# New Jersey Meadowlands Commission

## Meadowlands Environmental Research Institute
New Jersey

# Screening Level Ecological Risk Assessment of Contamination in Wetlands Considered for Restoration in Hackensack Meadowlands District

## Final Report

Photo courtesy of NJMC/MERI

**ENSR International**

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# LIST OF ACRONYMS

| | |
|---|---|
| °C | Degrees Centigrade |
| AFDW | Ash Free Dry Weight |
| As | Arsenic |
| ASTM | American Society of Testing and Materials |
| AUF | Area Use Factor |
| AWQC | Ambient Water Quality Criteria |
| BAF | Bioaccumulation Factor |
| CBR | Critical Body Residue |
| Cd | Cadmium |
| COC | Chain-Of-Custody |
| COPC | Chemicals Of Potential Concern |
| Cr | Chromium |
| CSM | Conceptual Site Model |
| Cu | Copper |
| DQA | Data Quality Assessment |
| DQO | Data Quality Objective |
| ED | Exposure Duration |
| EEQ | Environmental Effects Quotient |
| EPC | Exposure Point Concentration |
| ERA | Ecological Risk Assessment |
| ERED | Environmental Residue Effects Database |
| ER-L | Effects Range-Low |
| ER-M | Effects Range-Median |
| GC/MS | Gas Chromatography/ Mass Spectrometry |
| GIS | Geographical Information Service |
| H’ | Shannon-Wiener Diversity Indices |
| Hg | Mercury |
| HQ | Hazard Quotient |
| ID | Identification |
| IRM | Information Resource Management |
| J’ | Pielou’s Evenness |
| KFM | Kearny Marsh |
| L | Least |
| LCS | Laboratory Control Sample |
| LCV | Lowest Chronic Value |
| LEL | Low Effect Level |
| LOAEL | Low Observed Adverse Effect Levels |
| LOEL | Lowest Observed Effect Levels |
| M | Moderate |
| MERI | Meadowlands Environmental Research Institute |
| MS/MSD | Matrix Spike/Matrix Spike Duplicate |
| Ni | Nickel |
| NJMC | New Jersey Meadowlands Commission |
| NOAEL | No Observed Adverse Effect Levels |
| NOED | No Observable Effects Dose |
| ORM | Oritani Marsh |
| ORNL | Oak Ridge National Laboratory |
| PAH | Polynuclear Aromatic Hydrocarbons |

| | |
|---|---|
| Pb | Lead |
| PBT | Persistent, Bioaccumulative, Toxic |
| PCB | Polychlorinated Biphenyls |
| PE | Performance Evaluation |
| PEC | Probable Effect Concentration |
| PRG | Potential Remedial Goals |
| PML | Protective Media Level |
| PT | Proficiency Testing |
| QA | Quality Assurance |
| QA/QC | Quality Assurance/Quality Control |
| QAM | Quality Assurance Manager |
| QAPP | Quality Assurance Project Plan |
| QC | Quality Control |
| RBM | Riverbend Marsh |
| RPD | Relative Percent Difference |
| RUF | Root Uptake Factor |
| S | Severe |
| SAW | Sawmill Creek Wildlife Management Area |
| SCV | Secondary Chronic Value |
| SEL | Severe Effect Level |
| SHS | Secaucus High School Marsh |
| SLERA | Screening-Level Ecological Risk Assessment |
| SOP | Standard Operating Procedure |
| SQL | Sample Quantitation Limit |
| TDD | Total Daily Dosage |
| TEC | Threshold Effect Concentration |
| TOC | Total Organic Carbon |
| TRVs | Toxicity Reference Values |
| U.S. EPA | U.S. Environmental Protection Agency |
| USACE | U.S. Army Corps of Engineers |
| WMA | Wildlife Management Area |
| Zn | Zinc |

# EXECUTIVE SUMMARY

The NJMC owns or manages approximately 3400 acres of wetlands (Figure 1), and is actively acquiring more wetlands for preservation and/or restoration. The objective of this Project is to assess the ecological risks from contamination at these sites to help guide decisions about restoration of these sites and to build capacity to comprehensively assess additional wetlands in the future. The Project included the development of a database of historic data, a screening level ecological risk assessment (SLERA) for several trophic levels, development of rapid assessment curves based on food web modeling, and efforts to correlate contaminant concentrations with measures of benthic community health and laboratory measured toxicity.

The objective of this Project is to assess the ecological risks from contamination at the Meadowlands wetlands to help guide decisions about restoration of these wetlands and to develop a capacity to assess additional wetlands which may be considered for acquisition or restoration.

A sampling program was completed to supplement existing data available from previous recent wetland studies. Five study wetland were selected for sampling based on a review of historic data, and included Kearny Freshwater Marsh (KFM), Oritani Marsh (ORM), Riverbend Marsh Wetland Preserve (RBM), Sawmill Creek Wildlife Management Area (SAW), and Secaucus High School Marsh (SHS). Sampling locations are identified in the table below. Samples were collected for toxicity testing and bulk chemistry analysis. Samples were also collected at RBM and KFM for benthic community analysis, as no benthic data had been previously collected for these wetlands. The sediment samples were analyzed for total metals (cadmium, Cd; chromium, Cr; copper, Cu; mercury, Hg; lead, Pb; nickel, Ni; zinc, Zn; arsenic, As), pesticides, polychlorinated biphenyls (PCB's), polynuclear aromatic hydrocarbons (PAHs), total organic carbon (TOC), grain size, % moisture and pH by the MERI Laboratory (Cd, Cr, Cu, Fe, Pb, Ni, Zn, Hg, grain size) and AccuTest Laboratory (all other parameters). Toxicity testing included both 28-day and 10-day growth and survival tests. Study design and methods were detailed in a Quality Assurance Project Plan (QAPP) that was approved by U.S. Environmental Protection Agency (U.S. EPA) prior to sampling.

| Location | Description |
|---|---|
| RBM-01 | In open water 30' wide area bounded by common reed (*Phragmites australis*) and punctuated by *Spartina alterniflora* hummocks. |
| RBM -02 | In mosquito ditch, 6-8' wide and bounded by thickly growing *Phragmites* |
| SHS -01 | Along the main channel within the marsh |
| SHS -02 | Along the main channel within the marsh |
| SHS -03 | Along the main channel within the marsh |
| SAW -01 | Northeast side of channel (roughly 80' wide) approximately 1500 feet from confluence with the Hackensack River |
| SAW -02 | Open water area near western spur of NJ Turnpike |
| ORM -01 | In a tidal creek along the southwest border of the marsh |

| Location | Description |
|---|---|
| ORM -02 | In a tidal pool along the southeast side of the marsh |
| KFM -01 | Along western edge of the marsh near cattails and loosestrife |
| KFM -02 | In middle of the wetland in open water |
| KFM -03 | Near the railroad bed at the northern end of the site near islands of *Phragmites*. |

The following provides a brief summary of key conclusions drawn from this project:

**Benthic Community**

The benthic community found at Riverbend included faunal elements that are typical of high estuarine, low salinity marshes throughout the eastern United States. There is no evidence of pollution or stress-related indicator species. The benthic community at Kearny Marsh appeared to be impoverished. More than 55 benthic taxa have been identified from the five marshes for which benthic data are available. All of these taxa observed during the current Project and past studies are commonly found in estuarine systems. The most commonly represented groups included 12 insects, 12 polychaetes, 8 amphipods, 5 isopods, 4 gastropods, and 3 bivalves. Density, diversity, and evenness varied considerably within the wetlands sampled as part of past studies (Mill Creek, Oritani, and Secaucus High School marshes) with an order of magnitude difference in density between locations in a given wetland. Previous studies included sampling of mudflats, tidal creeks, marsh plain, ponded/open water, channels, high marsh, and mosquito ditches. Samples collected as part of this study were primarily located in open water, mosquito ditches, and/or channels. The diversity of habitat types makes comparison among the wetlands more difficult.

**Toxicity Testing**

Toxicity testing results were mixed. None of the test sediments produced 100% mortality. As might be expected, there were differences in survival and growth between the 28-day and 10-day test results. Survival and growth were not always similar (e.g., diminished survival and growth in the same sample). The 28-day mean survival rate for SAW-01, RBM-01, and SHS-01 were less than and significantly different than those for the negative control. RBM-01 exhibited greater than 50% mortality for the 28-day test, while greater than 50% mortality was reported for ORM-02 and SHS-01 in the 10-day tests. More than 25% mortality was observed for SAW-01, ORM-01, RBM-02, and SHS-01 during the 28-day tests and for samples from ORM-01, RBM-02, and SHS-02 for the 10-day test. However the survival in RBM-02 and SHS-02 were not significantly different from the control. All samples from Kearny Marsh shown significantly lower growth than the control for the 10-day test. This same pattern was not evident in the 28-day test. Survival was not significantly different from the control for any of the samples in the 28-day test, but was significantly different in KFM-01 for the 10-day test. SHS-02 and SHS-03 sediments showed no significant difference in survival or growth from the controls in either the 28-day or 10-day tests.

## Screening Level Ecological Risk Assessment (SLERA)

Evaluation of sediment quality in the five study wetlands indicated widespread exceedance of benthos-based sediment screening values for metals in all wetlands, with the exception of chromium and iron in Kearny and Oritani Marshes. Several of the metals are also known to bioaccumulate (cadmium, chromium, mercury). The pesticides 4,4'-DDE and alpha-chlordane were selected as Contaminants Of Potential Concern (COPCs) since their maximum concentrations provided the greatest exceedances among detected pesticides (e.g., DDT and related fractions). The pesticide 4,4'-DDE is distributed widely throughout the wetlands, whereas alpha-chlordane was detected in Kearny Marsh. PAHs were selected as they are a widely distributed contaminant in the wetlands and many of the individual PAH fractions were above the screening level, as was the cumulative total PAHs (tPAHs). PCBs were detected, primarily as Aroclor 1248, 1254, and 1260, and were retained as a COPC due to concern regarding their levels and inherent bioaccumulative properties and potential adverse effects on higher wildlife. Similar results were observed for data from previous wetland studies, however the magnitude of exceedance was typically much greater for sediment data from past studies.

Evaluation of sediment quality in the five study wetlands indicated widespread exceedance of phytotoxicity-based sediment screening values. Organic contaminants do not exceed the available benchmarks for tPAHs and tPCBs. No phytotoxicity values were available for the pesticides. Comparison of the metals to available phytotoxicity screening benchmark values indicated widespread exceedances for the metals with the exceptions of cadmium and iron. Similar results were observed for data from previous wetland studies.

Four representative wildlife species were evaluated using a maximum Hazard Quotient (HQ). Food web models indicate potential concern regarding maximum exposure of all COPCs except chromium, alpha-chlordane, 4,4'-DDE, and tPAHs for the mink, and all COPCs except arsenic, alpha-chlordane, and tPAHs for the heron. The results for muskrat indicate potential concern regarding exposure of arsenic (HQ = 2.55) and mercury (HQ = 1.47), however the magnitude of the HQ suggests only a slight potential for risk. The results of the food web models for mallard indicate potential concern regarding exposure of mercury (HQ = 1.47), but not other COPCs.

## Wildlife Risk Curves

Wildlife risk curves were developed for the four target receptor species, by modeling the total daily dose over a range of sediment and water concentrations. Sediment data from the current Project as well as past studies were pooled and 25$^{th}$, 50$^{th}$, 75$^{th}$ and 100th percentile concentrations were calculated for each COPC. Historic water quality data were also pooled, and similar percentile concentrations for water concentrations determined. These sediment and water concentrations were then input into the food web models resulting in the prediction of four corresponding total daily doses (TDD) for each COPC and receptor species. These predicted TDDs were then compared to Toxicity Reference Values (TRVs) to calculate a Hazard Quotient. The calculated HQs were then plotted against the sediment concentrations to form the wildlife risk curves. These curves allow for the analysis of the range of potential ecological risk that may result over the observed range of sediment concentrations in the Meadowlands wetlands.

The wildlife risk curves indicate that the potential risk associated with different COPCs differs significantly in magnitude and pattern of risk with increasing sediment concentrations and that these patterns also differ depending on the receptor/trophic pathway involved. In some cases the level of estimated risk is so low for some receptor-COPC combinations that it is near zero. In general, greater risk are predicted for the piscivores (heron, mink) than for the herbivores (mallard, muskrat) over the majority of the COPCs, with the mink having the highest HQs and the mallard the lowest. Higher risks are associated with the metals than with the organic COPCs. For most COPCs (cadmium, chromium, copper, mercury, zinc, alpha-chlordane, 4,4'-DDE, tPCBs), the risk levels rise slightly as sediment concentrations rise from 25th to 75th percentile, with a large increase as the 100th percentile is reached. Comparison of the median (50th percentile) HQ indicates that little or slight risk (HQ < 2) would be predicted for most COPCs under average sediment and surface water concentrations with the exception of arsenic.

**Relationships Between Benthic Community, Contaminants, and Toxicity**

No clear and definitive relationships emerge between sediment toxicity or sediment chemistry and benthic community parameters. The lack of obvious relationship may be caused by a number of factors including physical habitat, water chemistry, and potential overestimation of risk from sediment chemistry, and differences between the sensitivity of test organisms and indigenous populations to sediment contaminants. Likewise, it is possible that the sample sites selected reflect only a small segment along the continuum of sediment contaminant concentrations.

This Project aimed to development methods that would support rapid assessment of potential ecological risk in wetlands that NJMC might be considering for acquisition, management or restoration. The findings of the study point to the need for additional work to improve the predictive capacity of wildlife risk curves as well as screening for benthic/aquatic life risk.

# 1.0 INTRODUCTION

The Hackensack Meadowlands District that encompasses more than 30 square miles, and contains about 8,400 acres of open space, wetlands (mostly brackish tidal marshes) and waterbodies located in a heavily urbanized and industrialized setting. These 8,400 acres were rezoned for conservation because of the Master Plan developed by the New Jersey Meadowlands Commission (NJMC, a New Jersey State agency). As a result of activities such as development, dredging, draining, mosquito control, landfilling, and industrial pollution, wetlands have been modified and contaminated to varying degrees. Contaminants such as pesticides, polychlorinated biphenyls (PCBs), polynuclear aromatic hydrocarbons (PAHs) and heavy metals have been detected in wetlands. These contaminants, particularly bioaccumulative ones, may pose risks to fish and wildlife feeding and reproducing in the wetlands.

The NJMC owns or manages approximately 3,400 acres of wetlands, and is actively acquiring more wetlands for preservation and/or restoration. The NJMC and others have conducted baseline studies of several marshes in the Meadowlands that included chemical analyses of water and sediments/soils and biologic inventories. These sites range from areas that are relatively clean to areas that may have high concentrations of contaminants. Some of these wetlands have already been restored; others are currently or potentially under consideration for restoration. Restoration includes activities such as restoration of appropriate hydrology, the creation of biological habitats, the control of invasive plant species, and the reintroduction of native marsh vegetation.

The objective of this Project is to assess the ecological risks from contamination at the Meadowlands wetlands to help guide decisions about restoration of these wetlands and to build capacity to comprehensively assess additional wetlands in the future. ENSR International with the NJMC/Meadowlands Environmental Research Institute (MERI) undertook this project. Specific objectives of the project are to:

1. Develop a geographic database of the chemical and biological data that has already been collected during environmental assessments of several marshes in Meadowlands;

2. Apply U.S. EPA screening level ecological risk assessment (SLERA) for each contaminant detected at each marsh and identify contaminants of potential concern (COPCs) for additional food web modeling to determine if animals at several trophic levels using the marshes may be at risk;

3. Apply the procedure over a range of concentrations representative of the Meadowlands for each contaminant for the same set of animals to develop “standard curves” relating sediment contaminant concentration to a measure of ecological risk; this may be used to quickly estimate ecological risk in future assessments;

4. Correlate benthic diversity with sediment contaminant concentrations to look for biologic affects of contamination;

5. Conduct sediment toxicity tests to test the validity and predictive power of ecological risk screening.

This report documents the work completed and major findings of the project. Section 2.0 describes the sampling program and methods employed in sample analysis. Section 3.0 documents the results of field measurements, benthic community analysis, sediment chemistry, and toxicity testing. Section 4 discusses the results of ecological risk assessment and the development of risk curves to allow for prediction of potential toxicity. Section 5.0 discusses the development of risk curves to aid in rapid assessment of potential risk. Section 6.0 provides a qualitative discussion of interrelationships between benthic community composition, sediment chemistry and whole sediment toxicity testing. Uncertainty and limitations of the analyses and conclusions are discussed throughout the report and are summarized in Section 7.0. Project conclusions are provided in Section 8.0. A list of acronyms used throughout this document is provided on page vii.

# 2.0 SAMPLING PROGRAM

## 2.1 Sampling Design Overview

The purpose for the sample collection is to provide the NJMC/MERI with data that can be used to assess the ecological risks from contamination at sites in the Meadowlands. These assessments will help guide decisions about restoration and build capacity to comprehensively assess additional wetlands in the future. Samples were collected for toxicity testing, bulk chemistry analysis and benthic community analysis. Design and methods were detailed in a Quality Assurance Project Plan (QAPP) that was approved by U.S. Environmental Protection Agency (U.S. EPA) prior to sampling. Sampling and analyses were completed in accordance with this approved QAPP.

## 2.2 Study Area

NJMC owns and manages a number of wetland areas including Kearny Freshwater Marsh (KFM), Oritani Marsh (ORM), Riverbend Marsh Wetland Preserve (RBM), Sawmill Creek Wildlife Management Area (SAW), Secaucus High School Marsh (SHS), Skeetkill Marsh (SM), Harrier Meadows (HM), Mill Creek (MC), and 8-Day Swamp (8-Day). Five wetlands of interest were identified for sampling based on a review of available data. These wetlands include Kearny Freshwater Marsh (KFM), Oritani Marsh (ORM), Riverbend Marsh Wetland Preserve (RBM), Sawmill Creek Wildlife Management Area (SAW), and Secaucus High School Marsh (SHS). Brief descriptions are provided in the following sections based on the descriptions and summaries provided in *Hackensack Meadows, New Jersey, Biodiversity: A Review and Synthesis* (Hudsonia, 2002). Additional information on each of the individual wetlands is available on the NJMC/MERI "Digital Meadowlands" electronic database.

Kearny Freshwater Marsh (KFM)

Kearny Freshwater Marsh is an approximately 311 acres (ac) marsh bounded by the defining upland ridge to the west, NJ transit Boonton Line and the Belleville Turnpike to the north, the Transco pipeline to the east, and the abandoned railroad to the south.. It provides a freshwater to slightly brackish (mean salinity of 1.8 ppt; HMDC, 2002) environment. The marsh was formerly dominated by common reed (*Phragmites australis*), but when a pumping station on the Hackensack River was retired, runoff (and poorly maintained culverts) caused the water level in this area to risk, slowly killing the once dominant *Phragmites*. The site is now mostly open water with islands of vegetation. Dominant wetland species include common reed, broad-leaved cattail (*Typha latifolia*), purple loosestrife (*Lythrum salicaria*), rose mallow (*Hibiscus palustris*), and marsh fleabane (*Pluchea odorata*). In the open water environments duckweed (*Lemna minor*), water shield *(Brasenia schreberi*), arrow *arum (Peltandra viriginica*), pickerelweed (*Pontederia cordata*) and white (*Nymphaea odorata*) and yellow-water lily (*Nuphar spp.*) are found. (Hudsonia, 2002). This marsh is the historic and current site of important rookeries for black-crowned and yellow-crowned night-herons, respectively (Kane and Gibbons, 1997).

Oritani Marsh (ORM)

Oritani Marsh is a 225-acre triangle of the Berry's Creek Marsh bounded by NJ Transit Bergen Line, Berry Creek Canal and the western spur of the NJ Turnpike. Part of this site is tidally influenced with most tidal areas with mosquito control ditches. Its flora is dominated by thickly growing *Phragmites*, although stands of saltmeadow cordgrass (*Spartina patens*) still persist. Other important species include cattails, spikerush (*Eleocharis* spp.), and marsh-marigold (*Caltha palustris*) (Hudsonia, 2002).

Riverbend Marsh Wetlands Preserve (RBM)

Riverbend Marsh Wetland Preserve is an estuarine marsh of 58 ac located along the eastern side of the Hackensack River in the southeast portion of Secaucus. It is adjacent to Malanka Landfill in South Secaucus. The site is dominated by common reed, but has patches of high marsh vegetation including saltmeadow cordgrass, spikegrass, (*Distichlis spicata*) and glasswort (*Salicornia sp.*) (Hudsonia, 2002).

Secaucus High School Marsh (SHS)

The Secaucus High School Marsh is a small portion (38 ac) of a once much larger estuarine marshland located along the eastern side of the Hackensack River in the northern portion of Secaucus. This marsh is dominated by a *Phragmites* monoculture, except along the Hackensack where small patches of *Spartina* sp. are present (TAMS, 2001a).

Sawmill Creek Wildlife Management Area (SAW)

Sawmill Creek Wildlife Management Area is a large (741 ac) complex of estuarine marsh and mudflat/open water located between the western spur of the New Jersey Turnpike and the Hackensack River in Kearny and Lyndhurst. The marsh vegetation is dominated by saltmarsh cordgrass (*Spartina alterniflora)*, saltmeadow cordgrass, and common reed (Hudsonia, 2002). This state-protected wetland is a prime site for migrant and over-wintering waterfowl.

### 2.3 Sediment Sampling Locations

Figure 2-1 presents the wetland study areas that include one freshwater and eight estuarine marshes owned or managed by NJMC. These wetlands include: Secaucus High School Marsh (SHS), Oritani Marsh (ORM), Riverbend Marsh Wetlands Preserve (RBM), Sawmill Creek Wildlife Management Area (SAW), Kearny Freshwater Marshes (KFM), Mill Creek (MC). Skeetkill Marsh (SM), and Harrier Meadow (HM). The sediment sampling locations selected for analysis included locations in Secaucus High School Marsh (SHS), Oritani Marsh (ORM), Riverbend Marsh Wetlands Preserve (RBM), Sawmill Creek Wildlife Management Area (SAW), and Kearny Freshwater Marshes (KFM). The two sediment sampling locations at Sawmill Creek were identified based on historic information as “reference” locations. Figures 2-2 through 2-6 illustrates the sample locations within each wetland.

Sampling sites for sediment chemistry analysis and toxicity tests were selected after a review of the existing sediment chemistry data for all of the marshes. The proposed sediment sampling locations were in rough proximity to previous sediment chemistry samples. This was done to provide an expected gradient of sediment contamination and allow a better opportunity to correlate sediment chemistry with toxicity while doing risk assessment. Samples for benthic analysis were collected at Riverbend and Kearny Marshes because no benthic analyses had been completed previously at these marshes.

The upper 10cm (approximate) of sediment were used for toxicity testing since this layer represents the most biologically active portion of sediment where contaminants are likely to be most bioavailable.

Sampling locations were located in the field using Global Positioning System (GPS). The coordinates for sampling locations are provided below:

| Location | Latitude | Longitude | Description |
|---|---|---|---|
| RBM-01 | 40º45’15.23” N | 74º05’17.57”W | In open water 30’ wide area bounded by common reed (*Phragmites australis*) and punctuated by *Spartina alterniflora* hummocks. |
| RBM -02 | 40º45’09.89”N | 74º05’19.39”W | In mosquito ditch, 6-8’ wide and bounded by thickly growing *Phragmites* |
| SHS -01 | 40º48’20.73”N | 74º02’50.14”W | Along the main channel within the marsh |
| SHS -02 | 40º48’15.04”N | 74º02’51.02”W | Along the main channel within the marsh |
| SHS -03 | 40º48’12.44”N | 74º02’47.67”W | Along the main channel within the marsh |
| SAW -01 | 40º45’54.39”N | 74º05’45.91”W | Northeast side of channel (roughly 80’ wide) approximately 1500 feet from confluence with the Hackensack River |
| SAW -02 | 40º46’30.99”N | 74º06’02.10”W | Open water area near western spur of NJ Turnpike |
| ORM -01 | 40º47’58.14”N | 74º05’13.76”W | In a tidal creek along the southwest border of the marsh |
| ORM -02 | 40º47’43.02”N | 74º05’01.74”W | In a tidal pool along the southeast side of the marsh |
| KFM -01 | 40º45’44.08”N | 74º07’55.32”W | Along western edge of the marsh near cattails and loosestrife |
| KFM -02 | 40º45’37.67”N | 74º07’43.60”W | In middle of the wetland in open water |
| KFM -03 | 40º45’48.89”N | 74º07’18.63”W | Near the railroad bed at the northern end of the site near islands of *Phragmites*. |

## 2.4 Sample Collection Methods

The vessel used to access the sampling sites was a 21' Privateer with a 115 Hp engine. The Privateer was used at high tide to ensure water levels were high enough to enter and exit. Sampling in Kearny Marsh was conducted using a 14-foot aluminum skiff with a 15 horsepower engine. The latitude and longitude of each location was recorded using a Trimble ProXRS Global Positioning System (GPS) unit.

### 2.4.1 Sediment for Bulk Chemistry Analysis and Toxicity Testing

Sediment samples were collected using an Ekman Dredge (6" x 6" x 6") affixed to an approximately five-foot long pole. Sediment was placed in a decontaminated stainless steel bowl and a stainless steel spoon was used to homogenize and transfer sediment to collection jars. Latex gloves were worn during transfer procedure. Samples to be analyzed at the MERI lab were placed in widemouth glass 2 liter containers, labeled on the lid and jar, and immediately placed in coolers with wet ice. Samples to be analyzed by Accutest Laboratories were placed in 8 ounce glass amber bottles, labeled appropriately and chilled. Both lab sample containers were wrapped in bubble wrap prior to cooler placement. Samples for toxicity testing were placed in 3-gallon plastic buckets supplied by B-H Laboratories, labeled on the lid and container, sealed, and stored appropriately on deck. Samples were chilled upon return to shore due to large sized of container.

A duplicate sample was collected at location KFM-SD-03. At the end of each day, sediment samples were delivered to Accutest, Inc. and MERI laboratory for analysis (see Appendix A for Chain of Custody (COCs) forms). A rinseate blank was collected at Sampling Station KFM-SD-02 to determine the potential transfer of contaminants from sampling equipment to the samples. Six one-liter, amber jars were filled and one plastic container with $HNO_3$ was filled with lab supplied deionized water using the Ekman. The sample was transferred to a stainless steel bowl with a spoon and placed in containers supplied by Accutest.

Decontamination of gear (e.g., Ponar/Ekman, stainless steel spoons, stainless steel bowls) was initiated after collections were completed at each site. The Ekman and Ponar were rinsed clean with site water between each sample to avoid cross contamination of samples. Latex gloves were worn during decontamination. Jaws of both Ekman and Ponar, once decontaminated, were sealed shut during transfer to the next site/station.

### 2.4.2 Sediments for Benthic Analysis

. Sediment samples for benthic analysis were collected with a stainless steel petite Ponar/Ekman grab sampler in triplicate per ENSR Standard Operating Procedures 5204. Samples were collected at RBM-01, RBM-02, KFM-01, and KFM-02. Benthic samples were sieved in the field using a 500-micron mesh sieve. The Ponar or Ekman was rinsed with site water for each sample to ensure that the whole sample was placed in the sieve. Buckets were also rinsed with site water when transferred to the sieve workstation. The sieve workstation consisted of a milk crate used as a pedestal with a galvanized steam washtub placed on top. The sample was transferred to the sieve and gently rinsed with site water. Site water was placed in the tub prior to sieving. Sieving was conducted in a gentle swirling motion until all but particles larger than 500 microns remained. Samples were then placed in plastic wide mouth containers and labeled on lid and container. Benthic samples were preserved with 10% buffered formalin.

### 2.5 Field Measurements

Field measurements, including temperature, salinity, dissolved oxygen (DO), DO%, turbidity, Oxidation Reduction Potential (ORP), specific conductivity and pH were recorded using a Hydrolab Mini Sonde Water Quality Multiprobe Sensorflex Parameter Expansion System. Water parameters were measured at the water's surface at each location and were recorded on the attached data logs (Appendix A). Other parameters recorded included weather, sampling start and stop times, tide levels and names of surveyors.

### 2.6 Benthic Sample Analysis Methods

Cove Corporation performed Benthic lab analyses. Voucher specimens were collected as part of this analysis. The raw data, a list of the vouchers, and the QC Report prepared by Cove is included in Appendix D.

Using the data provided by Cove Corporation, ENSR completed an analysis of the benthic community by constructing dominant species lists, calculating of Shannon-Wiener Diversity Indices (H'), and Pielou's Evenness (J') for each assessment site sampled by ENSR (Riverbend Marsh and Kearny Marsh).

### 2.7 Sediment Chemical Analysis Methods

The sediment samples were analyzed for total metals (cadmium, Cd; chromium, Cr; copper, Cu; mercury, Hg; lead, Pb; nickel, Ni; zinc, Zn; arsenic, As), pesticides, polychlorinated biphenyls (PCB's), polynuclear aromatic hydrocarbons (PAHs), total organic carbon (TOC), grain size, % moisture and pH by the MERI Laboratory (Cd, Cr, Cu, Fe, Pb, Ni, Zn, Hg, grain size) and AccuTest Laboratory (all other parameters). Table 2-1 summarizes the laboratory methods used for sample analysis. Analyses were performed in accordance with the Quality Assurance Project Plan developed for the project and approved by U.S. EPA (ENSR, 2003).

## 2.8 Sediment Toxicity Testing Methods

Whole sediment toxicity testing was conducted on sediment samples collected from twelve locations in the Hackensack Meadowlands. These included samples from Kearny Freshwater Marsh (KFM), Oritani Marsh (ORM), Riverbend Marsh Wetland Preserve (RBM), Sawmill Creek Wildlife Management Area (SAW), and Secaucus High School Marsh (SHS). With the exception of Kearny Marsh, the marshes sampled were estuarine with moderate salinity. As such, different test conditions (e.g., salinity of overlying water and/or test organism) were used to perform the toxicity tests for the freshwater and estuarine samples. Whole sediment toxicity testing was conducted 1 through 29 October 2003 using the amphipod, *Hyalella azteca* (H. *azteca*). Whole sediment toxicity testing was conducted 3 through 13 October 2003 using the freshwater midge, *Chironomus tentans* (C. *tentans*). Whole sediment toxicity testing was conducted 12 through 22 October 2003 using the marine annelid, *Neanthes arenaceodentata* (N. *arenaceodentata*). The sediment samples were collected by ENSR personnel and transported to B-H Laboratories' Spring City, Pennsylvania laboratory. The objective of this testing was to determine chronic toxicity of the sediments to H. *azteca*, C. *tentans* and N. *arenaceodentata*.

All testing was conducted in accordance with the QAPP (ENSR, 2003) and individual testing SOPs developed by B-H Laboratories (provided in Appendix I of the QAPP). The testing procedures are consistent with those outlined in:

- Ingersoll, C.G., G.T. Ankley, G.A. Burton, F.J. Dwyer, R.A. Hoke, T.J. Norbert-King and P.V. Winger. 1994. Methods for Measuring the Toxicity and Bioaccumulation of Sediment-associated Contaminants with Freshwater Invertebrates. EPA/600/R-94/024. U.S. EPA Office of Research and Development, Duluth, MN,
- ASTM. 1995. Standard Test Methods for Measuring the Toxicity of Sediment-Associated Contaminants with Fresh Water Invertebrates. E1706-95. American Society for Testing and Materials, Philadelphia, PA; and
- ASTM. 1994. Standard Guide for Conducting Sediment Toxicity Tests with Marine and Estuarine Polychaetous Annelids.E1611-94. American Society for Testing and Materials, Philadelphia, PA.

The following provides a brief overview of the basic methods employed. A detailed discussion of methods is provided the toxicity testing report prepared by B-H Laboratories (Appendix C).

### 2.8.1 Test Organisms

<u>Amphipod, *Hyalella azteca* (10-ppt; representative of salinity observed in the sampled wetlands)</u>: Immature H. *azteca* (13-days old) used in testing were obtained from Aquatic Research Organisms (ARO), a commercial culture facility located in Hampton, New Hampshire. This species is a representative invertebrate and a United States Environmental Protection Agency (EPA) accepted test

organism. The amphipods used for testing were hatched on 18 September 2003 and held at the culture facility until shipment on 30 September 2003. The amphipods were acclimated to a salinity 10-ppt prior to shipping to B-H Laboratories Biological Services Division.

Amphipod, *Hyalella azteca* (moderately-hard reconstituted water; surrogate for freshwater): Same as for Freshwater amphipod (10 ppt) treatment, except the amphipods were placed in a plastic cubitainer with moderately-hard reconstituted water and shipped overnight to B-H Laboratories.

Larval Midge, *Chironomus tentans*: C. *tentans* larvae (14-days old) used in testing were obtained from Aquatic Research Organisms (ARO), a commercial culture facility located in Hampton, New Hampshire. This species is a representative invertebrate and United States Environmental Protection Agency (EPA) accepted test organism. Prior to shipping, the amphipods were fed rabbit pellets during acclimation and holding at ARO. Midge larvae (14-days old) were placed in a plastic cubitainer with moderately hard re-constituted water and shipped overnight to B-H Laboratories' Biological Services Division in Spring City, Pennsylvania.

Marine Annelid, *Neanthes arenaceodentata*: N. *arenaceodentata* (two to three weeks old) used in testing were obtained from California State University at Long Beach, a non-commercial culture facility located in Long Beach, California. This species is a representative invertebrate and U.S. EPA accepted test organism. Annelids were placed in a plastic cubitainer with 25-ppt seawater and shipped overnight to B-H Laboratories' Biological Services Division in Spring City, Pennsylvania.

### 2.8.2 Negative Control Sediment

Sediment to be used as the negative controls was collected from Blue Marsh Reservoir, a site known to exhibit minimal toxicity, located in Lower Heidelberg, Pennsylvania. The sediment was collected on 25 September 2003 and transported to the Spring City laboratory. The sediments were collected from a cove south of the boat launch. The sediments were collected approximately five-feet from the shoreline in two-feet of water.

### 2.8.3 Test Sediments

ENSR International, Inc. personnel, collected sediments on 15 through 17 September 2003. at RBM -01, RBM -02, SHS -01, SHS -02, SAW -01, SAW -02, SAW -03, ORM- 01, ORM -02, KFM -01, KFM -02, and KFM-SD-03. Sites no(s). SAW-SD-01 and SAW-SD-02 were designated as reference sediments. Sediment samples were picked up in Lyndhurst, NJ by B-H Laboratories' personnel and transported to B-H Laboratories' Spring City office on 15, 16 and 17 October 2003. Chain-of-Custody forms accompanied all sediments. All sediments were stored at 1° to 4.4°C in their original containers when not being used. Sediments were warmed to test temperature prior to test initiation. Prior to use, the sediments were homogenized and sieved through a No. 6 (3.35-mm) stainless steel sieve to remove large pebbles, stones, twigs, plant clumps, and indigenous macroinvertebrates.

### 2.8.4 Overlying Water

Reconstituted water: Reconstituted water was prepared in accordance to procedures outlined in EPA/600/R-99/064 to be used as acclimation and overlying water for the toxicity tests. Water was decanted through a mixed bed de-ionizing system (U. S. Filter, Inc.) consisting of carbon, mixed resin-bed and membrane filters. A 190-liter Nalgene tank was filled with about 150-liters of de-ionized water. In a five-gallon bucket of de-ionized water, 9.5-grams of $CaSO_4$ and 9.5-grams $CaCl_2$ was mixed into solution by vigorously aerating for 30-minutes or until the salts dissolve. Into a second 5-gallon bucket of de-ionized water, 5.7-grams of $MgSO_4$, 18.2-grams of $NaHCO_3$ and 0.76-grams KCl was mixed by aerating for 30-minutes or until the salts dissolve. The two five-gallon aliquots were added to the 150-liters of de-ionized water and the volume was brought to 190-liters with de-ionized water. Reconstituted water was aerated for at least 24-hours prior to use.

Salt Water (10-ppt and 25-ppt): Natural seawater was collected at the Manasquan Inlet in Brielle, New Jersey. The seawater was collected on 24 September 2003 by B-H laboratories personnel. Salinity of water collected at this NJDEP approved site was 32 ppt at the time of sampling. The 10-ppt and 25-ppt seawater was prepared by filtering the natural seawater through a 0.5-μm polypropylene string-wound cartridge filter. The natural seawater was adjusted to 10-ppt or 25-ppt by incremental additions of de-ionized water. The water was aerated for 24-hours. After aerating for 24 hours, the salinity was measured and the salinity was adjusted either by adding additional seawater or de-ionized water to achieve the target salinity. The water was aerated an additional 24-hours prior to use as acclimation or overlying water. Natural 10-ppt salt water was prepared in 190-liter batches.

### 2.8.5 General Testing Protocol

Sediment toxicity tests were conducted in chambers that were placed in a temperature-controlled environmental chamber. Test chambers were 300-mL or 1000-mL high-form lip-less Kimax beakers. Prior to testing, sediment samples were prepared as described above. An appropriate volume of homogenized sediment and overlying water was placed into each test chamber on day –1 (day before test initiation). Eight replicate chambers were tested for each of the sediments. After an overnight settling period, the water quality (e.g., pH, alkalinity, conductivity, temperature, dissolved oxygen) of the overlying water were determined and organisms were added in an unbiased manner to each test chamber. Organisms in the test chambers were fed with the appropriate amount and type of food (per the testing protocols). Water renewals were accomplished by siphoning off the old solution and replacing it with fresh overlying water renewed overlying water twice daily (A.M. and P.M.). Test chambers were not aerated as water renewal maintains sufficient dissolved oxygen. Any dead organisms observed during the renewal process were removed and recorded. Appropriate water quality measurements (e.g., dissolved oxygen, temperature, pH, ammonia) were taken and recorded as described in the testing protocols.

The tests were terminated at the end of 10-days or 28-days, depending on the test. Surviving organisms within each replicate were collected with a U.S. Standard No. 50 stainless steel sieve. The organisms were counted, rinsed with de-ionized water and transferred as a group to pre-weighed pans. The number of surviving organisms was recorded. Organisms were dried and each pan containing organisms was weighed on an Ohaus Model AP250D Plus analytical balance. The total organism dry weight was divided by the number of surviving organisms to obtain the average organism weight per replicate.

# 3.0 RESULTS

The results of benthic sampling, bulk sediment chemistry and toxicity testing are described in this Section. The results presented in this section support assessment and analyses discussed in Sections 4.0 through 6.0 of this report.

## 3.1 Field Measurements

Field measurements taken at the time of sediment collection for benthos and chemical analysis and toxicity testing included temperature, salinity, conductivity, dissolved oxygen, pH, oxidation reduction potential (ORP), and total dissolved solids. Table 3-1 provides a summary of field measurements taken. Water temperature at the time of sampling ranged from 19.7 °C to 25.3 °C. As is evident from the salinity measurements ranging from 0.9 ppt to 1.3 ppt, the Kearny Marsh (KFM) samples were collected in an area with low salinity (near freshwater to slightly brackish). Salinities at other locations ranged from 5.6 ppt to 14.3 ppt. Dissolved oxygen supersaturation was observed at Kearny Marsh (KFM) and one of the Oritani Marsh sampling locations. Elsewhere dissolved oxygen ranged from 2.2 to 5.9 mg/L. The pH ranged from 6.95 to 8.66 S.U., and was greatest at those sampling locations where dissolved oxygen supersaturation was evident. These combined results suggest significant photosynthetic activity at those locations. ORP ranged from 286 to 358 millivolts, and was generally lower at the freshwater sampling locations. Total dissolved solids ranged from 1.1 to 1.5 g/L at Kearny Marsh, and from 6.3 to 15.5 g/L at the other sampling locations.

## 3.2 Benthic Community Composition

Invertebrates identified from sediment samples collected at Kearny and Riverbend marshes in September 2003 are shown in Table 3-2. A total of five taxa were found in the six samples collected at Kearny Marsh, including immature chironimids and oligochaetes which each might contain several species. The remaining taxa, the sand fly *Culicoides* sp., the amphipod *Gammarus palustris*, and the nereid polychaete *Laeonereis culveri*, all typical estuarine species, were present in very low numbers. Density (? = 2827 ind./ $m^2$), diversity (0.52), and evenness (0.26) were all very low at this site (Table 3-2).

The benthic fauna at Riverbend Marsh was richer in species than at Kearny Marsh and displayed greater diversity and evenness. The density of animals was very great with up to 10 times as many individuals per square meter (> 30,000 ind./ $m^2$) at Riverbend Marsh than at Kearny Marsh or three other marshes, Secaucus High School, Oritani, and Mill Creek, sampled previously by others, whose parameters are given in Table 3-3. Riverbend Marsh was one of the most diverse studied with a diversity of H'> 2. Diversities of only a few samples at Mill Creek Marsh exceeded this value. An important factor that may contribute to some of the observed differences between the various marshes is the types of habitats sampled. Previous studies included sampling of mudflats, tidal creeks, marsh plain, ponded/open water, channels, high marsh, and mosquito ditches. Samples collected as part of this study were primarily located in open water, mosquito ditches, and/or channels.

In addition to the fauna found at Kearny Marsh, organisms at Riverbend Marsh included the burrowing anemone, *Edwardsia elegans*, six additional polychaete species, three additional amphipod species, two isopods, one bivalve, *Macoma balthica*, and one snail, *Spurwinka salsa*. All of these taxa are common inhabitants of estuarine systems.

Dominant species at Kearny and Riverbend Marshes are shown in Table 3-4. Oligochaetes were the dominant taxon at Kearny Marsh and the second-most common taxon at Riverbend. The snail, *Spurwinkia salsa*, was the dominant at Riverbend. This gastropod species was found in all six samples collected at Riverbend but was absent from Kearny Marsh and had not been seen at the historical samples collected at Mill Creek, Oritani, or Secaucus High School Marshes (Tams, 2001a; Louis Berger, 2001; HMDC, 1997). Benthos sampling data for Sawmill Creek and Eight-day marshes were available but not used in these comparisons. Sawmill Creek data were from 1987 and deemed to old to provide useful comparison to current benthos sample or comparison to sediment chemistry collected during this study. Data for Eight-day swamp (Weis and Weis, undated) were not identified to similar taxonomic levels as other data, and the dates of sample collection were unknown. The insect taxa *Chironomidae* pupae/larvae and *Culicoides* sp. (by density) ranked second and third at Kearny Marsh and fifth and seventh at Riverbend. Polychaetes *Laeonereis culveri* and *Hobsonia florida* ranked third and fourth at Riverbend Marsh.

Table 3-5 gives a summary of species found both at Kearny and Riverbend in 2003 and at the historical sites visited in 1997 and 2000. Approximately, 55 taxa have been identified from the five marshes. This is an approximate number as several of the taxa might actually be the same species (i.e. *Cyathura* and *Cyathura polita*) and other taxa probably are comprised of several species (i.e. *Chironomidae* and Oligochaeta). All of these taxa are those commonly found in estuarine systems. The most commonly represented groups include 12 insects, 12 polychaetes, 8 amphipods, 5 isopods, 4 gastropods, and 3 bivalves. Chironomids were dominant in the Secaucus High School samples, while copepods and a common polychaete were dominant in the Oritani marsh samples, and Mill Creek samples were dominated by an amphipod, chironomids, and a common polychaete. A description of some of the more common and interesting species, especially polychaetes follows.

*Spurwinkia salsa* (Pilsbry, 1905) was the dominant benthic invertebrate found at Riverbend Marsh in September 2003. This deposit feeding gastropod snail inhabits high marsh pools from Maine to Connecticut and possibly to the DELMARVA peninsula in salinities of 0-29‰. This is a euryhaline snail endemic to the upper reaches of estuaries (Davis et al., 1982). Habitats include tidal marsh creeks and pools, intertidal marsh turf below mean highwater level, and shallow subtidal waters of channels and embayments bordered by tidal marshes. S. *salsa* can live in brackish marshes where salinities are always below 20‰ and often drops to 0‰ in spring, as well as in Spartina salt marshes at or near their furthest reaches up the estuary (i.e. inland) where seasonal salinities range broadly (0-20‰)(Davis et al. 1962). In Maine and Massachusetts, as in the Meadowlands, they were found living in association with aquatic insects and euryhaline estuarine endemics of marine origin, such as, *Hobsonia florida*, *Manayunkia aestuarina*, *Marenzellaria viridis*, *Gammarus* sp., *Cyathura polita*, midge larvae, and oligochaetes.

*Laeonereis culveri* (Webster, 1879). *Laeonereis culveri* found at both Kearny and Riverbend marshes, is an estuarine polychaete that is distributed from Connecticut to the Gulf of Mexico. The species occurs in low saline estuarine habitats including tidal flats, sandy shoals, and marshes and is highly adapted to survive in such environments. Mazurkiewicz (1975) studied the larval development and ecology of L. *culveri*. Females deposit their eggs within their mucoid tubes. Development is entirely benthic with planktonic larvae never observed. Larvae were found in the upper 2 cm of flocculent sediment on tidal flats from June to September in Connecticut. Laboratory experiments suggest that at 22ºC, development from fertilized eggs to the 20-setiger juvenile stage takes approximately 45 days. This is a stage of development where most of the adult morphology is present. A total of 190 days is needed to develop to the 80-setiger stage or one where sexual maturity might be indicated. Adults died after spawning, so success of the annual reproductive cycle will dictate status of the adults in the benthic communities the following year. Some observations of larval behavior by Mazurkiewicz (1975) suggest that L. *culveri* larvae have some capability of selecting favorable sediments in which to live. He noted that if overlying water becomes foul, the larvae assume a swim-crawl mode and move to more favorable conditions. Klesch (1970) conducted a less comprehensive study of L. *culveri* in Texas. He found that the species had a biannual spawning in contrast to the annual summer spawning in Connecticut waters. It is likely that populations in New Jersey marshes are more similar to the Connecticut pattern.

*Hobsonia florida* (Hartman, 1951), found at Mill Creek, Oritani, and Riverbend marshes, is one of the most common estuarine polychaetes occurring in intertidal and subtidal muds from Maine to Florida; the species is also known from the Pacific Northwest. The species forms part of the infauna in salt marshes, estuaries, and river mouths, often in high densities. H. florida is found in a wide range of salinities from <1‰ to 30‰ with other common estuarine invertebrates (Pettibone, 1977). The species has been known under several names including *Amphicteis gunneri floridus*, A. *floridus*, and *Hypaniola grayi* Pettibone, 1953, and H. *florida*. The current classification of the species and referral to Hobsonia was by Banse (1979). H. *florida* is small, 8-15 mm long and lives in tubes consisting of mud, sand, and bits of animal and plant debris (Zottoli, 1974). The larval development was described by Zottoli (1974). Working in New Hampshire, he found that spawning occurred from late May to early September. The eggs were released into the tube of the female and developed there to the 2-setiger stage in about 2 days, after which they left and continued to develop in the mud. Approximately 36 days were required for development to proceed to 18 setigers or the juvenile stage.

*Heteromastus filiformis* (Claparède, 1864). This species is a widespread benthic infaunal polychaete that often inhabits intertidal muds and is subject to low oxygen conditions. The species is not typically found in marsh communities and only one specimen (at Riverbend) was found in the five marshes tabulated in Table 3-5.

*Manayunkia aestuarina* (Bourne, 1883) is a small (1-3 mm) tube-dwelling sabellid polychaete known from intertidal and subtidal muds in Europe and North America. Among the five marshes compared in Table 3-5, this species was found only at Riverbend Marsh where it ranked sixth in abundance. The species is a surface deposit feeder, using cilia on its tentacles to gather sediment particles. On the east coast of the North America, the species ranges from Maine to the Carolinas. Bell (1982) reported on the biology and ecology of this species from a South Carolina salt marsh. She found that the species attained its highest densities in the fall, with lesser peaks in winter and spring; densities were lowest in winter months. In Boston Harbor, Massachusetts, Trueblood et al., (1994) reported highest densities in August and September with no distinct peaks in other seasons. This suggests that northern populations are more seasonal than southern populations. Bell (1982) noted that embryos and juveniles were present year round, suggesting more or less continuous reproduction. For a small species, the eggs are large, sexes are separate, fertilization is by transfer of spermatophores, and the development is entirely within the tubes of the females.

*Nereis succinea* (Frey and Leuckart, 1847). This is one of the most common nereids in the northern hemisphere and the subject of numerous physiological, embryological (as *Nereis limbata*) and ecological investigations are reviewed by Pettibone (1963). The species is euryhaline and well known from all three coasts of the United States, and was found at Oritani Marsh. It ranges from the high intertidal to shallow subtidal depths, but is most common in the mid-intertidal range. The species occurs in estuaries and embayments in a wide variety of habitats including sand, mud, or among rocks. Individuals construct U-shaped burrows, which they frequently abandon, forming new ones. Adults undergo complete heteronereid metamorphosis and swarm to the surface of the water in large numbers. There are distinct morphological differences between males and females. There is evidence of lunar synchrony in the spawning cycle during the periods of the new moon from June to September. Males are smaller and faster moving than females and usually appear earlier in the evening than the females. Males swim in rapid, narrowing circles around the females shedding sperm. This stimulates the females to release eggs. Adults die after spawning. Eggs are about 140 µm in diameter and develop into planktotrophic larvae. Metamorphosis to bottom crawling forms occurs at the 4-6 setiger stage. Some aspects of the biology were reviewed by Pettibone (1963) but there is an extensive literature on the species.

*Streblospio benedicti* (Webster, 1879). *Streblospio benedicti* is one of the most common spionid polychaetes occurring in intertidal mud flats in eastern North America. The species is very common in salt marshes where it inhabits silt-encrusted tubes and feeds from surface sediments and nearbottom water with its prehensile palps. In the present study, this species was found at

Oritani and Riverbend Marshes. The reproduction and larval development has been studied extensively for S. *benedicti* (Blake and Arnofsky, 1999). Females brood embryos in dorsal pouches from which larvae are eventually released into the plankton. Two distinct forms of larval development have been described. In the first, larvae are released into the plankton where they remain for up to 45 days feeding on phytoplankton. In the second, the larvae are released from the pouches at a much later stage of development; they do not feed, and settle shortly thereafter. Both types of development are known from the eastern United States. Dauer (1984) has described the feeding morphology and behavior of this important species.

*Marenzellaria viridis* (Verrill, 1873). This species is believed to be native to the northeastern United States, but has been introduced into northern Europe and most recently San Francisco Bay. The species was formerly known as *Scolecolepides viridis*. Maciolek (1984) revised the nomenclature. In the present study, M. *viridis* was found only at Oritani Marsh where it was the second most common taxon. The species is capable of building dense populations in soft sediments from the intertidal to shallow subtidal. The adults and larvae are tolerant of very low salinities and thus well suited to colonizing high estuarine habitats such as marshes and back bay environments. This salinity tolerance also accounts for its dispersal to Europe by ballast water transport. George (1966) described the larval development based on specimens from Nova Scotia. The eggs are relatively large (200-260 µm) and spawned directly into seawater where they are fertilized and develop as planktotrophic larvae. Development to settling juveniles takes about 45 days.

*Boccardiella ligerica* (Ferroniére, 1898). *Boccardiella ligerica* is a widely distributed species that is opportunistic in sediments overlain with waters of very low salinity including near freshwater. In data compiled for the five marshes in Table 3-5, B. *ligerica* was found only in Riverbend Marsh. The species was redescribed by Blake and Woodwick (1971) (as *Boccardia*) and has been reported from all three coasts of North America. Little is known concerning its biology. As part of a recent study of estuarine tributaries in South Carolina, the species was one of the dominant species (ENSR, unpublished reports).

The benthic community found at Riverbend includes faunal elements that are typical of high estuarine, low salinity marshes throughout the eastern United States (Pettibone, 1963, 1977; Zottoli, 1974; Mazurkiewicz, 1975; Maciolek, 1984). As indicated in the comments for indicator species, all are adapted to these environments. The current sampling provided minimal evidence of pollution or stress-related indicator species such as the polychaetes *Capitella capitata* (none observed) or, possibly, *Polydora cornuta* (one individual observed in two of the replicates at RBM-02). *Polydora cornuta* was observed in Mill Creek in 1997 (pers. Comm. Brett Bragin, 2003).

The benthic community at Kearny Marsh appears to be impoverished and is possibly impacted by the high contaminant loads. This statement is tempered, however, by the fact that the species composition of the insects was not determined.

## 3.3 Bulk Sediment Chemistry

#### 3.3.1.1 2003 Sediment Sampling

Bulk sediments were analyzed for metals (cadmium, Cd; chromium, Cr; copper, Cu; mercury, Hg; lead, Pb; nickel, Ni; zinc, Zn; arsenic, As), pesticides, polychlorinated biphenyls (PCB's), polynuclear aromatic hydrocarbons (PAHs), total organic carbon (TOC), grain size, % moisture and pH. Table 3-6 provides the results for general sediment chemistry for the individual samples analyzed. Chemistry data for the individual samples is provided in Appendix E, Table E-1. Table 3-7 provides summary statistics (minimum and maximum) for each of the five wetlands from which samples were taken. For the purposes of summary statistics, only detected concentrations are considered in the minimum and maximum statistics. When a contaminant was not detected (ND) in any sample summary statistics are identified as Not Applicable (NA), however if a contaminant was detected at one sample location and not another, one half of the analytical detection limit was used for any non-detect concentrations in the calculation of summary statistics.

Maximum and mean concentrations of total PAHs were greatest in Riverbend Marsh and were lowest in the Secaucus High School Marsh. Few pesticides were detected in any of the marshes. Chlordane (alpha (cis)-) was detected only in the Kearny Marsh while 4,4'-DDE was detected in all marshes. Other detected pesticides included 4,4'-DDD, 4,4'-DDT and methoxychlor. Of the seven PCB aroclors analyzed, three were not detected in any of the marshes (aroclor 1016, 1221, and 1232). Aroclor 1242 was detected only in Riverbend Marsh. Aroclors 1248 and 1258 were detected in all of the marshes. Maximum and Mean total PCB concentrations were greatest in Oritani Marsh and lowest in Secaucus High School Marsh. All the metals analyzed were detected in all the marshes sampled. With the exception of iron and lead, the greatest maximum concentrations were observed in Oritani Marsh. These data were used in the Screening Level Ecological Risk Assessment (SLERA) presented in Section 4.0.

#### 3.3.1.2 Historic Sediment Sampling

A number of studies, which included wetland sediment sampling and analysis, and in most cases comparison to applicable aquatic life criteria have been performed on the Meadowlands Wetlands. These studies included the following wetlands:

- Kearny Marsh - Langan EES (1999)
- Oritani Marsh - Louis Berger (2001)
- Secaucus High School Marsh – TAMS (2001a)
- Skeetkill Creek Marsh – ECI (1997)

- Riverbend Wetland Preserve – TAMS (2001b)
- Mill Creek Wetland – HMDC (1997)
- Eight-Day Swamp – Weis, J. and P. Weis (undated)
- Harrier Meadows – ECI (1997)

Data from these studies were used to support this project and are provided in Appendix E. The number of sampling sites, depth of sediment samples, parameters analyzed, detection limits, and methods of reporting varied between the various studies. For consistency all data were compared to estuarine sediment benchmark values; primarily from effects range-low (ER-L) (Long et al., 1995). Additional sediment benchmark sources included the low effect levels (LELs) from the Ontario Ministry of the Environment (OMOE) (Persaud et al., 1996) if no ER-L was available for a given contaminant. The following provides a brief summary of pertinent observations regarding the reported sediment chemistry data for each study:

Riverbed Wetland Preserve (TAMS, 2001b):

- Samples were collected from 15 locations at multiple depths (0-6", 6-12", 12-18", 0-24" composite). Two field duplicate samples were collected and analyzed.
- Samples were analyzed for volatile organic compounds (VOCs), semi-volatile components (SVOCs), pesticides, PCBs, metals, cyanide, particle size, percent solids, pH, total petroleum hydrocarbons (TPH), and total organic carbon.
- Detection limits for SVOCs and certain pesticides often exceeded the applicable screening criteria.
- Total organic carbon concentrations ranged from 38,000 to 190,000 mg/kg.
- Where detected, 4,4-DDT, aroclor-1248, aroclor-1260, chromium, copper, lead, mercury, nickel, and zinc exceeded the applicable screening criteria in a majority of samples.

Secaucus High School Wetlands Mitigation (Tabs, 2001a)

- Samples were collected from 8 locations at multiple depths (0-6", 6-36"). Two field duplicate samples were collected and analyzed.
- Samples were analyzed for SVOCs, pesticides, PCBs, metals, cyanide, particle size, percent solids, total petroleum hydrocarbons (TPH), and total organic carbon.
- Detection limits for some SVOCs and certain pesticides often exceeded the applicable screening criteria.
- Total organic carbon concentrations ranged from 21,000 to 160,000 mg/kg.

- Where detected, 4,4-DDT, aroclor-1248, aroclor-1260, arsenic, chromium, copper, lead, mercury, nickel, and zinc exceeded the applicable screening criteria in a majority of samples.

Oritani Marsh

- Samples were collected from 18 locations at multiple depths (0-6”, 6-36”). Two field duplicate samples were collected and analyzed.
- Samples were analyzed for SVOCs, pesticides, PCBs, metals, cyanide, particle size, percent solids, total petroleum hydrocarbons (TPH), and total organic carbon.
- Detection limits for some SVOCs, PCBs, and pesticides often exceeded the applicable screening criteria.
- Most pesticides were not detected; although 4,4-DDE was detected and exceeded applicable criteria in 23% of the samples.
- Where detected, several SVOCs, arsenic, chromium, copper, lead, mercury, nickel, silver, and zinc exceeded the applicable screening criteria in a majority of samples.

Skeetkill Marsh

- Samples were collected from 5 locations.
- Samples were analyzed for pesticides, PCBs, and metals.
- 4,4’-DDE was detected at concentrations above the applicable criteria in 80% of the samples. 4,4’-DDD, 4,4’-DDT, a-chlordane were detected, but only exceeded applicable criteria in 20 – 40% of the samples.
- Where detected, PCBs (aroclor-1254 and aroclor-1260), arsenic, chromium, copper, lead, nickel, and zinc exceeded the applicable screening criteria in a majority of samples. Mercury was reported as non-detected; however the analytical detection limits were not presented in the data table.

Harrier Meadow Marsh

- Samples were collected from 5 locations.
- Samples were analyzed for metals only.
- Arsenic, chromium, nickel, and zinc were detected in all samples, but did not exceed the applicable screening criteria. Copper and lead were detected in all samples and only exceeded the applicable screening criteria in 20 and 40 percent of the samples, respectively. Mercury was detected in only one sample, and exceeded the applicable screening criteria at that location..

Mill Creek Marsh

- Samples were collected from 28 locations.
- Samples were analyzed for pesticides, PCBs, total petroleum hydrocarbons, cyanide, phenol, and metals.
- 4,4'-DDE, 4,4'-DDT were detected in 35 and 40 percent of the samples, respectively. Approximately 20 percent of the DDT concentrations exceeded applicable criteria, while only 2 percent exceeded the applicable screening criteria for DDE.
- Aroclor-1254 was detected in 65 percent of the samples, and exceeded the applicable screening criteria in 61 percent of the samples (95 of samples were aroclor-1254 was detected).
- Nickel and silver were detected in all samples at concentrations above applicable criteria. Mercury was not detected in 10 percent of the samples, and exceeded applicable screening criteria in 77 percent of the samples. Chromium, copper, lead, and zinc were detected in all samples, and in 83 to 90 percent of the samples.

Eight-day Marsh

- Samples were collected from 17 locations at one centimeter depth intervals (1 to up to 32 centimeters below the sediment surface).
- Samples were analyzed for selected metals only (mercury, arsenic, cadmium, chromium, copper, lead and zinc).
- Sediment concentrations of mercury and copper exceeded the applicable screening criteria at all sampling locations within the upper 4 centimeters of the sediment. Less than 9 percent of the samples from the upper 4 centimeters were below the screening criteria for chromium. Approximately 95 percent of samples from the upper 4 centimeters exceeded the cadmium screening criteria, and 98 percent of samples from the upper 4 centimeters exceeded criteria for arsenic, lead, and zinc.
- Sediment criteria were exceeded for the seven metals at all depths sampled at eleven of the seventeen sampling locations (65%).
- A pattern of increase concentration with sample depth was evident for several metals at some locations with concentrations as much as 10 times higher in the deeper sediments. However, this pattern did not appear to be consistent for a given sampling location, or for any given metal.

Detection limits for PAHs in several of the studies (Oritani, Riverbend, Mill Creek, Secaucus High School) were greater than the applicable screening criteria and nearly ten time higher than those used in the current study, making meaning full comparisons difficult. The same was true for PCB and pesticide detection limits in the Oritani, Riverbend, and Secaucus High School studies. VOCs were typically below detection limits in those wetland studies where analyzed.

Concentrations of metals were consistently lowest at Harrier Meadows, and greatest at Eight-day swamp and Kearny Marsh. Riverbend, Kearny, Oritani and Secaucus High School marshes were sampled as part of historic studies as well as the current project. Only surface samples were collected as part of this Project, therefore comparisons to historic studies are limited to surface sediment data (0 – 6 inches) from the historic studies. A comparison of data between these studies reveals that in most cases, concentrations of metals for this Project were typically within the range of historically reported values. Exceptions include cadmium at Riverbend and Oritani marshes, and chromium, lead, and zinc in Oritani Marsh where concentrations reported in this Study were greater than those reported historically as well as for mercury at Secaucus High School marsh which was less than the concentrations previously reported. Where metal concentrations observed during this Project were within the range of those previously observed, maximum concentrations were lower than those reported, in some cases by several times or several orders of magnitude (e.g., chromium in KFM reporting a maximum concentration of 49 mg/kg for this Project versus 5950 mg/kg for the historic study; and 280 mg/kg versus 1400 mg/kg at SHS; arsenic at RBM of 16.1 mg/kg versus 84 mg/kg; mercury in KFM of 0.52 mg/kg for this study versus 152 mg/kg from the previous study). In other cases, the maximum concentrations between this Project and past studies were similar.

Comparisons were not made for PAHs, PCBs or pesticides given the issues associated with elevated detection limits for the past studies which analyzed for these contaminants.

### 3.4 Toxicity Testing

Toxicity tests were completed on whole sediment samples collected from various wetlands. Sample sites were the same as those for bulk chemistry discussed in the previous subsection. With the exception of Kearny Marsh, all wetlands sampled were estuarine. Tests included a 28-day test using H. azteca (for both estuarine and freshwater sediment samples), and 10-day tests using C. *tentans* for freshwater sediment samples and N. *arenaceodentata* for estuarine sediment samples. Results are compared to a negative control. Negative control sediments are sediments known to produce at least 90% survival. Sawmill Creek (SAW) was identified as a reference site based on information supplied by NJMC/MERI. However, as noted in the bulk chemistry results as well as the toxicity results below, the SAW-01 and SAW-02 samples did not reflect an “unimpaired” condition. Given the nature of the Meadowlands today, it is unlikely that an “unimpaired” condition exists in the Meadowlands. As such, all test results are compared to the negative control rather than using the “reference” samples for comparison.

### 3.4.1 Toxicity Testing Results

Tables 3-8 and 3-9 present the results for the 28-day and 10-day toxicity tests for estuarine and freshwater sites, respectively. Each table presents both percent survival and mean weight per test organism at test initiation. Detailed results are provided in Appendix C of this report. Because mortality of some individuals throughout the testing period may provide greater availability of food for the remaining individuals, biomass is normalized to the number of organisms at test initiation. Biomass (as a measure of growth) is presented in Tables 3-8 and 3-9.

The negative control exhibited a 90% mean survival rate for the 28-day toxicity test for estuarine conditions. The 28-day mean survival rate for SAW-01, RBM-01, and SHS-01 were less than and significantly different than those for the negative control. Biomass was 0.54 mg for the negative control, but was greatest in ORM-02 (0.99 mg) and SAW-01 (1.02 mg). RBM-01 exhibited the lowest survival (36.25%) and lowest biomass (0.33 mg) among the 28-day test samples. Biomass for the RBM-01 sample was significantly different from that of the negative control. In the Kearny Marsh 28-day tests run with reconstituted water (freshwater condition), there was no significant difference between the test samples and the negative control for either survival or biomass.

Mean survival in the 10-day test negative control for estuarine conditions was 92.5%. Percent survival in ORM-01, ORM-02, RBM-01, and SHS-01 were lower than and significantly different from the negative control. The RBM-01 sample exhibited the lowest mean survival at 25%. Survival was less than 56% for the above samples. Mean biomass in the negative control was 0.33 mg. Biomass in SAW-01 and RBM-02 were lower and significantly different from the negative control. Mean biomass in other samples were less than the negative control, but were not statistically significantly different from the negative control. The survival rate for the freshwater test samples were all above 75%, and only the KFM-01 was less than and significantly different from the negative control. However, biomass was lower and significantly different from the negative control for all the Kearny Marsh test samples.

### 3.4.2 Application and Uncertainty of Sediment Toxicity Tests

Toxicity tests with sediments have evolved into an important tool in the evaluation of sediment quality. Unlike chemical analyses or biological community assessments, from which the toxicity of sediment contaminants can only be inferred, toxicity tests provide direct empirical evidence of the biological consequences of exposure to contaminated sediment. Toxicity studies can be a critical component in assessing real-world ecological risk since the physical and chemical characteristics of sediments play a significant role in the bioavailability of contaminants. High concentrations of organic carbon, for example, may reduce the bioavailability of some materials (e.g., copper). Conversely, very low levels of organic carbon may increase bioavailability, but may also result in poor habitat (and thus less exposure), particularly if the substratum is highly consolidated with few interstitial spaces suitable for benthic organisms.

Like all toxicity tests, a sediment toxicity study is a unique scientific investigation, the goal of which is to control all variables except the test medium, or sediment. As such, each toxicity test will be somewhat different than all previous and succeeding tests. This is true even if variables such as test chambers, temperature, lighting, and overlying water remain the same. Each set of tests used a different batch of organisms that may be of slightly different age and/or of slightly different health than other batches. New batches of food may also be of slightly different quality than previous batches. Variability among tests is, therefore, to be expected. In addition, even within a given test and treatment, the responses of individual replicates can vary substantially. It is not unusual to have low survival or growth in one or two replicates while the other replicates from the same sediment show a much better response. The resulting variance contributes to uncertainty in interpreting toxicity test results. Clearly defined results (e.g., 100% mortality) are easy to interpret, but responses that cluster around the threshold of statistical significance are more difficult to describe and complicate risk assessments and/or mitigation plans. For that reason, toxicity test results that show only marginal responses should be interpreted with caution and reviewed in light of the other legs of a Sediment Triad – chemical analyses and biological community assessments.

Toxicity testing was completed in accordance with the QAPP and associated SOPs with some exceptions.

- Hyalella azteca were 13-days old at initiation. The B-H Laboratories standard operating procedure for the Hyalella azteca test (ATL/SEDHACHR.050) states that organisms will be 7-8 days old for a 28-day test. Organism age is within range for a 10-day test (7-14 days); there is no EPA-mandated age range for a 28-day test. Since all organisms were within the same age range, this deviation from the SOP probably did not affect test outcome.
- Chironomus tentans was 14-days old at test initiation. The report states that the organisms were 2nd to 3rd instar, which is the target metamorphic stage of C. tentans at test initiation. However, the USEPA sediment testing manual (page 43) states that organisms 12.5 days old or older are probably 4th instar. Without head capsule measurements, or another acceptable method of determining instar, it cannot be determined if organisms were of the correct stage at initiation. Older organisms have been found to be less sensitive to some toxicants. This could be a deviation that might reduce the sensitivity of test organisms to the test sediment.
- Dry weight for amphipods and chironomids should be determined at 60 to 90°C, rather than 100°C. However, drying at 100°C is more typical for other toxicity tests and since all organisms were dried at the same temperature, this deviation should not have any effect on test outcome.

Sediments were screened to remove large material (e.g., large pebbles, stones, twigs and plant clumps) and homogenized prior to subsampling for toxicity test replicates. However, as wetland sediments likely include un-decomposed or partially decomposed plant material, it is possible that contaminants may not be equally distributed in the whole sediment sample. This may result in variability among the replicates.

A discussion of potential relationships between sediment chemistry, benthic community, and the above described toxicity testing is provided in Section 6.0.

# 4.0 ECOLOGICAL RISK ASSESSMENT

## 4.1 Introduction

As part of the evaluation of the ecological health of the Meadowlands wetlands of interest, ENSR used sediment and surface water quality data from the NJMC/MERI wetland database (See Appendix F; (Langan EES, 1999; Louis Berger, 2001; TAMS, 2001a; TAMS, 2001b; ECI, 1997a; ECI, 1997b; HMDC, 1997; Weis, J. and P. Weis, undated) as well as data collected as part of this study to perform a screening-level ecological risk assessment (SLERA). The purpose of the SLERA was to provide a conservative evaluation of the potential risks to ecological receptors posed by constituents in sediments and surface water in the wetlands under consideration, and to identify wetland area and chemicals of potential concern (COPCs) which warrant further investigation. The SLERA followed U.S. EPA guidance on ecological risk assessments (U.S. EPA, 1997). Food web modeling results from the SLERA were also used to generate the standard curves for rapid risk assessment described in Section 5.0. The approach and content of the SLERA were described in the approved *Quality Assurance Project Plan for Meadowlands Wetland Ecological Risk Assessment Project, Revision 1* (ENSR, 2003).

### 4.1.1 SLERA Guidance and Organization

The SLERA for this project was conducted using methods adapted from relevant risk guidance, including the following:

- Framework for Ecological Risk Assessment (U.S. EPA, 1992);
- Intermittent "ECO Update" Bulletins of U.S. EPA; and
- Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments (Interim Final) (U.S. EPA, 1997).

In should be noted that there are some significant differences between the formal process outlined by the Ecological Risk Assessment Guidelines for Superfund (ERAGS) (USEPA, 1997) and this project. These differences are summarized below:

- The results of the Meadowland ERA project are to be used for planning and risk management decisions rather than as ecological risk assessment conducted under the Superfund (CERCLA) program;
- The ERAGS guidance provides a methodology to investigate a hazardous waste site at which the contaminants of potential concern (COPCs) may or may not be fully characterized. ERAGS directs an inclusive, conservative approach by which a large universe of contaminants is initially considered and conservatively screened in the SLERA to see which contaminants

require further investigation in the Baseline Ecological Risk Assessment (BERA). Based on the results of the SLERA, a potentially large number of contaminants may be brought forward into the BERA, including those based on exceedance of a media-based benchmark and those due to a lack of appropriate screening values. Final decisions regarding which contaminants are considered COPCs are made in the BERA. This is done to assure that all chemicals from all potentially liable responsible parties (PRPs) are accounted for and to assure that remediation will address all unacceptable chemical-based risks;

- In contrast, this project, while relying on the ERAGS as a useful guidance document for an ecological risk assessment framework, is not driven by the same regulatory program, has different objectives, and is confined by available budget for this study. In this case, this project is intended to conduct a pragmatic assessment of potential ecological risk, based mostly on available historic data and knowledge of historical impacts, with limited supplemental sediment and benthic sampling, and a defined budget. The output of the ecological risk assessment will provide NJMC/MERI with a better understanding of the relative levels of risk for a variety of receptors over a gradient of sediment contaminant concentrations.

The ERAGS anticipate completion of all eight steps of the U.S. EPA's eight-step process for ecological risk assessment under Superfund toward the end of site cleanup. This SLERA compares to Steps 1 and 2 of U.S. EPA's eight-step process for ecological risk assessment under Superfund (Figure 4-1). As noted above, the objective here is to develop tools useful in planning and decision-making rather than remediation of the wetlands. Should additional investigation be warranted to achieve the project objectives, such work would be scoped and addressed in future phases of work.

#### 4.1.2 Organization of the SLERA

The SLERA is organized into the three following major sections suggested by U.S. EPA's Framework for Ecological Risk Assessment (U.S. EPA, 1992). These sections are Problem Formulation, Analysis, and Risk Characterization. A brief description of the content and purpose of these sections are given below.

1) **Problem Formulation** — provides the basis for decisions regarding the scope and objectives of the risk assessment. Information is collected and evaluated to develop a conceptual site model (CSM) of the site, in which the contaminants and potential exposure pathways are initially identified. This planning step contributes to the following processes:

- definition of risk assessment objectives of the SLERA;
- brief description of the of the Meadowlands wetlands and ecological receptors;
- brief description of contaminant sources and affected media;
- identification of chemical contaminants to be evaluated as COPCs;

- selection of potential receptors and receptor pathways;
- selection of assessment and measurement endpoints; and
- formalization of these selections via development of a CSM.

2) **Analysis of Exposure and Effects** — quantifies the magnitude, frequency, type, and duration of exposures of ecological receptors to COPCs at the site. Information is collected to:

- describe the statistical treatment of the environmental media data and selection of the exposure point concentrations (EPCs) to be evaluated;
- discuss the bioavailability of chemicals in the species' exposure media [Note: chemicals assumed to be 100% bioavailable for purposes of this SLERA] ;
- select the benchmarks to be used to relate chemical concentrations in the relevant environmental media to potential adverse ecological effects; and
- provide details on the assumptions and exposure factors to be used in the evaluation of wildlife receptors.

3) **Risk Characterization** — synthesizes the exposure and effects information to estimate the probability for and extent of adverse ecological effects, identifies uncertainty factors and evaluates potential effect on risk characterization findings, and identified potential data gaps. Data and supporting scientific judgments and/or modeling results are interpreted, so that:

- qualitative and quantitative statements about potential ecological risk can be advanced;
- sources of uncertainty are identified and the influence estimated;
- the site's potential to pose adverse ecological risk to biota and habitats can be determined;
- COPCs and/or exposure pathways that need to be further evaluated are identified.

The SLERA is organized in the following manner: Introduction (Section 4.1), Problem Formulation (Section 4.2), Analysis of Exposure and Effects (Section 4.3), Risk Characterization including a discussion of uncertainty (Section 4.4), and Summary and Conclusions (Section 4.5). References are provided in Section 9.0. Additional supporting data calculations and correspondence are contained in the Appendices.

## 4.2 Problem Formulation

Problem formulation is the initial systematic planning phase of the ecological risk assessment process. It provides the basis for the approach and methodology to be used as well as defining the specific scope and objectives of the risk evaluation. The problem formulation phase includes site characterization, identification of potential ecological receptors of interest, identification of potential exposure pathways for those receptors, identification of contaminants to be further evaluated, selection of assessment and measurement endpoints, and development of the CSM for the SLERA.

### 4.2.1 Selection of Ecological Receptors

Figures 2-2 through 2-6 illustrate the sampling locations within each wetland area. The Meadowlands wetland study area includes one freshwater and eight estuarine marshes. As described in Section 2.2, the focus of the SLERA is based on information and data from one freshwater and four estuarine wetlands. As part of the problem formulation, available documentation was reviewed to briefly characterize the habitat and receptors available within these wetlands. Sources of potential information that were consulted included:

- Previous wetland studies and reports conducted or sponsored by NJMC/MERI (see list provided in Section 3.3.1.2);
- NJMC/MERI "Digital Meadowlands" electronic database;
- NJ DEP Geographical Information Service (GIS) maps;
- USGS topographic quadrangle maps;
- Hudsonia, Ltd. 2002. Hackensack Meadowlands, New Jersey, Biodiversity: A Review and Synthesis. Prepared for Hackensack Meadowlands Partnership. August 2002; and
- other available site reports, maps, and information.

The following sections provide an overview of the wetlands of interest and the ecological receptors (plants, birds, mammals, fish, and invertebrates) which reside there. Much more extensive documentation is available in a variety of reports, documents, and databases at the NJMC/MERI website (http://cimic.rutgers.edu/meri/).

#### 4.2.1.1 Wetlands Habitats of Interest

The five wetlands (See Section 2.1) selected for this assessment and include Kearny Freshwater Marsh (KFM), Oritani Marsh (ORM), Riverbend Marsh Wetland Preserve (RBM), Sawmill Creek Wildlife Management Area (SAW), and Secaucus High School Marsh (SHS).

#### 4.2.1.2 Ecological Receptors of Interest

Potential ecological receptors of interest for this project include both aquatic communities (fish, benthic invertebrates) and wildlife receptors (birds, mammals). A brief summary of the receptor classes and some of the important species are given below.

Fish Community

The fish community in the Meadowlands wetlands is comprised of species adapted to the range of euryhaline conditions found in the tidal estuarine and freshwater wetlands and includes both resident and migrant (transient) species. An extensive fish survey and species inventory was conducted over a two year study in 1987-1988, which found 34 species (HMDC, 1989). The more abundant and commonly occurring fish were mummichog (*Fundulus heteroclitus*), Atlantic silverside (*Menidia menidia*), inland silverside *(Menidia beryllina*), white perch (*Morone americana*), blueback herring (*Alosa aestivalis*), Atlantic tomcod (*Microgadus tomcod*), brown bullhead (*Ictalurus nebulosus*), pumpkinseed (*Lepomis gibbosus*), American eel (*Anguilla rostrata*), bay anchoy (*Anchoa mitchilli*), striped killifish (*Fundulus majalis*) and striped bass (*Morone saxatalis*). The catch was greatly dominated by mummichog (85-91% of total catch) with silverside species, and white perch also common. Another fish survey and inventory is currently being completed (Bragin et al., in prep) which will update the species list for the wetllands and adjacent Hackensack River and several of its tributaries. The lower Hackensack River system has been declared Essential Fish Habitat by the National Marine Fisheries Service (NMFS) for red hake, black sea bass, Atlantic butterfish, and three flounder species (Day et al.,, 1999)

Aquatic Invertebrates

Aquatic invertebrates were sampled as part of the sampling and inventory survey conducted over 1987-1988 (HMDC, 1989). More recent studies (2000 – 2001) have also been completed for many of the marshes. Aquatic and epibenthic forms included several shrimp species; including mysid (*Neomysis americana*), grass (*Palaemonetes pugio*), and sand *(Crangon septemspinosa*), as well as several species of amphipods. A large number of blue crab (*Callinectes sapidus*) were taken as were polychaetes, oligochaetes, and mollusks. White-fingered mud crab (*Rhitropanopeous harrissii*) and fiddler crab *(Uca pugilator*) were also present. Aquatic insect larvae, midges (particularly Chironomidae and Ceratopogonidae) and mosquito (Aedes spp.) were found in abundance in many areas within the marshes.

Birds

Bird inventories have identified 260 bird species known to occur in the Meadowlands, including resident, migrant (spring and fall), breeding (summer) and wintering birds (Hudsonia, 2002). This diversity is due to the wide range of habitats afforded by the interspersed freshwater and brackish wetlands with upland and residual forested areas. Waterfowl and shorebirds also occur in large numbers during seasonal migration periods.

Common resident wetland-associated nesting bird species include swamp sparrow (*Melospirza georgiana*), savannah sparrow (*Passerculus sandwichensis*), red-winged blackbird *(Agelaius phoeniceus*), and marsh wren (*Cistothorus palustris*). Waterfowl species that are typically observed include Canada goose (Branta canadensis), American black duck (*Anas rubripes*), *mallard (Anas platyrhynchos*), canvasback (*Aythya valisineria*), greater scaup (*Aythya marila*), gadwall *(Anas strepera*), ruddy duck (*Oxyura jamaicensis*), green-winged teal (*Anas crecca*), and northern pintail (*Anas acuta*). Wading birds that are common to the Meadowlands include black-crowned night-heron (*Nycticorax nycticorax*), American bittern (*Botaurus lentiginosus*), glossy ibis (*Plegadis falcinellus*), great egret (*Casmerodius albus*), green heron (*Butorides striatus*), least bittern (*Ixobrychus exilis*), snowy egret (*Egretta thula*), great blue heron (*Ardea herodias*), and yellow-crowned night-heron (*Nyctanassa violacea*) (Hudsonia, 2002).

<u>Mammals</u>

Mammal surveys conducted in the late 1980 – early 1990 indicated that about 20 species of mammals were observed in the Meadowlands. These included those found primarily in the primary wetland areas as well as those found along adjacent/abutting road and rail beds, pipeline corridors, landfills, and other elevated areas (Hudsonia, 2002). The muskrat (*Ondatra zibethicus*) is probably the most important mammal species in the marshes. Other rodent species include white-footed mouse (*Peromyscus leucopus*), meadow jumping mouse (*Zapidus hudsonius*), meadow vole (*Microtus pennsylvanicus*), house mouse (*Mus musculus*), Norway rat (*Rattus norvegicus*), woodchuck (*Marmota monax*), eastern chipmunk (*Tamis striatus*), and eastern grey squirrel (*Sciurus carolinensis*). Other mammals included masked *shrew (Sorex cinerus*), eastern mole (*Sqalopus aquaticus*), eastern cottontail (*Sylvilagus floridanus*), raccoon (*Procyon lotor*), opossum (*Didelphis virginiana*), striped skunk (*Mephitis mephitis*), red fox (*Vulpes vulpes*), mink (*Mustela vison*), little brown bat *(Myotis lucifugus*) and white-tailed deer (*Odocoileus virginianus*).

#### 4.2.1.3 Species of Special Interest

The presence of federally or state-listed species (i.e., threatened/endangered species or species of special concern) in the Meadowlands was assessed. No federally listed threatened or endangered species are currently known to occur in the Meadowlands (Hudsonia, 2002). However, a number of state-listed species are potentially present. Based on the New Jersey Department of Environmental Protection (NJDEP) Natural Heritage program listing in October 2001, 42 species potentially present in the Meadowlands were listed as endangered and threatened species or were listed as rare species; including 33 bird, 3 amphibian, 1 reptile, 2 fish and 3 plant species (Hudsonia, 2002). According to the NJMC Master Plan (NJMC, 2003), twelve endangered, threatened or declining species of birds, one such reptile (wood turtle), and seven such plant species have been sighted within the Meadowlands District.

#### 4.2.1.4 Selected Representative Ecological Receptors

These species are likely to occur in the Hackensack Meadowlands wetlands and may forage on prey species at several trophic levels in the area, including vegetation, fish, and shellfish (e.g., fiddler crab). The factors considered in the selection of the ecological receptors included: (1) potentially exposed to chemical stressors in surficial media at the site; (2) presumed to play key roles in site ecosystems; (3) have complex life cycles that include sensitive life stages; and (4) conservatively representative of the range of trophic levels in food webs and food chains in the Hackensack Meadowlands. The selected receptors included wetland vegetation, benthic macroinvertebrates, mallard duck, muskrat, great blue heron, and mink.

Wetland Vegetation

The vegetation found in the estuarine (ORM, RBM, SAW, SHS) and freshwater (KFM) tidal wetlands found in the Meadowlands was selected as a receptor community. This includes the assemblage of common plants found in these areas, including, but not restricted to: saltmeadow and saltmarsh cordgrass (*Spartina*), common reed (*Phragmites*), cattail (*Typha*), loosestrife (*Lythrum*), spikerush (*Eleocharis*), and spikegrass (*Distichlis*). For a fuller listing of species see description of wetlands provided in Section 2.2. Evaluation of this receptor community assesses the potential influence of sediment quality on the overlying plant community.

Benthic Macroinvertebrates

[Benthic macroinvertebrates in the tidal wetlands were selected as a receptor community. This is a diverse community that includes macroinvertebrates in the water column (shrimp species), epifaunal (crab species), and in-faunal (midge larvae). Evaluation of this receptor community assesses the potential influence of sediment quality on the benthic macroinvertebrate community health.

Mallard (*Anas platyrynchos*)

This surface feeding duck was selected as a representative avian herbivore species for evaluation of potential risks associated primarily with sediment exposure in the wetlands. The mallard is a common surface feeding duck that inhabits freshwater and saltwater ponds and wetlands. They feed mainly aquatic plants, often filtering through soft mud for the seeds (U.S. EPA, 1993a,b). The mallard is a migratory bird, but may be present in the study area during a majority of the year. Mallards are recreationally hunted and were selected to allow the evaluation of an herbivorous exposure pathway for the aquatic sites.

Muskrat (*Ondatra zibethicus*)

The muskrat was selected as a representative mammalian herbivore wildlife species for evaluation of potential risks associated primarily with sediment exposure in the wetlands. The muskrat is a common mammal that inhabits ponds, streams, swamps, and a variety of other aquatic habitats. They feed mainly on the roots and basal portions of aquatic plants, but may also consume shoots, bulbs, tubers, stems, and leaves (U.S. EPA, 1993a,b). Selection of the muskrat allows evaluation of an herbivorous exposure pathway.

Great Blue Heron *(Ardea herodias)*

The great blue heron was selected as a representative avian carnivore for the evaluation of potential risks associated with exposure to sediment and surface water contaminants within the wetlands. Great blue herons inhabit a variety of freshwater and marine areas, including marshes, lakes, streams, and wetlands. While fish are the preferred prey, herons will also consume amphibians, reptiles, crustaceans, insects, birds and mammals (U.S. EPA, 1993a,b). Many herons migrate south during the coldest months in the northeast, leaving in October or November and returning in February or April (U.S. EPA, 1993a,b), although great blue herons are reported to remain in the area year round. Selection of the great blue heron allows evaluation of a piscivorous exposure pathway in the aquatic systems.

Mink (*Mustela vison*)

The mink was selected as a representative mammalian carnivore species for evaluation of potential risks associated with sediment and surface water exposure in wetlands. It is sensitive to PCBs and similar chemicals, and it serves as a bioindicator of mercury pollution in aquatic habitats. It was selected to evaluate the exposure of compounds in surface water and sediment due to its association with aquatic habitats and its opportunistic feeding habits which include fish, amphibians, crustaceans, shorebirds, insects, small mammals (*e.g.* shrews), and other prey items that may bioaccumulate compounds from sediments. The mink is the most abundant and widespread carnivorous mammal in North America. They feed on aquatic prey such as fish, frogs and invertebrates. Mink are active year-round and are found in a variety of aquatic habitats including rivers, streams, lakes, and swamps (U.S. EPA, 1993a, U.S. EPA, 1993b). Selection of the mink allows evaluation of a piscivorous exposure pathway for the aquatic sites.

### 4.2.2 Selection of Exposure Pathways

Potential complete exposure pathways for ecological receptors were identified through documents and maps. Exposure pathways for several groups of ecological receptors were identified as potentially relevant. Each exposure pathway includes a potential source of COPC, an environmental medium (surface water, hydric soil or sediment), and a potential exposure route. Following U.S. EPA guidance, incomplete routes of exposure were not evaluated in the SLERA. This approach is used to focus the risk evaluation on exposure pathways that are considered to be potentially complete and for which there are adequate data pertaining to the receptors, exposure, and toxicity for completion of the risk analysis. The selected exposure pathways are discussed below.

#### 4.2.2.1 Sources of Contaminants

For the Meadowlands wetlands, the sources of contaminants are due to the historical and current land use activities in the Meadowlands. The Meadowlands were widely used for waste and fill disposal during a period from the late nineteenth century through current times. In addition to these disposal activities, there was significant alteration of the hydrological and drainage patterns of the wetlands, due to the building of embankments for roads and railways, agriculture, residential encroachment, and mosquito control projects. The soils and sediments have been impacted by direct dumping and discharges as well as by stormwater runoff and groundwater leachate from bordering areas. In addition, the wetlands are susceptible to any spills or releases to the Hackensack River. Furthermore, pollutants entering Newark Bay from the polluted Passaic River, Arthur Kill, and New York Bay may be transported upstream into the Hackensack River through tidal activity.

Currently, there are approximately 2,500 acres of solid waste landfills in the Meadowlands and seven National Priority List (NPL) sites within 2 miles (Hudsonia, 2002). There are also numerous (50+) active industrial discharges, two power generating plants, three sewage treatment plants, and a number of combined sewer overflows, and emergency overflows within the Meadowlands District. All of these current and legacy discharges have led to on-going challenges to the water and sediment quality in the Meadowlands wetlands. Due to the widespread nature of the contamination, the range of potential contaminants includes heavy metals, PAHs and other hydrocarbons, PCBs, and other organic chemicals and the affected media include surface water, hydric soils, and sediments.

#### 4.2.2.2 Exposure Pathways

As part of the SLERA, it is necessary to identify and select exposure pathways through which ecological receptors may be potentially affected by relevant media at the site. The SLERA risk evaluation focused on the exposure pathways for which (1) there was a complete exposure pathway between affected media and ecological receptor, (2) the chemical exposures were the highest and most likely to occur and (3) there were adequate data pertaining to the receptors, exposure pathways, and toxicity for completion of risk analyses. Based on evaluation of the site media and ecological receptors, the following complete ecological exposure pathways were evaluated in the SLERA:

- Direct contact of sediment/hydric soil by wetland vegetation;
- Direct contact and ingestion of sediment by benthic invertebrates (e.g., shellfish);

- Direct contact with sediment and surface water by aquatic vertebrates (i.e., fish); and
- Ingestion of contaminated prey items (plants, prey species ) by vertebrate wildlife receptors (e.g., birds and mammals)

Not all exposure pathways will be evaluated. For example, the inhalation pathway, while complete, was not considered, due to the nature of the contaminants (not a large number of volatile chemicals) and the difficulty of finding toxicity reference values for this pathway. For depiction of the selected exposure pathways refer to Conceptual Site Model (CSM) (Figure 4-2).

### 4.2.3 Identification of COPCs

As noted above, the historic and current land uses have led to the introduction of a wide spectrum of chemicals into the surface water and sediments in the Meadowlands wetlands. Chemical contaminants under consideration are based on previous sampling and analytical data collected within the Meadowlands wetlands study areas. In general, these include priority pollutants (e.g., heavy metals, organics, other) for both surface water and sediment. Some ancillary data (e.g., total organic carbon, hardness) is also available for some wetlands for consideration in the SLERA. Available qualified chemical data identified in the NJMC/MERI database were evaluated. Data contained in the NJMC/MERI database were considered sufficiently validated for purposes of the SLERA unless otherwise noted. The specific data that were used in the SLERA are described in Section 4.3.

#### 4.2.3.1 Approach for Identifying COPCs

As noted earlier (Section 4.1), U.S. EPA's guidance (U.S. EPA, 1997) directs an inclusive, conservative approach by which the entire universe of contaminants at a site is initially considered and conservatively screened in the SLERA to see which contaminants require further investigation in the BERA. In contrast, the Meadowlands SLERA project, while relying on the U.S. EPA guidance as a useful guidance document for an ecological risk assessment framework, is conducting a pragmatic assessment of potential ecological risk, based mostly on available historic data, with limited supplemental sediment and benthic sampling, and a defined budget. The results of the SLERA will be used for risk management decisions and not to direct regulatory compliance and/or cleanup activities.

Screening level analysis for benchmarks protective of benthic communities and wetland vegetation were completed for all detected sediment and surface water contaminants. However, it was considered appropriate to limit full evaluation (i.e., food web modeling and development of risk curves) to a limited sub-set of the potential COPCs. Based on contractual constraints a maximum of 10 COPCs could be examined as part of food web modeling, however an eleventh COPC was added to accommodate the key contaminants of concern. These Meadowlands-wide COPCs were selected from among the list of contaminants which exceeded the screens for aquatic, benthic, and vegetation receptors for the Meadowlands wetlands of interest. Chemicals that did not have an exceedance of any media benchmark in the wetlands sampled in 2003 were eliminated from further investigation. Those chemicals that exceeded benchmarks were considered for selection as COPCs, but not all chemicals exceeding benchmark were selected as COPCs. Details of the screening of contaminants against benchmarks for protection of aquatic, benthic and plant receptors are provided in Section 4.3.2 and 4.3.3. The final selection of Meadowlands-wide COPCs was based on a number of factors including:

- distribution and magnitude of the sediment benchmark exceedances with focus on the five wetlands of interest – Kearny Freshwater Marsh (KFM), Oritani Marsh (ORM), Riverbend Marsh Wetland Preserve (RBM), Sawmill Creek Wildlife Management Area (SAW), and Secaucus High School Marsh (SHS);
- the contaminant concentration relative to reference/background data;
- distribution of COPCs between major contaminant classes (i.e., metals, organics, persistent bioaccumulation (i.e., accumulation in an organism as a result of uptake) or and toxic (PBT) chemicals);
- potential for biomagnification (accumulation to higher concentrations at higher trophic levels though dietary accumulation); and,
- best professional judgement (BPJ) (this included consideration of exceedance in other Meadowlands wetlands including Skeetkill Marsh, Harrier Meadows Marsh, Kearny Marsh (estuarine component), and other locations).

To support the identification of COPCs to carry forward into the risk assessment, particularly the food web modeling, comparison was made between sediment screening benchmarks and contaminants detected in the 2003 sediment sampling in the Meadowlands wetlands of interest. Two sets of comparison were made: (1) between sediment concentrations and sediment screening benchmarks protective of benthic invertebrates (see Section 4.3.3), and (2) between sediment concentrations and sediment screening benchmarks protective of vegetation (see Section 4.3.2). COPCs were selected using data from these analyses as well as the other factors described above.

#### 4.2.3.2 Results of COPC Selection

Based on this evaluation, eleven chemicals were carried forward into the food web modeling as Meadowlands-wide COPCs including:

- seven heavy metals (arsenic, cadmium, chromium, copper, lead, mercury, zinc);
- two pesticides (alpha-chlordane, 4,4'-DDE);
- polychlorinated biphenyls (evaluated as total PCBs); and
- polynuclear aromatic hydrocarbons (evaluated as total PAHs).

Refer to Sections 4.3.2 and 4.3.3 for screening analyses used to support the COPC selection.

### 4.2.4 Development of the Conceptual Site Model (CSM)

The end product of the problem formulation step is the development and refinement of the CSM. The CSM for the Site describes the origin, fate, transport, exposure pathways, and receptors of concern. The ecological CSM graphically portrays a series of working hypotheses regarding how the COPCs might pose hazards to the ecosystem and ecological receptors at the site. The SLERA focuses on those pathways for which (1) chemical exposure are the highest and most likely to occur; and (2) there are adequate data pertaining to the receptors, exposure pathways, and ecotoxicity.

The CSM for the Meadowlands wetlands is presented in Figure 4-2. The CSM indicates that contaminants enter the Meadowlands wetlands from historic and current sources (e.g., landfills, industrial areas, transportation corridors) via spills, releases, surface erosion and stormwater runoff, as well as groundwater discharge to the wetlands. These chemicals may also be transported to the wetlands by tidal flushing from contaminant sources (both upstream and downstream) within the Hackensack River watershed as well as sources from outside the watershed. The affected media are the sediments, hydric soils, and surface water. Exposure to ecological receptors in the wetlands to these media occurs from direct contact, ingestion or root uptake and indirectly from ingestion of plant material or fish and benthic invertebrate prey tissue. As shown in Figure 4-2, the SLERA evaluated fish and benthic invertebrates, aquatic plants, semi-aquatic mammalian and avian herbivores, and semi-aquatic mammalian and avian piscivores.

### 4.2.5 Selection of Biological Endpoints to be Assessed

As part of the CSM, ecologically-based assessment and measurement endpoints relevant to the protection of natural resources at the Meadowlands wetlands were developed (Table 4-1). Assessment endpoints describe the characteristics of an ecosystem that have an intrinsic environmental value that is to be protected (i.e., protection of warmwater fish community; no potential risk to endangered species). Typically, assessment endpoints and receptors are selected for their potential exposure, ecological significance, economic importance, and/or societal relevance (Table 4-1).

Because assessment endpoints often cannot be measured directly, a set of surrogate endpoints (measurement endpoints) are generally selected for ecological risk assessment that relate to the assessment endpoints and have measurable attributes (e.g., comparison of media concentration to screening benchmarks, results of food web models). These measurement endpoints provide a metric for evaluating potential effects of chemicals on the ecosystem components at risk.

The measurement and assessment endpoints evaluated in the SLERA are presented in Table 4-1. These endpoints were selected to (1) provide an evaluation of relevant analytical data; (2) provide a risk characterization integrating several trophic levels; and (3) serve as the basis for refinement of COPCs that may, if appropriate, be used in further tier(s) of ecological risk assessment. The broad nature of assessment endpoints proposed for this project is appropriate for a SLERA. However, such assessment endpoints may need to be refined if further assessment (e.g., BERA) were pursued.

The four assessment endpoints selected for the SLERA were:

- Assessment Endpoint #1 - Protection and maintenance of wetland plant community;
- Assessment Endpoint #2 - Protection and maintenance of estuarine sediment-associated aquatic receptors (benthic invertebrate communities);
- Assessment Endpoint #3 - Protection and maintenance of estuarine surface water aquatic receptors (invertebrate and fish communities); and
- Assessment Endpoint #4 - Protection and maintenance of a vertebrate wildlife community.

The four measurement endpoints corresponding to the above assessment endpoints were:

- Measurement Endpoint #1 - Comparison of contaminant concentrations in the sediment, to screening benchmarks for wetland plant receptors.
- Measurement Endpoint #2 - Comparison of contaminant concentrations in the sediment to sediment benchmarks for estuarine benthic receptors.
- Measurement Endpoint #3 - Comparison of estuarine surface water concentrations to chronic ambient water quality criteria or similar conservative benchmarks. Screening of sediment is also used to assess this endpoint.
- Measurement Endpoint #4 –Comparison of the total daily dose (TDD) estimated using an exposure or food web model to a Toxicity Reference Value (TRV) to estimate potential risk. The selected semi-aquatic receptors in the estuarine water area were two birds (mallard, great blue heron) representing herbivorous and carnivorous feeding habitats and two mammals (muskrat and mink) representing herbivorous and carnivorous feeding habitats, respectively. Fish are addressed as prey items for carnivorous wildlife receptors.

### 4.3 Analysis of Exposure and Effects

The analysis portion of the SLERA is based on the CSM developed in Problem Formulation. The purpose of this section is to quantify the magnitude, frequency, type, and duration of exposures by ecological receptors to site contaminants. Information is collected to define chemical sources and chemical partitioning among environmental abiotic media and organisms; estimate the bioavailability of contaminants in the relevant exposure media; and attempt to relate chemical concentrations in the relevant environmental media to the potential to produce adverse ecological effects.

The ecological exposure assessment involves the identification of potential exposure pathways and an evaluation of the magnitude of exposure of identified ecological receptors. The ecological effects assessment describes the potential adverse effects associated with the identified contaminants to ecological receptors and reflects the type of assessment endpoints selected. The methods that were used to identify and characterize ecological exposure and effects are described in the following subsections.

### 4.3.1 Development of Exposure Point Concentrations (EPCs)

The COPCs identified in Section 4.2.3.2 based on sediment screening (see Sections 4.3.2 and 4.3.3) are listed in Table 4-2, along with their minimum, maximum, and arithmetic mean concentration among the sediment samples.

For benchmark screening and food web modeling in the SLERA, the maximum exposure point concentration (EPC) was established as a conservative "worst-case" estimate of exposure for ecological receptors. The EPCs represent the contaminant concentration in various media (e.g., water, sediment, prey) to which various receptors may be exposed. Qualified sediment data from the five wetlands of interest was considered for this portion of the task. Available surface water data from previous wetland studies (Langan EES, 1999; Louis Berger, 2001; TAMS, 2001a; HMDC, 1997) were used as a surrogate for surface water COPC exposure. If a contaminant was not detected in a particular wetland, it was not included in the summary statistics for that wetland.

For ecological effects based screening for communities, the maximum detected concentration in each media from individual wetlands was the evaluated EPC. For the food web modeling, data from all wetlands was pooled and summary statistics calculated to determine EPCs. The maximum concentration of each constituent in each media (sediment and surface water) was used as the EPC in the food webs. To calculate the average, a value of one-half the sample quantitation limit (SQL) was assigned to all samples in which a contaminant was not detected.

### 4.3.2 Wetland Vegetation Receptor Evaluation

To evaluate the assessment endpoint developed for vegetation receptors, analytical chemistry results were compared to benchmark screening values. Maximum detected sediment/hydric soil concentrations from each wetland were compared against soil screening value derived for the protection of plant species. As recommended in NJAC 7:26E-3.11 (Technical Requirements for Site Remediation: Site Investigation – Ecological Evaluation), Contaminant Hazard Reviews compiled by the U.S. Fish and Wildlife Service and documents published by the Oak Ridge National Laboratory (ORNL) (Opresko, et al., 1993; Efroymson, et al., 1997) were reviewed. Phytotoxicity benchmarks for the detected contaminants were identified in Toxicological Benchmarks for Screening Potential

Contaminants of Concern for Effects on Terrestrial Plants (Efroymson, et al., 1997) and are provided in Table 4-1.

Evaluation of sediment quality in the five wetlands of interest indicated widespread exceedance of phytotoxicity-based sediment screening values (Table 4-4a). Based on this comparison, it can be seen that the organic contaminants do not exceed the available benchmarks for tPAHs and tPCBs. No phytotoxicity values were available for the pesticides, but by their very design, pesticides are not likely to be particularly harmful to vegetation. Based on these factors, it is possible to eliminate the organic COPCs from further investigation.

Comparison of the metals contaminants to available phytotoxicity screening benchmark values indicated widespread exceedances for the metals with the exceptions of cadmium and iron. Cadmium was below the benchmark in a majority of the samples and there was no phytotoxicity benchmark for iron. The remaining metals exceeded the benchmarks with the largest relative magnitude of exceedance exhibited by chromium and the lowest by copper and nickel.

Previous studies did not compare reported sediment concentrations to applicable phytotoxicity criteria. Therefore, historic data from previous wetland studies were compared to similar phytotoxicity screening criteria (Table 4-4b). Issues associated with elevated analytical detection limits for pesticides, PAHs and PCBs, preclude meaningful comparison with phytotoxicity criteria and current sampling efforts for these contaminant classes.

Maximum and mean sediment metal concentrations from previous wetland studies were compared to the phytotoxicity criteria used in this Project. Maximum copper, lead, and mercury concentrations exceeded applicable phytotoxicity criteria for all previous wetlands studied. Mean and maximum arsenic, cadmium, chromium, nickel, and zinc concentrations exceeded the criteria for all but one or two of the wetlands sampled. Mean concentrations for mercury exceeded the criteria in all wetlands sampled, while mean copper and lead concentrations exceeded the criteria in all but one wetland sampled. The magnitude of exceedance was greatest for mercury and lowest for nickel. Eight-day swamp and Kearny Marsh exhibited this greatest magnitude of exceedance for the metals analyzed, while Harrier Meadows reported the lowest magnitude of exceedance for the metals analyzed.

The characterization of the potential risk identified in this section is described in Section 4.4.1.

### 4.3.3 Aquatic and Benthic Receptor Evaluation

A number of measurement endpoints were used to evaluate the assessment endpoints developed for aquatic and benthic receptors. The effects-based screening focused on the surface water and hydric soils/sediments collected from within the wetlands and compared the maximum detected concentrations against appropriate screening values. Analytes exceeding these conservative screening values were retained and most selected for more detailed assessment in the food web models.

#### 4.3.3.1 Comparison to Sediment Benchmarks

As recommended in NJAC 7:26E-3.11, estuarine sediment benchmark values were obtained primarily from effects range-low (ER-L) and effects range-median (ER-M) values from Long et al. (1995). Additional sediment benchmark sources included the low effect levels (LELs) and severe effect levels (SELs) from the Ontario Ministry of the Environment (OMOE) (Persaud et al., 1996), and consensus-based Threshold Effect Concentrations (TECs) and Probable Effect Concentration (PECs) (MacDonald et al., 2000). Where appropriate, sediment screening values were adjusted to reflect the exposure area-specific total organic carbon (TOC) content. Historic sediment data were reviewed to determine the appropriate TOC value (downloaded from http://cimic.rutgers.edu/ecorisk/; Langan EES, 1999; Louis Berger, 2001; TAMS, 2001a, b)). Approximately 100 TOC measurements from four of the wetlands (nothing from Sawmill) were identified, ranging from 1.2% to 76% with a median of 5.1%. Therefore, 5% TOC was selected as an appropriate wetland concentration. Use of this relatively low TOC value results in lower screening values than if a higher TOC value were used, and hence contributes to the conservativeness of the screening. When available, estuarine sediment screening values were selected before freshwater values. Selected benchmarks are provided in Table 4-1.

Evaluation of sediment quality in the five wetlands of interest indicated widespread exceedance of benthos-based sediment screening values for metals in all wetlands, with the exception of chromium and iron in KFM and ORM (Tables 4-5a and 4-5b). Several of the metals are also known to bioaccumulate (cadmium, chromium, mercury). Chromium was included at the specific request of NJMC/MERI staff to allow evaluation of site-specific chromium tissue samples derived from an on-going research effort. Iron was not selected as a COPC due to its ubiquity in the natural environment, the low level of exceedance of sediment benchmark (2X), and its role as an essential element. Nickel, although found at levels exceeding screening benchmarks, was not selected based on BPJ, since experience with SLERA in other wetland environments indicates that it is usually not very bioavailable and other metals are more likely to contribute to upper trophic level risk.

The pesticides 4,4'-DDE and alpha-chlordane were selected since their maximum concentrations exceeded the benchmark values by the greatest amount among detected pesticides (e.g., DDT and related fractions). The two pesticides differ in that 4,4'-DDE is distributed widely throughout the wetlands, whereas alpha-chlordane was detected in KFM. PAHs were selected as they are a widely distributed contaminant in the wetlands and many of the individual PAH fractions were above the screening level, as was the cumulative total PAHs (tPAHs). PCBs were detected, primarily as Aroclor 1248, 1254, and 1260, and were retained due to concern regarding their levels and inherent bioaccumulative properties and potential adverse effects on higher wildlife.

Historic data from previous wetland studies were also compared to similar screening criteria (see Section 3.3). Screening results generally identified similar contaminants as exceeding applicable screening criteria. Issues associated with elevated analytical detection limits for pesticides, PAHs and PCBs, preclude meaningful comparison with current sampling efforts for these contaminant classes. As noted in Section 3.3, sediment metal concentrations observed in this Project for wetlands sampled as part of previous studies were typically within the range of observed values from these previous studies. However, in many cases the maximum observe metal concentrations from previous studies were several times to several orders of magnitude higher than those reported for this Project.

The characterization of the potential risk identified in this section is described in Section 4.4.2.

#### 4.3.3.2 Comparison to Surface Water Quality Criteria

Surface water analytical results available from previous wetland studies (Langan EES, 1999; Louis Berger, 2001; TAMS, 2001a; HMDC, 1997) were compared against federal Ambient Water Quality Criteria (AWQC) (U.S. EPA, 2002) or 2002 New Jersey Surface Water Quality Standards (NJAC 7:9b). When these values were not available, other sources were consulted. These included: secondary chronic values (SCVs), lowest chronic values (LCVs) from Oak Ridge National Laboratory (ORNL) (Suter and Tsao, 1996), and Ecological Screening Levels derived by U.S. EPA Region 5 (U.S. EPA, 2003). When available, estuarine values were selected before freshwater values. The selected surface water quality benchmarks are provided in Table 4-6.

Table 4-7 provides a comparison of average and maximum surface water concentrations for Secaucus, Sawmill, Kearny, and Oritani Marshes to the selected screening values. This comparison indicated that where data was available, the maximum EPC for all the metal COPCs - arsenic, cadmium, copper, chromium, lead, mercury, and zinc exceeded the screening value and needed to be further considered. No organic surface water data was identified during the review.

The characterization of the potential risk identified in this section is described in Section 4.4.3.

#### 4.3.3.3 Critical Body Residue Benchmark Comparison

Organisms may experience adverse effects from chemicals that bioaccumulate or biomagnify. Adverse effects may include reproductive impairment, reduced growth, mortality, or behavioral changes. The bioaccumulation factors used in the food web modeling to estimate tissue concentrations (body burden) in prey items (e.g., wetland invertebrates) are provided in Table 4-8. Comparing estimated or measured body burden (tissue concentration) to Critical Body Residues (CBRs) taken from the scientific literature can be used to assess potential risk. CBRs are literature-derived tissue concentrations that are thought to represent toxicological thresholds and can be used to evaluate potential impacts to communities.

To evaluate potential impacts to resident crab/shellfish species, predicted invertebrate tissue concentrations modeled as prey items in the food web, were compared against CBRs identified in the scientific literature. For the evaluation of CBRs, whole body No Observable Effects Dose (NOED) values were selected as the primary CBR value. NOEDs were obtained from the U.S. Army Corps of Engineers' Environmental Residue Effects Database (ERED) (http://www.wes.army.mil/el/ered/index.html). ERED is available to the public and allows the user to set variables (i.e., habitat, species, chemical, age) and query the database from selected peer-reviewed literature. ERED searches were completed during November and December 2003. CBRs for marine invertebrate species (i.e., crabs, mussels, urchins) were selected preferentially.

The CBRs for mortality, growth and reproduction for marine invertebrates are summarized in Table 4-9. These CBRs were used for comparison with the model-predicted tissue concentrations in the fiddler crab (Table 4-10). The model-predicted tissue concentrations in fiddle crab exceeded the maximum CBRs for six of the eleven COPCs. However, model-predicted tissue concentrations of chromium were substantially greater than measured tissue concentrations in blue crab (*Callinecte sapidus*) from the Meadowlands wetlands (Konsevick and Reidel, 1993). Konsevick and Reidel (1993) documented chromium concentrations ranging from 0.18 to 10.5 mg/kg$_{dw}$. Assuming 80% moisture content, these values would equate to 0.9 to 52.5 mg/kg$_{ww}$, while the model predicted concentration of chromium was 198.7 mg/kg$_{ww}$. This suggests that the model assumptions are highly conservative and likely overestimate contaminant body burden in prey species.

### 4.3.4 Wildlife Receptor Exposure Analysis

The process of assessing exposure for wildlife receptors in the study area involved estimating the potential dosage for each relevant potentially complete exposure route, and summing these estimates to derive an expected total daily dosage (TDD) for each receptor. The extent of exposure depends on factors such as the type of food consumed, feeding rates, habitat preference, and home range. The food web models for this SLERA were developed to evaluate the following two potentially complete exposure pathways:

- Incidental ingestion of surface water and sediment; and
- Ingestion of contaminated prey items.

Food web modeling was used to evaluate potential risks due to COPCs in sediment and surface water to the following representative mammalian and avian wildlife species: mallard (*Anas platyrhynchos*); muskrat *(Ondatra zibethicus);* great blue heron *(Ardea herodias);* and mink (*Mustela vison*). These receptors are described in Section 4.2.1.2 g.

#### 4.3.4.1 Food Chain Modeling

In general, exposure assumptions (body weights, food ingestion rates, relative consumption of food items, etc.) for each of these species were obtained from the EPA's Wildlife Exposure Factors Handbook, Volumes I and II (U.S. EPA, 1993a, 1993b). The exposure factors used in the food web modeling are provided in Table 4-11.

To estimate potential exposure to these wildlife receptors from contaminants in surface water and hydric soil, a TDD was estimated for each species. The TDD calculation considers the following factors:

- maximum concentrations of the COPCs in the food items that the species would consume;
- estimated amounts of abiotic media (e.g., hydric soil) that the target species would incidentally ingest;
- the relative amount of different food items in its diet, body weight;
- exposure duration (ED);
- species-specific areas use factors (AUFs), which were defined as the ratio of the organisms home range to the site surface area);
- and food ingestion rates.

Prey items for wildlife species evaluated in the food web exposure models include fish, invertebrates, and plants. In the absence of site-specific tissue data, tissue concentrations for COPCs in prey items were estimated using literature-derived biota-sediment accumulation factors (BSAFs), root uptake factors (RUFs), and bioaccumulation factors (BAFs) obtained from U.S. EPA sources, when available (i.e., U.S. EPA, 1999). The uptake and bioaccumulation factors used in the food web modeling to estimate tissue concentrations in prey items were provided previously in Table 4-8.

#### 4.3.4.2 Comparison to Wildlife Toxicity Reference Values and Benchmarks

Toxicity Reference Values (TRVs) were determined for each COPC for avian and mammalian receptors. The TRV relates the dose of a respective COPC from oral exposure with an adverse effect. The selected TRVs represent a threshold for sub-lethal effects to wildlife receptors. Sub-lethal effects are defined as measurement endpoints related to impairment of reproduction, growth, or survival. In general, ecological exposure to compounds is more likely to result in potential sub-lethal effects, rather than lethal effects. Ecotoxicological literature values, generally no observed adverse effect levels (NOAELs), were obtained from Sample et al. (1996). Studies which used the site-specific target wildlife receptors (e.g., mink) were preferentially selected and cited. When studies for these species were not available, studies using alternative species were used. Adjustment factors were used to scale NOAELs for surrogate test species to an equivalent NOAEL for receptor species using the body weight ratio algorithms of Sample et al. (1996). The absence of toxicity information for a COPC will be discussed as part of the uncertainty analyses. TRV benchmarks used to assess potential risk to wildlife receptors are provided in Table 4-12.

Modeled Total Daily Dose (TDD) for each COPC and target wildlife receptor is summarized in Table 4-13. These modeled TDDs are compared to applicable reference criteria in the next section. Details regarding individual components of the TDD are described in Appendix E Tables E8 – E12. As illustrated in Table 4-13, the comparison of modeled TDD for each wildlife receptor to its respective TRV reveals that TRVs are exceeded for the mink and heron for all COPC metals except chromium in the case of the mink and arsenic for the heron. Muskrat and mallard TDD for mercury only marginally exceeded the applicable TRV. For organic COPCs, the only TRVs exceeded were for PCBs in the mink, and PCBs and DDE in the heron. No organic COPC TRVs were exceeded for the muskrat or mallard.

Exposure estimates were not calculated for historic sediment data. However, such data were incorporated into food web modeling for the development of wildlife risk curves, as described in Section 5.0. The risk curves provided in Section 5.0 provide insight into the relationship between risk (as expressed by a Hazard Quotient) and modeled total daily dose over a range of sediment and water concentrations.

Characterization of the risk associated with the exposures identified in this section is provided in Section 4.4.4.

### 4.4 Risk Characterization

This section presents the risk characterization for ecological receptors potentially exposed to surface soil and sediment at the Meadowlands wetlands of interest. The ecological risk characterization summarizes the results of the ecological effects evaluation (Section 4.3), considers potential sources of uncertainty (Section 4.5), and provides interpretation of the ecological significance of findings. The results of the SLERA risk analysis were analyzed and interpreted to determine the likelihood of adverse environmental effects, and to determine whether a conclusion of no significant risk could be reached for each of the four assessment endpoints evaluated. The results of these efforts were interpreted in terms of potential current ecological risk and to provide directions for future decision-making (e.g., about the potential for the restoration of wetlands) in the Meadowlands.

Risk results for the four assessment endpoints are presented and discussed. Two general approaches for ecological risk characterization were employed.

- Sediment and surface water concentrations in excess of conservative screening values were identified and the level of exceedance, as measured by the ratio of the EPC to the screening value or Environmental Effects Quotient (EEQ), identified and discussed. When the EEQ was less than one, it was assumed that exposure to the COPC would not be associated with adverse effects to single receptors. When the EEQ was greater than one, the potential for risk could not be excluded; and

- Potential risks to the representative vertebrate wildlife species associated with ingestion (as determined using food web models) were evaluated using the hazard quotient (HQ) approach. The HQ was calculated for each receptor/COPC pair by dividing the estimated Total Daily Dose (TDD) by the corresponding TRV. When the HQ was less than one, it was assumed that exposure to the COPC would not be associated with adverse effects to single receptors (i.e., inhibited growth, reproduction and survival of the individual organism) or wildlife populations. When the HQ was greater than one, this indicated that there was a potential for adverse effects to a receptor because of COPC exposure (or that the potential for risk could not be excluded).

Where the potential for risk exists (or could not be excluded) the ecological significance of these results is discussed. The results of the SLERA are described and summarized on the based of the four identified assessment endpoints of the CSM (see Figure 4-2). The results of each assessment endpoint are reviewed and discussed with regard to potential adverse risks posed to receptors. Where applicable, the cumulative effects of multiple measurement endpoints for an assessment endpoint (Section 4.2.5) will be discussed.

Results of the risk characterization of each of these assessment endpoints are summarized below.

### 4.4.1 Endpoint # 1 – Protection of aquatic wetland plant species

Assessment Endpoint #1 was the protection of plant communities in Meadowlands wetlands of interest. The single measurement endpoint was the comparison of the hydric soil/sediment contaminant Exposure Point Concentrations (EPCs) to appropriate phytotoxicity-based screening values for detected contaminants as described in Section 4.3.2. Data were compared against selected phytotoxicity benchmarks in Tables 4-4a and 4-4b. Risk to wetland vegetation was characterized using EEQs, which are the ratio of the EPC to the screening value (Tables 4-14a and 4-14b for the current study and historic studies, respectively). Where the calculated EEQ is less than one, it is assumed no risk exists for that particular receptor.

The exposure and effects assessment (Section 4.3.2) indicated that phytotoxicity benchmarks were exceeded for all metals at least once in all the studied wetlands. Comparison of the maximum sediment values for the 2003 study wetlands indicated that chromium had the highest EEQ (509.5) and copper (1.8) the lowest. Similar results were evident for the historically studied wetlands; however the magnitude of EEQs were an order of magnitude higher (EEQ for chromium = 6745; EEQ for copper = 18.2). In contrast, based on the calculated EEQs for the 2003 study wetlands (Table 4-14a), it can be seen that no risk exists for organic contaminants (EEQ < 1 for tPAHs and tPCBs). No risk was evident for PCBs for historically studied wetlands; and no qualified data was available to assess tPAH concentrations for the historic studies. As mentioned in Section 4.3.2, pesticides were assumed not be of a concern for vegetation. Based on these factors, it is possible to eliminate the organic COPCs from further investigation for wetland plants.

While some of the EEQs are elevated, the magnitude of the exceedance does not necessarily correlate directly with adverse effects. Further evaluation of the significance of the sediments is limited since no well-recognized soil severe effect level (SEL) benchmark concentrations have been established for plants. Recall, that comparison to SELs (when available), which reflect adverse impact in at least 50% of the studies examined, can provide additional insight as to the severity of potential risk.

It is important to note that while chemicals are typically eliminated because the have not exceeded a screening level benchmark, the exceedance of that benchmark by a COPC does not necessarily signify actual ecological risk, but does preclude a finding of no potential for risk. While there may be a conservative presumption of risk, in practice it indicates that the COPC needs to be further investigated, usually with special attention to site-specific factors and confirmation through other lines of evidence (e.g., field measures of plant community health) before adverse ecological risk can be concluded.

### 4.4.2 Endpoint # 2 - Protection of Estuarine Benthic Macroinvertebrate Receptors

Assessment Endpoint #2 was the protection of macroinvertebrate receptors in the estuarine sediments in the Meadowlands wetlands of interest. The two measurement endpoints used for this assessment endpoint were comparison of maximum EPC to estuarine screening values and the comparison of body burden to critical body residues (CBRs).

#### 4.4.2.1 Sediment Screening Analysis

The exposure and effects assessment (Section 4.3.3) indicated that benthos-based sediment screening values were exceeded for all metals in all the studied wetlands. The single measurement endpoint was the comparison of the maximum EPC to estuarine sediment screening values, as expressed by a calculated EEQ (Table 4-14a). The calculated EEQs based on the maximum EPC for sediment were greater than one for all COPCs, and ranged from 81.3 (tPCBs) to 2.0 (tPAHs). Based on this evaluation, the COPCs exceeded the screening values and would need to be further considered. Table 4-14b summarizes EEQs for qualified historic wetland study data. Maximum EEQs for the historic wetland data were greater than 10.0 for all COPCs except alpha-chlordane, and were typically an order of magnitude greater than those observed in the 2003 study wetlands.

To gain additional insight into the potential magnitude or risk represented by the concentrations in the wetlands of interest, the maximum hazard quotients were compared to severe effect level benchmarks (e.g., ER-M or SEL values from Long et al., 1995 or Persaud et al., 1996). These severe or probable effect levels are concentrations of COPC typically found in association with degraded or impacted benthic communities. This comparison is shown in Table 4-15. It shows that the 4,4'-DDE, tPAH, arsenic, cadmium, and copper do not exceed the SELs for the 2003 study wetlands, while alpha-chlordane (EEQ = 1.3) chromium (1.4) and lead (2.6) slightly exceeded the SELs. Those COPCs that significantly exceed the SEL benchmarks in the 2003 study wetlands included mercury (8.7), tPCBs (10.3) and zinc (16.8).

Maximum sediment concentrations from the historic wetland studies data exceeded the SELs for all metals, as well as PCBs. The disparity in the results between the current study wetlands and those sampled as part of historic studies is not easily explained. Data were rechecked against original data electronic data files to ensure that the discrepancy was not associated with a unit-conversion error or other data entry error. No errors were found. Maximum metal concentrations in the historic studies were all for samples from Eight-day Swamp and Kearny Marsh. Only maximum concentrations for copper and zinc were co-located at the same sampling location in Eight-day Swamp. Concentrations reported for both of these wetlands range over several orders of magnitude, suggesting that the elevated concentrations are not likely a result of misreported units in the original data sets. A comparison of 25$^{th}$, 50$^{th}$, and 75$^{th}$ percentile concentrations between the 2003 study wetlands and those for the historic wetland studies reveals concentrations that are much more closely aligned typically differing by less than a factor of 2. Maximum concentrations for the 2003 study wetlands were in most cases within a factor of 2 of the 75$^{th}$ percentile concentrations from prior studies. Some of the observed differences between maximum concentrations from this study and historic studies may reflect the actual differences in environmental concentrations within and among the various wetlands.

Using data from the 2003 study wetlands, tPCBs, mercury and zinc should be considered as potential risk "drivers" (i.e., chemicals that are more likely to be responsible for community impacts) for the Meadowlands wetland systems. These same COPCs would also be identified as primary risk drivers using the 75$^{th}$ percentile concentrations from historic studies. Use of maximum historic concentrations would suggest that chromium and cadmium may also be important risk drivers.

#### 4.4.2.2 Critical Body Residues (CBRs)

The potential impact of sediment concentrations on a crustacean species (i.e., fiddler crab) was evaluated. Although not found in many of the Meadowlands wetlands, fiddler crabs were used as to represent a key guild and prey category for higher trophic level wildlife. The fiddler crab was selected since it is a common estuarine species, is in direct contact with the sediments (e.g., burrowing activities) and is appropriate surrogate for the blue crab, for which data on site-specific tissue residues of chromium are available; thus allowing a site-specific check of body residues predicted by the food web modeling. Evaluation of the potential ecological risk to this organism by sediment provides an indirect measure of potential prey availability for the mink and heron receptors.

The potential ecological risk posed by wetland sediment concentrations to invertebrate omnivores was assessed by comparison of modeled fiddler crab tissue concentrations to CBRs (see Section 4.3.3.3) associated with adverse effects to other marine invertebrates (Table 4-10).

Comparison of the modeled invertebrate tissue concentration to the range of literature values of tissue concentrations associated with adverse ecological effects indicates that the modeled concentrations typically exceed the lowest levels of the effect range and often the upper bounding values as well. In particular, modeled values of arsenic, copper, chromium, lead, zinc, and 4,4'-DDE would indicate potential concern. Modeled values of cadmium, PAHs, alpha-chlordane, and tPCBs are within the range of effects. Only the predicted concentrations of mercury are well below those identified in the literature as being problematic.

As is discussed further in the Uncertainty Section (see below), the use of uptake factors to estimate the potential tissue concentration only provides an approximate estimate. The use of site-specific tissue samples is far preferable because tissues integrate the combined effects and variability of localized populations, habitats, substrates, and COPC bioavailability (both in surface water and sediment). In most cases, it is expected that modeled tissue concentrations are more likely to overestimate exposure. This appears to be the case for the Meadowlands wetlands. For example the range and mean of chromium in blue crab from Sawmill Creek (SAW) and Berry Creek wetlands (ORM is part of this wetland complex) were 0.9 to 10.5 and 6.9 mg/kg, respectively (note: MERI tissue data was converted to wet weight basis)(Konsevick and Reidel, 1993). These Meadowlands wetland specific tissue residues are far below the modeled concentration of 199 mg/kg, by from 1 to 2 orders of magnitude. This comparison indicates that further investigation of ecological risk at the Meadowlands wetlands may wish to consider collection of site-specific tissue samples to reduce the amount of uncertainty and provide a better approximate of potential risk.

### 4.4.3 Endpoint #3 - Protection of Estuarine Surface Water Receptors

Assessment Endpoint #3 was the protection of estuarine fish and aquatic invertebrate (water column and benthic) receptors in the Meadowlands wetlands of interest. The single measurement endpoint was the comparison of the surface water EPC to estuarine surface water screening values, chronic Ambient Water Quality Criteria (AWQC) for marine waters as provided in NJAC 7:9b (Table 4-7). The surface water EPC was based on pooled data taken from several previous water quality sampling efforts (Langan EES, 1999; Louis Berger, 2001; TAMS, 2001a; HMDC, 1997). Both the number of measured parameters and their detection limits varied between efforts, which adds considerable uncertainty to their interpretation. However, they do provide an indication of which chemicals in the surface water are likely to be elevated relative to others and provide an approximate indication of potential risk.

The EEQ for all the metal COPCs - arsenic, cadmium, copper, chromium, lead, mercury, and zinc were greater than one indicating a potential risk for aquatic life (fish and invertebrates) and needed to be further considered. Several COPCs were elevated substantially over the screening criteria, especially lead (EEQ = 63.0) and copper (EEQ = 248.4). The maximum concentration contributing to the copper EEQ was 700 µg/L and the maximum concentration contributing to the lead EEQ was 510 µg/L, both reported in Secaucus High School Marsh (SHS) (Tams, 2001). These maximum concentrations were observed a one of four sampling locations within the marsh, with the three remaining samples from this wetland showed non-detection for copper and lead. Likewise, these reported SHS maximum concentrations are an order of magnitude greater than concentrations reported in other wetlands. While, these data may be suspect, EEQs for these contaminants would still exceed 30 if the next greatest concentration were used. The EEQ values are conservatively based on the total metal and not the dissolved fraction (which was not available for many locales) and likely overestimate potential risk (see discussion of uncertainty in Section 4.5). NJDEP Surface Water Quality Standards include translators for total metals concentrations, which depending on the metal would equate the dissolved metal concentration with 84 – 99% of the total metal concentration.

### 4.4.4 Endpoint # 4 - Protection of Semi-Aquatic Wildlife Receptors

Assessment Endpoint #4 was protection and maintenance of semi-aquatic wildlife populations of avian and mammalian receptors. The potential risk posed to wildlife receptors (avians and mammals) was evaluated for site-related contaminants in environmental media in the estuarine environment (surface water, sediment, prey food items). Four representative species were evaluated using a maximum Hazard Quotient (HQ) (Table 4-16). As described in Section 4.3.4, total daily dosage (TDD) or the amount of each COPC that a receptor would intake on a daily basis via the identified exposure pathways was estimated using food web models. These calculated TDDs were then compared to Toxicity Reference Values (TRVs). No observed adverse effect levels (NOAELs) were selected as the TRVs for this assessment (Section 4.3.4). A HQ was calculated by dividing the TDD by the TRV for each COPC and receptor. Where the HQ is less than 1.0, risk is assumed to be absent. Where the HQ is greater than 1.0, there is a potential for risk, and further investigation may be required to further assess the potential risk.

The selected semi-aquatic receptors in the estuarine water area were two birds (mallard, great blue heron) and two mammals (muskrat, and mink) representing herbivorous and carnivorous feeding habitats, respectively. The results are discussed below by wildlife receptor.

#### 4.4.4.1 Mink

The mink represents exposure to a semi-aquatic mammalian carnivore. The results of the food web models for the mink indicate potential concern regarding maximum exposure of all COPCs except chromium, alpha-chlordane, 4,4'-DDE, and tPAHs (Table 4-16). The exceeded HQs range from HQ =

2.03 (lead) to HQ = 443 (mercury). Further evaluation of this receptor and its potential exposure pathways of concern are warranted.

#### 4.4.4.2 Muskrat

The muskrat represents exposure to a semi-aquatic mammalian herbivore. The results of the food web models for muskrat indicate potential concern regarding maximum EPC exposure of arsenic (HQ = 2.55) and mercury (HQ = 1.47) (Table 4-16). All other contaminants had HQs < 1 and do not pose a potential concern to this receptor. Further evaluation of this receptor and its potential exposure pathways of concern are warranted, but the level of potential risk is perceived to be slight, even under highly conservative assumptions.

#### 4.4.4.3 Mallard

The mallard represents potential COPC exposure to a semi-aquatic avian herbivore. The results of the food web models for mallard indicate potential concern regarding maximum EPC exposure of mercury (HQ = 1.47). All other COPCs had HQs < 1 and do not pose a potential concern to this receptor (Table 4-16). Further evaluation of this receptor and its potential exposure pathways of concern are warranted, but the level of potential risk is perceived to be slight, even under highly conservative assumptions.

#### 4.4.4.4 Great blue heron

The Great blue heron represents exposure to a semi-aquatic avian omnivore. The results of the food web models for the heron indicate potential concern regarding maximum EPC exposure of all COPCs except arsenic, alpha-chlordane, and tPAHs (Table 4-16). The exceeded HQs ranging from HQ = 1.9 (copper) to HQ = 1,190 (mercury). Further evaluation of this receptor and its potential exposure pathways of concern are warranted.

### 4.5 Uncertainty

Uncertainty is introduced into the SLERA in several places throughout the process. Every time an assumption is made, some level of uncertainty is introduced into the risk assessment. This is particularly true for this SLERA, where the conservative assumptions are designed to provide an extremely conservative (maximum) exposure term to the receptors; with the presumption being that if no potential risk is indicated under such stringent conditions, there is unlikely to be risk under any foreseeable circumstances. However, it is important to remember that the finding of potential risk under the maximum exposure scenario does not, in itself, indicate that ecological risk is present under actual site-specific conditions. To evaluate the potential influence of the conservative assumptions, the critical assumptions are examined and risk results compared to scenarios for which more realistic site-specific factors are evaluated. In accordance with U.S. EPA guidance (U.S. EPA, 1989), the potential major sources of uncertainty associated with the risk characterization process are identified and the potential influence on the risk results considered.

There are many potential sources of uncertainty in the risk assessment process; some are more important than others. Some of the sources of uncertainty in the SLERA are common to both aquatic and terrestrial assessments, while other refer to specific endpoints. Some major areas of uncertainty for the Meadowlands SLERA include:

- Site Characterization (e.g., the level of site characterization, the quality of the analytical data);
- Selection of Analytes (e.g., use of chronic toxicity, sensitivity of receptors);
- Exposure Assessment (e.g., assumptions about the frequency, duration, and magnitude of exposure for the receptors);
- Ecological Effects (e.g., the availability and potential range of dose-response data); and
- Risk Characterization (interpretation of the significance of risk results to local species and communities).

The significance of background or local conditions for the aquatic environment is also considered. These various uncertainties were assessed and considered as to whether they potentially over or under-estimate risk for this site. Discussions of selected sources of uncertainty are given below.

### 4.5.1 Uncertainty Associated with Site Characterization

The environmental media (surface water, sediment) in the Meadowlands wetlands of interest were considered sufficiently but not well-characterized for purposes of the SLERA. Data was available for the sediment in the selected wetlands of interest and were the basis of EPCs for the SLERA. The surface water EPC was based on pooled data taken from several prior water quality sampling efforts. Both the number and type (total vs. dissolved) of measured water quality parameters and their detection limits varied between efforts. The surface water data was used in estimating the EPCs of prey items (macroinvertebrates and fish). In estimating the EPCs, it was necessary to assume that the analytical chemistry results accurately reflect the exposure conditions within the wetlands of interest. This includes several standard assumptions:

- Sampling locations within the wetlands were assumed to represent contaminant concentrations within the entire wetland;
- Variability due to sampling and analytical analysis was assumed to be less than normal variability;

- Concentrations reported in the analytical results were assumed to be representative of the whole medium sample.
- If these assumptions were not met, the collected samples could either over- or underestimate the true exposure and potential for risk to ecological receptors. In one case (i.e., KFM), a single sediment sample is the basis for potential ecological concern regarding alpha-chlordane.

Surface water data were taken directly from the NJMC/MERI environmental database and are not co-located in time and spatial coordinates with the 2003 sediment sampling. The surface water data is assumed to provide adequate characterization of the media and may or may not reflect current site conditions. Surface water sampling was beyond the scope of this project.

### 4.5.2 Uncertainty Associated with Analyte Screening/Selection Process

The initial selection of contaminants was based on comparison of a maximum value to a conservative criterion or benchmark (often based on protection of a sensitive species not found on-site). This method of selection results in establishing a conservative "worst-case" estimate of potential ecological risk, which may not accurately portray actual ecological risks expected by on-site species. On-site species will encounter a variable spectrum of environmental conditions by averaging exposure through utilization of the site as opposed to being restricted to areas of the highest contamination (i.e., "hotspots"). The usefulness of the conservative-based screening is to identify wetlands and/or media where no adverse ecological effects would be expected under the worst case and which can then be eliminated from further consideration.

The aquatic risk assessment used toxicity values based on chronic effects from the NJ WQS or U.S. EPA to analyze the potential for ecological risk to estuarine communities. Chronic toxicity values were used as screening benchmarks because it was assumed that surface water and sediment indicator species would experience continuous, chronic exposure. The assumption of chronic exposure may be realistic for the sediment-associated species, but may be relatively conservative for the surface water species. Differential species sensitivity to the contaminants may result in the screening benchmarks being underestimates or, more likely, overestimates of potential acute and chronic toxicity for many aquatic organisms.

The use of total metals for estimating exposure in surface water samples is likely to be a conservative estimate of metal bioavailability. Current U.S. EPA guidance indicates that the dissolved fraction of metals is more appropriate for comparison to water quality standards (U.S. EPA, 2002). The use of total metals for screening purposes would tend to overestimate potential ecological risk to aquatic receptors.

For many some COPCs, appropriate media-specific screening values were not available and these contaminants were conservatively carried forward in the SLERA (e.g., phytotoxicity benchmarks for organics). Since the contaminants that were not screened out were included in as contaminants in the food webs, this uncertainty extends into the food web models.

### 4.5.3 Uncertainty Associated with Exposure Assessment

Considerable uncertainty exists with the exposure assessment for the ecological receptors. This uncertainty exists for the community receptors and individual wildlife receptors. These were considered separately for wildlife receptors in the terrestrial the aquatic ecosystems.

Exposure to aquatic and semi-aquatic receptors has considerable uncertainty due to exposure duration, habitat use factors, and bioavailability of sediments. These are described below.

Chronic toxicity values were used as screening benchmarks for aquatic receptors because it was assumed that ecological receptors would experience continuous, chronic exposure. Exposure in the aquatic environment is unlikely to be continuous for many fish species in the riverine and estuarine habitat, as they are generally transitory and are likely to move in and out of the regions of the Hackensack River adjacent to the Meadowlands. In addition, the fishery and macroinvertebrate habitat is considered degraded due to many factors within the wetlands and Hackensack River watershed including physical habitat (dredging and filling), current and historic dumping, landfilling, leachate of toxics, restricted or impounded flows, local non-point sources, and downstream tidally dispersed, hazardous waste-impacted surface water and sediment.

The potential exposure of several semi-aquatic receptors would be significantly reduced due to the large sizes of their home ranges. The use of a conservative assumption of Exposure Duration (ED) = 1.0 and Area Use Factor (AUF) = 100% would overestimate the potential risk to semi-aquatic receptors. The insertion of more realistic habitat use factors would significantly reduce potential exposure, particularly for the piscivores.

The bioavailability of several sediment contaminants, especially metals, may be overestimated. It has been shown that acid volatile sulfides (AVS) have a high binding capacity for divalent metals (e.g., Cr, Cu, Pb, Ni, Zn) in sediments, thereby reducing or eliminating the bioavailability to sediment-associated receptors. AVS concentrations are high in anaerobic sediment environments; such is likely to exist in the tidal wetlands of the Hackensack River and its tributaries, since they contain highly organically enriched sediments where biological demand is likely to be high. Therefore, AVS concentrations in the estuarine sediments may reduce the potential risk to sediment receptors in the wetlands. This information was not available for the SLERA, so 100% bioavailability was conservatively assumed.

### 4.5.4 Uncertainty Associated with Ecological Effects

The dose-response values used for the several species were extrapolated from data on similar species because no direct dose-response information was available for the representative species. This extrapolation involved conservative assumptions for several factors including allometric scaling, application of safety factors, and use of NOAELs; thus, it is likely that the scaled dose-response values chosen will result in overestimates of the potential for adverse effects.

A major source of uncertainty in the application of the hazard quotient method is the availability and/or source of toxicity data used for benchmark concentrations (e.g., NOAEL or LOAEL). Typically, the lowest data points among the available ecotoxicological database were conservatively selected as the screening benchmark concentrations. The lowest data point observed in the laboratory may not be representative of the actual toxicity which might occur in the environment. Using the lowest reported chronic toxicity data point as a benchmark concentration (as was done in this assessment with a NOAEL-based TRV) is a very conservative approach, especially when there is a wide range in reported toxicity values for the relevant species.

Another source of uncertainty is the estimate of uptake and/or bioaccumulation from environmental media into plants and wildlife. In an ecological risk assessment context, the best data to estimate bioaccumulation of contaminants in tissues will always be site-specific data. Ideally, plant and animal tissues should be collected from multiple areas within the contaminated site and from reference areas (preferable at locations where soil or sediment samples are also collected). In the absence of site-specific tissue data, uptake factors (UFs) or models may be used to estimate tissue concentrations, but may under or overestimate potential exposure. As indicated by the comparison of site-specific results of the tissue residues for chromium in blue crab vs. that modeled for invertebrates for food web models, the U.S. EPA-approved uptake factors used likely overestimate potential exposure.

### 4.5.5 Uncertainty Associated with Risk Characterization

Extrapolation of the potential for community, population, or ecosystem effects from the examination of one or more representative species is a major source of uncertainty for both the aquatic and terrestrial analyses. The underlying assumption is that potential effects on one representative species are consistent with the effects on similar species and representative of the potential for effects on the particular ecosystem being investigated. It is assumed that if the representative species were not affected, the potential for ecosystem-level effects are also unlikely.

The representative species were chosen to evaluate potentially high exposure to environmental media at the site, resulting from trophic level, habitat preferences, or limited home range. The effect of these assumptions may be to overestimate the potential for adverse ecological effects to other species.

It should be recognized that ecological risk typically considers potential risk to communities or populations. It is difficult to predict how an adverse effect on an individual organism might affect the ecosystem as a whole. If adverse effects are predicted for an individual, it does not necessarily mean that the community, population or ecosystem will be similarly affected. Even if one subset of the ecosystem is impacted in a localized area, it may not be a perceptible impact to the overall ecosystem (e.g. loss of individual fish may not affect resident population) is highly unlikely to affect resident populations at the Site. However, in some cases, particularly for threatened/endangered species or species of special concern, (e.g., great blue heron) the loss of individual organisms may mean a conclusion of unacceptable ecological risk at the site.

A number of interpretation approaches for HQ values have been proposed and should be considered in evaluating the ecological significance of HQs to populations and communities (e.g., Barnthouse et al., 1982, Barnthouse et al., 1986; U.S. EPA, 1 1988, 1992; Suter, 1993). For example, calculated HQ values should be sorted into the three categories according to the following thresholds and ecological receptors. These are identified below along with interpretation of the level of ecological risk concern associated with HQ values as follows

A: HQ >10 for risk to common species, or HQ >1.0 for risk to protected species

-> Recommended for additional risk investigation (i.e., ERA Tier 2) to reduce uncertainty or to consider potential effects of remediation; probable concern for potential ecological risk

B: HQ >1.0 to 10 for risk to common species, or HQ ~1.0 (e.g., 0.9 to 1.1) for risk to protected species

-> Considered for additional risk investigation (i.e., ERA Tier 2); possible concern for potential ecological risk.

C: HQ ≤1.0 for risk to common species, or HQ <1.0 for risk to protected species.

-> Not considered for additional risk investigation; negligible concern for ecological risk.

For interpretation of Category "A" HQs, the traditional HQ threshold of 1.0 is changed to 10.0 for common species because the appropriate assessment endpoint is at the population level where an adverse effect on a single individual of the population is not expected to be ecologically significant, and because the effect of applying multiple conservative assumptions tends to overestimate potential risk. The threshold of 1.0 is retained for protected species where the loss of an individual is assumed to have an adverse effect on survival of the species. The level of ecological risk concern associated with the three sets of HQ values may be interpreted as follows:

Several elements should be considered in a weight-of-evidence approach to assess the ecological significance of the calculated HQ values for measurement endpoints in Categories A and B. These factors include the magnitude of calculated risk, the concurrence of multiple measurement endpoints, the “weight” of the measurement endpoint; and the status of the receptor (threatened, endangered, or common species).

In summary, target levels of risk for ecological receptors are site-specific and should reflect the ecological significance of the risk. The threshold between acceptable and unacceptable risk is dependent upon the assessment endpoints (which should reflect risk management) and the uncertainties associated with the exposure concentration or dose and the effects concentration or dose.

### 4.6 SLERA Risk Conclusions

A SLERA was conducted for the Meadowlands wetlands to evaluate their potential to pose adverse ecological risk to ecological communities and wildlife receptors. Aquatic communities and wildlife receptors at the Meadowlands wetlands were evaluated. Four assessment endpoints were selected to provide a wide spectrum of potential exposures and receptors.

Surface water aquatic receptors, sediment-associated benthic receptors, and semi-aquatic wildlife receptors were evaluated in the Meadowlands wetlands. Potential risks evaluated included direct contact to environmental media (surface water, sediments) and bioaccumulative effects (food items). For purposes of this SLERA, the COPCs evaluated were limited to 11 contaminants including seven heavy metals (arsenic, cadmium, chromium, copper, lead, mercury, zinc), two pesticides (alpha-chlordane, 4,4’-DDE), tPCBs, and tPAHs.

All assessment endpoints were evaluated using conservative assumptions in the selection of contaminant concentration, exposure terms, screening benchmarks, and ecological effects. Under this conservative screening approach and using conservative assumptions for food web modeling, potential ecological risk was indicated for the four assessment endpoints. These results indicate that further ecological risk investigation or remedial activities may be warranted.

Due to these conservative maximum exposure assumptions, there is a great deal of uncertainty associated with the risk estimates. The sources and potential influences of this uncertainty were described and the results of this conservative screen were further evaluated to provide a better interpretation of the potential risk at the Site. It is likely that alternative assumptions or using site-specific factors may lead to a significantly lesser degree of potential risk.

The results of the SLERA are also further extrapolated as the standard risk curves described in Section 5.0, and in a qualitative analysis of benthic community composition as a function of sediment chemistry and toxicity in Section 6.0.

# 5.0 WILDLIFE RISK CURVES

Using the results of the SLERA for the Meadowlands wetlands of interest (Section 4.0), the food web models were used to generate “wildlife risk curves” for selected COPCs for four ecological receptors (mallard, heron, muskrat, mink). The wildlife risk curves relate the predicted HQs (HQ = food web modeled TDD/TRV) for a particular sediment concentration and receptor species to different sediment and water COPC concentrations (i.e., 25$^{th}$, 50$^{th}$, 75$^{th}$, and 100$^{th}$ percentile concentrations calculated from pooled data). These curves are intended to allow NJMC/MERI to quickly evaluate sediment data collected in the future as to potential ecological risk implications.

## 5.1 Construction of the Wildlife Risk Curves

As noted previously, the Meadowlands-wide COPCs were arsenic, cadmium, chromium, copper, lead, mercury, zinc, alpha-chlordane, 4,4’-DDE, tPAHs and tPCBs Sediment data for each COPC were pooled from the wetland sites sampled as part of this study and in surficial (0 – 6”) samples collected during previous studies (Langan EES, 1999; Louis Berger, 2001; TAMS, 2001a; TAMS, 2001b; ECI, 1997a; ECI, 1997b; HMDC, 1997; Weis, J. and P. Weis, undated). Surface water data available from previous wetland studies (Langan EES, 1999; Louis Berger, 2001; TAMS, 2001a; HMDC, 1997) were also pooled for each COPC. .

The pooled sediment data was used generate a frequency distribution curve to identify the 25th, 50th, 75th and 100th percentile sediment concentration values for each COPC. These values are presented in Table 5-1.

Surface water data used for calculation of percentile distribution for metals COPCs were based on pooled data taken from previous wetland studies that included water quality sampling efforts (see Appendix E, Table E2). Since there were no available surface water quality data for organic COPCs, organic COPC concentrations were predicted based from potential dissociation from the sediments (see Appendix E, Table E12 for details of the equilibrium partitioning calculation). For example, the 25$^{th}$ percentile for surface water concentration for tPCBs was based on the predicted concentration dissociating from the 25$^{th}$ percentile sediment concentration. For calculation purposes a site-specific sediment TOC value of 5% was used. This value represents the median values (range of 1.2% to 76%) of approximately 100 TOC measurements from four wetlands (KFM, ORM, RBM, SHS) (Langan EES, 1999; Louis Berger, 2001; TAMS, 2001a, 2001b). The representative surface water values are presented in Table 5-1. Analogously, the 25$^{th}$ prey (fish) concentration was based on bioaccumulation from the 25$^{th}$ percentile surface water concentration.

These surface water and sediment percentile concentrations were input into the food web models using the exposure parameters and assumptions described in Section 4.3, to provide estimates of concentrations in media and forage or prey items (Table 5-2). Using these media and estimated forage or prey COPC concentrations, HQs were calculated for the wildlife receptors (i.e., mallard, great blue heron, muskrat, mink) corresponding to the 25$^{th}$, 50$^{th}$, 75$^{th}$, and 100$^{th}$ percentile concentrations.

## 5.2 Risk Curves for Wetland Wildlife Receptors

The results of the food web models and HQs generated using the percentile media and prey concentrations (Table 5-2) are presented in Table 5-3. Hazard quotients (HQ = modeled total daily dose divided by the applicable TRV) are provided for the four wildlife receptors of interest. All HQs greater than 1.0 are shown in bold-italic text in the table. HQs developed for a range of concentrations (Table 5-3) provides a better characterization of the potential range of risk posed by site COPCs than that afforded by consideration of just the maximum EPC (Table 4-16).

To more easily understand the pattern of potential risk from the distribution of sediments at the Meadowlands wetlands of interest, the HQ values were plotted against sediment concentrations to develop a series of wildlife risk curves for the COPCs. These risk curves are presented for metals in Figure 5-1 through 5-7, for pesticides in Figures 5-8 and 5-9, for tPAHs (Figure 5-10), and for tPCBs (Figure 5-11). The risk curves for all four wildlife receptors are displayed on the same figure for each COPC.

In these curves, the risk HQ is calculated for the four media concentrations corresponding to the 25$^{th}$, 50$^{th}$, 75$^{th}$, and 100% concentrations and the curves represent interpolation of the expected risk at intermediate media concentrations. Note that there is no attempt to predict HQs for media concentrations lower than the 25$^{th}$ percentile.

Inspection of these risk curves (Figures 5-1 through 5-11) indicates that the potential risk associated with different COPCs differs significantly in magnitude and pattern of risk with increasing sediment concentrations and that these patterns also differ depending on the receptor/trophic pathway involved. In some cases the level of estimated risk is so low for some receptor-COPC combinations that it is virtually indistinguishable from the X-axis. As will be discussed later, some of the COPC curves are also significantly affected by the influence of the surface water concentration.

Several points can be made in general for these wildlife risk curves. In general, greater risk are predicted for the piscivores (heron, mink) than for the herbivores (mallard, muskrat) over the majority of the COPCs, with the mink having the highest HQs and the mallard the lowest. Higher risks are associated with the metals than the organic COPCs. For most COPCs (cadmium, chromium, copper, mercury, zinc, alpha-chlordane, 4,4'-DDE, tPCBs), the risk levels rise slightly as sediment concentrations rise from 25$^{th}$ to 75$^{th}$ percentile, with a large increase as the 100$^{th}$ percentile is reached. Comparison of the median (50$^{th}$ percentile) HQ indicates that little or slight risk (HQ < 2) would be predicted for most COPCs under average sediment and surface water concentrations with the exception of arsenic.

Since the COPC exposure includes contributions from both sediment and surface water, the potential relative contribution of each media was investigated for selected COPCs. It will be recalled that the surface water concentrations of metals were derived from the NJMC/MERI database from a number of previous water quality surveys and the organic surface water concentrations were predicted based on dissociation from sediment. Since this is a large source of uncertainty, the amount of risk predicted if this source was removed by setting the surface water concentrations used in modeling to zero. The resulting HQ values absent the influence of surface water concentrations are shown in Table 5-4. A side-by-side comparison of risk with and without the water contribution (the calculated percentile concentrations) is provided for the heron and mink in Table 5-5. Figures 5-12 through 5-15 display the same comparison for the two piscivores for arsenic, lead, zinc, and tPCBs, respectively.

It can be seen that for lead, there is little influence of the surface water COPCs, since the majority of the risk is due to the sediment or invertebrate ingestion (Figure 5-13). For the other three, the influence of the water has a significant effect on risk, particularly at the 100$^{th}$ percentile (maximum) concentrations. For some COPCs (arsenic (Figure 5-12); tPCBs (Figure 5-15)), greater effect is shown in the HQ for the mammalian receptor (mink), whereas, for zinc, the avian receptor (heron) shows greater effect (Figure 5-14).

These comparisons indicate that the wildlife risk curves are very sensitive to the surface water concentration for some COPCs. As discussed earlier (see Section 4.5), the total fraction was used for the EPC for the surface water metals. This suggests that additional sampling of the surface water in the Meadowlands wetlands of interest using the dissolved fraction and/or with "clean" metal sampling methods is likely to refine and potentially, significantly reduce the estimated risk for the wildlife receptors. An alternative to this re-sampling would be to use the median surface water values as a background risk for all receptors independent of sediment concentrations.

# 6.0 RELATIONSHIPS BETWEEN BENTHIC COMMUNITY, CONTAMINANTS, AND TOXICITY

The initial scope for the project planned that a Sediment Quality Triad analysis be performed using the biology, toxicity, and sediment chemistry data. The Sediment Quality Triad concept was introduced by Long and Chapman (1985), Chapman (1986), and Chapman et al. (1986) as a means to link biological degradation to chemical contamination. Typically, the triad consists of using the results of sediment chemistry, toxicity testing, and benthic community structure, although other parameters have also been used. The triad may be used to determine the extent of pollution-induced degradation, to identify problem areas of sediment contamination, and possibly to make predictions about where degradation might occur based on levels of existing contamination.

For use of this method it is desirable that the three data components be collected concurrently to avoid introduction of error associated with temporal differences. The scope of this project, however, was limited to collection of benthic samples at only two of the five marshes for which sediment chemistry and toxicity data were collected. Historic benthic community analysis was available for the other three marshes, but was not paired with complete sediment chemistry and toxicity testing data. Given the available budget for this initial project, it was decided that historic benthic data would be used where possible to support a Sediment Quality Triad analysis.

The key element in applying the Sediment Quality Triad is the availability of data for appropriate reference stations and test stations that reflect a wide range of perceived conditions (ENSR, 2001). In this case reference stations should reflect unimpaired baseline conditions. As noted in previous sections, while it was initially anticipated that Sawmill Creek would serve as an appropriate reference condition, the sediment chemistry and toxicity testing results indicate that this marsh does not represent unimpaired conditions. Because the three types of data were not available from an appropriate reference station, it is not possible to apply the classic Triad analysis to the current data set from the Meadowlands. Moreover, due to the sediment quality and toxicity results from Sawmill Creek, standardization of other location's results to the "reference" location would likely led to erroneous conclusions. Therefore, in lieu of that analysis, a brief qualitative summary comparing the benthic infaunal community parameters with toxicity and sediment chemistry results follows.

## 6.1 Benthic Community and Sediment Toxicity

Infaunal samples collected at Riverbend Marsh in September 2003 showed extremely high densities of infauna as well as higher diversity, evenness, and species number than found in most historical samples collected at Secaucus H.S., Oritani, and Mill Creek marshes. Nevertheless, survival for both the amphipod *Hyalella azteca* (28 days at 10 ppt) and the polychaete *Neanthes arenaceodentata* (10 days at 25 ppt), in sediments from RBM-01 were significantly ($\alpha$=0.05) lower than survival in the negative control sediment (Table 3-8). In a comparison of the two stations sampled at Riverbend, survival at RBM-01 was less than at RBM-02 for both the H. *azteca* (45% less) and N. *arenaceodentata* (58% less) test organisms. Infaunal samples collected at the individual stations did not differ in diversity, evenness, or number of taxa, although the density and total number of organisms was higher at the RBM-01 location than at RBM-02.

Infaunal samples collected at Kearny Marsh in September 2003 showed low densities of infauna compared to Riverbend Marsh. However, the density of infauna was greater than that observed for Secaucus High School, Oritani and Mill Creek Marshes. The diversity, evenness, and species number was within the range observed in historical samples collected at Oritani, Secaucus High School, and Mill Creek marshes. The 28-day survival for each of the test sediment sites was not significantly different from the negative control. Likewise while the 10-day test survival was lower than the negative control for all samples, only KFM-01 was significantly different from the control.

Growth data for the 28-day and 10-day tests showed mixed results. For the 28-day test mean weight (normalized to number of organisms at test initiation) was greater than the negative control for all test sediments except RBM-01. Weights for the test sediments were consistently lower than the negative control for all sites except RBM-01, ORM-02, and SHS-02 for the 10-day test. Sediments from SAW-01 and RBM-02 exhibited significantly lower growth than the negative control.

## 6.2 Benthic Community and Sediment Chemistry

A cumulative Ecological Effect Quotient (EEQ) was calculated to serve as a means of comparing benthic diversity with sediment chemistry results. The EEQ is calculated by dividing the observed maximum detected concentration for each COPC by the corresponding screening criteria (i.e., LEL) for each contaminant, and summing the ratios for each sampling location. Using the EEQ allows for comparisons to a single number rather than a large number of individual contaminants. Recognizing that different contaminant classes may pose different risk, separate EEQs were also calculated for metals and PCBs. Figure 6-1 illustrates the toxicity testing results (survival and growth) againts the cumulative EEQ. No clear relationship between contaminant levels (as expressed by EEQ) and measured toxicity is evident.

Figures 6-2 and 6-3 illustrate the EEQs and benthic community parameters. Historic data were included in this analysis where all chemical contaminants measured were consistent with those in the 2003 study and for which paired benthic data was available. Again there is not obvious relationship between the benthic community measures and sediment contaminant concentrations.

## 6.3 Analysis Conclusions

No clear and definitive relationships emerge between sediment toxicity or sediment chemistry and benthic community parameters. At some locations the results appear to support the premise of lower diversity, evenness and density is evident where sediment contaminant levels are elevated. However, there are an equal number of locations that did not appear to follow this pattern. A variety of factors may contribute to these conclusions. For example, hydrology, water chemistry and physical habitat may be more significant factors in defining benthic community structure and composition. Past studies as well as the current project sampling effort have sampled multiple habitat types (e.g., mudflats, channels, open water, high marsh, marsh plain, etc.) which introduce inherent variability in community composition because of habitat type alone. Hence differences in physical habitat type, habitat structure/quality or water chemistry not addressed under this project may impact benthic communities, and would not be reflected in the toxicity tests or sediment chemistry performed. Alternatively, observed toxicity may be the result of compounds not measured. However, more likely, the use of EEQs calculated using maximum concentration and Lowest Effects Level (LEL) and/or differences between total concentration and bioavailable concentrations may overestimate potential risk and mask any true relationships.

# 7.0 UNCERTAINTY AND LIMITATIONS

As noted in Section 2.0, the work completed for this project was completed in accordance with a Quality Assurance Project Plan (QAPP). This section presents a discussion of key issues relative to uncertainty and limitations of the data and conclusions.

Sediment Sampling and Field Measurements

- Sediment sampling and handling and field measurements were completed in accordance with the QAPP, and results were in conformance with the quality control requirements.
- GPS coordinates recorded for Kearny Marsh sampling location KFM-01 deviated from the actual field location (as indicated by field sampling personnel). The location of this sampling station was moved to reflect the actual approximate sampling location.

Benthic Sample Analysis

- Benthic sample analysis was completed in accordance with the QAPP, and results were in conformance with the quality control requirements.

Analytical Testing

- Sediment sample chemical analysis was completed in accordance with the QAPP, and results were in conformance with the data quality objectives.

Toxicity Testing

- Toxicity testing was completed in accordance with the QAPP and associated SOPs with some exceptions.
  - *Hyalella azteca* were 13-days old at initiation. The B-H Laboratories standard operating procedure for the *Hyalella azteca* test (ATL/SEDHACHR.050) states that organisms will be 7-8 days old for a 28-day test. Organism age is within range for a 10-day test (7-14 days); there is no EPA-mandated age range for a 28-day test. Since all organisms were within the same age range, this deviation from the SOP probably did not affect test outcome.

  - *Chironomus tentans* was 14-days old at test initiation. The report states that the organisms were 2$^{nd}$ to 3$^{rd}$ instar, which is the target metamorphic stage of *C. tentans* at test initiation. However, the USEPA sediment testing manual (page 43) states that organisms 12.5 days old or older are probably 4$^{th}$ instar. Without head capsule measurements, or another acceptable method of determining instar, it cannot be determined if organisms were of the correct stage at initiation. Older organisms have been found to be less sensitive to some toxicants. This could be a deviation that might reduce the sensitivity of test organisms to the test sediment.

- Dry weight for amphipods should be determined at 60 to 90°C, rather than 100°C. However, drying at 100°C is more typical for other toxicity tests and since all organisms were dried at the same temperature, this deviation should not have any effect on test outcome.

- Chironomid dry weights should also be determined at 60 to 90°C. However, as long as all organisms were dried at the same temperature, this deviation should not impact test results.

■ Sediments were screened to remove large material (e.g., large pebbles, stones, twigs and plant clumps) and homogenized prior to subsampling for toxicity test replicates. However, as wetland sediments likely include un-decomposed or partially decomposed plant material, it is possible that contaminants may not be equally distributed in the whole sediment sample. This may result in variability among the replicates.

Screening Level Ecological Risk Assessment (SLERA) and Standard Risk Curves

Uncertainty associated with the SLERA is described in detail in Section 4.4.2, including the implications of various assumptions and sources of uncertainty that are summarized below:

■ The environmental media (surface water, sediment) in the Meadowlands wetlands of interest were considered sufficiently but not well-characterized for purposes of the SLERA. Data was available for the sediment in the selected wetlands of interest and were the basis of Exposure Point Concentrations (EPCs) for the SLERA.

■ Surface water data were taken directly from the NJMC/MERI environmental database and are not co-located in time and spatial coordinates with the 2003 sediment sampling.

■ EPCs estimation assumes that the analytical chemistry results accurately reflect the exposure conditions within the wetlands of interest.

■ Screening against benthic and wetland vegetation benchmarks was used to select COPCs for further analysis. For purposes of this SLERA, the COPCs evaluated were limited to 11 contaminants including seven heavy metals (arsenic, cadmium, chromium, copper, lead, mercury, zinc), two pesticides (alpha-chlordane, 4,4'-DDE), tPCBs, and tPAHs.

■ Selection of COPCs was based, in part, on comparison of a maximum value to a conservative criterion or benchmark (often based on protection of a sensitive species not found on-site).

- Use of maximum values provides for a conservative "worst-case" estimate of potential ecological risk, which may not accurately portray actual ecological risks expected by on-site species.
- The aquatic risk assessment used toxicity values based on chronic effects to analyze the potential for ecological risk to estuarine communities.
- The sensitivity of different species to the contaminants may result in underestimates or overestimates of potential acute and chronic toxicity for many aquatic organisms.
- The use of total metals for estimating exposure in surface water samples is likely to be a conservative estimate of metal bioavailability.
- For many some COPCs, appropriate media-specific screening values were not available and these contaminants were conservatively carried forward in the SLERA (e.g., phytotoxicity benchmarks for organics).
- The surface water EPC was based on pooled data taken from individual wetland studies. Both the number and type (total vs. dissolved) of measured water quality parameters and their detection limits varied between efforts.
- Potential risks evaluated included direct contact to environmental media (surface water, sediments) and bioaccumulative effects (food items). It was assumed that other complete exposure pathways do not significantly contribute to risk.
- The potential exposure of several semi-aquatic receptors would be significantly reduced from that used in the food web modeling (i.e., that a receptor would spend 100% of their time under maximum exposure conditions) due to the large sizes of their home ranges.
- The bioavailability of several sediment contaminants, especially metals, may be overestimated. It has been shown that acid volatile sulfides (AVS) have a high binding capacity for divalent metals (e.g., Cr, Cu, Pb, Ni, Zn) in sediments, thereby reducing or eliminating the bioavailability to sediment-associated receptors.
- The dose-response values used for the several species were extrapolated from data on similar species because no direct dose-response information was available for the representative species. This extrapolation involved conservative assumptions for several factors including allometric scaling, application of safety factors, and use of NOAELs; thus, it is likely that the scaled dose-response values chosen will result in overestimates of the potential for adverse effects.
- Using the lowest reported chronic toxicity data point as a benchmark concentration (as was done in this assessment with a NOAEL-based TRV) is a very conservative approach, especially when there is a wide range in reported toxicity values for the relevant species.

- The estimate of uptake and/or bioaccumulation from environmental media into plants and wildlife is another source of uncertainty. Site specific tissue data was not available. In the absence of site-specific tissue data, uptake factors (UFs) or models are used to estimate tissue concentrations.
- Extrapolation of the potential for community, population, or ecosystem effects from the examination of one or more representative species assumes that potential effects on one representative species are consistent with the effects on similar species and representative of the potential for effects on the particular ecosystem being investigated.
- The representative species were chosen to evaluate potentially high exposure to environmental media at the site and may overestimate the potential for adverse ecological effects to other species.
- Ecological risk typically considers potential risk to communities or populations. If adverse effects are predicted for an individual, it does not necessarily mean that the community, population or ecosystem will be similarly affected.
- The standard risk curves were developed using the food web models. The uncertainties associated with these models (described above), are carried through to the standard risk curves.
- The standard risk curves are very sensitive to the surface water concentration for some COPCs. Surface water sampling was beyond the scope of this project. Therefore, historic water quality data was pooled and maximum concentrations were selected for us in food web modeling.

Analysis of Relationships Between Benthic Community, Contaminants, and Toxicity

- The analysis of relationships between benthic community composition, contaminant concentrations and whole sediment toxicity incorporates the uncertainty (described above) associated with each of these contributing elements.
- The data necessary to complete a Sediment Quality Triad analysis was not available, as no adequate reference samples were available.
- The weight of evidence analysis relies in part on historic data. For example, historic benthic data was used for all locations except Riverbend and Kearny Marshes. The actual location of sample collection may vary from those used for sediment chemistry and toxicity testing as part of this study. Likewise, use of this data assumes that the benthic community observed several years ago is representative of that which might be observed at the sediment sampling locations used in this study.

# 8.0 CONCLUSIONS

The following provides a brief summary of key conclusions drawn from this project:

Benthic Community

- The benthic community found at Riverbend includes faunal elements that are typical of high estuarine, low salinity marshes throughout the eastern United States. There is no evidence of pollution or stress-related indicator species such as the polychaetes *Capitella capitata* or, possibly, *Polydora cornuta.*
- The benthic community at Kearny Marsh appears to be impoverished and is possibly impacted by the high contaminant loads. This statement is tempered, however, by the fact that the species composition of the insects was not determined.
- More than 55 taxa have been identified from the five marshes for which benthic data are available. All of these taxa observed during the current Project and past studies are commonly found in estuarine systems. The most commonly represented groups include, 12 insects, 12 polychaetes, 8 amphipods, 5 isopods, 4 gastropods, and 3 bivalves.
- Density, diversity, and evenness varied considerably within the wetlands sampled as part of past studies (Mill Creek, Oritani, and Secaucus High School marshes) with an order of magnitude difference in density between locations in a given wetland.
- Previous studies included sampling of mudflats, tidal creeks, marsh plain, ponded/open water, channels, high marsh, and mosquito ditches. Samples collected as part of this study were primarily located in open water, mosquito ditches, and/or channels.

Toxicity Testing

- Toxicity testing results were mixed. None of the test sediments produced 100% mortality. As might be expected, there were differences in survival and growth between the 28-day and 10-day test results. Survival and growth were not always similar (e.g., diminished survival and growth).
- The 28-day mean survival rate for SAW-01, RBM-01, and SHS-01 were less than and significantly different than those for the negative control. RBM-01 exhibited greater than 50% mortality for the 28-day test, while greater than 50% mortality was reported for ORM-02 and SHS-01 in the 10-day tests.
- More than 25% mortality was observed for SAW-01, ORM-01, RBM-02, and SHS-01 during the 28-day tests and for samples from ORM-01, RBM-02, and SHS-02 for the 10-day test. However the survival in RBM-02 and SHS-02 were not significantly different from the control.

- All samples from Kearny Marsh shown significantly lower growth than the control for the 10-day test. This same pattern was not evident in the 28-day test. Survival was not significantly different from the control for any of the samples in the 28-day test, but was significantly different in KFM-01 for the 10-day test.
- SHS-02 and SHS-03 sediments showed no significant difference in survival or growth from the controls in either the 28-day or 10-day tests.

Screening Level Ecological Risk Assessment (SLERA)

- Evaluation of sediment quality in the five wetlands of interest indicated widespread exceedance of benthos-based sediment screening values for metals in all wetlands, with the exception of chromium and iron in Kearny and Oritani Marshes. Several of the metals are also known to bioaccumulate (cadmium, chromium, mercury).
- The pesticides 4,4'-DDE and alpha-chlordane were selected as COPCs since their maximum concentrations provided the greatest exceedances among detected pesticides (e.g., DDT and related fractions). The pesticide 4,4'-DDE is distributed widely throughout the wetlands, whereas alpha-chlordane was detected in Kearny Marsh.
- PAHs were selected as they are a widely distributed contaminant in the wetlands and many of the individual PAH fractions were above the screening level, as was the cumulative total PAHs (tPAHs).
- PCBs were detected, primarily as Aroclor 1248, 1254, and 1260, and were retained as a COPC due to concern regarding their levels and inherent bioaccumulative properties and potential adverse effects on higher wildlife.
- Evaluation of sediment quality in the five wetlands of interest indicated widespread exceedance of phytotoxicity-based sediment screening values. Organic contaminants do not exceed the available benchmarks for tPAHs and tPCBs. No phytotoxicity values were available for the pesticides. Comparison of the metals to available phytotoxicity screening benchmark values indicated widespread exceedances for the metals with the exceptions of cadmium and iron.
- All assessment endpoints were evaluated using conservative assumptions in the selection of contaminant concentration, exposure terms, screening benchmarks, and ecological effects. Under this conservative screening approach and using conservative assumptions for food web modeling, potential ecological risk was indicated for the four assessment endpoints. These results indicate that further ecological risk investigation may be warranted.
- Due to these conservative maximum exposure assumptions, there is a great deal of uncertainty associated with the risk estimates. The sources and potential influences of this uncertainty were described and the results of this conservative screen were further evaluated to provide a better interpretation of the potential risk at the Site. It is likely that alternative assumptions or using site-specific factors may lead to a significantly lesser degree of potential risk.

- Four representative species were evaluated using a maximum Hazard Quotient (HQ). Food web models indicate potential concern regarding maximum exposure of all COPCs except chromium, alpha-chlordane, 4,4'-DDE, and tPAHs for the mink, and all COPCs except arsenic, alpha-chlordane, and tPAHs for the heron. The results for muskrat indicate potential concern regarding exposure of arsenic (HQ = 2.55) and mercury (HQ = 1.47), however the magnitude of the HQ suggests only a slight potential for risk. The results of the food web models for mallard indicate potential concern regarding exposure of mercury (HQ = 1.47), but not other COPCs.

<u>Wildlife Risk Curves</u>

- The wildlife risk curves indicate that the potential risk associated with different COPCs differs significantly in magnitude and pattern of risk with increasing sediment concentrations and that these patterns also differ depending on the receptor/trophic pathway involved. In some cases the level of estimated risk is so low for some receptor-COPC combinations that it is near zero.
- In general, greater risk are predicted for the piscivores (heron, mink) than for the herbivores (mallard, muskrat) over the majority of the COPCs, with the mink having the highest HQs and the mallard the lowest.
- Higher risks are associated with the metals than with the organic COPCs.
- For most COPCs (cadmium, chromium, copper, mercury, zinc, alpha-chlordane, 4,4'-DDE, tPCBs), the risk levels rise slightly as sediment concentrations rise from 25$^{th}$ to 75$^{th}$ percentile, with a large increase as the 100$^{th}$ percentile is reached.
- Comparison of the median (50$^{th}$ percentile) HQ indicates that little or slight risk (HQ < 2) would be predicted for most COPCs under average sediment and surface water concentrations with the exception of arsenic.

<u>Relationships Between Benthic Community, Contaminants, and Toxicity</u>

- Sufficient appropriate data was not available to support a classic Sediment Quality Triad analysis. Therefore a qualitative analysis was provided.
- No clear and definitive relationships emerge between sediment toxicity or sediment chemistry and benthic community parameters. The lack of obvious relationship may be caused by a number of factors including physical habitat, water chemistry, potential overestimation of potential risk from sediment chemistry, and differences between the sensitivity of test organisms and indigenous populations to sediment contaminants.

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