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APPENDIX D

ECOLOGICAL RISK ASSESSMENT (NOAA)
EVALUATION OF RAYMARK SUPERFUND DATA FOR PRG DEVELOPMENT (SAIC)
EVALUATION OF ECOLOGICAL RISK TO AVIAN AND MAMMALIAN RECEPTORS IN THE VICINITY OF UPPER AND MIDDLE FERRY CREEK

RAYMARK INDUSTRIES, INC.

PHASE II
ECOLOGICAL RISK ASSESSMENT

Final

Prepared for:	U.S. Environmental Protection Agency Region I JFK Federal Building Boston, Massachusetts 02203
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Questions regarding this risk assessment may be directed to Michael Buchman (NOAA) or to Region 1, U.S. Environmental Protection Agency.

˙XECUTIVE SUMMARY

This report presents results of an ecological risk assessment for environmental receptors associated with Ferry Creek and the Housatonic River in Stratford, Connecticut. Ferry Creek historically received wastewater discharges from the Raymark Industries, Inc. (Raymark) facility located at 75 East Main Street in Stratford, Connecticut. Raymark discharged waste through an underground culvert which drained a series of sludge-settling lagoons located at the facility. This culvert entered the upper reaches of Ferry Creek. In addition, sludge containing hazardous substances was periodically removed from the lagoons and used as fill material at various locations throughout Stratford for many years. Raymark waste has been found in and adjacent to wetlands, Ferry Creek, and the Housatonic River.

This Ecological Risk Assessment (ERA) addresses the risk to ecological receptors from hazardous substances originating from Raymark that were released to Ferry Creek, portions of the Housatonic River, and associated wetlands. The primary ecological receptors considered as endpoints are either aquatic biota or avian species that are linked to the aquatic habitat through the food chain.

Based upon a preliminary screening risk assessment, the following compounds were chosen as initial Contaminants of Concern (CoCs):

arsenic	polychlorodibenzo-p-dioxins (PCDDs)
cadmium	
chromium	polychlorodibenzo-p-furans (PCDFs)
copper	
lead	polynuclear aromatic hydrocarbon (PAHs)
mercury	
nickel	polychlorinated biphenyls (PCBs)
silver	
zinc	dichloro-diphenyl-trichloro-ethane (DDT)

Four areas were identified in the screening level ERP (SLERA), based upon known waste-disposal patterns and previously collected data, as potentially at risk due to exposure to site-related contaminants:

- upper reaches of Ferry Creek,
- lower reaches of the creek,
- Housatonic River at the mouth of the creek, and
- wetlands next to the Housatonic River near the Housatonic Boat Club, to the south.

These areas include four distinct ecological communities:

- *Spartina*-dominated estuarine wetland,
- *Phragmites*-dominated freshwater wetland
- tidally-influenced stream system with a fluctuating salinity gradient,
- tidally-dominated saline stream/river system.

The following species were selected as ecological receptor species of concern in the Conceptual Model in the SLERA: benthic infauna, blue crab, American oyster, striped bass, black-crowned night heron, and the Atlantic piping plover. These selections were based on the preliminary exposure estimates and risk calculations presented in the SLERA, including a calculation of hazard quotients (HQs) using maximum likely exposure concentrations and reference toxicity values (RTVs). This ERA was conducted based upon the conceptual model of potential ecological risks at the site identified in the SLERA. Four assessment endpoints were developed for evaluation in the ERA based upon factors including ecological relevance, susceptibility to stressors at the site, and representation of management goals

- Survival, growth, reproduction, and appropriate indigenous benthic community (both infauna and epibenthic) composition in Ferry Creek, the Housatonic River near the mouth of Ferry Creek, and the wetlands associated with those areas;
- Survival, growth, and reproduction of oysters in the seed beds in the Housatonic River adjacent to the mouth of Ferry Creek;
- Protection of fish species from adverse reproductive effects and mortality; and,
- Protection of avian species foraging in the area from adverse growth and reproductive effects and mortality.

Measurement endpoints were chosen as the means by which these four assessment endpoints would be evaluated. The following measurement endpoints were chosen:

- Concurrent analysis of bulk sediment chemistry, toxicity to amphipods exposed to bulk sediments, and evaluation of the benthic macro-invertebrate community (i.e., sediment triad);
- Toxicity to oyster larvae exposed to bulk sediments;
- Analysis of fish tissue body burdens of CoCs for comparison with benchmark values; and,
- Evaluation of estimated daily dosage of CoCs to the heron and blackbird, modeled from intakes of fish, fiddler crab, mummichog, sediment, and water for comparison with benchmark values (i.e., a food-web model).

A field-sampling plan was developed to evaluate these measurement endpoints. Due to finite resources, tradeoffs were made in allocating the level of field and laboratory effort among the measurement endpoints, which were reflected in the field sampling design.

Three field-sampling areas were chosen to represent the study area: Upper Ferry Creek (creek and wetland habitats); Lower Ferry Creek; and the wetlands associated with the Housatonic River, near the Housatonic Boat Club. A reference area was chosen in a large wetland area adjacent to Milford Point, on the far side of the river opposite the mouth of Ferry Creek. Because it was known from previous sampling conducted under the remedial investigation (RI) that contaminant concentrations in the study-area sediments were heterogeneous, station locations were chosen within each area to represent a range of contaminant concentrations. Reference stations were selected in an attempt to match habitat type, salinity, and grain size.

The study was designed to optimize the data collected with the finite resources available, and the possible conclusions are those which can be made only within the limitations of the scope of the study conducted.

Results of this ecological risk assessment indicate that the following conclusions can be drawn from the sampling results and evaluation of information available:

- The benthic community assemblages are divided into four groupings. The reference stations form a group with the highest abundance and greatest diversity. Next, the two Upper Creek stations (SD13 and SD20) form a second group which were the most impacted and where the benthic community was dominated by only one or two species. The boat club station (HB23) and one Lower Ferry Creek station (SD19) form a third, intermediate group. This group had depressed diversity and was dominated by only three to four species. The other Lower Ferry Creek station (SD07) appeared to group with the reference station due to the number of species present. However, samples from this station exhibited depressed diversity and were dominated by the polychaete worm *Capitella*. This station had lower richness and evenness of species than the reference stations. The seemingly high number of species present at this station was due to the rare occurrence of only one or two individuals of a given species in only one to two of the four grabs taken. This illustrates that a simple count of species is a deceptive measure of diversity.

  Clearly, adverse impacts to the benthic community are observable within the entire site area. The most significant alterations of the benthic community occur within the Upper Ferry Creek area.

- Risk to the benthic community was also indicated by results from bulk-sediment toxicity tests. The amphipod bulk-sediment toxicity test identified three samples as "toxic"—those taken from the two upper creek stations, SD13 and SD21, plus the one from the lower creek station, SD07. When samples from these areas were compared against one another, both the lower and upper creek area samples exhibited lower survival than either the reference or boat club samples.

- Comparisons of sediment chemistry with amphipod mortality suggest that total PCBs, dioxins, Cu, and Pb may be causal agents. Although total PAH concentrations apparently did not contribute to lethality, avoidance of test samples appears to be related to total PAH content.

- The oyster larvae toxicity test was another approach used to evaluate risk to the benthic community as well as to the oysters themselves. Samples from only three site-related stations and one reference were tested. This test identified two samples as "toxic"—the sample taken from the boat club wetlands station, HB23, and the one taken from the upper creek station, SD13. Samples from these stations had higher incidence of abnormal development and mortality than in the reference sample. The mean response observed from the three site-related samples, as a group, indicated diminished viability.

- Overall, combined mortality (abnormality plus mortality) was predicted quite accurately by sediment chemistry Hazard Quotients. Adverse responses correlated highly with Cu and Pb (which are apparently covariates), total PCBs, total DDT, and dioxins. Relative to other stations, the sample from SD13 contained some of the highest concentrations of Cu, PCBs, DDTs, total PAHs, and dioxins.

- Potential impacts to fish were assessed primarily using the HQ approach with comparison to Maximum Allowable Tissue Concentrations (MATCs). Body burdens of CoCs measured in fish in the study area were compared with these toxicological benchmarks. The two fish species assessed were mummichog collected during this assessment and white perch collected in Selby and Frash ponds in October 1993. The white perch tissue was not analyzed for all CoCs; particularly, dioxins.

  The evaluation of this endpoint was limited not only by the lack of dioxin data for white perch, but also the lack of MATCs in the literature. MATCs could be located for only seven of the CoCs: PCBs, DDT+dichloro-diphenyl--ethane (DDE), mercury (Hg), cadmium (Cd), total PAHs, polychlorodibenzo-p-dioxins (PCDDs), and polychlorodibenzo-p-furans (PCDFs). A comparison of the maximum body burdens among site-related mummichog and white perch revealed only three Hazard Quotients (HQ) that exceeded 1: DDT+DDE in white perch, plus Cd and PAHs in mummichog. The Cd HQ was 4.38 for fish collected in Upper Ferry Creek, upstream of station SD13. Due to data gaps, the risk assessment for predatory fish, such as white perch, should be considered incomplete.

- Risk to fish was also assessed indirectly by comparing concentrations of CoCs in water to appropriate chronic ambient water quality criteria (AWQC). AWQC (US EPA 1993) were exceeded for Cu, chromium (Cr), Pb, Hg, silver (Ag), zinc (Zn), and total PCBs. Based on a comparison of maximum concentrations of CoCs in the water and toxicological data for fish, concentrations of Cu, Pb, Hg, and Zn observed in surface-water samples may cause adverse chronic effects.

- To assess avian risk, the dietary dosage of CoCs was estimated using a food-web model. The results from this exposure model were then compared by HQ calculation with Reference Toxicity Values (RTVs). Upper 95% confidence intervals or maximum observed values were used for estimating doses. Data from Upper and Lower Ferry Creek were combined and treated as one exposure area due to the foraging habits of the species.

  Dietary doses of chromium and lead calculated in the exposure model for black-crowned night heron were the only CoCs associated with site-related samples that exceeded their respective RTVs. Sediment concentrations of lead contributed most of the dose of this element in Ferry Creek, while crab ingestion was the major route for exposure at the boat club wetlands. HQs for chromium were the largest for any CoC, up to 3.5. The sum of HQs for chlorinated CoCs (i.e., dioxins, PCBs, and DDTs) was less than 1. Given the conservative nature of the assessment and the degree of exceedance of RTVs, CoCs apparently do not pose a substantial risk to these birds.

  The results of food-web modeling for avian species indicate that the exposure scenario modeled for red-winged blackbird does not pose a risk to this species because no HQ exceeded 1.

To further relate the results of the observations of biological impacts to the chemical concentration in sediment, HQs were calculated using published sediment-quality guideline concentrations of CoCs observed in sediment. Those samples identified as toxic by various biological measurement endpoints were among those with the highest HQs in this comparison.

These samples also had greater mean concentrations of numerous CoCs than the samples that did not exhibit toxic response. The CoCs that were most elevated with respect to either the sediment guideline or reference-area concentrations were Cu, Pb, PCBs, and dioxins/furans. Additionally, the responses observed from the various biological endpoints were in general agreement with one another. This weight of evidence confirms that the biological responses observed are the result of general contaminant concentrations in sediment.

The findings of the ERA indicate that there is an unacceptable risk to the benthic community, with potential for indirect impacts to those organisms dependent on a healthy benthos.

1.1 UNCERTAINTIES

It should be noted that there are uncertainties surrounding the conclusions made in this ERA that are associated with constraints both of the specific study design and the state-of-the-art of risk assessment. Therefore, certain conclusions must be interpreted in the context of their associated uncertainties. The greatest number of factors which affect the uncertainty of the risk assessment are associated with the food-web model for CoC exposure to avian receptors. The reader is cautioned to review the full Uncertainty Assessment (Section 9.0) in this report for discussion of the factors influencing uncertainty.

2.0 INTRODUCTION

This ERA summarizes the findings and conclusions of an investigation on the effects of contaminants from Raymark Industries on the biota of Ferry Creek and the Housatonic River near the mouth of the creek, plus associated wetlands, in Stratford, Connecticut **(Figure 2-1)**. This report has been prepared based upon investigations and interpretations made by the National Oceanic and Atmospheric Administration (NOAA) at the request of Region I of the U.S. Environmental Protection Agency (US EPA). NOAA, in its role as a trustee for certain natural resources, has developed an expertise in ecological evaluations. This expertise is made available to EPA through a technical support interagency agreement.

2.1 BACKGROUND

Ferry Creek historically received wastewater discharge over many years from Raymark Industries, Inc. (referred to as 'Raymark' or 'the facility'). Raymark manufactured automotive friction material from 1919 to 1989 at their 75 East Main Street location in Stratford, Connecticut. Materials used in the processes at the facility contained asbestos, metals, phenol-formaldehyde resins, and various adhesives. Wastes generated included asbestos, Pb solids, acids, caustics, and general wastewater. Discharges were released primarily through an underground culvert draining a series of sludge-settling lagoons at the facility. This culvert empties into the upper reaches of Ferry Creek **(Figure 2-2)**. Also, large volumes of sludge containing hazardous substances were removed from the lagoons during the 1970s and early 1980s and used as fill material at various locations throughout Stratford. Raymark waste has been found in wetlands and on soils adjacent to Ferry Creek and the Housatonic River and has migrated into aquatic habitats.

Remedial Investigation (RI) sampling efforts within Ferry Creek, Housatonic River, and associated wetlands have been conducted by the EPA since 1992. **Figure 2-3** indicates the extent of sediment sampling conducted during 1992–1994 for chemical analysis. These sampling results confirmed that contaminants migrated into aquatic habitats and adjacent wetlands. Elevated concentrations of barium (Ba), Cu, Pb, Zn, PCBs, and dioxins have been observed. Areas that were found to have elevated contaminant levels within the area (Figure 2-3) include the following:

- stations at the head of Ferry Creek, near Ferry Boulevard;
- the small inlet in Upper Ferry Creek (near station SD13);
- along the west bank of Lower Ferry Creek, near the side inlet;
- portions of the wetlands near the Housatonic Boat Club (near station HB23); and
- other areas as identified in the RI.

Since the concentrations of contaminants observed during the initial sampling efforts of the RI often exceeded sediment screening concentrations expected to pose some degree of risk to aquatic ecological receptors, a SLERA was conducted to determine the likelihood of adverse ecological impacts due to exposure to each site-related CoC (EVS Environment Consultants, Inc. [EVS] 1995). This assessment was based on conservative and generic assumptions concerning the nature of exposure and risk to ensure a high degree of confidence associated with any findings of negligible risk. This conservative SLERA, however, confirmed a likelihood that some ecological receptors were potentially at risk.

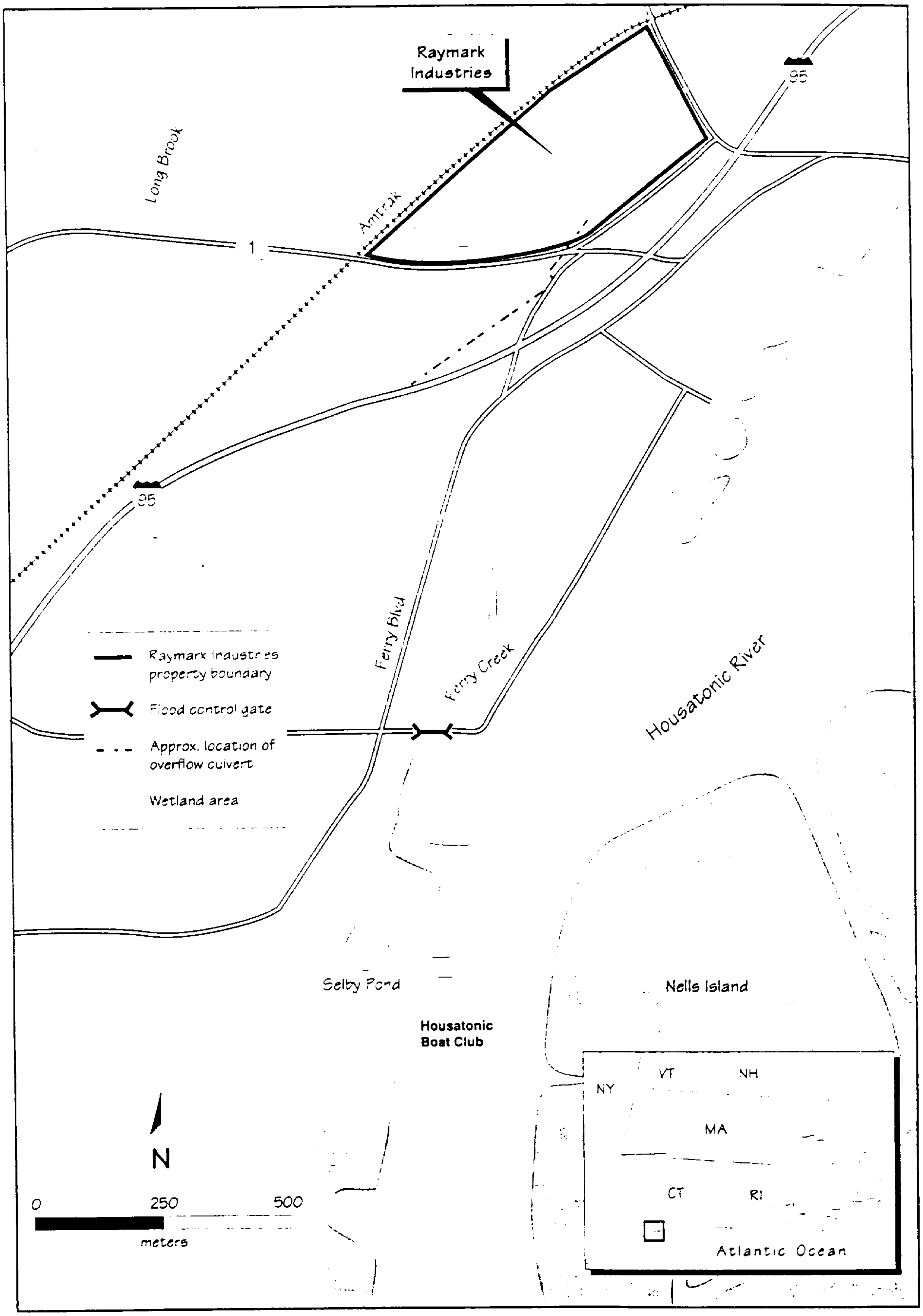

Figure 2-1. Location of Raymark Industries, Ferry Creek, Housatonic River, and adjacent wetlands.

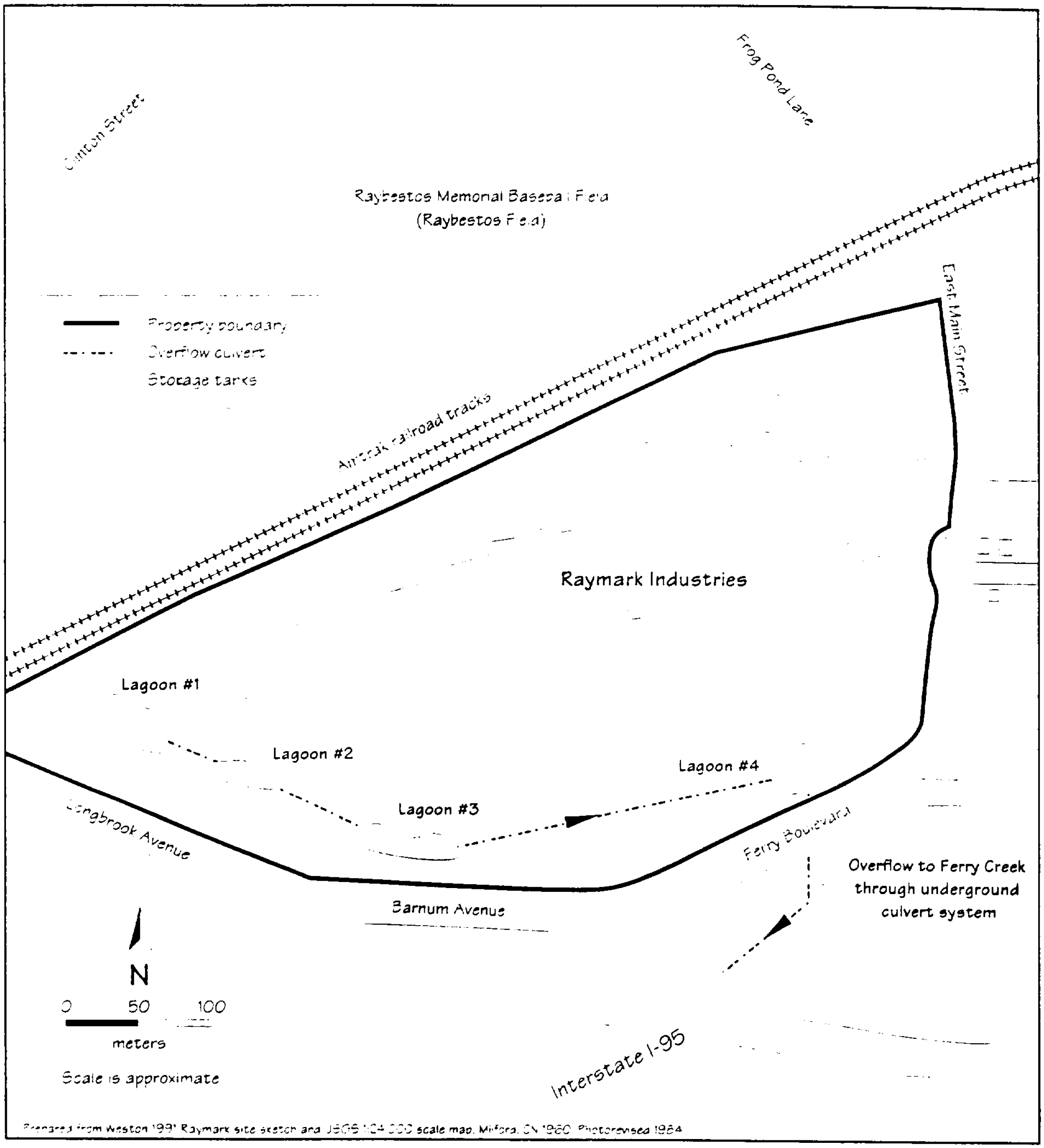

Figure 2-2. Detail of Raymark Industries in Stratford, Connecticut.

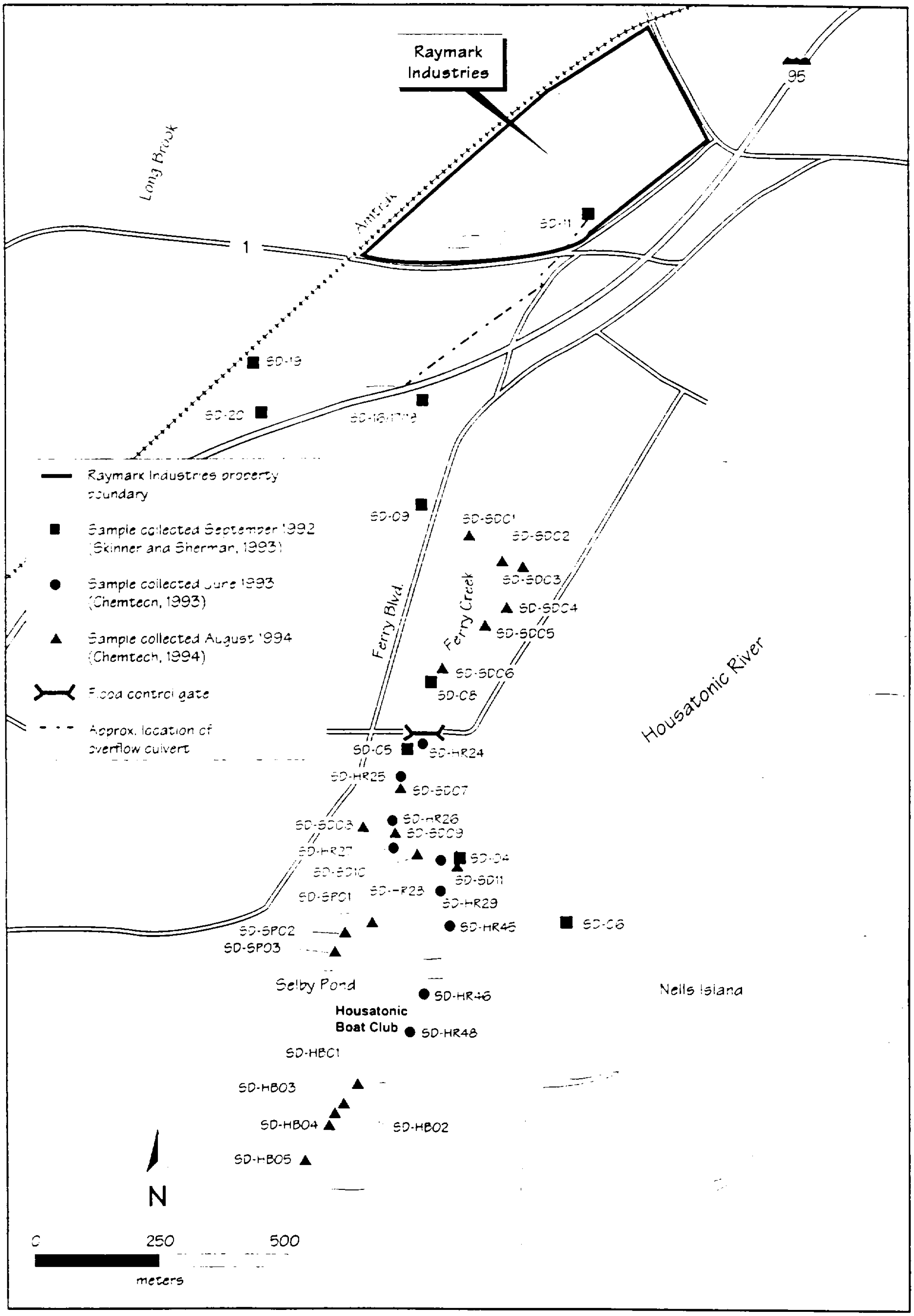

Figure 2-3. Sediment sampling stations downstream from the Raymark site sampled between 1992 and 1994.

2.2 OBJECTIVES

This ERA expands upon the Screening Level Assessment. It addresses risk to ecological receptors from hazardous substances released to Ferry Creek, portions of the Housatonic River, and associated wetlands. Because this assessment focuses on aquatic pathways and exposures, the primary ecological receptors considered are either aquatic biota or avian species that are linked to aquatic habitats through the food chain. This assessment uses site-specific information along with appropriate assumptions to refine estimates of risk made during the Screening Level Assessment. These refinements more accurately reflect site-specific conditions and the associated potential for risk to ecological receptors present within the habitats of concern.

3.0 PROBLEM FORMULATION

The problem-formulation phase of an ERA is the process by which the preliminary hypotheses are generated regarding the potential for ecological effects to occur as the result of exposure to specific stressors. Through a structured process, problem formulation facilitates the development of appropriate assessment endpoints, a conceptual model for the site, and an analysis plan including suitable measurement endpoints. In addition, the CoCs are defined in the problem-formulation stage.

The problem formulation and conceptual site model for this ERA were based largely on the results of the field investigation performed as part of the RI and the SLERA. The conceptual model describes the transport and transformation of CoCs from their release to points of exposure where organisms may come in contact with them. The conceptual model highlights the primary pathways by which contaminants reach environmental receptors and the likely locations and types of these exposures. The conceptual model also discusses the modes of toxicity in the organisms potentially impacted. The development of a conceptual model of the site is iterative, interactive, and concurrent with problem formulation.

3.1 CONCEPTUAL SITE MODEL

The conceptual site model summarizes

- waste source and CoCs;
- transport pathways of CoCs (physical and chemical);
- key habitats and ecological receptors;
- exposure pathways for ecological receptors;
- toxicological information on the CoCs; and
- risk hypotheses.

The overall problem formulation and the conceptual model result in selection of assessment and measurement endpoints for an ERA. Because the risk to the ecosystem cannot be addressed, key components of the system are identified. The viability of these key components essentially generates the assessment endpoints. Measurement endpoints are those parameters or metrics that are related to the assessment endpoints and can be directly measured. These indicators are then directly assessed as surrogates for the assessment endpoints. Ultimately, the conceptual model provides theoretical verification that the measurement endpoints used to evaluate the assessment endpoints are based on appropriate exposure pathways and will provide an adequate estimation of the risks to the ecosystem.

3.2 CONTAMINANTS OF CONCERN (CoC)

3.2.1 Waste Sources

Materials used in the processes at the Raymark facility contained asbestos, metals, phenol-formaldehyde resins, and various adhesives. Wastes generated from production activities included asbestos and Pb solids, acids, caustics, and wastewater. Typically, production wastes were discharged into a series of four unlined lagoons where solids were allowed to settle. The resulting overlying water was discharged from the fourth lagoon to a storm-water culvert leading to Ferry Creek. Before 1970, accumulated asbestos and Pb solids were removed from the lagoons and disposed on the Raymark facility as fill material. During the 1970s and early 1980s, solids were annually removed and disposed at various locations throughout Stratford.

In addition to the lagoons, numerous above- and underground storage tanks on the property had been used for storing raw materials, process wastewater, and fuels. Several leaks and spills from these tanks have been documented and may have contributed to the contamination at the site (Weston 1993). The types of wastes stored in these containers and the sources of the waste materials were not specified.

The primary contaminants found in soil collected from locations on the Raymark facility include metals, PCBs, PAHs, and dioxin/furans. Dioxins are thought to have been a trace contaminant in the cutting oils used at the facility. These were the contaminants evaluated as CoCs during the Screening Level Risk Assessment (EVS 1995).

3.2.2 Selection of CoCs

The concentrations of contaminants in sediment, tissue, and surface water in Ferry Creek, portions of the Housatonic River, and associated wetlands observed during the sampling conducted for the RI were reviewed to select CoCs. Soil and groundwater data were not reviewed for CoCs selection because aquatic pathways and receptors are the focus of this ERA.

Selection of CoCs in the SLERA (EVS 1995) were based on two primary guidelines:

(1) Exceedance of the Effects Range-Low (ERL) concentrations in sediment (Long & Morgan 1992.
(2) Exceedance of the screening toxicity equivalency quotient[†] (TEQs) expressed as equivalents of 2,3,7,8-tetrachloro-dibenzo-p-dioxin (TCDD) in sediment.

When screening guidelines were not available, contaminants were included as CoCs if they were detected in fish or shellfish tissue from historic site samples. No changes were made to the list generated in the SLERA. A full listing of each contaminant is also presented in **Table 3-1**. Brief toxicity profiles for these CoCs are given in **Table 3-2**.

[†] The combined toxic potential of dioxins, furans, and PCBs that could contribute to mixed-function oxidative enzyme mediated toxicity is expressed as the summation (TEQ) of the product of individual isomer concentrations and their toxic equivalence factor (TEF). TEFs are ratios that normalize the toxic response of one isomer to that of 2,3,7,8-TCDD. Only dioxins and furans have been factored into TEQs for this assessment.

Table 3-1. Contaminants of concern evaluated in the Phase-II ERA.

Contaminants of Concern				
Metals and Metalloids	PAH	PCDD and PCDF	Pesticides	PCB
Arsenic	Acenaphthene	penta	DDD	Aroclor 1016
Cadmium	Acenaphthylene	through	DDE	Aroclor 1221
Chromium	Anthracene	hepta	DDT	Aroclor 1232
Copper	Benz(a)anthracene	chloro-		Aroclor 1242
Lead	Benzo(a)pyrene	dioxins		Aroclor 1248
Mercury	Benzo(b)fluoranthene	and		Aroclor 1254
Nickel	Chrysene	furans		Aroclor 1260
Silver	Dibenz(a,h)fluoranthene			Aroclor 1262
Zinc	Fluoranthene			Aroclor 1268
	Fluorene			
	2-Methylnaphthalene			
	Naphthalene			
	Phenanthrene			
	Pyrene			

Table 3-2. General ecotoxicity of selected CoCs.

COC	Toxic Effects
Arsenic (Eisler 1988a, Mance 1987)	• Reduced survival and reproduction impairment in fish and aquatic invertebrates • Reduced survival, physiological dysfunction, carcinogenesis, mutagenesis, and teratogenesis in birds and mammals
Cadmium (Eisler 1985)	• Reduced growth, reduced survival, reproductive impairment, respiratory disruption, and molt inhibition in marine organisms at low ambient concentrations • Avian species comparatively resistant at low doses; reproductive impairment and growth retardation, anemia, and testicular damage at higher doses
Chromium (Eisler 1986a)	• Reduced survival and reproductive impairment in aquatic invertebrates and reduced survival and growth retardation in fish • Avian species relatively resistant; teratogenesis and reduced growth and reduced survival at relatively high, long-term doses
Copper (Mance 1987, ATSDR 1990a)	• mortality and reduced growth in aquatic invertebrates, and mortality and behavioral changes in fish; invertebrates generally more sensitive than fish • Mortality, developmental effects, genotoxic effects, and carcinogenesis in birds and mammals
Lead (Eisler 1988b)	• Reproductive impairment, reduced biomass, and reduced survival in aquatic invertebrates • Anemia, enzyme inhibition, teratogenesis, and reduced growth and survival in fish • Mortality, neurotoxicity, muscular paralysis, inhibition of heme synthesis, kidney and liver damage, and reproductive impairment in birds • Reproductive toxin in mammals, and carcinogenic in some mammals

Table 3-2. continued . . .

COC	Toxic Effects
Mercury (Eisler 1987a, Mance 1987)	• Mortality, reproductive impairment, and neurotoxicity in fish • Mortality, growth retardation, and behavioral effects in aquatic invertebrates • Mortality, neurotoxicity, and teratogenesis in birds • Carcinogenic in some mammals • General effects at low doses in both invertebrates and vertebrates
Nickel (Mance 1987, ATSDR 1993a)	• Mortality and deformity in fish • Mortality, abnormal development, and reduced larval growth in aquatic invertebrates • Mortality; immunological, neurological, developmental, and reproductive effects; genotoxicity and carcinogenesis in birds and mammals
Silver (Mance 1987, ATSDR 1990b)	• Mortality, abnormal development, and reduced growth in aquatic invertebrates • Larval mortality, growth retardation, premature hatch, and deformity in fish • Mortality and neurological effects in birds and mammals
Zinc (Eisler 1993, ATSDR 1992)	• Mortality, abnormal growth and development, reproductive impairment, and reduced larval settlement in aquatic invertebrates • Mortality, growth retardation, teratogenesis, and reproductive impairment in fish • Mortality, immunological, developmental, and reproductive effects; genotoxicity and carcinogenesis in birds and mammals
PCDDs/PCDFs (Eisler 1986b)	• Mortality, growth retardation, and fin necrosis in fish at very low exposure concentrations • Mortality, severe emaciation, loss of appetite, muscular incoordination, tremors, spasms, convulsions, and chick edema disease at very low doses in birds • Reproductive impairment, embryo toxicity, and developmental deformities in birds
PAH (Eisler 1987b)	• Carcinogenic in fish; reproductive impairment and emergence in aquatic invertebrates • Toxicity most pronounced among crustaceans and least pronounced among teleosts • Reduced embryo survival and development in birds • Mutagenic, carcinogenic, and teratogenic in birds and mammals
DDT, DDD, and DDE (Adams et al. 1987, Hose et al. 1989, Smith & Cole 1973, Word et al. 1987)	• Mortality and behavioral effects in aquatic invertebrates • Mortality, reproductive impairment, and teratogenicity in fish and elevated tissue concentrations • Reproductive impairment (i.e., eggshell thinning) in birds
PCB (Giesy 1994, Eisler 1986c)	• Reproductive impairment in fish and aquatic invertebrates • Reproductive, behavioral, mutagenic, carcinogenic, and teratogenic effects in some birds and mammals
Phenol (Clement Assoc. 1985)	• Mortality, reproductive effects, and developmental effects in aquatic species • Physiological effects and organ damage in birds and mammals
Bis(2-ethylhexyl) phthalate (ATSDR 1993b, Ozretich et al. 1983, Mayer & Sanders 1973)	• Mortality, and reproductive and behavioral effects in aquatic species • Mortality; developmental, reproductive, genotoxic, and carcinogenic effects in birds and mammals

3.3 CONTAMINANT TRANSPORT AND EXPOSURE PATHWAYS

Contaminated material associated with the Raymark facility originated from two primary sources: Discharge from waste lagoons, and waste material and sludge used for fill. Wastewater from the lagoons was discharged directly into Ferry Creek. Historical fill operations relocated contaminated material to soils and wetlands throughout the Stratford area, including Ferry Creek and wetlands near the boat club. Pathways from these primary sources have been eliminated by removal and remedial measures including the diversion of overland flow around the lagoons, capping of lagoon 4, the cessation of fill activities, and capping of the entire facility. However, these historical releases, in combination with environmental transport mechanisms, have led to contamination of secondary sources (i.e., receiving media). These secondary sources are primarily aquatic sediments and wetland soils.

Ferry Creek, the Housatonic River, two nearby ponds, and associated wetlands and sediments are locations where these secondary media have been contaminated. Although some processes may decrease bioavailability of contaminants to specific receptors (e.g., volatilization and sorption), other processes (e.g., dissolution and bioaccumulation) can increase the bioavailability of the contaminants. The relative importance of each chemical and physical process is determined by site-specific and chemical-specific conditions. Each of these interactive and competing influences at a given location combine to determine the overall bioavailability of the contaminants present and their potential for adverse biological impact. The geographic scale that can be predicted to have homogeneous conditions of bioavailability and potential risk is determined by the degree of variability in those parameters that control contaminant distribution and bioavailability. In addition, the behavior of receptor species also determines which potential exposure media or pathways may be significant. Together, these processes define certain key pathways for the exposure or uptake of CoCs by ecological receptors. **Figure 3-1** depicts the primary routes of contaminant transport and exposure in the areas of interest.

3.3.1 Exposure Pathways

To determine whether an ecological receptor may be adversely impacted by a CoC, exposure pathways were evaluated for each CoC/species combination. The exposure routes evaluated in this ERA include direct contact with contaminated sediment or water and ingestion of CoCs associated with food, sediments, and water. Dermal absorption by avian ecological receptor species was not evaluated because of the large uncertainties associated with this pathway. Exposure routes from sediment through either direct contact or ingestion are discussed below. A summary of exposure routes for each CoC and species of concern or species group (e.g., benthic macroinvertebrates) is discussed below. These exposure routes or scenarios are also characterized in **Figure 3-2**.

Aquatic Species—Wetland and creek sediments act as exposure points via direct contact or ingestion by benthic and epibenthic macroinvertebrates and wetland insects (Figure 3-2). Oyster larvae in the Housatonic River at the mouth of the creek could also be exposed when contaminants are transported out of Ferry Creek. Macroinvertebrates also serve as exposure points through trophic transfer to mummichog, heron, and other predators. These pathways are identified because the COCs have a high affinity for solids and because some COCs bioaccumulate in tissues.

Exposure via air and groundwater was assumed to be of secondary importance based on the chemical-physical properties of the CoCs. These pathways will not be considered further in this ERA.

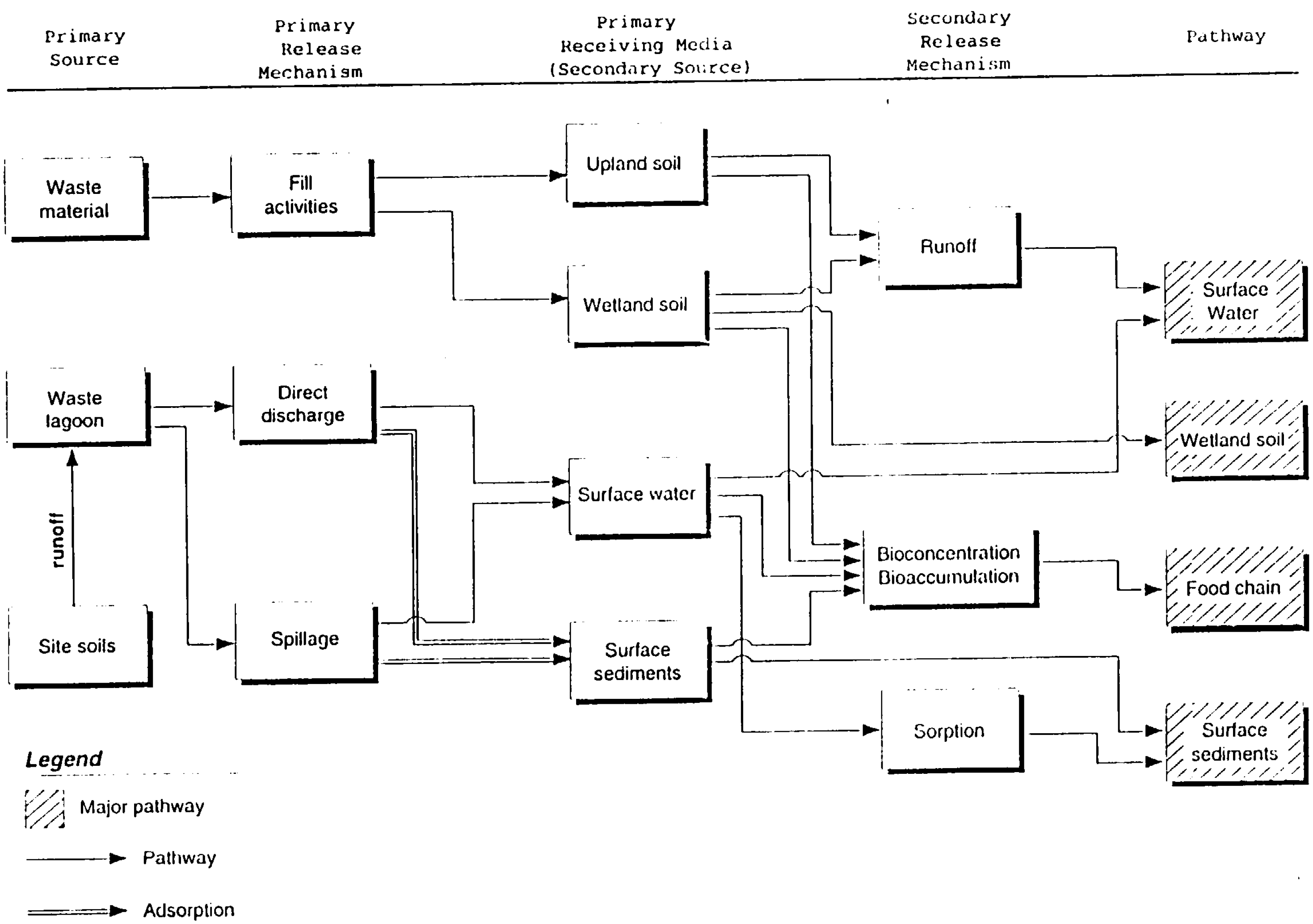

Figure 3-1. Primary contaminant pathways from the Raymark Industries site

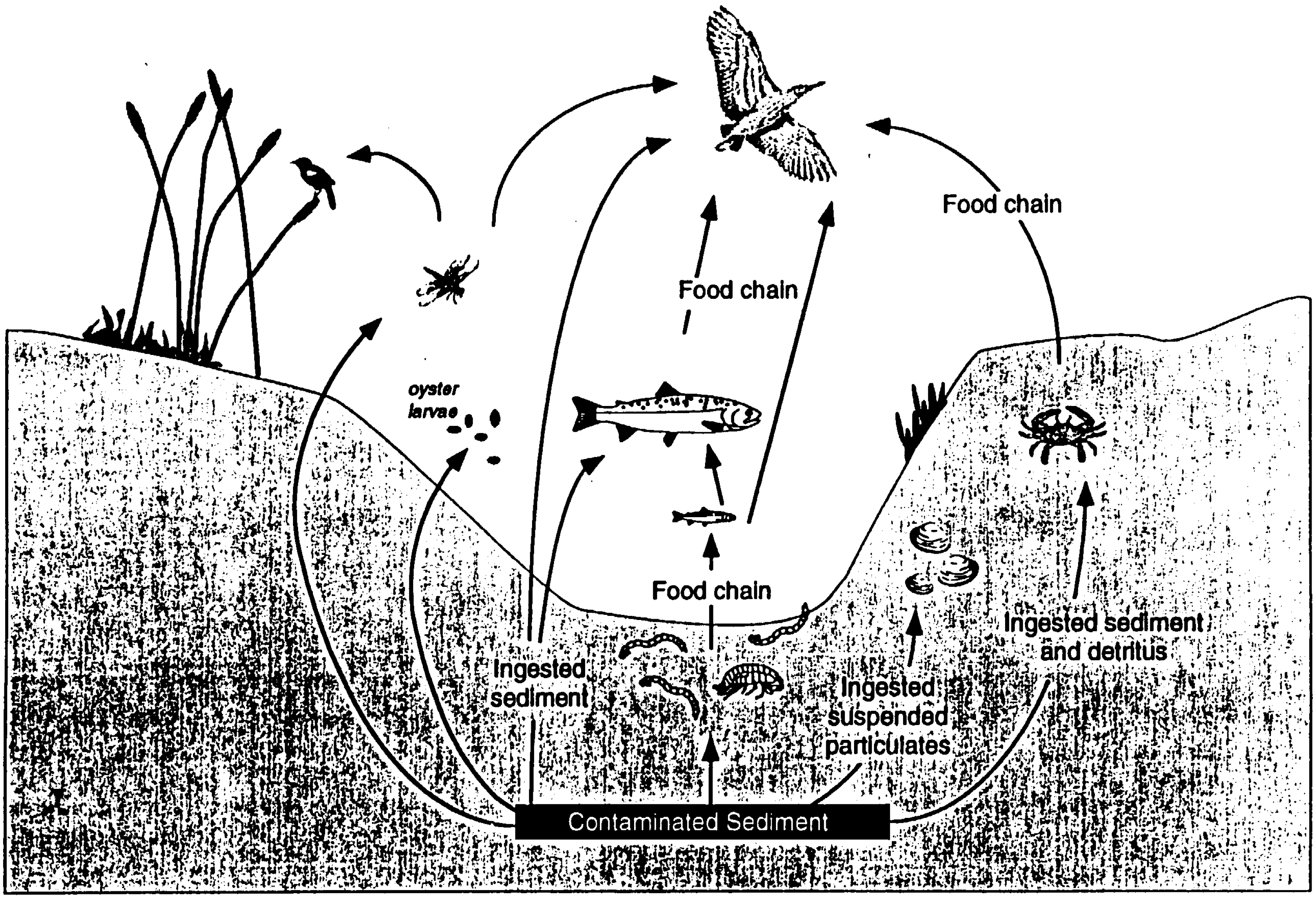

Figure 3-2. Generalized contaminant exposure scenarios for ecological receptor species of concern at the Raymark Industries site.

Based on these considerations, the exposure pathways retained for further consideration in this assessment are:

(1) Uptake from contaminated wetland, creek, and river sediments, and

(2) Uptake through the food chain within these contaminated habitats.

Avian Receptor Species—The primary exposure pathway for avian receptor species is through consumption of prey that have bioaccumulated site-related CoCs (Figure 3-2). For instance, black-crowned night heron could ingest CoCs through consumption of fish and fiddler crabs that are present in Ferry Creek and the Housatonic Boat Club wetlands. Red-winged blackbirds could ingest CoCs through consumption of terrestrial and emerged aquatic insects present in the wetlands. Black-crowned night herons can also be exposed through incidental sediment ingestion while feeding on crustaceans. Since black-crowned night herons feed directly in an aquatic environment, surface-water ingestion is considered an exposure pathway as well. For red-winged blackbirds, it is of secondary concern due to their feeding habitats.

3.3.2 CoC Bioavailability Profiles

Bioavailability from Water—The speciation of trace element CoCs in the water column and partitioning between aqueous and particulate phases are primary determinants of bioavailability and toxicity of these contaminants. Speciation in the water column is a function of the chemical and physical conditions, including pH, water hardness, temperature, dissolved oxygen, alkalinity, and total suspended solids. For metals, the relative concentrations of major ions and competing metal ions, and dissolved and total organic matter also determine speciation of these CoCs. With knowledge regarding the range and magnitude of these variables at each station, the speciation of some trace-metal contaminants can be measured analytically or predicted using speciation modeling with inherent uncertainties.

Increasing salinity and pH can substantially affect the speciation of metals. Generally, increasing salinity results in increased complexation by inorganic ligands. Partitioning behavior (i.e., dissolution or sorption) of both metals and organic compounds can also change dramatically in transition zones from fresh- to more saline waters, depending on a large number of site-specific variables. These variables include the dynamics of colloidal iron, natural organic matter, and particulate-matter settling and resuspension. Major shifts in partitioning behavior occur in transition from freshwater to very low-salinity conditions (only a few parts per thousand). The conditions measured in surface water during the August 1995 sampling round are summarized in **Table 3-3**.

In general, CoCs associated with suspended particles are not as bioavailable as dissolved CoCs (DiToro et al. 1991). Therefore, the dissolved concentration can be a better indicator of the acutely toxic fraction than the total concentration of contaminants measured in surface water. Typically, only those species that are freely dissolved (i.e., not complexed) are acutely toxic, although some exceptions exist. Even dissolved species (i.e., those that pass through a 0.45-μm filter) may be complexed with organic or inorganic ligands, or they may be associated with colloids. However, contaminated suspended particles can be ingested by planktonic organisms and may be bioavailable by that route of exposure. Contaminated particulates also settle out and become part of the sediment matrix, where their bioavailability may be altered drastically.

Table 3-3. Water quality conditions at each sampling location.

Sample Location	Temp. (°C)	Spec. Cond. (µm ohms/cm)	pH (standard units)	Oxygen (mg/L)	Salinity (ppt)
GM07	28	40	6.03	11.00	15
GM08	29.90	40.80	6.28	7.40	20
HB01	23.80	26.50	7.09	8	17
HB02	24.90	27.70	7.10	5	13
HB06	26.9	280*	6.09		19
HB8A*					
HB09†					
HB10	26.70	29.80	7.94	6.50	20
HB11*					
HB12	28.70	32	6.99	3	22.50
HB23	22.9	33.20	5.32	5.50	18
HB24*					
RF01	24.60	2.02	6.53	6	1
RF02	24.20	34.2	5.57	7.80	15.50
RF03	25.90	29.90	6.27	6.50	14
RF04*					
RF05*					
RF06*					
SD01	26.70	17.02	7.78	8.20	10
SD04†					
SD06	25.50	19.50	5.68	6.40	10
SD07	23.20	28.30	7.33	6.40	15
SD09	24.70	24.30	7.47	8	13
SD10	24.90	22.80	6.97	6.60	12
SD12	25.40	21.70	7.93	7.60	17
SD13	24.30	7.20	6.73	3.8	4
SD14	24.10	19.90	5.72	4.2	10
SD16*					
SD19	23.4	28.10	7.47	6.50	17
SD20	22	3.40	6.64	5.20	2
SD21	22.3	8.56	5.96	4.6	4
SD22	25.70	21.80	7.91	7.60	16
SD23	27.90	22.10	5.50	4.80	12
SD24	27.50	17.8	5.74	6.50	10
SD25	23.70	29.50	7.20	6.60	17
SD26*					
SD27*					
SD28	25.20	28.50	7.39	6.20	16
SD29	23.80	29.30	7.58	6.2	17
SD30	26.10	27.10	7.49	6.7	15
SD31	25.10	20.50	7.89	7	18.5
SD32	25	20.70	7.94	7	18
SD33	26.70	11.31	7.44	6.60	7.50
SD34	26	14.20	7.26	5.70	11
SD35	26.4	11.79	7.40	6.40	8.00
SD36	26.3	14.73	7.65	6	10
SD37	26.4	10.7	7.52	6	8
SD38*					

* A sediment sample was collected from this location, but no surface water was available to collect field measurement data.

Bioavailability from Sediment—Bioavailability from sediment is also primarily a function of partitioning between interstitial water and the various components of the sediment matrix. The partitioning between these sediment particles and interstitial water is critical to predicting the acute toxicity of a CoC in situ.

Several techniques have been used to predict partitioning of contaminants in sediment. Chemical and physical conditions within sediments are dynamic and complex, and bioavailability is therefore very site-specific. Consequently, these techniques have sought to identify the most critical factors to be considered when predicting partitioning, even though it is commonly recognized that a large number of factors influence the final result. The two theories most commonly applied are equilibrium partitioning for hydrophobic organic compounds (e.g., PCBs and PCDD/PCDF]) and the acid volatile sulfide (AVS) sequestering of divalent metals (i.e., Cd, Cu, Pb, Ni, and Zn). These two theories form the basis of EPA's proposed sediment-quality criteria, which are still in the development and verification stage.

In brief, the equilibrium partitioning theory assumes that organic coatings on sediment particles, as represented by TOC measurements, are the predominant determinant of the partitioning behavior of hydrophobic organic compounds between sediment particles and interstitial water (DiToro et al. 1991). This theory is based on the binding affinity of these hydrophobic CoCs for organic ligands. Therefore, by normalizing the concentration of hydrophobic organic compounds measured to the measured concentration of TOC, a better indication of the bioavailability of these compounds is obtained than by using the concentrations on a dry mass basis. Since sedimentary environments are complicated by a wide variety of organic matter, sediment particle surfaces, chemical gradients, physical resuspension, diffusion processes, and biological behavior, the theory is generally believed to provide only a general indication of partitioning and not a definitive prediction. An additional requirement for estimating the partitioning of hydrophobic organic compounds into interstitial water is knowledge of the partitioning constants (e.g., K_{OC}) for each individual COC. Use of literature values for these constants can introduce considerable uncertainty into the prediction unless these constants are measured at the facility (Brannon 1995).

The AVS theory also relies on the assumption that the dissolved interstitial metal concentration is also related to the abundance of a controlling phase, or sequestering agent, in the sediment matrix. According to this model, this sequestering element is assumed to be AVS, which is predominantly iron sulfides. The model states that if the AVS concentration is greater than the concentration of SEM, acute toxicity will not be observed (Di Toro et al. 1990) since the AVS sequesters all of the metals present. SEM are theoretically defined as metals whose divalent ions form more stable bonds with sulfide than does iron (Fe) (i.e., Cd, Cu, Ni, Pb, and Zn).

While the AVS theory has successfully predicted the acute toxicity of sediment contaminated with Cd and Ni (Ankley et al. 1991, Carlson et al. 1991) plus Zn and Pb (Casas & Crecelius 1994), success predicting the toxicity of Cu-contaminated sediments has been mixed (Ankley et al. 1993). Results with Hg, while theoretically an SEM, are limited and results to date indicate that interactions with organic matter and methylating microorganisms may be more important in affecting Hg bioavailability (NOAA 1995).

There are several possible explanations for the mixed results observed in experimental tests of the AVS theory:

(1) Other solid phases (e.g., Fe and manganese oxides) and other complexing ligands (e.g., natural organic matter) in sediment systems may successfully compete for dissolved metals, or

(2) Organisms may alter the condition of their immediate environment, thereby exposing themselves to conditions different from those measured in the bulk sediment (e.g., different AVS concentrations or pH).

In addition, AVS theory does not work for numerous elements, including Cr and arsenic, that are generally not associated with sulfides in sediments. Despite these limitations, AVS is considered a suitable screening tool for sediment toxicity on a site-by-site basis for certain metals. Direct measurement of metal species within interstitial water is also recommended where possible.

Bioavailability from Ingestion—If contaminated particles or food are ingested, only a fraction of the total concentration of CoCs associated with the ingested item is generally assumed to be assimilated. The proportion of any given CoC assimilated varies according to the CoC in question, the species involved, their feeding behavior, the pH of their gut, enzyme activity, enzyme induction levels, and so on. Unfortunately, this type of information is quite scarce, and determining the bioavailable fraction for a particular CoC in diets of specific organisms is generally not possible.

3.4 Ecological Communities Potentially at Risk

Four areas were selected as representative of those identified as potentially posing a risk to ecological receptors from site-related contaminants. These areas are the upper reaches of Ferry Creek, the lower reaches of the creek, portions of the Housatonic River, and wetlands adjacent to the Housatonic River near the Housatonic Boat Club (Figure 2-1). Receptors that use these areas include both aquatic and terrestrial species whose diets and potential exposures are closely tied to open water and wetland habitats.

3.4.1 Ferry Creek

Ferry Creek is located approximately 600 meters (m) from the Raymark facility. Ferry Creek has been divided into two reaches—Upper Ferry Creek, above the tide gate at Broad Street; and Lower Ferry Creek. The two reaches were evaluated separately because of differences in the influences of the tidal regime. The tide gate is situated on Ferry Creek approximately 200 m from the Housatonic River and is equipped with flapper gates (Figure 2-3). This gate largely restricts anadromous fish passage to the upper portions of Ferry Creek. However, the flapper gates are often stuck open by debris washing downstream. Tidal incursions of Housatonic River water do occur in Ferry Creek, as indicated by salinities. At high tide, salinity just upstream of the gate has been measured as high as 25 ppt, while only 1 ppt was measured at the head of the creek near the storm-water culvert draining the Raymark facility (Table 3-3). Salinities in the Housatonic River near Ferry Creek range from 0 ppt on the surface during high-flow periods to 25 ppt near the sediment during low-flow periods. Despite the flapper gates, some limited fish passage beyond the tide gate is likely, as with saline water incursion.

A variety of fish and invertebrate species use Lower Ferry Creek and the associated wetlands **(Table 3-4)**. Important anadromous and catadromous species using the creek include alewife, American shad, blueback herring, hickory shad, rainbow smelt, striped bass, and white perch. Dominant fish species of Lower Ferry Creek include Atlantic menhaden, bay anchovy, black seabass, striped killifish, mummichog, inland and Atlantic silversides, summer and windowpane flounder, and spotted hake. Important invertebrate species include the blue crab, fiddler crab, Eastern oyster, blue mussel, and soft and hardshell clams (Kaputa 1995; Aarestad 1994, 1995).

Table 3-4. Aquatic species associated with the lower Housatonic River, lower Ferry Creek below the tide gate, and the Housatonic Boat Club wetlands.

Species		Habitat Use			Fisheries	
Common Name	Scientific Name	Spawning /Mating	Nursery Ground	Adult Forage	Comm. Fishery	Recr. Fishery
Marine/Estuarine Species						
American sandlance	*Ammodytes americanus*		✓	✓		
Atlantic croaker	*Micropogonius undulatus*		✓			
Atlantic herring	*Clupea harengus*		✓	✓		
Atlantic mackerel	*Scomber scombrus*		✓			
Atlantic menhaden	*Brevoortia tyrannus*		✓	✓		✓
Atlantic silverside	*Menidia menidia*	✓	✓	✓		
Atlantic sturgeon	*Acipenser oxyrhynchus*		✓	✓		
Atlantic tomcod	*Microgadus tomcod*		✓	✓		
Bay anchovy	*Anchoa mitchilli*		✓	✓		
Black sea bass	*Centropristis striata*		✓	✓		✓
Bluefish	*Pomatomus saltatrix*		✓	✓		✓
Butterfish	*Peprilus triacanthus*		✓			
Crevalle jack	*Caranx hippos*		✓			
Cunner	*Tautogolabrus adspersus*		✓			
Fourbeard rockling	*Enchelyopus cimbrius*		✓			
Four-spine stickleback	*Apeltes quadracus*	✓	✓	✓		
Fourspot flounder	*Paralichthys oblongus*			✓		✓
Grubby	*Myoxocephalus aenaeus*		✓			
Hogchoker	*Trinectes maculatus*	✓	✓	✓		
Inland silverside	*Menidia menidia*	✓	✓	✓		
Inshore lizardfish	*Synodus foetens*		✓			
Little skate	*Raja erinacea*			✓		
Mummichog	*Fundulus heteroclitus*	✓	✓	✓		
Naked goby	*Gobiosoma bosc*		✓	✓		
Nine-spine stickleback	*Pungitius pungitius*			✓		
Northern kingfish	*Menticirrhus saxatilis*		✓			
Northern pipefish	*Syngnathus fuscus*	✓	✓	✓		
Northern puffer	*Sphoeroides maculatus*		✓			
Northern searobin	*Prionotus carolinus*		✓	✓		
Oyster toadfish	*Opsanus tau*	✓	✓	✓		
Rock gunnel	*Pholis gunnellus*		✓	✓		
Scup	*Stenotomus chrysops*		✓			
Sheepshead minnow	*Cyprinodon variegatus*	✓	✓	✓		
Smallmouth flounder	*Etropus microstomus*		✓			
Spanish mackerel	*Scomberomorus maculatus*	✓				
Spot	*Leiostomus xanthurus*		✓			
Spotted hake	*Urophycis regia*		✓			
Striped killifish	*Fundulus majalis*	✓	✓	✓		
Summer flounder	*Paralichthys dentatus*		✓	✓		✓
Tautog	*Tautoga onitis*		✓			

Table 3-4 continued

Species		Habitat Use			Fisheries	
Common Name	Scientific Name	Spawning /Mating	Nursery Ground	Adult Forage	Comm. Fishery	Recr. Fishery
Marine/Estuarine Species						
Three-spine stickleback	*Gasterosteus aculeatus*		✓	✓		
Weakfish	*Cynoscion regalis*					
Windowpane flounder	*Scophthalmus aquosus*		✓	✓		✓
Winter flounder	*Pleuronectes americanus*		✓	✓		✓
Anadromous/Catadromous Species						
Alewife	*Alosa aestivalis*	✓	✓	✓		
American eel	*Anguilla rostrata*	✓	✓	✓	✓	✓
American shad	*Alosa sapidissima*	✓	✓	✓		
Blueback herring	*Alosa aestavalis*	✓	✓	✓		
Hickory shad	*Alosa mediocris*			✓		
Rainbow smelt	*Osmerus mordax*	✓	✓			
Striped bass	*Morone saxatilis*		✓	✓		✓
White perch	*Morone americana*	✓	✓	✓		✓
Invertebrate Species						
Atlantic rock crab	*Cancer irroratus*	✓	✓	✓		
Blue crab	*Callinectes sapidus*		✓	✓		✓
Blue mussel	*Mytillis edulis*	✓	✓	✓		
Eastern oyster	*Crassostrea virginica*	✓	✓	✓	✓	
Green crab	*Carcinus maenas*	✓	✓	✓		
Hard-shelled clam	*Mercenaria mercenaria*			✓		
Horseshoe crab	*Limulus polyphemus*			✓		
Lady crab	*Ovalipes ocellatus*	✓	✓	✓		
Mud crab	*Panopeus spp.*	✓	✓	✓		
Sand shrimp	*Crangon septemspinosa*	✓	✓	✓		
Shore shrimp	*Palaemontes spp.*	✓	✓	✓		

Wetlands associated with Ferry Creek primarily form corridors along the creek, but are limited in size because of adjacent development along the creek banks. The wetland area present in the portion of the creek above the tide gate is largely disturbed and is predominantly composed of common reed grass (*Phragmites communis*), jewelweed (*Impatiens capensis*), bindweed (*Polygonum* spp.), seabeach roach (*Atriplex arenaria*), and poison ivy (*Rhus radicans*; DeLong 1993).

There was no information on avian species' use of habitat specifically within the Ferry Creek zone. However, observations have been recorded at the Milford Point Audubon Center, just across the Housatonic River from the creek (discussed below). Species use between these two areas is likely similar due to physical proximity and similar habitat. Also, during numerous site visits, black-crowned night heron and red-winged blackbirds were observed near the creek (Svirski 1997).

3.4.2 Housatonic River

The Housatonic River provides habitat for numerous migratory and estuarine fish and invertebrate species (Table 3-4). The most common fish species living in the Housatonic River near the facility include mummichog, Atlantic silverside, four-spine stickleback, naked goby, winter flounder, little skate, northern pipefish, and American eel. Common species found on a seasonal basis in the lower Housatonic River estuary include striped bass, bay anchovy, Atlantic menhaden, black seabass, small mouth flounder, Atlantic tomcod, summer flounder, bluefish, striped searobin, northern puffer, tautog, and blue crab. Anadromous runs of alewife, blueback herring, American shad, hickory shad, and rainbow smelt commonly enter the Housatonic River in spring to access suitable freshwater spawning grounds farther upstream.

Bluefish, found in the lower Housatonic River from May to November, are predatory fish that feed on Atlantic silverside and mummichog and support a popular sportfishery near the facility. Striped bass and blue crab are also seasonal predators that feed on Atlantic silverside and mummichog. Striped bass are present in the Housatonic estuary during spring and fall to feed on the herring runs in the river. Other predatory species include summer flounder, black seabass, white perch, hickory shad, weakfish, Atlantic herring, and striped searobin (MacLeod, pers. commun., 1995).

Recreational fish species such as crevalle jack, scup, weakfish, northern kingfish, black seabass, spot, Atlantic croaker, butterfish, and tautog use the lower Housatonic River primarily as nursery grounds for juveniles. Therefore, recreational fishing for these species in this area is not significant. However, adjacent areas in Long Island Sound do have important recreational fisheries for some of these species. These fisheries depend on the Housatonic River to support fish in their juvenile life-history stages. Although windowpane flounder and spot are not targeted directly as recreational species, they are harvested as bycatch in the important summer flounder recreational fishery. Seals also have been observed in the lower Housatonic River by fisheries biologists, although exact species identification is unavailable at this time (Kaputa 1995).

An important commercial larval bed for eastern oyster (*Crassostrea virginica*) cultivation is in the lower Housatonic River near the mouth of Ferry Creek. This oyster fishery is regulated under a State of Connecticut transplanting program. Oyster spat are annually collected and transplanted to certified offshore areas in Long Island Sound, where they grow to maturity in 3 to 4 years before being commercially harvested. Approximately 30,000 to 130,000 bushels of oyster spat are transplanted each year from the lower Housatonic River (Volk 1995). In addition to the oysters, other bivalves–particularly mussels–are found in the area.

Estuarine intertidal wetlands along the Housatonic River are largely undisturbed and dominated by smooth cordgrass (*Spartina alterniflora)* and salt meadow hay *(Spartina patens).* Milford Point is a prominent estuarine intertidal wetland that occupies about 245 hectares opposite the mouth of Ferry Creek on the Housatonic River. A number of bird species have been observed at the Milford Point Audubon Center. Nests of the Atlantic Coast piping plover *(Charadrius melodus)*, a federal threatened species, have been observed at Milford Point (Milton, pers. commun., 1995). A second large coastal tidal wetland near the study area is the Great Meadows, about 8 km downstream of the mouth of Ferry Creek. This wetland is a known nesting area for the least tern *(Sterna paradisaea)*, a State threatened species, and the Atlantic piping plover (DeLong 1993).

3.4.3 Housatonic Boat Club Wetlands

There are substantial wetlands located next to and south of the Housatonic Boat Club on the west shore of the Housatonic River, just south of the mouth of Ferry Creek. The wetlands are estuarine, intertidal wetlands largely undisturbed and dominated by smooth cordgrass (*Spartina alterniflora*) and salt meadow hay (*Spartina patens*). During low tide, the channels of this wetland are completely drained, so that only temporary habitat is available for fish. Salinities in the channels range from 15 to 25 ppt. These channels are likely to be inhabited by a variety of fish species such as alewife, American shad, blueback herring, hickory shad, rainbow smelt, striped bass, and white perch. Dominant fish species would include Atlantic menhaden, bay anchovy, black seabass, striped killifish, mummichog, inland and Atlantic silversides, summer and windowpane flounder, and spotted hake. Important invertebrate species include the blue crab, fiddler crab, Eastern oyster, blue mussel, soft and hardshell clams (Kaputa, pers. commun., 1995; Aarestad, pers. commun., 1994, 1995). As with Ferry Creek, information on bird species was not available for this area. However, it is again likely that the same bird species observed at Milford Point may also use the Housatonic Boat Club wetlands for forage areas.

3.5 SELECTION OF ENDPOINTS & REPRESENTATIVE RECEPTOR SPECIES

Numerous species of aquatic invertebrates, fish, and birds, as indicated in Table 3-2, could potentially be exposed to the CoCs in the areas of interest. Because a risk assessment cannot investigate all of these potential receptors, representative species were selected in the SLERA (EVS 1995) from the general suite of receptor species known to exist in the study area. These representative species were chosen based on the assumption that they were most likely to be the receptors at potential risk due to their life history or ecological niche. The following species were selected for evaluation as receptors in the SLERA: benthic infauna, blue crab, American oyster, striped bass, black-crowned night heron, and the Atlantic piping plover. Preliminary exposure estimates and risk estimates, including a calculation of HQ using maximum likely exposure concentrations and RTV, were modeled from existing information. The results presented in the SLERA concluded that there was potential risk to these evaluated ecological receptors. This ERA responds to recommendations for a complete risk assessment for these species. However, for this ERA some of these endpoints were further refined during work-plan preparation to use other, surrogate species as measurement endpoints.

3.5.1 Selection of Assessment Endpoints

Assessment endpoints represent an explicit statement of the environmental values that are to be protected. More specifically, they are statements addressing the viability of the communities, populations, species, or habitats of particular concern at a site due to their susceptibility to CoCs associated with releases from the facility. Based on results from the SLERA and the problem formulation, four assessment endpoints were selected. These assessment endpoints form the basis for this Phase- II ERA and were agreed upon by participating agencies. The four assessment endpoints are:

- Survival, growth, reproduction, and appropriate indigenous benthic community (both infauna and epibenthic) composition in Ferry Creek, the Housatonic River near the mouth of Ferry Creek, and the wetlands associated with those areas;
- Survival, growth, and reproduction of oysters in the seed beds in the Housatonic River at the mouth of Ferry Creek;
- Protection of fish species from adverse reproductive effects and mortality; and

- Protection of avian species foraging in the area from adverse growth or reproductive effects and mortality.

3.5.2 Selection of Measurement Endpoints

Assessment endpoints are linked to testable hypotheses by measurement endpoints. Measurement endpoints are the metrics or parameters that can be related back to an assessment endpoint, that characterize the status of that assessment endpoint, and that can be directly investigated in the risk assessment process. They must be directly measurable and responsive to the attributes of the CoCs in question.

To assess whether elevated CoCs in sediment and wetland soils are posing a risk to the benthic community, a sediment-quality triad approach was used. The triad is a weight-of-evidence approach based on three different measures of sediment quality: bulk sediment chemistry, sediment toxicity, and benthic community structure. This triad analysis was conducted at four stations in Ferry Creek, one station in the Housatonic Boat Club wetlands, and two reference stations. Concentrations of CoCs in the sediments, sediment toxicity to an amphipod, and benthic community structure at these stations were compared with values obtained from a reference location. The coincidence of elevated concentrations of CoCs, greater sediment toxicity, and benthic community alterations in Ferry Creek relative to reference areas was the measure of impacts to the benthic community. Additional stations from each of the areas of interest were also sampled and tested for chemical content and amphipod mortality to provide additional supporting information.

Acute toxicity tests using oyster larvae were performed to assess whether recruitment of oyster spat may be reduced by elevated CoCs in wetland or creek sediments that would be scoured during a storm event (and subsequently transported to the Housatonic River). Sediment was collected from sampling stations in Upper and Lower Ferry Creek and the Housatonic Boat Club wetlands. Mortality and abnormal development of larvae were measured to determine the potential for reduced recruitment.

Site-specific fish tissues were collected to assess whether reproduction and survival of resident fish species are being adversely affected through consumption of prey that have bioaccumulated CoCs. These tissues were then analyzed for chemical content. Mummichog was identified as a surrogate for low-trophic-level omnivorous fish. Fish tissues were collected from the entire length of Ferry Creek. To evaluate a higher trophic-level fish species, historical tissue and sediment data were reviewed for white perch (*Morone americana*). The perch had been collected from nearby ponds, one of which was suspected to be impacted by Raymark waste. Measured contaminant concentrations in fish tissue were compared with available benchmark values. Maximum acceptable tissue concentrations (MATC) are the benchmarks that are expected to be protective of reproductive effects or mortality. Risk was inferred by calculation of an HQ, i.e., the ratio of the measured concentrations versus the MATC. An HQ greater than 1 represents an exceedance of the benchmark tissue concentration. HQs less than 1 are expected to be protective of those adverse impacts represented by the benchmark value (e.g., reproductive effects or mortality).

To assess the potential for reduced reproduction and mortality in avian receptor species, an HQ approach was also used. For birds, however, concentrations of their diet were compared with doses expected to produce no adverse effects based on laboratory or other field studies. These doses are referred to as RTVs. Site-specific tissue concentrations (i.e., mummichog, fiddler crab, and terrestrial and emergent aquatic insects) were measured for use as input variables in an avian food web model. Additionally, data on ingestion of surface water and sediment were incorporated into the food-web model for heron so that all potential pathways were evaluated. An estimated daily dose was calculated for the black-crowned night heron and red-winged blackbird, and then compared with literature-derived benchmark RTVs. In this manner, an HQ was calculated. An HQ greater than 1 represents an exceedance of the benchmark dose by the estimated dose. Ratios less than 1 are then expected to be protective of those adverse impacts reflected by the benchmark dosage.

The measurement endpoints selected for this ERA differ slightly from those suggested by the SLERA. For example, the fiddler crab was selected as a surrogate for the blue crab, and the mummichog was substituted for the striped bass. These species not only served as appropriate surrogates, but were also substituted to improve the probability of success during the field effort to obtain relevant field data that would have broader use. Due to sampling constraints, the federally threatened Atlantic piping plover was eliminated. The red-winged blackbird was added during work-plan development as a measurement endpoint to address issues regarding potential risk to insectivorous birds in the study area. Also, evaluation of tissue body burdens of contaminants in white perch was added as a measurement endpoint for an assessment endpoint of trophic transfer to predatory fish species.

3.6 SPECIES PROFILES

The final list of measurement endpoints and representative ecological receptor species provides a diverse combination of species known to occur in the area that may be sensitive to the effects of the CoCs. The benthic infauna were chosen because they provide a resource base for higher-level consumers, are highly sensitive to many of the CoCs, and can represent an integrative, long-term measure of impacts. Fiddler crabs were chosen as a representative epibenthic macro-invertebrate because they are important in the diet of herons, they could be used as a surrogate for blue crab, they have a limited home range, and their omnivorous diet links them closely with the sediment (Ricketts & Calvin 1968). The eastern oyster was selected because maintenance of oyster spat beds is an important resource in the area, and many of the metal and metalloid CoCs can reduce the recruitment of oyster spat due to embryo toxicity. The mummichog was chosen because it is an important food source for birds such as herons, has a limited home range, and can serve as a surrogate for lower-trophic-level fish species. Data on contaminants in tissue of white perch were evaluated because many of the CoCs are known to biomagnify. The black-crowned night heron (a species of concern in the State of Connecticut) and red-winged blackbird were selected because of their feeding habits. The heron is an opportunistic, primarily aquatic feeder, while the blackbird is primarily an insectivore.

A brief discussion of the natural history of the receptor species is presented below. These life-history characteristics are considerations when determining the exposure potential of receptor species to CoCs at the facility.

3.6.1 Aquatic Species Profiles

Benthic Organisms—Benthic organisms live in or on the sediments, and are very important members of the marine ecological system. The benthic community is much richer than the pelagic community—157,000 species versus about 3,000 (Thorson 1971). Soft, sedimentary bottoms are deceptive: They appear dull and relatively lifeless, yet when even the smallest of organisms are counted (<0.5 mm), benthic communities may number over a half million per square meter.

Diverse, abundant infaunal and epibenthic organisms are necessary to maintain healthy estuarine communities. Benthic communities provide considerable biomass to support ecological food webs in estuaries. Members of the benthic community are also important processors of organic matter. Although organisms within these communities are largely immobile, numerous benthic or epibenthic species have planktonic larval stages. Successful settling and colonization of sediment by larvae usually require particular conditions suited to the species, including lack of stressors. Studies have shown that sensitive genera within the benthic and epibenthic community, such as amphipods, are among the first species to disappear from polluted areas (Lamberson et al. 1992).

Benthic communities are usually segregated by factors such as depth, grain size, salinity, exposure, and organic carbon. These types of physico-chemical factors combine to define a habitat niche conducive to only certain assemblages of benthic species. This is especially true of salt marshes which, like those found along Ferry Creek, are regularly flooded by estuarine water plus occasionally flooded from uplands by rainwater. Few species can tolerate the fluctuating conditions of these salt marshes. Polychaete worms; bivalve molluscs, especially the ribbed mussel (*Modiolus d.*); pulmonate snails (those with lungs instead of gills), other snails including the periwinkle (*Littorina l.*); and larger, foraging crustaceans are the most common benthic macro-organisms observed in East Coast salt marshes (Berrill & Berrill 1981).

Fiddler crabs—Two species of fiddler crabs are quite common to the flats and banks of salt marshes. Both the sand fiddler *(Uca pugilator)* and the mud fiddler *(Uca minax)* are very tolerant of fluctuating salinities. Fiddler crabs are well suited to tidal marsh conditions because they have primitive lungs rather than gills and are able to withstand long periods of submergence without oxygen. Fiddlers are famous for the breeding behavior of males: during low tide, males stand at the mouths of their burrows and wave their singularly large claw rhythmically in the air to attract females. A mated female extrudes eggs that are then carried under the tail until they are released into the water when fully mature. Larvae that survive metamorphosis settle to the bottom and begin foraging. Fiddler crabs are omnivores. Their feeding behavior, burrowing, and limited mobility combine to make these species good indicators of local benthic stress.

Eastern oyster—The eastern oyster *(Crassostrea virginica)* inhabits estuaries, drowned river mouths, and areas behind barrier beaches. Adults are completely sessile; their distribution depends upon where free-swimming larvae are successful at settling. Adult oysters typically live in clumps in which they are the dominant organism (Sellers & Stanley 1984). Temperature primarily initiates spawning of eastern oysters. Eggs and sperm are discharged into open water. Mass spawning provides concentrations of spawn needed to ensure fertilization when sexual products are discharged freely into open water. After fertilization, oyster larvae are free-swimming in the water column for 2 to 3 weeks before settling to the bottom and attaching to a solid object (preferably, other oyster shells). Dispersion of the larvae during this time depends upon local currents, but larvae may be transported long distances before settling (Sellers & Stanley 1984, Quayle 1988).

Mummichog—The mummichog *(Fundulus heteroclitus)* is a euryhaline species that inhabits shallow and low-salinity salt marsh flats, estuaries, and tidal areas, often found in schools near submergent or emergent vegetation. The species tolerates a wide range of salinities and temperatures. Mummichog are year-round residents of these habitats; there is no evidence that they engage in regular or predictable migrations. The home range of mummichog in tidal creeks is believed to be limited; in one study, the majority of individuals in a population exhibited a home range of 36 m near the bank of a tidally influenced creek (Lotrich 1975). Mummichogs spawn in shallow nearshore waters; eggs are deposited in clutches on the outer sides of aquatic plants, on masses of algae, in sand and mud substrate, and on mussel shells. The mummichog are opportunistic, omnivorous feeders, consuming a variety of amphipods and other small crustaceans, molluscs, polychaetes, insect larvae, and vegetable matter. They are preyed upon by a variety of animals, fishes, and birds, due in part to the availability of the species in schools in shallow inshore waters. They are reportedly consumed by kingfishers, otter, mink, and brook trout (Scott & Crossman 1973, Scott & Scott 1988).

White Perch—The white perch *(Morone americana)* was not included as an ecological receptor for the field-sampling effort undertaken as part of this ERA because of resource constraints. Rather, existing tissue contaminant data were reviewed to assess the potential for bioaccumulative CoCs to be transferred to predatory fish species. White perch live in a variety of habitats ranging from estuaries of high or low salinities, rivers, lakes, and ponds. The species tolerates a wide range of salinities and temperatures. Many populations are anadromous, but others live and spawn in low-salinity estuaries, and others are landlocked. Anadromous migration to fresh or brackish water is required only for marine populations. Rising temperatures in the spring stimulate spawning, but there are apparently no preferred spawning habitats. The species will spawn in waters that are tidal or nontidal, clear or turbid, fast or slow, with bottom substrates ranging from clays to gravel. Spawning usually occurs in freshwater, but has been observed in brackish waters at salinities of 4.2 ppt or less. Inshore zones of estuaries and creeks are used as nurseries. Adults generally live in the same areas, farther offshore in deeper water. Except for spawning movements, adults apparently do not migrate (Stanley & Danie 1983). Juvenile white perch are opportunistic demersal feeders, consuming microplankton and aquatic insect larvae. Larger white perch are opportunistic predators consuming fish, insect larvae, spawn of other fish, and crabs (Scott & Crossman 1973). White perch have been observed in ponds and estuarine areas both within and near the study area.

3.6.2 Avian Species Profiles

Black-crowned night heron—The black-crowned night heron *(Nycticorax nycticorax)* is common throughout the United States, and its breeding range includes the northern United States and Canada. Distribution of this species depends on suitable wetland habitat for breeding (Davis 1993). After the breeding season has ended, herons in the northern part of the range migrate south in late September or October (Davis 1993), although some birds winter in New England (Bent 1926; Palmer 1962; Ohlendorf et al. 1978, as cited in Davis 1993). Although black-crowned night herons have been documented in Connecticut during the Christmas Bird Count, they were not likely the same birds that had nested in the area (Parsons, pers. commun., 1995). The birds that nest in an area do not remain during the winter due to replacement migration (Parsons, pers. commun., 1995). Herons arrive in the Northeast by the end of March where they establish breeding colonies associated with large wetlands.

The black-crowned night heron is medium in size relative to other herons. As adults, they range from 58 to 66 cm (23–26 inches [in]) long and weigh 500–907 g for males (1.6–2 lb) or 727–884 g for females (1.6–1.9 lb) (Terres 1991). Females are typically slightly smaller than males. The sexes have similar plumage.

The black-crowned night heron is a social nester often found in mixed colonies. Herons will nest in areas associated with virtually any type of water body. Nests are constructed in trees, cattail marshes on prairies, or in clumps of tall grass on dry ground (Terres 1991).

Black-crowned night herons feed in shallow weedy areas of ponds, creeks, and marshes where aquatic vegetation provides cover for fish, invertebrates, and amphibians. They are primarily nocturnal feeders, but will hunt during the day when feeding nestlings. This species exhibits feeding-site fidelity and will use the same feeding site repeatedly (Parsons, pers.

commun., 1995; Gross 1923, as cited in Davis 1993). Tide affects the selection of foraging areas, as birds will fly farther during high tide to reach a foraging area (Custer & Osborn 1978, as cited in Davis 1993). Gross (1923) determined that grassy salt-marsh areas were the most important foraging areas for herons.

Black-crowned night herons are opportunistic feeders and consume a variety of aquatic and terrestrial species. They have been documented feeding on small terrestrial mammals, snakes, lizards, and chicks of other bird species (US EPA 1995). They are primarily piscivorous but will also eat molluscs, crustaceans, and insects, whatever is most available (Palmer 1962).

The nearest black-crowned night heron colony is located on Charles Island, about 3.5 miles (5.6 km) east of Ferry Creek. This species has been seen during daylight hours perched in trees near Ferry Creek and feeding in the creek. These herons are likely feeding on aquatic prey species that may have accumulated elevated concentrations of CoCs in their tissues.

Red-winged Blackbird—The red-winged blackbird *(Agelaius phoeniceus)* is a ubiquitous, marsh-dwelling bird. The red-wing lives in marshes and sloughs or along sluggish streams where bushes and small trees provide perching and nesting habitat (Terres 1991). Its breeding range extends from Alaska to Costa Rica (Oriens 1987). In winter this species leaves the northern part of its breeding range and winters over much of the United States, particularly in the southern states (Terres 1991). They rarely winter north of Connecticut, but regularly overwinter as far as southern Texas (Bent 1958). Red-winged blackbirds are often observed in wetlands during spring and early summer. Males arrive in New England in March to establish breeding territories in the center and along the perimeters of marshes. Groups of males frequent open fields where they feed on vegetation before insects become available (Bent 1958).

The red-winged blackbird is in the Icteridae or blackbird family. They range from 19 to 23 cm) (7.5–9 in) long, with a wingspan of 30–37 cm (12–14.5 in) as adults. Males and female adults weigh on average 65 g (2.2 oz) and 43 g (1.5 oz), respectively (Terres 1991).

Male blackbirds defend a territory for breeding and feeding. These territories may range in size from 0.03 to 0.23 hectares (0.08–0.57 acres). Female red-winged blackbirds forage extensively off their territories, and will fly to other areas where a richer food source is available (Oriens 1987). However, during the nesting season, males spend nearly the entire day on their territories (Oriens 1987).

Red-winged blackbirds nest in cattails, rushes, bushes, trees, and in some cases on the ground in dense grass. Generally 3 to 5 eggs are laid between March and July. Young birds have developed fully and are ready to leave the nest 11 days after hatching. This allows the adults to produce a second brood. Nestlings are fed insects—primarily mayflies, caddis flies, and lepidopteral larvae (Allen 1914, as cited in Bent 1958). Gabrielson (1914, as cited in Bent 1958) lists a variety of insects fed to young red-wings including crickets, beetles, mayflies, flies, spiders, worms, grasshoppers, and moths.

This species forages in either wetlands or upland areas, depending on the season (Oriens 1987). Birds feed in upland areas until eggs incubate, after which they remain in the marsh area (Bent 1958). During the nesting season male and female blackbirds feed on insects in the marshes or wetlands, but will fly to upland areas to feed on insects, fruits, and seeds. Blackbirds feed in marshes during early morning and late afternoon. They are primarily ground feeders but also pick insects off vegetation and consume flying insects. During late summer and fall, this species joins grackles, cowbirds, and starlings to feed on weed seeds and waste grain in open fields (Terres 1991). On a year-round basis their diet consists of 73% vegetable matter and 27% animal matter (Beal 1900, as cited in Bent 1958). During the spring and summer this species consumes about 40% and 50% insects, respectively (Martin et al. 1951).

Red-winged blackbirds are common in the Ferry Creek and Housatonic Boat Club wetlands as well as the reference areas during the nesting season (March through July). While nesting, they consume and feed their nestlings insects that may contain elevated concentrations of CoCs. As noted in Table 3-2, most of the CoCs associated with the Raymark facility impair reproductive success.

4.0 FIELD SAMPLING DESIGN

A field sampling plan was developed to generate the additional site-specific data necessary to assess the risks to biota from the Ferry Creek area, the Housatonic River, and associated wetlands. This sampling plan was consistent with the measurement endpoints outlined in Section 3.5 and designed to fill gaps in data gathered during the field sampling effort under the RI. This section discusses the sampling objectives, sampling methods, and analytical methods for each of the following components of the field study:

- sediment-quality triad, including sediment chemistry, toxicity tests, and benthic community analysis;
- concentrations of CoCs in mummichog;
- concentrations of CoCs in fiddler crabs; and
- concentrations of CoCs in insects.

Sediment, water, and tissue samples were collected from four general areas selected because of their varying hydrologic and habitat features as well as site-related history:

- Upper Ferry Creek, upstream of the tide-control gate **(Figure 4-1)**;
- Lower Ferry Creek, downstream of the tide-control gate to the creek's mouth at the Housatonic River **(Figure 4-2)**;
- wetlands near the Housatonic Boat Club **(Figure 4-3)**; and
- reference areas, including Milford Point and Beaver Brook **(Figure 4-4)**.

Sampling stations within each area were chosen to reflect the range of habitat conditions and contaminant concentrations in sediment in that area. Previous sampling and analysis of sediment indicated distinct areas of elevated concentrations of CoCs due to nearby disposal of Raymark waste. Therefore, it was expected that contaminant concentrations within each of these areas would be rather heterogeneous and would not display simple dilution gradients. Two locations within the reference area were chosen; one representative of high-salinity (Milford Point) and one of low-salinity conditions (Beaver Brook). **Table 4-1** shows the number of sampling stations and samples collected for each medium.

All sediment and tissue samples were analyzed for the CoCs presented in Table 3-1, except as noted in this section. In addition, conventional parameters considered potential indicators of contaminant bioavailability (e.g., grain size, TOC) and SEM/AVS were measured in the sediment. Percent lipid and percent moisture were measured in tissue samples.

4.1 *SEDIMENT TOXICITY*

Sediment toxicity was evaluated by analyzing CoC content of sediment, by toxicity testing with amphipods and oyster larvae, and by evaluating resident benthic macroinvertebrates.

4.1.1 Sediment Sampling Objectives

The sediment-quality triad includes synoptic measures of sediment chemistry, sediment toxicity, and benthic community structure. Sediment-quality triad analyses were conducted at seven locations that were selected based on historical sediment-sampling data to achieve a

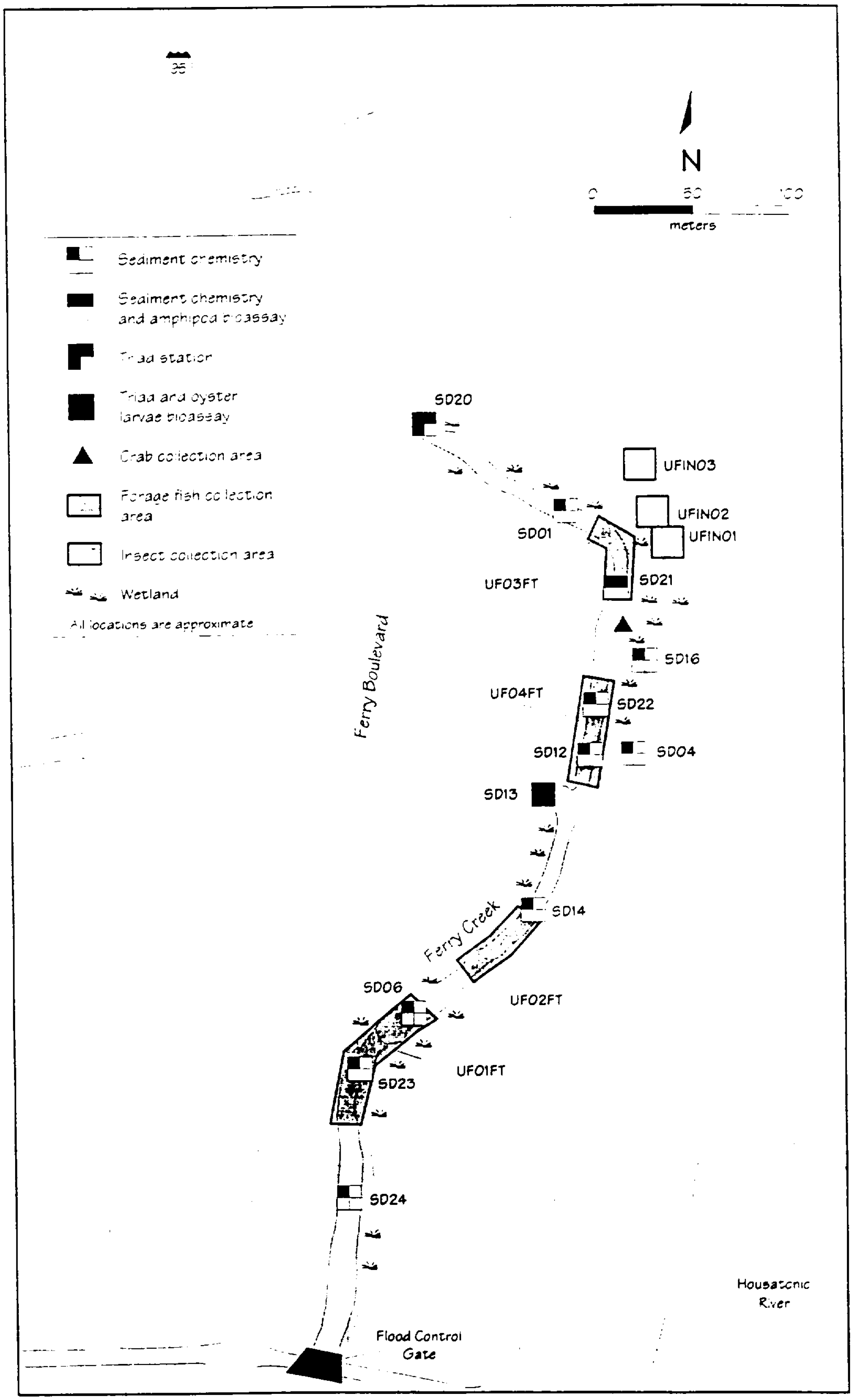

Figure 4-1. Locations of sampling stations in Upper Ferry Creek area.

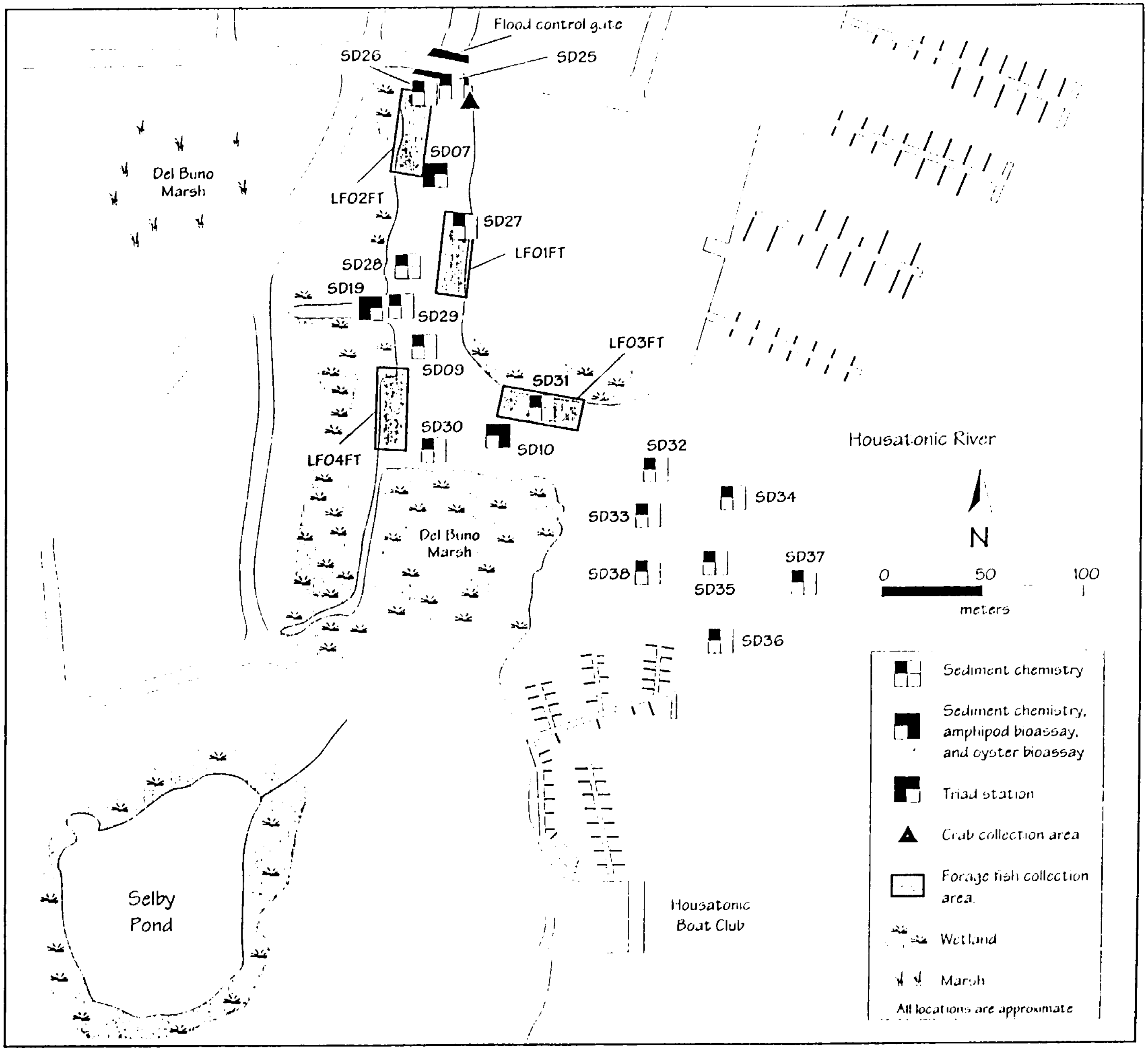

Figure 4-2. Locations of sampling stations in the Lower Ferry Creek area.

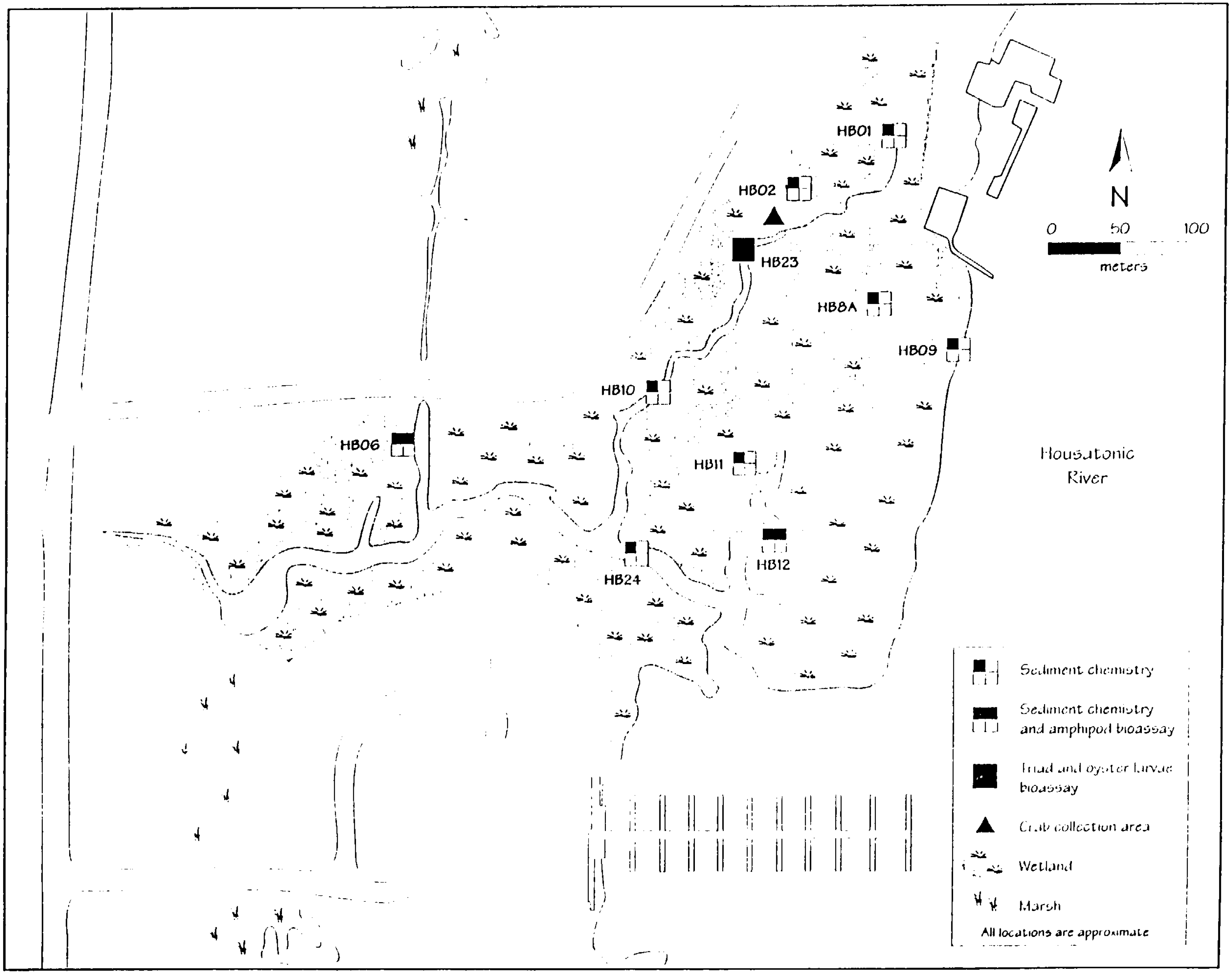

Figure 4-3. Locations of sampling stations in the Housatonic Boat Club wetlands area.

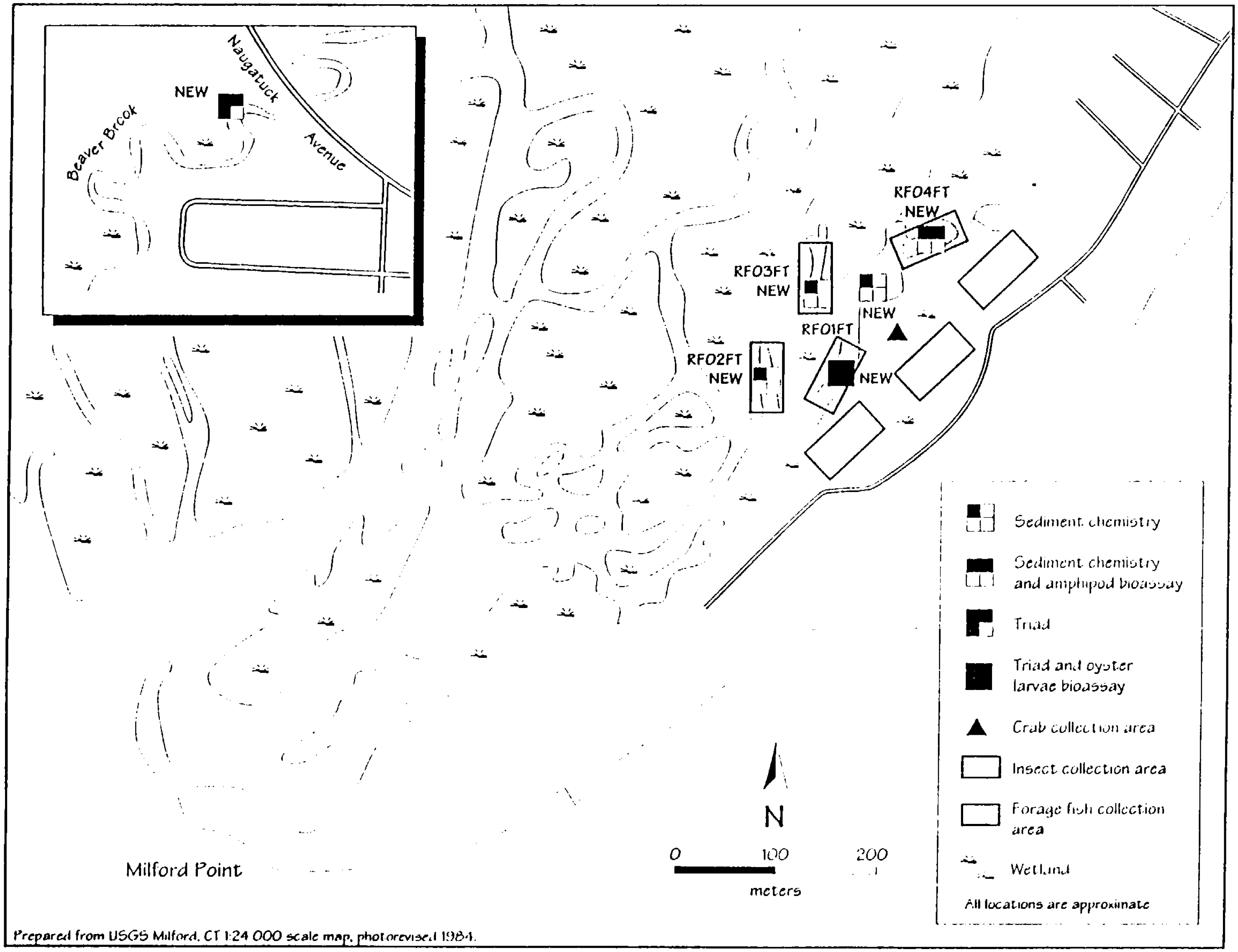

Figure 4-4. Locations of sampling stations in the reference zone areas of Milford Point and Beaver Brook (inset).

Table 4-1. Number of sampling stations and samples collected per media per area.

Media	Upper Ferry Creek	Lower Ferry Creek	Housatonic Boat Club	Reference
Sediment chemistry	12	12	10	6
Benthic community	2	2	1	2
Amphipod bioassays	3	3	3	3
Oyster bioassay	1	1	1	1
Fish tissue	4	4		4
Crab tissue	1	1	1	1
Insect tissue	3 composited			3 composited

range of CoC concentrations in sediment (Table 4-1; Figures 4-1 through 4-4). Toxicity testing with sediment samples was conducted with the estuarine amphipod, *Leptocheirus plumulosus*.

A standard 10-day static bulk-sediment toxicity test protocol (Method E1367-92 of the American Society for Testing and Materials; ASTM 1994a) was followed. In addition to the seven triad stations, five additional stations were sampled concurrently for amphipod toxicity and chemical content to provide three amphipod toxicity measurements per area. Oyster larval toxicity tests were conducted using sediments with a range of CoC concentrations to assess the potential risk to the oyster spat beds. The standard 48-hour larval toxicity-test protocol was followed (ASTM Method E724-89; ASTM 1994b). It was hypothesized that substantial masses of CoCs in Raymark waste and sediment in Ferry Creek and the Housatonic Boat Club wetlands historically and in the future could be mobilized and released to the river. If this release occurred at a critical time in larval development, oyster recruitment to the spat beds could be reduced. Three site-related stations, one from each area and one reference station, were selected for oyster larval toxicity testing (Table 4-1). One station was near the mouth of a highly contaminated ditch entering Ferry Creek. This station was selected to represent a worst-case event, such as the occurrence of a large storm flushing this sediment to the mouth of Ferry Creek (Figure 4-1). The second station was near the mouth of Ferry Creek, close to the oyster spat beds (Figure 4-2). The third station was in the Housatonic Boat Club wetlands at a location slightly less contaminated than the Upper Ferry Creek station (Figure 4-3).

Benthic community structure was assessed, identified to the lowest reasonable taxon, based on four samples per station at each of the seven sediment triad stations. Comparative measures of benthic community structure include species richness, total abundance, abundance of major groups, absence of sensitive taxa, and presence of pollution-tolerant species. The stations were selected based on an expected range of CoC concentrations, plus expected similarities in salinities, grain size, and TOC. Two reference area stations were chosen to cover a salinity gradient: one station was located in Beaver Brook, south of Naugatuck Avenue, as representative of a low-salinity site; and one station was at a higher-salinity area along Milford Point (Figure 4-4).

4.1.2 Sampling Methods

Haliburton NUS (HNUS) Corporation (under contract to the EPA to conduct the RI) was responsible for laboratory arrangements for chemical and physical analyses of sediment samples. Environment Consultants, Inc. (EVS) was responsible for laboratory arrangements for toxicity testing of sediment samples. Sediment sampling was conducted in August 1995 according to the HNUS sampling and analysis plan (SAP; HNUS 1994) with the following exceptions or additions:

- Sediment samples for chemical analysis and toxicity testing were collected using a stainless-steel Eckman grab sampler following sampling protocols detailed in ASTM Method D4343-84 (ASTM 1993).

- After collecting a grab sample, water present on the surface of the grab was siphoned off before the sediment sample was collected.

- After collecting the first grab, subsamples of sediment were removed for volatile organic compound (VOC) and SEM/AVS analyses. These subsamples were placed in appropriate containers, leaving zero head space in the containers after filling. The remaining sediment from this grab was composited with sediment from subsequent grabs to obtain the volume of sediment necessary for the chemical analyses and bioassays. Sediment from these grabs was placed in a stainless-steel mixing bowl and homogenized with a stainless-steel spoon to achieve a uniform texture and color before transferring to the appropriate sample containers.

Sediment samples for benthic community analysis were collected using a stainless-steel Eckman grab sampler. Four grab samples were collected but were not composited. For each grab, the sample was sieved using creek water. The sample was comprised of the portion retained on a 0.5-mm mesh screen. Any debris remaining on the screen was also retained for examination.

Procedures for collecting, handling, and storing sediment chemistry samples are described in detail in the HNUS SAP (HNUS 1994). Field quality assurance samples were collected as described in the HNUS SAP (HNUS 1994). All collection, handling, storage, and analysis procedures for benthic community samples are described in detail in the Work Plan and Field Sampling Plan for the Raymark ERA (EVS 1995). All sediment samples were analyzed for CoCs, SEM/AVS, TOC, and grain size as described in the HNUS SAP (HNUS 1994).

4.1.3 Bioassay Testing Methods

The short-term amphipod toxicity test was conducted by exposing *Leptocheirus plumulosus* to nine sediment samples collected in the study area and three reference sediment samples for 10 days, according to procedures and quality assurance/quality control (QA/QC) performance standards described in ASTM Method E1367-92 (ASTM 1994a). Test quality control is described in detail in the Work Plan and Field Sampling Plan for the Raymark ERA (EVS 1995). Laboratory endpoints for the amphipod test were survival and avoidance.

The oyster larvae toxicity test was conducted by exposing *Crassostrea gigas* larvae to three site-related sediment samples collected in the study area and one reference sediment for 48-hours according to procedures and QA/QC performance standards described in ASTM Method E724-89 (ASTM 1994b). The work plan indicated that this test would be conducted using the eastern oyster species, *C. virginica*; however, viable spawning adults of this species were not available when the test was conducted. This issue is discussed later in the ERA. Quality control for the oyster larvae test is described in detail in the Work Plan and Field Sampling Plan for the Raymark ERA (EVS 1995). Measurement parameters for the oyster larvae test were mortality and abnormal development. Combined mortality is calculated as the sum of these two, since abnormal larvae are presumed not to be viable.

Benthic invertebrate samples were sorted and identified to their lowest practical taxonomic level, following QC procedures outlined in the Work Plan and Field Sampling Plan for the Raymark ERA (EVS 1995).

4.1.4 Data Analysis

Amphipod bioassay data were statistically analyzed using a single-factor analysis of variance (ANOVA) followed by Fisher's positive, least-significant-difference (PLSD) multiple-comparisons procedure to identify the differences between stations located in the areas of interest and the reference area. The multiple-comparisons procedure included a correction for tied ranks and unequal sample sizes, when applicable. The purpose of this comparison was to identify those individual samples which, presumably due to their content of CoCs, had responses significantly different than those observed in the reference sample. Individual samples were identified as "toxic" by virtue of diminished survival. Statistical comparisons were made between mean responses observed in the five laboratory replicates of each single sample versus those observed in the appropriate reference area sample(s). Because the purpose of these tests was to identify samples indicative of adverse impact to biota (i.e., samples with average performance worse than the reference station), the post-ANOVA analyses used one-tailed tests. Tests of normality and heterogeneity of variance were performed by the laboratory.

Because of the potential effect of grain size on amphipod survival, the amphipod results were first partitioned based on percent fines. The samples with grain size less than or equal to 51% fines (i.e., those from stations HB06, SD20, SD21, and SD07) were grouped with the sample from the reference station RF01 (with fines of 51%). The samples with grain size greater than 51% fines (i.e., those from stations SD10, HB12, HB23, SD19, SD13) were grouped with the samples from the reference stations RF02 and RF03 (with 64% and 65% fines, respectively). In the latter case, the two reference stations were pooled as one group within the Kruskal-Wallis and multiple-comparisons test; in this way, the reference group is more representative of the reference area. This pooling of the reference stations also increases the statistical power of the multiple-comparisons tests because of the larger sample size of the reference group relative to the other stations.

Non-parametric statistical comparisons (Kruskal-Wallis with multiple contrasts) were conducted of mean survival and avoidance among the three stations each in Upper and Lower Ferry Creek, and the boat club wetlands versus the reference stations. The purpose was to determine whether overall, site-related responses were statistically significantly different from the reference.

For the oyster larvae test, statistical comparisons were made between mean responses observed in the five laboratory replicates of the single sample from each site-related area versus those observed in the controls, and also against the means from the reference area sample. The purpose of this comparison was to identify those samples which, by virtue of their content of CoCs, had toxic responses statistically significantly different from those observed in the controls. A Students t-test was also performed between mean responses observed in the three site-related samples as one population (i.e., Upper and Lower Ferry Creek and the Housatonic Boat Club wetlands) versus the value of the mean response observed from the reference sample. This test was to further address the question of whether site-related sediments were capable of causing adverse ecological impacts. Tests of normality and heterogeneity of variance were performed by the laboratory.

Benthic community indices were statistically analyzed using a single-factor ANOVA followed by Fisher's PLSD multiple-comparisons procedure to locate the differences between the site-related stations and the reference station. Comparisons were made with the appropriate high- or low-salinity reference station. Stations were identified as impacted on the basis of one-tailed tests indicating depressed diversity.

For the avian food-web model, estimates of central tendency of sediment concentrations of CoCs were required. The purpose of this exercise was to assess whether the arithmetic mean was a good measure of central tendency for the distribution. To determine the distribution of contaminants in sediments, the probabilistic form of the data is required. Since summary statistics were required for the data in various measurement bases and normalizations (sediments in both wet weight and TOC normalized dry weight), the distribution of the data was evaluated for each case. A combination of Quantile-Quantile (QQ) plots and histograms were used in the evaluation. To maximize the number of samples available for this assessment, the data were first standardized within each set (by subtracting the mean and dividing by the standard deviation) and then pooled across all areas. The resulting population of 46 sediment samples was compared with standard normal and lognormal distributions via the QQ-plots. This analysis is based on the assumption that the underlying processes affecting the distribution of contaminants are the same in all areas, and that only the location and scale of the distribution, and not the form itself, will vary among the different areas.

To determine the appropriate measure of central tendency, the distributions of the data were assessed based on the linear relationship in the QQ-plots and the pattern of deviation from a straight line. Even samples (particularly those of size <20) generated from a known normal distribution can show a fair degree of variability in the linearity of a QQ-plot. In environmental data, large numbers of non-detecteds in the dataset create a skewed, censored distribution (heavier left tail) than would be present in a normal distribution. Similarly, high concentrations from several hot spots can create a heavier right tail than expected. Professional judgment was used to determine whether or not the deviance from the hypothesized distribution was meaningful. Plots that showed either an S-shape (indicating heavier tails, or fewer observations around the mean than expected under normality) or reverse S-shape (indicating a distribution with a heavier center, or more observations around the mean than expected under normality) were not considered a valid basis on which to reject normality. The purpose of this exercise was to assess whether the arithmetic mean was a good measure of central tendency for the distribution; hence, a distribution that was basically symmetrical and unimodal, even if its proportions varied slightly from that of a normal sample, was generally accepted as approximately normal. When necessary to clarify the extent and nature of deviation from normality, histograms were also reviewed. When normality was rejected, the data were log-transformed and normality assessed using the QQ-plots on the transformed data. The conclusions from this investigation were that the CoC concentrations in sediments are all approximately lognormally distributed.

Only the individual contaminants detected in more than 20% of the samples were assessed for the underlying distributional form. For contaminants not meeting this criterion, the distributional form was not estimated and maximum values were used as input to the components of the ERA. Because of very high detection limits in some samples (sometimes exceeding detected values in other samples), one-half the detection limits were used as input to the QQ-plot analysis and in the calculation of total PCBs, total PAHs, and total DDTs. For the low-end detection limits, this effectively pulls concentrations down below what may be a reasonable level. However, it was necessary to dampen the impact of the very high detection limits because of their strong influence on the distributions. Because the elevated detection limits and their treatment represent a source of uncertainty associated with the laboratory precision, they should not be given as much weight as detected values in the data set.

The conclusion of lognormal distribution for sediment contaminant concentrations indicates that the geometric mean was a suitable measure of the central tendencies of the distribution. Consequently, the maximum value and a 95% upper confidence limit on the geometric mean were computed. In each case, a critical value from the t-distribution was applied to account for the small sample sizes within each area, and the variance was estimated from the sample.

4.2 FISH BIOACCUMULATION

Bioaccumulation of site-related CoCs in fish tissue confirms bioavailability of the CoCs and represents potential food-web transfers of these CoCs. When the levels of these CoCs in the fish are sufficiently elevated, their presence is also a risk to the fish themselves.

4.2.1 Sampling Objectives

Mummichog were collected to compare CoC concentrations in their tissue to MATCs to assess risk to the fish themselves. Fish tissue data were also collected to estimate exposure to the black-crowned night heron. Four composite samples were collected by seining at stations in each of three areas (Upper Ferry Creek, Lower Ferry Creek, and the reference area), for a total of 12 composite samples. No mummichog collection stations were located in the Housatonic Boat Club wetland since it drains completely during low tide.

An additional objective of the fish sampling study was to determine the relationship between the concentrations of CoCs in all three mummichog collection areas and the respective concentrations of CoCs in sediments sampled within those areas. This relationship, known as the biota sediment-accumulation factor (BSAF) links CoC concentrations in tissue and sediment and helps define protective goals for the assessment endpoints related to these data. Each of the three areas sampled represents a different habitat for the mummichog. Locations of seine stations were based partially on an expectation (based upon data collected during previous samplings) of the fish being exposed to a gradient of sediment contamination.

Limited historical data on CoC concentrations in white perch were collected during past studies (US EPA unpubl. 1994). Measurements included concentrations of Cd, Pb, Hg, Ni, PCBs, and DDT in offal of white perch collected from Frash Pond and Selby Pond in Stratford, Connecticut. These data were compared with MATCs to assess potential impacts to large, predatory fish.

4.2.2 Sampling Methods

Mummichog were collected using a two-person minnow seine. Three to five seine hauls were conducted at each fish sampling station identified in Figures 4-1, 4-2, and 4-4, or enough hauls to produce sufficient tissue volume for analysis. Only mummichog >40 mm long (those assumed to be adults) were retained for analysis. Each sample consisted of a composite of 30 to 60 fish, depending on the number of fish needed to obtain at least 100 g in wet weight. Total length was measured to the nearest tenth of a millimeter, and body weight was measured to the nearest centigram. Reproductive status of fish was noted. Fish were also examined for gross lesions. Field QA procedures used are described in detail in the Work Plan and Field Sampling Plan for the Raymark ERA (EVS 1995).

Whole body concentrations of CoCs and percent lipids were determined in fish. Analytical methods are described in detail in the Quality Assurance Project Plan, Appendix B of the Work Plan and Field Sampling Plan for the Raymark ERA (EVS 1995).

4.2.3 Data Analysis

To determine which summary statistic best describes the distribution of contaminants in fish tissues, the probabilistic form of the data was evaluated. Summary statistics were required for two types of measurements (wet weight and normalized to lipid content), thus the distribution was evaluated for each case. The same combination of QQ-plots and histograms used in the evaluation of sediment data were used to evaluate the distribution of fish tissue concentrations.

The conclusion from this evaluation is that the fish-tissue body burdens are all approximately normally distributed. The fish-tissue body burdens in some cases showed a fair amount of variability in the straightness of the QQ-plot; however, because of small sample sizes, the presence of repeated values (as detection limits), and the lack of extreme high-end values, these deviations were not considered meaningful enough to reject normality.

The conclusion of normal distributions for fish tissue indicate that the arithmetic mean was a suitable measure of the central tendencies of the distributions. Consequently, the maximum value and a 95% upper confidence limit on the arithmetic mean were computed. In each case, a critical value from the t-distribution was applied to account for the small sample sizes within each area and the fact that the variance was estimated from the sample.

4.3 *AVIAN EFFECTS*

Birds with two different feeding strategies were viewed as having potential risk from dietary exposure to site-related CoCs, insectivores, and piscivores. These feeding strategies may result in greater exposure to contaminated sediment than those that may occur with birds' other feeding habits. Red-winged blackbirds that nest in and near the study area were chosen to represent insectivores, while black-crowned night herons, which feed primarily on fish and crustaceans and defend their feeding territory, were chosen as representative of piscivorous birds. The goal of the field sampling for the avian-assessment endpoint was to provide data on CoC concentrations in their diet.

4.3.1 Sampling Objectives

Samples of insects, fish, and fiddler crabs were collected and analyzed for CoC body burdens to estimate doses of contaminants to herons and blackbirds. During the breeding season, red-winged blackbirds are almost exclusively insectivores, whereas herons typically ingest fish and benthic invertebrates. The exposure from ingestion of contaminated prey was compared with RTVs to calculate the potential risk for reproductive impairment and other adverse effects.

Collection of the four mummichog composite samples per area is described in Section 4.2.2. These composite fish samples represent prey items that would potentially bioaccumulate a range of CoC concentrations from sediment and across areas where herons forage. Fiddler crab were collected at single stations in each area. Stations were located to represent a range of contaminant concentrations in sediment and various habitat conditions. Stations were located in Upper Ferry Creek, in Lower Ferry Creek, in the Housatonic Boat Club wetlands, and at the reference area (Figures 4-1, 4-2, 4-3, and 4-4). Individual crabs were composited into one sample from each station and the samples were analyzed for the target CoCs (Table 3-1).

Samples of the insect community were collected from two locations: the wetlands adjacent to Upper Ferry Creek and the Milford Point reference area (Figures 4-1 and 4-4). The sampling design was originally intended to analyze for CoCs in multiple, composite samples of insect tissue separately from each area. However, despite intensive sampling effort, the sample sizes obtained were insufficient to submit multiple composite samples for all chemical analyses with suitable detection limits. Therefore, samples from within each area were composited to form a single sample representative of that entire area. These two samples were then analyzed for all targeted CoCs. Sample numbers, locations, and types are summarized on Table 4-1.

4.3.2 Sampling Methods

Fiddler crabs were collected by hand from within about 50 m of the designated sampling locations on Figures 4-1 through 4-4. Enough crabs were collected at each sampling location to achieve the 100 g of sample material required for analysis. At least 40 individuals were composited in each sample from each area. Two different species were observed and collected in the different habitats: *Uca pugnax* was collected from Lower Ferry Creek, the Housatonic Boat Club wetlands, and the reference area; whereas, *Uca minax* was collected from the Upper Ferry Creek area.

Insects were collected by sweep-netting vegetation within each sampling area using standard insect collection nets. All species of insects were retained. Attempts were made to collect insects using portable ultraviolet light traps at night, but were unsuccessful. After insects were captured, they were transferred to clean containers and killed with ethyl acetate vapors.

Analytical methods for tissue are described in detail in the Quality Assurance Project Plan, Appendix B of the Work Plan and Field Sampling Plan for the Raymark ERA (EVS 1995).

Sections:
5.0 Avian Food-Web Model
6.0 Exposure Assessment
(pages 57-104)
are available
in a separate file (size: 3 MB).

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Section:
7.0 Effects Assessment
(pages 105-134)
is available
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Sections:
8.0 Risk Characterization
9.0 Uncertainty Assessment
10.0 References
11.0 Acronyms
Appendices A & B
(pages 135-177)
are available
in a separate file (size: 3 MB).

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Appendix C &
Draft Evaluation of Raymark Superfund Data
for PRG Development
(pages 178-232)
are available
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Draft Evaluation of Raymark Superfund Data
for PRG Development Continued
(5.0 References & Figures and Tables)
(pages 233-285)
is available
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Final Report:
Evaluation of Ecological Risk to Avian & Mammalian Receptors
in the Vicinity of Upper & Middle Ferry Creek,
Stratford, CT, Sept.1999
(pages 286-339)
is available
in a separate file (size: 4 MB).

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Appendix E:
Ground Penetrating Radar Survey
(pages 340-370)
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