otters are determined as the average of values for channel and non-channel locations combined (Table 5-9). The mercury concentration reported in Table 5-9 is multiplied by 0.0059 to estimate the methylmercury concentration. Concentrations of methylmercury and Aroclor® 1268 in water potentially ingested by river otters (and all other receptors) are estimated as the mean concentrations in surface water sampling stations at the Site (Stations 1–6, Table 5-1). Average total methylmercury concentrations for these stations is 1.35×10^{-6} mg/L. Aroclor® 1268 was not detected at any station; therefore, half the detection limit of 0.001 mg/L is used as a representative value.

Raccoon—Raccoons can inhabit a broad range of areas, and their prey items reflect this diversity. A study on St. Catherine's Island, Georgia, indicates that crustaceans, particularly fiddler crabs, are an important food item for raccoons in salt marshes (75 percent of total intake), with mussels and smooth cordgrass constituting most of the remainder of their diet (Harman and Stains 1979). The same study also found that raccoons foraged only 31 percent of the time in salt marshes, with the remainder of the time spent in wooded areas (54 percent) and grass/scrub areas (15 percent). Therefore, raccoons at the LCP Site are assumed to consume fiddler crabs (75 percent of diet), oysters (20 percent), and smooth cordgrass (5 percent) from the marsh, but to spend only 30 percent of their foraging time in the marsh, for an area use factor of 0.3. Average food ingestion rate is 0.16 kg/day dry weight, as estimated from the allometric equation presented by Nagy (1987) for all mammals and an average female body weight of 2.78 kg (Silva and Downing 1995). The sediment ingestion rate is estimated at 9.4 percent of the food ingestion rate (Beyer et al. 1994), or 0.015 kg/day. Water ingestion is estimated to be 0.25 L/day based on allometric scaling of body size (U.S. EPA 1993c).

The home range of female raccoons on St. Catherine's Island is approximately 40 ha, which is smaller than the area of the marsh at the LCP Site. However, as noted above, marsh habitats only comprise about 30 percent of the total home range of raccoons in

coastal regions of Georgia. Raccoons could potentially use different areas of the Site, and risk may differ among areas. The potential for unacceptable risk to raccoons is therefore modeled using chemical residue data in prey species collected from prey transects in different parts of the LCP Site (see Figure 4-6, Table 5-10). Prey species collected along Transect 2 are used to estimate the potential for unacceptable risk to raccoons with a home range that includes the section of the marsh north of the causeway and east of Purvis Creek. Prey species collected along Transect 3 are used to estimate the potential for unacceptable risk to raccoons foraging in the southwestern quadrant of the marsh. Oysters were not collected at Transects 2 or 3; therefore, chemical concentrations in tissue of native oysters collected at six locations (Stations SW-01, SW-02, SW-03, SW-04, SW-05, and SW-06) during the surface water investigation were used as an estimate of concentrations in oysters for both these locations. Metals data for native oysters are presented in Table 1.1-4 of Volume II of this ERA, and Aroclor® 1268 data are presented in Table 1.1-6. The potential for unacceptable risk is not evaluated at Transect 1 because it is largely separated from terrestrial habitats by the meandering tidal channels and therefore expected that raccoons will not use this area of the marsh.

To quantify exposure through the sediment ingestion route, methylmercury and Aroclor® 1268 concentrations in sediment at the transect locations are estimated as the average concentration for the zone of the marsh, as identified in Appendix D, within which that transect is located .For Transect 2, the values from Zone 3 (Table 5-9) are used. For Transect 3, the values from Zone 2 (Table 5-9) are used. In all cases, the mercury concentration reported in Table 5-9 is multiplied by 0.0059 to estimate the methylmercury concentration.

Wood Stork—Wood storks are colonial nesting wading birds. No wood stork colonies are located at the LCP Site, but adults from several colonies in the vicinity of Brunswick forage at the LCP Site. Wood storks are tactile feeders that feed primarily on small to medium size fish (2–25 cm), although they also occasionally consume crustaceans and arthropods (Ogden et al. 1976). The diet of wood storks foraging at the LCP Site marsh is presumed to be 45 percent spot, 45 percent killifish, 5 percent fiddler crab, and 5 percent brown shrimp. Average food ingestion rate is 0.092 kg/day dry weight, as estimated from the allometric equation presented by Nagy (1987) for non-passerines and an average female body weight of 2.05 kg (Dunning 1993). The sediment ingestion rate is estimated at 2 percent of the food ingestion rate, or 0.0018 kg/day, based on Beyer et al. (1994). Water ingestion is estimated at 0.095 L/day, based on allometric scaling of body size (U.S. EPA 1993c).

Wood storks use foraging locations within a 50-km radius of their colony site (USFWS 1996), which equals a potential foraging range of 0.79 million ha. The nearest colony to the LCP Site is located 5.5 km away on St. Simon's Island (Bryan 1996, pers. comm.). Because this is a coastal colony, approximately half the potential foraging range is anticipated to be open ocean and, therefore, not suitable habitat. This would leave approximately 0.40 million ha of potential foraging habitat. The marsh at the LCP Site covers

195 ha, or approximately 0.05 percent of the total foraging range. Even allowing for non-suitable habitat within that range (urban areas, non-wetland areas), the LCP Site is unlikely to encompass more than 1 percent of the total foraging range of a nesting wood stork (Bryan 1996, pers. comm.). Therefore, an area use factor of 0.01 is used in the wood stork exposure model.

Wood storks at the LCP Site forage primarily along the banks of tidal creeks at low tides (Bryan 1996, pers. comm.). Therefore, in the exposure model, mean concentrations of methylmercury and Aroclor® 1268 in sediment samples collected from channel stations are used to reflect the mean chemical levels to which an individual foraging wood stork could potentially be exposed (Table 5-9). Mercury concentrations in Table 5-9 are multiplied by 0.0059 to estimate methylmercury concentrations.

Clapper Rail—Clapper rails are common inhabitants of Atlantic coast salt marshes. Clapper rails feed primarily on fiddler crabs and periwinkles (Heard 1982). Diet for clapper rails in the LCP Site marsh is presumed to be 75 percent fiddler crabs, 15 percent periwinkles, and 10 percent terrestrial insects. Average food ingestion rate is 0.020 kg/day dry weight, as estimated from the allometric equation presented by Nagy (1987) for non-passerines and an average female body weight of 0.27 kg (Dunning 1993). The rate of incidental soil ingestion by clapper rails during foraging is not known, but because of their probing foraging technique, a rate of 10 percent of the daily food intake is estimated based on other avian species with similar foraging techniques, as listed by Beyer et al. (1994). Water ingestion is estimated to be 0.025 L/day, based on allometric scaling of body size (U.S. EPA 1993c).

The home range size of the clapper rail varies across different parts of its geographic range. A home range 274 m in diameter (6 ha) was reported for clapper rails in South Carolina (Blandin 1963), and this estimate is used for clapper rails at the LCP Site marsh. Because this value is smaller than the size of the LCP Site, an area use factor of 1 was selected for clapper rails. However, the small home range also indicates that individual clapper rails would be limited to subsections of the Site and that different risk estimates would apply for birds inhabiting varying parts of the marsh. Therefore, the potential for unacceptable risk to clapper rails is modeled using chemical residue data in prey species collected from prey sampling transects in different parts of the LCP Site (see Figure 4-6, Table 5-10). Prey species collected along Transect 1 are used to estimate the potential for unacceptable risk to clapper rails inhabiting sections of the marsh west of Purvis Creek. Prey species collected along Transect 2 are used to estimate the potential for unacceptable risk to clapper rails with a home range that includes the section of the marsh north of the LCP causeway and east of Purvis Creek. Prey species collected along Transect 3 are used to estimate the potential for unacceptable risk to clapper rails foraging in the southwestern quadrant of the marsh. Terrestrial insects were not collected at Transects 1 or 3; therefore, chemical concentrations in tissue of insects collected at Transect 2 were used as a conservative estimate of concentrations in insects for both of these locations. PCB concentrations were not measured in any insects collected from the marsh transects as part

of this study. Therefore, the average concentration of Aroclor® 1268 in two samples of grasshoppers (0.64 mg/kg dry wt) collected from the marsh by EPA was used as an estimate of PCB concentrations in insects from all three transect locations.

Methylmercury and Aroclor® 1268 concentrations in ingested sediment at the transect locations are estimated as the average concentration for the zone of the marsh within which that transect is located. Concentrations from Zone 4 are used for Transect 1, concentrations from Zone 3 are used for Transect 2, and concentrations from Zone 2 are used for Transect 3 (Table 5-9). In all cases, mercury concentrations are multiplied by 0.0059 to estimate methylmercury concentrations.

Marsh Wren—Marsh wrens are common inhabitants of salt marshes, and densities up to 48 pairs/ha have been reported in Georgia salt marshes (Kale 1965). The diet of the marsh wren consists almost entirely of terrestrial insects, with mollusks and arthropods forming the remainder. For the food-web exposure model, the diet is assumed to be 95 percent insects and 5 percent periwinkles. Average food ingestion rate is 0.0027 kg/day dry weight, as estimated from the allometric equation presented by Nagy (1987) for passerines and an average female body weight of 0.00941 kg (Kale 1965). Incidental soil ingestion is estimated as 1 percent of the food ingestion rate because wrens forage extensively among the blades of cordgrass, which minimizes potential soil ingestion. Water ingestion is estimated to be 0.0026 L/day, based on allometric scaling of body size (U.S. EPA 1993c).

During the breeding season, the home range of the marsh wren is approximately 0.006–0.17 ha (Kale 1965). Birds may range more widely during the winter, but still use an area smaller than the size of the LCP Site; therefore, an area use factor of 1 was selected for this species. The small home range also indicates, however, that individual wrens would be limited to subsections of the marsh at the Site and that different risk estimates would apply for birds in various parts of the marsh. Therefore, the potential for unacceptable risk to wrens is modeled using chemical residue data in prey species collected from prey transects in different parts of the Site following the procedure described in the preceding section for clapper rails. Sediment concentrations are estimated following the procedure described in the preceding section for clapper rails.

Diamondback Terrapin—The diamondback terrapin is a common inhabitant of Atlantic coast salt marshes. The diet of terrapins consists primarily of mollusks and crustaceans. Tucker et al. (1995) reported that 76–79 percent of the diet by volume consisted of the salt marsh periwinkle, with the remainder being fiddler crabs, blue crabs, and bivalve molluscs. For the exposure model, the diet is considered to be 78 percent periwinkles, 12 percent fiddler crabs, 5 percent blue crabs, and 5 percent oysters. Average food ingestion rate is 0.0006 kg/day dry weight, as estimated from the allometric equation presented by Nagy (1987) for insectivorous iguanid lizards and an average body weight of 0.143 kg reported by Allen and Littleford (1955). Nagy did not develop allometric scaling models

for other types of reptiles; therefore, the equation for iguanid lizards is used as the best available estimate of food ingestion by terrapins. Incidental sediment ingestion is estimated at 6 percent by weight of food intake, which is based on the closest approximation of 5.9 percent for the eastern painted turtle (*Chrysemys picta*) (Beyer et al. 1994). Terrapins are assumed not to ingest site water.

The foraging range of the diamondback terrapin has not been extensively studied. Terrapins use estuarine areas as foraging habitat and upland areas as nesting habitat. Nesting terrapins have been reported to travel from 0.25 km (Palmer and Cordes 1988) to 8 km (Hurd et al. 1979). Based on this range of values, it is likely that individual terrapins may remain entirely within the marsh at the LCP Site. Therefore, an area use factor of 1 is used for diamondback terrapins in the food-web exposure model.

5.4 WILDLIFE FOOD-WEB EXPOSURE ESTIMATES

Results of the wildlife food-web exposure models are presented in Tables 5-11 through 5-17. These estimates are for ingestion of CoPCs through the dietary pathway based on exposure data and assumptions as summarized above.

5.5 PREVIOUS INVESTIGATIONS OF BODY BURDENS IN WILDLIFE AT THE LCP SITE

Sprenger et al. (1997) present results of body-burden analyses for clapper rails and diamondback terrapins collected at the LCP Site. Body-burden analyses offer an indication of the extent of exposure to CoPCs that is occurring in wildlife receptors; although by themselves, body-burden analyses do not indicate the likelihood of occurrence of adverse effects resulting from exposure. A complete description of the body-burden results is presented by Sprenger et al. (1997), and a summary is included here to provide some insight into exposure and to supplement results provided by the food-web exposure estimates described above. All values are expressed on a dry-weight basis.

5.5.1 Clapper Rail

Seven clapper rails were collected in the south marsh at the LCP Site, and seven birds were collected at a reference site. Breast muscle, carcass, and liver were analyzed for mercury and PCBs (specifically Aroclor® 1268), and feathers were analyzed for mercury only. Mercury was detected in every tissue analyzed, both in birds from the LCP Site and birds from the reference area. Rails from the LCP Site had a mean mercury concentration of 5.1 mg/kg in breast muscle, 5.08 mg/kg in the rest of the carcass, 15.6 mg/kg in liver, and 11.25 mg/kg in feathers. Rails from the reference area had a mean mercury concentration of 1.60 mg/kg in breast muscle, 1.08 mg/kg in the rest of the carcass, 3.47 mg/kg in liver, and 3.6 mg/kg in feathers. The mean concentration of PCBs was 98.2 mg/kg in breast muscle of clapper rails from the LCP Site, 27.82 mg/kg in carcass, and 25.2 mg/kg

TABLE 5-11. EXPOSURE VALUES FOR FOOD-WEB MODELING OF POTENTIAL FOR UNACCEPTABLE RISK TO WEST INDIAN MANATEE IN THE MARSH AT THE LCP SITE

WEST INDIAN MANATEE (*Trichechus manatus*)

Body weight (kg)	449	Sediment ingestion rate (kg/day)	0.15
Food ingestion rate (kg/day)	7.42	Water ingestion rate (L/day)	0
Food Item	Percent of diet	Area Use Factor	0.1
Cordgrass	100	Time Use Factor	0.5

		Sediment		Cordgrass		Total Food	Total
Chemical	Location	Conc. (mg/kg)	Exp. (mg/kg-day)	Conc. (mg/kg)	Exp. (mg/kg-day)	Exp. (mg/kg-day)	Exp. (mg/kg-day)
Polychlorinated biphenyls (Aroclor® 1268 concentration)	Entire marsh (channels only)	2.5	0.00004	0.039	0.00003	0.00003	0.00007
Mercury (methylmercury concentration)	Entire marsh (channels only)	0.009	1.5×10^{-7}	0.004	0.000003	0.000003	0.000004

Note: All sediment and tissue concentrations are expressed as dry weight values.

TABLE 5-12. EXPOSURE VALUES FOR FOOD-WEB MODELING OF POTENTIAL FOR UNACCEPTABLE RISK TO RIVER OTTER IN THE MARSH AT THE LCP SITE

RIVER OTTER (*Lutra canadensis*)

Body weight (kg)	6.7	Sediment ingestion rate (kg-day)	0.0066
Food ingestion rate (kg-day)	0.33	Water ingestion rate (L/day)	0.55
		Area Use Factor	0.66

Food Item	Percent of diet
Spot	35
Killifish	35
Fiddler crabs	15
Blue crabs	15

		Sediment		Water		Spot		Killifish		Fiddler Crab		Blue Crab		Total Food	Total
Chemical	Location	Conc. (mg/kg)	Exp. (mg/kg-day)	Conc. (mg/L)	Exp. (mg/kg-day)	Conc. (mg/kg)	Exp. (mg/kg-day)	Conc. (mg/kg)	Exp. (mg/kg-day)	Conc. (mg/kg)	Exp. (mg/kg-day)	Conc. (mg/kg)	Exp. (mg/kg-day)	Exp. (mg/kg-day)	Exp. (mg/kg-day)
Polychlorinated biphenyls (Aroclor® 1268 concentration)	Entire marsh	2.9	0.002	0.0005	2.70×10^{-5}	2.06	0.023	3.80	0.043	1.18	0.006	1.83	0.0089	0.081	0.083
Mercury (methylmercury concentration)	Entire marsh	0.015	0.00004	1.35×10^{-6}	7.29×10^{-8}	1.05	0.012	0.79	0.009	0.26	0.0013	7.76	0.038	0.060	0.060

Note: All sediment and tissue concentrations are expressed as dry weight values.

TABLE 5-13. EXPOSURE VALUES FOR FOOD-WEB MODELING OF POTENTIAL FOR UNACCEPTABLE RISK TO RACCOON IN THE MARSH AT THE LCP SITE

RACCOON (*Procyon lotor*)

Body weight (kg)	2.78	Sediment ingestion rate (kg/da	0.015
Food ingestion rate (kg day)	0.16	Water ingestion rate (L/day)	0.25
		Area Use Factor	0.3

Food Item	Percent of diet
Fiddler crabs	75
Oysters	20
Cordgrass	5

		Sediment		Water		Fiddler Crab		Oyster		Cordgrass		Total Food	Total
Chemical	Location	Conc. (mg/kg)	Exp. (mg/kg-day)	Conc. (mg/L)	Exp. (mg/kg-day)	Conc. (mg/kg)	Exp. mg/kg-day)	Conc. (mg/kg)	Exp. (mg/kg-day)	Conc. (mg/kg)	Exp. (mg/kg-day)	Exp. (mg/kg-day)	Exp. (mg/kg-day)
Polychlorinated biphenyls (Aroclor® 1268 concentration)	LCP Transect 2	1.9	0.0031	0.0005	1.34×10^{-5}	1.53	0.020	0.18	0.0006	0.032	0.00003	0.020	0.024
	LCP Transect 3	3.4	0.0055	0.0005	1.34×10^{-5}	0.81	0.010	0.18	0.0006	0.03	0.00003	0.011	0.017
Mercury (methylmercury concentration)	LCP Transect 2	0.013	2.1×10^{-5}	1.35×10^{-6}	3.62×10^{-8}	0.28	0.0036	1.06	0.0036	0.0010	8.59×10^{-7}	0.0073	0.0073
	LCP Transect 3	0.017	2.7×10^{-5}	1.35×10^{-6}	3.62×10^{-8}	0.20	0.0026	1.06	0.0036	0.0020	1.72×10^{-6}	0.0062	0.0062

Note: All sediment and tissue concentrations are expressed as dry weight values.

TABLE 5-14. EXPOSURE VALUES FOR FOOD-WEB MODELING OF POTENTIAL FOR UNACCEPTABLE RISK TO WOOD STORK IN THE MARSH AT THE LCP SITE

WOOD STORK (*Mycteria americana*)

Body weight (kg)	2.05	Sediment ingestion rate (kg/day)	0.0018
Food ingestion rate (kg/day)	0.092	Water ingestion rate (L/day)	0.095
Food Item	Percent of diet	Area Use Factor	0.01
Spot	45		
Killifish	45		
Fiddler crabs	5		
Shrimp	5		

		Sediment		Water		Spot		Killifish		Fiddler Crab		Shrimp		Total Food	Total
Chemical	Location	Conc. (mg/kg)	Exp. (mg/kg-day)	Conc. (mg/L)	Exp. (mg/kg-day)	Conc. (mg/kg)	Exp. (mg/kg-day)	Conc. (mg/kg)	Exp. (mg/kg-day)	Conc. (mg/kg)	Exp. (mg/kg-day)	Conc. (mg/kg)	Exp. (mg/kg-day)	Exp. (mg/kg-day)	Exp. (mg/kg-day)
Polychlorinated biphenyls (Aroclor® 1268 concentration)	Entire marsh (channels only)	2.5	2.3×10^{-5}	0.0005	2.3×10^{-7}	2.06	0.0004	3.80	0.0008	1.18	0.00003	0.81	1.8×10^{-5}	0.0012	0.0013
Mercury (methylmercury concentration)	Entire marsh (channels only)	0.009	8.0×10^{-8}	1.35×10^{-6}	6.3×10^{-10}	1.05	0.0002	0.79	0.0002	0.26	5.9×10^{-6}	0.41	9.3×10^{-8}	0.0004	0.0004

Note: All sediment and tissue concentrations are expressed as dry weight values.

TABLE 5-15. EXPOSURE VALUES FOR FOOD-WEB MODELING OF POTENTIAL FOR UNACCEPTABLE RISK TO CLAPPER RAIL IN THE MARSH AT THE LCP SITE

CLAPPER RAIL (*Rallus longirostris*)

Body weight (kg)	0.27	Sediment ingestion rate (kg/day)	0.0020
Food ingestion rate (kg/day)	0.020	Water ingestion rate (L/day)	0.025
		Area Use Factor	1

Food Item	Percent of diet
Fiddler crabs	75
Periwinkles	15
Terrestrial insects	10

		Sediment		Water		Fiddler Crab		Periwinkle		Insect		Total Food	Total
Chemical	Location	Conc. (mg/kg)	Exp. (mg/kg-day)	Conc. (mg/L)	Exp. (mg/kg-day)	Conc. (mg/kg)	Exp. (mg/kg-day)	Conc. (mg/kg)	Exp. (mg/kg-day)	Conc. (mg/kg)	Exp. (mg/kg-day)	Exp. (mg/kg-day)	Exp. (mg/kg-day)
Polychlorinated biphenyls (Aroclor® 1268 concentration)	LCP Transect 1	3.8	0.028	0.0005	4.6×10^{5}	1.19	0.07	0.078	0.0009	0.64	0.005	0.072	0.10
	LCP Transect 2	1.9	0.014	0.0005	4.6×10^{5}	1.53	0.09	0.089	0.0010	0.64	0.005	0.091	0.11
	LCP Transect 3	3.4	0.025	0.0005	4.6×10^{-5}	0.81	0.05	0.038	0.0004	0.64	0.005	0.054	0.08
Mercury (methylmercury concentration)	LCP Transect 1	0.010	0.0001	1.35×10^{6}	1.2×10^{-7}	0.31	0.02	0.032	0.0004	0.019	0.0001	0.018	0.018
	LCP Transect 2	0.013	0.0001	1.35×10^{6}	1.2×10^{-7}	0.28	0.02	0.027	0.0003	0.019	0.0001	0.016	0.016
	LCP Transect 3	0.017	0.0001	1.35×10^{6}	1.2×10^{-7}	0.20	0.01	0.021	0.0002	0.019	0.0001	0.012	0.012

Note: All sediment and tissue concentrations are expressed as dry weight values.

TABLE 5-16. EXPOSURE VALUES FOR FOOD-WEB MODELING OF POTENTIAL FOR UNACCEPTABLE RISK TO MARSH WREN IN THE MARSH AT THE LCP SITE

MARSH WREN (*Cistothorus palustris*)

Body weight (kg)	0.00941	Sediment ingestion rate (kg/da	0.000027
Food ingestion rate (kg/day)	0.0027	Water ingestion rate (L/day)	0.0026
		Area Use Factor	1

Food Item	Percent of diet
Terrestrial insects	95
Periwinkles	5

		Sediment		Water		Insect		Periwinkle		Total Food	Total
Chemical	Location	Conc. (mg/kg)	Exp. (mg/kg-day)	Conc. (mg/L)	Exp. (mg/kg-day)	Conc. (mg/kg)	Exp. (mg/kg-day)	Conc. (mg/kg)	Exp. (mg/kg-day)	Exp. (mg/kg-day)	Exp. (mg/kg-day)
Polychlorinated biphenyls (Aroclor® 1268 concentration)	LCP Transect 1	3.8	0.011	0.0005	1.4×10^{-4}	0.64	0.17	0.078	0.0011	0.17	0.19
	LCP Transect 2	1.9	0.005	0.0005	1.4×10^{-4}	0.64	0.17	0.089	0.0013	0.17	0.18
	LCP Transect 3	3.4	0.010	0.0005	1.4×10^{-4}	0.64	0.17	0.038	0.0005	0.17	0.18
Mercury (methylmercury concentration)	LCP Transect 1	0.010	2.9×10^{-5}	1.35×10^{-6}	3.7×10^{-7}	0.019	0.0052	0.032	0.0005	0.006	0.006
	LCP Transect 2	0.013	3.7×10^{-5}	1.35×10^{-6}	3.7×10^{-7}	0.019	0.0052	0.027	0.0004	0.006	0.006
	LCP Transect 3	0.017	4.7×10^{-5}	1.35×10^{-6}	3.7×10^{-7}	0.019	0.0052	0.021	0.0003	0.006	0.006

Note: All sediment and tissue concentrations are expressed as dry weight values.

TABLE 5-17. EXPOSURE VALUES FOR FOOD-WEB MODELING OF POTENTIAL FOR UNACCEPTABLE RISK TO DIAMONDBACK TERRAPIN IN THE MARSH AT THE LCP SITE

DIAMONDBACK TERRAPIN (*Malaclemys terrapin*)

Body weight (kg)	0.143	Sediment ingestion rate (kg/day)	0.00004
Food ingestion rate (kg/day)	0.0006	Water ingestion rate (L/day)	0
		Area Use Factor	1

Food Item	Percent of diet
Periwinkles	78
Fiddler crabs	12
Blue crabs	5
Oysters	5

		Sediment		Periwinkle		Fiddler Crab		Blue Crab		Oyster		Total Food	Total
Chemical	Location	Conc. (mg/kg)	Exp. (mg/kg-day)	Conc. (mg/kg)	Exp. (mg/kg-day)	Conc. (mg/kg)	Exp. (mg/kg-day)	Conc. (mg/kg)	Exp. mg/kg-day)	Conc. (mg/kg)	Exp. (mg/kg-day)	Exp. (mg/kg-day)	Exp. (mg/kg-day)
Polychlorinated biphenyls (Aroclor® 1268 concentration)	Entire marsh	2.9	0.0007	0.068	0.0002	1.18	0.0006	1.83	0.00039	0.18	0.00004	0.0012	0.0020
Mercury (methylmercury concentration)	Entire marsh	0.015	3.7×10^{-6}	0.027	0.0001	0.26	0.0001	7.76	0.0016	1.06	0.0002	0.0021	0.0021

Note: All sediment and tissue concentrations are expressed as dry weight values.

in liver. For birds from the reference area, the mean concentration of PCBs was 0.84 mg/kg in breast muscle, 1.85 mg/kg in carcass, and 0.85 mg/kg in liver.

5.5.2 Diamondback Terrapin

Eight terrapins were collected from the vicinity of the LCP Site during two sampling periods in 1995. Carcass, liver, eggs, and hatchlings were analyzed for mercury and PCBs (specifically Aroclor® 1268). Aroclor® 1268 was detected in carcasses at concentrations ranging from 1.7 to 620 mg/kg, and in liver at concentrations ranging from 12 to 3,500 mg/kg. Mercury was detected in carcasses at concentrations ranging from 1.8 to 15 mg/kg, and in liver at concentrations ranging from 11 to 330 mg/kg. Eggs taken from gravid females had mean Aroclor® 1268 concentrations ranging from 28.6 to 477 mg/kg, and mean mercury concentrations ranging from 0.87 to 4.5 mg/kg. Hatchlings produced from one of the female terrapins had a mean Aroclor® 1268 concentration of 12.6 mg/kg and a mean mercury concentration of 2.1 mg/kg. A positive correlation was noted between the PCB concentration in reproductively active females and their eggs and young, but the relationship was less strong for mercury.

5.6 AERIAL SURVEYS OF WADING BIRD ABUNDANCE

Aerial surveys of wading bird abundance and spatial distribution at the LCP Site and at a reference site on Hawkins Creek were conducted by Larry Bryan of the Savannah River Ecology Laboratory, University of Georgia. Six species of wading birds were seen on both sites. Great egrets (*Casmerodius albus*), snowy egrets (*Egretta thula*), and wood storks were most commonly observed. Great blue herons (*Ardea herodia*) were consistently present, but in low numbers. White ibis (*Eudocimous albus*) and little blue herons (*Egretta caerulea*) were occasionally observed in high numbers, but their presence during surveys was infrequent. For all surveys (low and high tide), the three dominant wading bird species (great egrets, snowy egrets, and wood storks) and all six species combined were present in significantly higher numbers at the LCP Site than at the Hawkins Creek site. Surveys indicated that birds used tidal creeks almost exclusively, with few observations recorded in the vegetated marsh. All species except white ibis and little blue herons were present in significantly higher numbers during low tides than high tides at the LCP Site. At the LCP Site, wood storks were typically found in the smaller intertidal creeks, the confluence of those creeks with larger order creeks, and mud flat openings at the origins of the first order creeks. Wood storks were more commonly observed in the marsh north of the causeway than in the southern part of the marsh. They also exhibited a preference for the part of the LCP Site that is distant from industrial operations. A complete report on the wading bird aerial survey is presented in Appendix A.

6. EFFECTS ASSESSMENT

Potential adverse effects of CoPCs on aquatic and terrestrial ecological receptors at the LCP Site are evaluated in the following sections. Toxicity tests with site water and field bioaccumulation studies with native species are used to evaluate effects of CoPCs on water-column organisms. Site-specific data from toxicity tests of site sediment and a survey of macrobenthic species abundance are used to evaluate effects of CoPCs on benthic organisms. When possible, concentration-response (or dose-response) relationships are quantified based on site-specific data. Potential adverse effects on wildlife receptors are evaluated using food-web exposure models in the risk characterization section.

6.1 EVALUATION OF DIRECT TOXICITY OF CoPCs TO WATER COLUMN ORGANISMS

Direct toxicity of CoPCs to water-column organisms was evaluated in a laboratory toxicity study and in a field bioaccumulation study.

6.1.1 Laboratory Toxicity Study

The laboratory toxicity study used standard EPA protocols to estimate chronic toxicity of water-borne chemicals to marine organisms. The marine organisms used were mysids and sheepshead minnows. Early life stages of both species (juvenile mysids and embryonic/larval sheepshead minnows) were used. Measurement endpoints for these tests are survival, growth, and sometimes fecundity of mysids, as well as survival and teratogenicity for sheepshead minnows.

6.1.1.1 Mysids

Mysids exposed for 7 days to ambient surface water (channel water) from the nine study area stations and two reference stations were characterized by survival (Table 6-1) that ranged from 78 percent (Turtle River near the mouth of Purvis Creek and the Clubbs Creek reference station) to 90 percent (upper range of Turtle River). The survival rate of control organisms was 88 percent, which is greater than the specified value of 80 percent for acceptability of test results (U.S. EPA 1988b). Survival of mysids exposed to water from study area stations, including stations with mercury concentrations greater than the Georgia standard (25 ng/L), was not significantly lower than survival of reference or control organisms. Similarly, there were no statistically significant reductions in growth (weight) of organisms exposed to water from study area stations as compared to reference or control organisms (Table 6-2). Mean weight of control organisms was 0.248 mg, as

TABLE 6-1. STATISTICAL ANALYSIS OF SURVIVAL OF MYSIDS EXPOSED TO SURFACE WATER FROM THE LCP SITE AND REFERENCE STATIONS

1. Number of Survivors

Sampling Station	Replicate (r)[a] 1	2	3	4	5	6	7	8	Mean ($\bar{x}$)	Variance (s^2)
Laboratory Control Water	4	4	4	4	5	4	5	5	4.38 (88%)	0.27
LCP Site Stations										
LCP Ditch (No. 1)	4	5	5	5	5	3	3	3	4.12 (82%)	0.98
Purvis Creek headwaters (No. 4)	4	5	3	4	5	5	4	5	4.38 (88%)	0.55
Upper Purvis Creek (No. 3)	5	4	5	4	4	4	5	3	4.25 (85%)	0.50
Middle Purvis Creek (No. 2)	4	4	4	4	4	5	4	5	4.25 (85%)	0.21
Lower Purvis Creek (No. 5)	5	4	4	5	4	4	5	4	4.38 (88%)	0.27
Gibson Creek (No. 9)	5	5	5	5	4	2	4	5	4.38 (88%)	1.13
Turtle River – mouth of Purvis Creek (No. 6)	5	2	4	5	3	4	4	4	3.88 (78%)	0.98
Turtle River – upper range (No. 7)	4	5	5	5	5	5	5	2	4.50 (90%)	1.14
East River (No. 8)	3	5	4	4	4	5	4	5	4.25 (85%)	0.50
Reference Stations										
Clubbs Creek (No. 11)	4	4	4	4	5	3	3	4	3.88 (78%)	0.41
Jointer Creek (No. 10)	5	4	5	3	5	4	4	4	4.25 (85%)	0.50

2. Cochran's (C) Test for Homogeneity of Variances of Survival Data[b]

$$C(\text{cal.}) = \frac{s^2(\text{max.})}{\sum s^2} = \frac{1.14}{7.44} = 0.15 \text{ ns,}$$

as compared to C (tab.) = 0.23
for $P = 0.05$, k = 12, and v = 7

TABLE 6-1. (cont.)

3. Parametric One-Way Analysis of Variance (ANOVA) of Survival Data[b]

Source of Variation	Degrees of Freedom (df)	Sum of Squares (SS)	Mean Square (MS)	F (cal.)
Sampling stations (plus control)	t - 1 = 11	3.36	0.31	0.50 ns,
Error	t (r - 1) = 84 tr - 1 = 95	52.13 55.49	0.62	
			as compared to F (tab.) = 1.92 for P = 0.05, 11 numerator df, and 84 denominator df	

Note: Toxicity data are abstracted from Volume II, Section 2 and pertain to 7-day-old mysids exposed for 7 days to undiluted channel water collected from 11 sampling stations distributed throughout the study area and a laboratory control. Water was replaced daily during the study.

Channel water was collected from sampling stations on June 26 (Days 1 and 2 of study), June 28 (Days 3, 4, and 5), and July 1 (Days 6 and 7). Grab samples of water were collected about 0.5 m below the air/water interface from approximately 2 hours on either side of slack low water. The water column at sampling stations was well mixed, as evidenced by differences in subsurface vs. bottom salinity of less than 5 ppt (typically less than 1-2 ppt).

[a] Each replicate (r) consisted of five mysids at initiation of toxicity study.

[b] Cochran's test demonstrated that variances of survival data were not significantly different (as indicated by the symbol "ns" associated with the value for C [cal.]). Consequently, a parametric analysis of variance (ANOVA) was used to evaluate the significance of mean differences in survival among sampling stations. The ANOVA documented the absence of significant differences (indicated by the symbol "ns" associated with F [cal.]).

TABLE 6-2. STATISTICAL ANALYSIS OF GROWTH (WEIGHT) OF MYSIDS EXPOSED TO SURFACE WATER FROM THE LCP SITE AND REFERENCE STATIONS

1. Mean Weight (mg dry weight)

	Replicate (r)[a]								Mean	Variance
Sampling Station	1	2	3	4	5	6	7	8	$(\bar{x})$	(s^2)
Laboratory Control Water	0.23	0.27	0.21	0.26	0.23	0.24	0.29	0.25	0.248	0.0006
LCP Site Stations										
LCP Ditch (No. 1)	0.16	0.24	0.25	0.31	0.28	0.16	0.18	0.18	0.220	0.0033
Purvis Creek headwaters (No. 4)	0.17	0.19	0.19	0.21	0.28	0.27	0.25	0.31	0.234	0.0020
Upper Purvis Creek (No. 3)	0.28	0.25	0.28	0.21	0.26	0.22	0.23	0.11	0.230	0.0030
Middle Purvis Creek (No. 2)	0.24	0.28	0.28	0.23	0.23	0.31	0.21	0.32	0.262	0.0016
Lower Purvis Creek (No. 5)	0.32	0.24	0.22	0.29	0.26	0.20	0.29	0.25	0.259	0.0016
Gibson Creek (No. 9)	0.27	0.31	0.27	0.28	0.18	0.12	0.21	0.32	0.245	0.0048
Turtle River – mouth of Purvis Creek (No. 6)	0.28	0.14	0.25	0.25	0.24	0.16	0.23	0.20	0.219	0.0023
Turtle River – upper range (No. 7)	0.20	0.30	0.27	0.32	0.34	0.31	0.34	0.13	0.276	0.0056
East River (No. 8)	0.17	0.25	0.24	0.24	0.24	0.30	0.29	0.26	0.249	0.0016
Reference Stations										
Clubbs Creek (No. 11)	0.21	0.26	0.21	0.26	0.31	0.12	0.16	0.30	0.229	0.0044
Jointer Creek (No. 10)	0.30	0.25	0.27	0.19	0.29	0.26	0.24	0.24	0.255	0.0012

2. Cochran's (C) Test for Homogeneity of Variances of Weight Data[b]

$$C(\text{cal.}) = \frac{s^2(\text{max.})}{\sum s^2} = \frac{0.0056}{0.0326} = 0.17 \text{ ns,}$$

as compared to C (tab.) = 0.23
for $P = 0.05$, k = 12, and v = 7

TABLE 6-2. (cont.)

3. Parametric One-Way Analysis of Variance (ANOVA) of Weight Data[b]

Source of Variation	Degrees of Freedom (df)	Sum of Squares (SS)	Mean Square (MS)	F (cal.)
Sampling stations (plus control)	t - 1 = 11	0.028	0.0025	0.93 ns,
Error	t (r - 1) = 84	0.228	0.0027	
	tr - 1 = 95	0.256		
			as compared to F (tab.) = 1.92 for P = 0.05, 11 numerator df, and 84 denominator df	

Note: Toxicity data are abstracted from Volume II, Section 2 and pertain to 7-day-old mysids exposed for 7 days to undiluted channel water collected from 11 sampling stations distributed throughout the study area and a laboratory control. Water was replaced daily during the study.

Channel water was collected from sampling stations on June 26 (Days 1 and 2 of study), June 28 (Days 3, 4, and 5), and July 1 (Days 6 and 7). Grab samples of water were collected about 0.5 m below the air/water interface from approximately 2 hours on either side of slack low water. The water column at sampling stations was well mixed, as evidenced by differences in subsurface vs. bottom salinity of less than 5 ppt (typically less than 1–2 ppt).

[a] Each replicate (r) consisted of five mysids at initiation of toxicity study. Mean weight for each replicate is the quotient of total weight of surviving mysids and number of mysids initially in replicate.

[b] Cochran's test demonstrated that variances of weight data were not significantly different (as indicated by the symbol "ns" associated with the value for C [cal.]). Consequently, a parametric analysis of variance (ANOVA) was used to evaluate statistical significance of mean differences in weight among sampling stations. The ANOVA documented the absence of statistically significant differences (indicated by the symbol "ns" associated with F [cal.]).

contrasted to EPA's acceptability criterion of at least 0.20 mg. Lastly, all female mysids that survived exposure to water from all sampling stations (and the laboratory control) contained eggs in either their oviducts or brood sacs.

6.1.1.2 Sheepshead Minnows

Sheepshead minnow embryos and larvae exposed for 9 days to ambient surface water (Table 6-3) experienced post-hatch survival that varied from 62 percent (Jointer Creek reference station) to 87 percent (laboratory control). Survival of laboratory controls surpassed the 80 percent criterion for acceptability of test results (U.S. EPA 1988b). The results of statistical analysis presented in Table 6-3 show that survival of fish exposed to water from study area stations is statistically similar to survival of fish exposed to water from both reference and control stations. In addition, only a single deformed fish was observed even though 660 fish were employed in the study.

6.1.2 Field Bioaccumulation Study

The field bioaccumulation study used native and hatchery-produced American oysters to monitor uptake of xenobiotic chemicals at the nine study area and two reference stations. A secondary objective of the bioaccumulation study was to monitor survival, growth, and general condition of oysters at the stations.

Survival of hatchery oysters deployed in the study area for 90 days (Deployment 1) was unexpectedly poor, with some of the lowest survival (2 percent) characteristic of the Jointer Creek reference station (Table 6-4). Survival of Deployment 2 and Deployment 3 oysters (including oysters placed in the LCP Ditch) was substantially higher. However, samples from only one study area station were retrieved for each of Deployments 2 and 3. The generally poor survival and resulting limited sample size of hatchery oysters precludes quantitative evaluation of their survival and growth. Low survival also precludes quantitative evaluation of total solids, lipid content, and body-burden data because environmental stress may have altered their metabolism and contaminant uptake rate. The low survival was apparently not related to site CoPCs and is believed to have been the result of temperature and salinity stress. All oyster study data are presented in Volume II, Section 1.1.

6.2 EVALUATION OF DIRECT TOXICITY OF CoPCs TO BENTHIC ORGANISMS

Benthic macroinvertebrate assemblages were evaluated at two stations at the LCP Site and two stations at the Jointer Creek reference area. At each of the sampling areas, one station was located in mid-channel and the other station was located on the channel bank. Taxonomic results were reported for 20 samples (4 stations [2 onsite and 2 at the reference area] with 5 sample replicates at each station) by the taxonomic laboratory.

TABLE 6-3. STATISTICAL ANALYSIS OF POST-HATCH SURVIVAL OF EMBRYONIC/ LARVAL SHEEPSHEAD MINNOWS EXPOSED TO SURFACE WATER FROM THE LCP SITE AND REFERENCE STATIONS

1. Number of Survivors

Sampling Station	Replicate[a] 1	2	3	4	Mean ($\bar{x}$)	Variance (s^2)
Laboratory Control Water	13	12	14	13	13.00 (87%)	0.67
LCP Site Stations						
LCP Ditch (No. 1)	10	12	9	9	10.00 (67%)	1.99
Purvis Creek headwaters (No. 4)	10	14	13	9	11.50 (77%)	5.66
Upper Purvis Creek (No. 3)	9	10	11	10	10.00 (67%)	0.67
Middle Purvis Creek (No. 2)	8	10	12	10	10.00 (67%)	2.66
Lower Purvis Creek (No. 5)	12	11	10	10	10.75 (72%)	0.92
Gibson Creek (No. 9)	14	10	10	10	11.00 (73%)	4.00
Turtle River – mouth of Purvis Creek (No. 6)	13	13	11	13	12.50 (83%)	1.00
Turtle River – upper range (No. 7)	11	9	11	11	10.50 (70%)	1.00
East River (No. 8)	9	12	10	10	10.25 (68%)	1.59
Reference Stations						
Clubbs Creek (No. 11)	11	13	12	10	11.50 (77%)	1.66
Jointer Creek (No. 10)	12	8	10	7	9.25 (62%)	4.97

2. Cochran's (C) Test for Homogeneity of Variances of Survival Data[b]

$$C(\text{cal.}) = \frac{s^2(\text{max.})}{\sum s^2} = \frac{5.66}{26.79} = 0.21 \text{ ns,}$$

as compared to C (tab.) = 0.33
for $P = 0.05$, k = 12, and v = 3

TABLE 6-3. (cont.)

3. Parametric One-Way Analysis of Variance (ANOVA) of Survival Data[b]

Source of Variation	Degrees of Freedom (df)	Sum of Squares (SS)	Mean Square (MS)	F (cal.)
Sampling stations (plus control)	t - 1 = 11	53.73	4.88	2.19*,
Error	t (r - 1) = 36 tr - 1 = 47	80.25 133.98	2.23	

as compared to F (tab.) = 2.08 for $P = 0.05$, 11 numerator df, and 36 denominator df

4. Tukey's (w) Test (Comparison) of Survival Data[b,c]

Sampling Station	Mean Number of Survivors ($\bar{x}$)
Laboratory control	13.00
Turtle River – mouth of Purvis Creek (No. 6)	12.50
Clubbs Creek reference station (No. 11)	**11.50**
Purvis Creek headwaters (No.4)	11.50
Gibson Creek (No. 9)	11.00
Lower Purvis Creek (No. 5)	10.75
Turtle River – upper range (No. 7)	10.50
East River (No. 8)	10.25
LCP Ditch (No. 1)	10.00
Upper Purvis Creek (No. 3)	10.00
Middle Purvis Creek (No. 2)	10.00
Jointer Creek reference station (No. 10)	**9.25**

$$w = q\sqrt{\frac{\text{error MS}}{r}}$$

$$= 4.95\sqrt{\frac{2.23}{4}}$$

$$= 3.70$$

Note: Toxicity data are abstracted from Volume II, Section 2 and pertain to embryonic/larval fish exposed for 9 days to undiluted channel water collected from 11 sampling stations distributed throughout the study area and a laboratory control. Water was replaced daily during the toxicity study.

Channel water was collected from sampling stations on June 26 (Days 1 and 2 of study), June 28 (Days 3, 4, and 5), July 1 (Days 6 and 7), and July 2 (Days 8 and 9). Grab samples of water were collected about 0.5 m below the air/water interface from approximately 2 hours on either side of slack low water. The water column at sampling stations was well mixed, as evidenced by differences in subsurface vs. bottom salinity of less than 5 ppt (typically less than 1–2 ppt).

[a] Each replicate (r) consisted of 15 embryonic fish at initiation of toxicity study.

TABLE 6-3. (cont.)

[b] Cochran's test demonstrated that variances of post-hatch survival data were not statistically significant (as indicated by the symbol "ns" associated with the value for C [cal.]). Consequently, a parametric ANOVA was employed to evaluate statistical significance of mean differences in survival among sampling stations. The ANOVA documented the presence of statistically significant differences (indicated by the symbol "*" associated with F [cal.]) and generated the error mean square (2.23) used in Tukey's test of the sources (causes) of the differences.

[c] According to Tukey's test, which employs a "familywise" error rate, any two sets of survival data are significantly different if the difference in their mean ($\bar{x}$) values is greater than Tukey's criterion of 3.70. In this table, mean ($\bar{x}$) values associated with the same vertical line are not significantly different, whereas mean ($\bar{x}$) values not associated with the same line are significantly different.

TABLE 6-4. SURVIVAL AND GROWTH OF HATCHERY OYSTERS DEPLOYED AT THE LCP SITE AND REFERENCE STATIONS

Sampling Station	Deployment 1 (90-day exposure) Survival (%)	Deployment 1 (90-day exposure) Mean ($\bar{x}$) growth of survivors (mm, shell length)	Deployment 2 (78-day exposure) Survival (%)	Deployment 2 (78-day exposure) Mean ($\bar{x}$) growth of survivors (mm, shell length)	Deployment 3 (70-day exposure) Survival (%)	Deployment 3 (70-day exposure) Mean ($\bar{x}$) growth of survivors (mm, shell length)
LCP Site Stations						
LCP Ditch (No. 1)	10 (19 of 200 oysters)	+0.2	26 (28 of 108 oysters)	+1.1	--	--
Purvis Creek headwaters (No. 4)	22 (22 of 100 oysters)	-0.5	--	--	--	--
Upper Purvis Creek (No. 3)	8 (8 of 100 oysters)	+3.6	--	--	--	--
Middle Purvis Creek (No. 2)	4 (4 of 100 oysters)	-0.2	--	--	19 (19 of 100 oysters)	+2.2
Lower Purvis Creek (No. 5)	14 (14 of 100 oysters)	-0.5	--	--	--	--
Gibson Creek (No. 9)	2 (2 of 100 oysters)	+4.5	--	--	--	--
Turtle River—mouth of Purvis Creek (No. 6)	--	--	--	--	--	--
Turtle River—upper range (No. 7)	1 (1 of 100 oysters)	+1.0	--	--	--	--
East River (No. 8)	3 (3 of 100 oysters)	+3.3	--	--	--	--
Reference Stations						
Clubbs Creek (No. 11)	13 (13 of 100 oysters)	+0.6	--	--	29 (29 of 100 oysters)	+1.8
Jointer Creek (No. 10)	2 (2 of 100 oysters)	+5.0	--	--	30 (39 of 129 oysters)	+6.0

Footnotes on next page.

TABLE 6-4. (cont.)

Note: Three deployments of hatchery oysters were used during the study. Deployment 1 occurred on June 13 (LCP Ditch and Purvis Creek headwaters) and June 14 (other nine stations), 1996. Deployment 2 occurred on June 25, 1996, in only the LCP Ditch (6 days after an accidental discharge of wastewater from the onsite treatment facility to the ditch). Deployment 3 occurred on July 3, 1996, at Clubbs Creek reference station, Jointer Creek reference station, middle Purvis Creek, and the Turtle River near the mouth of Purvis Creek. Hatchery oysters were retrieved from sampling stations on September 11, 1996 (LCP Ditch and Purvis Creek headwaters), and September 12, 1996 (remaining nine stations except Turtle River at the mouth of Purvis Creek, where all oyster deployments were lost). Retrieved hatchery oysters ranged in size from 71 to 125 mm in shell length.

-- - information not generated

Taxonomic evaluations were made to the lowest practical level by experienced taxonomists using the appropriate taxonomic literature.

The results of the benthic macroinvertebrate study are summarized in Tables 6-5 and 6-6. The original data for this study are presented in Volume II, Section 1.9. The total number of taxa, total number of individuals, and mean density of individuals in sediment collected onsite and at the reference area were similar, and in some cases, the results from the onsite macroinvertebrate enumeration were slightly higher than the results from the reference area; however, the small sample size precludes a statistical evaluation of the data (Table 6-6).

Direct toxicity of CoPCs to benthic organisms was also evaluated using sediment toxicity tests with the amphipod *Leptocheirus plumulosus*. Results of that evaluation are presented in Appendix B.

6.3 EVALUATION OF FOOD-WEB RELATED TOXICITY OF MERCURY AND PCBs TO WILDLIFE

The following sections contain toxicity profiles for mercury and PCBs, followed by a discussion of TRVs used to assess potential risk to wildlife receptors.

6.3.1 Toxic Effects of Mercury and PCBs

The toxic effects of mercury and PCBs to wildlife receptors is basically a function of:

- The environmental medium that contains the chemical
- The form of the chemical in the environmental medium
- The concentration of the chemical form in the medium
- The duration of exposure of the receptor to the medium and associated chemical form
- The route (e.g., direct contact, ingestion) by which the receptor assimilates the chemical form
- The physiological disposition of the chemical form and the receptor (i.e., the toxic mechanism and the receptor's sensitivity).

The remainder of this section addresses the general ecotoxicological properties of mercury and PCBs.

TABLE 6-5. SUMMARY OF TAXA IDENTIFIED DURING THE BENTHIC MACROINVERTEBRATE STUDY FOR THE LCP SITE AND REFERENCE STATIONS

Phyllum	Class	Order	Family	Genus/Species
Annelida		Oligochaeta	Tubificidae	*Tubificidae*
		Polychaetae	Capitellidae	*Capitella capitata*
				Mediomastus
			Nereidae	*Nereis succinea*
			Phyllodocidae	*Eteone succinea*
				Phyllodocidae
Arthropoda	Crustacea	Amphipoda		*Amphipoda*
			Aeginellidae	*Aeginelllidae*
			Ampelliscidae	*Ampelisca*
			Melitidae	*Melita*
		Decapoda	Palaemonidae	*Palaemonetes pugio*
		Isopoda	Anthuridae	*Cyathura polita*
		Tanaidacea	Paratanaidae	*Paratanaidae*
Arthropoda	Insecta	Coleoptera		*Coleoptera*
		Diptera	Dolichopodidae	*Dolichopodidae*
Mollusca		Gastropoda	Nassariidae	*Ilyanassa obsoleta*
Rhynchocoela				*Rhynchocoela*

TABLE 6-6. SUMMARY OF QUANTITATIVE RESULTS OF THE BENTHIC MACROINVERTEBRATE STUDY FOR THE LCP SITE AND REFERENCE STATIONS

Station	Total No. of Taxa[a]	Total No. of Individuals	Mean Density (no. individuals/m^2)
Onsite			
Mid-channel	7	47	2,089
Channel bank	7	19	844
Jointer Creek Reference Station			
Mid-channel	9	30	1,333
Channel bank	5	18	800

[a] Values are based on the pooled data for the five replicate samples collected at each station.

6.3.1.1 Mercury

Mercury is a naturally occurring element that exists in the environment in different chemical forms. The predominant form of mercury in water, soil, and sediment is ionic mercury (Hg^{2+}). Some of the ionic mercury can exist in a free ionic form, but most is bound by adsorption or chemical bonding to organic matter. Elemental mercury (Hg^0) is usually a minor component of total mercury in water, soil, and sediment but can be the primary component at the LCP Site because mercury was released in the elemental form. Elemental mercury is relatively stable in the environment, but it is subject to volatilization and oxidation to Hg^{2+}. Methylmercury (CH_4Hg^-) represents a small fraction of total mercury but is of major importance because of its toxicity and tendency to bioaccumulate. Methylmercury is formed primarily as a byproduct of the metabolism of sulfate-reducing bacteria whereby a methyl group is bound to a mercury atom in free ionic form.

Toxic effects of methylmercury in aquatic life typically include impairment of the central nervous system functions (WHO 1989), which often manifests itself in reduced survival (sometimes by reduced ability to avoid predators), growth, and/or reproduction. The magnitude of toxicity varies among taxa, with annelids, fishes, gastropods, and crustaceans being characterized by an increasing magnitude of response (Chlorine Institute 1992). Within a taxon, larvae and early developmental stages (excluding eggs) usually exhibit greatest sensitivity to mercury (Eisler 1987; WHO 1989; Chlorine Institute 1992).

Methylmercury in birds has been demonstrated to affect various organ systems, including the central nervous system, liver, kidneys, immune response system, and, particularly, the reproductive system (WHO 1989; Chlorine Institute 1992). Young birds (e.g., embryos) are more sensitive to mercury residues than older birds (Eisler 1987; Chlorine Institute 1992).

Although mammals (excluding humans) have not been as well studied as birds, target organ systems for methylmercury toxicity are likely to be similar. In mammals, the fetus is the most sensitive life stage, and a substantial latent period of weeks, months, or even years sometimes occurs between cessation of mercury exposure and the onset of symptoms of mercury over-exposure (Eisler 1987).

6.3.1.2 PCBs

PCBs are a complex group of chemicals consisting of 10 different homolog classes (monochlorobiphenyls to decachlorobiphenyls) that are distinguished by the number of chlorine atoms attached (or substituted) to the biphenyl molecule. There are 209 different PCB congeners, which reflect the different number (homolog) or location (isomer) of attached chlorine ions. PCBs were manufactured in the United States under the trade name Aroclor. Different Aroclors, which consist of different mixtures of congeners, were given a four-digit code (e.g., Aroclors® 1254, 1268) in which the last two digits usually indicate the chlorine content (by percent weight).

In general, PCBs are relatively insoluble in water, extremely lipophilic, and highly stable in biological systems (U.S. EPA 1980). The Aroclor of interest at the LCP Site, Aroclor® 1268, consists primarily of highly chlorinated biphenyl molecules containing 8, 9, and 10 chlorine molecules. Consequently, in comparison to most other Aroclors, Aroclor® 1268 is characterized by higher specific-gravity (1.604–1.611), lower solubility in water, and slower biological depuration rate (U.S. EPA 1980; Pruell et al. 1988).

PCBs bioaccumulate and biomagnify in the food chain, thus the primary route of exposure for wildlife is through consumption of prey tissue containing the compound. Toxic effects of accumulated body burdens of PCBs in both aquatic life and wildlife include liver damage, tumors, a wasting syndrome, neurotoxicity, reproductive failure, immunotoxicity, birth defects, and death (Eisler 1986). Toxic response depends on the congener composition of the mixture and its purity; the species, age, and sex of the exposed animal; and the route and duration of exposure (Safe 1994). Chlorine-related characteristics (both amount of chlorine and substitution positions of chlorine atoms) of PCBs change with time and location as PCBs are transported and transformed in the environment (U.S. EPA 1980).

Toxicologically adverse body burdens of PCBs in aquatic life vary greatly as partial functions of the specific Aroclor®, aquatic species, and measurement endpoint evaluated. For example, starry flounder exposed to "total" PCBs in the field exhibited reduced reproductive success when eggs contained 0.2 mg/kg PCBs (data reported by NOAA 1994, dry weight versus wet weight measurement was not indicated). Conversely, survival of juvenile sheepshead minnows exposed to Aroclor® 1016 in the laboratory was not affected at a whole-body concentration in fish of 230 mg/kg wet weight (Hansen et al. 1975).

6.3.2 Derivation of Toxicity Reference Values

The approach used to derive TRVs is to evaluate pertinent studies and then select the most appropriate study and use it for all taxonomically related species. Ideally, the study chosen to derive the TRV should indicate a chronic effect level for a relevant ecological endpoint (e.g., decreased fecundity, reduced survivorship) and should bound the response threshold by determining both a no-observed-adverse-effect level (NOAEL) and a lowest-observed-adverse-effect level (LOAEL) dose. The NOAEL values are used in the wildlife food-web exposure risk evaluations as a conservative estimate of the dose at which no adverse effects are expected to occur. Studies used to derive TRVs for avian and mammalian receptors are described below. No suitable studies were found that could be used to determine TRVs for mercury or PCBs in reptiles, and hazard quotients for diamondback terrapins are determined using TRVs presented by Sprenger et al. (1997).

Organomercurial species, such as methylmercury, are more toxic to organisms than inorganic forms of mercury. Thus, the TRVs for mercury described below are based on studies using methylmercury.

6.3.2.1 Derivation of Toxicity Reference Values for Birds

Methylmercury—A LOAEL dose in the diet of 0.064 mg/kg-day was derived for potential effects of methylmercury dicyandiamide on mallard ducks in a multi-generational study with reproductive success as a measurement endpoint (Heinz 1979). A NOAEL dose of 0.032 mg/kg-day was derived by applying a LOAEL-to-NOAEL uncertainty factor of 2, as recommended by U.S. EPA (1993a) because the LOAEL dose appeared to be near the threshold for dietary effects.

The TRV dose is scaled using allometry to derive NOAEL doses for other avian receptors (wildlife species NOAEL [$NOAEL_W$]). The concept of allometric scaling is based on species-specific differences that exist in broad-based physiological processes (e.g., metabolic rate) that have a general relation to toxicological responses (Sample et al. 1996). The $NOAEL_W$ can be calculated by using the adjustment factor for differences in body size:

$$NOAEL_W = NOAEL_T\left(\frac{bw_t}{bw_w}\right)^{1/3}$$

Where bw_t and bw_w are body weights of the test and wildlife species, respectively. The mean body weight of mallards used by Heinz (1979) was 1 kg. Mean body weights of wildlife species are provided in Section 5.3.2.8 *Life History Characteristics of Receptor Species*. The NOAEL dose for wood storks (bw_w=2.05 kg) is 0.025 mg/kg-day, the NOAEL dose for clapper rails (bw_w=0.27 kg) is 0.049 mg/kg-day, and the NOAEL dose for marsh wrens (bw_w=0.00941 kg) is 0.15 mg/kg-day.

Aroclor® 1268—Lillie et al. (1974) examined the effects of seven Aroclor mixtures, including Aroclor® 1268 on adult body weight gain, egg production, egg weight, eggshell thickness, and egg fertility of white leghorn hens. Each Aroclor was administered separately at 20 mg/kg in the diet for 9 weeks. At that dose level, Aroclor® 1268 had no adverse effect on any measured endpoint. Average food consumption of hens receiving the 20 mg/kg Aroclor® 1268 dose was 0.125 kg/day, and average hen weight was 1.95 kg. Based on these values, the test species NOAEL ($NOAEL_T$) for Aroclor® 1268 is 1.28 mg/kg-day. This unbounded NOAEL dose represents a chronic exposure during a critical life stage (reproduction). Hens receiving 20 mg/kg of Aroclor® 1254 in the diet experienced decreased egg production and hatchability of fertile eggs, but 2 mg/kg Aroclor® 1254 had no effect on any reproductive parameter. The 2 mg/kg dose level yields a bounded $NOAEL_T$ of 0.127 mg/kg-day based on average food consumption of 0.124 kg/day and average hen weight of 1.95 kg. The 20 mg/kg dose level yields a bounded test species LOAEL ($LOAEL_T$) of 1.22 mg/kg-day based on average food consumption of 0.119 kg/day and average body weight of 1.95 kg. Comparison of the Aroclor® 1254 and 1268 $NOAEL_T$ values indicates that Aroclor® 1268 is about 10-fold less toxic than Aroclor® 1254.

The study by Lillie et al. (1974) permits direct derivation of a TRV for Aroclor® 1268 without extrapolating from results obtained with less chlorinated Aroclor mixtures. Allometric scaling of the $NOAEL_T$ for Aroclor® 1268 results in a $NOAEL_W$ of 1.26 mg/kg-day for wood storks, 2.47 mg/kg-day for clapper rails, and 7.56 mg/kg-day for marsh wrens.

6.3.2.2 Derivation of Toxicity Reference Values for Mammals

Methylmercury—A bounded $NOAEL_T$ of 0.032 mg/kg-day ($LOAEL_T$=0.16 mg/kg-day) was derived for methylmercury in a multi-generational study in which rats (bw_t=0.35 kg) were exposed to methylmercury chloride in the diet and reproduction (pup viability) was the measurement endpoint (Verschuuren et al. 1976). Allometric scaling of the $NOAEL_T$ results in a $NOAEL_W$ of 0.0029 mg/kg-day for West Indian manatees (bw_w=449 kg), a $NOAEL_W$ of 0.012 mg/kg-day for river otters (bw_w=6.7 kg), and a $NOAEL_W$ of 0.016 mg/kg-day for raccoons (bw_w=2.78 kg).

Aroclor® 1268—No references were found pertaining to chronic toxicity tests in which mammalian species were exposed to Aroclor® 1268; therefore, a TRV for use in the exposure modeling was conservatively derived from a study by Aulerich and Ringer (1977) in which mink (bw_t=1 kg) were exposed through the diet to 0.14, 0.69, or 2.06 mg/kg-day Aroclor® 1254 for 4.5 months. Aroclor® 1254 at 0.69 and 2.06 mg/kg-day in the diet reduced the number of live offspring. No effect was observed at 0.14 mg/kg-day; therefore, this dose was considered a bounded chronic $NOAEL_T$ ($LOAEL_T$=0.69 mg/kg-day). Allometric scaling of the $NOAEL_T$ results in a $NOAEL_W$ of 0.018 mg/kg-day for West Indian manatees, 0.075 mg/kg-day for river otters, and 0.10 mg/kg-day for raccoons.

6.4 EVALUATION OF THREATENED AND ENDANGERED SPECIES

Field studies of threatened and endangered species conducted during 1996 were limited to aerial surveys of wading bird abundance and distribution at the LCP Site marsh. Studies were also planned to evaluate chemical concentrations in eggs and hatchlings of clapper rails and diamondback terrapins to derive site-specific TRVs for reproductive endpoints that could be used in food-web exposure models for threatened and endangered species (i.e., wood storks and sea turtles). The clapper rail study was not performed because field workers had difficulty locating nests in the dense vegetation.

The diamondback terrapin study was initiated, but only a limited number of nests or laying females were located. Only three clutches of eggs were obtained, two from reference areas (Clutches A and D) and one from the LCP Site (Clutch E). The eggs from all three clutches were artificially incubated in the laboratory. Clutch A produced seven hatchlings, and Clutches D and E each produced five hatchlings. Hatchlings and unhatched eggs were

grouped by clutch and analyzed for CoPCs. The results of the analyses are presented in Volume II, Section 1.8. The small sample sizes obtained from the reference area and the marsh at the LCP Site preclude a meaningful statistical comparison of CoPC concentrations in eggs or hatchlings from the two areas.

7. RISK CHARACTERIZATION

The unique nature of response actions at the Site (i.e., the fast-tracking of multiple removal actions) has allowed risk assessments for the Site to reflect conditions at the Site following completion of the removal actions. In that light, this ERA differs from most ERAs in that it accounts for the completed removal activities in the uplands and marsh areas and considers only post-removal data (e.g., original characterization data outside the lateral extent and vertical limits of excavation). In addition, data for portions of the marsh that are currently identified for inclusion on the upcoming marsh removal actions were not included in the risk assessment database. In particular, biological data collected from within the 13-acre area immediately adjacent to the upland portion of the Site and the outfall ditch from the LCP facility to Purvis Creek (both of which are planned for inclusion in the upcoming removal action) were not used in this ERA as they are not reflective of site post-removal conditions.

7.1 RISK ESTIMATION

Estimation of ecological risks associated with CoPCs at the Site is based on the integration of the results of the exposure and effects assessments. In keeping with the format of the risk assessment produced in support of removal actions (Sprenger et al. 1997), each assessment endpoint developed in that removal risk assessment is evaluated below based on findings relevant to all measurement endpoints.

Risk characterization for food-web exposure models is based on hazard quotients obtained by dividing the modeled estimate of the site-specific dose by the corresponding TRV. A hazard quotient of less than or equal to 1.0, based on a NOAEL TRV, indicates that the evaluated chemical poses no significant risk for a receptor as a result of cumulative exposure through all components of the food web. A hazard quotient of greater than 1.0 indicates a potential for unacceptable risk to ecological receptors. However, the result must be considered with regard to the uncertainty associated with the parameters evaluated as part of the model. Results of the food-web exposure models for mercury and PCBs are summarized in Table 7-1 and are discussed for each receptor species in the following sections.

7.1.1 Maintenance of Ecological Health of the Salt Marsh Community

For this assessment endpoint, there are weight-of-evidence components from a number of measurement endpoints. Toxicity tests and benthic community structure based upon site-specific direct measurements are superior evidence components to bulk media

TABLE 7-1. HAZARD QUOTIENTS OF POTENTIAL FOR UNACCEPTABLE RISK TO WILDLIFE RECEPTORS IN THE MARSH AT THE LCP SITE

		Polychlorinated Biphenyls			Mercury		
Species	Location	Total Exposure (mg/kg-day)	TRV (mg/kg-day)	HQ	Total Exposure (mg/kg-day)	TRV (mg/kg-day)	HQ
West Indian Manatee	Entire marsh (tidal channel only)	0.00007	0.018	0.004	0.000004	0.0029	0.001
River Otter	Entire marsh	0.083	0.075	1.1	0.060	0.012	5.0
Raccoon	Transect 2	0.024	0.10	0.24	0.0073	0.016	0.46
	Transect 3	0.017	0.10	0.17	0.0062	0.016	0.39
Wood Stork	Entire marsh	0.0013	1.26	0.001	0.0004	0.025	0.02
Clapper Rail	Transect 1	0.10	2.47	0.04	0.018	0.049	0.37
	Transect 2	0.11	2.47	0.04	0.016	0.049	0.33
	Transect 3	0.08	2.47	0.03	0.012	0.049	0.24
Marsh Wren	Transect 1	0.19	7.56	0.03	0.006	0.15	0.04
	Transect 2	0.18	7.56	0.02	0.006	0.15	0.04
	Transect 3	0.18	7.56	0.02	0.006	0.15	0.04
Diamondback Terrapin	Entire marsh	0.0020	0.323	0.006	0.0021	0.50	0.004

Note: TRV - toxicity reference value (NOAEL)
HQ - hazard quotient

concentrations and body burdens data, and food-web exposure models. These superior evidence components revealed no discernible site-related effects.

The laboratory toxicity study reported here found no statistically significant effects on water-column organisms at the LCP Site. This conclusion is based on essentially chronic toxicity tests with sensitive life stages of two types of marine organisms (mysids and sheepshead minnows), with reproductive capacity used as one of several ecologically relevant measurement endpoints. The long-term (partial chronic) laboratory sediment toxicity study reported in Appendix B found that concentrations of lead, mercury, and Aroclor® 1268 in marsh sediments from within the removal area did not correlate with significant effects on survival or growth of the marine amphipod *Leptocheirus plumulosus*. Sediment toxicity tests were also conducted by Sprenger et al. (1997) using *Leptocheirus plumulosus* and *Penaeus vannamei* (an estuarine shrimp species). Results of these acute toxicity tests did not indicate that site-related mortality threats existed for benthic and epibenthic species. Tests performed by Sprenger et al. (1997) were intended to evaluate effects of metals and PAHs, and were not directly used to evaluate Aroclor® 1268 or mercury. Additionally, finfish embryo toxicity tests performed by Sprenger et al. (1997) with sediment indicated that a few embryos responded with lesions, but no correlation was found between sediment chemical concentrations and response.

An investigation of benthic community abundance and diversity performed by Sprenger et al. (1997) found no alterations in the benthic community associated with chemicals present in the marsh at the Site. Qualitatively similar results were seen in a study conducted as part of this ERA, but sample sizes are too small to permit statistical evaluation. In addition to these investigations, results of food-web exposure models presented in this ERA suggest that dose-based exposure risks are at acceptable levels for most salt marsh receptors.

Fiddler crabs collected by EPA in May of 1995 at three of four sampling locations in the marsh at the LCP Site were reported to have lower lipid content relative to crabs from the reference area (Sprenger et al. 1997). In October 1995, EPA collected fiddler crabs at two locations in the marsh and at one reference location. These crabs did not display large differences in lipid content among stations. Lipid content declined in crabs at all locations in October relative to crabs collected in May. Reduction in lipid content in fiddler crabs at the Site was considered an indicator of reduced "fitness" by Sprenger et al. (1997) and was cited as evidence of an adverse impact for this assessment endpoint. However, no cause-and-effect relationships were established between decreased lipid content and ecologically relevant endpoints, such as decreased fecundity or reduced survivorship. Therefore, this measurement endpoint is given low weighting in the weight-of-evidence approach used here.

Overall, for chronic exposure levels, there are no discernible exposure risks for the ecological health of the salt marsh community.

7.1.2 Protection of Long-Term Health and Reproductive Capacity of Aquatic Reptiles

The potential for unacceptable risks for this assessment endpoint is most appropriately based on food-web exposure, as estimated for the diamondback terrapin. The food-web model was predicated on site-specific assumptions and data. Hazard quotients are much less than 0.01 for both mercury and PCBs (Table 7-1). On the basis of the exposure model results, there are no chronic exposure risks to aquatic reptiles associated with either mercury or PCBs at the LCP Site. Additionally, Sprenger et al. (1997) reported that terrapin histology did not display any degeneration or abnormality known to be associated with CoPCs. Sample sizes were small, however, and Sprenger et al. (1997) could infer no conclusive judgment regarding the status of the population from this information.

7.1.3 Protection of Long-Term Health and Reproductive Capacity of Omnivorous Mammals

The potential for unacceptable risks for this assessment endpoint is most appropriately based on food-web exposure, as estimated for raccoons. The food-web model was predicated on site-specific assumptions and data. The home range of a raccoon is smaller than the areal extent of the marsh at the LCP Site. Therefore, the exposure scenario evaluated risk for raccoons with foraging ranges located in two different parts of the marsh, based on sediment and prey data collected at Transects 2 and 3. Hazard quotients for mercury or PCBs were almost identical at both transects, and were less than 1 in all cases. Ingestion of prey was the most important route of exposure. The majority of the exposure to mercury and PCBs was attributable to consumption of fiddler crabs and oysters. On the basis of the exposure model results, there are no chronic exposure risks associated with exposure to mercury or PCBs for omnivorous mammals at the LCP Site.

7.1.4 Protection of Long-Term Health and Reproductive Capacity of Piscivorous Mammals

The potential for unacceptable risks for this assessment endpoint is most appropriately based on food-web exposure, as estimated for the river otter. The food-web model was predicated on site-specific assumptions and data. The exposure scenario evaluated risk for a river otter foraging throughout the marsh at the LCP Site. Mean concentrations of mercury and PCBs in abiotic media and prey species were considered indicative of the chemical concentrations to which a river otter might be exposed as a result of its foraging behavior, which effectively integrates variations in chemical burdens occurring within different regions of the marsh. Hazard quotients were 1.1 for PCBs and 5.0 for mercury. Consumption of prey was the principal route of exposure. Exposure to PCBs was primarily due to consumption of killifish (52 percent of total exposure), while consumption of blue crabs contributed substantially to the total exposure to mercury (63 percent of total exposure). On the basis of the exposure model results, there is a potentially

unacceptable chronic exposure risk associated with exposure to PCBs and mercury for piscivorous mammals inhabiting the LCP Site.

7.1.5 Protection of Long-Term Health and Reproductive Capacity of Birds

The potential for unacceptable risks for this assessment endpoint is most appropriately based on food-web exposure, as estimated for the clapper rail and marsh wren. The food-web models were predicated on site-specific assumptions and data. The home range of clapper rails and marsh wrens is smaller than the areal extent of the marsh at the LCP Site. Therefore, the exposure scenario evaluated the potential for unacceptable risk for rails and wrens foraging in three different parts of the marsh. Hazard quotients for exposure to mercury and PCBs were much less than 1 for both clapper rails and marsh wrens at all transects. Based on the results of the food-web exposure model, no chronic risks associated with exposure to mercury and PCBs were found for birds inhabiting the LCP Site. In support of this line of evidence, biomarker histopathology performed by Sprenger et al. (1997) for clapper rails inhabiting the marsh did not indicate specific toxicity or a specific uniform degeneration in tissue related to CoPCs.

7.1.6 Protection of Long-Term Health and Reproductive Capacity of Fishery Resources

The potential for unacceptable risks for this assessment endpoint is based on results of toxicity tests with fishes and potential prey as well as comparisons of chemical body burdens of fishes to literature-based effect levels, the two lines of evidence most highly recommended for this assessment endpoint by Sprenger et al. (1997). The toxicity tests are designed to evaluate the direct toxicity of chemicals in water or sediment to fishes and their prey, while body-burden comparisons address risks (including reproductive risks associated with maternal transfer of contaminants) to fishes exposed to contaminated water, sediment, and food over long periods.

Results of laboratory toxicity tests performed with embryonic/larval sheepshead minnows indicated that survival of fish exposed to water from stations at the LCP Site was statistically similar to survival of reference and control fish. In addition only 1 of 660 tested fish exhibited deformities. Survival, growth, and fecundity of mysids (potential invertebrate prey of fishes) exposed to water from stations at the LCP Site were also statistically similar to survival of reference and control organisms. These results are consistent with results of a "range-finding/exploratory" toxicity test reported by Sprenger et al. (1997) in which no discernible dose-related effects were noted with embryonic Japanese medaka exposed to sediment collected from several locations at the LCP Site (most of which were situated within the highly contaminated removal area). The low rate of lesion formation observed in this preliminary test suggests that CoPCs anywhere in the LCP Site are not associated with risk at the population level to finfish health or reproduction capacity.

Body burdens of mercury in whole killifish collected from a location outside of the removal area (Location LCP 43) averaged 1.0 mg/kg dry weight or 0.24 mg/kg wet weight, while mean mercury body burden in fillets of spot obtained from upstream Purvis Creek was 1.5 mg/kg dry weight or 0.31 mg/kg wet weight (Sprenger et al. 1997). These mercury body burdens are substantially less than the whole body burdens cited by Sprenger et al. (1997) to cause toxicological effects in fishes—i.e., mortality, decreased appetite, and decreased activity in rainbow trout at 4 to 27 mg/kg wet weight (Matida et al. 1971); and increased mortality, deformities, and decreased growth in brook trout at 5 to 7 mg/kg wet weight (McKim et al. 1976).

Body burdens of Aroclor® 1268 in whole killifish collected from Location LCP 43 averaged 3.74 mg/kg dry weight or 0.88 mg/kg wet weight, while mean Aroclor® 1268 body burden in spot filets obtained from upstream Purvis Creek was 2.8 mg/kg dry weight or 0.58 mg/kg wet weight (Sprenger et al. 1997). Although studies documenting toxic body burdens of Aroclor® 1268 to fishes were not identified, the studies cited by Sprenger et al. (1997) for toxicity of other Aroclors® indicate that a wide margin of safety exists at the LCP Site. Indeed, whole body burdens reported to be toxic to fishes are 200 mg/kg of Aroclor® 1016 causing mortality in fry of sheepshead minnows (Hansen et al. 1975); 170 mg/kg Clophen A50 inhibiting reproductive development of minnows (Bengtsson 1980, reference not provided by Sprenger et al. [1997]); and 46 mg/kg of Aroclor® 1254 causing mortality in spot (Hansen et al. 1971). There is no indication if these toxicological thresholds are reported in dry weight or wet weight. However, even if expressed in dry weight, the thresholds are from one to two orders-of-magnitude greater than body burdens of fishes at the LCP Site.

The results of toxicity tests and body-burden evaluations (presented above) document the absence of long-term health and reproductive risks to fishery resources at the LCP Site.

7.1.7 Protection of Fishery Nursery Function

Lines of evidence for this assessment endpoint are the same as for the health and reproductive capacity of fishery resources. Based on the above-presented results of toxicity tests and body-burden evaluations, there are no risks to the fishery nursery function at the LCP Site.

7.1.8 Protection of Individual Threatened or Endangered Marine Turtles

Lines of evidence for this assessment endpoint are the same as for salt marsh aquatic reptiles. It is important to note that marine turtles are primarily open-water species that may forage in tidal marshes but spend most of their time in larger coastal bays or in offshore ecosystems. Thus, for these species, using risk findings based on assumptions for the diamondback terrapin species yields a conservative risk estimate. Based on an evaluation of the terrapin measurement endpoints (food-web exposure model, histopathology findings),

there are no long-term, site-related exposure risks to threatened or endangered sea turtles at the LCP Site.

7.1.9 Protection of Individual Wood Storks

The potential for unacceptable risks for this assessment endpoint is most appropriately based on food-web exposure. The food-web model was predicated on site-specific exposure assumptions and data. The exposure scenario evaluated risk for a wood stork foraging throughout the marsh at the LCP Site. Mean concentrations of mercury and PCBs in abiotic media and prey species were considered indicative of the chemical concentrations to which a wood stork might be exposed as a result of its foraging behavior, which effectively integrates variations in chemical burdens occurring within different regions of the marsh. Hazard quotients for exposure to mercury and PCBs were much less than 1 for wood storks. Based on the results of the food-web exposure model, no chronic risks associated with exposure to mercury and PCBs were found for wood storks foraging the LCP Site. In support of this line of evidence, biomarker histopathology for another avian species (clapper rails) performed by Sprenger et al. (1997) did not indicate specific toxicity or a specific uniform degeneration in tissue related to CoPCs.

7.1.10 Protection of Individual West Indian Manatees

The potential for unacceptable risks for this assessment endpoint is most appropriately based on food-web exposure. The food-web model was predicated on site-specific exposure assumptions and data. The exposure scenario evaluated risk for a manatee foraging along borders of tidal channels throughout the marsh at the LCP Site. Mean concentrations of mercury and PCBs in abiotic media and vegetation were considered indicative of the chemical concentrations to which a manatee might be exposed as a result of its foraging behavior, which effectively integrates variations in sediment chemistry and chemical burdens of cordgrass occurring along tidal channels within different regions of the marsh. Hazard quotients for exposure to mercury and PCBs were much less than 1 for manatees. Based on the results of the food-web exposure model, no chronic risks associated with exposure to mercury and PCBs were found for West Indian manatees inhabiting the LCP Site.

7.1.11 Protection of Individual Shortnose Sturgeons

Shortnose sturgeon are anadromous and spawn primarily in large rivers with considerable fresh water flow (Crance 1985). Adult fish are not known to frequent the Turtle River (or Purvis Creek). Suitable spawning grounds do not exist in the vicinity of the LCP Site. Thus, the presence of individual sturgeon in the marsh at the LCP Site would be short term, and only transient adults or subadults (no eggs, larvae, or early-stage juveniles) are expected.

Risks for this assessment endpoint are also identified by analogy to findings for other finfish species. Results of laboratory toxicity tests performed with sheepshead minnows, mysids, and Japanese medaka exposed to site water or sediment do not indicate risk to shortnose sturgeon that might migrate through the LCP Site. Similarly, results of body-burden evaluations of finfishes chronically exposed to mercury and PCBs outside the removal area at the Site do not indicate risk to transient sturgeons.

Site-specific evaluation of shortnose sturgeon habitat requirements and measurement endpoints for finfishes indicates that there are no exposure risks to shortnose sturgeon at the LCP Site.

7.2 COMPARISON WITH ESTIMATION OF RISK PRESENTED BY SPRENGER ET AL. (1997)

The objective of the ERA by Sprenger et al. (1997) was to conduct a risk assessment with the focus on EPA Removal Program objectives. This ERA builds on the ERA performed by Sprenger et al. (1997) that was completed in advance of planned marsh removal actions. The objective of this ERA is to evaluate the potential ecological effects of CoPCs in the marsh under post-removal conditions; thus, it addresses marsh areas outside the removal area. Despite having different objectives, the two risk assessments use similar methods to assess ecological risk, except that the ERA by Sprenger et al. (1997) used more conservative assumptions than this ERA to minimize the chances of underestimating risk. Thus, for exposure and effects parameters for which site-specific information was lacking, assumed values reported by Sprenger et al. (1997) were consistently made in the direction of conservatively estimating risk. However, both risk assessments are valid evaluations of risk and reflect a range of conservative exposure or effects assumptions used in the calculation of hazard quotients. The important parameters and the distinctions between the two risk assessments include:

1. Estimating risk based on location-specific chemical concentrations in sediment and prey items versus area-weighting to account for species-specific foraging patterns in the marsh

2. Applying an area use factor of 1 for wildlife receptors versus use of fractional values for species with foraging ranges larger than the area of the marsh (i.e., West Indian manatee, river otter, and wood stork) or with foraging ranges that include upland areas (i.e., raccoon)

3. Modeling mercury exposure based on total mercury concentrations in sediment and prey tissue, but modeling effects based on TRVs for methylmercury, the more toxic form of mercury (i.e., assuming that all mercury at the LCP Site is methylmercury) versus modeling exposure and effects based on methylmercury

4. Modeling PCB exposure based on Aroclor® 1268 concentrations in sediment and prey tissue, but modeling effects based on TRVs for the presumably more toxic Aroclor® 1254 or Clophen® A50 versus modeling exposure and effects based on Aroclor® 1268.

The purpose of this section is to compare risk estimates reported by Sprenger et al. (1997) with estimates derived via this risk assessment approach and to explain apparent major differences in risk estimates between the two approaches.

Food-web exposure models were used by Sprenger et al. (1997) in their risk assessment to establish exposure-response relationships between sediment concentrations of Aroclor® 1268 and mercury and biological responses along a concentration gradient. To do this, the risk assessment approach used co-located sediment, water, and prey tissue samples collected from single sampling stations in food-web exposure models similar in format to the exposure models used in this risk assessment. Five of the stations used in the risk assessment of Sprenger et al. (1997) were physically located within the area of the marsh covered by the removal action, specifically Locations 17-18, 19-20, 35, M-1, and M-2.

As part of the field investigation for this ERA, body burdens in prey species were analyzed along a marsh transect (T-4) lying within the removal action area (see Figure 4-6). Prey tissue data from this transect were not used in this risk assessment because they are not reflective of sitewide post-removal conditions. However, for the purposes of comparing the results of this risk assessment (which considered sitewide post-removal conditions) and the Sprenger et al. (1997) risk assessment (which considered pre-removal conditions within the removal action area), the potential for unacceptable risk to receptors that may have existed in this part of the marsh prior to removal actions can be calculated based on body-burden data for prey species collected along Transect T-4 and sediment chemistry data collected at the marsh grid stations lying within the removal zone. Three of the receptor species (raccoon, clapper rail, and marsh wren) have home range sizes small enough to be contained within the removal area and are considered in this retrospective evaluation of risk. Similar evaluations of potential chronic risk are not performed for West Indian manatee, river otter, wood stork, or diamondback terrapin. The home range of these species is larger than the removal area; therefore, it was considered appropriate to use the site-specific assumptions of this baseline ERA, to assume that none of these four receptors would have foraged exclusively in the removal area.

Concentrations of methylmercury and Aroclor® 1268 in prey species tissues at Transect 4 are shown in Figure 4-6. Aroclor® 1268 concentrations were not measured in insects collected from this transect. Therefore, the average concentration in two samples of grasshoppers (0.64 mg/kg dry weight) collected from the marsh by Sprenger et al. (1997) was used. Fifty-nine of the marsh grid stations shown in Figure 4-3 are in the removal zone. These stations are A2–3, B1–4, C1–5, D2–5, E2–5, F2–5, G2–5, H1–5, I1–5, J1-5, K1–5, L1–5, M1–2, N1–2, O1–2, and P1. The mean total mercury concentration in surface sediment at these stations is 99.2 mg/kg dry weight (all data presented in Table 1.2-2 of Volume II of this report). The mean methylmercury concentration is estimated to be

0.59 percent of this value, or 0.58 mg/kg. The mean Aroclor® 1268 concentration in sediment at these stations is 17.4 mg/kg dry weight. Exposure values for the species are the same as those listed in receptor species life history profiles presented in Section 5.3.2.8 *Life History Characteristics of Receptor Species*. The values are summarized in Table 7-2 and values used by Sprenger et al. (1997) in the risk assessment in support of the removal action are also provided for comparison. As indicated in the table, assumptions made regarding exposure parameters (i.e., body weights, ingestion rates, area use factors, and diet) are generally similar between the two risk assessments. Results of the removal area food-web exposure models using Transect T-4 sediment, water and tissue data are presented in Table 7-3 for raccoons, Table 7-4 for clapper rails, and Table 7-5 for marsh wrens.

In the ERA in support of the removal action, Sprenger et al. (1997) presented risk estimates for all receptor species using acute values and LOAEL values as the basis of the TRVs, as shown in Appendix M, *Hazard Quotient Calculations LCP Site, Brunswick, GA April 1997*. To compare risk estimates derived by the two different assessment approaches, LOAEL values are used in addition to NOAEL values for calculating risk estimates to wildlife species in the removal area in this ERA. The test-species LOAEL values are indicated in Section 6.3.2.1 *Derivation of Toxicity Reference Values for Birds* and Section 6.3.2.2 *Derivation of Toxicity Reference Values for Mammals*, except for Aroclor® 1268 in birds, where the study produced only an unbounded NOAEL and, therefore, no LOAEL can be calculated. The LOAEL TRV doses were scaled using allometry to derive doses for wildlife receptors following the approach used above for scaling of NOAEL values. The LOAEL TRVs are presented in Table 7-2. Risk estimates for wildlife species are shown in Table 7-6. Hazard quotients calculated from exposure models in this risk assessment and from models in Sprenger et al. (1997) appear quite different. These discrepancies are due largely to differing assumptions made between the two approaches with regard to the four important parameters outlined above. Differences between the two risk approaches are discussed below for each species.

7.2.1 Raccoon

Major differences in hazard quotient calculations for raccoons between this risk assessment and that of Sprenger et al. (1997) relate to decisions made with regard to important parameters 1, 2, and 3 described above. These differences are described in detail in this section.

In their exposure models, Sprenger et al. (1997) assumed that all receptors forage exclusively within the area subject to marsh removal actions and, therefore, assigned an area use factor of 1 for all species. Models used in this ERA investigation consider an area use factor of 1 as inappropriate for raccoons because they vacate the marsh during high tides. As noted by Harman and Stains (1979) in their study of raccoons on St. Catherine's Island, Georgia, marsh habitats only comprise about 30 percent of the home range of raccoons. If this empirical data regarding the area use factor was applied to the model

TABLE 7-2. COMPARISON OF ASSUMPTIONS MADE FOR FOOD-WEB EXPOSURE MODELING CALCULATIONS FOR ASSESSMENT OF POTENTIAL FOR UNACCEPTABLE RISK TO SELECTED WILDLIFE RECEPTORS POTENTIALLY INHABITING THE AREA OF MARSH SUBJECT TO REMOVAL ACTIONS

	Raccoon		Clapper Rail		Marsh Wren	
	Sprenger et al. (1997)	This ERA	Sprenger et al. (1997)	This ERA	Sprenger et al. (1997)	This ERA
Body weight (kg)[a]	2.00	2.78	0.25	0.27	0.01	0.00941
Food ingestion rate (kg/day dry weight)[b]	0.16	0.16	0.019 0.028	0.020	0.003	0.0027
Sediment ingestion rate (kg/day dry weight)[a]	0.015	0.015	0.001	0.0020	0.00006	0.000027
Water ingestion rate (L/day)[a]	0.18	0.25	0.07	0.025	0.0027	0.0026
Area Use Factor[a]	1	0.3	1	1	1	1
Diet[a]	Fiddler crab: 90 percent Periwinkles: 10 percent	Fiddler crab: 75 percent Oysters: 20 percent Cordgrass: 5 percent	Fiddler crab: 80 percent Periwinkles: 20 percent	Fiddler crab: 75 percent Periwinkles: 15 percent Terrestrial insects: 10 percent	Grasshopper: 100 percent	Terrestrial insects: 95 percent Periwinkles: 5 percent
PCB LOAEL TRV (mg/kg-day)[a]	0.13	0.49	0.28	--	0.28	--
PCB NOAEL TRV (mg/kg-day)	--	0.10	--	2.47	--	7.56
Mercury LOAEL TRV (mg/kg-day)[a]	0.25	0.080	0.12	0.099	0.12	0.30
Mercury NOAEL TRV (mg/kg-day)	--	0.016	--	0.049	--	0.15

Note: ERA - ecological risk assessment
LOAEL - lowest-observed-adverse-effect level
NOAEL - no-observed-adverse-effect level
PCB - polychlorinated biphenyl
TRV - toxicity reference value

[a] Sprenger et al. (1997) values summarized in Appendix M, *Hazard Quotient Calculations, LCP Site Brunswick, GA, April 1997.*

[b] Sprenger et al. (1997) dry weight food ingestion rates presented in Appendix B, *Exposure Profiles, LCP Site Brunswick, GA, April 1997.*

TABLE 7-3. EXPOSURE VALUES FOR FOOD-WEB MODELING OF POTENTIAL FOR UNACCEPTABLE RISK TO RACCOONS POTENTIALLY INHABITING THE AREA OF MARSH SUBJECT TO REMOVAL ACTIONS

RACCOON (*Procyon lotor*)

Body weight (kg)	2.78	Sediment ingestion rate (kg/day)	0.015
Food ingestion rate (kg-day)	0.16	Water ingestion rate (L/day)	0.25
Food Item	Percent of diet	Area Use Factor	0.3
Fiddler crabs	75		
Oysters	20		
Cordgrass	5		

		Sediment		Water		Fiddler Crab		Oyster		Cordgrass		Total Food	Total
Chemical	Location	Conc. (mg/kg)	Exp. (mg/kg-day)	Conc. (mg/L)	Exp. (mg/kg-day)	Conc. (mg/kg)	Exp. mg/kg-day)	Conc. (mg/kg)	Exp. (mg/kg-day)	Conc. (mg/kg)	Exp. (mg/kg-day)	Exp. (mg/kg-day)	Exp. (mg/kg-day)
Polychlorinated biphenyls (Aroclor® 1268 concentration)	LCP Transect 4	17.4	0.03	0.0005	1.34×10^{-5}	73.2	0.94	6.4	0.022	1.8	0.0016	0.97	0.99
Mercury (methylmercury concentration)	LCP Transect 4	0.58	0.001	1.35×10^{-6}	3.62×10^{-8}	0.62	0.008	1.6	0.005	0.01	1.29×10^{-5}	0.014	0.015

Note: All sediment and tissue concentrations are expressed as dry weight values.

TABLE 7-4. EXPOSURE VALUES FOR FOOD-WEB MODELING OF POTENTIAL FOR UNACCEPTABLE RISK TO CLAPPER RAILS POTENTIALLY INHABITING THE AREA OF MARSH SUBJECT TO REMOVAL ACTIONS

CLAPPER RAIL (*Rallus longirostris*)

Body weight (kg)	0.27	Sediment ingestion rate (kg/day)	0.0020
Food ingestion rate (kg/day)	0.020	Water ingestion rate (L/day)	0.025
		Area Use Factor	1

Food Item	Percent of diet
Fiddler crabs	75
Periwinkles	15
Terrestrial insects	10

		Sediment		Water		Fiddler Crab		Periwinkle		Insect		Total Food	Total
Chemical	Location	Conc. (mg/kg)	Exp. (mg/kg-day)	Conc. (mg/L)	Exp. (mg/kg-day)	Conc. (mg/kg)	Exp. (mg/kg-day)	Conc. (mg/kg)	Exp. (mg/kg-day)	Conc. (mg/kg)	Exp. mg/kg-day)	Exp. (mg/kg-day)	Exp. (mg/kg-day)
Polychlorinated biphenyls (Aroclor® 1268 concentration)	LCP Transect 4	17.4	0.13	0.0005	4.6×10^{-5}	73.2	4.10	3.9	0.044	0.64	0.005	4.15	4.28
Mercury (methylmercury concentration)	LCP Transect 4	0.58	0.0043	1.35×10^{-6}	1.2×10^{-7}	0.62	0.03	0.11	0.0013	0.27	0.0020	0.038	0.040

Note: All sediment and tissue concentrations are expressed as dry weight values.

TABLE 7-5. EXPOSURE VALUES FOR FOOD-WEB MODELING OF POTENTIAL FOR UNACCEPTABLE RISK TO MARSH WRENS POTENTIALLY INHABITING THE AREA OF MARSH SUBJECT TO REMOVAL ACTIONS

MARSH WREN (*Cistothorus palustris*)

Body weight (kg)	0.00941	Sediment ingestion rate (kg/day)	0.000027
Food ingestion rate (kg/day)	0.0027	Water ingestion rate (L/day)	0.0026
		Area Use Factor	1

Food Item	Percent of diet
Terrestrial insects	95
Periwinkles	5

		Sediment		Water		Insect		Periwinkle		Total Food	Total
Chemical	Location	Conc. (mg/kg)	Exp. (mg/kg-day)	Conc. (mg/L)	Exp. (mg/kg-day)	Conc. (mg/kg)	Exp. mg/kg-day)	Conc. (mg/kg)	Exp. (mg/kg-day)	Exp. (mg/kg-day)	Exp. (mg/kg-day)
Polychlorinated biphenyls (Aroclor® 1268 concentration)	LCP Transect 4	17.4	0.05	0.0005	1.4×10^{-4}	0.64	0.17	3.9	0.055	0.23	0.28
Mercury (methylmercury concentration)	LCP Transect 4	0.58	4.0×10^{-5}	1.35×10^{-6}	3.7×10^{-7}	0.27	0.072	0.11	0.002	0.074	0.074

Note: All sediment and tissue concentrations are expressed as dry weight values.

TABLE 7-6. HAZARD QUOTIENT COMPARISON OF THIS ECOLOGICAL RISK ASSESSMENT AND THE ASSESSMENT BY SPRENGER ET AL. (1997) FOR SELECTED WILDLIFE SPECIES POTENTIALLY INHABITING THE AREA OF MARSH SUBJECT TO REMOVAL ACTIONS

	Polychlorinated Biphenyls					Mercury				
	This ERA			Sprenger et al. ERA[a]		This ERA			Sprenger et al. ERA[a]	
Species	Location	NOAEL HQ	LOAEL HQ	Location	LOAEL HQ	Location	NOAEL HQ	LOAEL HQ	Location	LOAEL HQ
Raccoon	Transect 4	9.9	2.0	LCP 17-18	27.3	Transect 4	0.94	0.19	LCP 17-18	1.9
				LCP 19-20	40.5				LCP 19-20	16.7
				LCP 35	26.7				LCP 35	9.2
Clapper Rail	Transect 4	1.7	NC	LCP 17-18	12.0	Transect 4	0.82	0.40	LCP 17-18	5.8
				LCP 19-20	17.5				LCP 19-20	57.1
				LCP 35	11.7				LCP 35	31.0
Marsh Wren	Transect 4	0.04	NC	M-1	56.1	Transect 4	0.49	0.25	M-1	36.6
				M-2	7.7				M-2	11.2

Note: ERA - ecological risk assessment
HQ - hazard quotient
LOAEL - lowest-observed-adverse-effect level
NC - not calculated
NOAEL - no-observed-adverse-effect level

[a] Sprenger et al. (1997).

presented by Sprenger et al. (1997), the hazard quotients for raccoons exposed to PCBs would decrease by 70 percent, although values at all three stations would still be higher than the LOAEL-based hazard quotient reported in this risk assessment. Both assessments report LOAEL-based hazard quotients greater than 1, suggesting a potential risk of exposure to Aroclor® 1268 for raccoons inhabiting the part of the marsh subject to removal actions.

Exposure doses reported by Sprenger et al. (1997) for raccoon exposure to mercury were incorrectly calculated for Locations 19–20 and LCP 35, and the revised hazard quotients reported in Table 7-6 are based on corrected exposure doses. As for hazard quotients for Aroclor® 1268, the hazard quotient for mercury in the model presented by Sprenger et al. (1997) could be adjusted to account for proportional use of the marsh by raccoons, as suggested by an area use factor of less than 1. Mercury hazard quotients calculated from the Sprenger et al. (1997) model are primarily larger than values in this risk assessment because their models base exposure on total mercury concentrations in sediment and prey tissue, but derive TRVs from studies using methylmercury. The assumption that all mercury in sediment and prey tissue exists in the methylated form is a conservative estimate of potential exposure to wildlife receptors. An alternative approach to deriving and applying toxicity thresholds, and the one applied in this risk assessment, is to use methylmercury-based thresholds for all wildlife receptors and use the measured methylmercury concentrations in sediments and prey tissue as the exposure dose.

As indicated in Section 5.3.2.7 *Mercury Speciation*, the maximum concentration of methylmercury in any marsh or channel surface sediment as a percentage of total mercury was 0.59 percent. Applying this value to the total mercury concentration in sediments reduces exposure to mercury through sediment ingestion by 170-fold. Mercury concentrations in prey items reported by Sprenger et al. (1997) in their exposure models are measured as total mercury. On this basis, the body burden in snails at Location 19–20 is very high. Prey tissue data collected at Transect 4 during this risk assessment indicate that methylmercury only comprises about 3.7 percent of the total mercury burden in periwinkles (Figure 4-6). Therefore, it appears appropriate to adjust mercury concentrations in snails by multiplying by a factor of 0.037. Additionally, snail mercury concentrations were not measured by Sprenger et al. (1997) at Location 19–20 (or at Location 35), but were extrapolated from results from Location 17–18 using a biota-sediment accumulation factor (BSAF). Snails collected during this ERA at Transect T-4, which is in the same localized area as Location 19–20, had a wet weight methylmercury concentration of 0.0776 mg/kg. Similarly for fiddler crabs, a major component of the raccoon diet, prey tissue data collected during this ERA from Transect T-4 indicate that methylmercury comprises, on average, about 34 percent of the total mercury burden (Table 1.7-1, Volume II). Therefore, it appears appropriate to adjust mercury content in fiddler crabs by multiplying by a factor of 0.34. Substituting the sediment methylmercury concentration, the periwinkle methylmercury concentration, and the fiddler crab methylmercury concentration into the exposure model; adjusting the area use factor to 0.3; and leaving other parameters unchanged would substantially reduce the hazard quotients for mercury in raccoons that are reported in Sprenger et al. (1997).

7.2.2 Clapper Rail

Major differences in hazard quotient calculations for clapper rails between this risk assessment and that by Sprenger et al. (1997) relate to decisions made with regard to important parameters 1, 3, and 4 described above. These differences are described in detail in this section.

For PCBs and mercury, variations in the sediment concentration used to estimate the dose received via incidental sediment ingestion account for some of the differences in hazard quotients derived from the two risk evaluations. Sprenger et al. (1997) used station-specific sediment concentrations to develop a concentration-response relationship. This study used an average sediment concentration calculated for the home range of a species. The latter approach was considered to better reflect the dose received via sediment ingestion for an individual bird that forages over its entire home range.

LOAEL-based hazard quotients for clapper rail exposure to PCBs as calculated by Sprenger et al. (1997) are about 7- to 10-fold larger than NOAEL-based hazard quotients calculated in this risk assessment. The study used to derive the Aroclor® 1268 TRV in this assessment (Lillie et al. 1974) did not determine a LOAEL concentration, and a comparison cannot be made on this basis. The differing risk estimates result, in part, because the PCB TRV cited by Sprenger et al. (1997) used Clophen® A50 as a surrogate for Aroclor® 1268 and the Clophen® A50 TRV is almost 10-fold lower than the TRV used in this risk assessment, which was based on the PCB of interest—Aroclor® 1268.

In selecting TRVs used in this risk assessment, several factors were weighted in the review of toxicological literature for selection of the most appropriate TRV. The order in which these factors are presented does not represent a hierarchical ranking, because all factors are considered simultaneously. Factors include:

- Taxonomic relationship between the test organism and the receptor species
- Use of laboratory organisms or domesticated species
- Ecological relevance of the study endpoints
- Route of exposure tested
- Duration of exposure tested
- Standardization of definitions and methods
- Study design and statistical robustness
- Data reporting standardization and completeness.

Based on these considerations, the study by Lillie et al. (1974) was selected for derivation of a TRV for Aroclor® 1268, instead of the study by Ulfstrand and Södergren (1971) that was used by Sprenger et al. (1997) as the basis of their TRV. The Ulfstrand and Södergren (1971) study used a Clophen® mixture, not an Aroclor® mixture, and the endpoints measured (i.e., abnormal nocturnal behavior and activity patterns) were considered a less relevant indicator of potential population-level responses than egg production and fertility, which were the measurement endpoints of the study by Lillie et al. (1974).

The TRVs used for mercury also differ between Sprenger et al. (1997) and this evaluation of risk. The TRV used in this study was based on a multigenerational study in mallard ducks with reproductive success as the measurement endpoint (Heinz 1979). This study is considered more ecologically relevant than Nicholson and Osborn (1984), which reported kidney lesions in juvenile starlings as the measurement endpoint, and which was used by Sprenger et al. (1997) as the basis of their TRV. The LOAEL values for the two studies are similar, however, and thus, the TRV selected is not a major factor contributing to differences in hazard quotients between the two assessments. As noted above for raccoons, Sprenger et al. (1997) calculated the exposure dose based on total mercury concentrations in sediment and prey, while the approach used in this evaluation was to use methylmercury concentrations because the TRV is based on methylmercury. Using the same approach that was used for raccoons (i.e., adjusting only the concentration of mercury in sediment, periwinkles, and fiddler crabs for the proportion of methylmercury and leaving all other exposure parameters the same as in Sprenger et al. [1997]) would substantially reduce the hazard quotients (reported in Sprenger et al. [1997]) for mercury in clapper rails.

7.2.3 Marsh Wren

Major differences in hazard quotient calculations for marsh wrens between this risk assessment and that by Sprenger et al. (1997) relate to decisions made with regard to important principles 1, 3, and 4 described above. These differences are described in detail in this section.

The studies used to derive TRVs for PCBs and mercury in marsh wrens are the same as those that were used for clapper rails, and the comments on the studies that were noted in the previous section apply equally for this receptor. For PCBs and mercury, variations in the sediment concentration used to estimate the dose received via incidental sediment ingestion account for some of the differences in hazard quotients derived from the two risk evaluations. Sprenger et al. (1997) used station-specific sediment concentrations to develop a concentration-response relationship. This study used an average sediment concentration calculated for the home range of a species. The latter approach was considered to better reflect the dose received via sediment ingestion for an individual bird that forages over its entire home range.

As discussed for clapper rails, different estimates of the TRV also greatly influence the results of the two exposure models for PCBs. Sprenger et al. (1997) cited a LOAEL-based TRV of 0.28 mg/kg-day, based on Clophen® A50, while the TRV used in this assessment was 7.56 mg/kg-day, based on the NOAEL for Aroclor® 1268. Differing assumptions regarding Aroclor® 1268 concentrations in ingested sediment and TRVs account for the large discrepancies between assessments for hazard quotients in marsh wrens.

Comparison of mercury hazard quotients for marsh wrens uses the same approach as described above for clapper rails. Sprenger et al. (1997) calculated the exposure dose based on total mercury concentrations in sediment and prey, while the approach used in this evaluation was to use methylmercury concentrations because the TRV is based on methylmercury. Using the same approach that was used for clapper rails (i.e., adjusting the concentration of mercury in sediment, periwinkles, and fiddler crabs for the proportion of methylmercury, and leaving all other exposure parameters) would substantially reduce the hazard quotient (reported in Sprenger et al. [1997]) for mercury in marsh wrens.

7.2.4 Conclusion

In conclusion, as discussed for the portion of the marsh subject to the removal actions, differences in risk estimates derived by Sprenger et al. (1997) and this risk assessment are not generally attributable to different assumptions about species-specific exposure parameters. The only exception is for raccoons where different area use factor values influence the risk estimates. The major reasons for differences in risk estimates between the two reports are due to differences in the TRVs, which are used to indicate effect for both PCBs and mercury, and differences in assumptions about the concentration of toxicologically important forms of mercury in sediment and selected prey tissue. The same reasons would also apply to consideration of potential ecological risk for the full marsh area as was done in this risk assessment.

7.3 UNCERTAINTY ANALYSIS

There are several major sources of uncertainty related to the assessment of risk to aquatic life and wildlife receptors at the LCP Site. These sources of uncertainty include parameters, model choice, and scenarios:

- Parameters
  - Representativeness of sampling locations
  - Representativeness of sample sizes of prey

- Characterization of body burdens in prey
- Exposure estimates for receptors

- Models
  - TRVs for receptors
- Scenarios
  - Choice of receptors.

7.3.1 Representativeness of Sampling Locations

Sampling locations, particularly for sediment, were unevenly distributed throughout the marsh, with a greater emphasis placed on locating stations in and around the area of the marsh south of the causeway and immediately west of the industrialized part of the plant. Areas of the Site west of Purvis Creek and north of the causeway, where CoPC concentrations are low, were less intensively sampled, particularly inner marsh areas. Averaging more sample station locations from the areas with highest contamination overstates the degree of contamination for receptors that forage over the entire marsh.

Reference station locations used for the surface water investigation had higher salinities than stations at the LCP Site. Higher salinities at these reference stations are probably attributable to their proximity to open ocean water and might confound interpretation of the results of toxicity and bioaccumulation studies conducted at these stations.

7.3.2 Representativeness of Sample Sizes of Prey

Chemical concentrations reported in prey species are based on an analysis of one composite sample per species at each transect. Compositing and the lack of replicate samples do not permit a statistical evaluation of the variation in chemical concentrations among individuals of a species. However, composite samples reflect the mean chemical concentrations to which a predator is exposed due to consumption of multiple prey items, during a single predation event or over a series of events. Therefore, chemical concentrations in composite samples are likely representative of concentrations in the wider prey populations at each transect.

7.3.3 Characterization of Body Burdens in Prey

Prey species were typically collected from three widely distributed transects in the marsh. However, oysters were not collected at the transects outside the removal area (i.e., Transects 1, 2, and 3), and terrestrial insects were collected at only Transect 2; thus, CoPC body burdens in these species were incompletely characterized. Concentrations of CoPCs

in native oysters collected as part of the surface water investigation were used in food-web exposure models to estimate concentrations in oysters elsewhere in the marsh. Although oysters were not collected at transect locations, the sample locations used provide good geographic coverage of the marsh. Therefore, body burdens in oysters from the six surface water sampling stations are probably reflective of levels in oysters that potentially occur at the transects.

Concentrations of CoPCs in killifish are determined based on analysis of individuals collected by EPA at one station (LCP-43) outside the section of the marsh subject to removal actions. Concentrations in killifish from this station are considered representative of concentrations in killifish from other areas of the marsh outside the removal area for the purposes of food-web exposure models used in this risk assessment. Thus, the models do not incorporate potential spatial variation in CoPC concentrations in killifish tissue. River otters and wood storks are considered to prey on killifish and to forage over the entire marsh. For these two receptors, exposure through consumption of killifish may be underestimated or over-estimated depending on how accurately the CoPC concentrations measured in killifish from Station LCP-43 predict concentrations in individuals from elsewhere in the marsh.

Body burdens in spot are based on analysis of fillets, not whole bodies. This might result in underestimation of CoPC concentrations in this species, particularly for PCBs, because they partition preferentially into lipids, which are likely to be of lesser constitutive importance in muscle tissue than other body compartments. If this assumption is true, exposure through the diet may be under-represented for river otters and wood storks because these species are both considered to prey on spot.

7.3.4 Exposure Estimates for Receptors

Exposure estimates are based on assumptions about the life history characteristics of the receptor species. In the absence of site-specific information on body weight, home range size, diet preferences, and food, water, and sediment ingestion rates for the receptors, information was obtained from literature sources or discussions with recognized experts for the various species. In several cases, in the absence of available information, best professional judgment was used to determine exposure parameters. The estimates used in the exposure models are considered reasonable, but because detailed information on life history parameters was not always available, a moderate degree of uncertainty is introduced into exposure estimates.

Area use factors are applied in the food-web exposure models to reflect the ratio of LCP Site area to home-range size of a receptor as reported in the literature. The home-range estimates presented in the literature represent the total space a receptor uses, which may incorporate areas receptors pass through, but where a receptor does not forage. As such, these estimates may underestimate the importance of preferential foraging areas. If the marsh at the LCP Site is a preferential feeding area for any of the receptors that were

modeled with an area use factor less than one, this could underestimate the actual exposure for these species.

A time use factor is applied in the manatee food-web exposure model to reflect that manatees migrate south to warmer waters during winter. To the extent that a manatee may occur year-round in the Brunswick area, application of a time use factor of less than 1 could underestimate the actual exposure for this species.

7.3.5 Toxicity Reference Values for Receptors

In comparison to more environmentally relevant Aroclors, Aroclor® 1268 has received less toxicological investigation, although some vertebrate test data are available. A peer-reviewed study of dose-based toxicity of Aroclor® 1268 to birds (Lillie et al. 1974) is available and was used directly to develop avian TRVs. Lillie et al. (1974) examined effects of seven Aroclor mixtures, including Aroclors® 1254 and 1268, on reproductive effects in hens. Based on results presented in the paper, NOAEL doses for Aroclors® 1254 and 1268 are 0.127 mg/kg-day and 1.28 mg/kg-day, respectively, suggesting that Aroclor® 1268 is about 10-fold less toxic than Aroclor® 1254.

No references were identified in the peer-reviewed literature regarding dose-based toxicity of Aroclor® 1268 that were useful in the derivation of TRVs for mammals. One study was identified that examined mammalian toxicity of Aroclor® 1268 relative to other PCBs. Kholkute et al. (1994) examined the effects of Aroclors® 1221, 1254, and 1268 and 3,3′,4,4′-tetrachlorobiphenyl (TCB) on *in vitro* fertilization in laboratory mice. Aroclors® 1221 and 1268 and TCB reduced oocyte fertilization rate at 1 μg/mL, while Aroclor® 1254 caused significant inhibition of fertilization at 0.1 μg/mL.

For mammals, TRVs should be developed systematically, based on the toxicity of Aroclor mixtures. Application of mammalian TRVs for more toxic Aroclor mixtures as compared to Aroclor® 1268 avoids inter-class extrapolations from birds to mammals but adds a conservative uncertainty to risk calculations. Indeed, the substantial differences in measured toxicity between Aroclor® 1268 and other Aroclors observed in mammals (Kholkute et al. 1994) and birds (Lillie et al. 1974) provides strong evidence that hazard quotients estimated in this ERA report for Aroclor® 1268 in mammals represent the most conservative estimate of site-specific ecological hazards and that, in the absence of a mammalian Aroclor® 1268 TRV, hazard quotients are probably overestimated by about an order of magnitude.

TRVs reported for test species are extrapolated to wildlife species by allometric scaling. Scaling of TRVs is based on the assumption that response to toxicants is a function of metabolic rate, which is a function of body size. Smaller animals with higher metabolic rates are assumed to be more resistant to toxic effects of chemicals because of more rapid detoxification, and hence have higher toxicity thresholds (higher TRVs) than larger-bodied

animals. If this fundamental assumption is incorrect, then use of this allometric relationship may lead to underestimation or overestimation of the true toxicity threshold.

7.3.6 Choice of Receptors

Receptors used in predictive risk assessments were selected because they were 1) common salt marsh species, 2) species occupying upper trophic levels of the salt marsh food web, or 3) threatened or endangered species that potentially occur in the marsh. Some species are commonly observed at the LCP Site marsh (e.g., clapper rails, marsh wrens), whereas others, although potentially occurring, are not commonly observed (e.g., river otter) and the extent to which these receptors use the Site is unknown. The frequency of occurrence of several of the threatened or endangered species reported to occur at, or near, the Site is also unknown. A manatee has been seen once in the Turtle River near the LCP Site. Occasional sightings of sea turtles have been reported in the Turtle River, but these are likely transient individuals that do not use the area regularly. The extent to which species seen in the Turtle River use the habitat at the Site is also not known.

Upper trophic-level receptors were also selected as representative species for assessing risk to similar species with similar foraging strategies and life history characteristics. If no significant risk is determined for the modeled receptors, it is also assumed that no risk exists for similar species. Differences in preferred prey and differences in habitat use among similar species may introduce some uncertainty in extrapolating results from a receptor to ecologically similar species.

7.4 CONCLUSIONS

This site-specific examination of chemical risks indicates that upon completion of the ongoing removal actions, there are no potential risks for any assessment endpoint except for exposure of piscivorous mammals, as represented by river otters, to Aroclor® 1268 and mercury. This potential risk to otters from exposure to Aroclor® 1268 is not considered to be ecologically significant given the uncertainty of basing the TRV on Aroclor® 1254. For mercury, the risk is based primarily on exposure through consumption of blue crabs. Although current conditions are generally protective of potential chronic risks, removal actions either already completed or planned for the LCP Site will further increase the margin of safety and lessen the potential for unacceptable risk below levels described in this site-specific reassessment. Completed removal actions have reduced plant site PCB and mercury transport to the marsh, and the planned marsh removal action will eliminate a substantial amount of PCB and mercury mass, which is currently available for potential environmental receptor exposure. The planned removal action, in combination with completed removal actions, will serve to substantially reduce PCB and mercury input from the LCP Site to the food web, resulting in the reduction of PCB and mercury estimated exposure and associated body burdens with the passage of time.

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