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GE Pittsfield
MAD002084093
R-9

February 28, 1995

Ms. J. Lyn Cutler
Section Chief, Special Projects
Bureau of Waste Site Cleanup
Department of Environmental Protection
436 Dwight Street
Springfield, MA 01103

Mr. Bryan Olson
U.S. Environmental Protection Agency
Waste Management Division
J.F. Kennedy Federal Building
HRR-CAN3
Boston, MA 02203

Re: DEP # 1-0147P; EPA Area 6
Housatonic River Site
Work Plan for Ecological Risk Assessment

Dear Ms. Cutler and Mr. Olson:

As discussed at our risk assessment meeting last October, GE's risk assessment consultants have prepared, on our behalf, a Work Plan for the Ecological Risk Assessment of the Housatonic River Site. Copies of that work plan are enclosed. The work plan proposes to rely in part on existing studies and also to conduct a number of additional studies in order to complete an ecological risk assessment for the Housatonic River and its floodplain.

One issue of particular note relates to the proposed schedule. As we advised you at our meeting last October, the proposed studies would need to begin by about April 1 in order to complete the majority of activities in 1995. This is due to seasonal constraints on several of the proposed activities. For example, the proposed fish reproductivity study requires that samples of white sucker and largemouth bass be collected during their breeding seasons (April and May, respectively). Similarly, the frog reproductivity study requires collection of frog samples in the spring, during their breeding season. As another example, the proposed study of great blue herons requires that our consultants work with the Massachusetts Division of Fish and Wildlife during their next regularly scheduled survey of heron hatching success,

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which is scheduled for June 1995. Other proposed studies are also seasonally dependent, as discussed in the work plan.

For these reasons, the schedule set forth in the work plan has been developed on the assumption that the Agencies' approval of the work plan is obtained by April 1. We realize that this is a very short time for review, particularly given the number of other documents pending with the Agencies. Unfortunately, if the Agencies' review is not completed until later, it will be necessary to develop and submit a wholly revised schedule. The current schedule cannot simply be shifted to later months in the same order due to the seasonal constraints on particular activities.

Please let us know how you would like to handle the review of this work plan. We would like to meet with you and the Agencies' ecological risk assessment personnel to discuss this work plan. We are available for such a meeting when the Agencies are ready to do so.

We look forward to discussing this work plan with you.

Sincerely yours,

John D. Ciampa /ese

John D. Ciampa
Project Manager

Enclosure

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WORK PLAN FOR THE ECOLOGICAL RISK ASSESSMENT OF THE HOUSATONIC RIVER SITE

TABLE OF CONTENTS

WORK PLAN FOR THE ECOLOGICAL RISK ASSESSMENT OF THE HOUSATONIC RIVER SITE

TABLE OF CONTENTS (CONT'D)

WORK PLAN FOR THE ECOLOGICAL RISK ASSESSMENT OF THE HOUSATONIC RIVER SITE

LIST OF FIGURES

LIST OF TABLES

ChemRisk®
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Louis Bolduc, Pittsfield Commissioner of Public Health (w/o encl.)
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Lenox, Lee, Stockbridge, Great Barrington, Sheffield (w/o encl.)
Housatonic River Initiative (w/o encl.)
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Work Plan for the Ecological Risk Assessment of the Housatonic River Site

February 28, 1995

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1.0 INTRODUCTION

This document constitutes a work plan for the performance of an ecological risk assessment for the Housatonic River site. It has been developed, on behalf of the General Electric Company (GE), by ChemRisk, the risk assessment division of McLaren/Hart Environmental Engineering, in conjunction with Jenkins, Sanders & Associates, S.G. Martin & Associates, Inc., Chadwick & Associates, and ENVIRON. This work plan has been developed for submission to the U.S. Environmental Protection Agency (EPA) and the Massachusetts Department of Environmental Protection (DEP), sometimes referred to jointly herein as the "Agencies."

The purpose of this risk assessment will be to evaluate whether a significant risk of ecological harm may be associated with exposure of ecological receptors to chemicals in soil, sediment, and surface water of the Housatonic River site that are related to past or present activities at the GE facility in Pittsfield, MA. As defined by the EPA (1992), ecological risk assessment is a process that evaluates the likelihood that adverse ecological effects may occur or are occurring as a result of exposure to one or more stressors. While ecological risk assessments cannot provide absolute proof of adverse impacts or lack thereof (EPA, 1989a), they can provide the quantitative basis for comparing and prioritizing risks, as well as a systematic means of improving the understanding of risks (Suter, 1993). They can be used to help identify environmental problems, establish priorities, and provide a scientific basis for regulatory actions (EPA, 1992).

This work plan describes GE's plan for evaluating potential risks to ecological receptors in the Housatonic River valley that may be presented by chemical releases from the GE facility. The ecological risk assessment for the Housatonic River site will generally follow the approach outlined by the EPA in its 1992 *Framework for Ecological Risk Assessment*, as well as the approach outlined in the Massachusetts Contingency Plan (MCP) (DEP, 1995) for environmental risk characterization. In accordance with the EPA (1992) framework, any physical, chemical, or biological entity that can induce an adverse response is referred to as a stressor in this work plan. The work plan provides a summary of the scientific literature related to potential effects of those stressors on environmental receptors, identifies receptors and endpoints of most importance based on literature studies and site-specific conditions, summarizes studies that have been conducted to date to evaluate the ecosystem, and proposes tasks to collect additional data in order to more fully characterize potential ecological risks.

An ecological risk assessment may be predictive or retrospective. Predictive assessments are generally performed in advance of the introduction of a potential stressor to an ecosystem, such as while a site is being selected for the construction of a waste disposal facility or in advance of a permitted discharge. Predictive assessments generally require the application of mathematical models of food webs to simulate the transfer of a chemical from sediment, soil, or water to lower levels of the food web and from these lower trophic levels to higher trophic levels. This modeling approach tends to focus on potential effects on individual organisms and assumes that effects predicted to individuals can result in effects at the population or community level. In contrast, retrospective assessments are conducted for sites where stressors were historically introduced, such as river reaches downstream of industrial facilities. At such sites, it is possible to use field observations and community-level data to determine if wildlife are adversely affected by the stressor.

While modeling is a useful tool for predicting potential effects, it generally requires substantial extrapolation. Due to the fact that specific data are generally not available for the species of interest, the toxicological data used are usually derived from studies of laboratory animals which must then be extrapolated to natural species. Interspecies extrapolations are known to introduce considerable uncertainty into an analysis. In addition to differences between species, the tremendous variation in natural populations compared with their laboratory counterparts will oftentimes limit the applicability of laboratory results to field situations. Furthermore, conditions in the wild vary considerably from laboratory conditions due to competition, habitat variability, and predation, and, as a result, laboratory data which are obtained under artificial conditions may be of limited use in predicting what will actually occur in the natural ecosystem. In addition, because the input data on toxicity that are required by such models are very limited and are not compiled in a single guidance document or database, it is necessary to conduct a comprehensive literature review, compiling a range of toxicity criteria. These studies and data must then be critically evaluated for applicability to the receptors and stressors of interest. In the absence of toxicity criteria handbooks, criteria for selecting input data and the values themselves may vary substantially among risk assessors.

Food web models are limited in their ability to accurately simulate varied food sources for different species and for individuals within a species (SAB, 1994). Typically, modelers may input the fraction of the diet assumed to be derived from a limited number of locations (e.g., contaminated vs. uncontaminated) and/or food group (e.g., vegetation, invertebrates, small mammals). Usually, these proportions are assumed to remain constant over time and to be uniform for all individuals within a species. In reality, the diets of most free-ranging individuals are extremely variable. Because models cannot adequately simulate the dynamics of feeding behavior, considerable uncertainty is introduced with respect to the transfer of chemicals up the food web. Conservative assumptions regarding diet are generally used to accommodate this uncertainty; however, the result is almost certainly an overprediction of potential exposure.

It is difficult, if not impossible, to adequately simulate the effects of both competition and predation in the natural environment. The fundamentals of population ecology assert that the health of a community is a function of all interactions within and among species (Begon and Mortimer, 1986), as well as of physical and chemical stressors. The primary types of interactions among species are competition and predation. While predation is often the focus of food web models, competition also plays a critical role in the utilization of limited resources, such as food, water, and breeding territory. By neglecting competition, food web models may substantially over- or underestimate actual exposure. Finally, modeling procedures cannot account for synergistic or antagonistic effects of more than one chemical stressor. Instead, the risk assessor is left with only two options: either consider the effects of two or more stressors to be additive, or ignore the potential effects of the secondary stressor. Either option can lead to underestimates if synergy is present.

In contrast with predictive modeling, the retrospective procedure ascertains whether diversity, density, and reproductivity of wildlife living in contaminated areas lie within normal ranges as reported in the literature, or whether they compare favorably to populations inhabiting uncontaminated reference areas. By monitoring the actual ecosystem potentially at risk, this approach determines whether wildlife are healthy and if the ecological system on which they depend is functional. However, a number of confounding factors can limit the effectiveness of the retrospective approach. For example, this process may not be sensitive enough to link adverse effects with specific environmental conditions, due to the natural variability and variety of additional stressors that may be present in a given ecosystem. In addition, it may not be able to provide insight into potential effects to receptors with low population densities or very large home ranges.

Given the high level of uncertainty associated with predictive modeling, such modeling will not be undertaken in this ecological risk assessment. At the same time, recognizing the potential insensitivities in the retrospective approach, several types of retrospective studies will be undertaken in order to develop multiple lines of evidence. Some of these studies will focus on evaluating biotic communities, while others will be aimed at assessing specific toxic effects on critical endpoints, notably reproductive success, for certain species.

This assessment will focus on the potential effects on receptors of chemical stressors in materials deposited in surface waters, sediments, and flood plain soils of the river system. Consistent with the MCP, EPA (1989a,b; 1994a,b), Suter (1990, 1993), Maughan (1993) and Burger (1994), this assessment will focus on adverse effects that are of biological or ecological significance at the subpopulation, community, or system-wide level. In most cases, this will be done by direct measurement of endpoints at the subpopulation, community, or system-wide level. However, in cases where direct measurements of population effects are not feasible, effects such as reproductive success, which are relevant to the population as a whole, will be evaluated.

1.1 Purpose and Objectives

The purpose and objectives of the tasks outlined in this work plan will be to determine whether historical releases from the GE facility to the Housatonic River site have or could be expected to adversely affect terrestrial or aquatic organisms which have potential for exposure to releases. A risk will not exist unless the stressor of concern is capable of causing an adverse effect and an ecological receptor has the opportunity to co-occur or come into contact with that stressor for a sufficient length of time and at a sufficient concentration to elicit the effect (EPA, 1992).

The risk assessment will identify those ecological receptors that have a significant potential for exposure to chemical stressors released from the GE facility. In addition, it will identify, based on the most current ecotoxicological literature, those taxonomic groups that are expected to be particularly sensitive to the effects of those chemical stressors, and the toxicological endpoints most likely to be of concern for those receptor groups. It will outline the linkages between receptors, their routes of exposure, and the critical effects (if any) resulting from those exposures. In addition, it will outline the linkages between measurement and assessment endpoints. Finally, it will provide the linkage between assessment endpoints and determination of risk of harm and will use a weight-of-evidence approach to determine whether there is a significant risk of harm to the ecosystem of the Housatonic River site.

1.2 Approach

Consistent with the approach suggested by EPA's *Framework for Ecological Risk Assessment* (EPA, 1992), the basic framework of organization followed for this evaluation will consist of three phases: (1) problem formulation; (2) data collection and analysis; and (3) risk characterization. The following descriptions of the components and objectives of each of these three phases are largely excerpted from EPA's (1992) *Framework Document*.

Problem Formulation

Section 2.0 of this work plan presents the problem formulation phase of the risk assessment. While this document is intended to provide a work plan for completing the ecological risk assessment, problem formulation must be considered early in the process to establish the goals, breadth, and focus of the risk assessment and to identify sampling needs. Problem formulation is essentially a planning process that identifies and describes the major factors to be considered in the assessment, such as the chemical and/or physical stressors of interest, their behavior in the environment, and their sources, distributions, and concentrations. It also identifies the ecosystem potentially at risk, its geography, topography, climate, vegetation, habitats, and species composition. Based on this information, endpoints - measurable responses of the ecosystem to the stressor(s) - are selected for study. The end product of the problem formulation stage is a conceptual model that identifies characteristics of the ecosystem to be monitored and data needed to evaluate those endpoints.

Data Collection and Analysis

During the data collection and analysis phase, the ecological data necessary for evaluating those endpoints selected during problem formulation are collected and analyzed. These data are critically evaluated in an effort to relate exposure to the stressor(s) and understand potential ecological effects. Based on the available data, an exposure profile is developed to characterize potential exposure of biota to the stressor(s). To the extent possible, the exposure profile characterizes the pathway, location, and timing of exposure events. In addition to the exposure profile, an ecological effects profile is developed during the analysis phase in which the observed or predicted effects of the stressor(s) are described. In addition to describing the actual effects observed or predicted, evidence of a causal relationship between the stressor(s) and the effects is evaluated.

Section 3.0 of the work plan discusses the data collection and analyses planned for the ecological risk assessment. It provides a summary of studies that have been conducted to date for the receptors and endpoints selected in the problem formulation stage, and outlines additional tasks which are proposed to fill data gaps and more fully characterize the ecosystem of the Housatonic River site.

Risk Characterization

The risk characterization phase develops quantitative or qualitative estimates of risk by integrating the exposure profile and effects profile. Risks are described for each endpoint and the overall ecological impact is determined by weight-of-evidence. In perhaps the most critical element of the risk assessment, the ecological significance of the predicted or observed effects is discussed. Finally, the risk characterization phase analyzes the uncertainty associated with each element of the assessment and summarizes overall confidence in the conclusions.

Section 4.0 of the work plan presents the approach that will be used to complete the risk characterization phase of the assessment. It discusses how the results of analyses completed during the data collection and analysis phase will be integrated and weighed in order to determine whether releases from the GE facility have or could be expected to adversely affect the river ecosystem.

1.3 Description of the Housatonic River Site

While the Housatonic River Site is described in detail in Section 2.0, the following description provides a brief introduction to the overall site. The Housatonic River is formed at the confluence of the West and East Branches of the Housatonic River in Pittsfield, Massachusetts. It flows south through western Massachusetts for approximately 49 river miles, where it enters the state of Connecticut approximately one mile north of Canaan, Connecticut. From there it continues an additional 83 miles south through Connecticut to Long Island Sound. A number of tributaries enter the river as it flows southward, including Hop Brook, the Williams River, the Green River, and Schenob Brook in Massachusetts, and the Ten Mile, Still, Shepaug, Pomperaug, and Naugatuck Rivers in Connecticut (Blasland & Bouck, 1991).

Land use on the flood plains of the Housatonic River is variable. Between the GE facility and Elm Street Bridge, land use is primarily urban, characterized by either commercial, industrial or undeveloped urban land with some high density urban neighborhoods composed primarily of one- and two-family homes. As one moves downstream towards Holmes Road, the land use is primarily residential with population density decreasing downstream. Below Holmes Road, the predominant land uses are forested land, wetlands, and agricultural land. Much of the forest and wetlands are owned and maintained by the Massachusetts Department of Fisheries and Wildlife (MDFW) as the Housatonic Valley Wildlife Management Area (HVWMA).

To the west of the river, in the area between New Lenox Road Bridge and Woods Pond Dam, there is a large tract of undeveloped land that is owned and maintained by the MDFW. This area is composed primarily of forest or wetlands, with a small number of shrub meadows. No developed roads or residential properties are included in this area. To the east of the River below New Lenox Road is October Mountain Road and the October Mountain State Park. There are a small number of residences along this road. South of Woods Pond, the river meanders through a number of small towns. Development in this area is primarily residential in the upper area and agricultural as the river moves towards Connecticut.

Sediment and flood plain sampling to date have indicated that polychlorinated biphenyls (PCBs), notably Aroclor 1260, constitute the primary chemical of concern. PCBs have been detected in surface water (at very low levels) and in sediments, flood plain soils, and biota of the river.

Sampling of flood plain soils indicates that the extent of soils with PCB levels higher than 1.0 ppm is generally contained within the approximate 10-year flood plain of the river and is for the most part found in the river stretch between the GE facility and Woods Pond Dam (Blasland & Bouck, 1991, 1992a). Similarly, the highest PCB sediment concentrations are found in depositional areas between Pittsfield and Woods Pond Dam, with the highest consistent concentrations detected in the area between New Lenox Road and Woods Pond Dam. PCBs have also been detected in the tissues of fish obtained from Woods Pond, Rising Pond and other downstream reaches of the river.

Although the Housatonic River site also encompasses Silver Lake, which is located adjacent to the GE facility, this work plan does not address Silver Lake. Historically, Silver Lake has received stormwater runoff from several municipal outfalls in addition to treated process water, non-contact cooling water, and stormwater from a portion of the GE facility (Blasland, Bouck & Lee, 1994a). As a result, Silver Lake contains more constituents at generally higher concentrations than those found in the Housatonic River. In addition, the ecosystem of Silver Lake is considerably different from that of the Housatonic River. Its location within a highly urbanized section of Pittsfield results in essentially no substantial habitat to support terrestrial wildlife. Thus, while it is possible that some birds could occasionally feed from the lake, it is unlikely, due to the high level of disturbance and human activity, that these birds would visit the area on a frequent basis, particularly when there are other less disturbed feeding areas nearby. Finally, the aquatic habitat of Silver Lake differs from that of the Housatonic in that it is a lotic ecosystem with deep sediments containing a number of different chemical contaminants. For these reasons, GE proposes to address the potential ecological risks at Silver Lake separately from the rest of the Housatonic River site. A separate proposal outlining the proposed approach for evaluating potential ecological risks at Silver Lake will be submitted at a later date.

1.4 Relationship of the Health and Environmental Assessment to the Regulatory Process

The RCRA Corrective Action Permit issued by the EPA to GE, effective January 3, 1994, requires that a Health and Environmental Assessment (HEA) be conducted for the site. Under that permit, GE is required to identify the environmental systems that may be exposed to constituents of concern, and to include a proposal for determining risks to the environment as a result of past or ongoing releases from the facility.

Similarly, the MCP requires that an environmental risk characterization be undertaken in order to determine whether a level of "no significant risk of harm to the environment" exists or has been achieved at a disposal site (310 CMR 40.0900, 40.0995). This analysis is required in order to provide information that will help regulatory agencies evaluate the need for remediation of the site. There are three basic approaches to ecological risk assessment under the MCP. The first two are chemical-specific (Methods and 1 and 2) and compare media concentrations with standards that have been developed by Massachusetts DEP. These methods may be used if contaminants on-site are limited to soils and groundwater. If, however, contaminants are present or are likely to migrate at potentially significant concentrations to other media for which standards have not been developed, then Method 1 cannot be used alone. In this case, there are two options available to the risk assessor: a) either use Method 1 for soils and groundwater and Method 3 for other media, or b) complete a Method 3 risk assessment that is site-specific and considers specific exposure patterns and contaminant distributions. Because sediments of the Housatonic River have been shown to contain PCBs, and because GE intends to conduct a site-specific evaluation of ecological risks at the site, this risk assessment will utilize a Method 3 analysis.

Within Method 3, there are two stages of evaluation. The first, known as a Stage I environmental screening, is used to identify all potential exposure pathways and to conduct a screening level analysis to determine if there is "potentially significant exposure" of environmental receptors to contaminated media. If it can be shown that this potential does not exist, then there is no need to conduct a Stage II analysis. If, however, the potential for significant exposure cannot be ruled out, it is necessary to proceed with a Stage II environmental assessment. Because the potential for wildlife exposure to contaminated sediments, surface water, and soils exists at the Housatonic River site, a Stage II analysis will be conducted and a site-specific assessment will be completed.

This risk assessment will be conducted in accordance with the RCRA Corrective Action Permit and the MCP. It will also take account of pertinent EPA and DEP guidance, including:

- EPA, 1992, *Framework for Ecological Risk Assessment*;

- EPA, 1989a, *Risk Assessment Guidance for Superfund, Volume II: Environmental Evaluation Manual,* Interim Final;

- EPA, 1989b, *Ecological Assessment of Hazardous Waste Sites: A Field and Laboratory Reference*;

- ECO Update Bulletins, a series of updates to EPA (1989a) issued intermittently by the Office of Solid Waste and Emergency Response; and

- The Massachusetts DEP's guidance for conducting ecological risk assessments (which is currently under development by the DEP) when it is finalized and available.

2.0 PROBLEM FORMULATION

As defined in EPA's (1992) *Framework for Ecological Risk Assessment*, problem formulation establishes the goals, breadth, and focus of the overall assessment. In its most complete form, problem formulation includes a preliminary characterization of exposure and effects and an examination of scientific data and data needs, policy and regulatory issues, and site-specific factors; its purpose is to define the feasibility, scope, and objectives for the ecological risk assessment. In short, this stage provides an early identification of key factors to be considered in order to produce a scientifically sound risk assessment. EPA (1992) recommends several interrelated efforts within the larger problem formulation stage. These include the definition of stressor characteristics, identification of the ecosystem potentially at risk, discussion of ecological effects, selection of endpoints, and development of a conceptual site model. Each aspect of problem formulation is necessary to the selection of methods of investigation and, as such, is critical to the development of the work plan.

2.1 Stressor Characteristics

A multitude of factors may act to stress the Housatonic River valley ecosystem, including both physical and chemical stressors. Physical stressors may include both natural and anthropogenic events, such as extremes in natural conditions, habitat alteration, direct manipulation of wildlife and resource management practices. Chemicals which were historically introduced to the Housatonic River system through industrial, municipal, and agricultural practices could potentially stress the ecosystem through direct toxic effects on the species present.

2.1.1 Physical Stressors

Several natural conditions may stress the Housatonic River valley ecosystem, including flooding, storms, and extreme temperatures. Flooding may result in drowned individuals and destruction of food resources, nests, and cover. Depending on the timing of floods, individuals may simply establish their breeding and foraging territories in different locations during periods of flooding. Storms, with their associated precipitation, winds, and extreme temperatures, can stress wildlife through starvation, freezing, drowning, and nest destruction or abandonment. Such extremes in weather are not unusual in Massachusetts.

The most evident anthropogenic physical stressor at the Housatonic River site is habitat alteration and destruction. Alteration of habitat along the Housatonic River is most apparent within the city of Pittsfield, upstream of the New Lenox Road bridge. Within this reach, residential, commercial, and industrial structures have been constructed, oxbows have been filled, native vegetation has either been completely removed or replaced with ornamental shrubs and grasses, and the river channel has been straightened for flood control purposes. In many areas upstream of the New Lenox Road bridge, these activities have resulted in the complete removal of suitable foraging and nesting habitats, thereby preventing wildlife from establishing territories in these areas. Where habitats have not been entirely destroyed, they have been substantially fragmented by buildings, parking lots, and roads. The roads which dissect the ecosystem of the Housatonic River valley limit the natural (and necessary) movements of wildlife as they forage, a situation analogous to an island existence (Diamond and May, 1976). The basic theory of island biogeography was elucidated by MacArthur and Wilson (1967). According to the theory, island size is of great importance to the number of species that will inhabit the island. Immigration and extinction rates intersect to define the equilibrium species number for an island. Smaller islands exhibit lower immigration rates and higher extinction rates, which lowers the equilibrium species number. Ecosystems that are fragmented, such as those in the upper Housatonic River valley, create small islands capable of supporting only small populations of a limited number of species (Lovejoy and Oren, 1981).

In the HVWMA downstream of the New Lenox Road bridge, physical stressors associated with development are replaced by hunting, trapping, and resource management practices. Wildlife species legally hunted in Massachusetts include: crow, turkey, ruffed grouse, pheasant, quail, various waterfowl, snowshoe hare, cottontail rabbit, opossum, raccoon, fox, coyote, bobcat, gray squirrel, jackrabbit, black bear, deer, bull frog, green frog, and snapping turtle (MDFW, 1992). Species legally trapped in Massachusetts include: fox, coyote, skunk, opossum, weasel, bobcat, fisher, mink, raccoon, otter, muskrat, and beaver (MDFW, 1991). Since HVWMA is actively managed to promote hunting and trapping and since representatives of nearly all levels of the food web may be legally hunted or trapped, these activities could stress the overall ecosystem if overharvesting were to occur. In addition to the direct effects of killing wildlife, resource management practices, such as mowing, may stress the ecosystem by limiting cover for feeding, nesting, and rearing young for those species not targeted for promotion. Additionally, competition for food resources is likely increased because of the mowing regularly conducted in HVWMA, while genetic and species diversity may be decreased.

2.1.2 Chemical Stressors

In addition to the physical stressors described above, chemical contaminants may also stress the ecosystem of the Housatonic River and flood plain. Most of the chemical sampling and analysis activities to date in these areas have focused on PCBs, since PCBs are believed to be the primary chemical constituent of concern in the Housatonic River system. These activities have shown the presence of PCBs in the sediments, surface water, flood plain soils, and biota of the river, as discussed further below.

Some limited sampling and analysis activities have also been conducted for other constituents. Specifically, as part of prior MCP investigations, the sediments and surface water upstream of, adjacent to, and downstream of the GE facility were sampled and analyzed for the Appendix IX+3 constituents (40 CFR Part 264). These data were reported to the Massachusetts DEP and EPA (Blasland & Bouck, 1991, 1992a). The Appendix IX+3 data on sediments showed the presence of a number of organic and inorganic constituents in sediments adjacent to or downstream of the GE facility. However, an evaluation of these data indicated that: (a) the inorganic constituents were either found at concentrations within the range of concentrations detected in upstream sediments or flood plain soils or else were found only in localized areas and not at further downstream locations; and (b) the organic constituents detected downstream of the GE facility were found only at low levels and in one or a few localized areas (Blasland & Bouck, 1992a). The Appendix IX+3 surface water data showed the presence of several inorganic constituents, as well as a few organic constituents (detected only under low-flow conditions). An evaluation of these data indicated that: (a) the downstream concentrations of the inorganic constituents were generally comparable to upstream levels; and (b) the few organic compounds detected during low-flow conditions were found only in localized areas and not in farther downstream locations (except for chlorobenzene, which was detected in the three farthest downstream locations sampled) (Blasland & Bouck, 1992a).

In general, these prior sampling data do not appear to indicate any system-wide distribution of non-PCB constituents in the Housatonic River. However, GE is currently conducting additional sampling of the river sediments, surface water, and flood plain soils for Appendix IX+3 constituents, under the MCP Supplemental Phase II/RFI Proposal (Blasland, Bouck & Lee, 1994a), which was approved by the Massachusetts DEP and EPA in September 1994. Results from this additional sampling have not been considered or evaluated in developing this work plan, since the sampling is not complete.

Based on the findings described above, this work plan focuses on PCBs as the principal chemical stressor of concern in the Housatonic River system. It should be noted, however, that the field and laboratory studies proposed herein, which will evaluate biota actually inhabiting the Housatonic River and floodplain, would also account for other constituents (if any) released by GE into the Housatonic River. In any case, in the event that the current sampling program should identify additional chemicals in the Housatonic River or flood plain that can be attributed to GE releases, and if such findings would affect the activities described in this work plan or require additional studies, an addendum to this work plan will be submitted to the Agencies to address those issues. Further, the Supplemental HEA Proposal/Risk Assessment Scope of Work to be submitted to the Agencies at the conclusion of the ongoing investigations will describe the procedures to be used to select, from all detected constituents, the potential constituents of concern for evaluation in the HEA/risk assessment.

Characteristics of PCBs

PCBs are a group of synthetic organic compounds composed of a biphenyl molecule with one to ten chlorine atoms. Their production and use were banned in July 1979. Commercially used PCB mixtures, which were marketed under the trade name Aroclor, were composed of individual polychlorinated biphenyl congeners. The Aroclors were differentiated based on the number of carbon atoms on the biphenyl molecules and the overall degree of chlorination. For example, Aroclor 1260 is a mixture of congeners with twelve carbon atoms on the biphenyl molecules and is 60 percent chlorinated (by weight). The PCBs detected in the Housatonic River and its flood plain consist primarily of Aroclor 1260; hence all discussions of PCBs in this report pertain to Aroclor 1260, unless otherwise noted.

PCBs occur as colorless or yellow crystals or oily liquids (ATSDR, 1993). In general, PCBs have low water solubility, high solubility in organic solvents, and are known for their insulating properties, thermal stability and resistance to oxidation and various chemical agents (ATSDR, 1993). The environmental fate and transport of PCBs involve adsorption to particulate and organic matter, volatilization, biodegradation, and photolysis. The low solubility, strong sorption to soils, and hydrophobicity of PCBs cause them to be extremely persistent and virtually immobile in soils (ATSDR, 1993). The presence of co-contaminants, particularly organic solvents, generally enhances leaching (Overcash et al., 1991).

Higher chlorinated Aroclors, such as Aroclor 1260, tend to resist aerobic biodegradation but are susceptible to anaerobic dechlorination (Brown et al., 1987a,b; Abramowicz and Brennan, 1991). In addition to the Aroclor mixture, biodegradation rates depend on concentration, type of microbial population, available nutrients, and temperature (ATSDR, 1993). Although volatilization of highly chlorinated Aroclors is generally limited by their strong adsorption to particulate matter, volatilization is enhanced in soils with low organic carbon, due to the lowered potential for adsorption (Shen and Tofflemire, 1980). Soil moisture also influences volatilization, since codistillation of PCBs with water enhances volatilization (ATSDR, 1993). Work by Haque et al. (1974) suggests volatilization half-lives from sand of 10, 25, and 40 days for tetrachlorobiphenyls, pentachlorobiphenyls, and hexachlorobiphenyls, respectively. Aroclor 1260 is composed of 12% pentachlorophenyls and 42% hexachlorophenyls, by weight (ATSDR, 1993).

Potential Sources of PCBs

As described by Blasland & Bouck (1991), PCBs were used at the GE facility in Pittsfield from 1932 until March 1977, as part of a flame-resistant, insulating liquid for select transformer applications. This synthetic oil, referred to as Pyranol, was used in less than five percent of transformer products manufactured at the facility. Before 1977, inadvertent releases of these materials reached the storm sewer system associated with the facility and were subsequently conveyed to the East Branch of the Housatonic River. A number of remediation projects were initiated in the late 1960s to reduce releases of PCBs, and the use of PCBs at the facility was discontinued in 1977. After 1977, ongoing remediation projects were supplemented with additional programs to control the release of PCBs to the river system.

Spatially, historic sources were likely clustered around the city of Pittsfield. As is the case with any publicly-accessible water body, other contributors of PCBs to the Housatonic River system may exist that are unrelated to the GE facility. Releases to the river could occur from many adjacent public, private, or commercial properties, or be conveyed to the river through permitted or unpermitted discharge points. These potential sources can be categorized as: industries and other commercial operations; permitted discharges; landfills and hazardous waste sites; and non-point sources. Additional potential sources exist along the branches of the Housatonic River, while still others exist both upstream and downstream of Pittsfield. Various industries and wastewater treatment plants (WWTPs) exist upstream of Woods Pond, near the towns of Lee and Lenox (between Woods Pond and Rising Pond), and between Rising Pond and the Connecticut border. Blasland & Bouck (1991) further details potential sources of PCBs in addition to GE.

Distribution of PCBs

The distribution and concentrations of PCBs in the sediments, surface water, flood plain soils, and biota of the Housatonic River have been measured on numerous occasions. In the early 1980s, Stewart Laboratories (1982) conducted a comprehensive sampling and analysis study of PCBs in the sediments, water, and biota of the river. Additional, more limited sampling efforts were carried out for GE by Blasland & Bouck in the late 1980s (Blasland & Bouck, 1991). In the early 1990s, pursuant to the MCP, Blasland & Bouck carried out another extensive study for GE, obtaining additional PCB data on sediments, surface water, flood plain soils, and biota (Blasland and Bouck, 1991, 1992a). Subsequently, additional flood plain soil sampling for PCBs was conducted at specific properties as part of an evaluation of the need for short-term measures (STMs) at certain flood plain properties (Blasland & Bouck, 1992c, 1993, 1994b) and again as part of avian and mammalian studies conducted by ChemRisk (1994) and discussed in Section 3.2. The results of these various studies are summarized below for each medium. GE is currently in the process of collecting more data on PCBs in sediments, surface water, flood plain soils, and biota in the river pursuant to the MCP Supplemental Phase II/RFI Proposal for the river site (Blasland, Bouck, & Lee, 1994a). Preliminary results from this investigation are not included in this summary, since the investigation is not complete. However, the results from that investigation will be considered, along with all prior data, in the ecological risk assessment for this site.

As discussed by Blasland, Bouck & Lee (1994a), data on PCBs in river sediments have been compiled into a single database that provides information on both the horizontal and vertical extent of PCBs in sediments of the Housatonic River. Additional sediment sampling is also planned as part of the RFI. The database demonstrates that significant quantities of sediments containing PCBs are located between the GE facility in Pittsfield and the Woods Pond Dam. PCBs are also present in river sediments downstream of Woods Pond Dam, but at substantially lower concentrations. The distribution of PCBs in sediments of the Housatonic River are summarized in Table 2-1. The sample locations and PCB results for all sediment sampling efforts in the river reaches between the GE facility and Woods Pond Dam are shown in Figures A-1 through A-11 in Appendix A. As illustrated on those figures, concentrations of PCBs in sediments are highly variable. This variability reflects the variability in the rivers's physical characteristics, flow conditions, and sedimentation characteristics.

As with the sediments of the Housatonic River, numerous studies conducted since the mid-1970s have produced an extensive data base regarding the occurrence of PCBs in the water column of the Housatonic River. The surface water data base indicates that PCBs in the water column measured during prior MCP Phase II activities were generally at concentrations close to the analytical detection limits. Approximately 30 percent of the results for total PCBs were reported as less than the detection limits. For filtered water samples, approximately 90 percent of the PCB data from the MCP investigation were reported as non-detect (Blasland & Bouck, 1991). The most recent water column monitoring program was conducted on a monthly basis between October 1990 and September 1991. These data indicate that, in the river reach between the GE facility and New Lenox Road bridge, concentrations of PCBs in the water column ranged from 0.03 to 0.39 μg/l, with an average concentration of 0.12 μg/l. Between New Lenox Road bridge and Woods Pond Dam, the concentrations of PCBs in the water column ranged from 0.06 to 0.58 μg/l, with an average concentration of 0.19 μg/l. Downstream of Woods Pond Dam, the concentration of PCBs in the water column ranged from 0.02 to 0.45 μg/l, with an average concentration of 0.11 μg/l (Blasland & Bouck, 1991).

Given the presence of PCBs in the sediments of the Housatonic River, PCB-containing sediments appear to have been transported and deposited onto the outlying banks and flood plain of the river during flooding events. Extensive flood plain soil sampling has shown that concentrations of PCBs greater than 1 ppm are generally contained within the approximate 10-year flood plain

Table 2-1. Distribution and Concentrations of PCBs in Surficial Sediment (mg/kg dry weight)

Reach	Minimum Concentration	Maximum Concentration	Mean Concentration
GE Facility to New Lenox Road	ND	290	26.7
New Lenox Road to Wood Pond Headwaters	ND	270	29.0
Woods Pond	ND	220	25.0
Woods Pond Dam to Rising Pond Headwaters	ND	22.0	4.57
Rising Pond	ND	37.0	5.81
Rising Pond Dam to Connecticut Border	ND	2.3	0.59
Connecticut Border to Stevenson Dam	ND	3.2	0.60

Source: Blasland and Bouck, 1992a; Lawler, Matusky & Skelly, 1994
ND: not detected

(Blasland, Bouck & Lee, 1994b). The sample locations and PCB results from all flood plain soil sampling efforts in the river reach between the GE facility and Woods Pond are illustrated in Figures A-12 through A-22 in Appendix A. Surficial flood plain soil samples collected between the GE facility and New Lenox Road bridge ranged from 0.05 to 230 mg/kg, with an average concentration of 20 mg/kg. Surface flood plain soil samples collected between New Lenox Road bridge and Woods Pond dam ranged from 0.05 to 75 mg/kg, with an average concentration of 10 mg/kg. However, it should be noted that most of these samples were collected immediately below New Lenox Road bridge. The limited number of samples collected further downstream of this area typically have lower PCB concentrations. Further floodplain transect sampling is currently being performed; this should provide more accurate data on the spatial distribution of PCB levels. Surface flood plain soil samples collected downstream of Woods Pond dam ranged from 0.05 to 4.0 mg/kg, with an average concentration of 0.6 mg/kg.

Since 1980, investigations of Housatonic River biota have generated data on concentrations of PCBs in fish, frogs, a turtle, and benthic macroinvertebrates. Fish samples collected by Stewart Laboratories, Inc. (1980, 1982) between Bailey Road Bridge and the headwaters of Woods Pond ranged in concentration from 2.2 to 228 mg/kg (net weight) and averaged 26 mg/kg. In November 1990, fish from Woods Pond and Rising Pond were sampled and individual skin-on fillets were analyzed for PCBs. The samples from Woods Pond (from the headwaters to Wood Pond dam) ranged in PCB concentration from 4.2 to 119 mg/kg; concentrations varied among species with mean concentrations ranging from 8.4 for yellow perch to 13.8 for brown trout. The samples from Woods Pond Dam to the Connecticut State line ranged in concentration from 6.1 to 33.0 mg/kg and averaged 18 mg/kg. PCB concentrations in fish in the Connecticut portion of the Housatonic River generally range from 0.03 to 29 mg/kg (wet weight basis), with an average of 3.3 mg/kg (Blasland and Bouck Engineers, 1991; The Academy of Natural Sciences of Philadelphia, 1993).

In 1982, a single composite sample of 12 frogs collected from Woods Pond was analyzed for total PCBs; the concentration detected was 4.4 mg/kg. In 1992, three seven-frog composite samples (leg muscle only) showed total PCB concentrations ranging from 2.2 mg/kg to 5.3 mg/kg, and averaged 4.0 mg/kg. A single turtle collected by Stewart Laboratories, Inc. (1982) from Woods Pond contained 2.1 mg/kg total PCBs. Finally, invertebrate sampling data collected in Connecticut are summarized in Table 2-2.

Table 2-2. Summary of Invertebrate Data Collected from Connecticut (mg/kg)[a,b]

Date	Filter Feeding[c] Larvae Total PCB conc.	Predatory[d] Larvae Total PCB conc.
1978	18.9	22.9
1979	8.2	5.6
1980	9.2	6.6
1981	6.2	4.6
1982	NA[e]	NA[e]
1983	NA	NA
1984	3.7	2.9
1985	0.5	0.8
1986	2.8	2.8
1987	7.9	5.2
1988	4.2	2.3
1989	5.6	4.6
1990	1.2	1.9

a. Reference: Pizzuto, 1991.
b. Samples consisted of approximately 20 gram composites.
c. Filter feeding larvae are caddisfly (*Hydropsychidae*).
d. Predatory insects are average values of hellgrammite larvae (*Corydalus*) and stonefly nymphs (*Perlidae*).
e. NA = Not analyzed.

2.2 Ecosystem Potentially at Risk

The ecosystem within which effects potentially occur provides the ecological context for the risk assessment. To that end, the geography, topography, climate, vegetative habitats, and food web of the ecosystem potentially at risk are described in this section. This risk assessment focuses on that portion of the Housatonic River and flood plain where the most substantial exposures are expected due to high PCB levels, while confounding factors are minimized. These attributes should ensure that any effects observed as a result of exposure to PCBs within the identified region are representative of the worst-case effects expected to occur anywhere along the Housatonic River. Conversely, if no effects are observed within the identified region, it would be reasonable to conclude that no effects would occur elsewhere along the Housatonic River as a result of PCB exposure.

As detailed in the preceding characterization of stressors, the reach of the Housatonic River with the highest concentrations of PCBs is between the GE facility and Woods Pond Dam. From the GE facility to the New Lenox Road bridge, terrestrial habitats have been substantially destroyed, altered, and/or fragmented, which would confound or make impossible any meaningful evaluation of terrestrial communities within this reach. For these reasons, as discussed further in Section 2.2.2, the present study will focus on the river reach between the GE facility and Woods Pond for purposes of evaluating the aquatic ecosystem, and on the river reach between the New Lenox Road bridge and Woods Pond for purposes of evaluating the terrestrial ecosystem. For the evaluation of wetland communities, the study area will include the river reach between the river's confluence with the West Branch and Woods Pond, which is the area containing the majority of wetlands. All of these river reaches are illustrated on Figures A-1 through A-11 (Appendix A). (We note, however, that the aquatic sampling will extend beyond the study area as necessary to obtain data from appropriate reference sites.)

2.2.1 General Description

The East Branch of the Housatonic River immediately downstream of the GE facility flows through an urban area where the water is shallow (average 1.3 ft. deep) and slow moving. The river is characterized by riffle/pool complexes with woody snags and a substrate of sand, silt and gravel. Further downstream of GE, the river widens and flows through a less developed area but remains a slow moving stream with riffles, pools, and snags. It is in this area that the East and West Branches of the Housatonic River converge. Between the New Lenox Road bridge and Woods Pond, the river is much deeper (up to 7 ft) with low water velocity and deep pools. This area is characterized by dense growths of overhanging bank vegetation.

The flood plain south of New Lenox Road to Woods Pond Dam is relatively wide. Approximately one-half mile south of New Lenox Road, the flood plain along the east bank of the river is confined by October Mountain, while the west bank has a relatively flat topography resulting in an extended flood plain (Blasland & Bouck, 1991). Within this reach, the Housatonic River is associated with substantial wetland flood plains and backwater areas. Here, the river is slow-mixing and meandering as it travels through relatively flat and level areas of local topography. Deposition of sediment within the backwater areas is pronounced; sediment depth averages 22 inches (Stewart Laboratories, Inc., 1982). For the most part, the channel is shallow and the stream banks are not well defined in this reach. In general, channels with these characteristics provide hydraulic conditions favorable for sediment deposition onto the flood plain (Blasland & Bouck, 1991). Downstream of the confluence of the East and West Branches, the Housatonic generally ranges in width from 75 to 120 feet, with gradients ranging from 0.03 percent to 0.43 percent (Chadwick & Associates, Inc., 1994). Selected water chemistry parameters at 10 locations on the Housatonic River are summarized in Table 2-3.

The climate of the Housatonic River valley is temperate, with warm, humid summers and cold winters. The NOAA weather station in Stockbridge, MA has compiled data on the normal climate: in this case, normal refers to the average of all daily values except extreme (i.e., record-breaking) values. The normal annual temperature is 46.3° F, the normal maximum annual temperature is 57.9° F, and the normal minimum annual temperature is 34.7° F. The normal temperature in July, the warmest month, is 68.9° F, while the normal temperature in January, the coldest month, is 21.4° F. Normal annual precipitation in the form of rain and snowfall is 43.47 inches per year, ranging from 2.84 inches in February to 4.36 inches in August (NOAA, 1992). Prevailing winds are from the west. The average growing season in the upper basin is 120 days (NERBC, 1980).

Table 2-3. Ranges of Selected Water Chemistry Parameters

Site[a]	Water Temp (°C)	D.O. (mg/L)	pH	Total Ammonia (mg/L)	Unionized Ammonia (mg/L)	Nitrate (mg/L)
Shallow Sites Upstream of GE Facility						
EP1	9-28	7.8-10.2	NM	NM	NM	NM
WB1	12-30	6.9-9.1	NM	NM	NM	NM
Shallow Sites Downstream of GE Facility						
EB2	11-27	6.6-9.4	NM	NM	NM	NM
HR1	11-32	6.7-8.8	7.9-8.3	ND-0.23	0-0.02	0.33-0.81
HR3	12-30	7.9-12.6	8.0-9.0	ND-0.08	0	0.92-2.40
HR4	13-29	6.6-19.2	7.9-8.5	ND-0.21	0	0.71-1.70
Deep Sites Downstream of GE Facility						
HR2	12-26	6.2-8.4	NM	NM	NM	NM
HR5	12-29	7.1-11.0	8.1-8.8	NM-0.14	0-0.01	0.68-1.40
HR6	14-29	6.0-15.2	8.0-8.6	NM-0.14	0	0.44-0.84
Woods Pond						
WP1	12-33	3.2-11.2	NM	NM	NM	NM

Note: Data collected by Blasland and Bouck Engineers at study sites on the Housatonic River system, 1993. Water temperature and dissolved oxygen (D.O.) measured weekly from 5/25/93 to 9/22/93. Other parameters measured monthly from 5/25/93 to 9/22/93. NM = Not measured. ND = Not Detected.

a. Sites correspond to sampling stations evaluated by Chadwick & Associates, Inc. (1994).

Based on Land Use and Property Ownership Maps (Blasland & Bouck, 1992b), National Wetlands Inventory maps, and field observations, much of the land between New Lenox Road and Woods Pond Dam that is potentially affected by a 10-year flood event is comprised of flood plain forests, shrub meadows, and cattail marshes. The vegetation of these three habitat types is described below.

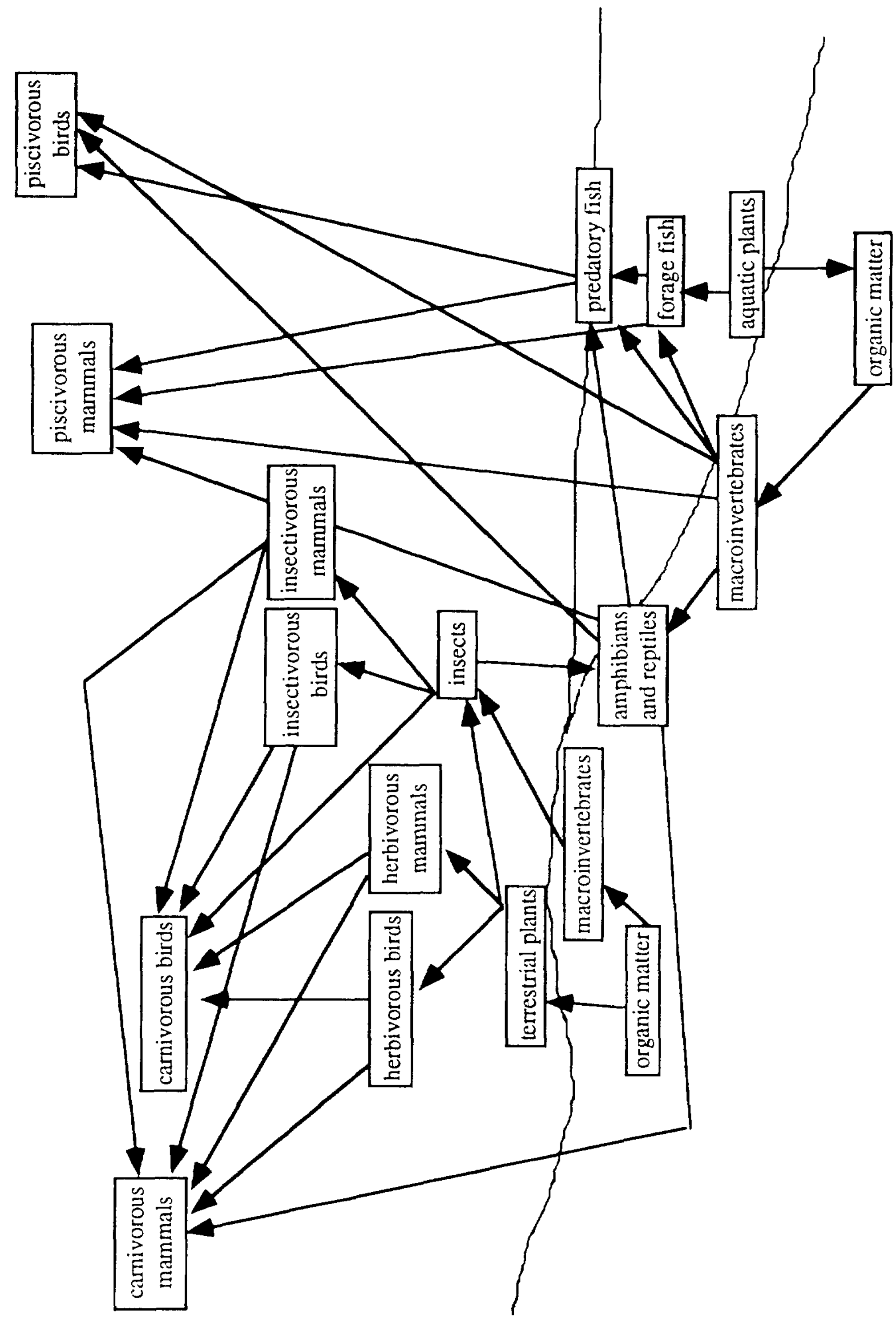
The Housatonic flood plain forest is dominated by mixed hardwood trees with scattered pine. It is intermittently flooded and has well developed ground, shrub, and tree strata. Ground cover is prevalent, consisting mainly of ferns. The shrub stratum, which covers approximately one-third of the flood plain forest, is relatively diverse and is dominated by arrowood and sugar maple. The tree stratum is well developed both in terms of density and diversity. The diversity is typical of a forest controlled by flooding; while many species of trees (balsam poplar, black oak, black cherry, white and yellow birch, ironwood, etc.) are represented, one species dominates (sugar maple). The ground, shrub, and tree strata together comprise a diverse ecosystem with high quality habitat for song birds and small mammals.

Shrub meadows along the Housatonic River between New Lenox Road and Woods Pond Dam are early successional ecosystems dominated by grass-forb-shrub communities. Within the region of interest, most shrub meadows are actively managed for wildlife and hunting by mowing and other practices. Vegetation is abundant, even in mowed areas. The ground cover is comprised of grasses and forbs, interspersed with occasional emergent shrubs.

As their name would suggest, cattail marshes are dominated by a single plant species, cattails, although limited numbers of red osier dogwood, willow, and silver maple are interspersed among the cattails. Arrowheads, water hyacinths, and lily pads are generally present in the backwaters of the river along the cattail marshes. Because the cattail stands are of limited size, frequently inundated with water, and are dominated by a single plant species, they do not likely support extensive or diverse terrestrial wildlife populations.

In addition to habitat characteristics, the species of animals present help define the ecosystem. Based on field work conducted since 1993, a schematic diagram, presented as Figure 2-1, was developed for the terrestrial and aquatic foodweb of the Housatonic River ecosystem.

Figure 2-1. Schematic of Housatonic Valley Food Web



2.2.2 Selection of Focused Study Areas

Given the large areal extent of the Housatonic River and flood plain, it is necessary to select focused study areas within the aquatic, terrestrial, and wetland systems for more detailed evaluation. The criteria used in making this selection were to identify areas that have the most substantial potential for exposure of the organisms and biotic communities to the chemical of potential concern (PCBs), while at the same time minimizing confounding factors.

Based on these criteria, and the characterization of stressors and the ecosystem presented above, the aquatic system study area selected is the reach of the Housatonic River from the GE facility in Pittsfield to Woods Pond. Because this reach is predicted to have the greatest potential for exposure of the aquatic community to PCBs, it is assumed that if an absence of risk is demonstrated for this reach, other (less affected) reaches would also be without risk. The converse, however, is not true. That is, if a risk is demonstrated in the study area selected, then an additional evaluation and possibly supplemental data may be required in order to determine the extent of risk (if any) in other, less affected reaches of the river.

A focused study area for the terrestrial ecosystem was also identified using the same criteria described above. Based on these criteria and the characterization of stressors and the ecosystem, the study area selected for the terrestrial community encompasses the approximate 10-year flood plain between the New Lenox Road bridge and Woods Pond Dam. In the upstream area between Pittsfield and New Lenox Road bridge, the flood plain of the Housatonic River is substantially more developed and the wildlife habitat more fragmented than downstream of the bridge. These characteristics of the habitat upstream of New Lenox Road bridge would likely confound any ecological evaluation of the terrestrial community. Thus, the area that best reflects the combination of high exposure potential and minimization of confounding factors is the reach between the New Lenox Road bridge and Woods Pond. As with the aquatic community, a finding of an absence of biologically significant risk in the terrestrial study areas would suggest that other, less affected areas would also be without risk. However, should there be a finding of potential risk within the terrestrial study area, it may be necessary to undertake a supplemental evaluation of less affected areas to determine the extent of risk (if any) in those areas.

The study area for the wetland ecosystem was identified based on consideration of the major wetlands and "high priority sites" within the approximate 10-year flood plain of the Housatonic River between the GE facility and Woods Pond dam. National Wetland Inventory (NWI) maps and the Massachusetts Natural Heritage Estimated Habitat Maps (which estimate the extent of wetland habitats for reported occurrences of state-listed animal species of special concern, as well as what are called "high priority sites" for rare wetland animals, plants, and/or significant natural areas/communities) were used to define the wetland study area. This area encompasses the approximate 10-year flood plain between the confluence of the East and West Branches and Woods Pond dam. Because the available maps do not identify wetlands or other high priority sites on the Housatonic River between the GE Facility and the confluence with the West Branch, this area was not included in the study area for the wetland ecosystem.

2.3 Ecotoxicological Effects

One of the critical components of the problem formulation phase is the determination of the taxonomic groups, species, and assessment endpoints to be evaluated. This determination requires an evaluation of the pertinent ecotoxicological literature in order to assess the sensitivity of various species and various toxicological endpoints to the chemical stressor of interest. To provide a sufficiently conservative assessment of the potential for ecological risks at a site, it is important to try to select several taxonomic groups that are sensitive to the chemical stressor of interest and, within those groups, to select highly sensitive toxicological endpoints that have biological or ecological significance at the subpopulation, community, or system-wide level.

This section provides an overview of the scientific literature as it pertains to the ecotoxicological effects of PCBs. Because the data to date have identified Aroclor 1260 as the primary chemical stressor of interest in the Housatonic River and flood plain, and because there appears to be considerable variability in the toxicological effects of different PCBs, this overview focuses on the demonstrated effects of the more highly chlorinated congeners and mixtures. In addition, because fungi, plants, invertebrates, reptiles, amphibians, fish, birds, and mammals all have some potential for exposure to PCBs in the Housatonic River ecosystem, the toxicologic effects of PCBs that are reported in the published scientific literature have been evaluated herein for each group of potential receptors. The published results of the available scientific studies are then used to select receptors of interest and assessment and measurement endpoints to be evaluated in the risk assessment.

2.3.1 Toxicity to Fungi

Fungi differ from both plants and animals and, therefore, have been assigned to their own phylogenetic kingdom. Like plants, fungi are predominately sedentary with plant-like cell walls. However, because fungi lack chlorophyll and cannot manufacture their own food, they must obtain complex high-energy nutrients from their environment through absorption.

Cole et al. (1979) reported that certain species of fungi may bioaccumulate PCBs; however, there are few data reported in the scientific literature pertaining to toxic effects associated with tissue concentrations in the fungi. Rather, studies of the toxicological effects of PCBs on fungi have generally evaluated adverse effects associated with PCB concentrations in laboratory media. Tejedor et al. (1979) and Murado et al. (1976) studied the growth of yeasts exposed to various PCB mixtures. Tejedor et al. (1979) reported that Aroclors 1232, 1242, 1248, 1254, and 1260 all inhibited growth of *Saccharomyces cerevisiae* at concentrations of 25 mg/L. Murado et al. (1976) reported that the growth of *Aspergillus flavus* in liquid cultures containing Aroclor 1254, ranging from 5 to 50 mg/L, was reduced in a dose dependent fashion, and reported that the magnitude of this effect was greatest for Aroclor 1232 and lowest for Aroclor 1260.

The concentrations that have been tested and shown to have no adverse effects on fungi are substantially higher than concentrations of PCBs that have been measured in the Housatonic River. As a result, toxic effects of PCBs to fungi should not be of concern for this ecological risk assessment.

2.3.2 Toxicity to Plants

Phytoplankton, including algae, diatoms, and other microscopic aquatic plants, are present in almost every aquatic environment. They form the basis of many aquatic food webs by acting as primary producers and as food sources for higher trophic-level organisms. It has been shown that phytoplankton can accumulate PCBs and, therefore, represent a means of introducing PCBs into the food web (Harding and Phillips, 1978; Keil et al., 1971; Fisher et al., 1976; Södergren, 1971; Morgan, 1972). Brown et al. (1982) demonstrated that the rate of uptake of Aroclor 1254 is partially dependent on organism size and that the extent of PCB accumulation appears to be species-specific. Stange and Swackhamer (1994) reported that the bioaccumulation of PCB isomers with log K_{ow} values of 6.0 or greater varied substantially among species, likely as a result of interspecies differences in lipid fraction of the cell membrane and dissolved organic carbon content. These authors reported bioaccumulation factors for individual PCB isomers in three species of phytoplankton that ranged from 1,000 to 100,000.

Despite the fact that bioaccumulation of PCBs by phytoplankton has been reported by a number of authors, there is little information regarding toxicity associated with those tissue concentrations. Rather, phytotoxicity has been generally studied based on the PCB concentrations in the surrounding water. Although Stange and Swackhamer (1994) observed that total PCB concentrations of 6.9 µg/L did not produce toxic effects in three phytoplankton species, other studies have shown that the growth of marine and freshwater algae is substantially reduced by higher concentrations PCB concentrations (Keil et al., 1971; Cole and Plapp, 1974; Glooschenko and Glooschenko, 1975; Hawes et al. 1976). The toxicity of different PCB congeners to algae varies considerably; in general the lower chlorinated biphenyl mixtures are more toxic than the more highly chlorinated mixtures (Craigie and Hutzinger, 1975; Hawes et al., 1976; Zullei and Benecke, 1978). In addition, there are substantial differences in sensitivity among species. It appears that toxic effects of PCBs to phytoplankton may occur at concentrations ranging from 1 µg/L (Moore and Harris, 1972) to 1000 µg/L (Mosser et al., 1972). It should be noted, however, that phytoplankton sensitivity has been shown to vary with temperature, geographic location, interspecific competition, cell density, osmotic pressure, and the presence of other toxicants (Fisher and Wurster, 1973, 1974; Fisher et al., 1973, 1974; Mosser et al., 1974, 1976; Cole and Plapp, 1974).

There is very little available information in the scientific literature regarding the uptake, accumulation, metabolism, and toxicity of PCBs in aquatic plants. In a review paper, Pal et al. (1980) reported that three species of aquatic plants bioconcentrated significant amounts of PCBs from water, while two species did not. Mahanty (1975) evaluated the toxicity of Aroclor 1242 in duckweed at concentrations of 0, 5, 10, 20, 30, 40, 50, and 100 mg/L in water and reported that PCB concentrations of 100 mg/L were lethal. The lowest dose that significantly reduced growth was 20 mg/L, indicating a LOEC of 20 mg/L and a NOEC of 10 mg/L for reduced growth for Aroclor 1242 in this species.

The majority of the scientific research regarding PCB toxicokinetics and toxicity in different species of terrestrial plants has focused on agriculturally important plant species, because of the potential human health concern related to the ingestion of foods containing PCBs. The majority of the PCBs entering terrestrial plants do so from atmospheric deposition of contaminated dusts or from volatilization from contaminated soil surfaces, rather than via translocation (Sheppard et al., 1991; Greichus and Dohman, 1980; Fries and Marrow, 1981). EPA (1994c) reports that translocation of PCBs from roots to above ground portions of plants is negligible compared with other mechanisms.

There is evidence suggesting that lower chlorinated PCB congeners with higher volatility are more readily available for absorption by plants than other more highly chlorinated and less volatile PCB congeners (Sawhney and Hankin,1984). Schonhen and Reiderer (1989) suggested that because of the low water solubility and high lipophilicity of highly chlorinated isomers and mixtures, PCBs tend to partition from the vapor state to inert lipophilic materials, such as the waxy covering of the leaves of vascular plants. However, several studies have shown that uptake through the leaves is very low (O'Connor et al. 1990). While absorption of PCBs through plant roots is negligible for terrestrial plants, it has been shown that root crops with peels containing a high lipid content may accumulate PCBs, but that accumulation is low (biomagnification factors of <1) compared to concentrations present in soils (Iwata et al, 1974; Wallnöfer et al., 1975; Iwata and Gunther, 1976; Lawrence and Tosine, 1977; Suzuki, 1977; Moza et al., 1979; Weber and Mrozek, 1979; Sawhney and Hankin, 1984; Sheppard et al., 1991; Webber et al., 1994). Finally, the level of PCBs in fruits of crop plants is several orders of magnitude less than that in the rest of the plant (Sawhney and Hankin, 1984; Cetinkaya et al., 1985).

Generally, PCBs are not toxic to terrestrial plants, except at very high concentrations. Most studies available in the published literature have evaluated the effects of PCBs on growth. Weber and Mrozek (1979) and Strek et al. (1981) reported that high concentrations of PCBs (1000 mg/kg Aroclor 1254 in soil) inhibited crop growth in sensitive species (soybeans and fescue), but that no effect on these species was observed at a soil concentration of 100 mg/kg. Strek et al. (1981) reported that corn and sorghum were unaffected at similar concentrations.

One study by Klekowski and Klekowski (1982) evaluated the frequency of somatic and gametophytic mutation in ostrich ferns growing within a PCB-contaminated floodplain of the Housatonic River. These authors reported that the gametophyte mutation frequency among those ferns was qualitatively different from the frequency among the controls. However, the mutation events measured in this study would not necessarily be passed on to future generations, as the ferns were tested during the haploid stage of their life cycle. At this stage, only a single copy of each gene is present in each cell. In contrast, during the diploid stage of the fern life cycle, each cell has two copies of every gene. A mutation in one gene of the pair may not affect the plant because diploid cells have the ability the excise mutations using the good gene as a template.

In summary, PCBs do not appear to be toxic to plants, except at very high levels. This is largely due to the fact that plants do not take up PCBs to an appreciable degree and those that are found in plant tissues are generally either the result of adsorption of PCB contaminated soils to root tissues, or volatilization from the soil to the waxy covering of the leaves. Because of this low level of uptake, it is necessary to have water concentrations greater than 1 μg/l and soil concentrations greater than 100 mg/kg before any adverse effects may occur. These conditions are not generally present in the Housatonic River ecosystem.

2.3.3 Toxicity to Invertebrates

Many aquatic and terrestrial invertebrates dwell in sediment and soil respectively, so they are in intimate contact with the substrates which constitute major sinks for PCBs. Some groups, such as filter–feeders, ingest sediments incidentally due to their feeding strategies. Others, such as terrestrial earthworms and aquatic oligochetes, ingest bulk sediment directly and this can act as an important, if not dominant, PCB uptake route (Lee, 1992).

Aquatic Invertebrates

Aquatic invertebrates have been reported to accumulate and bioconcentrate PCBs (Lowe et al., 1972; Sanders and Chandler, 1972; Courtney et al., 1978). PCBs bioconcentrate at different rates and accumulate to different levels in different species of aquatic invertebrates. BCFs in various species have been reported to range from approximately 10 in scud to as much as 101,000 in young oysters (Lowe et al., 1972; Nebeker and Puglisi, 1974). In one laboratory study, several freshwater invertebrates accumulated Aroclor 1254 residues up to 6300 times greater than those to which they were exposed (Mayer et al., 1977). When sea stars (*Asterias rubens*) were fed PCB–contaminated mussels for 5 months, the gonads of male and female sea stars contained 7 to 9 times the concentration of PCBs as the control sea stars and exhibited disturbed embryonic development of oocytes (den Besten et al., 1989). Bioaccumulation rates in invertebrates are site- and species-specific and are affected by various factors, such as food source, particulate size, organic carbon content, species lipid content, trophic level, surrounding environment, sex, age, and length of exposure (Sanders and Chandler, 1972; Harding and Addison, 1986; Larsson, 1986). In addition, two species were reported to have accumulated higher levels of the less-chlorinated PCB isomers than higher chlorinated ones over time (Lester and McIntosh, 1994). Dillon et al. (1990) compared the concentration of PCBs in invertebrates to those of other aquatic species and reported that aquatic invertebrates had concentrations one to several orders of magnitude lower than in fish.

Sediment-dwelling organisms have much higher direct exposures to PCBs than their pelagic (free swimming) counterparts because sediments have PCB concentrations at least three orders of magnitude greater than the overlying water (DiPinto et al., 1993). Lower chlorinated PCB mixtures are reported to be more toxic than those with higher chlorination (Stalling and Mayer, 1972; Nebeker and Puglisi, 1974; Sericano et al., 1991). The acute toxicity of PCBs to various aquatic invertebrate species can vary with sex, life-stage, and age (DiPinto et al., 1993 Kahn et al., 1976; Lowe et al., 1972; Courtney et al., 1978). In addition to lethality, other observed effects include growth inhibition and altered ion transport. For certain PCB isomers, however, the effects appear to be reversible.

A few laboratory studies have evaluated the effects of PCB spiked sediments on several different species of marine aquatic invertebrates. DiPinto et al. (1993) investigated the effect of exposing a species of copepod (*Microarthridian littorale*) for 96 hours to sediments containing 21 to 333 mg/kg of Aroclor 1254. The authors reported that 83 mg/kg of Aroclor 1254 significantly affected copepod survival, but concentrations of 42 mg/kg or less did not. They reported that the LC_{50} values for females and males were 251 mg/kg and 117 mg/kg, respectively. DiPinto et al. (1993) also examined the reproductive impairment of the copepods exposed to a range of 4 to 83 mg/kg Aroclor 1254 for 12 days. Egg production and total reproductive capacity was significantly reduced in 4.2 mg/kg spiked sediments and a reduction of larval copepods was observed in 8.3

mg/kg spiked sediments. In a study on the sand shrimp, McLeese and Metcalfe (1980) exposed adult shrimp to either Aroclor 1242 or 1254 for 96 hours. They reported a LC_{50} of >0.78 mg/kg for Aroclor 1242 and >3.4 mg/kg for Aroclor 1254. When a species of marine sandworm (*Nereis diversicolor*) was exposed for 128 days to an undefined mixture of PCBs at concentrations ranging from 18.7 to 89.6 mg/kg, the LT_{50} ranged from 31 to 48 days compared to the LT_{50} of 62 days for control sediments (Polikarpov et al., 1983). In a related experiment, these authors reported significant adverse effects on sandworm survival for concentrations of 14.7 mg/kg but not for concentrations of 1.7 mg/kg. A study on the exposure of an infaunal amphipod to Aroclor 1254 for 10 days reported an LC_{50} of 10.8 mg/kg (Swartz, et al., 1988).

MacDonald (1994) reported the results of a field study in which marine sediments were co-sampled for benthic invertebrates and PCB concentrations. The study concluded that the benthic invertebrate community structure was affected by PCBs because species richness, infaunal index, and echinoderm abundance were influenced by the sediment concentrations of Aroclor 1254. Low densities of echinoderms ($3.1\pm4.5/m^2$) were found when sites had PCB concentrations of 1.48±2.77 mg/kg whereas high densities (229 ± 60 per $0.1m^2$) were found in sites with 0.015±0.017 mg/kg PCBs. Similarly, low densities of arthropods and low species richness were found at sites with 4.88±4.09 and 1.05±2.5 mg/kg PCB. Although it is difficult to establish cause and effect relationships with these kinds of correlations, changes in invertebrate community structure are observed at PCB concentrations of similar magnitude to those which cause toxicity in laboratory experiments. Hence, community structure seems to be a reasonably sensitive endpoint.

Terrestrial Invertebrates

PCBs are rapidly taken up and bioconcentrated in earthworms (Diercxsens et al., 1985; Rodriquez-Grau et al., 1989; Goven et al., 1993). Crickets have also been shown to accumulate PCBs through air transfer to the crickets themselves or to the crickets' food, but body burden concentrations were less than 20% of those in soil (Paine et al., 1993). Crickets collected from a hazardous waste site were reported to have body burdens of only 0.31 mg/kg while shield-backed bugs and ants had body burdens of 13.9 and 60 mg/kg (Watson et al., 1985).

The acute toxicity of Aroclor 1254 to earthworms has been estimated to have a 5–day LD_{50} of 251 mg/kg in soil, corresponding to a whole body concentration of 930 mg/kg wet weight (Rhett et al., 1989), and a 14-day LC_{50} of 240 mg/kg in soil. The threshold body mass for reduced immunologic competence in earthworms has been reported to be 76 mg/kg (dry weight) of Aroclor 1254 (Rodriguez-Grau et al., 1989). The LD_{50} of Aroclor 1254 in house crickets was estimated to be 1,200 mg/kg (dry weight); however, PCB soil levels up to 9,320 mg/kg did not adversely affect crickets (Paine et al., 1993).

Terrestrial invertebrates are a predominant food source for many organisms higher up the food chain and thus the uptake and accumulation of PCBs by terrestrial invertebrates may be of potential ecological concern where soils are contaminated with PCBs. The data presented above, however, suggest that terrestrial invertebrates are less sensitive to the toxic effects of PCBs than are aquatic invertebrates.

2.3.4 Toxicity to Fish

Laboratory and field studies have been conducted to evaluate the effects of PCBs in fish (Harding and Addison, 1986; Halter and Johnson, 1974; Hansen et al., 1974; Nebeker et al., 1974; Mauck et al., 1978). Results of available studies indicate that PCBs make their way into fish by one of two major exposure routes: 1) by direct uptake from the water column and/or sediment and 2) by ingestion of contaminated prey (i.e., food web transfer). In reviewing the literature, it appears that concentrations of PCBs in water alone in the range of 1 ug/l (1 ppb) or lower do not adversely affect the survival of adult fish or their offspring (Hansen et al., 1974; Snarski and Puglisi, 1976). PCBs are hydrophobic and tend to adsorb to sediments rather than remaining dissolved in the water column. Accumulation of PCBs in fish as a result of food web transfer or secondarily, from incidental uptake of sediments, has been shown to affect the health of fish. Consequently, water concentrations alone may not be the best predictors of the potential toxicity of PCBs in fish; rather, tissue concentrations provide a better measure of exposure.

Once assimilated, PCBs tend to sequester in the fatty tissues (Lieb et al., 1974). Therefore, the degree of PCB uptake appears to be dependent upon the lipid content of different fish species, in addition to their behavior and the degree of chlorination of the PCBs to which the fish are exposed (Opperhuizen and Schrap, 1988; Rasmussen et al., 1990; Connolly, 1991). While uptake by fish appears to be rapid, elimination is slow and appears dependent upon species and gender. Elimination half-lives have been reported to range from 50 days to 92 weeks (Guiney et al., 1979; Vodicnik and Peterson, 1985; DeKock and Lord, 1988), depending upon the PCB and fish species studied.

In general, the acute toxicity of PCBs to fish varies with PCB mixture, species, age, sex, and condition. PCBs are more toxic to early life stages than to later life stages (Nebeker et al., 1974; Schimmel et al., 1974; Mauck et al., 1978) and toxicity varies among fish species (Hansen et al., 1971; Mayer et al., 1977). LC_{50}s derived from tests of longer duration are usually lower than those derived from shorter tests, indicating that PCBs are cumulative toxicants in fish (Harding and Addison, 1986). LC_{50}s range from 0.93 ug/l for Aroclor 1254 in sheepshead minnows to 15 ug/l for Aroclor 1242 in fathead minnows. The only LC_{50} reported for Aroclor 1260 is 3.3 ug/l in fathead minnows. The NOEC for Aroclor 1260 has been reported to be 1.3 ug/l (DeFoe et al., 1978).

While a number of studies have been undertaken to determine if PCBs affect the growth and survival of adult fish, it appears that these may not be critical toxicological endpoints. Rather, it appears that fertilization success, hatching success, and survival immediately post-hatch are more critical endpoints in fish. Sublethal effects such as alterations in normal gonad development and spawning may manifest themselves in adult fish, and maternal transfer of PCBs to the eggs has been illustrated by a substantial decrease in the body burdens of the adult female (Butler and Schutzmann, 1979; Niimi, 1983; Harding and Addison, 1986; Larsson et al., 1993). Spawned eggs may not be competent to be fertilized and those eggs which do become fertilized may not survive or hatch normally to develop into juvenile fish, especially in those species where a yolk sac is absorbed by the fry (Spies et al., 1994). Several authors have reported that concentrations of PCBs that affect the survival of offspring do not seem to affect the health of adult fish (Nebeker et al., 1974; Schimmel et al., 1974; Mauck et al., 1978).

Several authors have also evaluated the relationship between adult body burdens and spawning success. Bengtsson (1980) reported delayed spawning and hatchability in adult cyprinid minnows with whole body tissue concentrations of 170 mg/kg. Similar results were reported by Holm et al.

(1993) who observed lower spawning success in three–spined stickleback that had whole body concentrations of 289 mg/kg. No effect in spawning success was observed with sticklebacks having tissue concentrations of 102 mg/kg or lower.

A substantial body of evidence is available from field studies showing a relationship between PCB contamination of maternal fish and the hatchability of their eggs. Baltic flounder with ovaries containing more than 120 ng/g PCB exhibited a reduction in the viable hatch of their eggs, where viable hatch was measured as fertilization x hatching success (von Westernhagen et al., 1981). Johansson et al. (1970) found that coho salmon with PCB levels of 7.7 to 34 ug/g (on a lipid weight basis) exhibited a correlation between PCB levels and an egg mortality rate of 29 to 100% (measured as unfertilized eggs and non-viable embryos). In another study, Lake trout captured in the Great Lakes exhibited a significant relationship between embryonic mortality (measured as egg death) and the concentration of total PCBs in the eggs and adults (Mac et al., 1993). These authors also reported a negative correlation between PCB concentration and the percentage of eggs hatching successfully; however, they did not observe a relationship between PCBs and fry mortality.

Several additional studies have indicated an association between PCBs in fish eggs and adverse effects on both the hatching success of the eggs and the subsequent survival of fry. Monod (1985) collected charr from Lake Geneva and reported a significant and positive correlation between PCB levels in eggs and total mortality rates (measured from fertilization through to yolk sac resorption) provided that the concentrations of PCBs were expressed on a lipid weight basis. When total mortality was broken down into mortality taking place at different phases of development, there was a significant correlation between PCB levels and mortality in eggs during the early stages of embryo development, but no significant correlation was associated with later phases of development (Monod, 1985). Hansen et al. (1974b) reported that an average concentration of 170 µg/g of Aroclor 1254 in eggs of sheepshead minnows significantly reduced hatching success while 27 µg/g did not. They also reported that the survival rate of fry hatched from eggs containing 7.0 µg/g and higher was significantly reduced compared to the controls. The level of impact of PCB concentrations in eggs on reproductive success may be species-dependent, however, since 0.314 µg/g of Aroclor 1254 in lake trout eggs significantly reduced hatching from 67% to 29% and reduced survival of fry from 93% to 75% (Mac and Edsall, 1991). They further reported that egg concentrations of 0.173 µg/g did not significantly reduce either hatching or fry survival. Adverse reproductive effects were also associated with the presence of Aroclor 1254 in the eggs of the winter flounder (Black et al., 1988) Ankley (1991) correlated the PCB content of eggs from 10 Chinook salmon with their hatching success and found a statistically significant inverse relationship. In eggs from 4 of the 10 fish, PCB concentrations from 4 to 6 µg/g reduced hatching success from over 85% to less than 60%. The total PCB content of eggs of the starry flounder was also inversely correlated with embryological and hatching success (Spies et al., 1985).

To summarize, PCBs are more toxic to fish during their early life stages, from hatching to the period when fry are losing their dependance on the yolk sac and are beginning to feed independently. Therefore, the effects of PCBs on fish can be reasonably evaluated by measuring reproductive success. Impaired reproduction, low survival of fry, or reduced fitness of adult fish can also be related to the structure of a fish community within an aquatic ecosystem.

2.3.5 Toxicity to Amphibians and Reptiles

Amphibians and reptiles may be exposed to PCBs in an aquatic system. While amphibians are likely to be exposed via a combination of pathways including ingestion, dermal contact with water and sediment, maternal transfer to the eggs, and exposure of the eggs to contaminated sediments after oviposition, reptiles are primarily exposed to PCBs via the food web and maternal transfer to their eggs (Bryan et al., 1987a,b). Reptiles tend to bioaccumulate PCBs to a greater degree than amphibians (Olafsson et al., 1983, Niethammer et al., 1984). This may be due to their higher positions in the food web (Niethammer et al., 1984; Roger, 1990) or to the fact that certain amphibians have a high rate of fatty acid renewal which may increase their rate of PCB depuration relative to their rate of PCB uptake (Brown, 1964). However, despite their tendency to bioaccumulate higher levels of PCBs than amphibians, reptiles appear to be insensitive to their toxic potential (Bryan et al., 1987; Olafsson et al., 1983). Olafsson et al. (1983) reported that no adverse effects were observed in snapping turtles sampled, despite PCB tissue concentrations as high as 3,608 mg/kg.

Birge et al. (1978) reported that amphibians are less sensitive to PCBs than fish; however, it appears that the hatching success and post-hatching survival of amphibians may be adversely impacted when their eggs are exposed to water column concentrations of 1 µg/L or more. Birge et al. (1978) reported a LOAEL of 1.0 µg/L and a NOAEL of 0.1 µg/L in their study system.

However, direct comparisons with control amphibians were not provided by the authors and the use of acetone as a carrier solvent may have enhanced the bioavailability of PCBs tested or may be, itself, contributing to toxic effects (Mac and Seelye, 1981).

The only study which specifically addresses the potential estrogenic effects of PCBs in amphibians or reptiles relates to the modification of temperature-dependent sex determination in turtles. Bergeron et al. (1994) undertook a study to evaluate the estrogenic effects of certain PCBs by reversing sex in the red-eared slider turtle. Of the 11 PCB congeners tested in this study, only two, 2',4',5-trichloro-4-biphenylol and 2',3',4',5'-tetrachloro-4-biphenylol, were shown to significantly ($p<0.001$) reverse sex at a male producing temperature. It should be noted, however, that tri-and tetra-chlorinated biphenyls are not components of Aroclor 1260, the primary PCB mixture found in the Housatonic River sediments.

Potential reproductive effects, including hatching success and hatchling survivability, appear to be the most sensitive toxic endpoint of PCBs in amphibians and reptiles. In addition, despite their higher potential to bioaccumulate PCBs, reptiles appear to be considerably less sensitive to these toxic effects than amphibians. For these reasons, it appears that evaluation of reproductive effects in amphibians is the most critical endpoint to be considered when evaluating the effects of PCBs in these groups of organisms.

2.3.6 Toxicity to Birds

Food ingestion is the principal pathway through which insectivorous, carnivorous, and piscivorous birds are exposed to PCBs (Niethammer et al., 1984; Stromberg et al., 1994; Focardi et al., 1988). Herbivorous birds may be exposed to PCBs through incidental ingestion of soil or sediment particles, as they forage and preen.

While acute toxicity (LD_{50}) values have been reported to range from 604 to more than 6,000 mg/kg PCBs in diet (USFWS, 1985a), the vast majority of research during the past two decades has focused on sublethal, chronic endpoints. These endpoints include: hepatotoxicity and enzyme induction; neurotoxicity; endocrine and growth effects; immune system effects; mutagenicity;

reproductive impairment; and teratogenicity. While such effects have all been observed in the laboratory at high PCB doses, the most prevalent effect observed in the field is reproductive impairment, including embryotoxicity and aberrant parental incubation behavior.

Reproductive impairment has been attributed to higher chlorinated PCBs, such as Aroclor 1254 and 1260, since the 1970s. Three principal causes of reproductive impairment have been hypothesized -- eggshell thinning, abnormal parental incubation behavior, and embryonic mortality. However, repeatable and positive results are reported for PCBs only with respect to the latter two. Numerous laboratory studies (Heath et al, 1972; Peakall, 1971; Tucker and Crabtree, 1970; Scott et al., 1971; Dahlgren and Linder, 1971; Custer and Heinz, 1980; Riseborough and Anderson, 1975; McLane and Hughes, 1980) and field studies (Faber and Hickey, 1973; Lindvall and Lowe, 1980; Wiemeyer et al., 1988, 1993; Norheim and Kjos-Hanssen, 1984) on eggshell thinning in relation to PCB exposure have yielded negative or inconclusive results.

Many researchers have investigated the effects of PCBs on hatching and fledging success of young birds. Both negative and positive results have been reported and there is considerable variability among different species of birds and different PCBs. For example, reported dietary concentration NOAELs and LOAELs for this endpoint range from 3.0 mg/kg to 50 mg/kg and from 10 to 33 mg/kg, respectively. Summarized below are key laboratory feeding studies, egg injection studies, parental behavior studies, and field studies that addressed hatching and/or fledging success.

McLane and Hughes (1980) recorded clutch sizes and hatchability in captive screech owls fed 3.0 mg/kg Aroclor 1248. No adverse effects were observed ($p>0.05$). Custer and Heinz (1980) fed nine-month-old mallards a dietary dosage of 25 mg/kg Aroclor 1254 for at least one month before egg-laying and then allowed hens to incubate their own eggs. No detrimental effects on the number of hens laying, the date of first egg laid, clutch size, fertility or hatching and survival of ducklings were observed. Likewise, no significant reproductive effects were observed during long-term dietary exposures of 50 mg/kg Aroclor 1254 in Japanese quail and Northern bobwhites (NAS, 1979). In contrast, Peakall et al. (1972) observed adverse effects on hatching and fledging success in ring doves fed Aroclor 1254 at a dietary concentration of 10 mg/kg over two generations. Peakall and Peakall (1973) found that reproductive success was dependent on exposure of the female, while exposure of the male was unimportant. American kestrels given 33 mg/kg of dietary Aroclor 1254 for 62 to 69 days showed a significant decline in sperm concentration (Bird et al., 1983).

A number of egg injection studies have compared the embryotoxicity of different PCB congeners to different avian species (Brunstrom and Reutergardh, 1986; Brunstrom, 1988; Brunstrom et al., 1990; Brunstrom, 1991). It is worth noting that Hoffman et al. (1987) believed that injection studies generally suggest a greater sensitivity to foreign substances than reported in the wild and, as such, are probably not directly comparable to natural environmental accumulation in the egg. In addition, because these eggs are artificially incubated, the results offer no information on the possible effects of aberrant parental behavior on the survival of embryos. However, the results of such studies are useful in demonstrating the variability in sensitivity among different bird species.

Brunstrom and Reutergardh (1986) injected the yolks of eggs of pheasant, mallard, goldeneye, and black-headed gull with 3,3',4,4'-tetrachlorobiphenyl. The highest dose given was 0.1 mg/kg egg (mallard) or 1.0 mg/kg egg (pheasant, goldeneye, black-headed gull). Only in the pheasants' eggs was any effect on hatchability noted. While injection of 1.0 mg/kg resulted in the death of all the pheasant embryos, no decrease in the hatching rate occurred when 0.1 mg/kg was injected. Brunstrom (1988) next injected the yolks of eggs from duck, goose, herring gull and various breeds of chicken with 3,3',4,4'-tetrachlorobiphenyl. A NOAEL was not measured for chickens and the LOAEL was 5 μg/kg egg. In contrast, the highest dose administered (5,000 μg/kg egg for ducks and 1,000 μg/kg egg for geese and herring gulls) to other species did not affect the viability of the embryos and caused no gross abnormalities.

Brunstrom et al. (1990) tested the embryotoxicity of 3,3',4,4'-tetrachlorobiphenyl and 3,3',4,4',5-pentachlorobiphenyl by injecting either 1.0 or 0.1 mg/kg egg, respectively, into the yolk sacs of eggs of common eiders. No significant increase in embryo mortality was detected. Through comparisons with previous data, Brunstrom et al. (1990) showed that eider embryos are considerably less sensitive than chick embryos to either congener. Brunstrom (1991) also compared the toxicities of various polychlorinated biphenyls in chick embryos, and found that 3,3',4,4',5-pentachlorobiphenyl was the most toxic of 11 PCBs tested, followed by 3,3',4,4'-tetrachlorobiphenyl.

The relatively few studies that considered parental behavior as an endpoint demonstrate varied sensitivity among different species of birds. A dietary NOAEL of 25 mg/kg was reported for mallards (Custer and Heinz, 1980), while a dietary LOAEL of 10 mg/kg was reported for ring doves and mourning doves (Peakall and Peakall, 1973; Tori and Peterle, 1983). Custer and Heinz (1980) fed nine-month-old mallards a dietary dosage of 25 mg/kg Aroclor 1254 for at least one month before egg-laying and then allowed hens to incubate their own eggs. No detrimental effect on the hatching of fertile eggs or the survival of ducklings to three weeks of age were noted.

In contrast, Peakall and Peakall (1973) maintained second generation ring doves on a 10 mg/kg diet of Aroclor 1254 and observed the incubation behavior of pairs of treated female/untreated male and untreated female/treated male birds. Using egg temperature as a measure of nest attentiveness, the authors found that this dosage of Aroclor 1254 adversely affected both hatching and fledging success and was correlated with lower egg temperatures. Tori and Peterle (1983) observed the courtship behavior and reproductive effort of mourning doves fed dietary Aroclor 1254 of 0, 10, or 40 mg/kg for six weeks. These authors reported that doves fed 10 mg/kg spent twice as much time as controls in the courtship phase, but only 50% of these pairs completed the courtship phase and progressed into nest building and incubation.

In addition to the laboratory studies discussed above, a number of field studies have evaluated potential effects of PCBs on the breeding success of birds. Findings have varied considerably, depending on the species studied, the presence of co-contaminants, and the study design. Among the negative studies is work by Harris and Osborn (1981), which showed that female puffins that were exposed to PCBs and laid highly contaminated eggs (10-81 mg/kg PCBs) were indistinguishable in breeding success from control birds. Custer et al. (1983) studied black-crowned night-herons nesting in three New England and two North Carolina colonies and found that concentrations of DDE and PCBs in the eggs (ranging from 0.8 to 10 mg/kg wet weight) did not affect clutch size, number of eggs hatched, or nestling survival. Similarly, Struger and Weseloh (1985) found that concentrations of PCBs ranging from 19-39 mg/kg in eggs of Caspian terns did not result in impairment of reproduction. Investigations at four great blue heron colonies on the coast of British Columbia (Elliott et al., 1989) also have not demonstrated a positive association between PCB concentrations in eggs and adverse reproductive effects. Weseloh et al. (1990) observed normal herring gull productivity, despite concentrations of PCBs in eggs that ranged from 35 to 140 mg/kg (wet weight). King et al. (1991) found that PCB residue levels were not correlated with hatching success for Forster's terns and black skimmers in Lavaca Bay, Texas. Harris et al. (1993) evaluated hatching and fledging success of Forster's terns in Green Bay, WI and found no significant differences relative to a control colony at Lake Poygan. Stromberg et al. (1994) evaluated hatching success and sex ratios among deformed double-crested cormorants in Green Bay, WI with respect levels of PCBs; no relationship between PCB concentrations and reproductive outcome was found for eggs collected from successful, unsuccessful, and deformity-containing nests.

As described below, there have also been a number of field studies on the effects of PCBs on reproductive success that have had positive findings: however, these studies do not generally provide sufficient chemical and reproductive data to demonstrate that a causal relationship exists, since the presence of co-contaminants is commonly a confounding factor. Nonetheless, the ability to detect adverse reproductive outcome through field studies is an important assurance that the methodology is adequately sensitive for evaluating this endpoint.

Hoffman et al. (1987) collected Forster's tern eggs from an area with documented reproductive problems and compared them to eggs from an area where reproduction was normal. Adverse effects on hatching success, weight of hatchlings, increased liver to body weight ratio, and shorter femur length were reported in eggs that had measurable levels of PCBs and dioxins. Working with the same colonies of Forster's terns, Kubiak et al. (1989) also observed effects on incubation period, hatching success, hatchling weight and liver/body weight ratios. Kubiak et al. (1989) concluded that parental inattentiveness contributed to low reproductive success of Green Bay terns, based on their observation that a higher percentage of nests used by contaminated adults were abandoned or disappeared than occurred for uncontaminated adults. No statistically significant difference in the number of eggs that yielded fledging young was found. Very high concentrations of a multitude of man-made chemicals are present in Green Bay and statistical correlations were not conducted to determine the strength of the association between PCBs and reproductive impairment. Furthermore, in a more recent study of the same Forster's tern colonies, Harris et al. (1993) found that hatching and fledging success did not differ significantly from the control colony. The authors attributed their findings to reduced PCB loading to Green Bay.

Tillet et al. (1992) evaluated the effects of PCBs on double-crested cormorant eggs collected from the Great Lakes. Total concentrations of PCBs in the samples ranged from 0.05 to 14.8 mg/kg (wet weight). Mortality of cormorant embryos was between 9 and 39%. While there was a significant correlation between total concentrations of PCBs in composites of cormorant eggs from a colony and egg mortality rates from the same colony, the coefficient of determination (R^2) was only 0.319. Thus, much of the variance in egg mortality was unexplained. Becker et al. (1993) found that the hatching success of a highly contaminated common tern colony was similar to that of a less contaminated colony, and well within the range reported elsewhere. However, in comparing eggs sampled at random with eggs failing to hatch, there were significant differences between the amounts of total PCBs. In the failed eggs, concentrations of higher chlorinated PCBs were 20% higher than in the random sample.

In conclusion, while hepatotoxicity and enzyme induction, neurotoxicity, endocrine and growth effects, immunodeficiency, mutagenicity, and teratogenicity have been observed in laboratory studies to result from exposure of birds to high doses PCBs, the most sensitive effect consistently reported in both field and laboratory studies is reproductive impairment. This suggests that, except in extreme cases of acute poisonings, reproductive outcome is the endpoint most likely to be associated with actual concentrations of PCBs in the environment, and is, therefore, the most important endpoint to consider when evaluating risks to birds inhabiting the Housatonic River ecosystem.

2.3.7 Toxicity to Mammals

In general, PCBs are not particularly toxic to mammals following acute exposures, yet chronic exposure to many of the higher-chlorinated, planar congeners, as well as to the more highly chlorinated mixtures, has been demonstrated to result in frank effects on the reproductive abilities of mammals, especially of mink. Of mammals that have been tested for susceptibility to the effects of PCBs, mink have been shown to be most sensitive to the effects of PCBs (Ringer, 1983; Wren, 1991). The reason for this sensitivity is not known; however, interspecies variability in sensitivity to PCBs is common, even among closely related species (EPA, 1993).

Likely because of the sensitivity of minks to PCBs, most PCB ecotoxicity studies on mammals have focused on this species. Several studies have indicated that female adult mink (Platonow and Karstad, 1973; Bleavins et al., 1980) may be much more sensitive to impaired reproductive functions associated with PCB exposure than are males (Wren et al., 1987). It is also probable that the transfer of PCBs through lactation to whelps is at least partially responsible for the susceptibility observed in this age group (Bleavins et al., 1981; Brunstrom et al., 1991). There is limited evidence that laboratory species are less sensitive to the impacts of PCBs than are wild species (Linzey, 1987); it has been postulated that the likely exposure of wild mammals to multiple stressors not experienced by laboratory animals may compromise their ability to resist PCB toxicity (Wren et al., 1987).

Several investigators (Aulerich et al., 1971, 1973; Platonow and Karstad, 1973; Aulerich and Ringer, 1977; Hornshaw et al., 1983; Aulerich et al., 1986) have observed that the toxicity of metabolized PCBs (those in Great Lakes fish or in diets comprised of contaminated beef or rabbit) was greater than that observed for technical grade PCBs administered directly. However, many of these studies have flaws that potentially confound such conclusions. For example, the control mink in the Platonow and Karstad (1973) study suffered poor reproductive performance, and the control diets were shown to be contaminated with pesticides and other chlorinated organic compounds. Similarly, the Great Lakes fish diets of the Hornshaw et al. (1983) study were not analyzed for other contaminants, so it is not certain that the observed effects in this study can be totally attributed to PCBs.

Studies in a variety of mammal species have demonstrated that PCBs are not as toxic following acute exposures as they are following long-term exposures to lower concentrations of the higher-chlorinated, coplanar congeners, or to the more highly chlorinated mixtures (Ringer, 1983). Mortality (LD_{50}s) associated with acute dietary exposures in mink ranged from 1,000 mg/kg for Aroclor 1221 to 4,000 mg/kg for Aroclor 1254 (Aulerich and Ringer, 1977); LC_{50}s in rats have ranged from 1,000 to 20,000 mg/kg for various Aroclors (Kimbrough et al., 1978). Lethal dietary concentrations (LC_{50}s) associated with short-term exposures in mink have ranged from 31.5 mg/kg for exposures to Aroclor 1254 (Hornshaw et al., 1986) to 84 mg/kg for exposure to metabolized Aroclor 1254 (Aulerich et al., 1986). Exposure to lower concentrations of certain PCBs over long periods can also be fatal to mammals. Ringer et al. (1981) found that chronic exposure to dietary levels of Aroclors 1242 and 1254 between 5 and 10 mg/kg resulted in 50% mortality in mink; estimated LC_{50}s calculated for these data were 8.6 and 6.65 mg/kg, respectively.

Many of the more highly chlorinated commercial mixtures of PCBs have the potential to affect the reproductive abilities of mammals, especially of mink. For example, while chronic dietary Aroclor 1254 concentrations of 2.0 or 2.5 mg/kg resulted in significant adverse reproductive effects in mink (Aulerich and Ringer, 1977; Aulerich et al., 1985), chronic exposure to the same concentration of Aroclor 1016, 1221, and 1242 did not. Dietary exposure to 5 mg/kg of Aroclor 1242 was detrimental to reproduction (Aulerich and Ringer, 1977). Concentrations as high as 20 mg/kg of Aroclor 1016 had no effect on litter size in chronically exposed mink (Ringer, 1983). While the results of the Platonow and Karstad (1973) and Hornshaw et al. (1983) studies indicated that dietary concentrations of Aroclor 1254 as low as 0.64 or 0.48 mg/kg may impair reproduction, the many confounding factors in this study preclude the results from being considered conclusive.

Of all the congeners examined, 3,4,5-hexachlorobiphenyl, appears to be the most potent reproductive toxicant in mink; concentrations of 0.1 mg/kg and 0.5 mg/kg resulted in complete reproductive failure (Aulerich et al., 1985) and studies conducted on the relative reproductive potency of various commercial mixtures and congeners confirmed that 0-*ortho* and mono-*ortho* chlorinated biphenyls have the greatest adverse effects on mink reproduction (Backlin and Bergman, 1992; Kihlstrom et al., 1992).

The most sensitive effects associated with chronic exposure to PCBs in mammals appear to be the survival and growth of young (Aulerich and Ringer, 1977; Wren et al., 1987, Linzey, 1987, Wren, 1991). Hornshaw et al. (1983) observed inferior kit survival associated with maternal dietary exposures to Aroclor 1254 in fish at concentrations as low as 0.48 mg/kg. Wren et al. (1987) observed depressed growth in kits born to dams exposed to 1.0 mg/kg Aroclor 1254 in their diets.

The specificity of the effects of PCBs on mink reproduction and mortality is well established from laboratory experiments, but Wren (1991) found there was poor resolution of the information on effects using field data. Comparison of tissue levels observed in toxicity studies with those observed in the field is useful, as long as assessments of the health of wild mammals are made as well. Unfortunately, most of the studies that evaluated tissue levels in wild mammals (O'Shea et al., 1980; Henney et al., 1981; Foley et al., 1988) include little information on the health of individuals or populations. However, Wren et al. (1987) noted that the maximum liver level (3.1 mg/kg) observed in female mink that experienced reproductive failure in laboratory studies was lower than PCB levels that have been observed in wild mink (O'Shea et al., 1980; Henney et al., 1981; Foley et al., 1988). Wren et al. (1987) felt this comparison was important because it indicated that reproduction could be affected in wild mink if they have high tissue levels of PCBs.

In conclusion, chronic exposures of mammals to highly chlorinated mixtures of PCBs have been shown to cause reproductive impairment, particularly with respect to survival and growth of young. Mink are substantially more sensitive to these effects than most other mammals and, as a result, most PCB ecotoxicity studies have focused on minks. Therefore, it is reasonable to conclude that a critical mammal for evaluation is mink and that the critical effects are reproductive impairment and survival of young.

2.4 Selection of Receptors of Concern

An ecological receptor may be a species or group of species that is potentially affected by the presence of on-site stressors. While a multitude of ecological receptors may be observed or expected to be present in the Housatonic River site, in-depth evaluation of the health of all potential receptors present is not feasible. Therefore, species that are sensitive to and likely to be exposed to the chemical stressor of interest, and are representative of the variety of organisms and types of exposures present in the study area, need to be selected for study. By narrowing the number of species assessed, more careful evaluation of their health can be performed with a resulting increased certainty in the conclusions.

The selection of species for evaluation is one of the first and most important tasks in ecological risk assessment (EPA, 1991b). The choice of species is influenced by the needs of the study, the natural or available habitat on-site, the types of organisms that could be affected by the presence of contaminants, the distribution of contaminants, and the variable susceptibility of those organisms to the toxic effects of contaminants at the site.

In selecting receptors for the risk assessment for the Housatonic River site, the choice of species reflects the aquatic and terrestrial resources that are potentially at risk (EPA, 1991b). In making these selections, the following criteria have been considered:

- *Sensitivity of the species and life stages to the ecotoxicological potential of PCBs* - Evaluation of particularly sensitive species and life stages yields a conservative assessment of potential risk.

- *Size and location of the home range relative to the area of contamination* - Ideally, the home range of selected species should correspond to the area of contamination, because such species would experience among the highest exposures. Species which have home ranges that exceed the area of contamination should not necessarily be dropped from consideration, however, particularly if that species is sensitive to the effects of the contaminants.

- *Feeding preferences and behaviors* - Receptors should be selected that have a high potential for exposure, either because of the foods that they consume or because of behaviors such as nesting, burrowing, and preening.

- *Position on the food chain* - Selection of mid-to-high trophic level consumers allows for greater potential for bioaccumulation and biomagnification up the food chain. In addition, by focusing on higher level organisms, it is possible to indirectly evaluate potential impacts on the habitats and prey organisms on which those species depend.

- *Ecological value* - The extent to which the receptor group is critical to the function and structure of the community or ecosystem reflects its ecological value. For example, a receptor species or group that is the obligatory food source for another species or group, or is critical to the habitat requirements of an endangered species, or is critical to nutrient cycling would have high ecological value.

- *Social importance or economic value* - Species that may have high social or economic value include those that are valued for their aesthetic qualities (certain bird or plant species), as well as those that are valued as recreational resources (hunting, fishing, etc.)

- *Sufficient population density to allow for meaningful evaluation* - For field studies, species should be selected that are present in sufficient numbers to be identified and evaluated. Population sizes that are sufficient to test for statistical significance further augment the meaningfulness of the study.

- *Threatened, endangered, or rare species* - Any threatened, endangered or rare species that inhabit the site will be included as receptors of concern, in accordance with the MCP.

- *Minimization of confounding factors* - Receptors for which confounding factors that could confuse or mask the causes of any observed population or reproductive effects are minimized (e.g., hunting or trapping could reduce population sizes and skew the sex and age structures of populations of some species) help to limit the overall uncertainty of the assessment; and

- *Overall feasibility* - Characteristics, such as transient inhabitation of the site and absence of data in the literature, may preclude certain species from being studied for reasons of technical infeasibility.

There are a number of candidate receptors that could be evaluated in the risk assessment for the Housatonic River site. Each is discussed below, along with their strengths and weaknesses as candidates for evaluating the potential impacts of PCBs on the ecosystem of the Housatonic River valley. Receptors of interest are then selected based on the criteria described above.

2.4.1 Plants

As primary producers, plants constitute the base of all food webs. They are ubiquitous and may, under certain circumstances, be among the first organisms affected by on-site stressors (EPA, 1991). However, as discussed in Section 2.3.2 and as stated by EPA (1991), plants are not useful receptors of interest for hydrophobic compounds like PCBs, due to the fact that they cannot transport substantial amounts of these compounds, limiting their potential for uptake, and thus may react to exposure less rapidly.

Moreover, PCBs do not appear to be toxic to plants except at water concentrations greater than 1 µg/l and soil concentrations in the hundreds of mg/kg, both of which are higher than the levels that have generally been detected in the Housatonic River ecosystem (Section 2.1.2). While plants do comprise a critical foundation of the food web, the health of the plant community as the base of the food web will be evaluated indirectly through the assessment of potential risks to higher trophic level organisms which depend on it (as discussed below). For this reason, and because of their low uptake and lack of sensitivity to PCBs, plants have not been selected as receptors of concern for this assessment.

2.4.2 Invertebrates

Due to the low solubility, high lipophilicity, and low degradability of PCBs, they tend to bioaccumulate in both aquatic and terrestrial invertebrates. In aquatic systems, benthic invertebrates are capable of mobilizing PCBs from sediments as they ingest sediment-associated detritus. Similarly, in terrestrial systems, earthworms and other invertebrates may mobilize PCBs from soils as they ingest organic matter to which PCBs have adsorbed.

As discussed in Section 2.3.3, aquatic invertebrates appear to be more sensitive to PCBs than terrestrial invertebrates. Moreover, in the Housatonic River, the sediment PCB levels are generally higher than those in the flood plain soil. As a result, it makes sense to access potential PCB effects on aquatic invertebrates. Further, since highly chlorinated PCBs are more concentrated in sediments than in the water column, benthic invertebrates are exposed to greater PCB concentrations than are pelagic or planktonic organisms.

Benthic invertebrate community data lend themselves well to evaluating trends in water quality since changes in species richness, diversity, balance, abundance, and presence/absence of tolerant or intolerant species can be correlated to the quality of aquatic ecosystems (Bode et al., 1993). A diversity index is commonly employed as a measure of the relative health or stress of aquatic invertebrate communities (Klemm et al., 1990). The advantages of using benthic macroinvertebrates include their sensitivity to environmental impacts, their susceptibility to localized contamination due to their relative immobility, their abundance, and the fact that they are fairly easy and inexpensive to sample. Bioconcentration by benthic organisms is the first step in the biological transport of hydrophobic pollutants from the sediments to higher trophic levels, including fish, birds, mammals and humans (Lee, 1992). Benthic macroinvertebrates can accumulate PCBs to concentrations thousands of times higher than those in water samples (Gooch and Hamdy, 1983).

Aquatic invertebrate communities can be affected by factors such as sediment grain size, aeration, habitat, food source, water flow, temperature and water quality. To make comparisons between different sites, these factors should be accounted for as much as possible. Choosing sampling sites in areas of comparable habitat and sediment deposition in addition to co-sampling the sediments for benthic invertebrates, grain size, TOC, and redox potential will aid in determining whether differences in community parameters can be significantly correlated to chemical contaminants.

In terrestrial systems, earthworms and many other invertebrates live in direct contact with soil and may come into contact with contaminants that are either adsorbed to soil particles or dissolved in soil moisture. For example, earthworms ingest large quantities of soil during their normal life cycle and may encounter elevated concentrations of PCBs in localized surface soil areas, mobilizing them from soil as they ingest organic matter to which the PCBs have adsorbed (EPA, 1991). Many birds and mammals feed on terrestrial invertebrates; therefore these higher trophic level organisms may accumulate PCBs via the food chain.

While both aquatic and terrestrial invertebrates have been considered for evaluation, the data available on the effects of PCBs on aquatic invertebrates is much more substantial than the data for terrestrial invertebrates. This allows the data collected for the aquatic invertebrates in the Housatonic River to be compared with the data in the published literature. Further, as noted above, terrestrial invertebrates are less sensitive to the toxic effects of PCBs than aquatic invertebrates. Therefore, a risk assessment that is based on aquatic invertebrates should be protective of terrestrial invertebrates as well. For all these reasons, we have selected aquatic invertebrates as receptors of concern for this risk assessment.

2.4.3 Fish

Fish may come into contact with PCBs dissolved in the water column; however, this is not the primary route of exposure for most species. Bottom feeding fish may take up PCBs via the ingestion of benthic invertebrates and the direct ingestion of sediments during feeding. Predatory fish ingest PCBs while feeding on smaller fish.

There is a substantial body of data on the chemical sensitivity of fish to PCBs. Sensitivity to PCBs varies depending upon the life stage of the fish and the species being evaluated. Identification of species is straightforward and there is information readily available on their environmental requirements and life histories. Fish are relatively long lived and they may occupy many trophic levels. In addition, most temperate species of fish reproduce once a year resulting in relatively stable populations (EPA, 1991). It can be difficult to study fish, however, because they are mobile and are not confined to localized areas of contamination, quantitative samples are difficult to obtain, and their numbers are fewer than those of smaller organisms, leading to a greater chance of sampling error.

Omnivorous fish are species of fish that feed on a combination of food types. They may consume plant material and detritus as well as smaller fish and aquatic organisms. These fish, which include the white sucker, bullhead, and carp, have potential to take up PCBs via the food chain and via direct ingestion of sediments during feeding. Of this group, the white sucker is common in the Housatonic River. It is an omnivorous bottom feeder that primarily consumes benthic invertebrates, small mollusks and occasionally fish eggs. This species tends to feed in pools with cobble/rock substrates and will also scrape algae and diatoms from the rocks. It has potential for direct exposure to PCBs from sediments and also for food web exposure via the ingestion of sediment dwelling organisms. White suckers typically spawn in riffle areas and broadcast the eggs, but sometimes spawn in areas where streams enter bays or pools. There are no available data regarding PCB accumulation in the tissues of white suckers within the Housatonic River at this time. The white sucker is chosen as a receptor because it is a good representative of bottom feeding fish and is abundant throughout the Housatonic River.

Predatory fish generally feed on other fish and aquatic organisms during the adult stages of their lives. Many of the predatory fish species (e.g. largemouth bass, smallmouth bass, and trout) are game fish that are actively sought by recreational anglers. Because these fish are high trophic level organisms which feed on secondary consumers, they bioaccumulate PCBs via the food chain.

While all three of these species are known to inhabit this reach of the Housatonic River, largemouth bass are the most common. Smallmouth bass populations are more limited in size, making study of this species less feasible than largemouth bass. Trout populations are believed to be very small and, due to their temperative requirements, are restricted to colder portions of the river.

The largemouth bass, provides an excellent model for studying predatory fish in the upper Housatonic River. It is a lurking predator which feeds in vegetated areas and primarily consumes fishes, as well as crayfish and frogs. Largemouth bass create and defend nests in sand/gravel substrate. There are some data regarding PCB tissue concentrations for scaled, skin-on largemouth bass fillets in the Housatonic River (Stewart, 1982; Blasland and Bouck, 1991). The Stewart (1982) data indicate that total PCB (wet wt.) in largemouth bass fillets varies from 20 mg/kg in Woods Pond to 6.9 mg/kg in the reach from Rising Pond Dam to the state line (2,597 mg/kg lipid to 920 mg/kg lipid, respectively, when expressed as lipid-normalized PCB concentrations). Other largemouth bass caught in Woods Pond and Rising Pond were reported to have fillets with mean total PCB concentrations (in mg/kg) of 10.7 (± 7.9 SD) and 16.5 (± 12.1 SD) respectively, which corresponds to 1,119 (± 739 SD) and 890 (± 599 SD) mg/kg lipid, respectively (Blasland and Bouck, 1991). Largemouth bass is chosen as a receptor for this assessment due to the high trophic level it occupies, its abundance, and its demonstrated capacity to accumulate PCBs in the Housatonic River.

2.4.4 Amphibians and Reptiles

Amphibians may be found in a variety of terrestrial and aquatic habitats. Generally, amphibians are terrestrial as adults and move to aquatic habitats to breed. Most deposit their eggs in or near the water and the eggs hatch into free swimming and feeding larvae. Eventually these larvae metamorphose into their adult forms and move back into the terrestrial habitat where they continue to feed and grow. Because of their aquatic breeding and their adult and larval feeding behaviors, these organisms demonstrate a high potential for exposure. They may be exposed to PCBs via ingestion, dermal contact with water and sediment, and maternal transfer to the eggs. In addition, because amphibians are consumed by higher trophic level organisms, including birds, mammals, reptiles, and fish, they may provide a source of PCB transfer within the food web.

Certain reptiles may also be exposed to PCBs in an aquatic system. Snapping turtles have been proposed as an effective tool for monitoring changes in contaminant levels over time, due to their long life expectancy, their lifetime residence within a well defined area, their sedentary lifestyle, their omnivorous feeding behavior, and their tendency to bioaccumulate high levels of organochlorine contaminants with no apparent adverse effects (Olafsson et al., 1983). Turtles are primarily exposed to PCBs via the food web and maternal transfer to their eggs (Bryan et al., 1987a,b).

For most amphibians and reptiles, the primary route of exposure to PCBs is likely to occur via the food web. The variety of foods eaten by these organisms can greatly affect the level of exposure that they receive. While the diets of some amphibians may be limited to organisms lower on the food web, like insects and invertebrates, other amphibians, like bullfrogs, are omnivorous and consume crayfish, insects, invertebrates, plants, and, upon occasion, small fish and mammals (Korschgen and Baskett, 1963; Niethammer et al., 1984). The diets of reptiles are similarly variable. Although small reptiles may feed primarily on small insects and invertebrates, larger snakes and reptiles, like the snapping turtle, are tertiary consumers whose diets may include crayfish, fish, frogs, small mammals and birds (Niethammer et al., 1984; Rogers, 1990).

Despite their tendency to bioaccumulate higher levels of PCBs than amphibians, reptiles appear to be relatively insensitive to the toxic effects of PCBs (Olafsson et al., 1983; Bryan et al., 1987). In addition, while snapping turtles have been studied extensively to determine their suitability as biological monitoring tools, very little information is available to indicate the dietary, sediment, or body burden PCB concentrations that would result in specific adverse effects in this species. In those few studies that have evaluated toxicity in snapping turtles, the turtles have generally been exposed to multiple organic and inorganic contaminants, making it impossible to identify whether the toxic responses observed are related to PCBs (Albers et al., 1986; Bishop et al., 1991). Other studies have indicated that snapping turtles can tolerate tissue concentrations of hundreds or thousands of mg/kg without any adverse effects (Olafsson et al,, 1983). Furthermore, there are essentially no studies pertaining to effects of PCBs on other reptiles that have been published to date. Finally, snapping turtles are not generally consumed by higher trophic level organisms and thus do not represent a route of PCB food web transfer to higher trophic levels. For these reasons, they have not been selected as receptors of concern for this risk assessment.

Frogs, on the other hand, have been shown to be substantially more sensitive to the toxic effects of PCBs indicating that water column PCB concentrations of 1 μg/l or higher may, in certain cases, result in decreased hatching success and post-hatch survival (Birge et al., 1978). Because of this sensitivity to PCBs, their abundance in the Housatonic River system, and their importance as a food source for higher trophic level organisms, frogs have been selected as receptors of concern for this risk assessment.

2.4.5 Birds

There are numerous birds which inhabit the Housatonic River valley, including piscivorous, carnivorous, insectivorous, and herbivorous species. As discussed in Section 2.3.6, it appears that chronic, sublethal effects of PCBs in birds are the most important, with reproductive impairment, including embryotoxicity and aberrant parental incubation behavior being the most prevalent.

Piscivorous Birds

Piscivorous birds feed largely on fish and other aquatic biota. They have a high potential for exposure, due to the fact that the primary component of their diets is fish which tend to bioaccumulate PCBs. Piscivorous birds considered for this analysis include the great blue heron, belted kingfisher, osprey, and gulls and terns.

Great blue herons are colonial nesting birds that are rare or localized in Massachusetts during the breeding season (Veit and Petersen, 1993). During the winter, great blue herons inhabit coastal areas, and they spend the breeding season along shallow shores of ponds, lakes, streams, rivers, wet meadows, and marshes. These birds build nests approximately 40 m high in tall trees, in fairly dense colonies (DeGraaf and Rudis, 1986). DeGraaf and Rudis (1986) report that dozens of nests may be built in the crown of a single tree. Great blue herons commonly opt for open canopies and exposed limbs for nesting, presumably for ease of access. They feed on aquatic and terrestrial insects, fishes, amphibians, reptiles, and crustaceans. Great blue herons generally fish in shallow water, usually less than 50 cm deep, and prefer exposed mud flats and sandbars (USFWS, 1985c).

Most great blue heron colonies along the Atlantic coast are located in riparian swamp and beaver impoundments. Islands are common nest colony sites for great blue herons, presumably due to their isolation from human habitation and disturbance (USFWS, 1985c). USFWS (1985c) reports that "human disturbance, habitat destruction, and the resulting loss of nesting and foraging sites probably have been the most important factors contributing to declines in some great blue heron populations in recent years."

Great blue herons are good receptors for ecological risk assessment for a variety of reasons. First, great blue herons nest at several locations in Berkshire County that are within foraging range of the study area (pers. comm., 1994, Mr. Tony Gola, MDFW). Second, there is suitable foraging habitat for great blue herons in the Housatonic River valley (although the most common nesting habitat, beaver impoundments, is not known to occur within the study area). Third, because herons nest colonially, the feasibility of evaluating populations, rather than single individuals, is improved. Fourth, because herons are neither secretive nor camouflaged, the feasibility of finding and observing them is improved over certain other piscivorous birds (e.g., bitterns). Fifth, reproductive data for virtually all heronries in Massachusetts for at least the past decade are available through MDFW. For the reasons listed above, the great blue heron has been selected as a receptor of interest.

Belted kingfishers are solitary birds that nest on banks of water bodies containing small fish. Fish are the staple of their diets, although crayfish, insects, mollusks, and tadpoles are also consumed (DeGraaf and Rudis, 1986). Belted kingfishers prefer to forage in clear water for an unobstructed view of their prey (USFWS, 1985b). Not surprisingly, belted kingfishers are seldom seen on ponds or streams that are overgrown with thick vegetation that would obscure their vision. The most suitable locations for nests are well-drained banks with sandy soil and vertical or slightly overhanging faces (USFWS, 1985b).

Belted kingfishers are not a good receptors of concern because few areas of the Housatonic River between New Lenox Road and Woods Pond Dam offer suitable nesting habitat. Much of this reach of the Housatonic River is occluded by masses of pondweed and water buttercup, which are a deterrent to kingfisher foraging. Woody roots, which are common in the study area, impede nest excavation. Therefore, a population of adequate size for a statistically meaningful study would be difficult or impossible to locate within the reach of interest of the Housatonic River. Additionally, there is no information published in the scientific literature regarding the toxicity of PCBs to kingfishers or related species. Finally, great blue herons are more appropriate receptors of interest than belted kingfishers because of the greater availability of information on PCB toxicity to great blue herons, as well as the availability of habitat suitable for great blue herons in the Housatonic River valley.

Ospreys nest near large bodies of water that support abundant fish populations. Shallow areas of streams and shoals of lakes where fish are close to the surface are the preferred foraging areas (DeGraaf and Rudis, 1986). However, reservoirs and larger waterbodies often provide improved foraging conditions over rivers and lakes, due to abundant fish populations and reduced turbidity. Turbid waters or waters with extensive emergent and submergent vegetation reduce ospreys' prey visibility. If prey visibility is low or food resources are scarce, ospreys travel to distant food sources from the nest (USFWS, 1987).

The Housatonic River valley is not an optimal place to locate an osprey population of adequate size for this assessment. The backwaters of the Housatonic River are choked with aquatic vegetation in many places, and dense overhanging vegetation along the shorelines limits foraging opportunities for ospreys by reducing their view of the water surface. Also, the Quabbin Reservoir near Amherst and the Atlantic coast offer better osprey habitat than the Housatonic River. Veit and Petersen (1993) indicate that ospreys breed only near or along the coast of Massachusetts, and not in the Berkshires. The natural habitat preferences of osprey, as well as their willingness to travel long distances from their nests to forage, suggest that this species is less suited than the great blue heron for use as an receptor of interest for this ecological assessment.

Gulls and terns are colonial nesting piscivorous birds seasonally common to portions of the northeastern United States. These species feed on fish, garbage, insects, worms, and grubs. Their habitats vary from marshy areas to gravelly, sandy beaches; however they all nest and forage near water bodies (e.g. lakes, rivers, and oceans). As colonial species, gull and terns are relatively easily monitored for reproductive success. However, Veit and Peterson (1993) indicate that very few gulls and terns breed in western Massachusetts; rather, most of those present in Massachusetts are migrants. In light of these factors, gulls and terns are less favorable receptors of interest than great blue herons.

Carnivorous Birds

Carnivorous birds feed on a variety of animals, including mammals, birds, fish, amphibians, and reptiles. Carnivorous birds considered for this assessment include the bald eagle and other birds of prey, including owls, falcons, and hawks. Exposure to these birds occurs primarily via the ingestion of contaminated prey.

Bald eagles are rare in New England and considered endangered nationally. In Massachusetts, the only breeding bald eagle population is located in the Quabbin Reservoir, as the result of reestablishment efforts (Veit and Petersen, 1993). Most bald eagles in Massachusetts are only winter residents. Bald eagles are very sensitive to human disturbances and insulation from interference is a habitat requirement. Fish are the staple of the bald eagle's diet, although they also feed on small to medium mammals, large birds, turtles, and carrion. Bald eagles nest in large living trees on large bodies of water (DeGraaf and Rudis, 1986). USFWS (1986b) reports that bald eagles prefer relatively open canopy, old-growth forests. According to DeGraaf and Rudis (1986), the tree size, shape, and proximity to other nesting eagles are important in determining habitat quality. Bald eagles prefer unobstructed views of their prey, as would be common over large waterbodies with low turbidity and little overhanging vegetation (USFWS, 1986b).

The Housatonic River does not offer a habitat suitable for breeding bald eagles. Not only are the trees substantially smaller than those preferred by bald eagles, but the Housatonic River is choked with aquatic vegetation in many places, and has dense overhanging vegetation along the shorelines. Thus, nesting and foraging would be difficult or impossible for bald eagles. Also, human disturbances (e.g., roads, railroad tracks, municipal landfills, gravel pits, recreational activities, etc.) are common in the Housatonic River valley, likely deterring bald eagles from nesting in the region. Bald eagles were not chosen as indicator species because the lack of suitable habitat minimizes the likelihood of their exposure to PCBs.

Owls, falcons, and hawks are large birds of prey found in New England. These birds are generally carnivorous, feeding on mice, small mammals, amphibians, and birds. Large birds of prey tend to have relatively large territory sizes, large home ranges, and low population densities. For example, barred owls range from 86.2 to 369.2 ha, and have been found in population densities of three pairs per 9,320 ha. Red-tailed hawks, which are common in Massachusetts, also have very low densities (one pair per 570 ha), large home ranges (80 to 1040 ha), and large territory sizes (32.4 to 81 ha) (DeGraaf and Rudis, 1986).

Large birds of prey are not generally suitable indicators of localized environmental pollution, because they are present in low densities and occupy large feeding territories and home ranges. These characteristics prevent the determination of the extent of exposure to chemicals within the Housatonic River flood plain. It is unlikely that large carnivorous species would obtain food only from the relatively limited area of the 10-year flood plain. Finally, exposure of carnivorous birds of prey to PCBs is expected to be less than exposures of piscivorous birds to PCBs, given the greater levels of PCBs in the aquatic system, relative to the terrestrial system. Because the prey item of piscivorous birds would be expected to have higher concentrations of PCBs than the prey items of carnivorous birds, the evaluation of piscivorous birds allows a more conservative risk assessment. Therefore, large birds of prey have not been selected as receptors of concern for this assessment.

Insectivorous Birds

The most prominent insectivorous bird species of the Housatonic River system are perching birds and song birds of the taxonomical order Passeriformes (as known as passerines). Three examples of the species included in this group are: (1) the barn swallows of the Hirudinidae family, which eat flying insects caught over ponds, lakes, rivers, and fields; (2) wood thrushes of the Turdidae family, which eat insects from leaf litter and understory vegetation; and (3) vireos of the family Vireonidae, which feed on caterpillars, moths, beetles, bugs and ants from the undersides of leaf surfaces (Wallace and Mahan, 1975; DeGraaf and Rudis, 1986).

Many passerines prefer the habitat of bottomland hardwood forests with wet areas and some edge habitat. They tend to have relatively small territory sizes and high population densities. For example, territories of red-eyed vireos range from 0.3 to 1.2 ha; Eastern phoebes have territories of approximately 0.3 ha. The population density of American redstarts has been found to be approximately 35 pairs per 40 ha, and up to 63 territorial yellow warbler males have been observed in 40 ha (DeGraaf and Rudis, 1986).

Passerines that eat invertebrates are appropriate receptors of interest for this assessment for several reasons: a) they may be exposed to PCBs through the food web because they feed on invertebrates (which uptake PCBs from soil and sediment); b) breeding populations are expected in the Housatonic River study areas because the habitat requirements for many passerine birds are satisfied; c) home ranges are relatively small and well within the limits of the target area, allowing reference and target populations to be easily distinguished; and d) sufficient sample sizes to allow for statistically meaningful conclusions can be studied due to the high densities of passerine birds.

Herbivorous Birds

Herbivorous (plant-eating) birds in the Housatonic River valley include the ring-necked pheasant, wood duck, mallard, Canada goose, and American black duck. Due to the limited uptake of the more highly chlorinated PCBs by plants, any PCB exposure to these birds through diet would be minimal. However, it is possible that herbivorous birds could have some, limited PCB exposure through incidental ingestion of soil or sediment. With the exception of the Canada goose, all of the above species are actively hunted in Massachusetts (MDFW, 1992). Canada geese are hunted in states other than Massachusetts.

Ring-necked pheasants consume cultivated grains and weed seed, as well as fleshy fruits, insects, and buds and soft parts of herbaceous vegetation. Pheasants breed in open cultivated fields of grass or grain, fallow fields, bushy pastures, hedgerows by roadsides, cut over land, and open ungrazed woodlots. While ring-necked pheasants generally avoid wooded areas during the breeding season, they prefer areas with dense protective cover in the winter (DeGraaf and Rudis, 1986).

Some populations of ring-necked pheasants have fairly high population densities, as well as relatively small territory sizes (1.2 to 5.3 ha). DeGraaf and Rudis (1986) report population densities ranging from of one to fifty birds per 40 hectares. In Massachusetts, hunting can diminish ring-necked pheasant populations; however, populations are restocked regularly (Veit and Petersen, 1993). In Berkshire County, there are few probable or possible breeding pairs of pheasants (Veit and Petersen, 1993). Most breeding pairs of pheasants occupy eastern and central Massachusetts.

Ring-necked pheasants are not a suitable receptors of interest for this assessment for several reasons. First, it would be difficult to locate a breeding ring-necked pheasant population of adequate size for a quantitative study in western Massachusetts (Veit and Petersen, 1993). Second, population levels are actively managed through hunting and restocking; therefore, the effects of hunting may be mistaken for the effects of the chemical of interest. Restocking may also confound the effects being studied. Third, given the very limited uptake of PCBs by plants, the exposure potential of ring-necked pheasants be expected to be substantially lower than that of piscivorous and insectivorous bird species, both of which have already been selected as receptors of interest. Thus, the exposure potential and sensitivity of this species is low, relative to other avian species.

Wood ducks are fairly common throughout Massachusetts, primarily due to the abundance of artificial nesting boxes (Veit and Peterson, 1993). Natural cavities inhabited by wood ducks are generally in trees with diameter at breast height (DBH) of 35-40 cm and must be in trees at least 16 cm DBH (DeGraaf and Rudis, 1986). They feed on aquatic and terrestrial insects, acorns, hickory nuts, waste grains, seeds of aquatic plants, and fleshy fruits. Acorns are their principal fall and winter food source (DeGraaf and Rudis, 1986).

Wood ducks were not selected as an receptors of interest because their herbivorous feeding behavior minimizes exposure and because populations are actively managed. As primary consumers, the potential for wood ducks to be exposed to PCBs through food web transfer is substantially lower than for higher trophic level species, such as passerines or the great blue heron. Additionally, wood duck habitat is actively managed in HVWMA due to insufficient natural nesting capacities. Although several nesting boxes have been established in the Housatonic River area to allow wood ducks to inhabit the site, it would be virtually impossible to distinguish population effects caused by PCBs from the effects of poor natural nesting opportunities and active hunting. Again, the exposure potential and sensitivity of this species is low, compared to other avian species.

In addition to wood ducks, the waterfowl expected to inhabit the Housatonic River site on a regular basis are mallards, Canada geese, and American black ducks. While less common waterfowl species, such as blue-winged teal, common goldeneye, hooded merganser and common merganser, may stop over in the Housatonic River valley during migration, the three species listed above are most representative of local waterfowl. Mallards, Canada geese and black ducks are entirely herbivorous, feeding primarily on grasses, seeds, and waste grains. All three species forage over wide areas, frequently traveling from their breeding sites in marshes and along waterbodies to feed in upland meadows, farm land, and residential parks (DeGraaf and Rudis, 1986).

Mallards, Canada geese, and American black ducks were not selected as receptors of interest because their feeding ranges extend to upland meadows, farmlands, and residential parks, they are herbivorous, and they are hunted in the Housatonic River valley. These species would be inappropriate to use for this study because potential exposures to PCBs are expected to be low, as a result of their large foraging range and food preferences. Additionally, it would not be possible to distinguish population effects potentially caused by PCBs from the effects of hunting.

2.4.6 Mammals

Mammals are the largest animals inhabiting the Housatonic River site that may have potential for exposure to PCBs. While PCBs are not particularly toxic to mammals following acute exposures, chronic exposures may result in reproductive effects in mammals (Section 2.3.7). The sensitivity of mammalian species to toxic effects of PCBs is highly variable, with mink being substantially more sensitive than other mammals that have been tested.

The primary route of exposure of mammals to PCBs in the Housatonic River valley is through the food chain, although additional exposure may occur via direct contact or ingestion during feeding, burrowing, or grooming. Exposure to PCBs is highly variable among mammals due to the high variability in feeding behaviors of mammals; they may be piscivorous, carnivorous, insectivorous, omnivorous or herbivorous.

Piscivorous Mammals

Piscivorous mammals feed, at least in significant part, on fish. This group of mammals includes the mink and the river otter, both of which are actively trapped for their pelts.

Minks are described by both Godin (1977) and Linscombe et al. (1982) as solitary and unsociable, with extremely large home ranges. They are "tireless wanderers" that may range from one watershed to another when food or water is scarce (Godin, 1977; Hamilton and Whitaker, 1979). Linscombe et al. (1982) estimate that mink population densities range from 3 to 21.9 mink per 259 ha and that average home ranges are 2,630 m in stream length. The preferred habitat of minks includes stony shores with dense vegetation and wetlands with irregular and diverse shorelines (USFWS, 1986a). They are most often found along streams where there is an abundance of downfall or debris for cover and pools for foraging; log jams provide excellent foraging cover for mink (USFWS, 1986a). The food preferences of minks are broad and include fish, frogs, aquatic insects, snakes, small mammals, muskrats, and birds (Hamilton and Whitaker, 1979). Linscombe et al. (1982) report that "no one individual food item seems to be consistently more important in the mink diet." Korschgen (1958) and Sealander (1943) reported that fish comprised approximately 20 to 30 percent of the mink's diet.

Minks are partial piscivores and are thought to be highly sensitive to PCBs (Wren et al., 1987; USFWS, 1985c). They are difficult to evaluate in field studies because: a) they are solitary, suggesting that it would be difficult to locate a target population of adequate size to render statistically meaningful results; b) they occupy extensive home ranges, suggesting that exposure is limited and making it difficult to distinguish between potentially exposed and reference populations; and c) they are subject to a wide range of physical and chemical stressors, the effects of which may not be distinguishable from the effects of PCBs as a stressor. However, because of their apparent sensitivity to PCBs, mink have been selected as a receptor of concern for this risk assessment.

River otters have also been selected as receptors of concern for this assessment. River otters inhabit borders of streams, lakes or other wetlands in forested areas (DeGraaf and Rudis, 1986), and spend much of their time in water (Godin, 1977). The well-hidden dens of river otters may be located under large roots or a fallen tree, in a crevice, in an abandoned beaver or muskrat house, or in dense thickets bordering water (DeGraaf and Rudis, 1986). Openings to dens along lakes or streams are above water in the summer and generally below the ice in the winter (Godin, 1977). River otters primarily feed on aquatic animals, such as fish, frogs, crayfish, salamanders, and turtles (DeGraaf and Rudis, 1986).

Godin (1977) describes river otters as primarily nocturnal animals that are "great travelers, known to go 100 miles or more seeking new territory." Toweill and Tabor (1982) confirm that otters regularly travel long distances, reporting that the mean distance recorded was 9 to 10 km per night. Murray (1987) also documents very large home ranges of otters (15 km in width and 18.2 to 43.9 km in length, straight-line waterway distance). Toweill and Tabor (1982) report population densities as low as 5 otters per 40 square miles.

The naturally low population densities and low reproductive rates of otters make them particularly susceptible to human influences, such as habitat loss and overtrapping (Murray, 1987). Murray (1987) asserts that "humankind has altered or destroyed much of the otter's original habitat in

North America and remains its worst enemy." A Master's Degree thesis written by Mack (1985), considering otter populations in Colorado, ascribes historic declines in populations across the nation to habitat loss and overtrapping. These human impacts are generally considered to be of greater importance than pollution; adequate evidence does not exist to support the claim that pollution is a limiting factor of otter populations in general (Murray, 1987).

It should be noted that there are some factors which may confound field studies of river otter. First, they occupy extensive home ranges and consequently would feed in both contaminated and uncontaminated areas. Second, low populations densities, resulting from naturally low reproductive rates and human disturbance of habitat, may make it difficult to locate a target population of adequate size. Finally, it may be difficult to distinguish the population effects of trapping from the effects resulting from potential exposures to chemical stressors. However, because they feed on many aquatic organisms that uptake PCBs from river sediment, and thus have a relatively high potential for exposure, they have been selected as a receptor of concern for this assessment.

Carnivorous Mammals

Carnivorous mammals feed primarily on smaller mammals, as well as birds and frogs. Carnivorous mammals in this region of the country include the long-tailed weasel and the ermine, both of which are actively trapped.

Long-tailed weasels are voracious hunters that primarily feed on small mammals, such as bats and hares, as well as birds and their eggs and frogs (Godin, 1977). Home ranges from 12.1 to 162 ha have been reported (DeGraaf and Rudis, 1986) but, according to DeGraaf and Rudis (1986), home range size varies with food availability, cover type, and season. The preferred habitats of long-tailed weasels are open woodlands, brushlands, and rocky areas along the waterways (Godin, 1977).

Long-tailed weasels are poor receptors of interest because of their habitat preferences and large home ranges. The grassy and eroded shores of the Housatonic River clearly contrast with the rocky shores preferred by long-tailed weasels. Because of their large home ranges, exposure is limited and it would be difficult to distinguish between reference and target populations. Virtually no information is available in the scientific literature regarding exposure factor values or toxicity criteria for this species. Finally, exposures of carnivorous mammals to PCBs are expected to be lower than those of piscivorous mammals, given the lower concentrations of PCBs present in the terrestrial system, relative to the aquatic system. For these reasons, long-tailed weasels are less suitable receptors of interest than either minks or river otters.

Ermines are "tireless hunters," generally found in forested areas, close to waterways, in low brush, thickets, or rock piles (DeGraaf and Rudis, 1986; Godin, 1977). Mice, voles, rats, rabbits, shrews, birds, and frogs are hunted by ermines for food, although more prey generally is killed than is consumed (Godin, 1977). Ermines have been found to travel up to 4.8 km per night during a food shortage, while under normal conditions, home ranges are 12.1 to 16.2 ha (DeGraaf and Rudis, 1986).

Ermine were excluded from consideration as a receptor of interest because: a) the habitat of the Housatonic River valley is not suitable for ermines; thus, populations of sufficient size for a quantitative study probably would not occur whether or not chemicals are present; b) their propensity to travel long distances to feed would make it difficult to differentiate between reference and target populations; c) virtually no information is available in the scientific literature regarding exposure factor values or toxicity criteria for this species; and d) as discussed above with respect to long-tailed weasels, exposures of carnivorous mammals to PCBs are expected to be lower than exposures of piscivorous mammals, such as river otters and mink, which have already been selected as receptors of interest.

Omnivorous Mammals

Omnivorous mammals include a number of species that tend to feed on a variety of foods including smaller mammals, birds, insects, seeds/nuts, and plant materials. This group of mammals includes the marten, fisher, and raccoon.

Martens eat small mammals, birds, insects, and fruits (Strickland et al., 1982; DeGraaf and Rudis, 1986). Upland forests characterized by spruce, balsam and hemlock are their preferred habitat. DeGraaf and Rudis (1986) report that the average home range of martens is 0.65 to 2.6 km^2 and may be as high as 39 km^2. Although once found over much of northern North America, Godin

(1977) in *Wild Mammals of New England* states that "the marten has now been greatly reduced in range and numbers." Strickland et al. (1982) mapped the current distribution of martens as being restricted to north of the St. Lawrence River, listing dates of extirpation from more southerly areas ranging from the 1800s and the 1930s. Because martens are easily trapped, some researchers believe that trapping may have partially caused the decline in the northeast.

While martens have some of the attributes of good receptors of interest, they cannot be used as receptors for this assessment because they have been extirpated from areas south of the St. Lawrence River. Additionally, the habitat of the Housatonic River flood plain is not suitable for martens. Upland forest characterized by spruce, balsam, and hemlock are the preferred habitat, as opposed to the bottomland forests and cattail marshes of the Housatonic River flood plain. Therefore, it is very unlikely that a marten population exists in the Housatonic River flood plain. Additionally, virtually no information is available in the scientific literature regarding exposure factor values or toxicity criteria for this species and exposures and sensitivity of martens to PCBs are expected to be lower than those of minks or river otters.

Fishers are omnivorous that eat virtually anything that is available, including shrews, mice, squirrels, birds, toads, insects, berries, nuts, and carrion (DeGraaf and Rudis, 1986). In fact, despite its name, "about the only fish it eats are those it finds dead" (Godin, 1977). Fishers inhabit mixed softwood-hardwood forests and wetlands dominated by alder (DeGraaf and Rudis, 1986). Godin (1977) reports that these mammals are great travelers, with home ranges that are 12.8 to 24.0 km in diameter. While hunting, they may travel up to 96 km (DeGraaf and Rudis, 1986). If fishers inhabit the Berkshires, as suggested by Godin (1977) and implied by the trapping season in Massachusetts, it is likely that population densities are low, due to their very large home ranges.

Fishers are not favorable receptors of interest because of their large home ranges, and low (or nil) population densities due to possible extirpation during the 1800s and continued harvesting. Thus, locating a population of sufficient size for quantitative study may be difficult or impossible, and even if such a population could be identified, their large home ranges would prevent the differentiation of reference and target populations. Additionally, virtually no information is available in the scientific literature regarding exposure factor values or toxicity criteria for this species and this species is expected to have lower exposures and sensitivity to PCBs than either minks or river otters.

Raccoons are opportunistic feeders and consume a variety of animal and plant foods. Raccoons prefer fairly open mature hardwood forests in close proximity to streams, rivers, ponds, or lakes (Godin, 1977; DeGraaf and Rudis, 1986). Raccoons are primarily nocturnal with peak feeding activity before midnight. Although raccoons do not hibernate, they remain dormant throughout the winter in dens. Chapman and Feldhamer (1982) report that densities of raccoons can range from 5 to 20 raccoons per km^2. The principal causes of mortality in raccoons are related to the activities of man (e.g., hunting, trapping, automobiles) and malnutrition in winter (Chapman and Feldhamer, 1982).

Raccoons have not been selected as an receptors of interest for several reasons. First, their large home ranges and varied food habits would impede the determination of levels of exposure. Additionally, information on exposure factor values and toxicity criteria is extremely limited. Finally, because other species that have been selected as receptors of interest (i.e., minks and otters) are likely to have higher exposure levels and greater sensitivity to PCBs, inclusion of raccoons would not contribute substantially to the overall understanding of risk.

Insectivorous Mammals

Insectivorous mammals in the Housatonic River valley include white-footed mice, the short-tailed shrew and the masked shrew. These mammals feed on terrestrial invertebrates and emergent aquatic invertebrates and thus have potential for exposure via the food chain. In addition, because these mammals are prey species for a number of other avian and mammalian species, they provide an important link in the food web.

White-footed mice are commonly found in northeastern woodlands and thickets (Godin, 1977). Both the interiors and edges of deciduous, mixed, and coniferous forests are inhabited by white-footed mice (DeGraaf and Rudis, 1986). These are social mammals that may live in groups of two or more (Godin, 1977) and although they prefer to nest above ground in trees (Godin, 1977), tree cavities and spaces under stumps or logs are also used for nesting (DeGraaf and Rudis, 1986).

The diet of white-footed mice consists of insects, seeds, acorns, nuts, fruits, and tender green plants. In addition, because they live in and on the soil, they have increased opportunity for exposure through dermal contact and incidental ingestion. They do not travel far to forage or hunt. In fact home ranges are approximately 0.02 to 0.15 ha for females and 0.06 to 0.22 ha for males (DeGraaf and Rudis, 1986). These animals are abundant in woodlands and thickets and are easily trapped and studied.

White-footed mice were selected as receptors of interest for the following reasons: a) they feed in part on insects, and thus have increased opportunity for dietary exposure; b) they have small home ranges, which allow target and reference individuals to be distinguished; c) they live in and on the soil and therefore have increased opportunity for exposure through dermal contact and incidental ingestion; d) they are easily trapped; e) they are abundant in woodlands and thickets, like those of the Housatonic flood plain; and f) they are an important foodsource for higher trophic-level species.

The short-tailed shrew and the masked shrew are two shrew species commonly found in the damp forests and meadows of the northeastern United States. Short-tailed shrews prefer open habitats, in contrast with the damp deciduous forests preferred by the masked shrews. Both species nest in holes under logs, stumps, and rocks. Shrews travel little and, in some localities and periods, their population densities are high. The home range of short-tailed shrews is 0.2 to 0.51 ha. The home range of masked shrew is 0.04 ha. In favorable habitats, densities up to nine masked shrews per 22 ha have been reported, while short-tailed shrews have been found to have population densities up to 48 individuals per 119 ha (DeGraaf and Rudis, 1986).

The diet of short-tailed and masked shrews consists of insects, worms, spiders, snails, and plants (DeGraaf and Rudis, 1986). The extremely rapid metabolism of shrews forces them to feed continuously to avoid starvation. Because they live in and on the soil, they have increased potential for exposure through dermal contact and incidental ingestion. Shrew populations cannot be censused using mark and recapture methods because they starve to death in the traps. In addition, shrews that survive live trapping techniques may go into shock and die upon being handled, since their heart rates may rise to more than 1,000 beats per minute when startled. As a result, mark and recapture studies are not a feasible option for these species (National Audubon Society, 1991).

While mark and recapture studies are not a feasible option for these species, shrews were included as receptors of interest because: a) population sizes can be compared based on trap-nights of effort; b) they feed on insects and other invertebrates, which increases their potential for dietary exposure; c) they have small home ranges, which ensure that target and reference individuals can be distinguished; and d) they live in and on the soil and therefore have increased potential for exposure through dermal contact and incidental ingestion.

Herbivorous Mammals

Herbivorous mammals indigenous to the Berkshires primarily include white-tailed deer, black bear, moose, and the Southern red-backed vole. Because herbivorous mammals feed on plant material, potential for exposure and uptake of PCBs is minimal.

White-tailed deer prefer forest and swamp borders and are less sensitive to human development and disjointed habitat than either moose or bear. They browse on young woody deciduous plants and some conifers (DeGraaf and Rudis, 1986) and have a home range of 5.2 to 7.8 km^2 (DeGraaf and Rudis, 1986). The habitat requirements of black bear include large tracts of remote mixed deciduous-coniferous woodlands with thick understory, and swamps. The home range of black bears has a radius of 24 km (DeGraaf and Rudis, 1986), resulting in very low population densities. Moose prefer boreal forest interspersed with semi-open aquatic systems, such as swamps or lakes. They require some cover and aquatic plants for food (DeGraaf and Rudis, 1986). The home range of moose has a radius of 3.2 to 16 km. Reflective of their large home ranges, population densities tend to be quite low. In fact, the average moose density in northeastern United States is one individual per 13 km^2 (DeGraaf and Rudis, 1986).

Several factors suggest that these large, herbivorous mammals would not serve as good receptors of interest. First, exposure to PCBs is limited due to herbivorous feeding habits, in concert with large territories and home ranges extending far beyond the target areas. Large mammals generally spend limited amounts of time grazing on a relatively small section of the feeding territory. While feeding in these areas, potential exposure is minimal because food sources (i.e., plants) do not accumulate highly chlorinated PCBs to an appreciable degree. Second, within the 10-year flood plain of the Housatonic River valley, the preferred habitats of these large mammals are very limited. Human development has fragmented and destroyed large tracts of contiguous forest land, while farming has required the maintenance of meadows in place of forestland adjacent to the river. Therefore, it is unlikely that a sizable population of large mammals would naturally exist in the Housatonic River site, regardless of the presence of chemical contaminants. As a result, it is unlikely that there is an adequate number of individuals to provide the basis for statistically meaningful conclusions. It also would clearly be difficult to determine the extent to which individuals might be exposed to chemicals on the Housatonic River flood plain when their ranges extend over many kilometers. Finally, both the exposure potential and the sensitivity of these species to PCBs are lower than for mink and river otters; the selection of the latter allows for a more conservative evaluation of potential ecological risks.

Southern red-backed voles are among the most abundant mammals in the northeastern U.S. Southern red-backed voles prefer the mossy rocks, logs, and tree roots of cool, moist, coniferous or mixed forests (DeGraaf and Rudis, 1986). Voles nest in sheltered spots under boards, rocks, and logs, and commonly use the burrow systems of moles. According to DeGraaf and Rudis (1986), home ranges are small, approximately 0.10 to 1.44 ha. Their diet consists mainly of vegetative material, such as tender grasses, bulbs, cambium of roots and stems, seeds, and grains (DeGraaf and Rudis, 1986).

Although Southern red-backed voles are herbivorous, thus limiting the potential for dietary exposure, they live in close contact with the soil, which allows for exposure through dermal contact and incidental ingestion during burrowing and grooming activities. Further, Southern red-backed voles are a significant food source for owls, hawks, foxes, bobcats, mustelids, and raccoons, and hence alterations in vole populations could potentially affect all species that prey on them, including many mammals and birds of prey. In addition, they are generally abundant in habitats similar to many found in the Housatonic River valley and they have small home ranges that naturally restrict individuals to target or reference areas. They are also easily trapped and studied. For these reasons, Southern red-backed voles have been included as receptors of interest in this assessment.

2.4.7 Endangered and Threatened Species

The MCP requires that any designated species of special concern, threatened species, or endangered species inhabiting a disposal site be considered a receptor of interest in ecological risk assessment (310 CMR 40.0922 (3)). The RCRA Corrective Action Permit issued by EPA to GE has a similar requirement. While several such species, hereafter referred to as T&Es, have been identified in the literature as potentially inhabiting the Housatonic River valley, the presence of any of these organisms within the study areas will need to be ascertained through a combination of on-site observation, investigation of the Natural Heritage database, and communications with MDFW personnel. If it is determined that any T&Es are present or likely to be present within the study areas, these species will be considered receptors of concern and will be evaluated.

It may well be possible to evaluate the potential effects on such species by reliance on the activities proposed herein for other receptors of interest. For example, if the T&E species is in the same feeding guild as a receptor already being studied and does not have a greater exposure potential or sensitivity to PCBs than the receptor under study, the studies proposed herein will be used to the extent possible to address potential effects on the T&E species. If, however, it is determined that these studies are not adequate to address potential effects on the T&Es (if located) and that the latter must be separately evaluated, an addendum to this work plan will be developed and submitted to the Agencies for that purpose.

2.4.8 Summary of Selected Receptors of Concern

Based on the criteria outlined above, several candidate taxonomic groups have been selected as receptors of interest for this risk assessment, while others have been eliminated from further consideration.

Selected Receptors

The following species or groups of organisms have been selected as receptors of concern in the ecological risk assessment for the Housatonic River site:

Feeding Guild	Rationale for Selection
Aquatic invertebrates	Susceptible to localized contamination, food source for higher trophic levels, higher potential for exposure than terrestrial invertebrates.
Omnivorous fish White sucker	High potential for exposure through food and sediment, food source for higher trophic level organisms, large population size

Predatory fish Largemouth bass	High potential for exposure, top of aquatic food web, food source for higher trophic levels, large population size
Amphibians/reptiles Frogs	High potential for exposure, more sensitive to PCBs than reptiles, food source for higher trophic levels
Piscivorous birds Great blue heron	Relatively high potential for exposure, sensitivity to PCBs, available reproductive data
Insectivorous birds Passerines	High potential for exposure, appropriate home range
Piscivorous mammals Mink, river otter	Relatively high potential for exposure, high sensitivity to PCBs
Insectivorous mammals White-footed mouse, shrew	High potential for exposure via soils, food source for higher trophic levels, appropriate home range
Herbivorous mammals Southern red-backed vole	Potential for exposure via soils, food source for higher trophic levels, appropriate home range
Species of special concern, threatened, or endangered species	Required by MCP; may be evaluated by reliance on studies of other receptors

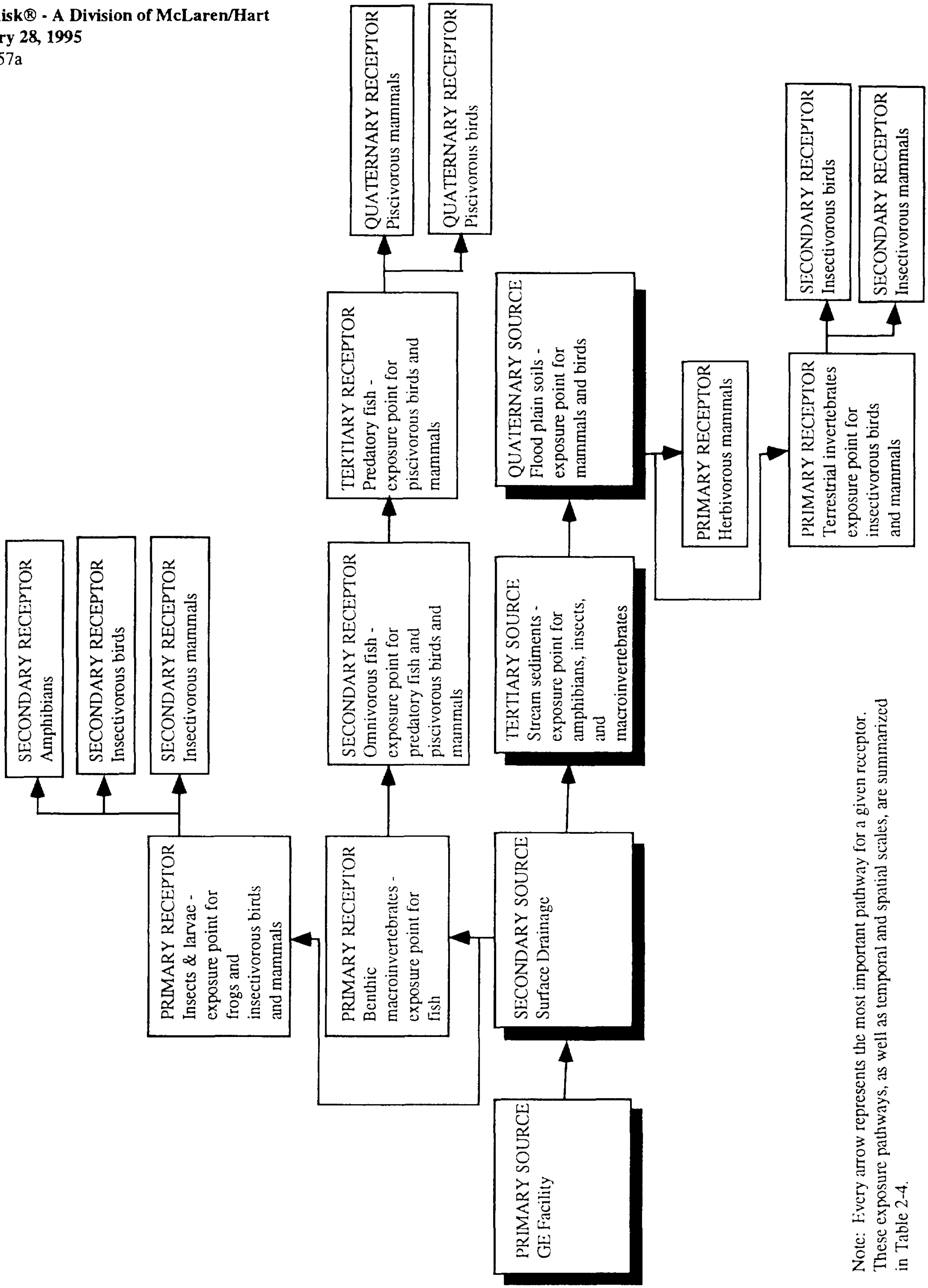
Figure 2-2 illustrates the linkages between the selected receptor groups and sources of exposure to PCBs. Table 2-4 further details the specific exposure pathways associated with each selected receptor group.

Eliminated Receptors

The following other candidates have been eliminated as receptors of concern for the Housatonic River ecological risk assessment.

Feeding Guild	Rationale for Elimination
Aquatic and terrestrial plants	Minimal uptake of PCBs by plants, indirectly assessed by evaluating higher trophic levels.

Figure 2-2. Relationships Between Selected Receptor Groups and Exposure Points



Note: Every arrow represents the most important pathway for a given receptor. These exposure pathways, as well as temporal and spatial scales, are summarized in Table 2-4.

Table 2-4. Exposure Pathway Summary

Receptor Level	Receptor Group	Representative Species	Exposure Pathways(a)	Spatial Scale	Temporal Scale
Primary	Benthic macroinvertebrates	Various taxa	Ingestion of and dermal contact with sediment	River bed downstream of facility	Year-round, with seasonal fluctuations
Primary	Herbivorous mammals	Southern red-backed vole	Incidental ingestion of and dermal contact with surface flood plain soil, maternal transfer	10-year floodplain of Housatonic River downstream of facility	Spring, summer, and fall
Secondary	Omnivorous fish	White sucker	Diet, maternal transfer	Water column downstream of facility	Year-round, with seasonal fluctuations; spring for hatching
Secondary	Amphibians	Frog species	Diet, maternal transfer, dermal contact with sediment	Wetlands of Housatonic River downstream of facility	Year-round, with seasonal fluctuations; spring for hatching
Secondary	Insectivorous birds	Passerines (American redstart, barn swallow, Eastern phoebe, rose-breasted grosbeak, American robin, red-winged blackbird, wood thrush, yellow warbler)	Diet, incidental ingestion of soil and sediment	10-year floodplain of Housatonic River downstream of facility	Spring, summer, and fall
Secondary	Insectivorous mammals	White-footed mouse, shrew	Diet, incidental ingestion of and dermal contact with surface flood plain soil, maternal transfer	10-year floodplain of Housatonic River downstream of facility	Spring, summer, and fall
Tertiary	Predatory fish	Largemouth bass	Diet, maternal transfer	Water column downstream of facility	Year-round, with seasonal fluctuations; spring for hatching
Quaternary	Piscivorous birds	Great blue heron	Diet, maternal transfer	Berkshire County	Spring, summer, and fall
Quaternary	Piscivorous mammals	River otter, mink	Diet, maternal transfer	Housatonic River valley	Year-round, with seasonal fluctuations; spring for kits
Unknown	Threatened and endangered species	Unknown	Unknown	Housatonic River valley downstream of facility	Year-round

Note:
(a) This table lists the most important exposure pathways for each receptor group. Certain exposure pathways are not listed because they would not contribute appreciably to exposure. These include inhalation (since PCBs are not very volatile) plant uptake (since plants minimally take up PCBs, given their insolubility) and gill transfer (since PCBs are insoluble).

Terrestrial Invertebrates	Lower potential for exposure than aquatic invertebrates, indirectly assessed by evaluating species that feed on them
Omnivorous fish Carp, bullhead	Small population, potential for exposure similar to suckers (selected receptor)
Predatory fish Smallmouth bass, trout	Trout range restricted to cold portions of river, potential for exposure to these species similar to that of largemouth bass (selected receptor)
Amphibian/reptiles Snapping Turtle	Lack of toxicological data, lack of sensitivity to PCBs, not a food source for higher trophic levels
Piscivorous birds Belted kingfishers, osprey	Relative to great blue heron, habitat is less suitable, population densities are lower, and toxicological information is more limited
Carnivorous birds Bald eagle, hawks, falcons, owls	Lack of suitable habitat, large home ranges, low population density, lower exposure than piscivorous birds
Herbivorous birds Pheasants, ducks, geese	Lower potential for exposure than insectivorous or piscivorous birds, populations actively managed
Carnivorous mammals Long-tailed weasel, ermine	Lower potential for exposure than piscivorous mammals, large home ranges, low population density
Omnivorous mammals Martens, fishers, raccoons	Lower potential for exposure than piscivorous mammals, large home ranges, low population density
Herbivorous mammals Deer, bear, moose	Lower potential for exposure than piscivorous mammals, very large home ranges

2.5 Selection of Assessment Endpoints

Information presented in the preceding sections on the characteristics of the ecosystem, receptors, and stressors and their ecological effects is used in this section to select ecologically based assessment endpoints. As defined by Suter (1990), an endpoint is a characteristic of an ecological receptor that may be affected by exposure to a stressor. Assessment endpoints are explicit expressions of the actual environmental value that is to be protected (EPA, 1992). That is, assessment endpoints are the specific effects on the selected receptors that are evaluated quantitatively in the risk assessment. They are usually comprised of an entity (population, community, or habitat) and a property (population number, community structure, ecological function).

Five criteria have been suggested for the evaluation of potential assessment endpoints (Suter, 1990):

- *unambiguous operational definition*, which provides direction for testing and modeling and includes a subject (e.g., a specific subpopulation or community) and an effect on the subject that is measurable (e.g., local extinction or reduction in the subpopulation);
- *accessibility to prediction and measurement*, which means that the response of an endpoint can be measured or estimated from measurements of related responses or component responses;
- *susceptibility to the hazardous agent*, which results from the potential for exposure and receptor's susceptibility to the exposure;
- *biological relevance*, which is determined by its importance to higher levels of the biological hierarchy; and
- *societal relevance*, which implies that the endpoint should be understood and valued by the decision maker and the public.

Several candidate assessment endpoints have been identified which meet the above criteria and which are relevant to subpopulations, community, or ecosystem-wide effects, as provided in the MCP. Clearly, the effect described by these assessment endpoints must be clearly associated with the stressor of interest. The candidate assessment endpoints are:

- *Change in community or population structure* - Community and population structure encompasses species composition, diversity, richness, density, population and subpopulation sizes, and age structure of populations. This assessment endpoint includes absence of an expected species and reduction in a population or subpopulation. A change in community structure may be relative to conditions prior to release of the stressor of interest and/or relative to conditions at one or more appropriate reference sites. A multitude of environmental conditions (many of which are unrelated to the presence of the stressors of interest) influence community structure (e.g., weather, altitude, season, latitude, food sources, predators, and human disturbances both at the site of interest and at alternate locations). Therefore, confounding factors must be carefully considered for this assessment endpoint. Additionally, ecologically significant population reductions include those involving keystone species, substantial food sources, and determinants of habitat types. Populations and subpopulations are subject to natural temporal and spatial variability.

- *Sublethal effects with potential population level implications, including reproductive impairment* - Reproductive endpoints that are measurable in individuals and also have the potential to result in population level effects include clutch sizes, litter sizes, hatching success, young survivability, fecundity, etc. Such reproductive endpoints implicitly account for bioaccumulation of a substance to a level that results in toxic effects in the measured species or a species higher in the food web, provided that reproductivity is the most sensitive endpoint for the receptors and chemicals of interest.

- *Toxic effects to rare and endangered species* - Toxic effects to organisms of endangered or threatened species may imperil the entire subpopulation or population, given its stressed condition and small size.

- *Habitat degradation or destruction* - This condition may be defined as the reduction of the area of a habitat or the reduction or elimination of structural vegetative components or critical features typically found within a habitat type. Habitat loss is a species-dependent endpoint: the existence and extent of habitat degradation or destruction will vary for different receptor species, depending on their habitat requirements.

- *Loss or diminishment of ecological function* - Ecological functions are the services provided by a community or ecosystem relative to the flow of energy and matter throughout the system.

As discussed above, the candidate assessment endpoints considered for this ecological risk assessment focus on harm to subpopulations or communities. The significance and suitability of each candidate assessment endpoint varies among receptors and their ecological niche or function.

In determining which of the candidate assessment endpoints to select for analysis, it is important to focus on the critical effects of the chemical of interest on the identified receptors of concern -- i.e., the types of effects to which those receptors are particularly sensitive and which are relevant to subpopulation, community, or ecosystem-wide impacts. Based on the ecotoxicological assessment of PCBs described above, it seems clear that the critical effects of PCBs for the selected receptors of interest are reproductive effects. For fish, amphibians, birds, and mammals, the critical effects relating to reproduction are:

- Fish - egg viability, hatching success, post hatch survival.
- Amphibians - hatching success, post-hatch survival; and
- Birds - reproductive impairment;
- Mammals - reproductive impairment, including survival of young;

In the case of benthic invertebrates, the most appropriate assessment endpoint appears to be change in community structure.

For this reason, in the present study, we have principally selected assessment endpoints that relate to reproductive success. Such endpoints include sublethal reproductive effects (including bioaccumulation to a reproductive toxic level and overt reproductive dysfunction) and change in the structure of a community or population. These assessment endpoints have been matched with selected receptors, based on the receptor's susceptibility to a particular effect and overall feasibility of measurement. Table 2-5 identifies the assessment endpoints selected for each receptor group. (The measurement endpoints listed in the table are discussed below.)

2.6 Selection of Measurement Endpoints

Measurement endpoints are measurable responses to a stressor that are related to the characteristics chosen as the assessment endpoints (Suter, 1990; 1993). When an assessment endpoint can be directly measured, the measurement and assessment endpoints are the same. In most cases, however, the assessment endpoint cannot be directly measured, so a measurement endpoint (or a suite of measurement endpoints) is selected than can be related, either quantitatively or qualitatively, to the assessment endpoint (EPA, 1992).

The relationships between measurement endpoints and assessment endpoints enable the risk assessor to use the results of field observations, bioassays and literature reviews to decide whether a risk of harm has resulted or is likely to result from the stressors. While using a measurement endpoint to approximate or estimate an assessment endpoint introduces some uncertainty into the assessment, that uncertainty will be minimized by selecting measurement endpoints that are closely related to the selected assessment endpoints, and by developing multiple lines of evidence. Multiple lines of evidence will be used to strengthen the support for the conclusions and to consider the weight-of-evidence.

Table 2-5. Summary of Selected Assessment and Measurement Endpoints

Receptor of Interest	Assessment Endpoint	Measurement Endpoint
Benthic macroinvertebrates	Change in community structure	Species diversity, number of taxa Species richness Population density
Fish (general, white sucker, largemouth bass)	Change in community structure	Fish community composition Biomass Species diversity Species richness
	Reproductive impairment	Egg viability of white sucker Egg viability of largemouth bass Hatching success of white sucker Hatching success of largemouth bass White sucker post-hatch survival Largemouth bass post-hatch survival
Amphibians (frogs)	Reproductive impairment	Frog hatching success Frog post-hatch survival
Insectivorous birds	Change in community structure	Bird community composition Species diversity Population density
	Reproductive impairment	American redstart clutch size Barn swallow clutch size Eastern phoebe clutch size Rose-breasted grosbeak clutch size American robin clutch size Red-winged blackbird clutch size Wood thrush clutch size Yellow warbler clutch size Number of young American redstart hatched Number of young barn swallows hatched Number of young Eastern phoebes hatched Number of young American robins hatched Number of young red-winged blackbirds hatched Number of young wood thrushes hatched Average hatching success across species
Piscivorous birds (great blue heron)	Reproductive impairment	Number of young great blue herons hatched
Insectivorous mammals (white-footed mouse, shrew)	Change in community structure	Density of white-footed mouse population Age structure of white-footed mouse population
	Reproductive impairment	White-footed mouse litter size Percent reproductive female shrews Age structure of white-footed mouse population
Herbivorous mammals (southern red-backed vole)	Change in community structure	Density of southern red-backed vole population Age structure of southern red-backed vole population
	Reproductive impairment	Percent reproductive female southern red-backed voles Age structure of southern red-backed vole population

Table 2-5. Summary of Selected Assessment and Measurement Endpoints (cont'd)

Receptor of Interest	Assessment Endpoint	Measurement Endpoint
Piscivorous mammals (mink, river otter)	Change in community structure	Trapping success of minks per unit effort Mink age structure Mink sex ratio
	Reproductive impairment	Mink age structure Mink sex ratio Trapping success of minks per unit effort Otter fecundity as a function of PCB tissue levels
Threatened and endangered species (if located)	Toxic effect - specific endpoints depend on species	Literature review Field survey Other endpoints depend on species located (if any) and may be addressed by endpoints described above for similar feeding guilds

Several different approaches for measurement are available, including chemical analyses, benchmark comparisons, toxicity quotient method, field surveys, toxicity tests, and biomarkers (EPA, 1989b). In order to match a measurement approach with a given receptor and assessment endpoint, the technical feasibility, strength of association with the assessment endpoint, specificity to the chemical, receptor, and site of interest, availability of an objective measure for judgement, and representativeness of the various measurement approaches were considered. A number of measurement endpoints were determined to be both technically feasible and chemical- and receptor-specific. These include the following:

- Benthos community size/structure;
- Fish community size/structure;
- Fish reproductive success;
- Frog reproductive success;
- Passerine community size/structure;
- Passerine reproductive success;
- Great blue heron reproductive success;
- Small mammal population size/structure;
- Small mammal reproductive success;
- Otter reproductive success;
- Mink population size/structure;
- Mink reproductive success; and
- Threatened and endangered species survey.

Table 2-5 shows more specifically the measurement endpoints which will be used to evaluate the selected assessment endpoints for each receptor of concern. That table lists, for each assessment endpoint specified, the measurement endpoints that most directly address that assessment endpoint. In some cases, the measurement endpoints listed also address, less directly, other assessment endpoints.

2.7 Wetland Assessment

Under the MCP, potential environmental receptors can include habitats as well as biota (310 CMR 40.0922(2)). Wetlands (if present) are specifically identified in the DEP's draft guidance for environmental risk characterization as sensitive habitats that should be described and evaluated with regard to their function as a wildlife habitat (e.g., wetlands are often primary habitats for threatened and endangered species), as well as any other wetland functions (e.g., cycling of nutrients) that could be adversely affected by the presence of chemical stressors.

In determining the most relevant assessment endpoints for the potential effects of PCBs on wetlands, it appears that the most sensitive endpoints would be those that relate to the wildlife organisms that inhabit the wetlands. Due to the limited uptake and bioaccumulation of the more highly chlorinated PCBs (e.g., Aroclor 1260) by plants, and the limited sensitivity of both aquatic and terrestrial plants to such PCBs (discussed above), plants do not appear to be a particularly sensitive indicator of potential PCB effects on wetlands. Rather, the wildlife inhabiting the wetlands would provide a more sensitive indicator of potential PCB effects. As such, the selection of wildlife receptor groups of concern discussed in Section 2.4 and the selection of the assessment endpoints for them, as discussed in Section 2.5, should also be applicable to wetlands within the study area. In addition, other potentially relevant assessment endpoints for wetlands include habitat degradation or destruction and loss or diminishment of ecological function (both of which are described in Section 2.5).

A number of measurement endpoints will be utilized to evaluate these assessment endpoints. As a first step, the extent and types of wetlands in the study area will be determined. This will be followed by a general plant and animal survey of representative wetlands. The purpose of this survey is to describe the plant and animal communities of the wetlands in order to determine the wildlife species present and to make a qualitative evaluation of the habitat for potentially affected wildlife species. To assess the potential effects of PCB on the wildlife species (and their habitat), it is expected that the same measurement endpoints used for the wetland wildlife receptor groups discussed above will apply, so that additional studies will not be necessary. In the event that a wetland species is identified that would not be adequately addressed by those studies, then it may be necessary to evaluate such species separately. Finally, a preliminary evaluation of wetland functions will be made using a standard technique. If that evaluation indicates the likelihood of significant impairment to a wetland function, then such function(s) will be assessed for their potential sensitivity to PCB exposure. These studies are described in Section 3.4.

2.8 Conceptual Model

As described by EPA (1992), the major focus of the conceptual model is the development of a series of working hypotheses regarding how the stressor might affect ecological components of the natural environment (NRC, 1986). The preceding text of Section 2.0 has presented a conceptual model of the Housatonic River ecosystem, including a preliminary analysis of the ecosystem, specific habitats, receptors, stressor characteristics, and ecological effects. This conceptual model has tracked the chemical stressors through environmental transport pathways and through the food web to the organisms that could potentially be exposed. As such, this conceptual model supports selection of the assessment endpoints and the critical links between assessment endpoints and measurement endpoints, and indicates that the receptors and endpoints selected for evaluation are appropriate to the overall risk assessment.

3.0 PROPOSED DATA COLLECTION AND ANALYSIS ACTIVITIES

This section presents proposed activities associated with collection and analysis of data on the aquatic, terrestrial and wetland communities of the Housatonic River. The overall approach to evaluating the potential impact of PCBs on each of these three ecosystems is discussed. A brief summary of the existing data is presented and the data gaps are identified. Finally, any additional studies which are deemed necessary to fill those gaps are described along with the steps involved in completing the proposed work.

3.1 Aquatic Communities

Potential damage to the aquatic ecosystem by PCB contamination of the Housatonic River is of concern because of the possibility of repercussions through the food chain from invertebrates to fish to birds and mammals. Within the aquatic ecosystem, benthic invertebrates and fish have been identified as receptors of concern. This risk assessment proposes to evaluate damage to the aquatic ecosystem by examining certain assessment endpoints--for benthic invertebrates, changes in community structure; and for fish, changes in community structure and reproductive impairment (see Section 2.5). We propose to rely upon existing studies (Chadwick, 1994) where appropriate and to conduct further studies to a) address specific concerns identified by the Agencies and b) obtain additional information to address the selected assessment endpoints identified for aquatic species. As discussed in Section 4.0, a "weight of evidence" approach will be taken in this risk assessment. The results of the previous studies on benthic invertebrates (Chadwick, 1994) and the proposed invertebrate/sediment co-sampling study described in this plan will act as independent lines of evidence for drawing conclusions about the health of the invertebrate community. Similarly, the previous study on fish communities in the Housatonic River (Chadwick, 1994) provides one piece of evidence, with the results of the fish reproductive study proposed herein acting as an additional line of evidence. Together, this information will be used to evaluate the health of the aquatic community.

The supplemental studies which are proposed here attempt to clearly identify the relationships between potential PCB exposures and the measurement endpoints. The measurement endpoints chosen for aquatic invertebrates were abundance, species richness, diversity and community structure. The measurement endpoints chosen for fish included species richness, diversity, density/biomass, community structure, and for one omnivorous and one carnivorous species, fertilization success, hatching success and fry survival.

3.1.1 Benthic Invertebrates

The initial evaluation of benthic invertebrate populations indicates that the invertebrate communities downstream of the GE facility are generally healthy (Chadwick, 1994). Substantial variability in density was observed between stations, but this was attributed to habitat conditions, rather than levels of PCBs. Moreover, the Shannon-Weaver diversity values were relatively consistent throughout all shallow water and deep water stations. However, after review of this study, the Agencies commented to GE that the study depended on historic PCB data, rather than co-sampled sediment PCB data, making it difficult to precisely relate population data to PCB exposure. This problem was compounded by the focus on sampling riffle or erosional areas at shallow water sites, which provide the greatest invertebrate abundance and species richness but have the least potential for PCB accumulation. The Agencies further noted that the upstream site was not an ideal reference site because, even though it was generally unimpacted by PCBs, it has been affected by anthropogenic activities to a greater extent than several of the downstream stations.

The Agencies' concerns can largely be addressed though the development of more precise relationships between data on PCB exposures and invertebrate population structure and by focusing collection in depositional areas. This will be accomplished through a field study in which PCB concentrations are determined on a split of the sediments from which the benthic invertebrate population data are derived. In this study, both benthic invertebrate population data and PCB (and other chemical) data will be obtained for sediments that are comparable in type and that contain a full range of PCB concentrations from non-detect (ND) to high. As a result of this sampling design, the PCB data will be both temporally and spatially linked to the invertebrate community data. This approach will allow the development of a stressor-response data set which will be compatible with the existing benthic invertebrate data and will be useful in defining site-specific population-level NOAELs. If no effects are observed, these data will provide additional support for the results from the previous benthic invertebrate study (Chadwick, 1994).

The steps involved in the proposed additional study are as follows:

Initial Selection of Study Sites

The existing sediment data from the general vicinity of the sampling sites identified by Chadwick (1994) as HR2, HR5, and HR6 will be reviewed in an attempt to locate approximately 15 locations in these areas which contain a gradient of PCB concentrations from ND to high. These particular sites will be sampled because they are located in areas which were found to represent a wide range of PCB sediment concentrations. HR2 (between the New Lenox Road Bridge and Woods Pond) was reported to have the highest recorded PCB levels in the river, HR5 (downstream of Rising Pond) exhibited intermediate PCB levels, and HR6 (near the Connecticut border) had the lowest reported PCB levels (Stewart, 1982). To avoid potential confounding factors, the emphasis will be placed on identifying shallow (< 3 feet deep) depositional areas with comparable habitats. If the data indicate that a full range of PCB concentrations in sediments (including several NDs) can be obtained within these portions of the Housatonic River, then no locations outside these river reaches will be identified. In the event that the data do not indicate any suitable sediment locations within these portions of the Housatonic River that are likely to have non-detectable levels of PCBs, then a suitable control area with comparable habitat will be identified--either elsewhere in the Housatonic River system (e.g., upgradient of the GE facility or on a tributary) or in a different river system.

Sediment/Benthic Invertebrate Sampling

Following the identification of at least 15 sediment locations with an apparent gradient of PCB concentrations from ND to high, each of these locations will be sampled by grab sampling of the top 3 inches of sediment. At each location, five replicate sediment samples will be taken using an Ekman grab sampler with a volume of $0.02m^2$. Each grab sample will be split into a subsample for chemical analysis and another subsample for benthic invertebrate analysis. A parallel grab sample will be taken adjacent to each chemical/biological grab sample for grain size and total organic carbon (TOC) analysis. Redox potential of the sediments will also be measured on site, and a water sample will also be collected at each site for general water quality analysis. The sediment and water sampled collected for chemical/physical analysis will be sent to the laboratory for such analyses. The invertebrate samples will be preserved for later analysis. All sampling will be conducted in accordance with the approved Sampling and Analysis Plan/Data Collection and Analysis Quality Assurance Plan (SAP/DCAQAP) (Blasland, Bouck & Lee, 1994c).

Chemical Analysis of Sediments

Sediment samples collected from each of the 15 initial sampling locations will be analyzed for grain size, TOC, and PCBs as well as chlorinated pesticide residues (e.g., DDE) which might represent a confounding factor. The water samples will be analyzed for general water quality parameters, including water temperature, dissolved oxygen, pH, ammonia, and nitrate. The analyses will conform to the SAP/DCAQAP.

Selection of Sites for Further Analysis

Based on a review of the sediment data for the 15 initial sampling locations, a final series of 5 to 10 sites will be selected for more detailed analysis. These sites will be chosen based upon the following criteria: a) the presence of a range of PCB concentrations in the sediments (from non-detectable to concentrations representing high levels found below Pittsfield); b) a consistent type of sediment, taking into account grain size and redox potential; c) an absence of confounding factors such as depth and TOC levels; and d) the use of stations exclusively within the Housatonic River if the previous three criteria can be met. It should be noted that the design of this study intentionally avoids the limitations of using upstream reference stations (unless necessary) by relying on internal variability in the downstream reach of the Housatonic River, in order to establish a stressor-response relationship.

Benthic Invertebrate Processing

After the results of the chemical analyses are used to identify the final sites, the preserved samples of invertebrates from those 5 to 10 sites will be processed. Processing includes separation from debris, sorting, identifying to the lowest taxonomic level, and counting. The abundance of each species per replicate per site and species richness (the number of taxa per replicate per site) will be enumerated. Diversity will be calculated according to the Shannon–Weaver Diversity Index (H') as a measure of the effects of stress on invertebrate communities.

Statistical Analysis of Benthic Invertebrate Data

This study has been designed to allow a rigorous evaluation of the relationships between benthic community structure and the concentrations of PCBs in the sediments. Initial statistical analyses will focus on determining if significant differences exist in the benthic community structure of the 5 to 10 stations representing the full range of PCB concentrations. This question will be addressed using MANOVA techniques. If significant differences in community structure are observed among the sites, analyses will be carried out to evaluate stressor-response relationships for PCBs and potential confounding factors (e.g., grain size, redox potential, TOC). This will be accomplished by using multiple regression techniques. If a significant stressor-response relationship exists between PCB concentrations and benthic community structure, then these analyses could be used to establish site-specific NOAEL and/or LOAEL values for the benthic community. All statistics will be performed using SPSS for Windows (v. 6.0).

Overall Evaluation of Benthic Invertebrate Data

As discussed in Section 3.1, the data will be used to facilitate a weight of evidence approach in evaluating the assessment endpoints. The previous study of the benthic invertebrate community (Chadwick, 1994) and the proposed co-sampling of invertebrates with PCBs in sediments will act as independent lines of evidence in the evaluation of the health of the benthic community inhabiting the Housatonic River. If no adverse relationship between sediment PCB concentrations and the associated benthic community structure is found, then this will be used to support the findings of the Chadwick (1994) study. If exposure-related effects are observed in the proposed new study, then population level NOAELs for this site may be defined.

3.1.2 Fish

As discussed previously, the critical effects of PCBs on fish are potential reproductive effects. Thus, the assessment endpoints selected to evaluate fish are reproductive impairment and change in community structure (which would reflect impaired reproduction).

These endpoints have already been addressed to some extent by the prior study (Chadwick, 1994). That study evaluated the overall community structure of fish populations in the Housatonic River, as well as the condition of the fish. It obtained data on species diversity and richness, population density and biomass, and fish condition at several sites in the Housatonic River, including some downstream of the GE facility and some upstream of (or not affected by) that facility. The species diversity and richness at the study sites compared very well to prior data on the Housatonic River and other rivers in the Northeast. The fish population/community parameters were generally similar at comparable sites upstream and downstream of the GE facility and did not show a relationship to the general levels of PCBs reported (in prior studies) in the sediments in the various river reaches sampled. These parameters were also found to be typical in comparison to other rivers in the Northeast.

Upon review of this study, the Agencies expressed several concerns about it. These concerns included the view that the study lacked sufficient sensitivity to identify specific toxic effects on fish, as well as the concern that the upstream control sites are affected by anthropogenic activities to a greater extent than several downstream sites and hence do not constitute good control sites.

These Agency concerns can be largely addressed by an additional study which focuses on evaluating the relationships between the reproductive potential of native fish and the exposure of those fish to PCBs. Given the ranges of the various fish species in the Housatonic, the low concentrations of PCBs in the water column and the heterogeneous distribution of PCBs in the sediments, it is difficult to develop precise stressor-response relationships for native fish from these data sets. This study will therefore use tissue residues of PCBs in resident species of fish as integrated measures of exposure. Resident species chosen should: a) be abundant at the stations below Pittsfield; b) occupy a relatively high trophic level; and c) be amenable to strip spawning and laboratory early-life-stage studies. Based on these and other criteria, the white sucker and largemouth bass were the species chosen (see Section 2.4.3). The overall objective of the fish study is to determine whether a significant relationship exists between the concentrations of PCBs in the tissues of individual fish and the reproductive potential of those fish. Reproductive potential will be evaluated by following spawning, fertilization and early development of eggs and embryos of resident fish. These early life stages are particularly sensitive to PCBs and these data sets can be readily extrapolated to population level effects. It is important to note that this fish study is intentionally designed to focus on the stressor-response relationship between fish tissue concentrations of PCBs and reproductive success and not to examine the relationship between sediment PCB concentrations at the sampling sites and fish body burdens of PCBs. The steps in the proposed fish study are outlined below.

Fish Collection

Samples of white suckers and largemouth bass will be collected during their breeding seasons (April and May, respectively) in river reaches which encompass at least five sites for each species. The purpose of this sampling will be to attempt to obtain fish that reflect a full range of PCB tissue concentrations from ND to high. Specifically, fish of each species will be collected from reaches in the vicinity of the sites identified by Chadwick (1994) as HR2, HR5, and HR6 and at least one appropriate off-site location that is not impacted by PCBs. The off-site control site for white suckers will likely be located in the Housatonic River upstream of the GE facility, and the off-site control location for largemouth bass will be a non-PCB contaminated local pond which is not connected to the Housatonic River system. All sites will be selected to have comparable habitat, depth, and availability of the target fish species.

At each site, approximately 4 to 8 pairs of reproductively mature, ripe fish per species will be collected, for a total of at least 20-40 pairs of fish of each species. Fish collection procedures will be in accordance with the SAP/DCAQAP (Blasland, Bouck & Lee, 1994c).

Fish Reproductivity Study

The two fish species chosen for consideration breed at different times, because their spawning is largely temperature-dependent. The white sucker spawns at water temperatures of 50-60°F while the largemouth bass spawns later when the water temperature reaches 62-68°F. Water temperature will be carefully monitored in order to determine when each species is likely to be ripe. Individual males and females will then be strip spawned and at least four breeding pairs selected randomly from fish collected at each station. Fertilized eggs will be transported from the field to the laboratory where fertilization success, hatching, embryonic development, and survival will be monitored as part of an early-life-stage study. For each species of fish, fertilized eggs from approximately four different pairs per sampling site will be incubated in laboratory water. The actual number of fish pairs will be dependent upon the number of fish caught. The egg mass from each fish pair will be divided into four subsets such that within-fish variation in rates of fertilization and hatching can be evaluated, in addition to between-fish and between-site variation. The hatchlings of both species can be expected to live off their yolk sacs for approximately 7 days, after which they begin to feed on their own. Consequently, the fish will be monitored for 14 days post-hatch, in order to evaluate their survival during this critical time in their development. Fertilization and hatching success, as well as fry survival, will be used as measurement endpoints. Specific parameters that will be measured include time to hatch, survival, gross sublethal effects, and the length and dry weight of the surviving fish.

Analysis of PCB Levels in Tissues and Eggs of Reproductive Fish

PCB concentrations will be measured in the whole tissues of the male and female fish caught for the spawning study. The tissues will also be analyzed for chlorinated pesticides which may be a confounding factor. Both fresh and dry weights and lipid content of the tissues will be measured to account for intra-and inter-specific variation in lipid content. As there may be substantial loss of PCBs with the eggs during spawning, an aliquot of eggs from each of the pairs of both fish species will also be analyzed for PCBs and chlorinated pesticides. This study will provide a basis for evaluating the relationship between PCB accumulation in adult fish and eggs and reproductive success in a resident species; as such, it will provide an important supplement to the existing fish population data.

Statistical Analysis of Fish Data

Again, the goal of the statistical analysis of the fish data is the description of the relationship between the body burdens of PCB in fish tissues and the reproductive success of those fish. In addition, the PCB concentration in the fertilized eggs will be related to their hatching success. This will involve the use of multiple linear regression procedures. If no statistically significant relationship between reproductive success and PCB concentrations in the fish or eggs is found, then it would be concluded that there is no significant effect of PCBs at the range of concentrations found in this study. If, however, a significant stressor-response relationship exists between PCB concentrations fish tissues or eggs and hatching success, these analyses could be used to establish site-specific fish residue NOAEL and/or LOAEL values. All statistics will be performed using SPSS for Windows (v. 6.0).

Overall Evaluation of Fish Data

As discussed in Section 3.1, the data will be used to facilitate a weight of evidence approach in evaluating the assessment endpoints. The previous study of the fish community (Chadwick, 1994) and the proposed fish reproduction study will act as independent lines of evidence in the evaluation of the health of the fish community inhabiting the Housatonic River. If no adverse relationship between tissue and egg PCB concentrations and fish reproduction is found, then this will provide additional evidence in support of the conclusions of the Chadwick (1994) study. If exposure-related effects are observed in the fish reproduction study, then a NOAEL for each of the two fish species at this site may be defined.

3.2 Terrestrial Communities

Historical flooding of the Housatonic River has resulted in the deposition of PCBs in the flood plain of the river between the GE facility in Pittsfield and Woods Pond Dam. As discussed in Section 2.1.2, concentrations of PCBs exceeding 1.0 ppm are generally contained within the approximate ten-year flood plain of the river. Within the river reach between the GE facility and the confluence with the West Branch, the 10-year flood plain of the river is generally confined within the banks or a narrow strip along the river bank; areas in this reach in which the flood plain is more substantial are generally areas that have been heavily developed, either commercially, residentially, or agriculturally, and thus do not provide adequate or suitable habitat for substantial populations of terrestrial ecological receptors. Between the confluence and the West Branch and New Lenox Road, the flood plain widens and there are undeveloped wetland areas within the 10-year flood plain of the river, but very limited terrestrial areas that have not been developed for residential or agricultural purposes. Thus, for an evaluation of the terrestrial receptors living near the Housatonic River, the risk assessment will focus on the area between New Lenox Road and Woods Pond dam, an area that has relatively large and undisturbed terrestrial areas within the 10-year flood plain and which provides suitable habitat for a number of avian and mammalian receptors.

The assessment of impacts to terrestrial wildlife will be conducted using a number of different approaches, and will consist of a number of existing field studies (ChemRisk, 1994; Organ, 1989) supplemented by additional field studies to evaluate whether PCBs in the terrestrial ecosystem of the Housatonic River are exerting an adverse effect on any of the receptors of concern known to inhabit that ecosystem.

In evaluating the potential adverse effects of PCBs, it is important to compare results of site-specific studies with the results of other studies conducted in areas where PCBs are not a contaminant of concern. However, the identification of a single reference area for comparison of results of all terrestrial evaluations of the Housatonic River site is infeasible, given the internal heterogeneity of the Housatonic system. Therefore, it has been necessary to select endpoint-specific reference areas, matched with those aspects of the Housatonic River floodplain system that are most critical for a given measurement endpoint. This strategy is termed multiple asymmetric referencing. Where possible, local, matched reference areas are used. However, when this approach is not feasible, due to a lack of availability of matched reference areas, literature based reference areas that closely match the on-site study areas are utilized. The selection of the reference area(s) to be used under each part of the terrestrial evaluation is discussed in the appropriate sections that follow.

3.2.1 Amphibians/Reptiles

As discussed in Section 2.3.5, relatively little information is available in the scientific literature on the toxic effects of PCBs on amphibians and reptiles. While both types of organisms experience a high potential for exposure to PCBs via sediment contact and food web exposure, reptiles appear to be far less sensitive to the toxic effects of PCBs than amphibians (Olafsson et al., 1984; Bryan et al., 1987). For this reason, as well as the abundance of frogs in the Housatonic River ecosystem, their potential for exposure, and their importance in both the aquatic and terrestrial food webs, frogs have been chosen as the receptor of concern for this risk assessment.

The limited data that are available in the published literature indicate that the most sensitive toxic endpoints tested in frogs are reproductive effects, including egg viability, hatching success, and early post-hatch survival of young (Birge et al., 1978). Thus reproductive effects constitute the assessment endpoint that will be evaluated for these organisms. To evaluate potential impacts on the reproductive effects endpoint, a frog reproduction study will be undertaken. The purpose of this study will be to determine whether egg mortality, eggs hatchability, normal development, and the percent of young surviving post hatch are being affected by the presence of PCBs in the Housatonic River ecosystem, compared to a reference ecosystem, and whether these effects, if any, are related to PCB levels in the egg masses.

Selection of study areas

Adult male and female bullfrogs will be collected in the spring from the Housatonic River, as well as from one or more matched reference areas. Preliminary site visits have indicated that there are substantial populations of bullfrogs present in the backwater areas above Woods Pond, the area of the river known to have the highest average levels of PCB sediment contamination. Thus, this is the PCB-affected study area from which frogs will be collected. Bullfrogs will also need to be collected from areas that have not been affected by GE facility releases. It is likely that these will be collected from the cattail marsh areas near Center Pond in Dalton and/or from the cattail marsh near Windsor Road in the Peru State Wildlife Management Area.

Study Design

- Twenty pairs of adult male and gravid female bullfrogs will be collected in April or May from the Housatonic River study area and from one or more reference areas, for a total of forty pairs of adults.

- After collection, frogs will be transported to the laboratory and females will be hormonally induced to complete development of their eggs. Eggs will then be removed from each female and inseminated by a male collected from the same location.

- Laboratory personnel will separate subsets of 20-40 fertilized eggs from each egg mass from both reference and Housatonic River frogs. These will be incubated in laboratory water which will enable the incubation conditions of all egg masses to be strictly controlled.

- Potentially confounding effects will be further minimized by ensuring that laboratory personnel are "blind" with respect to the origin of the egg masses.

- Laboratory personnel will count and remove hatched larvae daily, and will record all morphological abnormalities and mortality observed.

- The portions of the egg masses remaining after the subsets of eggs are removed for incubation will be submitted for PCB analysis.

- Female frogs will also be submitted for whole body PCB analysis.

- Data on maternal tissue and egg mass PCB levels will be tested for a statistically significant relationship with hatching success and post-hatch survivability. Hatching success and post-hatch survivability for egg masses collected from the Housatonic River will also be tested for statistically significant differences relative to egg masses collected from the reference area(s).

Results of these studies will be reported and interpreted as to whether it appears that the presence of PCBs in the Housatonic River ecosystem is adversely impacting the reproductive success of frogs that live there.

3.2.2 Insectivorous Birds

As discussed in Section 2.4.5, insectivorous birds were selected as a receptor group primarily because they may be exposed to PCBs through the diet. Exposure potential of insectivorous birds is augmented by their relatively small home ranges; birds with breeding territories within the extent of contamination are likely to derive 100% of their food from the Housatonic River site. Additionally insectivorous birds have been shown to be susceptible to PCB toxicity. Because the most sensitive endpoint for PCB toxicity in insectivorous birds appears to be reproductive impairment, this potential effect was selected as an assessment endpoint. It is assumed that reproductive impairment on a community-wide scale may result in changes in community structure. Therefore, change in community structure was selected as a second assessment endpoint for evaluating potential adverse effects of PCBs to insectivorous birds.

The measurement endpoints selected to evaluate reproductive impairment are: a) clutch sizes of American redstart, barn swallow, Eastern phoebe, rose-breasted grosbeak, American robin, red-winged blackbird, wood thrush, and yellow warbler; b) number of young hatched for American redstart, barn swallow, Eastern phoebe, American robin, red-winged blackbird, and wood thrush; and c) average hatching success across species. The measurement endpoints selected to evaluate changes in community structure are: a) bird community composition; b) species diversity and number of species; and c) population density.

The analysis of these measurement endpoints will largely rely on a previously conducted field study of insectivorous birds (ChemRisk, 1994). As part of the *Evaluation of the Terrestrial Ecosystem of the Housatonic River Valley*, ChemRisk (1994) evaluated all of the measurement endpoints listed above that have been selected for insectivorous birds. In order to address specific concerns raised by the Agencies, the existing data will be supplemented by additional statistical analyses. The study areas and reference data used in this effort are described below. This section then details the study designs employed previously, as well as the supplemental statistical analyses that are planned.

Study Areas and Reference Data

In the evaluation of avian reproductive impairment, the boundaries of the 10-year flood plain of the Housatonic River served as the boundaries between the target and reference nests. Target nests were typically located along the river banks, under the New Lenox Road bridge, and in the cattail stands in the river. Reference nests were located well outside of the 10-year flood plain, including along October Mountain Road, in the reference shrub meadow, and in the Crane Farm barn in Dawson. Feeding preferences and territory sizes of individual species monitored were evaluated in order to verify that reference birds would not be expected to visit the target area and vice versa. In addition, data from the literature (Ehrlich et al., 1988) on normal clutch sizes for each avian species were also used as reference data for comparison to observations of target nests.

For the evaluation of avian community structure, ChemRisk identified a specific target study area in a forested portion of the 10-year flood plain. This area is located south of the New Lenox Road bridge. It is 5.85 hectares in size and homogeneous in habitat. This site is the largest intact, undisturbed flood plain forest between New Lenox Road bridge and Woods Pond dam, the reach of the Housatonic River with both high concentrations of PCBs and suitable wildlife habitat.

After a great deal of investigation of the area within a 50 mile (or greater) radius of Pittsfield, no suitable reference areas were identified for the flood plain forest. The absence of a suitable reference site is in part a reflection of the target area's location within the HVWMA. This area differs substantially from land uses of surrounding areas, where human development, agricultural activity, and industrialization have drastically impacted the land, rendering it unsuitable for comparison to the target site. In such areas it would not be possible to distinguish between those human influences and PCB effects. While the Housatonic River upstream of Pittsfield, tributaries to the Housatonic River, and other river valleys (including the Hoosic and the Konkapot River valleys) were all considered for use as reference sites, no areas of comparable size, land use, and habitat type were located close enough to the target site to allow simultaneous studies to be conducted.

Because an adequate reference flood plain forest could not be located, a comprehensive literature search was conducted in an effort to identify suitable reference data for comparison to the avian community structure study. Because passerines are highly sensitive to subtle variations in habitat as they establish breeding territories, the most important criteria for selecting reference studies were habitat type (i.e., flood plain forest), habitat quality (e.g., forest maturity, isolation from human influence, size), and habitat structure (e.g., flooding regime, development of strata, canopy cover, dominance and abundance of plant species). A broad array of sources was searched, including reports of state and federal agencies and nongovernmental organizations, Master's and Doctoral dissertations, and scientific journals.

Based on the criteria described above, two series of breeding bird censuses were selected as the most suitable reference studies. These are breeding bird plot censuses that were conducted since 1981, as part of the Breeding Bird Census in a flood plain forest in Maryland (Criswell and Gauthey, 1983; Van Velzen and Van Velzen, 1985, 1986, 1987, 1988; Gauthey 1984, 1989, 1990, 1991, 1992) and in a flood plain forest in North Carolina (Crotteau et al., 1978, 1979; Christensen et al., 1980, 1982; Hall et al., 1984; Van Velzen and Van Velzen, 1985; Hall, 1990; Mueller and Hall, 1991; Mueller and Mueller, 1992; Mueller, 1993). The annual Breeding Bird Census is a nationwide effort jointly sponsored by the Association of Field Ornithologists and Cornell Laboratory of Ornithology. It includes detailed information on the habitats of the individual study plots, which facilitates comparisons with outside databases. The Breeding Bird Census shares objectives and methodologies virtually identical to those of the avian census conducted at the Housatonic flood plain forest. No comparable reference sites closer to the Housatonic River could be identified in the scientific literature. Although it would clearly be desirable to use reference sites located closer to the target area, these two series of censuses were determined to be appropriate reference studies because they are very similar to the target area with respect to habitat type, quality, and structure.

Study Design

As part of the *Evaluation of the Terrestrial Ecosystem of the Housatonic River Valley*, ChemRisk (1994) evaluated the reproductive success and community structure of an insectivorous bird community inhabiting the 10-year flood plain of the Housatonic River. Potential reproductive impairment in insectivorous birds was evaluated by measuring clutch size, numbers of young hatched, and overall hatching success for several species of common songbirds. Reproductivity data for birds living within the 10-year floodplain were compared to reproductivity data for birds living outside of the 10-year floodplain, as well as to ranges of normal clutch sizes reported in the scientific literature. Data were collected through a nest monitoring program conducted during the summer of 1993. The methodology involved: a) searching for and locating natural nests; b) identifying the bird species occupying the nest; c) marking the nest location with a numbered stake; d) determining whether the nest was located inside or outside of the 10-year flood plain; e) recording descriptive information on the nest number, location, setting, species, date and time located, number of eggs, number of young, behavior of parents, and other comments; and f) revisiting each nest every two to six days to monitor numbers of eggs and young, signs of predation or parasitism, and other relevant factors.

Analysis of the nest monitoring data first required the entry of all nesting data into a database including descriptive parameters for nest identification number, species, location with respect to the 10-year flood plain, number of eggs laid, number of young hatched, and whether the nest was determined to be active or inactive. The mean clutch sizes and number of young hatched for each species were tested for statistically significant differences between the target and reference nests. Both the Student t-test and Mann-Whitney U-test were used to test statistical significance, with a level of significance of 0.05.

In addition to comparing target and reference nests, average clutch sizes were compared to the literature (Ehrlich et al., 1988, and selected other papers) to determine whether reproduction throughout the Housatonic area might vary substantially from historic and/or nationwide observations, or whether differences might be apparent due to study design or observer bias. Because the records of observation (or means and standard deviations) were not available in Ehrlich et al. (1988), it was not possible to test for statistically significant differences.

Avian community structure was evaluated by censusing the density and diversity of birds in the flood plain forest target area. A modified version of the plot census technique of Svensson (1970) was applied. The Housatonic flood plain forest target area was divided into five one-hectare plots and, due to constraints on space, one 0.85-hectare plot. In an effort to minimize edge effects, the plots were established well within the boundaries of a homogeneous habitat, with a 7.6 m (25 ft) setback. Censuses were conducted on May 25 through May 28, 1993 between 5:30 a.m. and 9:00 a.m.

Prior to initiating each census, the observer recorded information pertaining to the time, weather conditions, and location. Each one hectare plot was censused with the observer entering the cell at approximately the midpoint of one edge and slowly proceeding along a route that allowed thorough coverage. As each plot was traversed, the common name, sex, and location within the plot of all birds observed visually or aurally were recorded on the data form. Following each census, the field observers double-checked data sheets for completeness and legibility and summarized their observations regarding likely numbers of breeding pairs of each species.

Individual observations from each of the four daily census runs were plotted on composite maps. The number of breeding pairs of each species in the 5.85 hectare area was then extrapolated to the number of pairs per 40 hectares of habitat to provide an estimation of breeding bird density (territories/40 ha) by individual species and by total number of breeding birds.

Species diversity was evaluated using Hill's diversity numbers (designated N0, N1, and N2), a series of indices which account for both the number of species and the equitability or evenness of allotment of individuals among the species. While a large number of diversity indices exists, the series of diversity numbers presented by Hill (1973) is probably the easiest and most meaningful to interpret from an ecological point of view (Ludwig and Reynolds, 1988).

The results of the studies described above serve as measurement endpoints that directly address the assessment endpoints of reproductive impairment and changes in community structure for insectivorous birds. In addition, supplemental statistical analyses will be conducted to address concerns raised by the Agencies in their review of the original report. Specifically, the agencies requested information on the distributions of data on clutch sizes and young hatched for each species (e.g., normal, lognormal), as well as on the power of the individual statistical tests. These additional analyses will be conducted on the existing data set and reported with the presentation of these measurement endpoints.

3.2.3 Piscivorous Birds

As previously discussed, piscivorous birds were selected as a receptor group for this ecological risk assessment because of their relatively high potential for exposure and their susceptibility to PCB toxicity. As detailed in Section 2.3.6, the most sensitive endpoint of toxicity for PCBs to birds is reproductive impairment, as a combined result of embryotoxicity and aberrant parental behavior. Therefore, reproductive impairment was selected as the assessment endpoint for piscivorous birds.

Of the piscivorous bird species likely to breed in the vicinity of the Housatonic River site, great blue herons were determined to be the most appropriate species on which to base measurement endpoints. MDFW has collected reproductive success data for great blue herons throughout the state since 1982. This extensive data set substantially adds to the understanding of actual reproductivity of exposed populations of great blue herons. Comparable data sets are not available for other piscivorous bird species. Moreover, great blue herons have established colonies within foraging distance of the Housatonic River site; thus, this species is known to inhabit the area and may potentially be exposed to PCBs in the river ecosystem.

One possible approach to evaluating potential risks to great blue herons might be to undertake a predictive toxicity quotient approach in an effort to develop a theoretical prediction of whether the population of great blue herons in Berkshire County is at risk of reproductive impairment, based on assumptions about heron behavior, assumed dietary intake of PCBs due to consumption of Housatonic River fish, and toxicity values derived from the literature. However, given the serious limitations and uncertainties in such predictive modeling (discussed above) and the availability of pertinent actual field data (i.e., an extensive data set from MDFW on the actual reproductive success of great blue herons throughout the state, together with the existence of known colonies of great blue herons within foraging distances of the Housatonic River), we have decided, instead, to attempt a direct field assessment of the actual reproductive success of great blue herons in the area. This study will focus on the number of great blue herons hatched per nest in colonies within foraging distance of the Housatonic Rive site, relative to distance from the site and relative to the number of great blue herons hatched per nest in reference colonies. This measurement endpoint will directly address the assessment endpoint of reproductive impairment of great blue herons. This study is discussed below.

Study of MDFW Heron Data

Since 1979, the MDFW has collected data on the hatching success of all known great blue colonies within Massachusetts (MDFW 1979; 1980; 1981; 1982; 1983; 1984; 1985; 1986a,b; 1987; 1989; 1991). Surveys were conducted annually through the mid 1980s, biennially during the late 1980s, and every five years thereafter. Surveys are overseen by the Natural Heritage and Endangered Species Programs. Sites are visited in late June and, using spotting scopes, the number of total platforms, active platforms, and young in each nest are counted. Several heronries are located within 34 km of the Housatonic River. (Thirty-four kilometers is reported by Henning et al. (1994a,b) to be the maximum distance that great blue herons travel from the colony to forage).

ChemRisk will analyze the MDFW great blue heron reproductive data with the objectives of determining whether significant differences in reproductive success are observed for those colonies located closest to the Housatonic River as compared to reference colonies, and whether a significant relationship is observed between distance from the Housatonic River site and reproductivity (i.e., whether the stressor and response are correlated). As detailed below, several activities will be completed in order to meet these objectives, including: a) completion of the June 1995 MDFW survey; b) preparation of a database that includes great blue heron reproductive data for all available years and colonies; c) linear modeling of data to evaluate statistical significance of predictors of reproductive success; and d) interpretation of data and presentation of results.

In order to ensure that the data presented in the ecological risk assessment are the most current available, and to take advantage of data from recently established great blue heron colonies in Lenox and Dalton, ChemRisk will utilize data from the MDFW survey planned for June 1995 (pers. comm., 1994, Tony Gola, MDFW). Clearly, these new data will help to ensure that the measurement endpoint reflects current conditions of the colonies. Furthermore, reproduction data for the two colonies in Lenox and Dalton, established since the last survey (in 1991), are expected to improve the power of the analysis of the relationship between reproductive success and distance from the Housatonic River site.

ChemRisk will work with MDFW in conducting the 1995 survey field work for Berkshire County and one or more reference colonies. Permission for ChemRisk to participate has been granted by MDFW (pers. comm., 1994, Tony Gola, MDFW). A ChemRisk ecologist will accompany the survey participants on site visits to all colonies in Berkshire County, as well as in one other county with nesting habitat comparable to that of Berkshire County. The selection of the most appropriate reference colonies will be based on conversations with Mr. Bradford Blodget, the state ornithologist, and Mr. Tony Gola, both of whom have directed the MDFW heron survey since its inception. To the extent possible, multiple candidate reference colonies will be selected, in order to minimize potential confounding effects. The ChemRisk ecologist participating in the 1995 survey field work will record parental behavior and colony condition, as well as factors that may influence the certainty of the overall assessment (e.g., observer bias, variability in potential for human disturbance at different colonies, etc.).

The second activity for this measurement endpoint is the development of a data base of information collected by MDFW from 1982 through 1995 on the reproductive success of great blue herons in Massachusetts. The data base will include the following fields of information: colony name, county, distance from Housatonic River site, year of observation, number of active nests, total number of young observed and estimated, and average number of young per nest. ChemRisk will verify the accuracy of the data entry, prior to sorting the data set by county, distance from Housatonic River site, and year.

As the third activity for the measurement endpoint, linear modeling will be conducted using Systat software (Systat, Inc., Evanston, IL) to evaluate the statistical significance of categorical (e.g., county, colony) and continuous (e.g., year, distance) predictors on reproductive success.

Interactions between predictors (e.g., simultaneous nonindependent combined effects of county and year) will also be evaluated as appropriate. All relationships will be evaluated for significance at the 0.05 (5 percent) level.

The final activity for this measurement endpoint involves the interpretation and presentation of results. Because all of the colonies located within Berkshire County are within 34 km of the Housatonic River site, while none of colonies located in counties other than Berkshire County are within 34 km of the site, tests for significant differences in reproductivity between counties will be used to compare reproductivity of potentially exposed and unexposed populations of great blue herons. In addition to county, year and interaction of county and year will be tested for significant relationships to the number of young per nest. ChemRisk will also determine whether reproductive success is significantly different among the colonies within Berkshire County as a function of these colonies' distances from the Housatonic River, the year, and the interaction of distance and year. ChemRisk will report the percentage of variance in young per nest that is explained by these three factors, as well as the significance of each factor's ability to predict the number of young per nest. In short, multiple statistical tests will be used to interpret the MDFW data, and the results will be used to directly evaluate the potential reproductive impairment of great blue herons breeding within 34 km of the Housatonic River site.

3.2.4 Insectivorous and Herbivorous Mammals

As discussed previously, insectivorous mammals were selected as receptors of interest primarily because of their potential for exposure to PCBs through the diet. Both insectivorous and herbivorous mammals may also be exposed to a lesser degree through incidental ingestion of and dermal contact with soil of the 10-year flood plain. These feeding guilds were also selected as receptor groups because they occupy home ranges that are smaller than the extent of contamination and serve as an important food source for higher trophic level species. The white-footed mouse, masked shrew, and short-tailed shrew were selected as representatives of insectivorous mammals and the Southern red-backed vole as a representative of herbivorous mammals.

While the ecotoxicological literature is quite limited with respect to wild mammals other than mustelids, Linzey (1987) found that reproductive success of white-footed mice was impaired as a result of chronic exposure to PCBs. Because these findings are consistent with effects to domesticated species of rodents and larger wild mammal species, changes in community structure and reproductive impairment were selected as assessment endpoints for these small mammals.

The measurement endpoints selected to evaluate changes in community structure are: a) population density for white-footed mice and Southern red-backed voles; and b) age structure of white-footed mouse and Southern red-backed vole populations. The measurement endpoints selected to evaluate reproductive impairment are: a) white-footed mouse litter size; b) percent reproductive female shrews and Southern red-backed voles; and c) age structure of white-footed mouse and Southern red-backed vole populations.

The analysis of these measurement endpoints will largely rely on a previously conducted field study (ChemRisk, 1994). As part of the *Evaluation of the Terrestrial Ecosystem of the Housatonic River Valley*, ChemRisk (1994) evaluated all of the measurement endpoints listed above that have been selected for insectivorous and herbivorous mammals. In order to address specific concerns raised by the Agencies, the existing data will be supplemented by white-footed mouse population data collected over the past 11 years by Jeff Boettner of the University of Massachusetts and his associates. The study areas utilized in the small mammal studies and in the reference studies are described below, followed by a discussion of the study designs employed previously and the supplemental comparisons that are planned.

Study Areas and Reference Data

As an initial step in the small mammal population and reproduction evaluations, ChemRisk identified specific study areas within the 10-year flood plain, as well as appropriate reference data. Because white-footed mice and Southern red-backed vole are predominantly forest-dwelling species, the target study area for them was the same target flood plain forest area that was used for the study of avian community structure. As described in Section 3.2.2, this target area was established in the flood plain forest south of the New Lenox Road bridge and is part of the largest undisturbed flood plain forest between that bridge and Woods Pond dam. Because an adequate reference flood plain forest could not be located, a comprehensive literature search was conducted in an effort to identify a suitable reference study for comparison to the white-footed mouse and Southern red-backed vole populations in the flood plain forest. A 27-month study conducted by Batzli (1977) in Illinois on white-footed mice and a 3-year study by Miller and Getz (1977) in

Connecticut on white-footed mice and Southern red-backed voles were chosen as reference studies for the small mammal studies. These studies had objectives similar to this analysis, adequate interpretations of findings for comparisons of results, and were conducted in flood plain forest habitats similar to the Housatonic River Valley flood plain.

Given the habitat preferences of the masked shrew and the short-tailed shrew, target and reference areas were established for these species in a shrub meadow habitat located south of the New Lenox Road bridge. The shrub meadow target area is located within the approximate 10-year flood plain, just north of the flood plain forest target area previously described. The reference shrub meadow is located outside of the 10-year flood plain, directly northeast of the Lenox Hunt Club and east of the railroad tracks.

Study Design

A mark-recapture study was conducted in the flood plain forest target area in order to evaluate the population structure of white-footed mice and Southern red-backed voles. A grid of Sherman live traps was established in the target area; traps were monitored for seven days and each mouse captured was marked. After recording the trapping location, reproductive condition, age, gender, weight, and general condition of each mouse it was released. Using the data collected on recaptures, population density estimates were calculated by age class (e.g. juvenile, subadults, and adults). These data were then compared to results of literature studies conducted by Batzli (1977) and Miller and Getz (1977).

The extremely rapid metabolism of shrews greatly limits the options available to study these species. Because shrews often starve to death within hours of being trapped, the only feasible options for studying shrew population structure is to compare captures per unit effort. To achieve this, a total of 30 pitfall traps were established along three transects in both the target and reference meadows and in the flood plain forest target area. Traps were monitored daily. Each species captured was identified and the location, age, gender, weight, and general condition of individuals were noted.

Following completion of the small mammal population structure study, five female white-footed mice, four female red-backed voles, five female short-tailed shrews (of which three were immature) and 13 female masked shrews (of which seven were immature) were collected for examination for placental scars and embryos.

After review of the 1994 ChemRisk report, the Agencies raised certain concerns regarding the interpretation of the results of the small mammal studies. One comment provided by the Agencies suggested that the reported population age structures of the small mammals, with greater proportions of juveniles compared to reference populations, could be attributed to premature death of adults as a result of exposures to PCBs. As summarized in Section 2.3.7, a large number of studies have demonstrated that reproductive impairment and survival of young are more sensitive endpoints for PCB toxicity in mammals than mortality of adults. In light of these data, it is evident that high proportions of juveniles compared to reference populations more likely reflect unimpaired reproduction in small mammals.

A second comment questioned whether population densities greater than the reference studies might be indicative of decreased predation by carnivorous birds and mammals, which might reflect population-level impacts of PCBs to higher trophic level organisms. This work plan strives to test this theory using several different lines of evidence. First, the potential risk of harm to key predators of small mammals (river otters, minks) will be assessed directly, as described below. While certain other predators -- specifically, birds of prey -- have not been selected as receptors for consideration in this assessment, the evaluation of great blue herons (discussed above) will offer a more conservative estimate of potential risk of harm to avian populations than would be provided by an evaluation of carnivorous birds, given the expected levels of PCBs in each bird species' prey items.

Additionally, to further address the Agencies' comments, the age structure and population density of white-footed mice in the Housatonic River site study area will be compared to supplemental data on the range of density and age structure of white-footed mice that are found in undisturbed ecosystems. Such data have been collected by Mr. Jeff Boettner and his colleagues at the University of Massachusetts; since 1986, they have tagged almost 20,000 small mammals (focusing on white-footed mice and deer mice) at 11 wooded sites in western Massachusetts and Cape Cod. Given the scope of his work, this database is expected to provide excellent information on the range of background population density and age structure. The results of these comparisons will be used to determine whether the relatively high population densities and proportions of juveniles that were observed in 1993 at the Housatonic River site are actually within the range of normal variation within the region. In short, the additional data will be used for purposes of more clearly interpreting the results of the Housatonic River site data collected in 1993.

3.2.5 Piscivorous Mammals

As discussed in Section 2.4.6, piscivorous mammals were selected as a receptor group of interest because of their relatively high potential for exposure and their susceptibility to PCB toxicity and to adverse reproductive effects in particular. Of the piscivorous mammals potentially inhabiting the Housatonic River valley, minks and otters have the highest potential for exposure through the consumption of Housatonic River fish. Additionally, evaluation of minks and otters is technically more feasible than for other piscivorous mammals as a result of trapping activity and the availability of data on PCB tissue residues (for otters). For these reasons, the potential effects of PCBs on piscivorous mammals will be evaluated using minks and otters as receptor species.

As noted in Section 2.3.7, the critical effects of PCBs on minks and otters relate to reproductive impairment, including survival of the young, and these effects will be evaluated through the assessment endpoints of reproductive impairment and change in community structure. For minks at least, a review of the ecotoxicological literature (discussed in Section 2.3.7) and the PCB concentrations found in fish in the Housatonic River (discussed in Section 2.1.2) indicates that, if one assumes that minks in the area obtain any significant portion of their diet from the Housatonic River fish, any literature-based evaluation or predictive modeling of the potential exposures and risks to minks due to such PCB exposure would predict a theoretical risk to minks in terms of reproductive impairment. This factor will be taken into account in the assessment of risks to piscivorous mammals. However, most of the toxicological studies reported in the literature have been conducted under artificial conditions in the laboratory and thus cannot simulate the competition, predation, and environmental conditions that affect the potential exposure of feral populations. For these reasons, we have elected to focus additional study efforts on field investigations of endpoints that would reflect reproductive success, in order to determine whether the results of such field investigations, which take into account true exposure to the animals, would bear out the predictions based on the literature.

For this purpose, a field study of mink community structure (which would reflect reproductive success) will be conducted. In addition, to further evaluate the assessment endpoints for piscivorous mammals, we will use the results of an existing retrospective field study of otters, including those in the Housatonic River valley. Each of these studies is discussed below. Their results will be used to assess whether field data--in contrast to literature-based data--indicate a risk to piscivorous mammals.

3.2.5.1 Evaluation of Mink Trapping Data

Mink and otter are actively trapped in Massachusetts and their harvest is monitored by the Massachusetts Department of Fisheries and Wildlife. While the numbers of mink harvested in Massachusetts indicate that there are sizeable populations of mink living there, the data recorded by MDFW provide no means of comparing numbers of mink caught in the Housatonic River valley to numbers caught in other areas of Massachusetts, due to the fact that the statistics recorded do not allow for the estimation of success rates (numbers of animals trapped per unit of effort) to be made. As a result, while one can compare raw numbers from one area with those from another, these numbers cannot be considered directly comparable due to the fact that there is no indication as to the relative amount of effort required to collect those animals. For example, equal numbers of mink could be harvested from two different areas with very different mink population sizes, if more traps are set more nights at the area with the lower population. Thus, while it appears that the numbers of mink trapped in the Housatonic River valley upstream and downstream of the GE facility are comparable (Pers. comm. 1993, Tom Decker, MADFW), no real conclusions concerning the size of the mink population can be drawn based on existing data.

If, however, data were collected concerning trapping effort for mink, these data could be directly applied to determine whether concentrations of PCBs in the Housatonic River are adversely affecting the reproductive success of mink. For this reason, ChemRisk is proposing to undertake a study to evaluate the reproductive success and age/sex structure of mink populations in the Housatonic River valley. The purpose of this study will be to determine whether population sizes, age structure, and sex ratios of mink trapped near the Housatonic River are comparable to those parameters measured for mink trapped in another watershed area that has not been impacted by releases from the GE facility.

This study will consist of two distinct phases. The first phase will be a scoping study to collect information on the activities and behaviors of Massachusetts trappers in order to design an appropriate survey methodology for collecting the information needed to determine reproductive success and community structure of mink. The second phase will include the implementation of a survey methodology to collect specific information about the areas trapped, the number of mink collected, their ages and sexes, and the level of effort required to obtain those animals. Each of these phases is described in detail below.

Phase I

Because there is little information publicly available concerning the numbers, activities, and behaviors of Massachusetts trappers, it is not possible at this point to design a survey methodology for collecting the necessary data. For this reason, we propose to do a preliminary study of trappers in order to identify the best means of collecting the necessary information. This phase of the study will consist of the following steps:

- Contact the Massachusetts Department of Fisheries and Wildlife to identify contact persons at and locations of tagging stations in Massachusetts. Because MDFW does not keep separate files of trapping licenses sold each year, the information on the numbers of trappers in the state and their locations cannot easily be determined. It is hoped that, as part of the required tagging process, MDFW tagging stations keep files which will provide us with the information necessary to contact these individuals.

- If MDFW is unable to provide us with the requested information, we intend to contact the Bay State Trappers Association to enlist their help in identifying local trappers who may be contacted.

- A portion of the local trappers identified will be interviewed to determine where they set their traps and their levels of activity. Depending upon the size of the population and their geographic distribution, trappers may be interviewed by telephone, mail, or by personal interview, either singly or in groups.

- An appropriate data intake form will be developed to record the requested data; however, the nature of this survey form will depend upon the methodology used to collect these preliminary data and will need to be determined once the size and geographic distribution of the trapper population have been identified.

- During the interview, trappers will be asked to provide the following information:

  - the specific areas where they place their traps;
  - their general level of trapping effort (i.e., number of nights per week, number of weeks per year);
  - a general indication of the number of mink that they trap during the season;
  - their ability to differentiate among males and females and among different age groups;
  - the frequency with which they check their traps;
  - the frequency with which they visit the tagging station;
  - the location of the tagging station(s) that they use;
  - whether there are differences in the quality of pelt for males vs. females, young vs old mink, or different parts of the trapping season;
  - whether it is easier or more difficult to catch older or younger mink;
  - their willingness to participate in a trapping effort survey next winter; and
  - their preference for participation (e.g., questionnaire to be completed at the tagging station vs. weekly questionnaire or diary study).

The data collected during Phase 1 will then be used to evaluate the feasibility of conducting a trapping effort study and to determine the best methodology to be employed in designing and implementing such a study.

Phase 2

Based on the data collected in Phase 1, a survey will be designed for the purpose of collecting information on trapping effort, harvest rates, and age and sex ratios of trapped mink (if feasible). This survey will be designed to collect information from trappers who are known to trap mink in the vicinity of the Housatonic River as well as trappers who trap mink in a similar watershed in Massachusetts that has not been impacted by releases from the GE facility in Pittsfield. In conducting this study, it will be necessary to do the following:

- Work with MDFW to select a reference watershed area where trapping is known to occur but there is no known PCB contamination.

- Evaluate Phase 1 data to identify tagging stations that are used by Housatonic River valley trappers as well as reference area trappers.

- Identify numbers of trappers known to trap in the treatment and reference areas.

- Develop survey instrument for the collection of trapping effort data. Depending upon the results of the Phase 1 data evaluation, this survey methodology may include a panel study, a telephone survey, a mail survey, or a questionnaire to be completed during visits to the tagging station. While a tagging station questionnaire would be the optimal approach, this may not be appropriate if it is determined that trappers do not visit the tagging stations on a regular basis. If visits to the tagging station are infrequent, the trappers may not be able to recall with accuracy the number of nights of effort required to harvest the pelts being tagged, or the age or sex of those animals, resulting in recall bias of survey results.

- If it is feasible to distribute questionnaires at tagging stations, the questionnaires will be distributed at the tagging stations located closest to the Housatonic River, as well as at tagging stations proximate to the reference watershed.

- Collect information on the number of pelts tagged, the number of nights of effort required to collect those pelts, and the ages and sexes of the mink harvested from both treatment and reference areas.

- Calculate trapper effort, success rates, harvest rates, age structures, and sex ratios of mink trapped from both the treatment and reference areas.

- Evaluate whether there are statistical differences between the level of effort, success or harvest rates, age structure, or sex ratios from treatment and reference areas.

The results of these analyses will be used to evaluate the potential impacts of PCBs on reproductive success and population structure in mink. If success and harvest per level of effort, as well as sex and age ratios of mink, in the Housatonic River valley are similar to these same parameters measured in the reference area, this would provide field-based evidence that PCBs in the Housatonic River valley are not producing an adverse effect on the reproductive success of mink. Such evidence will be considered along with the literature data in assessing potential risks to mink.

3.2.5.2 Retrospective Study of River Otters

In his Ph.D. dissertation, John Organ (1989) examined PCB residues in livers of trapper-harvested river otters from watersheds in Massachusetts - including the Housatonic - and related the tissue concentrations measured to an indicator of reproduction (counts of corpus luteum). Rather than initiate a comparable study to examine the potential effects of PCBs on the fecundity of piscivorous mammals, Organ's (1989) study will be discussed in detail as part of the ecological risk assessment. Based on excerpts of Organ's dissertation (1989), the methodology is briefly described below.

In November and December of 1986 and 1987, MDFW collected 210 river otter carcasses from licensed fur trappers at pelt tagging stations throughout Massachusetts. These carcasses were frozen and transferred to the Massachusetts Cooperative Wildlife Research Unit for dissection; approximately 150 grams of liver tissue were removed from each carcass. Two canine teeth were removed from each skull for age determination using radiograph (Kuen and Berg, 1983), cementum (Fancy, 1980), and dentine (Driscoll et al., 1985) analyses. In addition, female reproductive tracts were removed, each ovary was sectioned, and corpora lutea were counted as a measure of fecundity.

Two different types of tissue analysis were conducted. First, the livers of 13 adult females were individually analyzed for PCB concentrations (following the method discussed below) to allow an evaluation of a potential relationship between PCB concentrations and fecundity, using simple and multiple regression analyses.

Second, liver tissues from sixty seven river otters trapped in 1986 were composited into 33 samples, while livers from 143 river otters trapped in 1987 were composited into 70 samples and analyzed. Composites were necessary due to limitations on numbers of samples that could be submitted for analysis. Composite samples were composed of liver tissue from otters trapped within the same watershed. In some cases, composites were further refined, consisting of samples from otters caught from the same vicinity within the watershed. Trappers were interviewed to pinpoint capture locations.

Liver tissue samples that were removed for organochlorine analyses were analyzed using electron capture gas chromatography. The method detection limit for PCBs was 0.05 ppm. Total PCBs were adjusted for liver lipid level of each sample using the following formula:

$$LA = TPCB/L$$

where LA is the lipid-adjusted total PCB concentration, TPCB is the unadjusted concentration of total PCBs, and L is the fraction of lipids in the liver.

Comparisons among watersheds were made using 96 samples representing 20 watersheds with more than one sample. Organ compared tissue concentrations from the various watersheds using both total PCBs and lipid-adjusted total PCBs. Due to the non-normal distribution of PCB data, common logarithm ($\log_{10}$) transformations were performed. Geometric means of wet weight PCBs were used in all data analyses, following the procedures of Foley et al. (1988) and Schmitt (1981). The least squares method of analysis of variance (ANOVA) was used to compare differences among watersheds and between years. Most computations were conducted using version 4 of Systat Multivariate General Linear Hypothesis and Statistics modules (Wilkinson, 1988). Statistics were reported following the protocol recommended by Tacha et al. (1982).

PCBs were detected in 101 of the 103 river otter liver samples analyzed; concentrations ranged from nondetectable (ND) to 22.0 mg/kg wet weight, with a geometric mean of 1.03 mg/kg. Significant differences were found between watersheds ($p=0.03$), but not between years ($p=0.43$). The geometric mean concentration of PCBs in the Housatonic watershed samples was 2.97 mg/kg. River otter samples from the Buzzard's Bay, Merrimack, Concord, Charles, and Connecticut watersheds also had geometric mean concentrations of PCBs equal to or greater than 2 mg/kg, a tissue level that has been reported to be associated with reproductive impairment in mink (Platanow and Karsted, 1973). Lipid-adjusted total PCB concentrations ranged from ND to 6.47 mg/kg, with a geometric mean of 0.36 mg/kg. Significant differences in lipid-adjusted PCB levels were detected both between watersheds (p=0.01) and between years (p=0.01). The lipid-adjusted geometric mean PCB concentration for Housatonic watershed otters was 1.23 mg/kg.

Fecundity was evaluated on the basis of corpora lutea counts. Average fecundity for 55 adult female river otters from 1986 and 1987 was 1.7 and ranged from zero to five. Thirteen adult females that were not composited for residue analysis, but were analyzed as single samples, were evaluated for relationship of total PCBs to fecundity, as indicated by numbers of corpora lutea. The average number of corpora lutea for this sample was 1.9, ranging from zero to five. Simple linear regression showed an insignificant relationship between total PCBs and fecundity (p=0.10, r=0.48, r^2= 0.23). Multiple regression was conducted, adding the watershed average for total PCBs as an independent variable in an attempt to explain variability. This also failed to significantly explain variability (p=0.15, r=0.57, R^2=0.32).

The results of Organ's (1989) study will be used in the ecological risk assessment for the Housatonic River site as an additional means of evaluating the assessment endpoint of reproductive impairment. ChemRisk's interpretation of the results will draw substantially from Organ's discussion.

3.3 Threatened and Endangered Species

As discussed in Section 2.4.7, the evaluation of threatened and endangered species (T&Es) is required under the MCP and the RCRA Corrective Action Permit for the Housatonic River site. The determination of which species will need to be evaluated, if any, will depend on the potential presence of such species within the Housatonic River ecosystem. The sources of information that will be used to determine the likely presence of T&Es in the aquatic, terrestrial, and wetland areas will include the following:

- The Massachusetts Division of Fish and Wildlife *Atlas of Estimated Habitats of State-Listed Rare Wetlands Wildlife*;

- The Natural Heritage Program's High Priority Sites of Rare Species Habitat and Exemplary Natural Communities;

- The list of endangered, threatened, and special concern species summarized in the 1991 *MCP Interim Phase II Report* (Blasland & Bouck, 1991); and

- The list of observed endangered, threatened, and special concern terrestrial animals summarized in the *Evaluation of the Terrestrial Ecosystem of the Housatonic River Valley* (ChemRisk, 1994).

These sources do not indicate the presence of any aquatic T&Es in the study areas. The above information will be updated at the time of the study by contacting the state Natural Heritage Program and by including the results of observations made during future site-visits. The information on T&Es (both from the Natural Heritage Program, as well as from observations made during preliminary plant and animal survey proposed for the wetlands) will be expanded to cover the area within the 10-year floodplain from the confluence of the East and West Branches to Woods Pond dam, including emergent and shrub/scrub wetlands. The design of this survey is described in detail in Section 3.4.2.

If agency records or the field sightings indicate the presence of any T&E species within the study areas, potential risks of harm to that species will be evaluated. It may be possible to address such potential risks by relying on the evaluation of the receptors that have already been selected, using the studies described above. This would be the case if the T&E species is in the same taxonomic group or feeding guild as receptors already being evaluated and does not have a greater exposure potential or sensitivity to PCBs. To the extent possible, given the availability of appropriate exposure and toxicological information, potential effects on T&Es will be addressed at the individual organism level, since effects on individual T&Es are of critical importance to the population given its already stressed condition. If existing studies/information are not adequate to address potential effects on T&Es, an addendum to this work plan will be developed and submitted to present additional approaches for the evaluation of such species.

3.4 Wetland Communities

As described in Section 2.7, the potential effects of PCBs on wetlands will be evaluated using both biological receptors that inhabit the wetlands (i.e., wildlife) and functional endpoints that are potentially affected by a chemical stressor. As discussed above, plants are not a sensitive indicator of PCB effects and therefore plants (wetland vegetation) will be considered only as they serve as a habitat for wildlife.

The first step in this assessment will be to characterize the extent and types of wetlands in the study area. The study area includes the area within the 10-year floodplain between the confluence of the East and West Branches of the Housatonic River and Woods Pond Dam. At least four existing data sources are available as resources for this characterization including: the National Wetlands Inventory (NWI) maps for the East Lee and Pittsfield East USGS quadrangles; the Soil Conservation Service's 1983 field investigations of soil types within Berkshire County; the floodplain and land use maps that have been prepared on GE's behalf as part of MCP Phase II activities (e.g., Blasland, Bouck & Lee, 1994d), and the maps in the Massachusetts Division of Fish and Wildlife *Atlas of Estimated Habitats of State-Listed Rare Wetlands Wildlife*.

From this information, representative wetlands (based on type, size, and location) will be chosen for preliminary plant and animal surveys. These surveys will provide an indication of dominant vegetative community structure (wildlife habitat) and will produce a list of observed wildlife species (including T&Es, if present). It is expected that the assessment of any observed T&Es can be done largely through the studies/assessment of aquatic and terrestrial receptor species described above. The survey, supplemented with available existing information, will also provide an initial determination of local biodiversity.

In addition, a general assessment of the functions and values of the wetlands in the study area will be conducted using the standard Wetland Evaluation Technique (WET) developed by the Army Corps of Engineers (Adamus et al., 1987). The same representative wetlands will be evaluated for functions and values as are evaluated for the preliminary plant and animal surveys.

The specific tasks proposed for the evaluation of the wetland communities of the Housatonic River ecosystem are discussed below.

3.4.1 Wetland Extent and Type Characterization

As noted above, available information will be used to describe the extent and types of wetlands in the study area and to choose representative wetlands for preliminary plant and animal surveys and functionality evaluation. A limited field reconnaissance of the major wetlands and/or major hydric soil areas identified from existing maps will be conducted. Each major area identified within the study area will be subjected to a routine limited on-site evaluation of hydric soil and plant communities, following the procedures outlined in the Routine Determinations section of the Corps of Engineers Wetlands Delineation Manual (COE, 1987). The results of this field reconnaissance survey will be used to establish the general wetlands boundaries (noted on field maps) and to assist in choosing representative wetlands for further evaluation. A rationale will be provided for the selection of representative wetlands.

3.4.2 Plant and Animal Survey

A preliminary plant and animal survey will be conducted for each representative wetland identified. Upon completion of the survey, the plant communities of the representative wetlands will be described on the basis of species number, species diversity (qualitative), relative density, strata and overall dominance, and the frequency of living to dead trees. The animal communities will be described on the basis of the species observed and expected (based on geography, season, and habitat), and to the extent possible, their association with different wetland types.

A qualified biologist will determine the dominant plant species in all applicable vegetation strata. The survey strategy will be a variation of the method described by the Army Corps of Engineers (Adamus et al., 1987) with the addition of a determination of the frequency of dead trees. The method consists of the following steps:

- Establishment of transects along which several random center points are established;

- Survey of tree species within a plot defined by a 30 foot radius from the randomly placed center points;

- Survey of saplings/shrubs and woody vine species within a plot defined by a 10 foot radius from the center points; and

- Survey of herbaceous species within a 3.28 x 3.28 foot quadrant with one corner touching the observation point and one edge adjacent to the transect line.

Representative emergent and shrub/scrub wetlands will be surveyed for birds by a qualified biologist using general sampling procedures for riparian communities as outlined in Ohmart and Anderson (1986) as follows:

- Birds will be censused during the early morning hours on consecutive mornings (weather permitting) using point counts.

- Three census plots per major wetland habitat type will be selected based upon aerial photography, field reconnaissance, and available maps. Surrounding land uses and access will be considered when selecting census plots. Plot centers and the 50-meter points in each of the four cardinal directions will be flagged and the plot center coordinates noted using a Global Positioning System (GPS) receiver. The order in which sampling locations are surveyed each day will be varied.

- All birds seen or heard within a 50-meter radius of the observer will be counted during an 8-minute sampling session (Manuwal and Carey 1991). Birds seen or heard outside of the 50-meter radius will be counted separately.

- Data collected for each observation will include species, observation type (visual or auditory), and activity. Other wildlife species observed during bird census will also be noted.

- Weather condition information will be recorded at the beginning of each census session and the general habitat characteristics of each census plot will be described.

In addition to the early morning census, one 15-minute nocturnal (one to four hours after sunset) survey will be conducted on each plot to census owls, rails, and other nocturnal species (Johnson et al. 1981; Marion et al. 1981; Connors 1986). Observation will consist of 5 minutes of passive listening, followed by 10 minutes of song/call playbacks of selected nocturnal bird species suspected of occurring in the area. All species heard will be noted, including non-avian species (e.g., frogs, mammals).

Mammals, reptiles, and amphibians will be surveyed along transects placed in each major wetland habitat type (McBee 1989; Bull 1981) using similar criteria as described for birds as follows:

- A number of transects will be established; the start and end points of each transect will be flagged and the coordinates noted using a Global Positioning System (GPS).
- Each transect will be surveyed once, during which all signs and direct observations of mammals, reptiles, and amphibians will be noted.
- Rocks and logs will be overturned to search for amphibians and snakes.
- Birds and invertebrates (e.g., butterflies), or their sign, will also be noted.
- The general habitat characteristics along each transect will be described, as will weather and survey conditions.

As the activity and movement of many mammals increase at dawn and dusk, major roads within or adjacent to the representative wetlands will be searched, by car, during these times. Observed species and their locations will be noted.

Identification of T&Es and their habitats will be targeted during the plant and animal surveys and during the field component of the WET analysis. The location of all sightings will be noted on field maps and GPS coordinates will be recorded.

ChemRisk's 1994 report will be used to obtain similar information for the forested wetlands, as will any other existing data on biota observed in the wetlands study area. The results of the bird, animal, and T&E surveys described above, as well as similar information for the forested wetlands reported by ChemRisk (1994), will be assembled and reviewed as described in Section 3.4.4.

3.4.3 Wetland Functionality Assessment

In order to determine which wetland functions are of particular importance in the study area and subsequently to consider whether PCBs could adversely affect these functions, two tasks will be performed. The first is a watershed-specific data search for information relative to the habitats, wildlife, vegetation, and recreational uses of wetlands within the study area. Both published and unpublished sources will be searched.

The second task is to conduct a preliminary evaluation of wetland functions and values in the field. This will be accomplished by performing a wetland functionality assessment (WET analysis) as per the protocols outlined in the *U.S. Army Corps of Engineers Wetland Evaluation Technique* manual (Adamus et al., 1987). Maps, aerial photographs, and field data on dominant vegetation, water quality, presence of special habitat features, etc. will provide the information needed for the WET model. The WET analysis evaluates a range of functions and values in terms of site-specific significance, effectiveness, and opportunity. Probability ratings of High, Moderate, or Low are assigned.

While all of the functions identified in the protocols will be evaluated in the modeling effort, only those that would be potentially affected by exposure to PCBs will be accompanied by a detailed narrative which explains the rationale behind the resulting rating for each function. The rationale will be provided for limiting the functions on the list to those potentially affected by PCBs. The potential effects of PCBs will be discussed for those functions identified.

WET also has a provision for assessing the suitability of wetland habitat for various species of birds, fish, and invertebrates. This assessment uses the same input data as the functional assessment, but utilizes different interpretive keys. Probability ratings are assigned by these keys in a manner similar to the functional assessment.

3.4.4 Evaluation of Data

As discussed previously in Section 2.7, since the wildlife inhabiting the wetlands appear to be the most sensitive indicator of potential PCB effects on the wetlands, the same assessment endpoints used for the wildlife species discussed above should likewise be applicable to the wetlands. In addition, the wetlands will be evaluated generally for habitat degradation and loss of ecological function. The data obtained through the steps outlined above will be evaluated with reference to those assessment endpoints as follows.

First, the results of the plant and animal survey described in Section 3.4.2, together with all other available data on biota observed in the wetlands, will be used to (1) identify the wildlife species present in the wetlands, and (2) make a qualitative evaluation of the habitat for potentially affected wetlands species. The wildlife species identified in the wetlands will be compared to the aquatic and terrestrial receptor species that have been considered and evaluated for the aquatic and terrestrial communities. To the extent possible, the evaluations of those species (as described above) will be used to characterize potential risks (if any) to the wetland wildlife species. In the event that a wetland species is identified that would not be adequately addressed by the above-described studies--e.g., due to greater potential for exposure or sensitivity to PCBs--then an attempt will be made to assess the risks of such species based on other available information. If such information does not exist, it may be necessary to develop a separate proposal to gather the necessary data to make such assessment.

The results of the WET functionality analysis described in Section 3.4.3 will be used to make an overall qualitative evaluation of wetlands functions. If that analysis indicates that any of the high-probability functions may have been impaired due to PCB exposure, a determination will be made as to whether additional, quantitative data are needed to better assess that wetland function and the extent to which it may have been affected by PCBs. If it is determined that such data are needed, a supplemental proposal for obtaining the necessary data will be developed and submitted.

4.0 RISK CHARACTERIZATION

In the risk characterization, the measurement endpoint results will be evaluated to determine whether a significant risk is present for each assessment endpoint. The data generated during the analysis phase will be evaluated in the context of the hypotheses developed during problem formulation. A summary of the receptors selected, the endpoints evaluated, the results that will be considered, and their general implications is included in Table 4-1.

Since multiple measurement endpoints will be evaluated for each assessment endpoint, the possibility exists that the findings of different measurement endpoints will differ. In the practice of risk assessment, and in other scientific disciplines, differences such as these are often resolved through a weight-of-evidence determination. Weight-of-evidence refers to the process of quantitatively or qualitatively integrating independent measurement endpoints and other corroborative information, as well as the strength and appropriateness of the underlying analyses, relative to the stated assessment endpoint. Indeed, the integration of results of all measurement endpoints based on the weight-of-evidence is a critical component of risk characterization. Weight-of-evidence will be applied separately to individual endpoints and evidence of harm will be differentiated from potential for harm.

In evaluating the weight-of-evidence, specific attributes of each measurement endpoint will be evaluated, such as:

- *Strength of association between the measurement endpoint and assessment endpoint* -- This attribute refers to the extent to which the measurement endpoint is representative of, correlated with, or applicable to the assessment endpoint. If there is no association between a measurement endpoint (e.g., a study that may have been performed for some other purpose) and the assessment endpoint of interest, then that study should not be used to evaluate the stated assessment endpoint.

- *Site-specificity* -- This attribute relates to the extent to which data, media, species, environmental conditions, and habitat types used in the study design reflect the site of interest. For example, the use of site-specific biota, water, and/or sediment in a study would provide for greater site-specificity than the use of stock organisms or synthetic media. Similarly, actual field studies in the affected area provide more site-specificity than theoretical extrapolation using literature values.

- *Stressor-specificity* -- This attribute relates to the degree to which the measurement endpoint is associated with the specific stressor(s) of concern. (Stressors might include a particular chemical, waste, or physical alterations.) Some measurement endpoints may

Table 4-1. Interpretation of Results by Measurement Endpoint

Receptor of Interest	Measurement Endpoint	Measurement Approach	Results to be Evaluated	Relevance of Results to Assessment Endpoint
Benthic macroinvertebrates	Species diversity, number of taxa	Invertebrate sampling, chemical analysis, processing, and statistical analysis of data	Shannon-Weaver Diversity Index for each station	Diversity is a direct measure of community structure
	Species richness	Invertebrate sampling, chemical analysis, processing, and statistical analysis of data	Number of taxa per replicate per station	Richness is a direct measure of community structure
	Population density	Invertebrate sampling, chemical analysis, processing, and statistical analysis of data	Abundance of each species per replicate per station	Density is a direct measure of community structure
Fish	Fish community composition	Electroshock sampling and identification of collected fish, with comparison of results to reference areas and/or reference data from the literature	Fraction of total number fish collected from each site that is represented by each feeding guild	Composition of community is a direct measure of community structure
	Population density	Electroshock sampling and identification of collected fish, with comparison of results to reference areas and/or reference data from the literature	Number of fish collected of each species from each sampling station	Population density is a direct measure of community structure
	Biomass	Electroshock sampling, identification and weighing of collected fish, with comparison of results to reference areas and/or reference data from the literature	Pounds of fish collected per acre, sorted by feeding guild	Biomass is a direct measure of community structure
	Species richness	Electroshock sampling and identification of collected fish, with comparison of results to reference areas and/or reference data from the literature	Number of taxa collected from each sampling station	Richness is a direct measure of community structure
	Egg viability of white sucker and largemouth bass	Fish collection, strip spawning, laboratory incubation of fertilized eggs, chemical analysis, and statistical analysis of data, with internal referencing	Fraction of fertilized eggs for multiple levels of exposure	Egg viability is a direct measure of reproductivity, reflecting fertility of adults

Table 4-1. Interpretation of Results by Measurement Endpoint

Receptor of Interest	Measurement Endpoint	Measurement Approach	Results to be Evaluated	Relevance of Results to Assessment Endpoint
Fish (cont'd)	Hatching success of white sucker and largemouth bass	Fish collection, strip spawning, laboratory incubation of fertilized eggs, chemical analysis, and statistical analysis of data, with internal referencing	Fraction of eggs that hatch for multiple levels of exposure	Egg hatchability is a direct measure of reproductivity, reflecting embryotoxicity
	White sucker and largemouth bass post-hatch survival	Fish collection, strip spawning, laboratory incubation of fertilized eggs, chemical analysis, and statistical analysis of data, with internal referencing	Fraction of fry that survive for 14 days post hatch for multiple exposure levels	Survival of young reflects toxicity and potential reproductive impairment at the population level
Amphibians (frogs)	Frog hatching success	Collection of frogs from site and reference area, laboratory incubation of fertilized eggs, measurement of hatching success	Fractions of eggs from frogs collected from Housatonic site and reference water bodies that hatch	Egg hatchability is a direct measure of reproductivity, reflecting potential embryotoxicity
	Frog post-hatch survival	Collection of frogs from site and reference area, laboratory incubation of fertilized eggs, measurement of morphological abnormalities and mortality of young	Fractions of young from frogs collected from Housatonic site and reference water bodies that die or have morphological abnormalities	Survival of young reflects potential toxicity and potential reproductive impariment at the population level
Insectivorous birds (passerines)	Bird community composition	Avian plot census technique of Svensson (1970) for flood plain forest target area, with comparison of results to reference data from literature	Taxonomic families and species observed	Composition of community is a direct measure of community structure
	Species diversity	Avian plot census technique of Svensson (1970) for flood plain forest target area, with comparison of results to reference data from literature	Hill's (1973) diversity indices	Diversity is a direct measure of community structure
	Population density	Avian plot census technique of Svensson (1970) for flood plain forest target area, with comparison of results to reference data from literature	Total number of pairs per 40 hectares of habitat	Population density is a direct measure of community structure

Table 4-1. Interpretation of Results by Measurement Endpoint

Receptor of Interest	Measurement Endpoint	Measurement Approach	Results to be Evaluated	Relevance of Results to Assessment Endpoint
Insectivorous birds (passerines) (cont'd)	Clutch sizes for 8 species	Monitoring of activity in natural nests of 8 species of insectivorous birds located within 10-year flood plain (target nests) and outside of flood plain (reference nests)	Average numbers of eggs laid per nest by species, relative to reference nests and reference data from the literature	Clutch size is a direct measure of reproductive success, reflecting potential physiological effects on adults
	Number of young hatched per nest for 6 species	Monitoring of activity in natural nests of 8 species of insectivorous birds located within 10-year flood plain (target nests) and outside of flood plain (reference nests)	Average numbers of young hatched per nest by species, relative to reference nests	Young hatched is a direct measure of reproductive success, reflecting potential behavioral effects on adults and potential embryotoxicity
	Average hatching success across species	Monitoring of activity in natural nests of 8 species of insectivorous birds located within 10-year flood plain (target nests) and outside of flood plain (reference nests)	Average proportion of eggs laid that hatch, averaged across all species	Hatching success is a direct measure of reproductive success, reflecting potential behavioral effects on adults and potential embryotoxicity
Piscivorous birds (great blue heron)	Number of young great blue herons hatched	Analysis of MDFW data for statistically significant relationship between distance from site and young hatched per nest	Numbers of young observed per nest, as a function of colony location relative to the site	Numbers of young per nest is a direct measure of reproductive impairment
Insectivorous mammals (white-footed mouse, shrew)	Density of white-footed mouse population	Mark-recapture study in flood plain forest target area with comparisons to reference data from literature	Number of individuals per hectare	Population density is a direct measure of community structure
	Age structure of white-footed mouse population	Mark-recapture study in flood plain forest target area with comparisons to reference data from literature	Proportions of juveniles, subadults, adults within the white-footed mouse population	Age structure is a direct measure of community structure; relatively low proportions of juveniles may also reflect reproductive impairment at the population level
	White-footed mouse litter size	Examination of adult females for number of placental scars and embryos with comparisons to reference data from literature	Average observed litter size per adult female	Litter size is a direct measure reproductive success
	Percent reproductive female shrews	Examination of adult females for number of placental scars and embryos with comparisons to results from reference study area	Number of reproductively active adult females compared to total number of adult females collected	Percent reproductive females is a direct measure of reproductive activity and potential impairment

Table 4-1. Interpretation of Results by Measurement Endpoint

Receptor of Interest	Measurement Endpoint	Measurement Approach	Results to be Evaluated	Relevance of Results to Assessment Endpoint
Herbivorous mammals (southern red-backed vole)	Density of southern red-backed vole population	Mark-recapture study in flood plain forest target area with comparisons to reference data from literature	Number of individuals per hectare	Population density is a direct measure of community structure
	Age structure of southern red-backed vole population	Mark-recapture study in flood plain forest target area with comparisons to reference data from literature	Proportions of juveniles, subadults, adults within the white-footed mouse population	Age structure is a direct measure of community structure; relatively low proportions of juveniles may also reflect reproductive impairment at the population level
	Percent reproductive female southern red-backed voles	Examination of adult females for number of placental scars and embryos with comparisons to reference data from literature	Number of reproductively active adult females compared to total number of adult females collected	Percent reproductive females is a direct measure of reproductive activity and potential impairment
Piscivorous mammals (mink, river otter)	Mink harvests per unit effort	Two-phase survey of licensed trappers regarding mink harvesting activity in Housatonic and reference watersheds	Minks harvested per unit effort in Housatonic and reference watersheds	When normalized to trapping effort, harvest success reflects mink population size, which is a direct measure of community structure
	Mink age structure	Information on age classes of minks harvested will be collected as part of harvesting survey	Ages classes of minks harvested per unit effort for Housatonic and reference watersheds	Mink harvests by age class reflect mink population structure, which is a direct measure of community structure; relatively low proportions of juveniles may also reflect reproductive impairment at the population-level
	Mink sex ratio	Information on gender of minks harvested will be collected as part of harvesting survey	Gender of minks harvested per unit effort for Housatonic and reference watersheds	Mink harvests by gender reflect mink population structure, which is a direct measure of community structure; relatively low proportions of females could result in reproductive impairment at the population level
	Otter fecundity as a function of PCB tissue levels	Presentation of methods, results, and conclusions of Organ (1989), which includes field survey and laboratory bioassay	Statistical correlation between corpus luteum counts and concentrations of PCBs in livers of adult female river otters	Relationship between fecundity and PCB exposure is a direct measure of a potential stressor-response relationship for reproductive impairment

Table 4-1. Interpretation of Results by Measurement Endpoint

Receptor of Interest	Measurement Endpoint	Measurement Approach	Results to be Evaluated	Relevance of Results to Assessment Endpoint
Threatened and endangered species (if located)	Literature review	Review of existing reports, contact Natural Heritage Program	Number and identification of T&Es likely to inhabit site	Risk of harm to T&Es requires understanding of species likely to inhabit site
	Field survey	Plant and animal transect surveys	Number and identification of T&Es likely to inhabit site	Risk of harm to T&Es requires understanding of species likely to inhabit site
	Other endpoints depend on species located (if any) and may be addressed by endpoints described below for similar feeding guilds	Unknown	Unknown	Unknown
Wetlands	Wetlands characterization	Review of available information; field reconnaissance	Determination of extent and type of wetlands in area of interest	Necessary first step in characterizing potential risks to wetlands
	Plants and animals present	Plant and animal transect survey in representative wetlands	Identification and description of plant and animal communities	Use to (1) identify wildlife species present, and (2) make general assessment of habitat for wildlife
	Same measurement endpoints as for above wildlife receptors	As above	As above	Wildlife likely to be most sensitive indicator of potential PCB effects on wetlands
	Wetlands functionality	WET analysis	WET results with probability ratings for wetlands functions	Use to assess likelihood of a high-probability wetlands function being adversely affected by exposure to PCBs

respond to a broad range of stressors so that it is difficult to interpret results with regard to the stressor of concern, while other measurement endpoints are more specific to a particular stressor.

- *Quality of data and overall study* -- This attribute reflects the degree to which data quality objectives and other recognized characteristics of high quality studies are met. The key factor affecting the quality of the data is the appropriateness of data collection and analysis practices. The key factor affecting the quality of the study is the appropriateness and implementation of the experimental design and the minimization of confounding factors. If data are judged to be of poor quality, the study would be rejected for use in the ecological risk assessment.

- *Availability of an objective measure for judging effects* -- This attribute relates to the ability to judge results of the study against well-accepted standards, criteria, or objective measures. Examples of objective measures include levels of statistical significance or endpoints that are well established by the scientific or regulatory community as measures of environmental harm.

- *Sensitivity of the measurement endpoint for detecting changes* -- This attribute relates to the ability to detect a response in the measurement endpoint. The sensitivity of the measurement endpoint may be affected by natural or analytical variability.

- *Spatial representativeness* -- This attribute relates to the degree of comparability between the study area, locations of measurements or samples, locations of stressors, and locations of ecological receptors and their points of potential exposure.

- *Temporal representativeness* -- This attribute relates to the temporal comparability between the measurement endpoint (when data were collected or the period for which data are representative) and the period during which effects of concern would occur. For example, if effects of a stressor are manifested in the summer, observations made in the winter would have a low temporal representativeness.

- *Quantitative* -- This attribute relates to the degree to which numbers can be used to describe the magnitude of response of the measurement endpoint to the stressor. Some measurement endpoints may yield qualitative or hierarchial results while others may be more quantitative.

- *Correlation of stressor to response* -- This attribute relates to the degree to which a correlation is observed between levels of exposure to a stressor and levels of response, and the strength of that correlation.

- *Use of a standard method* -- The extent to which the study follows specific protocols recommended by a recognized scientific authority for conducting the method correctly. Examples of standard methods are study designs or chemical measures published in the Federal Register or the Code of Federal Regulations, developed by ASTM, or repeatedly published in the peer-reviewed scientific literature.

For a given assessment endpoint, the quality of each measurement endpoint with respect to these attributes will be compared and those measurement endpoints with the highest quality for the most attributes will be given the greatest weight in the overall characterization of risk. Additionally, the ecological significance of the results will be interpreted, in an effort to place risk estimates in the context of the types and extent of anticipated effects. Aspects of ecological significance that will be considered include: a) nature and magnitude of effects; b) spatial and temporal patterns of effects; and c) recovery potential.

Finally, the risk characterization will include an uncertainty analysis which identifies and, to the extent possible, quantifies the uncertainty associated with each measurement endpoint and with the overall conclusions. The most significant sources of uncertainty will be identified and the relative significance of each will be discussed. The uncertainty analysis will provide an evaluation of the impact of the uncertainties on the overall assessment.

5.0 SCHEDULE

In order to complete the majority of activities proposed in Section 3.0 of this work plan in 1995, the studies would need to begin by approximately April 1, 1995, due to the seasonal constraints on several of the proposed field activities. A schedule has been developed based on the assumption that the Agencies' approval of this work plan is obtained by that date. That proposed schedule is shown on Figure 5-1. Under that schedule, most of the field and laboratory work described herein would be completed in 1995, followed by a few supplemental field/laboratory studies (as necessary), as well as analysis and evaluation of the data.

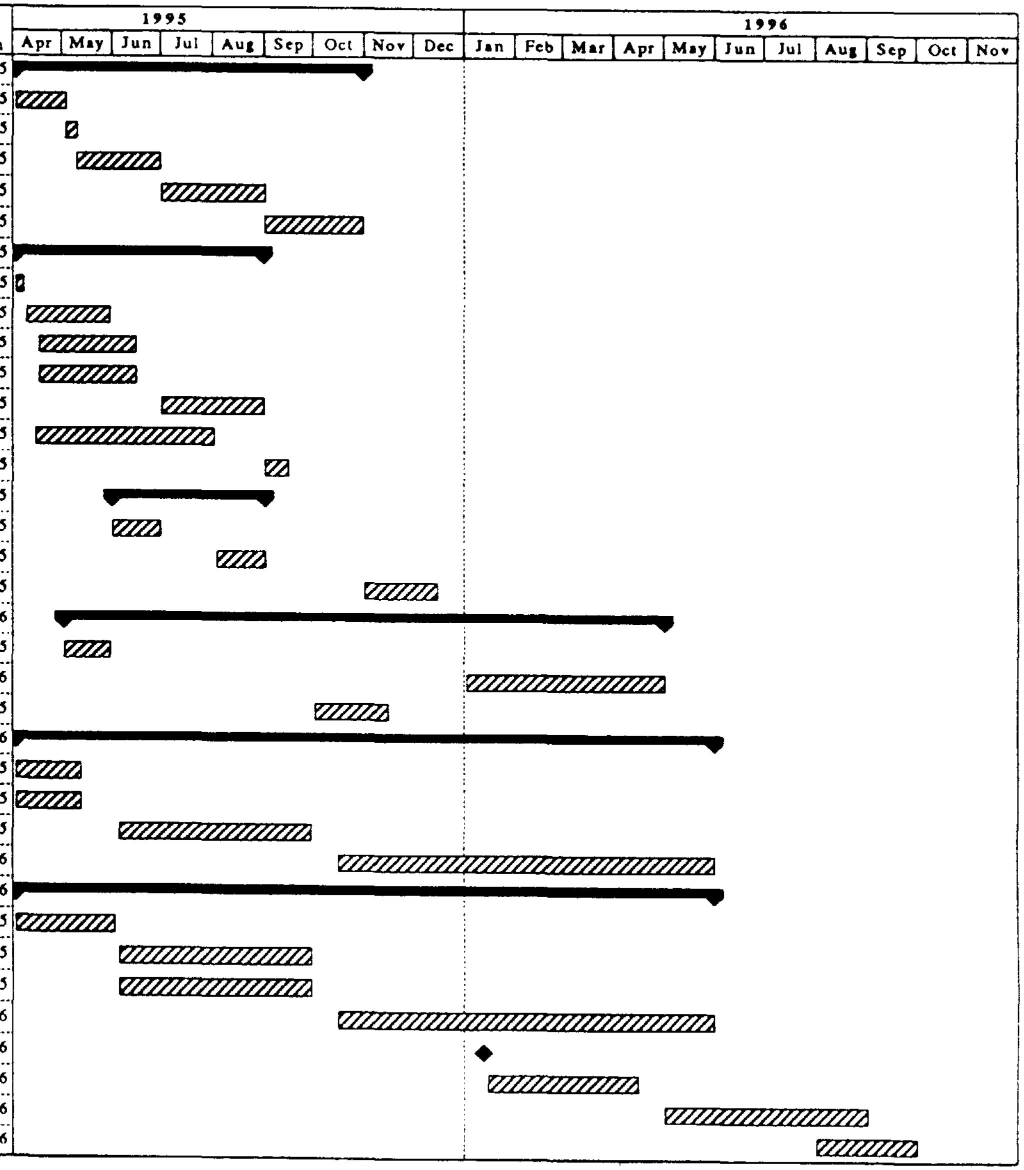
GE is currently scheduled to submit a Supplemental HEA Proposal/Risk Assessment Scope of Work for the Housatonic River site to the Agencies in January 1996. The schedule outlined on Figure 5-1 would allow that proposal to take into account in a substantial way the studies described herein. Although the overall ecological risk assessment would not be completed by that time, much of the work would be performed, and the remainder of the work could be completed while the Supplemental HEA Proposal/Risk Assessment Scope of Work is under review and during the HEA/Risk Assessment period following approval of that proposal.

If, however, the Agencies' approval of this work plan is not obtained until well after April 1, 1995, then it will be necessary to develop and submit a revised schedule. The timetable cannot simply be shifted to later months in the same order, due to the seasonal constraints on particular activities. Rather, a new schedule will need to be developed based on the actual approval date.

The proposed schedule also assumes that no supplemental proposals or studies will be necessary. If they are, the schedule would have to be extended appropriately.

Figure 5-1. Schedule of Activities for Completion of Housatonic River Ecological Risk Assessment
(Assuming Approval of Work Plan by April 1, 1995)

ID	Name	Scheduled Start	Scheduled Finish
1	Benthic Invertebrate Study	4/3/95	10/31/95
2	Initial Selection of Study Sites	4/3/95	5/3/95
3	Sediment/Invertebrate Sampling	5/3/95	5/10/95
4	Chemical Analysis of Sediments	5/10/95	6/30/95
5	Processing of Invertebrate Samples	7/1/95	9/1/95
6	Statistical Analysis of Data	9/1/95	10/31/95
7	Fish Hatching and Survival Study	4/3/95	8/31/95
8	Selection of Sampling Sites	4/3/95	4/7/95
9	Fish Collection	4/10/95	5/30/95
10	Fish Reproductivity Study	4/17/95	6/15/95
11	PCB Tissue and Egg Analysis	4/17/95	6/15/95
12	Statistical Analysis of Data	7/1/95	8/31/95
13	Frog Hatching and Survival Study	4/15/95	8/1/95
14	Additional Data Analysis for Insectivorous Birds	9/1/95	9/15/95
15	Great Blue Heron Reproductivity Survey & Analysis	6/1/95	9/1/95
16	Field Survey	6/1/95	6/30/95
17	Data Analysis and Interpretation	8/3/95	9/1/95
18	Additional Data Analysis for White-footed Mice	11/1/95	12/15/95
19	Evaluation of Mink Trapping Data	5/2/95	5/1/96
20	Phase 1	5/2/95	5/30/95
21	Phase 2	1/2/96	5/1/96
22	Summary of Retrospective Study on River Otters	10/2/95	11/15/95
23	Evaluation of Threatened & Endangered Species	4/3/95	5/31/96
24	Literature Search	4/3/95	5/12/95
25	Agency Consultation	4/3/95	5/12/95
26	Field Survey	6/5/95	9/29/95
27	Data Analysis	10/16/95	5/31/96
28	Wetland Community Assessment	4/3/95	5/31/96
29	Characterization of Type & Extent	4/3/95	6/2/95
30	Plant and Animal Survey	6/5/95	9/29/95
31	Wetland Functionality Assessment	6/5/95	9/29/95
32	Evaluation of data	10/16/95	5/31/96
33	Submittal of Supplemental HEA Proposal	1/12/96	1/12/96
34	Agency Review (estimated)	1/15/96	4/15/96
35	Risk Characterization*	5/1/96	9/1/96
36	Report Preparation*	8/1/96	10/1/96



*The time table for these tasks depends on the length of the Agency review of the Supplemental HEA Proposal, as well as on whether any supplement investigations are deemed necessary.

6.0 REFERENCES

Abramowicz, D.A. and M.J. Brennan. 1991. 4.5 Anaerobic and aerobic biodegradation of endogenous PCBs. In: *Biological Remediation of Contaminated Sediments, with Special Emphasis on the Great Lakes*. C.T. Jafvert and J.E. Rodgers (eds.). U.S. Environmental Protection Agency, Athens, GA. PB91-161679. January. 79-87.

Academy of Natural Sciences of Philadelphia. 1993. *PCB Concentrations in Fishes from the Housatonic River, Connecticut, in 1984 to 1992*. The Academy of Natural Sciences of Philadelphia, Philadelphia, PA. August.

Adamus, P.R., E.J. Clairain, Jr., R.D. Smith, and R.E. Young. 1987. *Wetland Evaluation Technique (WET); Volume II: Methodology*. Operational Draft Technical Report. U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

Albers, P.H., L. Sileo, and B.M. Mulhern. 1986. Effects of environmental contaminants on snapping turtles of a tidal wetland. *Arch. Environ. Contam. Toxicol.* 15:39-49.

Ankley, G.T., D.E. Tillitt, J.P. Giesy, P.D. Jones, and D.A Verbrugge. 1991. Bioassay-derived 2,3,7,8-tetrachlorodibenzo-p-dioxin equivalents in PCB-containing extracts from the flesh and eggs of Lake Michigan chinook salmon (*Oncorhynchus tshawytscha*) and possible implications for reproduction. *Can. J. Fish. Aquat. Sci.* 48:1685-1689.

ATSDR. 1993. *Toxicological Profile for Selected PCBs (Aroclor -1260, -1254, -1248, -1232, -1221, and -1016)*. U.S. Public Health Service for Toxic Substances and Disease Registry, Atlanta, GA.

Aulerich, R.J. and R.K. Ringer. 1977. Current Status of PCB toxicity to mink and effect on their reproduction. *Arch. Environ. Contam. Toxicol.* 6:279-292.

Aulerich, R.J., R.K. Ringer, and S. Iwamoto. 1973. Reproductive failure and mortality in mink fed on Great Lakes fish. *J. Reprod. Fert.* Suppl. 19:365-376.

Aulerich, R.J., R.K. Ringer, and J. Safronoff. 1986. Assessment of primary vs. secondary toxicity of Aroclor® 1254 to mink. *Arch. Environ. Contam. Toxicol.* 15: 393-399.

Aulerich, R.J., R.K. Ringer, H.L. Seagren, and W.G. Youatt. 1971. Effects of feeding coho salmon and other Great Lakes fish on mink reproduction. *Can. J. Zool.* 49:611-616.

Aulerich, R.J., S.J. Bursian, W.J. Breslin, B.A. Olson, and R.K. Ringer. 1985. Toxicological manifestations of 2,4,5,2',4',5'-, 2,3,6,2',3',6'-, and 3,4,5,3',4',5'-hexachlorobiphenyl and Aroclor 1254 in mink. *J. Toxicol. Environ. Health.* 15:63-79.

Backlin, B. and A. Bergman. 1992. Morphological aspects on the reproductive organs in female mink (*Mustela vison*) exposed to polychlorinated biphenyls and fractions thereof. *Ambio* 21(8):596-601.

Bailey, S. and P.J. Bunyan. 1972. Interpretation of persistence and effects of polychlorinated biphenyls in birds. *Nature (London)* 236:34. (Cited in Hamdy and Gooch, 1986)

Batzli, G.O. 1977. Population dynamics of the white-footed mouse in floodplain and upland forests. *Am. Field Mid. Naturalist* 97(1):18-32.

Becker, P.H., S.Schumann, and C. Koepff. 1993. Hatching failure in common terns (*Sterna hirundo*) in relation to environmental chemicals. *Environ. Pollut.* 79:207-213.

Begon, M. and M. Mortimer. 1986. *Population Ecology: A Unified Study of Animals and Plants.* Second Edition. Boston: Blackwell Scientific Publications.

Bengtsson, B-E. 1980. Long-term effects of PCB (Clophen A50) on growth, reproduction and swimming performance in the minnow, *Phoxinus phoxinus. Water Res.* 14:681-687.

Bergeron, J.M., D. Crews, and J.A. McLachlan. 1994. PCBs as environmental estrogens: Turtle sex determination as a biomarker of environmental contamination. *Environ. Health Persp.* 102(9):780-781.

Bird, D.M., P.H. Tucker, G.A. Fox, and P.C. Lague. 1983. Synergistic effects of Aroclor® 1254 and mirex on the semen characteristics of American kestrels. *Arch. Environ. Contam. Toxicol.* 12:633-640.

Birge, W.J., J.A. Black, and A.G. Westerman. 1978. *Effects of Polychlorinated Biphenyl Compounds and Proposed PCB-Replacement Products on Embryo-Larval Stages of Fish and Amphibians.* Prepared by Kentucky Water Resources Research Institute, Lexington for the Office of Water Research and Technology, Washington. DC.

Bishop, C.A., R.J. Brooks, J.H. Carey, P. Ng, R.J. Norstrom, and D.R.S. Lean. 1991. The case for a cause-effect linkage between environmental contamination and development in eggs of the common snapping turtle (*Chelydra S. serpentina*) from Ontario, Canada. *J. Toxicol. Environ. Health* 33:521-547.

Black, D.E., D.K. Phelps, and R.L. Lapan. 1988. The effect of inherited contamination on egg and larval winter flounder, *Pseudopleuronecties americanus. Mar. Environ. Res.* 25:45-62.

Blasland & Bouck Engineers, P.C. 1991. *MCP Interim Phase II Report/Current Assessment Summary for Housatonic River.* Blasland & Bouck Engineers, Syracuse, NY. December.

Blasland & Bouck Engineers, P.C. 1992a. *Addendum to MCP Interim Phase II Report/Current Assessment Summary for Housatonic River.* Blasland & Bouck Engineers, Syracuse, NY. August.

Blasland & Bouck Engineers, P.C. 1992b. *Housatonic River Floodplain STM Report: Land Use and Property Ownership Map.* Plates 6 and 7. Blasland & Bouck Engineers, Syracuse, NY. August.

Blasland & Bouck Engineers, P.C. 1992c. *Summary of Housatonic River Floodplain Property Sampling & Analysis.* Blasland & Bouck Engineers, Syracuse, NY. October.

Blasland & Bouck Engineers, P.C. 1993. *Report on January 1993 Housatonic River Floodplain Property Sampling and Analysis.* Blasland & Bouck Engineers, Syracuse, NY. February.

Blasland, Bouck & Lee, Inc. 1994a. *MCP Supplemental Phase II Scope of Work and Proposal for RCRA Facility Investigation of Housatonic River and Silver Lake.* Blasland, Bouck & Lee, Inc., Syracuse, NY. June.

Blasland, Bouck & Lee, Inc. 1994b. *Housatonic River Floodplain Properties, Results of Supplemental Site Characterization Sampling.* Blasland, Bouck, & Lee, Inc., Syracuse, NY. February.

Blasland, Bouck & Lee, Inc. 1994c. *Sampling and Analysis Plan/Data Collection and Analysis Quality Assurance Plan.* Blasland, Bouck, & Lee, Inc., Syracuse, NY. May.

Blasland, Bouck & Lee, Inc. 1994d. *MCP Phase II and STM Evaluation Flood Plain Sampling Results (series of maps).* Blasland, Bouck, & Lee, Inc., Syracuse, NY. March.

Bleavins, M.R., R.J. Aulerich, and R.K. Ringer. 1980. Polychlorinated biphenyls (Aroclors 1016 and 1242): Effects on survival and reproduction in mink and ferrets. *Arch. Environ. Contam Toxicol.* 9:627-635.

Bleavins, M.R., R.J. Aulerich, and R.K. Ringer. 1981. Placental and mammary transfer of polychlorinated and polybrominated biphenyl in the mink and ferret. In: *Avian and Mammalian Wildlife Toxicology*. D.W. Lamb and E.E. Kenaga (eds.). Philadelphia, PA: ASTM. pp. 121-131.

Block,W.M., L.A. Brennan, and R.J. Gutierrez. 1986. The use of guilds and guild-indicator species for assessing habitat suitability. In: *Wildlife 2000: Modelling habitat relationships of terrestrial vertebrates.* J. Verner, M.L. Morrison, and C.J. Ralph (eds.). Madison: University of Wisconsin Press. pp. 109-113. (Cited in EPA, 1991)

Block, W.M., L.A. Brennan, and R.J. Gutierrez. 1987. Evaluation of guild-indicator species for use in resource management. *Environ. Manage.* 11(2):265-269. (Cited in EPA 1991)

Bode, R.W., M.A. Novak, and L.E. Abele. 1993. *20 Year Trends in Water Quality of Rivers and Streams in New York State Based on Macroinvertebrate Data 1972–1992.* New York State Department of Environmental Conservation, Albany, NY.

Brown, G.W. 1964. The metabolism of amphibia. In: *Physiology of the Amphibia.* Moore, J.A. (ed.). New York, NY: Academic Press. (Cited in Niethammer et al., 1984)

Brown, M.P., J.J. McLaughlin, J.M. O'Connor, and K. Wyman. 1982. A mathematical model of PCB bioaccumulation in plankton. *Ecol. Model.* 15:29. (Cited in Hamdy and Gooch, 1986)

Brown, J.F., D.L. Bedard, M.J. Brennan, J.C. Carnahan, H. Feng, and R.E. Wagner. 1987a. Polychlorinated biphenyl dechlorination in aquatic sediments. *Science* 236:709-712.

Brown, J.F., R.E. Wagner, H. Feng, D.L. Bedard, M.J. Brennan, J.C. Carnahan, and R.J. May. 1987b. Environmental dechlorination of PCBs. *Environ. Toxicol. Chem.* 6:579-593.

Brunstrom, B. and L. Reutergardh. 1986. Differences in sensitivity of some avian species to the embryotoxicity of a PCB, 3,3'4,4'-tetrachlorobiphenyl, injected into eggs. *Environ. Pollut. (A)* 42:37-45.

Brunstrom, B. 1988. Sensitivity of embryos from duck, goose, herring gull and various chicken breeds to 3,3',4,4'-tetrachlorobiphenyl. *Poultry Science* 67:52-57

Brunstrom, B., D. Broman, and C. Naf. 1990. Embryotoxicity of polycyclic aromatic hydrocarbons (PAHs) in three domestic avian species, and of PAHs and coplanar polychlorinated biphenyls (PCBs) in the common eider. *Environ. Pollut.* 67: 133-143.

Brunstrom, B. 1991. Toxicity and EROD-inducing potency of polychlorinated biphenyls (PCBs) and polycyclic aromatic hydrocarbons (PAHs) in avian embryos. *Comp. Biochem. Physiol.* 100C(1/2):241-243.

Brunstrom, B., H. Hakansson, and K. Lundberg. 1991. Effects of a technical PCB preparation and fractions therof on ethoxyresorufin o-deethylase activity, vitamin A levels and thymic development in the mink (*Mustela vison*). *Parmacol. Toxicol.* 69:421-426.

Bryan, A.M., P.G. Olafsson, and W.B. Stone. 1987a. Disposition of low and high environmental concentrations of PCBs in snapping turtle tissues. *Bull. Environ. Contam. Toxicol.* 38:1000-1005.

Bryan, A.M., W.B. Stone, and P.G. Olafsson. 1987b. Disposition of toxic PCB congeners in snapping turtle eggs: Expressed as toxic equivalents of TCDD. *Bull. Environ. Contam. Toxicol.* 39:791-796.

Bull, E.L. 1981. Indirect estimates of abundance of birds. *Stud. Avian Biol.* 6:76-80.

Burger, J. 1994. How should success be measured in ecological risk assessment? The importance of predictive accuracy. *J. Toxicol. Environ. Health* 42:367-376.

Butler, P.A. and R.L. Schutzmann. 1979. Bioaccumulation of DDT and PCB in tissues of marine fishes. *Aquatic Toxicol.* ASTM STP 667:212-220.

Cetinkaya, M., R. Silwar, and W. Thiemann. 1985. Organochloro pesticide residues in coffee cherries, green coffee and in leaves of the coffee tree. *Chem. Mikrobiol. Technol. Lebesm.* 9:33-36.

Chadwick & Associates, Inc. 1994. *Aquatic Ecology Assessment of the Housatonic River, Massachusetts.* Chadwick & Associates, Inc., Littleton, CO.

Chapman, J. A., & Feldhamer, G. A. (Eds.). 1982. *Wild Mammals of North America: Biology, Management, and Economics.* Baltimore, MD: The John Hopkins University Press.

ChemRisk. 1993. Revised Risk Assessment to Evaluate the Need for Short Term Measures in the Flood Plain of the Housatonic River, Volumes I and II. McLaren/Hart-ChemRisk, Portland, ME. April.

ChemRisk. 1994. *Evaluation of the Terrestrial Ecosystem of the Housatonic River Valley*. McLaren/Hart-ChemRisk, Portland, ME. July.

Christensen, C., C. Dal Morin, S. Hall, D. Kennedy, J. Kimball, B. Lord, C. McLaughlin, and H. Mueller. 1980. Forty-third breeding bird census: 20. Floodplain forest. *Am. Birds* 34(1):49.

Christensen, S. Hall, A. Lindsey, K. Meyer, H. Mueller, and H. Wiley. 1982. Forty-fifth breeding bird census: 22. Floodplain forest. *Am. Birds* 36(1):58.

COE. 1987. *Corps of Engineers Wetlands Delineation Manual, Part IV.B* . U.S. Army Corps of Engineers, Environmental Laboratory, Department of the Army, Waterways Experiment Station, Vicksburg, MS. Technical Report Y-87-1. January.

Cole D.R. and F.W. Plapp. 1974. Inhibition of growth and photosynthesis in *Chlorella pyrenoidosa* by a polychlorinated biphenyl and several insecticides. *Environ. Ent.* 3(2):217-220.

Cole, M.A., P.B. Reichart, and D.K. Button. 1979. Reversible bioconcentration of monochlorobiphenyls by *Rhodotorula rubra*: Correlations with aqueous solubility of substrate. *Bull. Environ. Contam. Toxicol.* 23:44. (Cited in Hamdy and Gooch, 1986)

Connolly, J.P. 1991. Application of a food chain model to polychlorinated biphenyl contamination of the lobster and winter flounder food chains in New Bedford Harbor. *Environ. Sci. Technol.* 25(4):760-770.

Connors, P.G. 1986. Marsh and shorebirds. In: *Inventory and Monitoring of Wildlife Habitat*. A.Y. Cooperride, R.J. Boyd, and H.R. Stuart (eds.), Washington, DC: U.S. Department of the Interior. pp. 351-370.

Courtney, W.A.M. and W.J. Langston. 1978. Uptake of polychlorinated biphenyl (Aroclor 1254) from sediment and from seawater in two intertidal polychaetes. *Environ. Pollut.* 15:303. (Cited in Hamdy and Gooch, 1986)

Craigie, J.S. and O. Hutzinger. 1975. Effects of commercial chlorinated hydrocarbons and specific chlorobiphenyls on the growth of seven species of marine phytoplankton. *Chemosphere* 4:139. (Cited in Mahanty, 1986)

Criswell, J.H. and D. Gauthey. 1983. Forty-sixth breeding bird census: 15. Mature deciduous floodplain forest. *Am. Birds* 37(1):56.

Crotteau, M., C. Dal Morin, R. Fox, S. Giduz, S. Hall, K. Hopkins, D. Kennedy, H. Mueller, D. Newman, S. O'Bryan, D. Richards, P. Spiegel, H. Wiley, and H. Wilson. 1978. Forty-first breeding bird census: 28. Floodplain forest. *Am. Birds* 32(1):63.

Crotteau, M., C. Dal Morin, S. Hall, T. Herbert, D. Kennedy, L. Moore, H. Mueller, M. Schultz, B. Simpson, J. Whitehead, and H. Wilson. 1979 Forty-second breeding bird census: 21. Floodplain Forest. *Am. Birds* 33(1):62.

Custer, T.W. and G.H. Heinz. 1980. Reproductive success and nest attentiveness of mallard ducks fed Aroclor 1254. *Environ. Pollut. (Series A)* 21:313-318.

Custer, T.W., G.L. Hensler, and T.E. Kaiser. 1983. Clutch size, reproductive success, and organochlorine contaminants in Atlantic coast black-crowned night-herons. *The Auk* 100:699-710.

Dahlgren, R.B. and R.L. Linder. 1971. Effects of polychlorinated biphenyls on pheasant reproductions, behavior and survival. *J. Wildl. Manage.* 35:315-319. (Cited in Peakall and Peakall, 1973)

Decker, T. 1993. Memorandum to T. Abel, Associate Environmental Scientist, McLaren/Hart/ChemRisk®, Portland, ME from T. Decker, Certified Wildlife Biologist, Massachussetts Department of Fisheries and Wildlife, Westborough, MA re: Mink and Otter Harvest Information for Massachussetts. July 26.

De Kock, A.C. and D.A. Lord. 1988. Kinetics of the uptake and elimination of polychlorinated biphenyls by an estuarine fish species (*Rhabdosargus holubi*) after aqueous exposure. *Chemosphere* 17(12):2381-2390.

DeFoe, D.L., G.D. Veith, and R.W. Carlson. 1978. Effects of Aroclor 1248 and 1260 on the fathead minnow (*Pimephales promelas*). *J. Fish. Res. Bd. Can.* 35:997-1002.

DeGraaf, R.M. and D.D. Rudis. 1986. *New England Wildlife: Habitat, Natural History, and Distribution.* Broomall: U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station. General Technical Report NE-108.

den Besten, P.J., H.J. Herwig, D.I. Zandee, and P.A. Voogt. 1989. Effects of cadmium and PCBs on reproduction of the sea star *Asterias rubens*: Aberrations in the early development. *Ecotoxicology and Environmental Safety* 18:173–180.

DEP. 1995. Massachusetts Contingency Plan. 310 CMR 40.0900. Commonwealth of Massachusetts Department of Environmental Protection, Office of Research and Standards, Boston, MA. January.

Diamond, J.M. and R.M. May. 1976. Island biogeography and the design of natural reserves. In: *Theoretical Ecology*. R.M. May (ed.). Oxford: Blackwell Scientific Publishers.

Diercxsens, P., D. deWeck, N. Borsinger, B. Rosset, and J. Tarradellas. 1985. Earthworm contamination by PCBs and heavy metals. *Chemosphere* 14(5):511–522.

Dillon, T.M., W.H. Benson, R.A. Stackhouse, and A.M. Crider. 1990. Effects of selected PCB congeners on survival, growth, and reproduction. *Environ. Toxicol. Chem.* 9:1317-1326.

DiPinto, L.M., B.C. Coull, and G.T. Chandler. 1993. Lethal and sublethal effects of the sediment-associated PCB Aroclor 1254 on a meiobenthic copepod. *Environ. Toxicol. Chem.* 12:1909-1918.

Driscoll, K.M., G.S. Jones, and F. Nichy. 1985. An efficient method by which to determine age of carnivores, using dentine rings. *J. Zool.* 205:309-313.

Dunning, J. 1994. *Secrets of the Nest. The Family Life of North American Birds.* Boston, MA: Houghton Mufflin Company.

Ehrlich, P.R., D.S. Dobkin, and D. Wheye. 1988. *The Birders Handbook: A Field Guide to the Natural History of North American Birds.* New York: Simon and Schuster.

Elliott, J.E., R.W. Butler, R.J. Norstrom, and P.E. Whitehead. 1989. Environmental contaminants and reproductive success of great blue herons *Ardea herodias* in British columbia, 1986-87. *Environ. Pollut.* 59:91-114.

EPA. 1989a. *Risk Asssessment Guidance for Superfund. Vol II: Environmental Evaluation Manual Interim Final.* U.S. Environmental Protection Agency, Office of Emergency and Remedial Response, Washington, DC. EPA/540/1-89/001. March.

EPA. 1989b. *Ecological Assessments of Hazardous Waste Sites: A Field and Laboratory Reference Document.* U.S. Environmental Protection Agency, Environmental Research Laboratory, Corvallis, OR. EPA 600/3-89-013.

EPA. 1989c. *Risk Asssessment Guidance for Superfund. Vol. I: Human Health Evaluation Manual (Part A).* U.S. Environmental Protection Agency, Office of Emergency and Remedial Response, Washington, DC. EPA/540/1-89/002. December.

EPA. 1991. *Criteria for Choosing Indicator Species for Ecological Risk Assessment at Superfund Sites.* U.S. Environmental Protection Agency, Washington, D.C. EPA 101/F-90/051. April.

EPA. 1992. *Framework for Ecological Risk Assessment.* U.S. Environmental Protection Agency, Risk Assessment Forum, Washington, D.C. EPA/630/R-92/001.

EPA. 1993a. *Great Lakes Water Quality Initiative Criteria Documents for the Protection of Wildlife (Proposed).* U.S. Environmental Protection Agency, Office of Science and Technology, Washington, DC. PB93-154722. April.

EPA. 1993b. *Wildlife Exposure Factors Handbook. Volumes I and II.* U.S. Environmental Protection Agency, Office of Research and Development, Washington, D.C. EPA/600/R-93/187a and b. December.

EPA. 1994a. *Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments.* U.S. Environmental Protection Agency, Environmental Response Team, Edison, NJ. September 26.

EPA. 1994b. Field studies for ecological risk assessment. *Eco Update* 2(3). U.S. Environmental Protection Agency, Office of Solid Waste and Emergency Response. EPA 540-F-94-014. September.

EPA. 1994c. *Health Assessment Document for 2,3,7,8-Tetrachlorodibenzo-p-Dioxin (TCDD) and Related Compounds.* U.S. Environmental Protection Agency, Office of Research and Development, Washington, D.C. EPA/600/BP-92/001. August.

Faber, R.A. and J.J. Hickey. 1973. Eggshell thinning, chlorinated hydrocarbons, and mercury in inland aquatic bird eggs, 1969 and 1970. *Pest. Monit. J.* 7(1):27-36.

Fancy, S.G. 1980. Preparation of mammalian teeth for age determination by cementum layers: A review. *Wildl. Soc. Bull.* 8: 242-248.

Fisher, N.S. and C.F. Wurster. 1973. Individual and combined effects of temperature and PCB's on the growth of three species of phytoplankton. *Environ. Pollut.* 5:205. (Cited in Mahanty, 1986)

Fisher, N.S. and C.F. Wurster. 1974. Impact of pollutants on plankton communities. *Environ. Conserv.* 1:189. (Cited in Mahanty, 1986)

Fisher, N.S., E.J. Carpenter, C.C. Remsen, and C.F. Wurster. 1974. Effects of PCB on inter-specific competition in natural and gnotobiotic phytoplankton communities in continuous and batch cultures. *Microbiol. Ecol.* 1:39-50. (Cited in Mahanty, 1978)

Fisher, N.S., R.R. L. Guillard, and C.F. Wurster. 1976. Effects of chlorinated hydrocarbons pollutant on the growth kinetics of a marine diatom. In: *Modelling Biochemical Processes in Aquatic Ecosystems.* R.P. Canale (ed.). Ann Arbor, MI: Ann Arbor Science. pp. 305. (Cited in Mahanty, 1986)

Focardi, S., C. Leonzio, and C. Fossi. 1988. Variations in polychlorinated biphenyl congener compositions in eggs of Mediterranean water birds in relation to their position in the food chain. *Environ. Pollut.* 52:243-255 .

Foley, R.E., S.J. Jackling, R.J. Sloan, and M.K. Brown. 1988. Organochlorine and mercury residues in wild mink and otter: Comparison with fish. *Environ. Toxicol. Chem.* 7:363-374.

Freeman, H.C., G. Sangalang, and B. Flemming. 1982. The sublethal effects of a polychlorinated biphenyl (Aroclor 1254) diet on the Atlantic Cod (*Gadus morhua*). *The Science of the Total Environment* 24:1–11.

Fries, G.F. and G.S. Marrow. 1981. Chlorobiphenyl movement from soil to soybean plants. *J. Agric. Fd. Chem.* 29:757-759.

Frink , C.R., B.L. Sawhney, K.P. Kulp, and G.G. Fredett. 1982. *Polychlorinated Biphenyls in Housatonic River Sediments in Massachusetts and Connecticut: Determination, Distribution, and Transport: Bulletin 800.* Connecticut Agricultural Experiment Station. December. (Cited in Blasland & Bouck, 1991)

Gauthey, D. 1984. Forty-seventh breeding bird census: 22. Mature deciduous floodplain forest. *Am. Birds* 38(1):73.

Gauthey, D. 1989. Breeding bird census: 1988. 6. Mature deciduous floodplain forest. *J. Field Ornithol.* 60(1)Supplement:28-29.

Gauthey, D. 1990. Breeding bird census: 1989. 8. Mature deciduous floodplain forest. *J. Field Ornithol.* 61(1)Supplement:32.

Gauthey, D. 1991. Breeding bird census: 1990. 11. Mature deciduous floodplain forest. *J. Field Ornithol.* 62(1)Supplement:39.

Gauthey, D. 1992. Breeding bird census: 1991. 17. Mature deciduous floodplain forest. *J. Field Ornithol.* 63(1)Supplement:43-44.

Glooschenko, V. and W. Glooschenko. 1975. Effect of polychlorinated biphenyl compounds on growth of Great Lakes phytoplankton. *Can. J. Bot.* 53:653.

Godin, A.J. 1977. *Wild Mammals of New England.* Chester: Globe Pequot Press.

Gooch, J.A. and M.K. Hamdy. 1983. Uptake and concentration factor of Aroclor 1254 in aquatic organisms. *Bull. Environ. Contam. Toxicol.* 31:445–452.

Goven, A.J., G.S. Eyambe, L.C. Fitzpatrick, B.J. Venables, and E.L. Cooper. 1993. Cellular Biomarkers for measuring toxicity of xenobiotics: Effects of polychlorinated biphenyls on earthworm *Lumbricus terrestris* coelomocytes. *Environ. Toxicol. Chem.* 12:863-870.

Greichus, Y.A. and B.A. Dohman. 1980. Polychlorinated biphenyl contamination of areas surrounding two transformer salvage companies Colman, South Dakota - September 1977. *Pest. Monit. J.* 14(1):26-30.

Grue, C.E., et al., 1983. Assessing hazards of organophosphate pesticides to wildlife. *Transactions of the North American Wildlife and Natural Resources Conference.* 48:200-220. (Cited in EPA, 1991)

Guiney, P.D., M.J. Melancon, J.J. Lech, and R.E. Peterson. 1979. Effects of egg and sperm maturation and spawning on the distribution an delimitation of a polychlorinated biphenyl in rainbow trout (*Salmo gairdneri*). *Toxicol. Appl. Pharmcol.* 47:261-272. (Cited in Vodicnik and Peterson, 1985)

Hall, S., K. Meyer, H. Mueller, D. Stuckey, J. Trainer, J. Whitehead, and H. Wiley. 1984. Forty-seventh breeding bird census: 35. Floodplain forest. *Am. Birds* 38(1):77.

Hall, S. 1990. Fifty-third breeding bird census: 19. Floodplain forest. *J. Field Ornithol.* 61(1):41.

Halter, M.T. and H.E. Johnson. 1974. Acute toxicities of a polychlorinated biphenyl (PCB) and DDT alone and in combination to early life stages of coho salmon (*Oncorhynchus kisutch*). *J. Fish. Res. Board Can.* 31:1543-1547.

Hamilton, W.J. and J.O. Whitaker. 1979. *Mammals of the Eastern United States.* Second Edition. Ithaca: Comstock Publishing Associates.

Hansen, D.J., P.R. Parrish, J.I. Lowe, A.J. Wilson, and P.D. Wilson. 1971. Chronic toxicity, uptake, and retention of Aroclor® 1254 in two estuarine fishes. *Bull. Environ. Contam. Toxicol.* 6(2):113-119.

Hansen, D.J., P.R. Parrish, and J. Forester. 1974a. Aroclor 1016: Toxicity to and uptake by estuarine animals. *Environ. Res.* 7:363-373.

Hansen, D.J., S.C. Schimmel, and J. Forester. 1974b. Aroclor 1254 in eggs of sheepshead minnows: effect on fertilization success and survival of embryos and fry. *J. Proc. Annu. Conf. Southeast Assoc. Game Fish Comm.* 27:420–426.

Haque, R., D.W. Schmedding, and V.H. Freed. 1974. Aqueous solubility, adsorption, and vapor behavior of polychlorinated biphenyl Aroclor 1254. *Environ. Sci. Technol.* 8:139.

Harding, G.C. and R.F. Addison. 1986. Accumulation and effects of PCBs in marine invertebrates and vertebrates, Chapter 2. In: *PCBs and the Environment*, Volume 2. J.S. Waid (ed.). Boca Raton, FL: CRC Press.

Harding, L.W. and J.H. Phillips. 1978. Polychlorinated biphenyl (PCB) uptake by marine phytoplankton. *Mar. Biol.* 49:103. (Cited in Hamdy and Gooch, 1986)

Harris, M.P. and D. Osborn. 1981. Effect of a polychlorinated biphenyl on the survival and breeding of puffins. *J. Appl. Ecol.* 18:471-479. (Cited in Becker et al., 1993)

Harris, H.J., T.C. Erdman, G.T. Ankley, and K.B. Lodge. 1993. Measures of reproductive success and polychlorinated biphenyl residues in eggs and chicks of Forster's Terns on Green Bay, Lake Michigan, Wisconsin - 1988. *Arch. Environ. Contam. Toxicol.* 25:304-314.

Hawes, M.L., J.C.Kricher, and J.C. Urey. 1976. The effects of various Aroclor fractions on the population growth of *Chlorella pyrenoidosa*. *Bull. Environ. Contam. Toxicol.* 15(1):14-18.

Heath, R.G., J.W. Spann, J.F. Kreitzer, and C. Vance. 1972. Effects of polychlorinated biphenyls on birds. *Proc. XV int. Ornithol. Congr.* pp. 475-85. (Cited in Peakall and Peakall, 1973)

Henning, M.H., N.W. Harrington, N.M. Shear, and T.J. Iannuzzi. 1994a. Distributions for key exposure factors controlling the uptake of xenobiotic chemicals by great blue herons (*Ardea herodius*) through ingestion of fish. *Submitted to Environmental Toxicology and Chemistry*.

Henning, M.H., T.J. Iannuzzi, N.W. Harrington, and N.M. Shear. 1994b. Exposure factors controlling the uptake of xenobiotic chemicals by herons and egrets through ingestion of fish. *15th Annual Meeting of the Society of Environmental Toxicology and Chemistry (SETAC)*. Denver, Colorado. October 30-November 3.

Henny, C.J., L.J. Blus, S.V. Gregory, and C.J. Stafford. 1981. *PCBs and Organochlorine Pesticides in Wild Mink and River Otters from Oregon*. Patuxent Wildlife Research Center, Pacific Northwest Field Station, Corvallis, OR. pp. 1763-1780.

Heyer, W.R., M.A. Donnelly, R.W. McDiarmid, L.C. Hayek, and M.S. Foster. 1994. *Measuring and Monitoring Biological Diversity: Standard Methods for Amphibians.* Smithsonian Institution Press, Washington, DC.

Hill, M.O. 1973. Diversity and evenness: A unifying notation and its consequences. *Ecology* 54:427-432. (Cited in Ludwig and Reynolds, 1988)

Hoffman, R.D. 1978. The diets of herons and egrets in southwestern Lake Erie. In: *Wading Birds.* (Sprunt, A., J. Ogden, and S. Winckler, eds.) Natl. Audubon Soc. Res. Rep. 7: 365-369.
Hoffman, D.J., B.A. Rattner, L. Sileo, D. Docherty, and T.J. Kubiak. 1987. Embryotoxicity, teratogenicity, and aryl hydrocarbon hydroxylase activity in Forster's terns on Green Bay, Lake Michigan. *Environ. Res.* 42: 176-184.

Holm, G., L. Norrgren, T. Andersson, and A. Thurén. 1993. Effects of exposure to food contaminated with PBDE, PCN or PCB on reproduction, liver morphology and cytochrome P450 activity in the three-spined stickleback, *Gasterosteus aculeatus. Aquatic Toxicol.* 27:33-50.

Hornshaw, T.C., R.J. Aulerich, and H.E. Johnson. 1983. Feeding Great Lakes fish to mink: Effects on mink and accumulation and elimination of PCBs by mink. *J. Toxicol. Environ. Health.* 11:933-946.

Hornshaw, T.C., J. Safronoff, R.K Ringer, and R.J. Aulerich. 1986. LC_{50} test results in polychlorinated biphenyl-fed mink: Age, season, and diet comparions. *Arch. Environ. Contam. Toxicol.* 15: 717-723.

Iwata, Y., F.A. Gunther, and W.E. Westlake. 1974. Uptake of a PCB (Aroclor 1254) from soil by carrots under field conditions. *Bull. Environ. Contam. Toxicol.* 11:523-528.

Iwata, Y. and F.A. Gunther. 1976. Translocation of the polychlorinated biphenyl Aroclor 1254 from soil into carrots under field conditions. *Arch. Environ. Contam. Toxicol.* 4:44-59.

Johansson N., S. Jensen, and M. Olsson. 1970. PCB indications of effects on fish. *PCB Conference 1*, National Swedish Environment Protection Board, p. 59–67. (cited in Monod, 1985).

Johnson, R.R., B.T. Brown, L.T. Haight, and J.M. Simpson. 1981. Playback recordings as a special avian censusing techniques. *Stud. Avian Biol.* 6:68-75.

Kahn, M.A., R.M. Rao, and A.F. Novak. 1976. Adsorption of polychlorinated biphenyl (Aroclor 1254) on shrimp. *Bull. Environ. Contam. Toxicol.* 16:503. (Cited in Hamdy and Gooch, 1986)

Keil, J.E., L.E. Priester, and S.H. Sandifer. 1971. Polychlorinated biphenyl (Aroclor 1242): Effects of uptake on growth, nucleic acids, and chlorophyll of a marine diatom. *Bull. Environ. Contam. Toxicol.* 6:156. (Cited in Hamdy and Gooch, 1986)

Kihlstrom, J.E., M. Olsson, S. Jensen, A. Johansson, J. Ahlbom, and A. Bergman. 1992. Effects of PCB and different fractions of PCB on the reproduction of the mink (*Mustela vison*). *Ambio* 21(8):563-569.

Kimbrough, R., J. Buckley, L. Fishbein, G. Flamm, L. Kasza, W. Marcus, S. Shibko, and R. Teske. 1978. Animal toxicology. *Environ. Health Perspect.* 24:173-185.

King, K.A., T.W. Custer, and J.S. Quinn. 1991. Effects of mercury, selenium, and organochlorine contaminants on reproduction of Forster's terns and black skimmers nesting in a contaminated Texas bay. *Arch. Environ. Contam. Toxicol.* 20:32-40.

Klekowski, E.J. and E. Klekowski. 1982. Mutation in ferns growing in an environment contaminated with polychlorinated biphenyls. *Am. J. Bot.* 69(5):721-727.

Klemm, D.J., P.A. Lewis, F. Fulk, and J.M. Lazorchak. 1990. *Macroinvertebrate Field and Laboratory Methods for Evaluating the Biological Integrity of Surface Waters.* U.S. Environmental Protection Agency, Office of Water, Washington, DC. EPA/600/4–90/030.

Korschgen, L.T. 1958. December food habits of mink in Missouri. *J. Mammal.* 39:521-527. (Cited in Chapman, 1982)

Korschgen, L.J. and T.S. Baskett. 1963. Food of Impoundment- and stream-dwelling bullfrogs in Missouri. *Herpetologica* 19(2):89-99.

Kuen, D.W. and W.E. Berg. 1983. Use of radiographs to age otters. *Wildl. Soc. Bull.* 11:68-70.

Kubiak, T.J., H.J. Harris, L.M. Smith, T.R. Schwartz, D.L. Stalling, J.A. Trick, L. Sileo, D.E. Docherty, and T.C. Erdman. 1989. Microcontaminants and reproductive impairment of the Forster's tern on Green Bay, Lake Michigan - 1983. *Arch. Environ. Contam. Toxicol.* 18:706-727.

Larsson, P. 1986. Zooplankton and fish accumulate chlorinated hydrocarbons from contaminated sediments. *Can. J. Fish. Aquat. Sci.* 43:1463-1466.

Larsson, P., G. Bremle, and L. Okla. 1993. Uptake of pentachlorophenol in fish of acidified and nonacidified lakes. *Bull. Environ. Contam. Toxicol.* 50:653-658.

Lawler, Matusky & Skelly Engineers. 1994. *Housatonic River Connecticut Cooperative Agreement - Task IV.B, PCB Fate and Transport Model: Additional Monitoring and Model Verification.* Lawler, Matusky, and Skelly Engineers, Pearl River, NY. November.

Lawrence, J. and H.M. Tosine. 1977. Polychlorinated biphenyl concentrations in sewage and sludges of some waste treatment plants in Southern Ontario. *Bull. Environ. Contam. Toxicol.* 17:49. (Cited in Weber and Overcash, 1980)

Lee, H. 1992. Models, muddles, and mud: Predicting bioaccumulation of sediment–associated pollutants. In: *Sediment Toxicity Assessment.* G.A. Burton, Jr. (ed.), Boca Raton, FL: Lewis Publishers, pp. 267–293.

Lester, D.C. and A. McIntosh. 1994. Accumulation of polychlorinated biphenyl congeners from Lake Champlain sediments by *Mysis relicta. Environ. Toxicol. Chem.* 13(11):1825-1841.

Lewis, S.C., J.R. Lynch, and A.I. Nikiforov. 1990. A new approach to deriving community exposure guidelines from "No-Observed-Adverse-Effect Levels." *Reg. Toxicol. Pharmacol.* 11:314-330.

Lieb, A.J., D.D. Bills, and R.O. Sinnhuber. 1974. Accumulation of dietary polychlorinated biphenyls (Aroclor 1254) by rainbow trout (*Salmo gairdneri*). *J. Agric. Fd. Chem.* 22(4):638-642.

Lindvall, M.L. and J.B. Lowe. 1980. Effects of DDE, TDE, and PCBs on shell thickness of Western grebe eggs, Bear River Migratory Bird Refuge, Utah - 1973-74. *Pest. Monit. J.* 14(3): 108-111.

Linscombe, G., N. Kinler, and R.J. Aulerich. 1982. Chpater 31: Mink. In: *Wild Mammals of North America: Biology, Management, and Economics*. Chapman, J.A. and G.A. Feldhamer (eds.). Baltimore: Johns Hopkins University Press. 629-643.

Linzey, A.V. 1987. Effects of chronic polychlorinated biphenyls exposure on reproductive success of white-footed mice (*Peromyscus leucopus*). *Arch. Environ. Contam. Toxicol.* 16:445-460.

Lovejoy, T.E. and D.C. Oren. 1981. The minimum critical size of ecosystems. In *Forest Island Dynamics in Man-Dominated Landscapes*. R.L. Burgess and D.L. Sharpe (eds.). New York: Springer-Verlag.

Lowe, J.I., P.R. Parish, J.M. Patrick, Jr., and J. Forrester. 1972. Effects of the polychlorinated biphenyl Aroclor 1254 on the American oyster, *Crassostrea virginica*. *Mar. Biol.* 17:209. (Cited in Hamdy and Gooch, 1986)

Ludwig, J.A. and J.F. Reynolds. 1988. *Statistical Ecology: A Primer on Methods and Computing*. New York: John Wiley & Sons.

Mac, M.J. and C.C. Edsall. 1991. Environmental contaminants and the reproductive success of lake trout in the Great lakes: an epidemiological approach. *J. Toxicol. Environ. Health* 33:375-394.

Mac, M.J., T.R. Schwartz, C.C. Edsall, and A.M. Frank. 1993. Polychlorinated biphenyls in Great Lakes Lake Trout and their eggs: Relations to survival and congener composition 1979–1988. *J. Great Lakes Res.* 19(4):752–765.

Mac, M.J., and J.G. Seelye. 1981a. Potential influence of acetone in aquatic bioassays testing the dynamics and effects of PCBs. *Bull. Environm. Contam. Toxicol.* 27:359-367.

Mac, M.J. and J.G. Seelye. 1981b. Patterns of PCB accumulation by fry of lake trout. *Bull. Environ. Contam. Toxicol.* 27:368-375.

MacArthur, R.H. and E.O. Wilson. 1967. *The Theory of Island Biogeography*. Princeton: Princeton University Press.

MacDonald, D.D. 1994. *Sediment Injury in the Southern California Bight: Review of the Toxic Effects of DDTs and PCBs in Sediments.* MacDonald Environmental Sciences Ltd., Ladysmith, B.C., Canada. August.

Mack, C.M. 1985. *River Otter Restoration in Grand County, Colorado.* Master's Degree Thesis, Colorado State University, Fort Collins, CO. Fall.

Manuwal, D.A. and A.B. Carey. 1991. *Methods for Measuring Populations of Small Diurnal Forest Birds.* USDA Forest Service, General Technical Report PNW-GTR-278.

Mahanty, H.K. 1975. A study on the effects of polychlorinated biphenyl (Aroclor 1242) on an aquatic plant - *Spirodela oligorrhiza* Kurz. Helm. *Bull. Environ. Contam. Toxicol.* 14:558. (Cited in Pal et al., 1980)

Marion, W.R., T.E. O'Meara, and D.S. Maehr. 1981. Use of playback recordings in sampling elusive or secretive birds. *Stud. Avian Biol.* 6:81-85.

Mauck, W.L., P.M. Mehrle, and F.L. Mayer. 1978. Effects of the polychlorinated biphenyl Aroclor 1254 on growth, survival, and bone development in brook trout (*Salvelinus fontinalis*). *J. Fish Res. Bd. Can.* 35:1084-1088.

Maughan, J.T. 1993. *Ecological Asessment of Hazardous Waste Sites.* New York: Van Nostrand Reinhold.

Mayer, F.L, P.M. Mehrle, and H.O. Sanders. 1977. Residue dynamics and biological effects of polychlorinated biphenyls in aquatic organisms. *Arch. Environ. Contam. Toxicol.* 5:501-511. (Cited in Dillon et al., 1990)

McBee, K. 1989. Field surveys: Terrestrial vertebrates. In: *Ecological Assessment of Hazardous Waste Sites: A Field and Laboratory Reference.* W. Warren-Hicks, B.R. Parkhurst, and S.S. Baker, Jr. (eds.). Washington, DC: Environmental Protection Agency. EPA/600/3-89/013. pp. 8-58 - 8-72.

McFarland, V.A. and J.U. Clarke. 1989. Environmental occurrence, abundance, and potential toxicity of polychlorinated biphenyl congeners: Considerations for a congener-specific analysis. *Environ. Health Perspect.* 81:225-239.

McIntyre, G.A. 1953. Estimation of plant density using line transects. *J. Ecology* 41:319-330.

McLane, A.R. and D.L. Hughes. 1980. Reproductive success of screech owls fed Aroclor 1248. *Arch. Environm. Contam. Toxicol.* 9:661-665.

McLeese, D.W. and C.D. Metcalfe. 1980. Toxicities of eight organochlorine compounds in sediment and seawater to *Crangon septemspinosa*. *Bull. Environ. Contam. Toxicol.* 25: 921–928.

MDFW. 1991. *Abstracts of the 1992 Fish and Wildlife Laws.* The Commonwealth of Massachusetts Division of Fisheries and Wildlife, Boston, MA. September.

MDFW. 1979. *Field Investigation Report: Great Blue Heron Rookery Inventory.* Commonwealth of Massachusetts Division of Fisheries and Wildlife. October 1.

MDFW. 1980. *Field Investigation Report: Great Blue Heron Rookery Inventory, 1980.* Commonwealth of Massachusetts Division of Fisheries and Wildlife. June 30.

MDFW. 1981. *Field Investigation Report: Great Blue Heron Rookery Inventory, 1981.* Commonwealth of Massachusetts Division of Fisheries and Wildlife. June 30.

MDFW. 1982. *Field Investigation Report: Great Blue Heron Rookery Inventory, 1982.* Commonwealth of Massachusetts Division of Fisheries and Wildlife. July 16.

MDFW. 1983. *Field Investigation Report: Great Blue Heron Rookery Inventory, 1983.* Commonwealth of Massachusetts Division of Fisheries and Wildlife. July 3.

MDFW. 1984. *Field Investigation Report: Great Blue Heron Rookery Inventory Results.* Commonwealth of Massachusetts Division of Fisheries and Wildlife. October 16.

MDFW. 1985. *Field Investigation Report: Great Blue Heron Rookery Inventory Results.* Commonwealth of Massachusetts Division of Fisheries and Wildlife. December 16.

MDFW. 1986a. *Field Investigation Report: Great Blue Heron Rookery Inventory Results.* Commonwealth of Massachusetts Division of Fisheries and Wildlife. February 19.

MDFW. 1986b. *Field Investigation Report: Great Blue Heron Rookery Inventory, 1986.* Commonwealth of Massachusetts Division of Fisheries and Wildlife. November 25.

MDFW. 1987. *Field Investigation Report: Great Blue Heron Rookery Inventory, 1987.* Commonwealth of Massachusetts Division of Fisheries and Wildlife. October 9.

MDFW. 1989. *Statewide Great Blue Heronry Inventory, 1989.* Commonwealth of Massachusetts Division of Fisheries & Wildlife.

MDFW. 1991. *Memorandum: 1991 Great Blue Heronry Survey.* Commonwealth of Massachusetts Division of Fisheries & Wildlife. August 19.

MDFW. 1992. *Abstracts of the 1993 Fish and Wildlife Laws.* The Commonwealth of Massachusetts Division of Fisheries and Wildlife, Boston, MA. September.

Miller, D.H. and L.L. Getz. 1977. Comparisons of population dynamics of peromyscus and clethrionomys in New England. *J. Mammal.* 58(1):1-16.

Monod, G. 1985. Egg mortality of Lake Geneva charr (*Salvelinus alpinus* L.) contaminated by PCB and DDT derivatives. *Bull. Environ. Contam. Toxicol.* 35:531–536.

Moore, S.A. and R.C. Harris. 1972. Effects of polychlorinated biphenyl on marine phytoplankton communitites. *Nature* 240:356. (Cited in Hamdy and Gooch, 1986)

Morgan, J.R. 1972. Effects of Aroclor 1242 (a polychlorinated biphenyl) and DDT on cultures of an alga, protozoan, daphnid, ostracod, and guppy. *Bull. Environ. Contam. Toxicol.* 8(3):129-137.

Moriarty F. 1969. The effects of polychlorinated biphenyls on *Chorthippus brunneus* (Saltatoria: Acridae). *Entomol. Exp. Appl.* 12:206-210.

Mosser, J.L., N.S. Fisher, T.C. Teng, and C.F. Wurster. 1972. Polychlorinated biphenyls: Toxicity to certain phytoplankters. *Science* 175:191. (Cited in Mahanty, 1986)

Moza, P.N., I. Scheunert, W. Klein, and F. Korte. 1979. Studies with 2,4',5-trichlorobiphenyl-^{14}C and 2,2',4,4',6-pentachlorobiphenyl-^{14}C in carrots, sugar beets and soil. *J. Agric. Fd. Chem.* 27:1120-1124. (Cited in O'Connor et al., 1990)

Mueller-Dumbois, D. and H. Ellenberg. 1974. *Aims and Methods of Vegetative Ecology*. New York: John Wiley and Sons.

Mueller, H. and S. Hall. 1991. Breeding bird census: 1990. 16. Floodplain forest. *J. Field Ornithol.* 62(Supplement 1):42.

Mueller, H. and N. Mueller. 1992. Breeding bird census: 1991. 24. Floodplain forest. *J. Field Ornithol.* 63(Supplement 1):47-48.

Mueller, H. 1993. Breeding bird census: 1992. 20. Floodplain forest. *J. Field Ornithol.* 64(Supplement 1):27.

Murado, M.A., M.C. Tejedor, and G. Baluja. 1976. Interactions between polychlorinated biphenyls (PCBs) and soil microfungi. Effects of Aroclor 1254 and other PCBs on *Aspergillus flavus* cultures. *Bull. Environ. Contam. Toxicol.* 15:768. (Cited in Hamdy and Gooch, 1986)

Murray, J.A. 1987. *Wildlife in Peril: The Endangered Mammals of Colorado. River Otter, Black-footed Ferret, Wolverine, Lynx, Grizzly Bear, Gray Wolf.* Boulder: Roberts Rinehart, Inc. Publishers.

NAS. 1979. *Polychlorinated Biphenyls.* Rep. Comm. Assess. PCBs in Environ., Environ. Stud. Bd., Comm. Nat. Resource, National Research Council, National Academy Science, Washington DC. (Cited in USFWS, 1985a)

National Audubon Society. 1991. *The Audubon Society Pocketguides: Familiar Mammals, North America.* New York: Chanticleer Press, Inc.

Nebeker, A.V., F.A Puglisi, and D.L. DeFoe. 1974. Effect of polychlorinated biphenyl compounds on survival and reproduction of the fathead minnow and flagfish. *Trans. Am. Fish. Soc.* 4:722–728.

NERBC. 1980. *Housatonic River Basin Overview.* New England River Basin Commission. Boston, MA. September.

Niethammer, K.R., T.S. Baskett, and D.H. White. 1984. Organochlorine residues in three heron species as related to diet and age. *Bull. Environ. Contam. Toxicol. 33*:491-498.

Niimi, A.J. 1983. Biological and toxicological effects of environmental contaminants in fish and their eggs. *Can. J. Fish. Aquat. Sci.* 40:306-312.

NOAA. 1992. *Monthly Station Normals of Temperature, Precipitation, and Heating and Cooling Degree Day 1961-1990 Massachusetts Climatography of the United States No. 81*. National Oceanic and Atmospheric Administration, National Environmental Satellite, Date and Information Service, National Climatic Data Center, Asheville, NC. January.

Norheim, G. and B. Kjos-Hanssen. 1984. Persistent chlorinated hydrocarbons and mercury in birds caught off the west coast of Spitzbergen. *Environ. Pollut.* 33A:143-152. (Cited in USDOI, 1985)

NRC. 1986. *Ecological Knowledge and Environmental Problem-Solving: Concepts and Case Studies*. National Research council, National Academy Press, Washington, DC. (Cited in EPA, 1992)

O'Connor, G.A., D. Kiehl, G.A.Eiceman, and J.A. Ryan. 1990. Plant uptake of sludge-borne PCBs. *J. Environ. Qual.* 19:113-118.

Ohmart, R.D. and B.W. Anderson. 1986. Riparian habitats. In: *Inventory and Monitoring of Wildlife Habitat*. A.Y. Cooperrider, R.J. Boyd, and H.S. Stuart (eds.), Washington, DC: U.S. Department of the Interior. pp. 169-200.

O'Shea, T.J., T.E. Kaiser, G.R. Askins, and J.A. Chapman. 1980. Polychlorinated biphenyls in a wild mink population. In: *Worldwide Furbearer Conference Proceedings*, August 3-11, 1980, Frostburg, Maryland. pp. 1746-1751.

Olafsson, P.G., A.M. Bryan, B. Bush, and W. Stone. 1983. Snapping turtles - a biological screen for PCBs. *Chemosphere* 12(11/12):1525-1532.

Opperhuizen, A. and S.M. Schrap. 1988. Uptake efficiencies of two polychlorobiphenyls in fish after dietary exposure to five different concentrations. *Chemosphere* 17(2):253-262.

Organ, J.F. 1989. Mercury and PCB residues in Massachusetts river otters: Comparisons on a watershed basis. Ph.D. Thesis, University of Massachusetts, Amherst, MA.

Overcash, M.R., A.L. McPeters, E.J. Dougherty, and R.G. Carbonell. 1991. Diffusion of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin in soil containing organic solvents. *Environ. Sci. Technol.* 25(8):1479-1485.

Paine, J.M., M.J. McKee, and M.E. Ryan. 1993. Toxicity and bioaccumulation of soil PCBs in crickets: Comparison of laboratory and field studies. *Environ. Toxicol. Chem.* 12:2097-2103.

Pal, D., J.B. Weber, and M.R. Overcash. 1980. Fate of polychlorinated biphenyls (PCBs) in soil-plant systems. *Residue Reviews* 74:45-98.

Peakall, D.B. 1971. Effect of polychlorinated biphenyls (PCBs) on the eggshells of ring doves. *Bull. Environ. Contam. Toxicol.* 6:100-101. (Cited in Peakall and Peakall, 1973)

Peakall, D.B., J.L. Lincer, and S.E. Bloom. 1972. Embryonic mortality and chromosomal alterations caused by Aroclor 1254 in ring doves. *Environ. Health Perspect.* 1:103-104.

Peakall, D.B. and M.L. Peakall. 1973. Effect of a polychlorinated biphenyl on the reproduction of artificially and naturally incubated dove eggs. *J. Appl. Ecol.* 10: 863-868.

Pizzuto, E. 1991. Memorandum to David Rigg, Blasland & Bouck Engineers, P.C., from Ernest Pizzuto, Connecticut Department of Environmental Protection re: Invertebrate PCB samples, Hartford, CT. September 18.

Platanow, N.S. and L.H. Karstad. 1973. Dietary effects of polychlorinated biphenyls on mink. *Can. J. Comp. Med.* 37:391-400.

Polikarpov, C.G., P. Parsi, and S.W. Fowler. 1983. Chronic effects of a PCB (DP5) upon *Nereis diversicolor* in spiked Mediterranean sediments. *Rapp. Comm. int. Mer Medit.* 28(7):167–168.

Rasmussen, J.B., D.J. Rowan, D.R.S. Lean, and J.H. Carey. 1990. Food chain structure in Ontario lakes determines PCB levels in lake trout (*Salvelinus namaycush*) and other pelagic fish. *Can. J. Fish. Aquat. Sci.* 47:2030-2038.

Rhett, G., D.M.M. Adema, P. Roza, and R. Henzen. 1989. *Rate and Effects of PCB Accumulation on Eisenia foetida.* U.S. Army Corps of Engineers, Waterways Experiment States, Vicksburg, MS. September 21.

Ringer, R.K., R.J. Aulerich, and M.R. Bleavins. 1981. Biological effects of PCBs and PBBs on mink and ferrets - A review. In: *Toxicology of Halogenated Hydrocarbons.* M.A.Q. Kahn (ed.). New York: Permagon Press. pp. 329-343. (Cited in Ringer, 1983)

Ringer, R.K. 1983. ,Chapter 17: Toxicology of PCBs in mink and ferrets. In: *PCBs: Human and Environmental Hazards.* F. D'Itri and M. Kamrin (eds.). Woburn, MA: Butterworth Publishing. pp. 227-241.

Riseborough, R.W. and D.W. Anderson. 1975. Some effects of DDE and PCB on mallards and their eggs. *J. Wildl. Mgmt.* 39: 508-513. (Cited in Custer and Heinz, 1980)

Roberts, T.H., and C.J. O'Neil. 1985. Species selection for habitat assessments. *Transactions of the North American Wildlife and Natural Resources Conference* 50:352-362. (Cited in EPA, 1991)

Rodriguez-Grau, J., B.J. Venables, L.C. Fitzpatrick, A.J. Goven, and E.L. Cooper. 1989. Suppression of secretory rosette formation by PCBs in *Lumbricus terrestris*: An earthworm assay for humoral immunotoxicity of xenobiotics. *Environ. Toxicol. Chem.* 8:1201-1207.

Rogers, J.J. 1990. Observations on the snapper. *Mass. Wildlife* XL(2):2-7.

SAB. 1994. U.S. Environmental Protection Agency Science Advisory Board, Bioaccumulation Subcommittee of the Ecological Processes and Effects Committee and Drinking Water Committee. Briefing Material. April 28-29. Washington, DC.

Safe, S. 1990. Polychlorinated biphenyls (PCBs), dibenzo-p-dioxins (PCDDs), dibenzofurans (PCDFs), and related compounds: Environmental and mechanistic considerations which support the development of toxic equivalency factors (TEFs). *CRC Crit. Rev. Toxicol.* 21(1):51-88.

Safe, S. 1994. Polychlorinated biphenyls (PCBs): Environmental impact, biochemical and toxic responses, and implications for risk assessment. *Crit. Rev. Toxicol.* 24(2):87-149.

Sanders, H.O. and J.H. Chandler. 1972. Biological magnification of a polychlorinated biphenyl (Aroclor 1254) from water by aquatic invertebrates. *Bull. Environ. Contam. Toxicol.* 7:257. (Cited in Hamdy and Gooch, 1986)

Sawhney, B.L. and L. Hankin. 1984. Plant contamination by PCBs from amended soils. *J. Fd. Protect.* 47(3):232-236.

Schimmel, S.C., D.J. Hansen, and J. Forester. 1974. Effects of Aroclor 1254 on laboratory reared embryos and fry of sheepshead minnows (*Cyprinodon variegatus*). *Trans. Am. Fish. Soc.* 3:582-586.

Schmitt, C.J. 1981. Analysis of variance as a method for examining contaminant residues in fish: National pesticide monitoring program. Pages 270-298 In: *Aquatic Toxicology and Hazard Assessment: Fourth Conference.* D.R. Branson and K.L. Dickson (eds.). ASTM STP 737.

Schonherr J. and M. Reiderer. 1989. Foliar penetration and accumulation of organic chemicals in plant cuticles. *Rev. Environ. Contam. Toxicol.* 108:1-70. (Cited in Jensen et al., 1992)

Scott, M.L., D.V. Vadehra, P.A. Mullenhoff, G.L. Rumsey, and R.W. Rice. 1971. Results of experiments on the effects of PCBs on laying hen performance. Proc. Cornell Nutr. Conf. pp. 56-64. (Cited in Peakall and Peakall, 1973)

Sealander, J.A. 1943. Winter food habits of mink in southern Michigan. *J. Wildlife Management* 7(4):411.

Serafin, J.A. 1984. Avian species differences in intestinal absorption of xenobiotics (PCBs, dieldrin, Hg^{2+}). *Comp. Biochem. Physiol.* 78C(2):491-496.

Sericano, J.L., S.H. Safe, T.L. Wade, and J.M. Brooks. 1991. Toxicological significance of non-, mono- and di-ortho-substituted polychlorinated biphenyls in oysters from Galveston and Tampa Bays. *Environ. Toxicol. Chem.* 13(11):1797-1803.

Shen, T.T. and T.J. Tofflemire. 1980. Air pollution aspects of land disposal of toxic wastes. *J. Environ. Eng. Div. (Am. Soc. Civ. Eng.)* 106:211-226.

Sheppard, M.I., S.C. Sheppard, and B.D. Amiro. 1991. Mobility and plant uptake of inorganic ^{14}C and ^{14}C-labeled PCB in soils of high and low retention. *Health Physics* 61(4):481-492.

Smith, L.M., T.R. Schwartz, K. Feltz, and T.J. Kubiak. 1990. Determination of AHH-active polychlorinated biphenyls, 2,3,7,8-tetrachlordibenzo-p-dioxin and 2,3,7,8-tetrachlorodibenzofuran in Lake Michigan sediment and biota. The question of their relative significance. *Chemosphere.* 21(9):1063-1085.

Snarski, V.M. and F.A. Puglisi. 1976. *Effects of Aroclor 1254 on Brook Trout, Salvelinus fontinalis.* U.S. Environmental Protection Agency, Ecological Research Series, Washington, D.C. EPA/600/3-76-112. (Cited in Mayer et al., 1985)

Sodergren, A. 1971. Accumulation and distribution of chlorinated hydrocarbons in cultures of *Chlorella pyrenoidosa* (Chlorophyceae). *Oikos* 22:215. (Cited in Mahanty, 1986)

Spies, R.B., D.W. Rice, P.A. Montagna, and R.R. Ireland. 1985. Reproductive success, xenobiotic contaminants and hepatic mixed-function oxidase (MFO) activity in *Platichthys stellatus* populations from San Francisco Bay. *Mar. Environ. Res.* 17:117-121.

Spies, R.B., P. Thomas, M. Matsui, and D.E. Hinton. 1994. *Final Report for Southern California Fish Injury Studies.* Allied Marine Sciences, Inc., Livermore, CA and Industrial Economics Inc., Cambridge, MA. September.

Stalling, D.L. and F.L. Mayer, Jr. 1972. Toxicities of PCBs to fish and environmental residues. *Environ. Health Perspect.* 1:159. (Cited in Hamdy and Gooch, 1986)

Stange, K. and D.L. Swakhamer. 1994. Factors affecting phytoplankton species-specific differences in accumulation of 40 polychlorinated biphenyls (PCBs). *Environ. Toxicol. Chem.* 13(1):1849-1860.

Stewart Laboratories, Inc. 1982. *Housatonic River Study - 1980 and 1982 Investigations, volumes I and II.* Knoxville, TN. December. (Cited in Blasland & Bouck, 1991)

Stickel, W.H. 1975. Some effects of pollutants in terrestrial ecosystems. In: *Ecological Ticology Research.* A.D. McIntyre and C.F. Mills (eds.). New York: Plenum Press. pp. 25074. (Cited in EPA, 1991)

Stickel, W.H., L.F. Stickel, R.A. Dyrland, and D.L. Hughes. 1984. Aroclor 1254® Residues in birds: lethal levels and loss rates. *Arch. Environ. Contam. Toxicol.* 13:7-13.

Strek, H.J., J.B. Weber, P.J. Shea, E. Mrozek, and M.R. Overcash. 1981. Reduction of polychlorinated biphenyl toxicity and uptake of carbon-14 activity by plants through the use of activated carbon. *J. Agric. Fd. Chem.* 29(2):288-293.

Strickland, M.A., C.W. Douglas, M. Novak, and N.P. Hunziger. 1982. Chapter 29: Marten. In: *Wild Mammals of North America: Biology, Management, and Economics*. Chapman, J.A. and G.A. Feldhamer (eds.). Baltimore: Johns Hopkins University Press. pp. 599-612.

Stromberg, K.L., T.B. Dale, J.M. Larson, W.H. Karsov, D.R. Goldberg, and L. Sileo. 1994. Polychlorinated hydrocarbon (PCH) effects on double-crested cormorant reproduction in Green Bay, WI: Hypothesis tests. *15th Annual Meeting of the Society of Environmental Toxicology and Chemistry (SETAC)*, Denver, Colorado. October 30-November 3.

Struger, J. and D.V. Weseloh. 1985. Great Lakes Caspian terns: Egg contaminants and biological implications. *Colonial Waterbirds* 8:142-149. (Cited in Becker et al., 1993)

Suter, G.W, II. 1990. Endpoints for regional ecological risk assessments. *Environ. Manag.* 14(1):19-23.

Suter, G.W., II. 1993. *Ecological Risk Assessment.* Ann Arbor: Lewis Publishers.

Suzuki, M., N. Aizawa, G. Okano, and T. Takahashi. 1977. Translocation of polychlorobiphenyls in soil into plants: A study by a method of culture of soybean sprouts. *Arch. Environ. Contam.* 5:343-352.

Svensson, S. 1970. An international standard for a mapping method in bird census work. *Aud. Field Notes* 24(6):722-726.

Swartz, R.C., P.F. Kemp, D.W. Schults, and J.O. Lamberson. 1988. Effects of mixtures of sediment contaminants on the marine infaunal amphipod, *Rhepoxynius abronius*. *Environ. Toxicol. Chem.* 7:1013–1020.

Tacha, T.C., W.D. Warde, and K.P. Burnham. 1982. Use and inerpretation of statistics in wildlife journals. *Wildl. Soc. Bull.* 10:355-362.

Tejedor, M.C., M.A. Murado, and G. Baluja. 1979. Oxidative metabolism in *Saccharomyces cerevisiae* as affected by polychlorinated biphenyls. *Bull. Environ. Contam. Toxicol.* 22:439. (Cited in Hamdy and Gooch, 1986)

Thomas, P. 1988. Reproductive endocrine function in female Atlantic Croaker exposed to pollutants. *Marine Environ. Res.* 24:179–83.

Tillett, D.E., G.T. Ankley, J.P. Giesy, J.P. Ludwig, H. Kurita-Matsuba, D.V. Weseloh, P.S. Ross, C.A. Bishop, L. Sileo, K.L. Stromborg, J. Larson, and T.J. Kubiak. 1992. Polychlorinated biphenyl residues and egg mortality in double-crested cormorants from the Great Lakes. *Environ. Toxicol. Chem.* 11:1281-1288.

Tori, G.M. and T.J. Peterle. 1983. Effects of PCBs on mourning dove courtship behavior. *Bull. Environ. Contam. Toxicol.* 30:44-49. (Cited in USDOI, 1985)

Toweill, D.E. and J.E. Tabor. 1982. Chapter 36: River Otter. In: *Wild Mammals of North America.* Chapman, J.A. and G.A. Feldhamer (eds.). Baltimore: John Hopkins University Press. pp. 688-701.

Tucker, R. K. and D.G. Crabtree. 1970. *Handbook of Toxicity of Pesticides to Wildlife*. U.S. Dept. of Interior, Bureau of Sport Fisheries and Wildlife Resources Pub. No. 84. pp. 19. (Cited in Vos, 1972)

USFWS. 1980a. *Habitat as a Basis for Environmental Assessment.* Ecological Services Manual 101. U.S. Dept. of Interior, Fish and Wildlife Service. Washington D.C. (Cited in EPA, 1991)

USFWS. 1980b. *Habitat Evaluation Procedures*. Ecological Services Manual 102. U.S. Department of Interior, Fish and Wildlife Service. Washington D.C. (Cited in EPA, 1991)

USFWS. 1981. *Standards for the Development of Suitability Index Models*. Ecological Services Manual 103. U.S. Department of Interior, Fish and Wildlife Service. Washington D.C. (Cited in EPA, 1991)

USFWS. 1985a. *Polychlorinated Biphenyl Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review*. U.S. Department of Interior, Fish and Wildlife Service Biological Report 85(1.7) Contaminant Hazard Reviews Report No. 7. April.

USFWS. 1985b. *Habitat Suitability Index Models: Belted Kingfisher*. U.S. Department of the Interior, Fish and Wildlife Service, Washington, D.C. Biological Report 82(10.87). August.

USFWS. 1985c. *Habitat Suitability Index Models: Great Blue Heron*. U.S. Department of the Interior, Fish and Wildlife Service, Washington, D.C. Biological Report 82(10.99). July.

USFWS. 1986a. *Habitat Suitability Index Models: Mink*. U.S. Department of the Interior, Fish and Wildlife Service, Washington, D.C. Biological Report 82(10.127). November.

USFWS. 1986b. *Habitat Suitability Index Models: Bald Eagle (Breeding Season)*. U.S. Department of the Interior, Fish and Wildlife Service, Washington, D.C. Biological Report 82(10.126). October.

USFWS. 1987. *Habitat Suitability Index Models: Osprey*. U.S. Department of the Interior, Fish and Wildlife Service, Washington, D.C. Biological Report 82(10.154). September.

Van Velzen, W.T. and A.C. Van Velzen (eds). 1985. Forty-eighth breeding bird census. *Am. Birds* 39(1):109-114.

Van Velzen, W.T. and A.C. Van Velzen (eds). 1986. Forty-ninth breeding bird census. *Am. Birds* 40(1):69-71.

Van Velzen, W.T. and A.C. Van Velzen (eds). 1987. Fiftieth breeding bird census. *Am. Birds* 41(1):153-155.

Van Velzen, W.T. and A.C. Van Velzen (eds). 1988. Fifty-first breeding bird census. *Am. Birds* 42(1):145-148.

Veit, R.R. and W.R. Petersen. 1993. *Birds of Massachusetts*. C.W. Leahy (ed.). Lincoln: Massachusetts Audubon Society.

Vodicnik, M.J. and R.E. Peterson. 1985. The enhancing effect of spawning on elimination of a persistent polychlorinated biphenyl from female yellow perch. *Fund. Appl. Toxicol.* 5:770-776.

Von Westernhagen, H., H. Rosenthal, V. Dethlefsen, W. Ernst, U. Harms, and P.-D. Hansen. 1981. Bioaccumulating substances and reproductive success in Baltic flounder *Platichthys flesus*. *Aquatic Toxicol.* 1:85-99.

Wallace, G.J. and H.D. Mahan. 1975. *An Introduction to Ornithology*. New York: Macmillan Publishing Co., Inc.

Wallnoffer, P.R., G. Engelhardt, S.Safe, and O. Hutzinger 1973. Microbial hydroxylation of 4-chlorobiuphenyl and 4,4'-dichlorobiphenyl. *Chemosphere* 2:69. (Cited in Hamdy and Gooch, 1986)

Watson M.R., W.B. Stone, J.C. Okoniewski, and L.M. Smith. 1985. Wildlife monitors of the movement of polychlorinated biphenyls and other organochlorine compounds from a hazardous waste site. *Trans 41st Northeast Fish Wildl. Conf.* pp. 91-104.

Webber, M.D., R.I. Pietz, T.C. Granato, and M.L. Svoboda. 1994. Organic chemicals in the environment: Plant uptake of PCBs and other organic contaminants from sludge-treated coal refuse. *J. Environ. Qual.* 23:1019-1026.

Weber, J.B. and E. Mrozek. 1979. Polychlorinated biphenyls: Phytotoxicity, absorption, and translocation by plants, and inactivation by activated carbon. *Bull. Environ. Contam. Toxicol.* 23:412-417.

Weseloh, D.V., P. Mineau, and J. Struger. 1990. Geographical distribution of contaminants and productivity measures of herring gulls in the great lakes: Lake Erie and connecting channels 1978/79. *Sci. Tot. Environ.* 91:141-159.

Wiemeyer, S.N., C.M. Bunck, and A.J. Krynitsky. 1988. Organochlorine pesticides, polychlorinated biphenyls, and mercury in osprey eggs - 1970-79 - and their relationships to shell thinning and productivity. *Arch. Environ. Contam. Toxicol.* 17:767-787.

Wiemeyer, S.N., C.M. Bunck, and C.J. Stafford. 1993. Environmental contaminants in bald eagle eggs - 1980-84 - and further interpretations of relationships to productivity and shell thickness. *Arch. Environ. Contam. Toxicol.* 24:213-227.

Wilkinson, L. 1988. *Systat: The System for Statistics*. Systat, Inc. Evanston, IL.

Wren, C.D., D.B. Hunter, J.F. Leatherland, and P.M. Stokes. 1987. The effects of polychlorinated biphenyls and methylmercury, singly and in combination, on mink. I. Uptake and toxic responses. *Arch. Environ. Contam. Toxicol.* 16:441-447.

Wren, C.D. 1991. Cause-effect linkages between chemicals and populations of mink (*Mustela vison*) and otter (*Lutra canadensis*) in the Great Lakes Basin. *J. Toxicol. Environ. Health.* 33:549-585.

Zullei, N. and G. Benecke. 1978. Application of a new bioassay to screen the toxicity of polychhlorinated biphenyls on blue-green algae. *Bull. Environ. Contam. Toxicol.* 20:786-792.

Appendix A

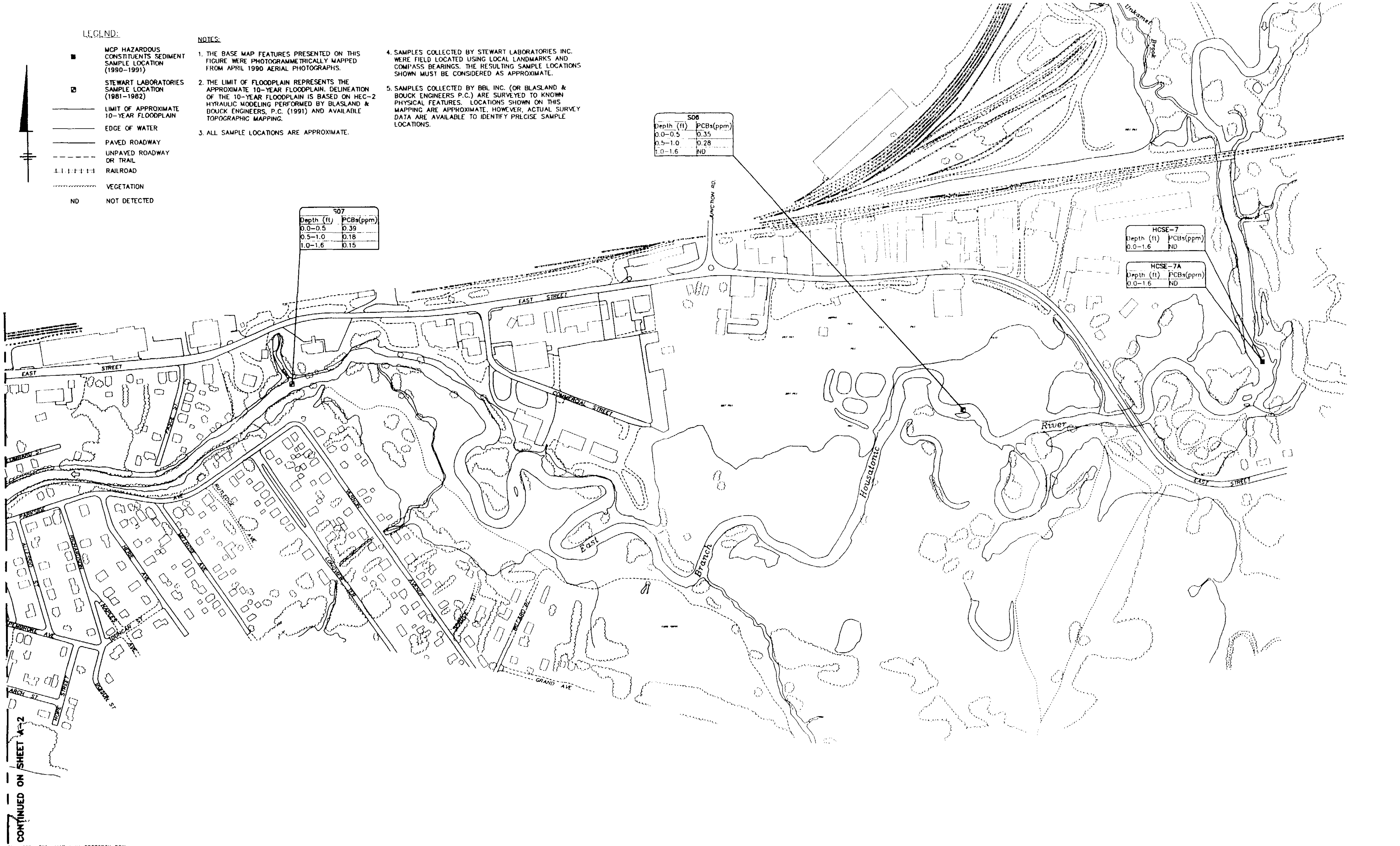
Sampling Maps

LEGEND:

- ■ MCP HAZARDOUS CONSTITUENTS SEDIMENT SAMPLE LOCATION (1990-1991)
- ⊠ STEWART LABORATORIES SAMPLE LOCATION (1981-1982)
- LIMIT OF APPROXIMATE 10-YEAR FLOODPLAIN
- EDGE OF WATER
- PAVED ROADWAY
- UNPAVED ROADWAY OR TRAIL
- RAILROAD
- VEGETATION
- ND NOT DETECTED

NOTES:

1. THE BASE MAP FEATURES PRESENTED ON THIS FIGURE WERE PHOTOGRAMMETRICALLY MAPPED FROM APRIL 1990 AERIAL PHOTOGRAPHS.
2. THE LIMIT OF FLOODPLAIN REPRESENTS THE APPROXIMATE 10-YEAR FLOODPLAIN. DELINEATION OF THE 10-YEAR FLOODPLAIN IS BASED ON HEC-2 HYRAULIC MODELING PERFORMED BY BLASLAND & BOUCK ENGINEERS, P.C. (1991) AND AVAILABLE TOPOGRAPHIC MAPPING.
3. ALL SAMPLE LOCATIONS ARE APPROXIMATE.
4. SAMPLES COLLECTED BY STEWART LABORATORIES INC. WERE FIELD LOCATED USING LOCAL LANDMARKS AND COMPASS BEARINGS. THE RESULTING SAMPLE LOCATIONS SHOWN MUST BE CONSIDERED AS APPROXIMATE.
5. SAMPLES COLLECTED BY BBL INC. (OR BLASLAND & BOUCK ENGINEERS P.C.) ARE SURVEYED TO KNOWN PHYSICAL FEATURES. LOCATIONS SHOWN ON THIS MAPPING ARE APPROXIMATE, HOWEVER, ACTUAL SURVEY DATA ARE AVAILABLE TO IDENTIFY PRECISE SAMPLE LOCATIONS.



CONTINUED ON SHEET A-2

I:ON=*,OFF=*REF*,*MAP_NUM,*PROPERTY,*ROW
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2/95 54-RLP YCC
10197418/TOPO/101ALG01.DWG

SCALE: 200' 0 200' 400'

NO ALTERATIONS PERMITTED HEREON EXCEPT AS PROVIDED UNDER SECTION 7209 SUBDIVISION 2 OF THE NEW YORK STATE EDUCATION LAW

No.	Date	Revisions	Init.

THE APPROXIMATE 10 YEAR FLOODPLAIN AS PRESENTED HAS NOT BEEN APPROVED BY FEMA FOR PURPOSES OF ASSESSING FLOOD DAMAGE RISKS. THIS INFORMATION SHOULD THEREFORE NOT BE USED FOR SUCH PURPOSES.

In charge of
Designed by
Drawn by
Checked by

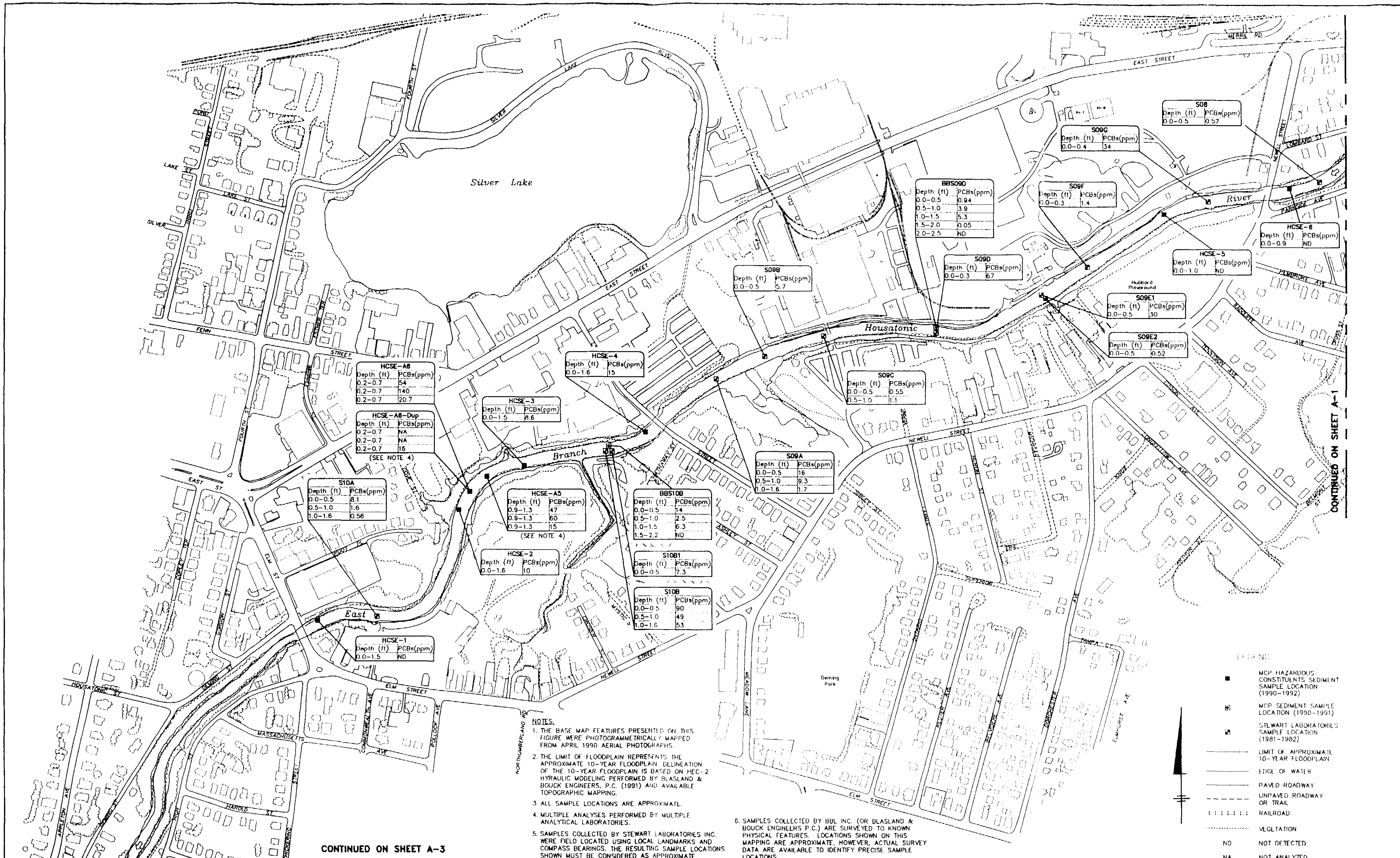
BLASLAND, BOUCK & LEE, INC.
ENGINEERS & SCIENTISTS

GENERAL ELECTRIC COMPANY • PITTSFIELD, MASSACHUSETTS

STEWART AND MCP PHASE II
SEDIMENT SAMPLING RESULTS

File Number 101.97.91
Date FEBRUARY 1995

A-1



NOTES:

1. THE BASE MAP FEATURES PRESENTED ON THIS FIGURE WERE PHOTOGRAMMETRICALLY MAPPED FROM APRIL 1990 AERIAL PHOTOGRAPHS.
2. THE LIMIT OF FLOODPLAIN REPRESENTS THE APPROXIMATE 10-YEAR FLOODPLAIN. DELINEATION OF THE 10-YEAR FLOODPLAIN IS BASED ON HEC-2 HYRAULIC MODELING PERFORMED BY BLASLAND & BOUCK ENGINEERS, P.C. (1991) AND AVAILABLE TOPOGRAPHIC MAPPING.
3. ALL SAMPLE LOCATIONS ARE APPROXIMATE.
4. MULTIPLE ANALYSES PERFORMED BY MULTIPLE ANALYTICAL LABORATORIES.
5. SAMPLES COLLECTED BY STEWART LABORATORIES INC. WERE FIELD LOCATED USING LOCAL LANDMARKS AND COMPASS BEARINGS. THE RESULTING SAMPLE LOCATIONS SHOWN MUST BE CONSIDERED AS APPROXIMATE.
6. SAMPLES COLLECTED BY BBL INC. (OR BLASLAND & BOUCK ENGINEERS P.C.) ARE SURVEYED TO KNOWN PHYSICAL FEATURES. LOCATIONS SHOWN ON THIS MAPPING ARE APPROXIMATE, HOWEVER, ACTUAL SURVEY DATA ARE AVAILABLE TO IDENTIFY PRECISE SAMPLE LOCATIONS.

LEGEND

- MCP HAZARDOUS CONSTITUENTS SEDIMENT SAMPLE LOCATION (1990–1992)
- MCP SEDIMENT SAMPLE LOCATION (1990–1991)
- STEWART LABORATORIES SAMPLE LOCATION (1981–1982)
- LIMIT OF APPROXIMATE 10-YEAR FLOODPLAIN
- EDGE OF WATER
- PAVED ROADWAY
- UNPAVED ROADWAY OR TRAIL
- RAILROAD
- VEGETATION
- ND NOT DETECTED
- NA NOT ANALYZED

1019741R/TOPO/101ALG02.DWG X:101X1X02,101ALX00 L:ON=*,OFF=*REF*,*MAP_NUM,*PROPERTY,*ROW 2/95 SA-REP YGC

SCALE: 200' 0 200' 400'

NO ALTERATIONS PERMITTED HEREON EXCEPT AS PROVIDED UNDER SECTION 7209 SUBDIVISION 2 OF THE NEW YORK STATE EDUCATION LAW

No.	Date	Revisions	Init

THE APPROXIMATE 10 YEAR FLOODPLAIN AS PRESENTED HAS NOT BEEN APPROVED BY FEMA FOR PURPOSES OF ASSESSING FLOOD DAMAGE RISKS. THIS INFORMATION SHOULD THEREFORE NOT BE USED FOR SUCH PURPOSES

In charge of ______
Designed by ______
Drawn by ______
Checked by ______

BLASLAND, BOUCK & LEE, INC.
ENGINEERS & SCIENTISTS

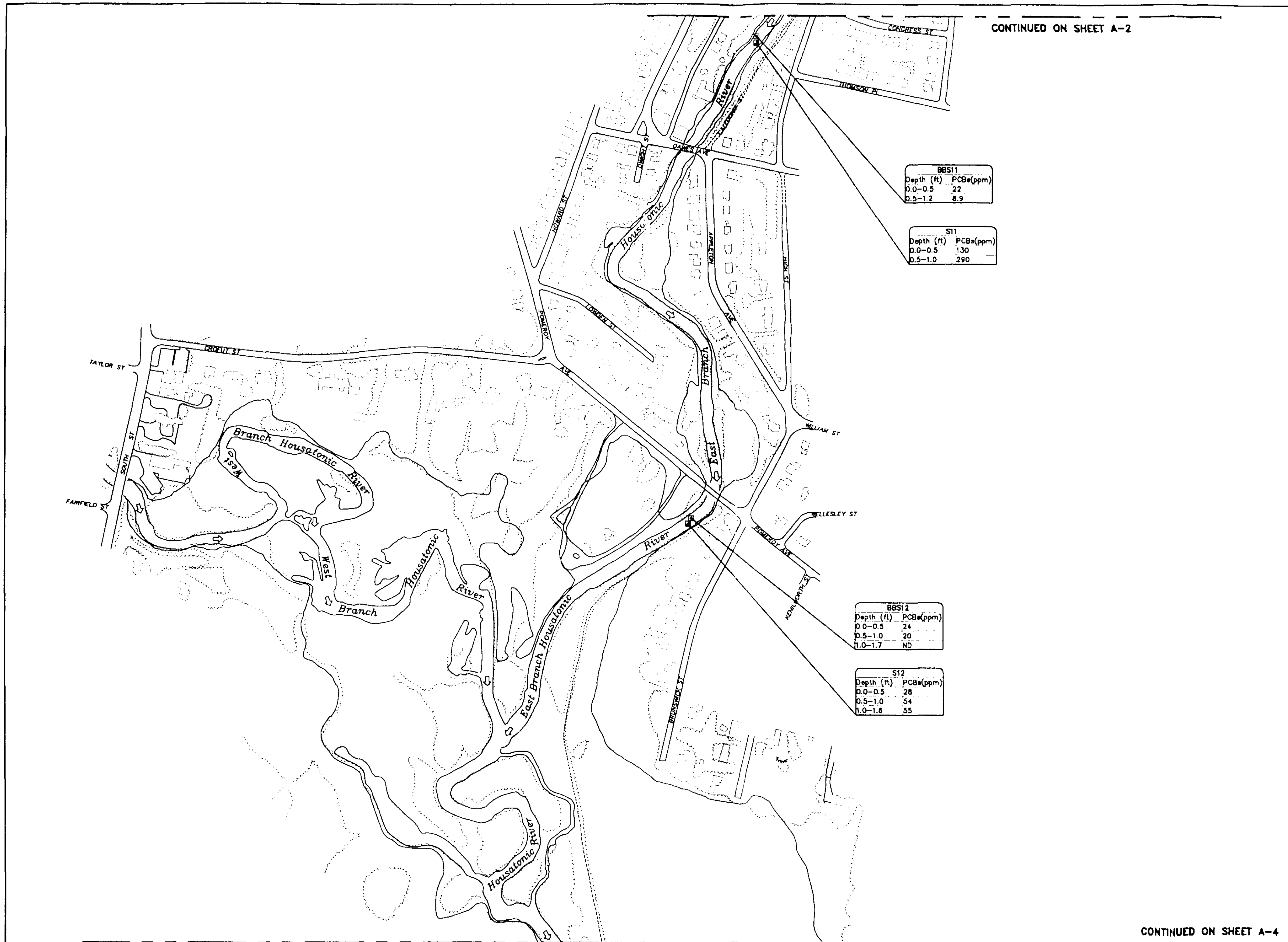
GENERAL ELECTRIC COMPANY • PITTSFIELD, MASSACHUSETTS

STEWART AND MCP PHASE II
SEDIMENT SAMPLING RESULTS

File Number 101.97.18
Date FEBRUARY 1995

A-2

CONTINUED ON SHEET A-2



CONTINUED ON SHEET A-4

Symbol	Meaning
⊡	MCP SEDIMENT SAMPLE LOCATION (1990-1991)
⊠	STEWART LABORATORIES SAMPLE LOCATION (1981-1982)
———	LIMIT OF APPROXIMATE 10-YEAR FLOODPLAIN
———	EDGE OF WATER
———	PAVED ROADWAY
- - - -	UNPAVED ROADWAY OR TRAIL
:::::::	RAILROAD
....	VEGETATION
ND	NOT DETECTED

NOTES:

1. THE BASE MAP FEATURES PRESENTED ON THIS FIGURE WERE PHOTOGRAMMETRICALLY MAPPED FROM APRIL 1990 AERIAL PHOTOGRAPHS.
2. THE LIMIT OF FLOODPLAIN REPRESENTS THE APPROXIMATE 10-YEAR FLOODPLAIN. DELINEATION OF THE 10-YEAR FLOODPLAIN IS BASED ON HEC-2 HYRAULIC MODELING PERFORMED BY BLASLAND & BOUCK ENGINEERS, P.C. (1991) AND AVAILABLE TOPOGRAPHIC MAPPING.
3. ALL SAMPLE LOCATIONS ARE APPROXIMATE.
4. SAMPLES COLLECTED BY STEWART LABORATORIES INC. WERE FIELD LOCATED USING LOCAL LANDMARKS AND COMPASS BEARINGS. THE RESULTING SAMPLE LOCATIONS SHOWN MUST BE CONSIDERED AS APPROXIMATE.
5. SAMPLES COLLECTED BY BBL INC. (OR BLASLAND & BOUCK ENGINEERS P.C.) ARE SURVEYED TO KNOWN PHYSICAL FEATURES. LOCATIONS SHOWN ON THIS MAPPING ARE APPROXIMATE. HOWEVER, ACTUAL SURVEY DATA ARE AVAILABLE TO IDENTIFY PRECISE SAMPLE LOCATIONS.

SCALE

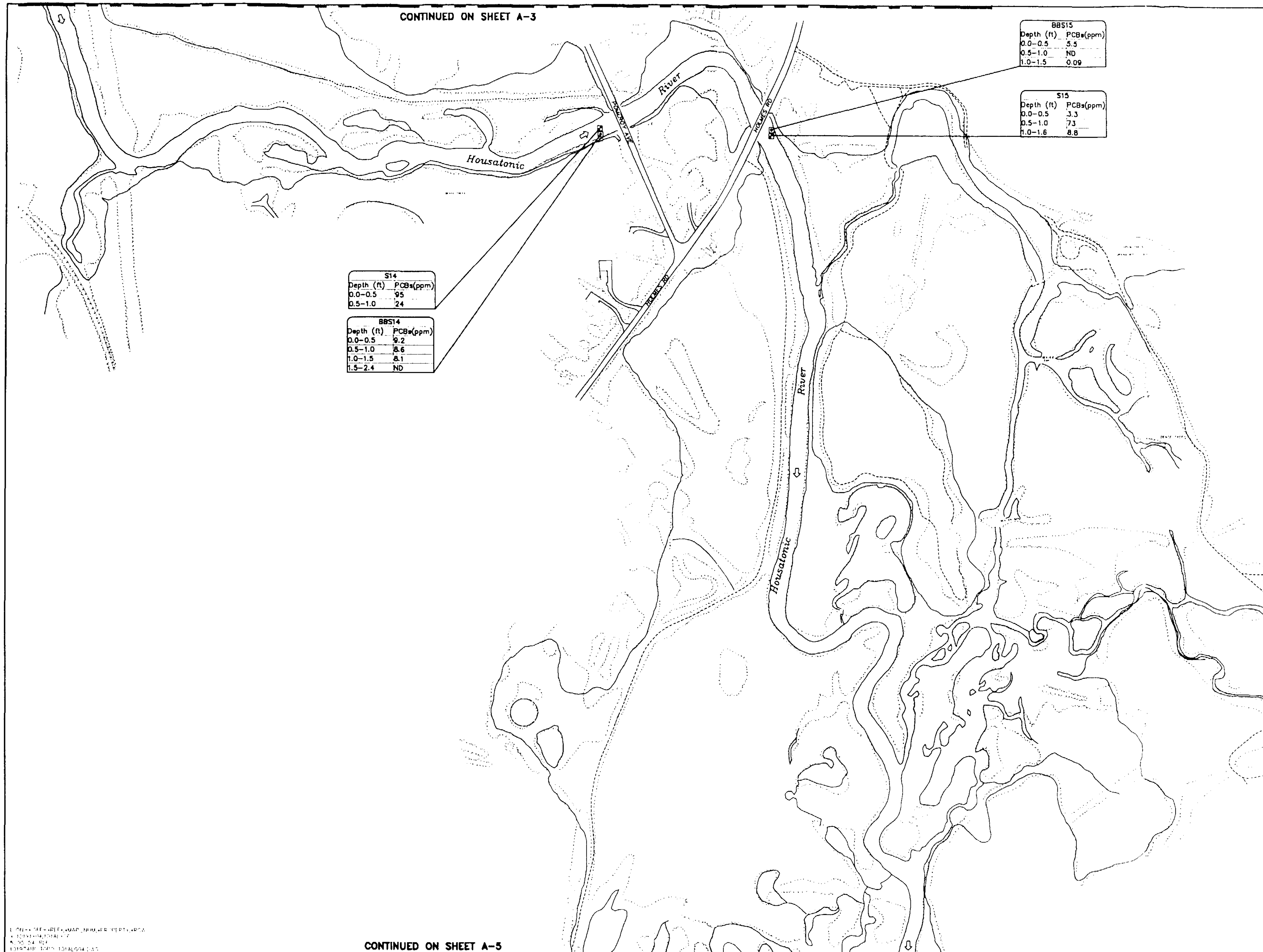
NO ALTERATIONS PERMITTED HEREIN EXCEPT AS PROVIDED UNDER SECTION 7209 SUBDIVISION 2 OF THE NEW YORK STATE EDUCATION LAW

BLASLAND, BOUCK & LEE, INC.
ENGINEERS & SCIENTISTS

STEWART AND MCP PHASE II
SEDIMENT SAMPLING RESULTS

A-3

CONTINUED ON SHEET A-3



MCP SEDIMENT SAMPLE LOCATION (1990-1991)
STEWART LABORATORIES SAMPLE LOCATION (1981-1982)
LIMIT OF APPROXIMATE 10-YEAR FLOODPLAIN
EDGE OF WATER
PAVED ROADWAY
UNPAVED ROADWAY OR TRAIL
VEGETATION
ND NOT DETECTED

NOTES:

1. THE BASE MAP FEATURES PRESENTED ON THIS FIGURE WERE PHOTOGRAMMETRICALLY MAPPED FROM APRIL 1990 AERIAL PHOTOGRAPHS.
2. THE LIMIT OF FLOODPLAIN REPRESENTS THE APPROXIMATE 10-YEAR FLOODPLAIN. DELINEATION OF THE 10-YEAR FLOODPLAIN IS BASED ON HEC-2 HYRAULIC MODELING PERFORMED BY BLASLAND & BOUCK ENGINEERS, P.C. (1991) AND AVAILABLE TOPOGRAPHIC MAPPING.
3. ALL SAMPLE LOCATIONS ARE APPROXIMATE.
4. SAMPLES COLLECTED BY STEWART LABORATORIES INC. WERE FIELD LOCATED USING LOCAL LANDMARKS AND COMPASS BEARINGS. THE RESULTING SAMPLE LOCATIONS SHOWN MUST BE CONSIDERED AS APPROXIMATE.
5. SAMPLES COLLECTED BY BBL INC. (OR BLASLAND & BOUCK ENGINEERS P.C.) ARE SURVEYED TO KNOWN PHYSICAL FEATURES. LOCATIONS SHOWN ON THIS MAPPING ARE APPROXIMATE, HOWEVER, ACTUAL SURVEY DATA ARE AVAILABLE TO IDENTIFY PRECISE SAMPLE LOCATIONS.

CONTINUED ON SHEET A-5

BLASLAND, BOUCK & LEE, INC.
ENGINEERS & SCIENTISTS

STEWART AND MCP PHASE II
SEDIMENT SAMPLING RESULTS

A-4

⊡	MCP SEDIMENT SAMPLE LOCATION (1990-1991)
⊠	STEWART LABORATORIES SAMPLE LOCATION (1981-1982)
———	LIMIT OF APPROXIMATE 10-YEAR FLOODPLAIN
———	EDGE OF WATER
———	PAVED ROADWAY
- - - -	UNPAVED ROADWAY OR TRAIL
········	VEGETATION

NOTES:

1. THE BASE MAP FEATURES PRESENTED ON THIS FIGURE WERE PHOTOGRAMMETRICALLY MAPPED FROM APRIL 1990 AERIAL PHOTOGRAPHS.
2. THE LIMIT OF FLOODPLAIN REPRESENTS THE APPROXIMATE 10-YEAR FLOODPLAIN. DELINEATION OF THE 10-YEAR FLOODPLAIN IS BASED ON HEC-2 HYRAULIC MODELING PERFORMED BY BLASLAND & BOUCK ENGINEERS, P.C. (1991) AND AVAILABLE TOPOGRAPHIC MAPPING.
3. ALL SAMPLE LOCATIONS ARE APPROXIMATE.
4. SAMPLES COLLECTED BY STEWART LABORATORIES INC. WERE FIELD LOCATED USING LOCAL LANDMARKS AND COMPASS BEARINGS. THE RESULTING SAMPLE LOCATIONS SHOWN MUST BE CONSIDERED AS APPROXIMATE.
5. SAMPLES COLLECTED BY BBL INC. (OR BLASLAND & BOUCK ENGINEERS P.C.) ARE SURVEYED TO KNOWN PHYSICAL FEATURES. LOCATIONS SHOWN ON THIS MAPPING ARE APPROXIMATE, HOWEVER, ACTUAL SURVEY DATA ARE AVAILABLE TO IDENTIFY PRECISE SAMPLE LOCATIONS.

BLASLAND, BOUCK & LEE, INC.
ENGINEERS & SCIENTISTS

STEWART AND MCP PHASE II
SEDIMENT SAMPLING RESULTS

A-5

CONTINUED ON SHEET A-5

CONTINUED ON SHEET A-7

Symbol	Description
⊠	STEWART LABORATORIES SAMPLE LOCATION (1981-1982)
———	LIMIT OF APPROXIMATE 10-YEAR FLOODPLAIN
———	EDGE OF WATER
———	PAVED ROADWAY
- - - -	UNPAVED ROADWAY OR TRAIL
:::::::	RAILROAD
·······	VEGETATION

NOTES:

1. THE BASE MAP FEATURES PRESENTED ON THIS FIGURE WERE PHOTOGRAMMETRICALLY MAPPED FROM APRIL 1990 AERIAL PHOTOGRAPHS.
2. THE LIMIT OF FLOODPLAIN REPRESENTS THE APPROXIMATE 10-YEAR FLOODPLAIN. DELINEATION OF THE 10-YEAR FLOODPLAIN IS BASED ON HEC-2 HYRAULIC MODELING PERFORMED BY BLASLAND & BOUCK ENGINEERS, P.C. (1991) AND AVAILABLE TOPOGRAPHIC MAPPING.
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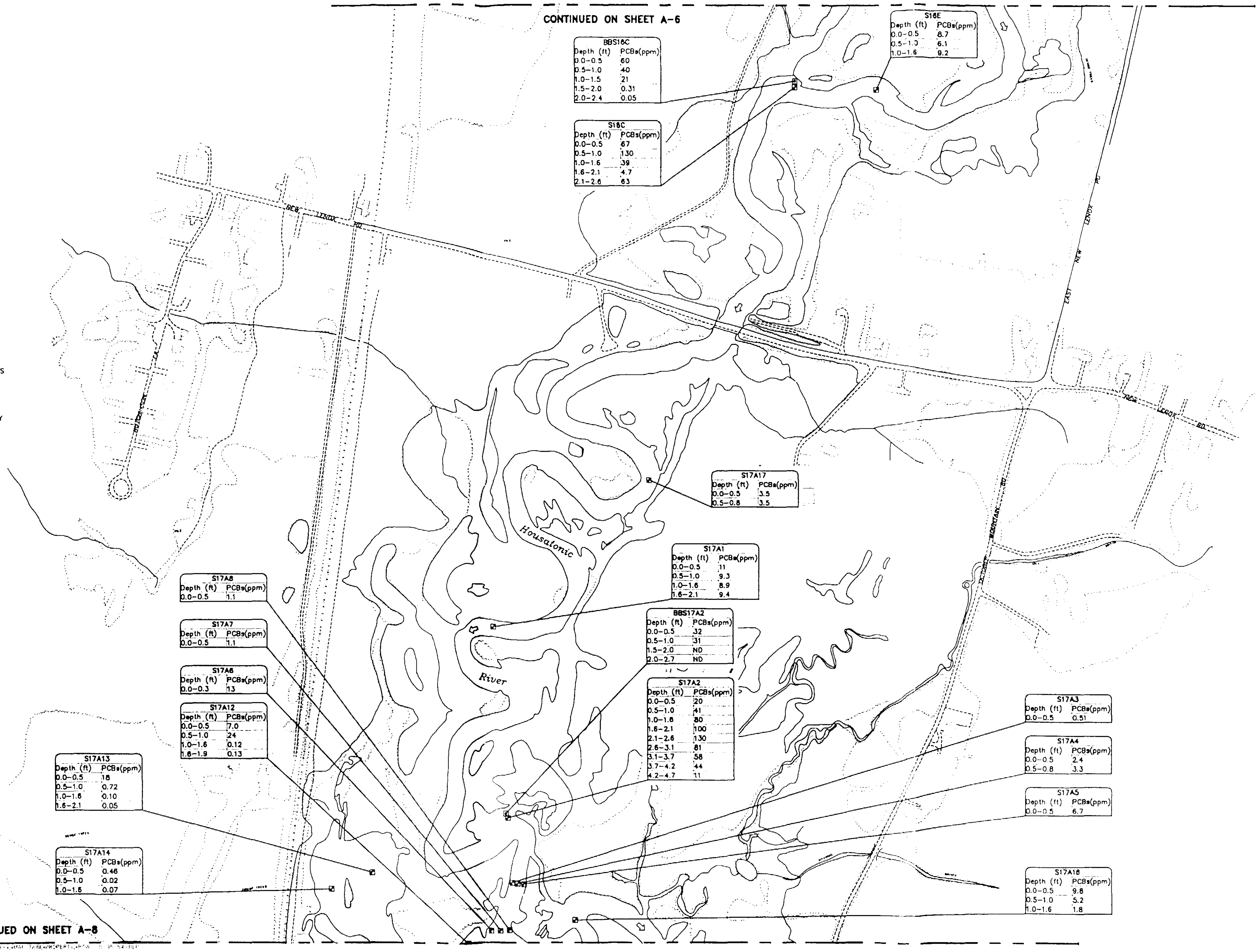
4. SAMPLES COLLECTED BY STEWART LABORATORIES INC. WERE FIELD LOCATED USING LOCAL LANDMARKS AND COMPASS BEARINGS. THE RESULTING SAMPLE LOCATIONS SHOWN MUST BE CONSIDERED AS APPROXIMATE.
5. SAMPLES COLLECTED BY BBL INC. (OR BLASLAND & BOUCK ENGINEERS P.C.) ARE SURVEYED TO KNOWN PHYSICAL FEATURES. LOCATIONS SHOWN ON THIS MAPPING ARE APPROXIMATE, HOWEVER, ACTUAL SURVEY DATA ARE AVAILABLE TO IDENTIFY PRECISE SAMPLE LOCATIONS.

BLASLAND, BOUCK & LEE, INC.
ENGINEERS & SCIENTISTS

STEWART AND MCP PHASE II
SEDIMENT SAMPLING RESULTS

A-6



LEGEND

- MCP SEDIMENT SAMPLE LOCATION (1990-1991)
- STEWART LABORATORIES SAMPLE LOCATION (1981-1982)
- LIMIT OF APPROXIMATE 10-YEAR FLOODPLAIN
- EDGE OF WATER
- PAVED ROADWAY
- UNPAVED ROADWAY OR TRAIL
- RAILROAD
- VEGETATION
- ND NOT DETECTED

NOTES:

1. THE BASE MAP FEATURES PRESENTED ON THIS FIGURE WERE PHOTOGRAMMETRICALLY MAPPED FROM APRIL 1990 AERIAL PHOTOGRAPHS (AREA NORTH OF NEW LENOX ROAD) AND FROM APRIL 1983 AERIAL PHOTOGRAPHS (AREA SOUTH OF NEW LENOX ROAD).
2. THE LIMIT OF FLOODPLAIN REPRESENTS THE APPROXIMATE 10-YEAR FLOODPLAIN. DELINEATION OF THE 10-YEAR FLOODPLAIN IS BASED ON HEC-2 HYRAULIC MODELING PERFORMED BY BLASLAND & BOUCK ENGINEERS, P.C. (1991) AND AVAILABLE TOPOGRAPHIC MAPPING.
3. ALL SAMPLE LOCATIONS ARE APPROXIMATE.
4. SAMPLES COLLECTED BY STEWART LABORATORIES INC. WERE FIELD LOCATED USING LOCAL LANDMARKS AND COMPASS BEARINGS. THE RESULTING SAMPLE LOCATIONS SHOWN MUST BE CONSIDERED AS APPROXIMATE.
5. SAMPLES COLLECTED BY BBL INC. (OR BLASLAND & BOUCK ENGINEERS P.C.) ARE SURVEYED TO KNOWN PHYSICAL FEATURES. LOCATIONS SHOWN ON THIS MAPPING ARE APPROXIMATE, HOWEVER, ACTUAL SURVEY DATA ARE AVAILABLE TO IDENTIFY PRECISE SAMPLE LOCATIONS.

BLASLAND, BOUCK & LEE, INC
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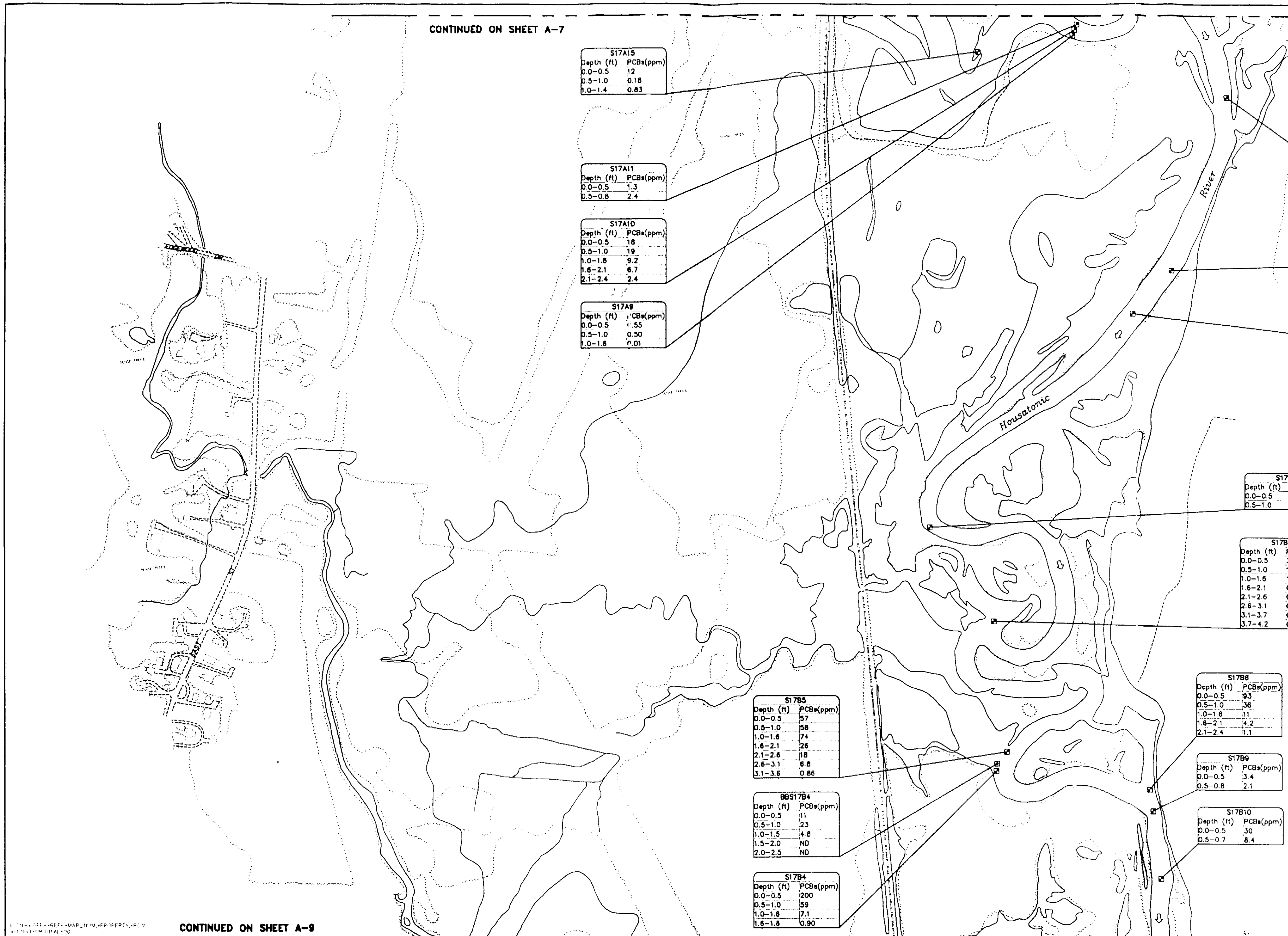
STEWART AND MCP PHASE II
SEDIMENT SAMPLING RESULTS

A-7



S17A15	
Depth (ft)	PCBs(ppm)
0.0–0.5	12
0.5–1.0	0.18
1.0–1.4	0.83

S17A11	
Depth (ft)	PCBs(ppm)
0.0–0.5	1.3
0.5–0.8	2.4

S17A10	
Depth (ft)	PCBs(ppm)
0.0–0.5	18
0.5–1.0	19
1.0–1.6	9.2
1.6–2.1	6.7
2.1–2.4	2.4

S17A9	
Depth (ft)	PCBs(ppm)
0.0–0.5	1.55
0.5–1.0	0.50
1.0–1.6	0.01

S17B1	
Depth (ft)	PCBs(ppm)
0.0–0.5	24
0.5–1.0	23
1.0–1.6	10
1.6–2.1	6.6
2.1–2.6	4.6

S17B2	
Depth (ft)	PCBs(ppm)
0.0–0.5	81
0.5–1.0	120
1.0–1.6	90
1.6–2.1	36

S17B7	
Depth (ft)	PCBs(ppm)
0.0–0.5	18
0.5–1.0	31
1.0–1.3	51

S17B8	
Depth (ft)	PCBs(ppm)
0.0–0.5	6.4
0.5–1.0	4.5

S17B3	
Depth (ft)	PCBs(ppm)
0.0–0.5	27
0.5–1.0	12
1.0–1.6	1.7
1.6–2.1	0.26
2.1–2.6	0.15
2.6–3.1	0.08
3.1–3.7	0.03
3.7–4.2	0.05

S17B6	
Depth (ft)	PCBs(ppm)
0.0–0.5	93
0.5–1.0	36
1.0–1.6	11
1.6–2.1	4.2
2.1–2.4	1.1

S17B9	
Depth (ft)	PCBs(ppm)
0.0–0.5	3.4
0.5–0.8	2.1

S17B10	
Depth (ft)	PCBs(ppm)
0.0–0.5	30
0.5–0.7	8.4

S17B5	
Depth (ft)	PCBs(ppm)
0.0–0.5	57
0.5–1.0	58
1.0–1.6	74
1.6–2.1	26
2.1–2.6	18
2.6–3.1	6.8
3.1–3.6	0.86

BBS17B4	
Depth (ft)	PCBs(ppm)
0.0–0.5	11
0.5–1.0	23
1.0–1.5	4.8
1.5–2.0	ND
2.0–2.5	ND

S17B4	
Depth (ft)	PCBs(ppm)
0.0–0.5	200
0.5–1.0	59
1.0–1.6	7.1
1.6–1.8	0.90

MCP SEDIMENT SAMPLE LOCATION (1990–1991)

STEWART LABORATORIES SAMPLE LOCATION (1981–1982)

LIMIT OF APPROXIMATE 10-YEAR FLOODPLAIN

EDGE OF WATER

PAVED ROADWAY

UNPAVED ROADWAY OR TRAIL

RAILROAD

VEGETATION

ND NOT DETECTED

NOTES:

1. THE BASE MAP FEATURES PRESENTED ON THIS FIGURE WERE PHOTOGRAMMETRICALLY MAPPED FROM APRIL 1983 AERIAL PHOTOGRAPHS.
2. THE LIMIT OF FLOODPLAIN REPRESENTS THE APPROXIMATE 10-YEAR FLOODPLAIN. DELINEATION OF THE 10-YEAR FLOODPLAIN IS BASED ON HEC-2 HYRAULIC MODELING PERFORMED BY BLASLAND & BOUCK ENGINEERS, P.C. (1991) AND AVAILABLE TOPOGRAPHIC MAPPING.
3. ALL SAMPLE LOCATIONS ARE APPROXIMATE.
4. SAMPLES COLLECTED BY STEWART LABORATORIES INC. WERE FIELD LOCATED USING LOCAL LANDMARKS AND COMPASS BEARINGS. THE RESULTING SAMPLE LOCATIONS SHOWN MUST BE CONSIDERED AS APPROXIMATE.
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BLASLAND, BOUCK & LEE, INC.
ENGINEERS & SCIENTISTS

STEWART AND MCP PHASE II
SEDIMENT SAMPLING RESULTS

A-8

CONTINUED ON SHEET 8

LEGEND

- MCP SEDIMENT SAMPLE LOCATION (1990–1991)
- STEWART LABORATORIES SAMPLE LOCATION (1981–1982)
- LIMIT OF APPROXIMATE 10-YEAR FLOODPLAIN
- EDGE OF WATER
- PAVED ROADWAY
- UNPAVED ROADWAY OR TRAIL
- RAILROAD
- VEGETATION
- ND NOT DETECTED

NOTES:

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2. THE LIMIT OF FLOODPLAIN REPRESENTS THE APPROXIMATE 10-YEAR FLOODPLAIN. DELINEATION OF THE 10-YEAR FLOODPLAIN IS BASED ON HEC-2 HYRAULIC MODELING PERFORMED BY BLASLAND & BOUCK ENGINEERS, P.C. (1991) AND AVAILABLE TOPOGRAPHIC MAPPING.
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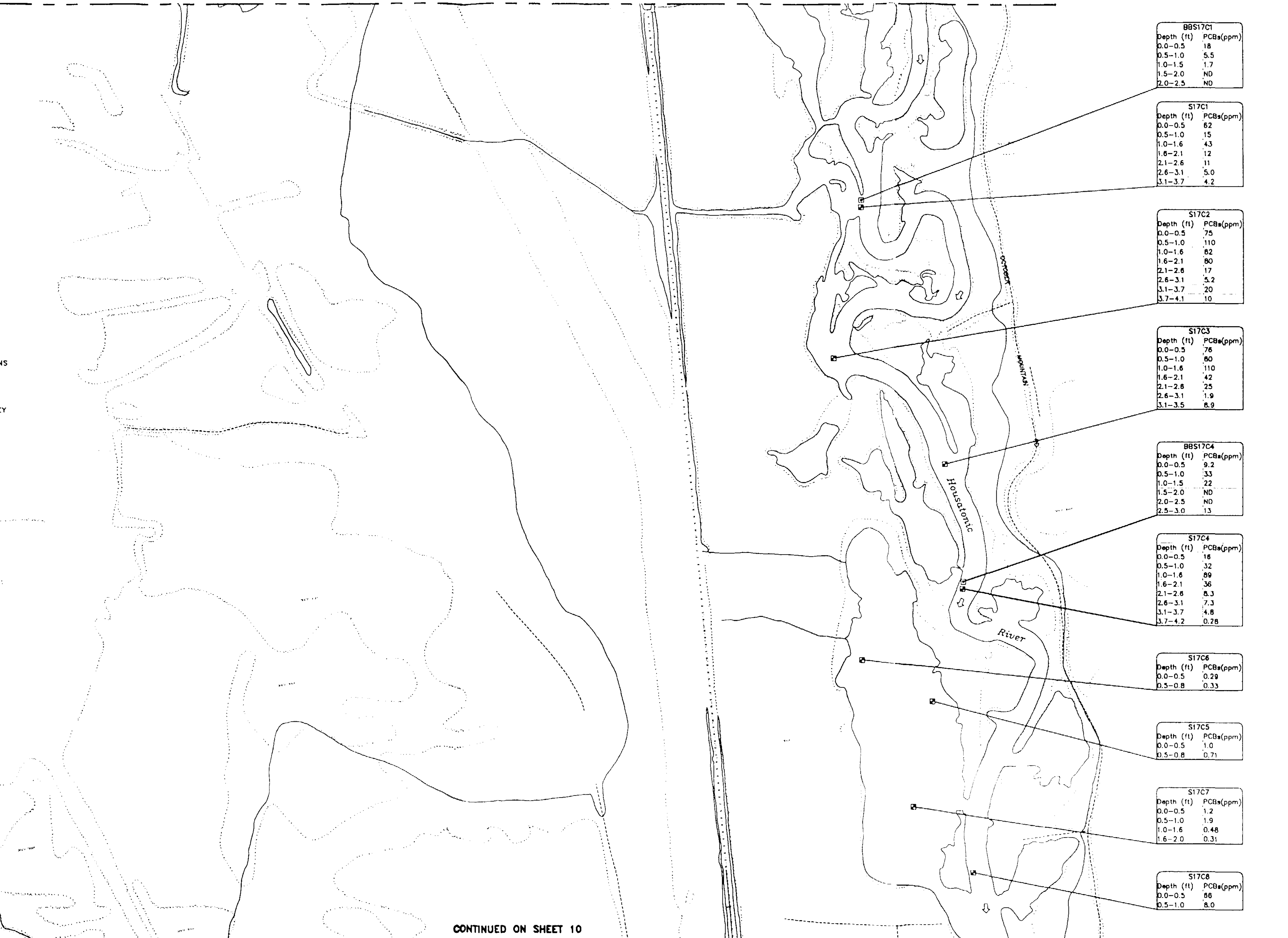


BBS17C1

Depth (ft)	PCBs(ppm)
0.0-0.5	18
0.5-1.0	5.5
1.0-1.5	1.7
1.5-2.0	ND
2.0-2.5	ND

S17C1

Depth (ft)	PCBs(ppm)
0.0-0.5	62
0.5-1.0	15
1.0-1.6	43
1.6-2.1	12
2.1-2.6	11
2.6-3.1	5.0
3.1-3.7	4.2

S17C2

Depth (ft)	PCBs(ppm)
0.0-0.5	75
0.5-1.0	110
1.0-1.6	82
1.6-2.1	80
2.1-2.6	17
2.6-3.1	5.2
3.1-3.7	20
3.7-4.1	10

S17C3

Depth (ft)	PCBs(ppm)
0.0-0.5	76
0.5-1.0	80
1.0-1.6	110
1.6-2.1	42
2.1-2.6	25
2.6-3.1	1.9
3.1-3.5	8.9

BBS17C4

Depth (ft)	PCBs(ppm)
0.0-0.5	9.2
0.5-1.0	33
1.0-1.5	22
1.5-2.0	ND
2.0-2.5	ND
2.5-3.0	13

S17C4

Depth (ft)	PCBs(ppm)
0.0-0.5	16
0.5-1.0	32
1.0-1.6	89
1.6-2.1	36
2.1-2.6	8.3
2.6-3.1	7.3
3.1-3.7	4.8
3.7-4.2	0.78

S17C6

Depth (ft)	PCBs(ppm)
0.0-0.5	0.29
0.5-0.8	0.33

S17C5

Depth (ft)	PCBs(ppm)
0.0-0.5	1.0
0.5-0.8	0.71

S17C7

Depth (ft)	PCBs(ppm)
0.0-0.5	1.2
0.5-1.0	1.9
1.0-1.6	0.48
1.6-2.0	0.31

S17C8

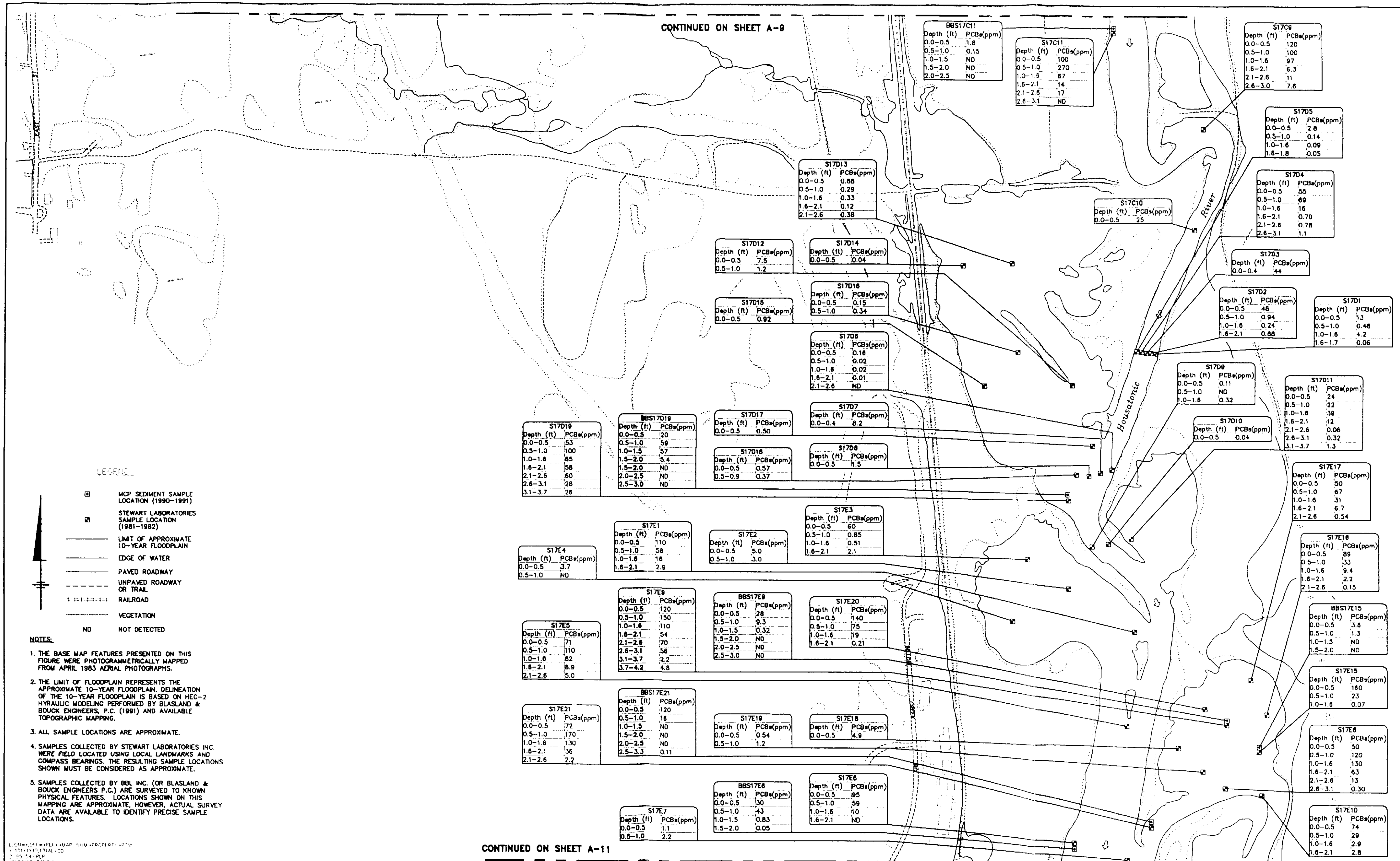
Depth (ft)	PCBs(ppm)
0.0-0.5	86
0.5-1.0	8.0

CONTINUED ON SHEET 10

BLASLAND, BOUCK & LEE, INC
ENGINEERS & SCIENTISTS

STEWART AND MCP PHASE II
SEDIMENT SAMPLING RESULTS

A-9



BLASLAND, BOUCK & LEE, INC
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STEWART AND MCP PHASE II
SEDIMENT SAMPLING RESULTS

A-10

LEGEND

⊡	MCP SEDIMENT SAMPLE LOCATION (1990–1991)
◪	STEWART LABORATORIES SAMPLE LOCATION (1981–1982)
———	LIMIT OF APPROXIMATE 10–YEAR FLOODPLAIN
———	EDGE OF WATER
———	PAVED ROADWAY
– – – –	UNPAVED ROADWAY OR TRAIL
‡‡‡‡	RAILROAD
········	VEGETATION
ND	NOT DETECTED

NOTES:

1. THE BASE MAP FEATURES PRESENTED ON THIS FIGURE WERE PHOTOGRAMMETRICALLY MAPPED FROM APRIL 1983 AERIAL PHOTOGRAPHS WITH THE EXCEPTION OF THE WOODS POND DAM WHICH WAS ADDED FROM ENGINEERING DRAWINGS.
2. THE LIMIT OF FLOODPLAIN REPRESENTS THE APPROXIMATE 10–YEAR FLOODPLAIN. DELINEATION OF THE 10–YEAR FLOODPLAIN IS BASED ON HEC–2 HYRAULIC MODELING PERFORMED BY BLASLAND & BOUCK ENGINEERS, P.C. (1991) AND AVAILABLE TOPOGRAPHIC MAPPING.
3. ALL SAMPLE LOCATIONS ARE APPROXIMATE.
4. SAMPLES COLLECTED BY STEWART LABORATORIES INC. WERE FIELD LOCATED USING LOCAL LANDMARKS AND COMPASS BEARINGS. THE RESULTING SAMPLE LOCATIONS SHOWN MUST BE CONSIDERED AS APPROXIMATE.
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CONTINUED ON SHEET A–10

S17E11	
Depth (ft)	PCBs(ppm)
0.0–0.5	120
0.5–1.0	86
1.0–1.6	64
1.6–2.1	24
2.1–2.6	5.8
2.6–3.1	2.7
3.1–3.7	5.7

S17E12	
Depth (ft)	PCBs(ppm)
0.0–0.5	44
0.5–1.0	5.2
1.0–1.6	0.20
1.6–2.1	0.06

S17E13	
Depth (ft)	PCBs(ppm)
0.0–0.5	94
0.5–1.0	160
1.0–1.6	42
1.6–2.1	2.3
2.1–2.6	14
2.6–3.1	0.32

S17E14	
Depth (ft)	PCBs(ppm)
0.0–0.5	59
0.5–1.0	56
1.0–1.6	5.8
1.6–2.1	2.3
2.1–2.6	1.7
2.6–3.1	1.0

S18A1	
Depth (ft)	PCBs(ppm)
0.0–0.5	150
0.5–1.0	140
1.0–1.6	63
1.6–1.8	6.1

BBS18A1	
Depth (ft)	PCBs(ppm)
0.0–0.5	41
0.5–1.0	4.4
1.0–1.5	ND
1.5–2.1	ND

S18A2	
Depth (ft)	PCBs(ppm)
0.0–0.5	62

BBS18A2	
Depth (ft)	PCBs(ppm)
0.0–0.5	13
0.5–1.0	0.07
1.0–1.4	ND

S18B1	
Depth (ft)	PCBs(ppm)
0.0–0.5	130
0.5–1.0	40
1.0–1.6	3.2
1.6–2.1	0.54
2.1–2.6	0.32
2.6–3.1	0.03
3.1–3.7	0.04

S18B2	
Depth (ft)	PCBs(ppm)
0.0–0.5	54
0.5–1.0	3.7
1.0–1.6	0.10
1.6–2.1	ND

BBS18B2	
Depth (ft)	PCBs(ppm)
0.0–0.5	17
0.5–1.0	0.11
1.0–1.5	ND
1.5–2.1	ND

S18B3	
Depth (ft)	PCBs(ppm)
0.0–0.5	0.83
0.5–1.0	0.05
1.0–1.6	0.47
1.6–2.1	0.53
2.1–2.6	0.09
2.6–3.1	0.76
3.1–3.7	0.02
3.7–4.2	0.10
4.2–4.7	ND

S18C1	
Depth (ft)	PCBs(ppm)
0.0–0.5	0.72
0.5–1.0	0.09
1.0–1.6	2.1
1.6–2.1	1.7
2.1–2.6	3.6
2.6–3.1	3.3
3.1–3.7	2.4
3.7–4.2	0.32
4.2–4.7	0.36
4.7–4.9	0.09

S18C2	
Depth (ft)	PCBs(ppm)
0.0–0.5	5.8
0.5–1.0	2.9
1.0–1.6	0.32
1.6–2.1	1.2
2.1–2.6	0.81
2.6–3.1	0.16
3.1–3.7	1.1

S18D1	
Depth (ft)	PCBs(ppm)
0.0–0.5	89
0.5–1.0	19
1.0–1.6	7.9
1.6–2.1	23
2.1–2.6	0.57
2.6–3.1	1.5
3.1–3.7	17
3.7–4.2	2.4
4.2–4.7	0.96
4.7–5.2	0.86
5.2–5.8	0.25
5.8–6.3	0.27
6.3–6.8	0.36
6.8–7.3	0.59
7.3–7.9	0.17
7.9–8.4	0.14

S18D2	
Depth (ft)	PCBs(ppm)
0.0–0.5	39
0.5–1.0	0.52
1.6–2.1	0.33
2.1–2.6	0.21

S18D3	
Depth (ft)	PCBs(ppm)
0.0–0.5	3.2
0.5–1.0	0.39
1.0–1.6	0.73
1.6–2.1	1.4
2.1–2.6	0.24

S18E1	
Depth (ft)	PCBs(ppm)
0.0–0.5	3.7
0.5–1.0	1.7
1.0–1.6	0.27
1.6–2.1	ND

S18E2	
Depth (ft)	PCBs(ppm)
0.0–0.5	7.3
0.5–1.0	0.84
1.0–1.6	0.15
1.6–2.1	0.21
2.1–2.6	0.30
2.6–3.1	0.21

S18E3	
Depth (ft)	PCBs(ppm)
0.0–0.5	9.5
0.5–1.0	1.6
1.0–1.6	0.69
1.6–2.1	0.52
2.1–2.6	0.25
2.6–3.1	0.40
3.1–3.7	ND
3.7–4.2	0.35
4.2–4.7	1.6
4.7–5.2	0.62
5.2–5.8	1.4
5.8–6.3	0.84

S18F1	
Depth (ft)	PCBs(ppm)
0.0–0.5	190
0.5–1.0	99
1.0–1.6	160
1.6–2.1	71
2.1–2.6	14
2.6–3.1	4.2
3.1–3.7	2.0
3.7–4.2	1.3
4.2–4.7	1.2
4.7–5.2	0.41
5.2–5.8	2.0
5.8–6.0	0.17

BBS18F1	
Depth (ft)	PCBs(ppm)
0.0–0.5	21
0.5–1.0	110
1.0–1.5	7.2
1.5–2.0	2.5
2.0–2.5	1.5

S18F2	
Depth (ft)	PCBs(ppm)
0.0–0.5	19
0.5–1.0	4.5
1.0–1.6	1.8
1.6–2.1	0.05

S18F3	
Depth (ft)	PCBs(ppm)
0.0–0.5	21
0.5–1.0	1.3
1.0–1.6	1.4
1.6–2.1	0.97
2.1–2.6	0.36

S18G1	
Depth (ft)	PCBs(ppm)
0.0–0.5	0.08
0.5–1.0	0.08
1.0–1.6	0.18

S18G2	
Depth (ft)	PCBs(ppm)
0.0–0.5	120
0.5–1.0	70
1.0–1.6	200
1.6–2.1	210
2.1–2.6	220
2.6–3.1	42
3.1–3.7	8.4
3.7–4.2	4.0
4.2–4.7	3.9
4.7–5.2	7.2
5.2–5.6	1.2

BBS18G2	
Depth (ft)	PCBs(ppm)
0.0–0.5	80
0.5–1.0	8.8
1.0–1.5	ND
1.5–2.0	ND
2.0–2.3	0.07

S18H1	
Depth (ft)	PCBs(ppm)
0.0–0.5	140
0.5–1.0	5.2
1.0–1.6	1.1
1.6–2.1	0.26

S18H2	
Depth (ft)	PCBs(ppm)
0.0–0.5	26
0.5–1.0	13
1.0–1.6	3.2
1.6–2.1	0.11
2.1–2.6	2.1
2.6–3.1	3.7
3.1–3.7	0.90
3.7–4.2	0.09
4.2–4.7	1.6
4.7–5.2	2.9
5.2–5.8	0.32
5.8–6.3	0.13
6.3–6.8	0.32

S18H3	
Depth (ft)	PCBs(ppm)
0.0–0.5	130
0.5–1.0	140
1.0–1.6	99
1.6–2.1	2.5
2.1–2.6	0.69
2.6–3.1	1.1
3.1–3.7	3.0
3.7–4.2	1.1
4.2–4.7	5.5
4.7–5.2	8.6
5.2–5.8	0.09
5.8–6.3	15
6.3–6.8	0.11
6.8–7.3	0.28
7.3–7.9	0.38
7.9–8.4	0.23
8.4–8.9	4.4
8.9–9.4	0.30
9.4–10	0.53

BBS18H3	
Depth (ft)	PCBs(ppm)
0.0–0.5	28
0.5–1.0	78
1.0–1.5	120
1.5–2.0	5.1
2.0–2.5	10

S18I1	
Depth (ft)	PCBs(ppm)
0.0–0.5	160
0.5–1.0	160
1.0–1.6	19
1.6–2.1	0.22
2.1–2.6	0.28
2.6–3.1	1.2
3.1–3.7	0.21
3.7–4.2	3.0
4.2–4.7	1.3
4.7–5.2	0.44
5.2–5.8	0.04
5.8–6.3	2.3
6.3–6.8	1.2

BBS18I1	
Depth (ft)	PCBs(ppm)
0.0–0.5	35
0.5–1.0	64
1.0–1.5	100
1.5–2.0	3.0
2.0–2.5	3.6

S18I2	
Depth (ft)	PCBs(ppm)
0.0–0.5	120
0.5–1.0	150
1.0–1.6	39
1.6–2.1	3.3
2.1–2.6	1.1
2.6–3.1	8.9
3.1–3.7	14
3.7–4.2	5.4
4.2–4.7	7.0
4.7–5.2	7.4
5.2–5.8	ND

BBS18I2	
Depth (ft)	PCBs(ppm)
0.0–0.5	52
0.5–1.0	120
1.0–1.5	180
1.5–2.0	20
2.0–2.5	8.2

S18I3	
Depth (ft)	PCBs(ppm)
0.0–0.5	170
0.5–1.0	110
1.0–1.6	4.8
1.6–2.1	0.65

BBS18I3	
Depth (ft)	PCBs(ppm)
0.0–0.5	1.5
0.5–1.0	ND
1.0–1.5	ND
1.5–2.0	ND

S18J1	
Depth (ft)	PCBs(ppm)
0.0–0.5	13
0.5–1.0	3.0
1.0–1.6	0.84
1.6–2.1	0.07

S18J2	
Depth (ft)	PCBs(ppm)
0.0–0.5	150
0.5–1.0	24
1.0–1.6	58
1.6–2.1	28
2.1–2.6	1.9

BBS18J2	
Depth (ft)	PCBs(ppm)
0.0–0.5	31
0.5–1.0	0.55
1.0–1.5	ND
1.5–2.0	ND

S18K1	
Depth (ft)	PCBs(ppm)
0.0–0.5	130
0.5–1.0	42
1.0–1.6	0.09
1.6–2.0	1.6

S18K2	
Depth (ft)	PCBs(ppm)
0.0–0.5	0.18
0.5–1.0	ND
1.0–1.3	0.01

S18K3	
Depth (ft)	PCBs(ppm)
0.0–0.5	94
0.5–1.0	4.1
1.0–1.6	1.5
1.6–2.1	0.08
2.1–2.6	0.09

S18L1	
Depth (ft)	PCBs(ppm)
0.0–0.5	27
0.5–1.0	4.0

S18L2	
Depth (ft)	PCBs(ppm)
0.0–0.5	9.7
0.5–1.0	0.06
1.0–1.3	1.3

S18M1	
Depth (ft)	PCBs(ppm)
0.0–0.5	41
0.5–1.0	1.5
1.0–1.6	0.06
1.6–2.1	0.13
2.1–2.6	0.66
2.6–3.1	0.27
3.1–3.7	ND
3.7–4.1	0.02

S18M2	
Depth (ft)	PCBs(ppm)
0.0–0.5	59
0.5–1.0	6.5
1.0–1.5	0.93

S18M3	
Depth (ft)	PCBs(ppm)
0.0–0.5	120
0.5–1.0	99
1.0–1.6	2.5
1.6–2.1	1.6
2.1–2.5	2.6

BBS18M3	
Depth (ft)	PCBs(ppm)
0.0–0.5	20
0.5–1.0	1.4
1.0–1.5	ND

S18P1	
Depth (ft)	PCBs(ppm)
0.0–0.3	2.1

S18P2	
Depth (ft)	PCBs(ppm)
0.0–0.5	120
0.5–1.0	80
1.0–1.2	31

BBS18P2	
Depth (ft)	PCBs(ppm)
0.0–0.5	22
0.5–1.0	50
1.0–1.5	43
1.5–2.0	30
2.0–2.5	67

S18P3	
Depth (ft)	PCBs(ppm)
0.0–0.2	7.4

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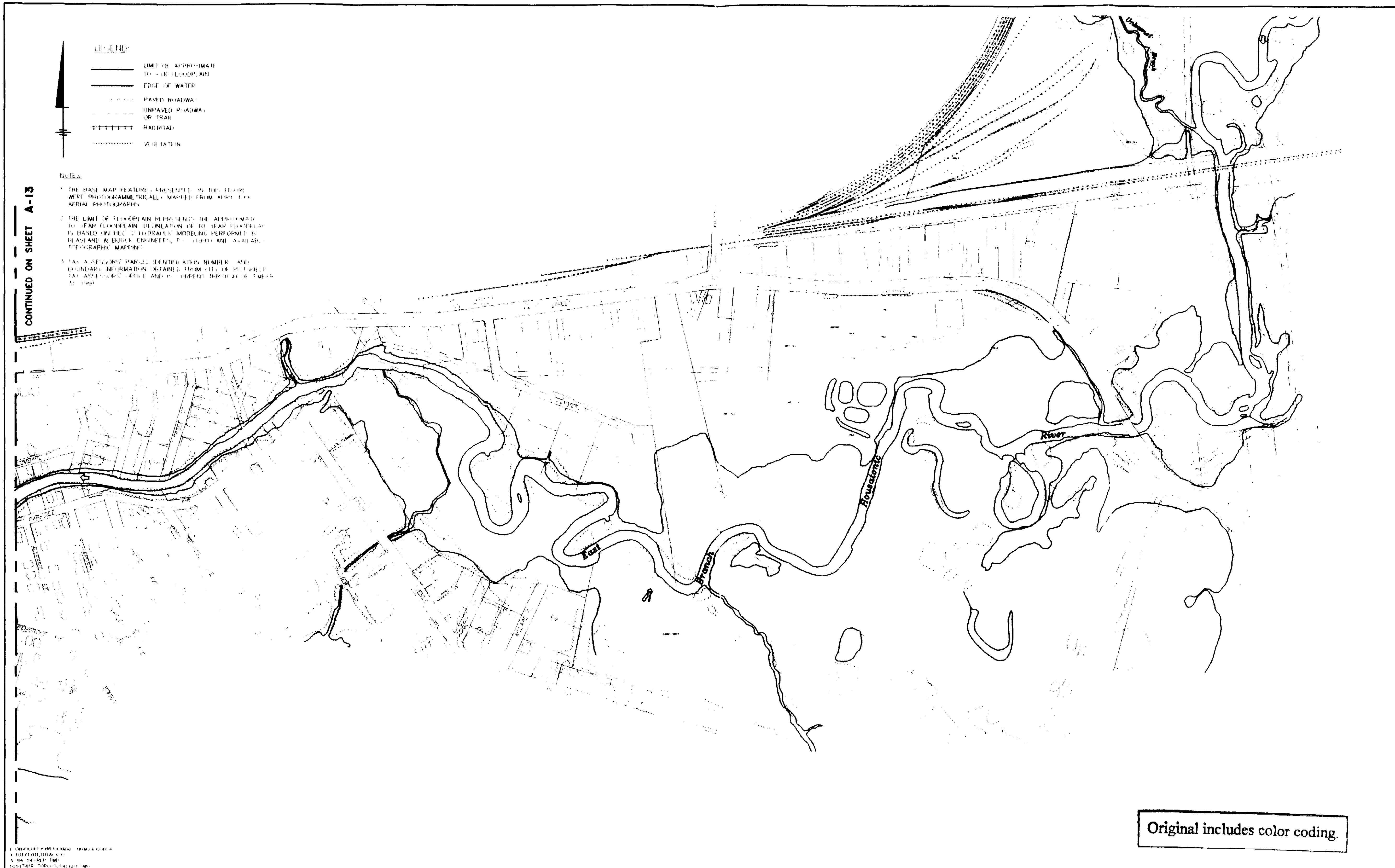
ENGINEERS & SCIENTISTS

STEWART AND MCP PHASE II
SEDIMENT SAMPLING RESULTS

A-11



CONTINUED ON SHEET A-13

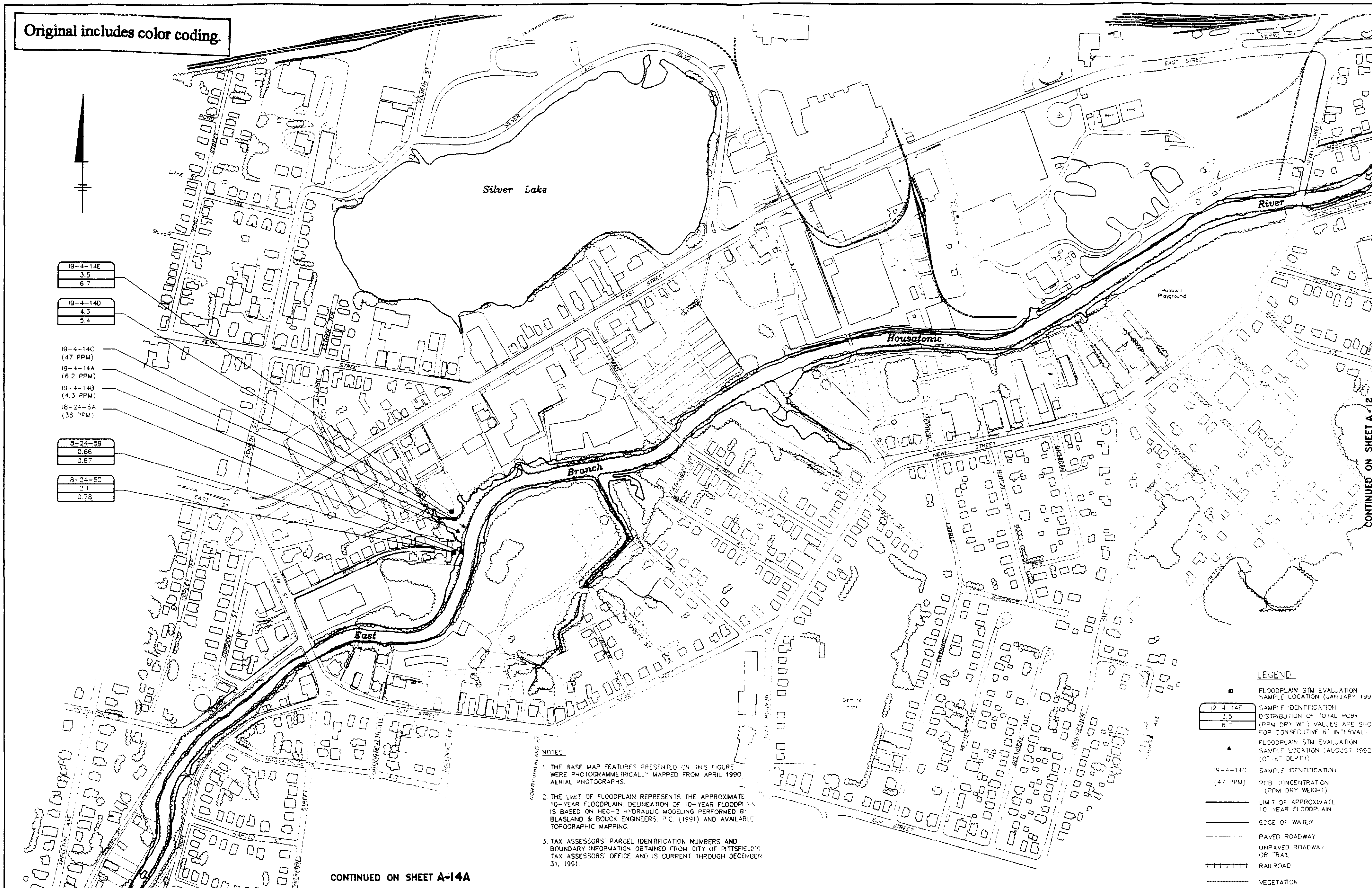
BLASLAND, BOUCK & LEE, INC

MCP PHASE II AND STM EVALUATION

FLOODPLAIN SAMPLING RESULTS

A-12

Original includes color coding.



LEGEND:

- FLOODPLAIN STM EVALUATION SAMPLE LOCATION (JANUARY 1993)
- 19-4-14E / 3.5 / 6.7 — SAMPLE IDENTIFICATION DISTRIBUTION OF TOTAL PCBs (PPM DRY WT.) VALUES ARE SHOWN FOR CONSECUTIVE 6" INTERVALS
- FLOODPLAIN STM EVALUATION SAMPLE LOCATION (AUGUST 1992) (0"-6" DEPTH)
- 19-4-14C — SAMPLE IDENTIFICATION
- (47 PPM) — PCB CONCENTRATION -(PPM DRY WEIGHT)
- LIMIT OF APPROXIMATE 10-YEAR FLOODPLAIN
- EDGE OF WATER
- PAVED ROADWAY
- UNPAVED ROADWAY OR TRAIL
- RAILROAD
- VEGETATION

NOTES:

1. THE BASE MAP FEATURES PRESENTED ON THIS FIGURE WERE PHOTOGRAMMETRICALLY MAPPED FROM APRIL 1990 AERIAL PHOTOGRAPHS.
2. THE LIMIT OF FLOODPLAIN REPRESENTS THE APPROXIMATE 10-YEAR FLOODPLAIN. DELINEATION OF 10-YEAR FLOODPLAIN IS BASED ON HEC-2 HYDRAULIC MODELING PERFORMED BY BLASLAND & BOUCK ENGINEERS, P.C. (1991) AND AVAILABLE TOPOGRAPHIC MAPPING.
3. TAX ASSESSORS' PARCEL IDENTIFICATION NUMBERS AND BOUNDARY INFORMATION OBTAINED FROM CITY OF PITTSFIELD'S TAX ASSESSORS' OFFICE AND IS CURRENT THROUGH DECEMBER 31, 1991.

1019741R/TOPO/101ACG02.DWG L:ON=*,OFF=*REF*,*MAP_NUM,GE*,OBC* X:101X1X02,101ACX00 3/94 54-RLP TWO

SCALE: 200' 0 200' 400'

NO ALTERATIONS PERMITTED HEREON EXCEPT AS PROVIDED UNDER SECTION 7209 SUBDIVISION 2 OF THE NEW YORK STATE EDUCATION LAW

No.	Date	Revisions	Init.

THE APPROXIMATE 10 YEAR FLOODPLAIN AS PRESENTED HAS NOT BEEN APPROVED BY FEMA FOR PURPOSES OF ASSESSING FLOOD DAMAGE RISKS. THIS INFORMATION SHOULD THEREFORE NOT BE USED FOR SUCH PURPOSES.

In charge of ________
Designed by ________
Drawn by ________
Checked by ________

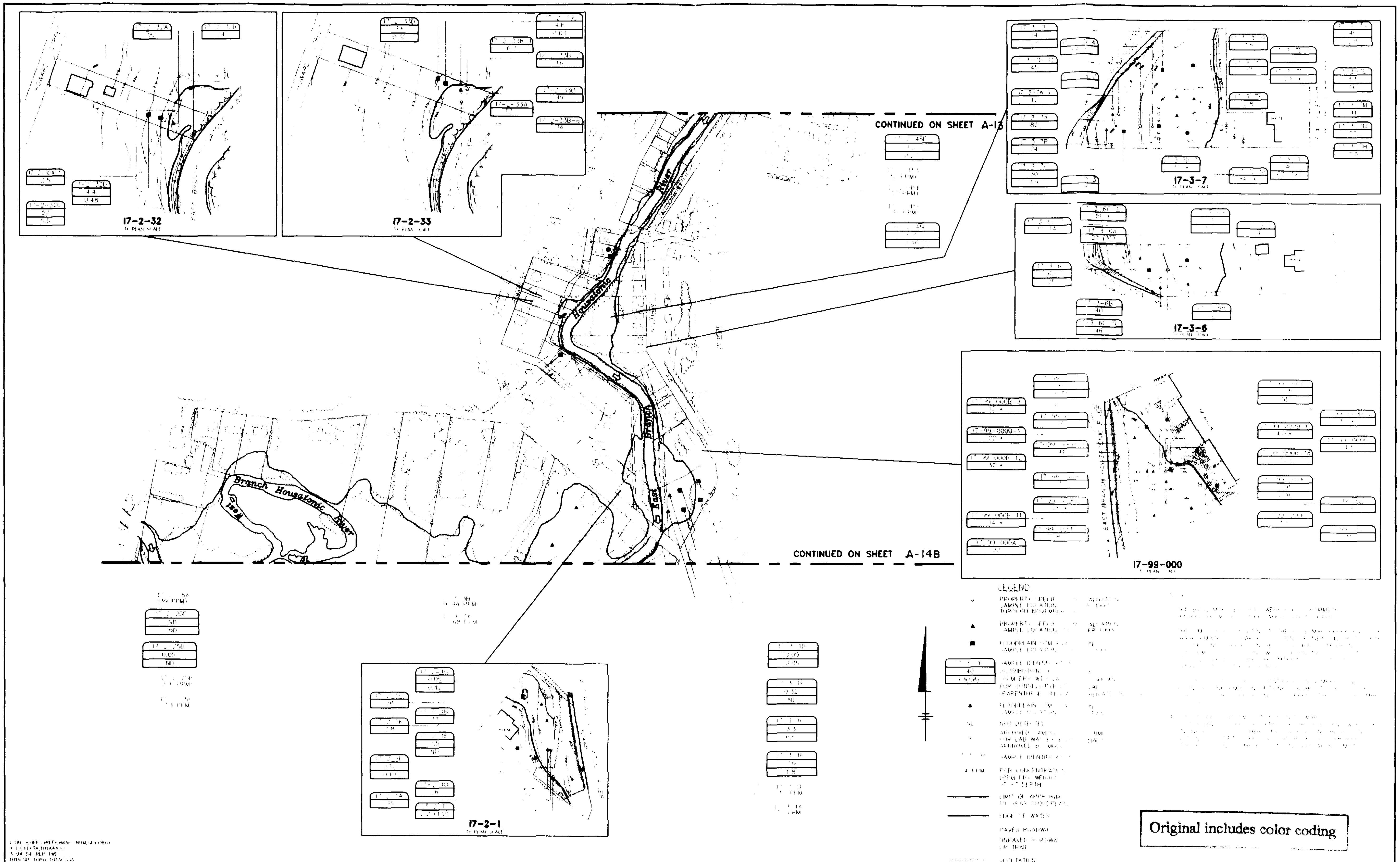
BLASLAND, BOUCK & LEE, INC.
ENGINEERS & SCIENTISTS

GENERAL ELECTRIC COMPANY • PITTSFIELD, MASSACHUSETTS

MCP PHASE II AND STM EVALUATION
FLOODPLAIN SAMPLING RESULTS

File Number 101.97.41
Date MARCH 1994

A-13



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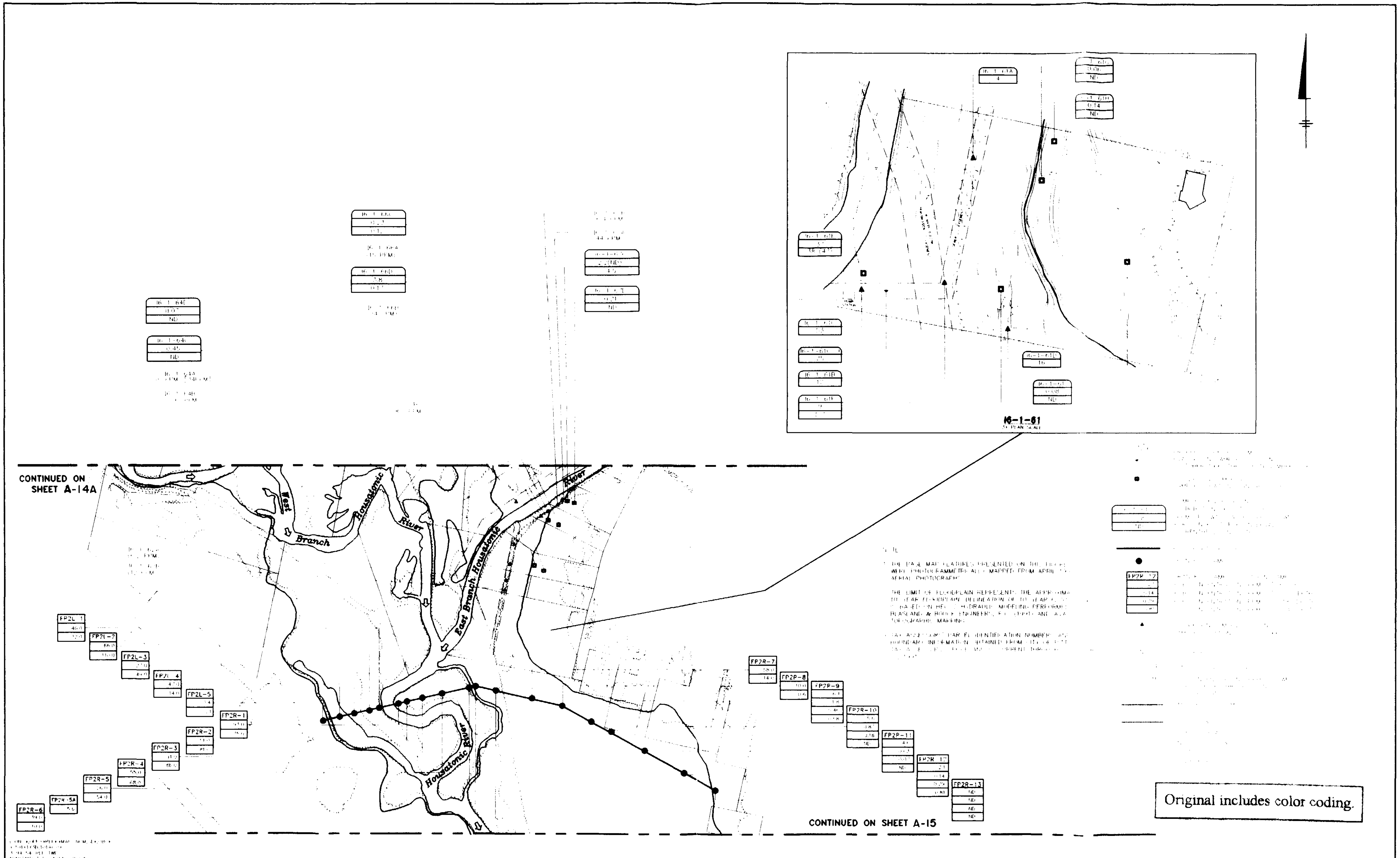
MCP PHASE II AND STM EVALUATION

FLOODPLAIN SAMPLING RESULTS

A-14A



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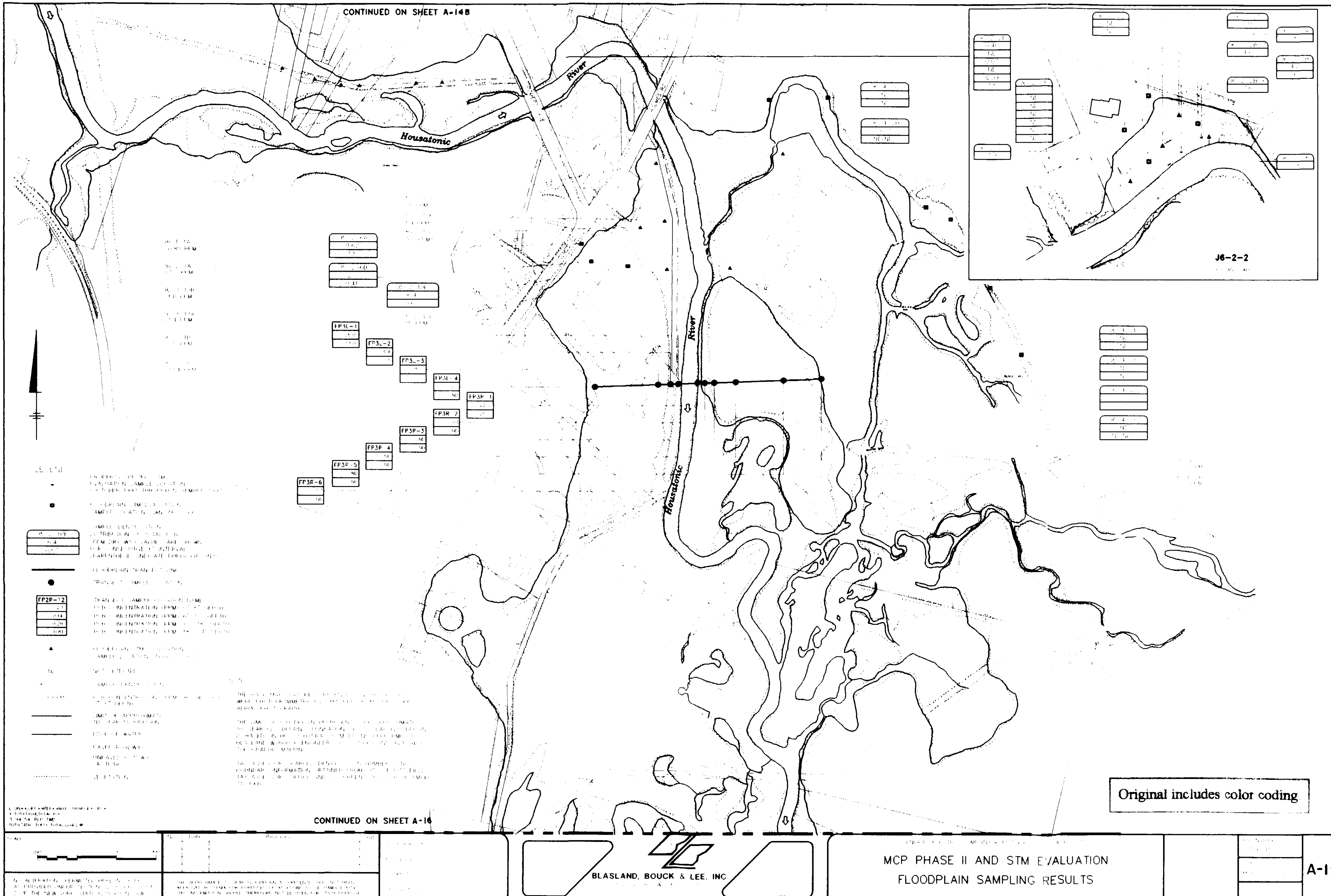
MCP PHASE II AND STM EVALUATION
FLOODPLAIN SAMPLING RESULTS

A-14B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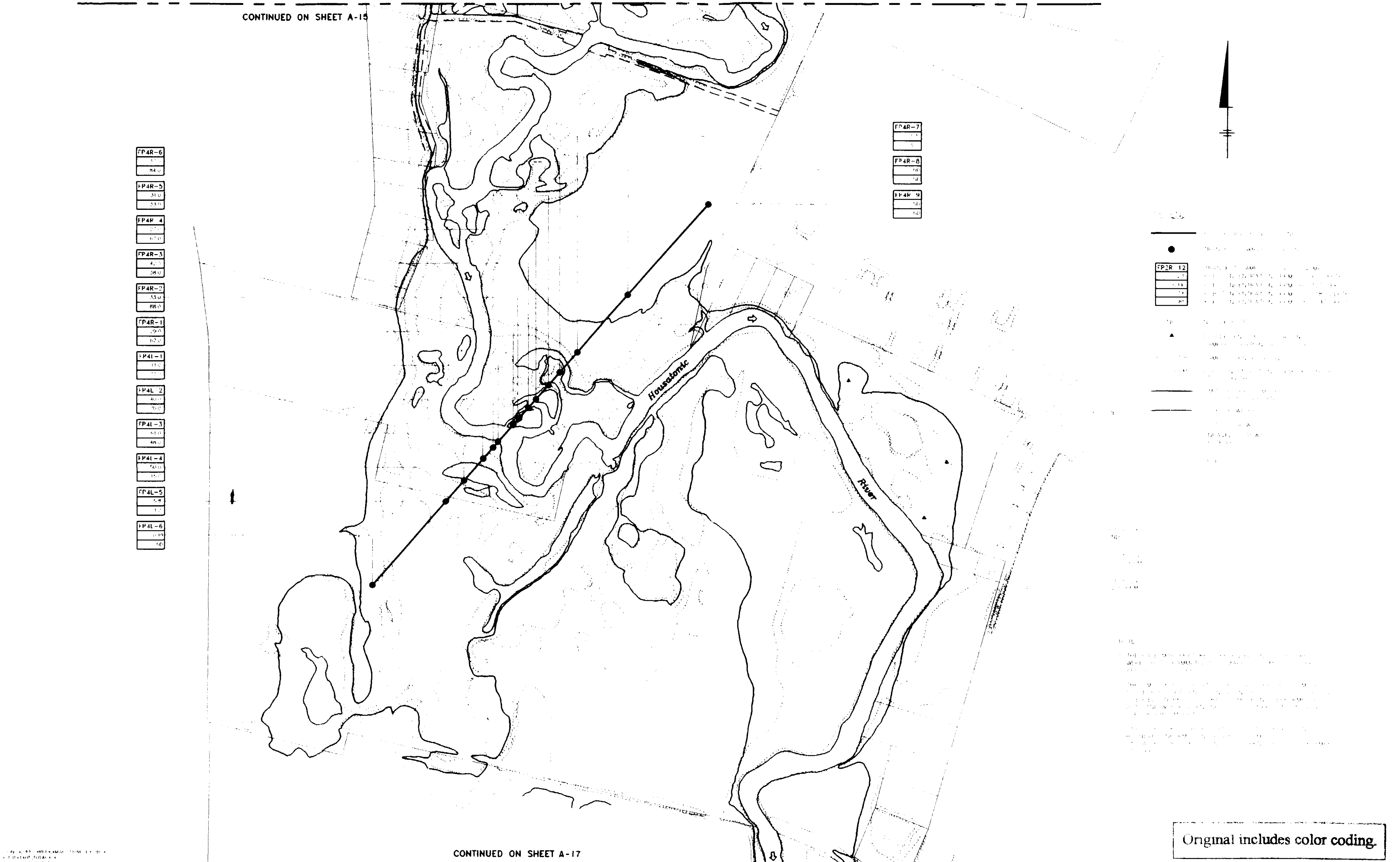
Housatonic
River
FP3L-1
FP3L-2
FP3L-3
FP3L-4
FP3P-1
FP3R-2
FP3P-3
FP3P-4
FP3P-5
FP3R-6
FP2P-12
J6-2-2
CONTINUED ON SHEET A-16
Original includes color coding
BLASLAND, BOUCK & LEE, INC
MCP PHASE II AND STM EVALUATION
FLOODPLAIN SAMPLING RESULTS
A-15

CONTINUED ON SHEET A-15



CONTINUED ON SHEET A-17

Original includes color coding.

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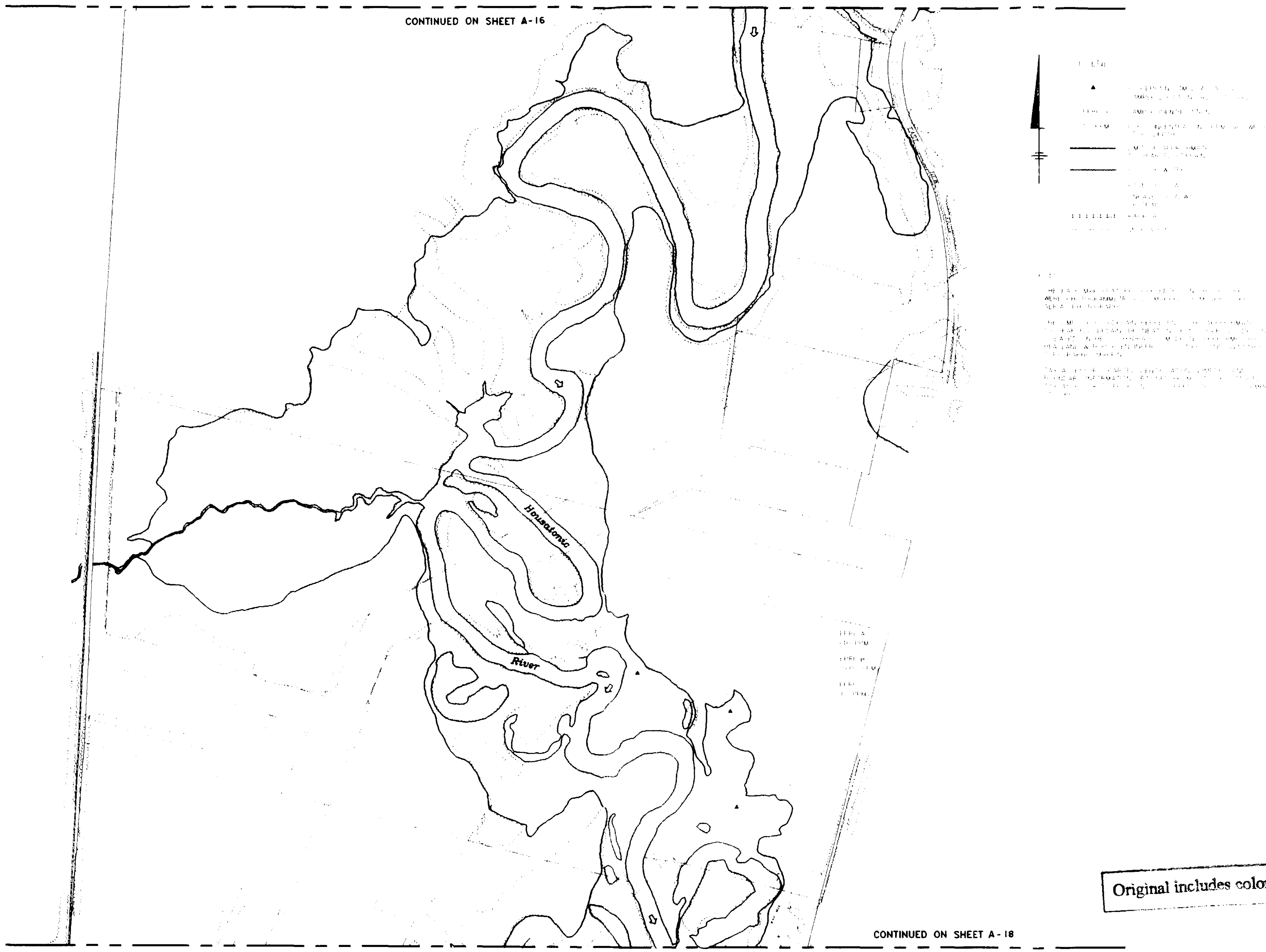
MCP PHASE II AND STM EVALUATION
FLOODPLAIN SAMPLING RESULTS

A-16

CONTINUED ON SHEET A-16



Original includes color coding

CONTINUED ON SHEET A-18

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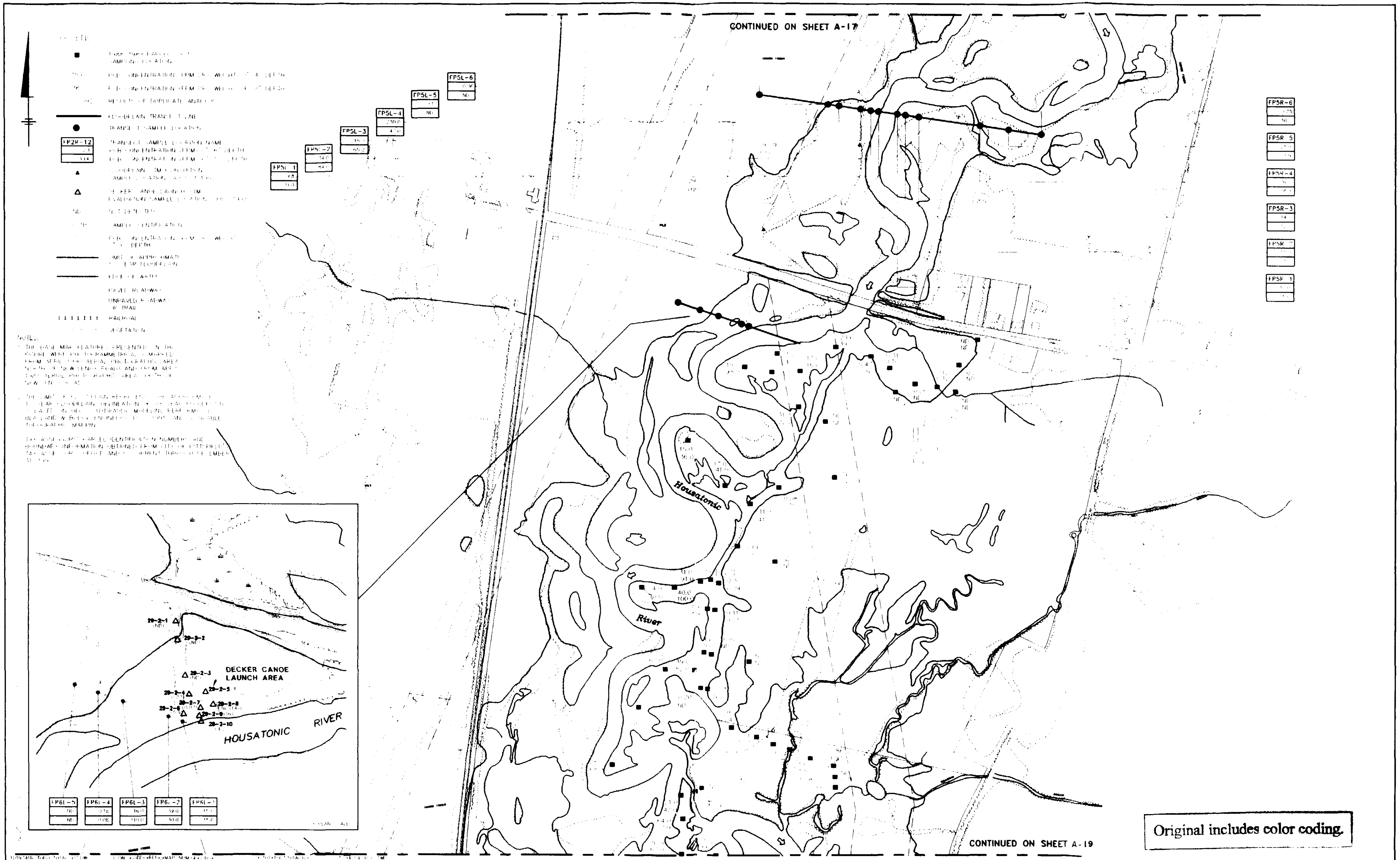
MCP PHASE II AND STM EVALUATION
FLOODPLAIN SAMPLING RESULTS

A-17



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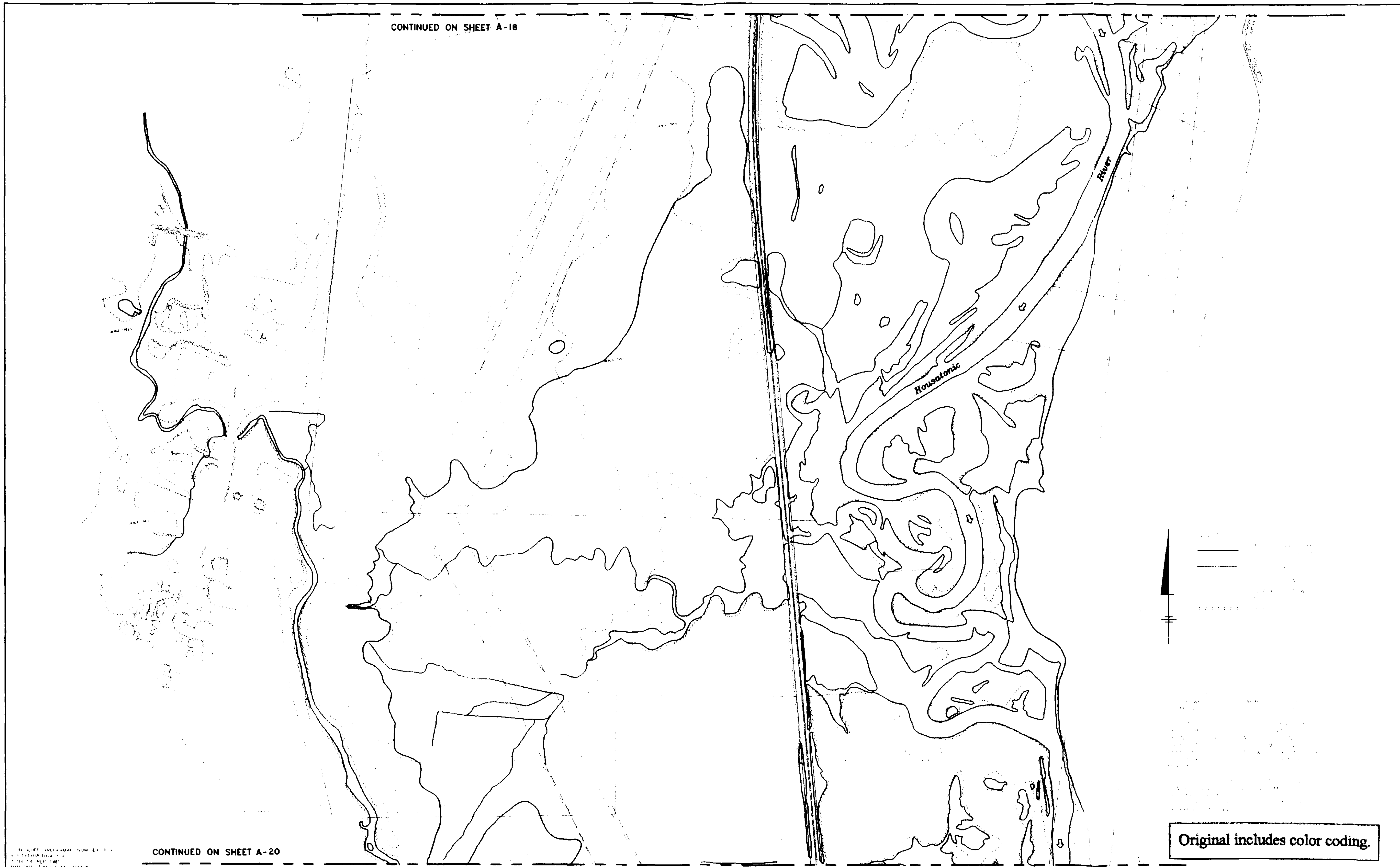
MCP PHASE II AND STM EVALUATION
FLOODPLAIN SAMPLING RESULTS

A-18



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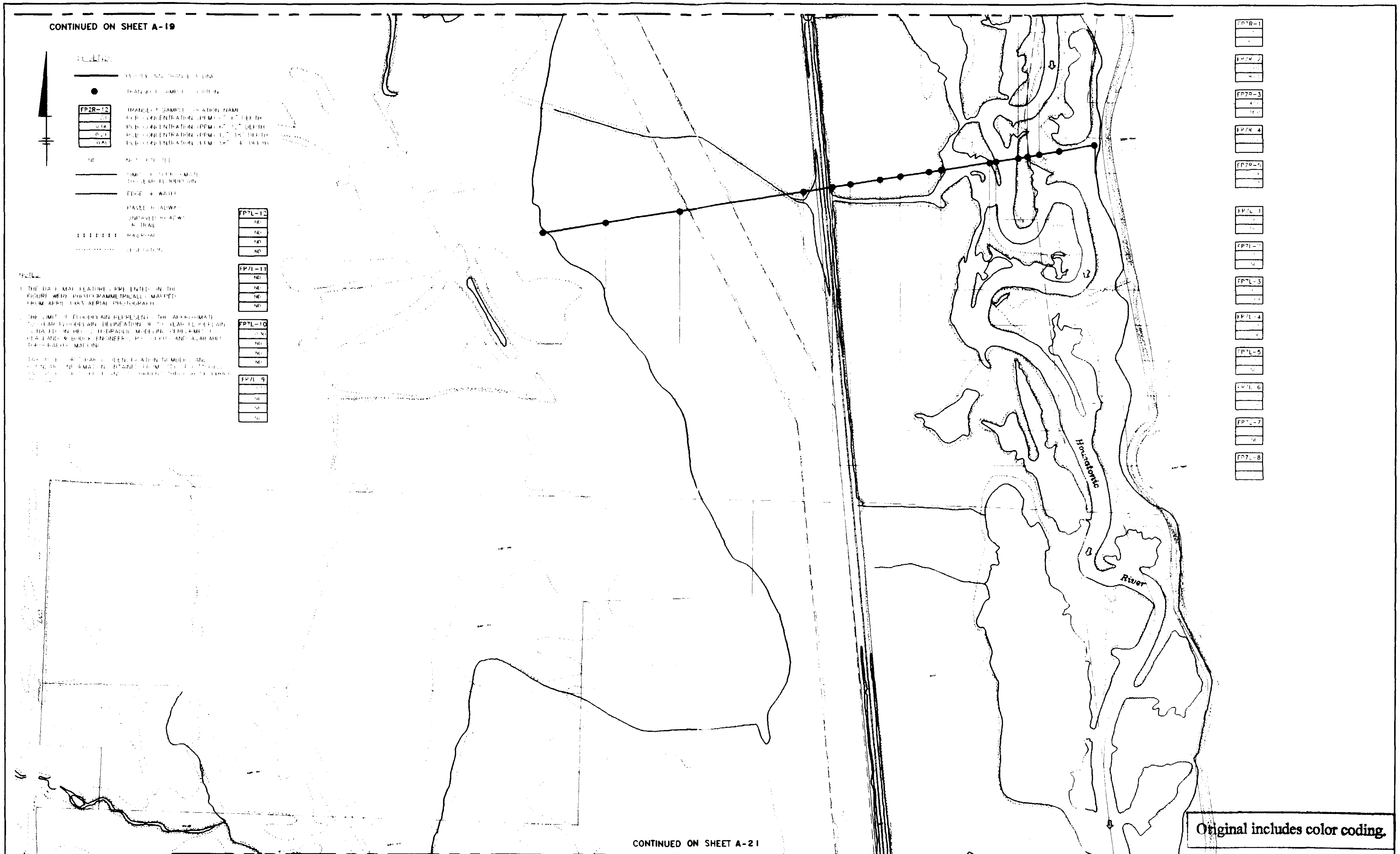
MCP PHASE II AND STM EVALUATION
FLOODPLAIN SAMPLING RESULTS

A-19



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MCP PHASE II AND STM EVALUATION

FLOODPLAIN SAMPLING RESULTS

A-20

Original includes color coding.

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MCP PHASE II AND STM EVALUATION

FLOODPLAIN SAMPLING RESULTS

A-21

CONTINUED ON SHEET A-21

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MCP PHASE II AND STM EVALUATION

FLOODPLAIN SAMPLING RESULTS

A-22