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6.1 INTRODUCTION

This Ecological Risk Assessment (ERA) was prepared to evaluate the potential risk to ecological receptors associated with the marine habitat surrounding the Maine Yankee Atomic Power Station (Maine Yankee). The ERA was prepared as part of the decommissioning of Maine Yankee, specifically as part of the Resource Conservation and Recovery Act (RCRA) process associated with closure of the site-related stormwater outfalls. In addition to assessing the potential baseline ecological risk, the ERA was prepared to develop sufficient information to make informed risk management decisions on an outfall-specific basis.

The framework and approach for the ERA follows the general guidelines outlined in the United States Environmental Protection Agency’s (USEPA) Guidelines for Ecological Risk Assessment, EPA/630/R-95/002F, dated April 1998. In addition, the following documents, which provide risk assessment guidance and/or technical information, were used in the development of the ERA:

- Intermittent “ECO Update Bulletins” of USEPA; and
- USEPA Region 1 Risk Updates.

The ERA discusses the potential for environmental impacts from site-related chemicals in the event that no remedial action is taken. Specifically, the ERA discusses the potential for existing site-related chemicals to impact near-shore intertidal and subtidal habitats of the Back River and Bailey Cove in the vicinity of outfalls 005, 006, 008, 009, 010, 011, and 012. The outfalls are related to areas of the site drained by storm drain systems, as listed in Table 3-1. Potential risks to the environment are assessed for each outfall individually, with the exception of outfalls 005 and 006, which were evaluated together due to their close proximity. The ERA only assesses potential risks to the marine environment since the outfalls discharge to these habitats. Terrestrial or freshwater wetland risks that may be associated with other potential sources on the site were not evaluated in the ERA.

In addition to the outfalls described above, another area of potential concern was identified by Maine DEP midway through the ERA process. The additional area was the silt spreading area west of the 345 kV Transmission Line. This area is an intertidal mudflat surrounded by a salt marsh and is also evaluated in the ERA.

A work plan for the ERA (Approach for Offshore Ecological Risk Assessment at Maine Yankee) was included as Appendix E of the Quality Assurance Project Plans (QAPP) for the Maine Yankee Decommissioning Project, RCRA Facility Investigation. During development of the work plan and discussions with regulators, it was determined that the ERA would be limited to the stormwater outfall areas, since no other exposure pathways were identified.
The work plan was generally followed, but close discussion, consultation, and coordination with the state and federal ecological risk assessors on this project occurred throughout the ERA process. As a result, some modifications to the approach were made in coordination with the state and federal risk assessors.

The fundamental change to the work plan was to employ a phased approach in the ERA. The phased approach consisted of an initial screening stage where chemical concentrations in sediment were screened to identify areas of concern where further investigation was warranted. The work plan called for sediment toxicity testing and benthic community structure analysis (BCSA) in addition to chemical analysis at all of the proposed sediment sampling locations. Through the phased approach, the number of locations requiring further investigation was narrowed down to three.

The results and conclusions of the initial sediment screening were presented in a technical memorandum to the Maine DEP in November 2001 (CH2M HILL, 2001b). The conclusion that only three of the sampling locations required further investigation in the form of sediment toxicity testing and BCSA were discussed and agreed upon by both the state and federal ecological risk assessors.

The second phase of the ERA was to collect further information from bulk sediment toxicity testing and benthic community structure from the three outfall locations identified as posing potential ecological risk to the benthic community. The same information was obtained from a reference location to aid in interpreting the results. The results of this phase of the ERA investigation were summarized in a technical memorandum (May 2002) addressing the ecological risk to the benthic community near each outfall (CH2M HILL, 2002a).

The third phase of the ERA was to assess the potential risk posed by bioaccumulative chemicals in the sediments. This phase was accomplished through a combination of chemical residue analysis of blue mussel, soft-shell clam, and mummichog tissue, and through modeling of potential uptake of these chemicals through the food web. The results of this phase of the ERA were presented in a technical memorandum to the Maine DEP in July 2002 (CH2M HILL, 2002b).

The results of the second and third phase of the ERA process were discussed with the state and federal regulators during a conference call on October 3, 2002. The general conclusions presented in the technical memoranda were agreed upon. It was determined that further assessment of the potential risk posed by polycyclic aromatic hydrocarbons (PAHs) to fishes was warranted, based on the nature of these chemicals. It was agreed that the results of all three memoranda and the additional PAH evaluation would be combined and used in the risk characterization phase of the full ERA.

6.2 PRELIMINARY PROBLEM FORMULATION

Problem formulation is the initial step of the ERA process. Problem formulation includes the preparation of an ecological site model, the identification of potential exposure pathways and ecological receptors, and the selection of the assessment and measurement endpoints to evaluate those receptors for which complete and potentially significant exposure pathways are likely to exist.
6.2.1 Ecological Site Model

Information on the habitat features of the site, and the fate and transport mechanisms of chemicals detected at the site, were used to build the ecological site model (Figure 6-1). The ecological site model addresses complete exposure pathways, ecological receptors, assessment endpoints, and measurement endpoints.

Environmental Setting

Maine Yankee is located on a peninsula that is bounded by the Back River to the east, mainland to the north, and Bailey Cove to the west (Figure 6-3). The site is approximately 13 miles inland from the open ocean. The coastline around the site varies between salt marsh and mudflat, with some rocky areas where the surface gradient is steepest. The eastern side of the site, where outfalls 008 through 012 are located, is characterized predominantly by a rocky shoreline with a moderately steep gradient; small patches of salt marsh are found along the immediate shoreline and mud flats are found in the vicinity of outfalls 008, 011, and 012. Little to no mudflat is present at outfalls 009 and 010, which are located on either side of the cooling water intake channel, because the shoreline is comprised of large boulders and rip-rap. Outfalls 005 and 006 are located on Bailey Cove, which is characterized by extensive mudflats.

The benthic invertebrate community of Montsweag Bay, which occurs south of the site, and Back River is both abundant and diverse. The invertebrate species of commercial or food value include the American lobster, the soft-shelled clam, the blue mussel, the blood worm, and the sand worm.

In summer, the most abundant finfish species in the area include the migratory alewives, blueback herring, and menhaden. Smaller but appreciable numbers of smelt, mackerel, and striped bass are also found in the area in summer. In winter, all of the above species leave the area except for smelt, which remain widely distributed throughout the estuary and are found at all depths. In spring and fall, large numbers of juvenile sea herring appear, but this species is completely absent in summer and winter.

The most abundant demersal (bottom-living) fish is the tomcod, which occurs in large numbers in lower Montsweag Bay in summer, but does not extend into the northern end of the Bay or in to the Back River during that time of year. Of secondary importance in abundance are winter flounder and smooth flounder. The grubby sculpin is a weak fourth in numerical importance. The last three species are more evenly distributed throughout the area than are the tomcod.

Most of the adult fish are concentrated in the central channel areas of the Bay and Back River. Juvenile flounder, alewives, and bluefish are found in flooded flats and a few species, such as mummichogs, silversides, and sticklebacks, are restricted to the shallow areas.

There are many nesting osprey in the area, including several nests on the site itself. In addition, Montsweag Bay, the Back River, and the surrounding areas provide abundant waterfowl habitat. Previous baseline surveys of migratory waterfowl in the area identified American black duck, bufflehead, and goldeneye as the three most abundant waterfowl species using the area (Maine Department of Inland Fisheries and Game, 1971). Other migratory waterfowl known to use the area include mallard, teal, scaup, scoters, common
merganser, Canada geese, and oldsquaw. The area also provides plentiful habitat for wading birds and shorebirds, such as great blue heron, snowy egret, and various sandpipers. In addition to osprey, other piscivorous birds, such as the belted kingfisher, frequently use the area, and it is also likely that bald eagles occasionally forage there as well. Finally, herring gulls and other gulls are also abundant in the area.

**Chemicals at the Site and Their Fate and Transport**

The Maine Yankee facility operated from 1972 to 1997. Over that time, minor spills and releases (primarily petroleum) and a few significant releases have occurred. Four significant releases have occurred including: (1) a release of an unknown amount of chromated water from the primary component cooling system to a storm drain in 1985, (2) a release of approximately 12,000 gallons of de-mineralized water containing sodium chromate in 1988, (3) an accidental release of approximately 200 gallons of low viscosity transformer oil (non-PCB) to the Back River in 1991, and (4) a release of kerosene to subsurface soils in the former Spare Generator Storage Building in 1994. These four releases were studied and remediated to the satisfaction of the Maine Department of Environmental Protection (DEP).

Previous site investigation activities have included sediment sampling at the stormwater outfalls located along the Back River and Bailey Cove. These outfalls receive discharge from roof drains and catch basins. Sediment samples collected at these outfalls have contained primarily petroleum products. Volatile organic compounds (VOCs) and metals were detected at low concentrations.

Based on a review of the previous sampling data, metals, semi-volatile organic compounds (SVOCs), and VOCs are the chemical classes of most potential concern. Several metals were detected in the sediments at Maine Yankee. Of these, arsenic, chromium, lead, mercury, and selenium will bioaccumulate to some degree; mercury (and in some cases selenium) is also known to biomagnify in aquatic food webs.

Most of the SVOCs detected in the sediments at Maine Yankee are PAHs. PAHs in aquatic sediments generally degrade more slowly than PAHs in the atmosphere. As the level of organic carbon in sediments increases, however, PAHs tend to become strongly adsorbed and thus have limited bioavailability. Biodegradation and biotransformation by benthic organisms are the most important biological fate processes for PAHs in sediments. Most animals and microorganisms can metabolize and transform PAHs to breakdown products that may ultimately experience complete degradation. PAHs with high molecular weights are degraded slowly by microbes and readily by multicellular organisms. Biodegradation probably occurs more slowly in aquatic systems than in soil.

### 6.2.2 Exposure Pathways and Potential Receptors

Exposure routes are the specific mechanism by which a chemical contacts or enters the body of a receptor. Exposure routes can include ingestion of water, sediment or prey with chemical body burdens, inhalation, and dermal absorption. Dermal and inhalation exposures for upper trophic level receptor species were not considered significant relative to ingestion exposures because of the general fate properties (e.g., relatively high adsorption to solids) of the chemicals previously detected at the site and the protection offered by feathers. Surface water ingestion was not considered as an exposure route due to the salinity of the water in the
Incidental ingestion of sediment during feeding, preening, or grooming activities is, however, considered in the risk estimates.

**Exposure Pathways**

The exposure pathways proposed for the ERA are conservative ones based on the environmental setting and potential habitat. The exposure pathways for four groups of ecological receptors (epibenthic, infaunal, fish, and avian predators) were identified during regulatory discussions and development of the work plan (Appendix E, Stratex, 2001d). These exposure pathways are shown in the ecological site model **Figure 6-1**. The model indicates that site-related chemicals have historically discharged from primary sources (outfalls) to the sediment at each outfall area. Sediment and animal tissue (biota) were identified as the primary media of concern to the ecological receptors. Potential routes of exposure leading to receptors are presented in **Figure 6-1** and explained as follows:

**Aquatic Biota.** Aquatic receptors (including mussels, clams, benthic invertebrates, and fish) may be exposed to chemicals in water, sediments, and pore water via partitioning across cell membranes, ingestion of sediments, and consumption of contaminated prey.

**Aquatic Avian Wildlife.** Avian receptors may be exposed to chemicals through the ingestion of food items that contain bioaccumulated chemicals from the sediment, and through incidental ingestion of contaminated sediment during foraging.

**Potential Receptors**

Ecological receptors are selected based on environmental sensitivity, trophic level/guild representation, societal value, likely exposure routes, and site use. The site characterization provided above identifies a number of estuarine communities and species located near the site that could be exposed to site-related chemicals from the outfalls, including:

- epibenthic communities in intertidal/subtidal areas;
- infaunal benthic communities in intertidal/subtidal areas;
- estuarine fishes; and
- aquatic predatory birds

Based on the estuarine communities and species potentially present and for which complete and potentially significant exposure pathways exist, the species/communities described below were selected as representative receptors for the ERA (Appendix E, Stratex, 2001d):

- The blue mussel (*Mytilus edulis*) - The blue mussel is a sessile invertebrate that attaches itself to its substrate by means of byssal threads. It lives in both intertidal and subtidal zones, attached to wharf pilings, sea walls, and rocks, often in great numbers. Subtidal beds are located almost exclusively in areas with good currents, especially around offshore islands and in the mouths of estuaries. The blue mussel feeds on phytoplankon by siphoning the seawater and suspended sediments. Blue mussels are the representative epibenthic species in the intertidal/subtidal environments that are potentially exposed to water-borne and particulate-bound contaminants.

- The soft shell clam (*Mya arenaria*) – The soft shell clam is a soft bottom burrower which inhabits shallow subtidal (10 m) estuarine waters and intertidal areas. Soft shell clams
inhabit stiff sands and muds that will not collapse against the shell valves when they are closed. Adult clams burrow deeply (as far as 30 cm), feeding through a long extensible siphon. Soft shell clams represent infaunal species in the intertidal/subtidal environment potentially exposed to bulk sediment and pore water contaminants.

- **Benthic community** – The benthic community, including mollusks, segmented worms, crustaceans, and fish, is an ecologically important collection of species. It is an important food source for birds, fish, and benthic and epibenthic invertebrates. The benthic community is potentially exposed to contaminants in bulk sediments, pore water, and the water column.

- **The mummichog** (*Fundulus heteroclitus*) – The mummichog is a small fish that lives in shallow, estuarine intertidal waters. It is found mainly in salt marshes and tidal creeks. Its entire life cycle is completed within the estuary and its diet consists mainly of benthic invertebrates. Therefore, it is a species that is potentially highly exposed to sediment-associated contaminants.

- **The shortnose sturgeon** (*Acipenser brevirostrum*) – The shortnose sturgeon is a federally-listed endangered species. It is an anadromous, bottom-feeding fish whose diet consists mainly of mollusks, supplemented by polychaetes, small benthic fish, benthic crustaceans and insect larvae. It is an indiscriminate-feeder that feels along the bottom using its barbels and "vacuums" up food items buried in the bottom sediments. The shortnose sturgeon feeds mostly at night or on windy days when turbidity is high and visibility low. At these times, it moves into shallow water (1-5 m) to forage. Feeding occurs in deeper water during the late summer (5-10 m) and winter (10-30 m) (Gilbert, 1989). The shortnose sturgeon requires temperatures less than approximately 22°C, and salinity less than 30 or 31 ppt (Gilbert, 1989). The shortnose sturgeon is potentially highly exposed to sediment-associated contaminants because of its feeding behavior (i.e., “vacuuming” the sediments for prey).

- **The osprey** (*Pandion haleaetus*) – The osprey is an avian predator whose major food is fish. The osprey feeds by hovering above water and diving for its prey. The osprey is a natural resource species of aesthetic importance. Osprey represent avian predators potentially exposed to contaminants through the aquatic food web.

- **The belted kingfisher** (*Ceryle alcyon*) – The belted kingfisher is an avian predator whose major food is fish. There is some overlap in food source between the osprey and the kingfisher; however, kingfishers generally feed on fish less than 7 inches in length, whereas ospreys generally feed on larger fish (USEPA, 1993b). Kingfishers typically feed in shallower areas where overhanging perches are available and over a much smaller range and, thus, might be more exposed to site-related chemicals. Therefore, the kingfisher was included to represent a second avian predator potentially exposed to contaminants through the aquatic food web.

- **The herring gull** (*Larus argentatus*) - The herring gull feeds on fish and shellfish at the surface and shorelines of waterbodies. The herring gull is the representative avian omnivore for the site.
6.2.3 Assessment and Measurement Endpoints

The conclusion of the problem formulation stage includes the selection of assessment and measurement endpoints, based on the ecological site model. Endpoints in the ERA define ecological attributes that are to be protected (assessment endpoints) and measurable characteristics of those attributes (measurement endpoints) that can be used to gauge the degree of impact that has or could occur. Assessment endpoints most often relate to attributes of biological populations or communities, and are intended to focus the risk assessment on particular components of the ecosystem that could be adversely affected by chemicals from the site (USEPA, 1998b). Assessment endpoints contain an entity (e.g., fish-eating birds) and an attribute of that entity (e.g., survival rate).

Because of the complexity of natural systems, it is generally not possible to directly assess the potential impacts to all ecological receptors present within an area. Therefore, receptor species (e.g., belted kingfisher) are often selected as surrogates to evaluate potential risks to larger components of the ecological community (guilds; e.g., piscivorous birds) represented in the assessment endpoints (e.g., survival and reproduction of piscivorous birds).

Assessment endpoints for the ERA are as follows:

- **Growth, survival, and reproduction of the benthic invertebrate community** - Benthic invertebrates serve as a forage base for many aquatic and semi-aquatic species. They also play an important role in the processing and breakdown of organic matter in aquatic systems. Because they have significant direct contact with, and may even consume sediment, benthic invertebrates may be highly exposed to site-related chemicals and develop body burdens. A benthic invertebrate community limited by chemical contamination would support fewer aquatic birds and fish. For the purposes of the ERA, the benthic invertebrate community was divided into two groups, epibenthic receptors (e.g., blue mussel) and infaunal (e.g., soft-shell clam) receptors.

- **Growth, survival, and reproduction of fish communities** – Fish are susceptible to direct chemical exposure from contaminated sediments. Fish communities also serve as a prey base for many aquatic and semi-aquatic organisms. The assessment endpoint selected for fish is adverse effects on the maintenance of fish populations within the habitats present. The mummichog and shortnose sturgeon were chosen as surrogate species to represent this endpoint. The mummichog is a common species in the area and the shortnose sturgeon is an endangered species known to occur in the area.

- **Growth, survival, and reproduction of aquatic bird communities** – Aquatic birds are most susceptible to bioaccumulated chemicals in prey organisms. The assessment endpoint selected for avian receptors is adverse effects on the maintenance of avian populations within the habitats present. The osprey, belted kingfisher, and herring gull were chosen as surrogate species to represent this assessment endpoint. The osprey and belted kingfisher are common species in the area and feed mainly on fish, making them appropriate choices to represent this assessment endpoint. The herring gull is a common omnivorous species in the area that feeds primarily on fish and shellfish along the shoreline, making it an appropriate surrogate species for this assessment endpoint.
Measurement Endpoints

Measurement endpoints are a measurable ecological characteristic that is related to each respective assessment endpoint (USEPA, 1997a). Each measurement endpoint is a measure of biological effects (e.g., laboratory toxicity test results). Commonly, biological responses in laboratory toxicity tests or in-field ecological measurements can be compared to reference data to determine whether there is an adverse effect associated with the observed chemical concentrations. The measurement endpoints chosen for each assessment endpoint are presented below.

<table>
<thead>
<tr>
<th>Assessment Endpoints</th>
<th>Measurement Endpoints</th>
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<tbody>
<tr>
<td>Epibenthic invertebrate growth, survival, and reproduction.</td>
<td>Comparison of hazard quotients for benthic invertebrates to a target HQ of 1.0. HQs are calculated by dividing sediment chemical concentrations by sediment screening benchmarks.</td>
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<tr>
<td></td>
<td>Comparison of blue mussel tissue residues at the Maine Yankee site with blue mussel tissue residues from a reference site.</td>
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<tr>
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<td>The following endpoints were also employed, where warranted (based on the results of the sediment and tissue screening process):</td>
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<tr>
<td></td>
<td>Statistical comparison of results of 28-day sediment laboratory toxicity tests (growth, survival, and reproduction) with the amphipod, <em>Leptocheirus plumulosus</em>, using Maine Yankee and reference sediment.</td>
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<tr>
<td></td>
<td>Comparison of benthic community structure at the Maine Yankee site with the benthic community structure at a reference site.</td>
</tr>
<tr>
<td>Infaunal invertebrate growth, survival, and reproduction.</td>
<td>Comparison of hazard quotients for benthic invertebrates to a target HQ of 1.0</td>
</tr>
<tr>
<td></td>
<td>Comparison of soft-shell clam tissue residues at the Maine Yankee site with soft-shell clam tissue residues from a reference site.</td>
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<tr>
<td></td>
<td>The following endpoints were also employed, where warranted (based on the results of the sediment and tissue screening process):</td>
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<tr>
<td></td>
<td>Statistical comparison of results of 10-day sediment laboratory toxicity tests (growth and survival) with the polychaete <em>Nereis virens</em>, using Maine Yankee and reference sediment.</td>
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<td></td>
<td>Comparison of benthic community structure at the Maine Yankee site with the benthic community structure at a reference site.</td>
</tr>
<tr>
<td>Growth, survival, and reproduction of fish.</td>
<td>Comparison of mean exposure HQs for shortnose sturgeon (derived using estimated tissue residues and literature-based critical tissue residue values) and mummichog (derived from measured tissue residues), to a reference HQ of 1.</td>
</tr>
<tr>
<td></td>
<td>Comparison of maximum sediment PAH concentrations with sediment concentrations linked with carcinogenic effects in fish.</td>
</tr>
<tr>
<td>Growth, survival, and reproduction of aquatic birds.</td>
<td>Comparison of mean exposure HQ for osprey, belted kingfisher, and herring gull (derived using literature-based toxicological data and site-specific chemical concentrations in fish and invertebrate tissue), to a reference HQ of 1. Exposure estimates include the chemical contribution from sediment</td>
</tr>
</tbody>
</table>
6.2.4 Summary of Available Data

Although limited sediment data were available from a previous site investigation, it was determined early in the ERA process that those data were of insufficient quality and quantity to prepare a thorough ERA or even a complete problem formulation. Therefore, additional surficial sediment (0-3.5 inches) samples were deemed necessary to properly assess the potential ecological risk posed by chemicals in the sediments. Subsequently, three intertidal and three subtidal sediment samples were collected at each outfall, where possible. Specific physical conditions at some outfalls required variations to this sampling plan, as follows:

- Due to the close proximity of Outfalls 005 and 006, and the extent of mudflats in this area, four intertidal samples were collected from the mudflats, and two subtidal samples were collected in the area of Outfalls 005 and 006 (Figure 2-10).
- Due to the close proximity of Outfalls 012 and N12, samples were collected at Outfall 012 only.
- No sediment was present in the intertidal area below Outfall 009, therefore intertidal sediment could not be collected.
- At Outfall 010 it was not possible to collect subtidal samples due to the scoured substrate next to the cooling water intake channel.

Sediment samples were analyzed for target compound list (TCL) SVOCs and VOCs, target analyte list (TAL) metals, grain size, total organic carbon (TOC), and percent moisture. Additional sediment was collected at each location for bulk sediment toxicity testing, if warranted based on the results of the chemical analyses. Four petite Ponar grab samples were also collected at each sampling location. The Ponar grab samples were sieved (0.5 mm), and preserved with formalin for future BCSA, if warranted.

In addition to sediment, biota samples were collected for tissue residue analysis. Blue mussels were collected from subtidal or low intertidal locations, and soft-shell clams were collected from intertidal mud flat locations. Mummichog were collected for tissue analysis along the shoreline adjacent to the facility. Biota were analyzed for metals, pesticides, PCBs, SVOCs, and percent lipids.

Sediment samples and biota samples were also collected at a reference site in Brookings Bay, beyond the influence of the Maine Yankee facility (Figure 2-4). This site was selected and approved by the regulators, after the original reference site identified in the work plan, which was located on the Damariscotta River, was determined to be unsuitable after an initial site visit. The Damariscotta River site was characterized by a primarily sandy bottom with few mudflats, and a higher salinity than is present in the Back River adjacent to Maine Yankee. Therefore, several sites in the Montsweag Bay area were visited to find an area containing substantial mudflats and salinity similar to that in the Back River. The Brookings Bay site
was selected as the reference site because the substrate and salinity were very similar to those at Maine Yankee and there were no obvious signs of potential sources of chemical contamination in the vicinity (e.g., industrial facilities or major roads).

At the reference site, sediment and biota samples were collected at three intertidal and three subtidal locations. Sediment and biota samples were analyzed for the same suite of parameters as the site samples.

Preliminary evaluations and discussions with the regulators during work plan development concluded that there were no surface water exposure pathways of concern. This is reflected in the ecological site model. Thus surface water was not sampled. The rationale for this relates to the small size of the outfall areas relative to the receiving water bodies (Back River and Bailey Cove), and the low volume and intermittent flow from the outfalls. The Back River and Bailey Cove are tidally influenced, which results in mixing and flushing of potential site-related chemicals.

6.3 SCREENING PHASE OF THE ECOLOGICAL RISK ASSESSMENT

The purpose of the screening phase of the ERA is to make an initial determination of the potential for risks based on conservative assumptions and methodologies. If such risks are possible, the results of the screening phase are then used to focus subsequent steps of the ERA process on the areas, chemicals, media, and receptors with the highest risk potential.

6.3.1 Sediment Benchmarks

The purpose of screening benchmarks is to establish chemical exposure levels that represent conservative thresholds for adverse ecological effects. The chronic screening values used for selection of chemicals of potential concern (COPCs) in sediment are presented in Table 6-1. The following hierarchy was used to select sediment screening values (Appendix E, Stratex, 2001d):

(1) If an effect range-low (ER-L) saltwater sediment screening value from the National Oceanographic and Atmospheric Administration (NOAA, 1999 and Long et al., 1995) was available, it was used preferentially;

(2) If saltwater ER-L values were not available, the saltwater threshold effect levels (TEL) from Environment Canada’s Interim Sediment Quality Guidelines (1995) were used;

(3) If saltwater ER-L or TEL values were not available, a freshwater sediment screening value (e.g., a lowest effect level) from the Ontario Ministry of the Environment (OMOE) (Persaud, et al., 1993), was used; and

(4) If none of the above screening values were available, then a variety of other literature sources were used to obtain screening values for selection of sediment COPCs.

For the purpose of adjusting TOC-dependent sediment quality criteria for selection of COPCs, sample-specific TOC values were used.
6.3.2 Sediment Benchmark Screening

The full data set was evaluated and analytes not detected in any samples were eliminated. Compounds detected in blanks and common laboratory contaminants (i.e. acetone and methylene chloride) were also eliminated from consideration. The next step was a comparison of results from individual samples to sediment screening values. Chemicals with concentrations less than the sediment screening values were eliminated from further consideration.

Comparison of detected chemicals to screening benchmarks are shown in Tables 6-2 through 6-8. The data for each habitat (i.e. subtidal or intertidal) at each outfall is represented in a separate table. Only chemicals that exceeded the benchmark in one or more samples in the group are presented in the table. There were no benchmark exceedences for intertidal sediments at Outfalls 008 and 012 or for subtidal sediments at Outfall 012. Therefore, no screening tables are presented for these areas. The data in the table are represented as “Benchmark Quotients” (BQ) or the sediment concentration divided by the benchmark. Thus a quotient of 1.0 represents a sediment concentration equal to the benchmark, a value greater than 1.0 represents a sediment concentration greater than the benchmark, and a value less than 1.0 represents a sediment concentration less than the benchmark.

6.3.3 Relationship to Reference Area

Naturally-occurring chemicals (i.e., metals) above sediment screening benchmarks were compared to concentrations present in the reference area. The chemicals present at the outfall stations at concentrations comparable to the reference stations were identified. Unless they were found to have a distinct distribution pattern, (e.g., highest chemical concentration immediately in front of the discharge and declining to the sides and offshore) these chemicals were eliminated from further consideration.

The BQ of four metals (arsenic, iron, mercury, and nickel) exceeded 1.0 at many stations, but with an average BQ for all outfall stations of 0.9 or less (Table 6-9). Two other metals (barium and zinc) slightly exceeded the screening benchmarks at a few locations. The concentrations of all of these metals at the reference site generally exceeded the concentration at the outfall stations. As indicated in Table 6-9, the mean “Reference Quotient” (site chemical concentration divided by the appropriate reference chemical concentration) was 0.8 or less for all of the metals with a maximum Reference Quotient of 2.4 and only 23% of the stations with a quotient above one. Based upon this analysis, the concentration of the metals at the outfall stations were considered to be at regional background levels and eliminated as possible COPCs.

6.3.4 Identification of Preliminary Chemicals of Potential Concern

After comparison to benchmarks and reference concentrations (inorganics), there were only a limited number of sampling stations and chemicals with Benchmark Quotients greater than 1.0. Consideration of other factors indicated that additional sediment sampling at some of these stations was not warranted, as summarized below:

- The benchmark for several SVOCs (anthracene, benzo(a)anthracene, indeno(1,2,3-cd)pyrene and fluorene) were exceeded at Stations 1 (Outfall 005/006 intertidal), 3
(Outfall 005/006 intertidal), 6 (Outfall 005/006 subtidal) and 18 (Outfall 009 subtidal). The exceedences were minimal (BQ of 1.3 or less). Based on this evaluation, there does not appear to be an elevated risk associated with these chemicals because the exceedences are minimal.

- Four SVOCs (benzo(a)anthracene, bis(2-ethylhexyl)phthalate, indeno(1,2,3-cd)pyrene, and benzo(g,h,i)perylene) exceeded screening benchmarks at Station 26 (Outfall 011, center intertidal) with BQs ranging from 1.1 to 1.5. The exceedences of benchmarks were minimal. Therefore minimal potential risk exists to the benthic community and further sampling was not recommended for this station.

Based upon the results of the sediment screening, no further sediment sampling or testing was deemed necessary at Outfalls 008, 011 and 012. Except for metals with higher concentrations at the reference site and a limited number of SVOCs as noted above, no chemicals exceeded benchmarks at these outfalls. Thus potential risk could be fully evaluated at these three outfalls using the bulk sediment chemistry results and biota tissue residue analysis without the need for further data collection.

There were 12 SVOCs detected at Station 20 (Outfall 010, center intertidal) that exceeded the screening benchmark by 3 to 23 times (Table 6-10). The ecological risk at this station could not be characterized from bulk sediment chemistry alone and toxicity testing at this site was warranted. Station 20 was located within the visible drainage channel leading from Outfall 010, whereas the other two intertidal stations were located outside of the channel on either side of Station 20. The concentrations at the other Outfall 010 stations were all well below benchmarks and/or comparable to concentrations at the reference area. Therefore, potential risk could be characterized based on bulk sediment chemistry at these two stations.

There were exceedences of SVOC benchmarks at four of the six stations at Outfall 005/006 (Table 6-10). At three of the four stations, there were two or fewer chemicals that exceeded the benchmarks (BQs of 1.2 or less); however, at Station 4 (intertidal), there were 8 exceedences, with BQs ranging from 1.2 to 5.8. Therefore, potential risk could not be adequately characterized at this station without additional information, and toxicity testing was conducted at this station. No toxicity testing was recommended for the other stations at Outfall 005/006 because of the limited number and magnitude of exceedences.

Station 16 (Outfall 009, subtidal) had the highest concentrations of SVOCs. Fourteen SVOCs exceeded the sediment screening benchmarks with BQs ranging from 4 to 44 (Table 6-10). Consequently, toxicity testing was recommended for this station. There was only one SVOC exceedence of a benchmark at other Outfall 009 stations (Station 18) and it was relatively minor (BQ=1.3).

### 6.3.5 Sediment Investigation at the Transmission Line (Silt Spreading) Area

An additional area was identified for investigation after the stormwater outfall samples were collected and evaluated. This area is the silt spreading area adjacent to the 345 kV Transmission Line. The area is a small intertidal mudflat surrounded by salt marsh. Three sediment samples were collected from the mudflat area and analyzed for SVOCs, PCBs, pesticides, and TAL metals. These data were compared with the sediment screening benchmarks used in the outfall evaluation and the maximum reference concentrations. Table
6-11 shows the BQs and RQs for those chemicals that exceeded the sediment screening values. The evaluation showed that several metals and two PAHs (acenaphthene and benzo(a)anthracene) were present in the sediment at concentrations exceeding the screening values. The metal concentrations were similar (less than 2 times reference) to the corresponding reference concentrations, with RQs ranging from 0.6 to 2.5 and only one RQ greater than 2.0. The highest RQ of 2.5 was for manganese, which had a corresponding BQ of only 1.3. Although benzo(a)anthracene was detected above the screening value, the concentrations of this chemical are consistent with those detected at Outfall 011 and at the reference site. Acenaphthene was detected above the screening value in one of the three samples, but was not detected in the other two samples, or in the duplicate sample at the same location. Therefore, it is unlikely that the isolated occurrence of this chemical represents a source of elevated ecological risk. Additional evidence to support this conclusion can be drawn from an alternative screening value for this chemical. A marine equilibrium partitioning (EqP)-based value of 1,100 µg/kg at 1 percent TOC is available from the EcoTox Thresholds (USEPA, 1996a). Since the concentration of this chemical is below the EqP value, the chemical is not likely to be bioavailable in sufficient quantities to present an ecological risk. Therefore, this evaluation suggests that there is minimal ecological risk in the sediments of the silt spreading area; therefore, no further investigation was recommended for this area.

6.3.6 Biota Tissue COPC Concentrations Relative to Reference and Screening Values

As part of the risk evaluation for the marine benthic community near the outfalls, soft-shelled clam, blue mussel, and mummichog samples were collected and their tissues were analyzed for chemical residues to assess potential risk from bioaccumulative chemicals.

Tissue Residue Screening Values

Since ingestion-based toxicity values for fish species are lacking for many chemicals, the critical body residue (CBR) approach was used instead of food web models to evaluate potential risk to the shortnose sturgeon and mummichog. Similarly, the CBR approach was used to evaluate the chemical residues measured in the clam and blue mussel samples as well. The CBR approach has several advantages, including: the integration of bioavailability and exposure from all routes by the exposed organism, and assumptions regarding steady state, equilibrium, or uptake kinetics are not required (USEPA, 2000a).

For many chemicals, the measured tissue residues were compared with tissue screening concentrations (TSCs) calculated using the methodology described in Shepard (1998). This method is based on the assumption that the USEPA’s ambient water quality criteria (AWQC), which are protective of 95% of all aquatic genera, can be used to calculate bioconcentrated tissue residues that, if not exceeded, should also be protective of 95% of all aquatic genera. The method is based on a one-compartment first-order kinetic toxicological model, using BCF values and the AWQC to calculate the TSCs. The validity of the calculated TSCs was confirmed by the author by comparing the TSCs with literature-derived effects concentrations. The author found that 94% of the literature values indicated that adverse effects only occur at tissue concentrations higher than the TSCs (USEPA, 1998b). Therefore the TSCs were deemed appropriate for a conservative screening evaluation.
Additional tissue benchmarks were compiled from tissue residue effects levels found in the literature for chemicals without a TSC.

**Clam and Mussel Tissue**

Clams and mussels were collected for tissue analysis from locations co-located with the sediment sampling locations at each outfall, except Outfalls 005/006 and Outfall 009. At Outfall 005/006, there was no mussel habitat close to the outfalls and corresponding with the sediment sampling locations. Therefore three additional clam samples were collected instead. At Outfall 009, there was no intertidal habitat present, so only mussels could be collected there. Clam and mussel tissues were also collected from the reference site. Potential risk attributable to the Maine Yankee facility was assessed by comparing tissue residue concentrations from the outfall locations with tissue residue concentrations at the reference location. This comparison resulted in Tissue Concentration Ratios (TCRs), which were calculated by dividing the tissue concentrations for a given outfall location by the maximum reference tissue concentration.

Potential risk was also evaluated by comparing chemical residue concentrations with tissue screening values. A hazard quotient (HQ) was calculated for each chemical by dividing the tissue concentration by the appropriate screening value. For many chemicals, the measured tissue residues were compared with TSCs calculated using the methodology described in (USEPA, 1998b). Summary tables of the tissue residue results for clams and mussels are presented in Tables 6-12 through 6-15. These tables present TCRs and HQs for the maximum chemical concentrations detected in each tissue type at each outfall (Tables 6-12 and 6-14). Tables 6-13 and 6-15 present TCR values for individual biota samples at each outfall. Chemicals that were not detected in any of the samples, or that had TCRs and HQs less than 1.0, are not shown in the tables.

For clam tissue, the majority of the metals, four pesticides, and several SVOCs were detected at concentrations corresponding to TCRs or HQs greater than 1.0 at one or more outfall locations (Table 6-12). Outfall 005/006 had the greatest number of TCRs above 1.0, with most of the metals, two pesticides, and nine SVOCs detected at maximum concentrations greater than the reference tissue concentrations. The majority of the TCRs greater than 1.0 were for clam tissue from locations MY06SD01 (closest to Outfall 006) and MY06SD02 (closest to Outfall 005). The other four sampling locations at this outfall had few TCRs greater than 1.0 (Table 6-13). Although many chemicals were detected at concentrations greater than the reference concentrations, most of these chemicals were below their tissue screening values or exceeded the screening value in the reference tissue as well. Most of the TCRs were between 1.0 and 2.0 and all but one TCR was below 3.5 (dieldrin). Clams collected at Outfall 008 had the lowest tissue chemical concentrations, with no chemicals detected at concentrations higher than the reference tissue concentrations (Table 6-13). Outfalls 011 and 012 had several TCRs greater than 1.0, but only one TCR greater than 2.0. Several SVOCs were detected in clam tissue from Outfall 010 at concentrations higher than the reference tissue concentrations (TCRs ranging from 1.1 to 1.8). As previously mentioned, no clams were collected at Outfall 009 because of the lack of intertidal habitat there.

For mussel tissue, several metals, pesticides, and SVOCs were detected at concentrations exceeding those in the reference tissue. Outfall 008 had the fewest chemicals with TCRs
above 1.0, with only three metals and two pesticides detected above reference tissue concentrations (Table 6-14). Of these chemicals, only chromium also exceeded the tissue screening value. The aluminum, chromium, and nickel tissue screening values were exceeded in all the tissue samples, including the reference. Therefore, it is likely that the screening value is overly conservative and the concentrations detected do not pose a significant risk to mussels.

Mussel tissue from Outfall 009 showed bioaccumulation of several SVOCs, with TCRs ranging from 1.3 to 4.3 (Table 6-15). In contrast, no SVOCs were detected above reference mussel tissue concentrations at Outfall 008, and only two SVOCs were detected at concentrations slightly above the reference tissue concentrations at Outfalls 010 and 011 and one SVOC at Outfall 012 (TCRs ranging from 1.03 to 1.3).

A few pesticides were detected at concentrations higher than reference concentrations in mussel tissue from each of the outfalls. However, none of the pesticide concentrations exceeded tissue screening values.

Although several metals were detected in mussel tissue at concentrations higher than in the reference tissue, only five metals also exceeded their tissue screening values (Table 6-14). Four of the five metals also exceeded the screening value in the reference tissue. Only mercury was detected in the reference mussel tissue at a concentration below the screening value (HQ of 0.8), but in the site mussel tissue (Outfall 012) at a concentration slightly higher than the screening value (HQ of 1.3). No mussels were collected at Outfall 005/006 because of the lack of mussel habitat close to the outfalls and sediment sampling locations.

Overall, the evaluation of the chemical residue in the clam and mussel tissue suggests that no elevated chemical residues (relative to the reference site) exist in clams and mussels at Outfalls 008, 011, and 012. The results further suggest that there are slightly elevated chemical concentrations in biota at Outfalls 005/006 and 010, and substantially elevated chemical concentrations in blue mussels at Outfall 009, compared with the reference site.

Few chemicals at a given outfall were found to have TCRs greater than 2.0 and also have HQs greater than 1.0. For soft-shell clams, only three chemicals exceeded these criteria; arsenic at Outfall 005/006, copper at Outfalls 005/006 and 012, and nickel at Outfalls 010 and 011. There were also five chemicals (cobalt, manganese, vanadium, anthracene, and phenanthrene) that had TCRs greater than 2.0, but for which there were no screening values available. For blue mussels, only one chemical, chromium, had a TCR greater than 2.0 and an HQ greater than 1.0. There were also several PAHs with TCRs greater than 2.0, but for which there were no screening values.

Mummichog Tissue

Since mummichog are a mobile receptor, samples of this species could not be collected from outfall-specific areas. Therefore, composite samples were collected from each side of Bailey Point using minnow traps placed along the shoreline. Mummichog were also collected at the reference site for comparative tissue concentrations. The chemical residues measured in the mummichog tissue samples were compared with reference mummichog chemical residues from the reference site in Brookings Bay. Tissue residues were measured from whole body composite samples of many individual fish. Care was taken to ensure that composite
samples were comprised of similar proportions of different sized fish [i.e., Reference Site: 50 percent large fish (6-8 cm), 50 percent small fish (4-6 cm); west side: 50 percent large, 50 percent small; east side: 36 percent large, 64 percent small (sample 1), and 37 percent large, 63 percent small (sample 2)]. The results of the chemical residue analyses for mummichog tissue are presented in Table 6-16. Tissue chemical residues from samples collected at the facility were compared with reference tissue concentrations by calculating TCRs. A TCR value greater than 1.0 indicates that the tissue residue for a given chemical is greater in the mummichog sample collected at the facility than in the reference mummichog sample. This evaluation revealed that many of the metals detected in mummichog tissue from the site were present at higher concentrations than in the reference tissue. However, in two of the mummichog samples, most of the metal concentrations were only slightly (less than two times) higher than in the reference sample. The third mummichog sample, one of the composite samples collected on the east side of the facility (Back River) contained several metals (barium, chromium, cobalt, iron, nickel, and vanadium) at concentrations substantially higher than those in the reference tissue.

When compared with the tissue screening effects values, the concentrations of two metals (copper and zinc) in the mummichog tissue slightly exceeded screening values (Table 6-16). However, since the magnitude of the exceedences was low (HQs ranging from 1.01 to 1.06) and the concentrations in the reference tissue were similar (HQs of 0.84 for copper and 0.99 for zinc), it is unlikely that these metals pose an elevated risk to mummichog and similar fishes in the area.

Several pesticides, two PCBs, and several PAHs were detected in all of the mummichog samples (Table 6-16). Three pesticides (DDT, alpha-BHC, and alpha-chlordane) and several PAHs were detected at concentrations slightly higher than reference concentrations in some of the samples. However, only two PAHs (acenaphthene and anthracene) were detected at concentrations higher than reference concentrations in all the samples (TCRs ranging from 1.2 to 1.7). Five other PAHs were present at concentrations slightly higher than reference concentrations in the mummichog sample from the west side of the facility (Bailey’s Cove). None of the pesticides or PCBs detected exceeded screening values and thus are unlikely to pose a risk to mummichog and similar fishes.

Screening values were not available for most of the PAHs detected. However, screening values for acenaphthene and naphthalene were available and all the detected PAH concentrations were below these screening values. Thus, the PAHs detected in the mummichog tissue likely do not pose a significant ecological risk. However, there is some uncertainty in using tissue chemical residues to evaluate potential risk from this group of compounds. PAHs are generally metabolized and depurated rapidly by fish. Thus, tissue chemical residues may underestimate potential effects from PAHs, particularly their carcinogenic properties. Therefore, additional evaluation was undertaken to evaluate the potential risk posed to fishes from PAH contamination. This discussion is presented in the Risk Characterization section of the ERA.

There were no chemicals detected in mummichog tissue with both TCRs > 2.0 and HQs > 1.0. However, there were four chemicals with TCRs > 2.0, but for which there were no screening values available. These chemicals were barium, cobalt, iron, and manganese.
6.3.7 Baseline Ecological Risk Assessment Data Collection Activities

Based upon the results of the screening phase, it was determined that additional information, including bulk sediment toxicity and benthic community structure, was warranted for some of the stations to support a baseline risk characterization. The decision as to where additional investigation was warranted was arrived at through coordination and concurrence with the state and federal regulators. The outfalls where additional study was deemed necessary included Outfall 005/006, Outfall 009, and Outfall 010, for the stations listed below:

- Station 16 (Outfall 009, subtidal)
- Station 4 (Outfall 005/006, intertidal)
- Station 20 (Outfall 010, center intertidal)
- Intertidal Reference Station 2 (for comparison). In order to fully evaluate and compare the toxicity testing results at the stations recommended above, it was necessary to perform toxicity testing on an intertidal station at the reference area.

A second round (November 2001) of sediment samples was collected for chemical analysis and toxicity testing at the stations listed above. Benthic grab samples, archived during the first round of sediment sampling (September 2001), were analyzed for BCSA at these stations as well. The results of the November 2001 analyses were compared with the initial chemistry results (Table 6-17). This comparison shows that the chemical concentrations in the sediment collected for the toxicity testing were generally similar to the concentrations in the first round of sediment sampling. However, the concentrations of SVOCs at MY06SD04 (Outfall 005/006) were generally lower in the second round, with none of the SVOCs exceeding benchmarks. Additionally, the two SVOCs, acenapthene and fluorene, which exceeded the screening benchmarks at only Outfall 005/006, Station MY06SD04, in the first round of sampling were not detected in the second round of sampling at this location. Fluorene was also detected at MY06SD16 (Outfall 009) at the highest concentration observed at any of the stations, although it was not detected there in the first round of sampling.

6.4 BASELINE PROBLEM FORMULATION

The final (baseline) problem formulation is a refinement of the preliminary (screening) problem formulation and is focused on defining the issues associated with the primary COPCs identified in the screening phase. The final problem formulation consists of an evaluation of the toxicity of key COPCs and a refined ecological site model.

6.4.1 Toxicity Evaluation

The classes of compounds represented by COPCs are limited to SVOCs. Based upon the screening results several SVOCs may pose a risk to populations of benthic invertebrates inhabiting certain areas near the facility.

Most of the SVOCs detected in the sediments at Maine Yankee are PAHs. PAHs were detected at concentrations exceeding the screening benchmarks at Outfalls 005/006, 009, 010, and 011. In aquatic environments, PAHs rapidly become adsorbed to organic and inorganic particulate materials and are deposited in sediments (Neff, 1985). Once adsorbed...
to sediment, PAHs can have limited bioavailability to aquatic organisms (Neff, 1985). However, PAHs deposited in sediments can be toxic to benthic invertebrates.

In aquatic environments, exposure to ultraviolet light can result in photomodification of some PAHs to products with increased polarity, water solubility, and toxicity compared to the parent compound (Duxbury et al., 1997). Ireland et al. (1996) showed that the photoinduced toxicity of PAHs to the daphnid, Ceriodaphnia dubia, occurred frequently during low-flow conditions and wet weather runoff, and was reduced in turbid conditions. In studies on the marine amphipod, Rhepoxynius abronius, ultraviolet radiation exposure enhanced the toxicity of fluoranthene and pyrene in sediments, but did not affect the toxicity of acenaphthene and phenanthrene (Swartz et al., 1997). Pelletier et al. (1997) found that the phototoxicity of individual PAHs (anthracene, fluoranthene, pyrene) to marine bivalves (Mulinia lateralis) and marine shrimp (Mysidopsis bahia) were 12 to >50,000 times that of conventional toxicity.

Fish may be at risk from chronic exposure to PAHs. PAH contamination in sediments has been shown to be correlated with histopathological abnormalities at a number of sites (Baumann et al., 1982; Malins et al., 1984 cited in Pastorok et al., 1994). Reductions in fish populations from acute exposures to areas of high PAH contamination is less likely; avoidance of areas with high PAH contamination has been demonstrated in some fish species (North et al., 1964 and Rice, 1973, cited in Pastorok et al., 1994).

The capacity to metabolize PAHs varies among organisms. Varanasi et al. (1985 cited in ATSDR, 1995) ranked the extent of benzo(a)pyrene metabolism by aquatic organisms as follows: fish > shrimp > amphipods > crustaceans > mussels. The fact that mussels are ranked last may be because mussels show no or limited mixed function oxidase (MFO) activity. MFO is an enzyme system responsible for the initiation of metabolism of various lipophilic organic compounds, including PAHs (Neff, 1985).

### 6.4.2 Refined Ecological Site Model

Results of the sediment screening indicated that metals and SVOCs pose a potential risk to the benthic community at some of the outfalls. The sources of these site-related chemicals are likely from historical spills and runoff from the facility, and subsequent discharge through the stormwater outfalls to the sediments in the vicinity of the outfalls. Receptors potentially at risk include benthic invertebrates, fish, and aquatic birds (Figure 6-1). Benthic invertebrates and fish may be exposed to chemicals via direct contact with chemicals in the sediment. Piscivorous birds and fish may be exposed through direct contact with the sediments as well, but are more likely exposed through ingestion of prey that contain bioaccumulative chemicals in their tissue. Fish may be at risk from the carcinogenic properties of PAHs in the sediments. This risk may not be apparent from tissue residues or ingestion-based toxicity values because these compounds are rapidly metabolized and their degradation products are most responsible for carcinogenic effects.
6.5 ECOLOGICAL EXPOSURE AND EFFECTS ASSESSMENT

Ecological exposure assessment is the process of estimating or measuring the amount of a COPC in environmental media (sediment, food items) to which an ecological receptor may be exposed.

COPC exposures were assessed for ecological receptors using the following methods:

- Measurement of chemical concentrations in sediments adjacent to outfalls.
- Dose calculations for upper trophic level receptors using food web models.

Ecological effects assessment is the process of estimating or measuring the potential adverse effects to ecological receptors associated with the COPCs.

Potential COPC effects were assessed for ecological receptors using the following methods (measurement endpoints):

- Comparisons of sediment COPC concentrations with sediment benchmarks;
- Evaluation of site tissue COPC concentrations relative to the reference location;
- Evaluation of biota tissue COPC concentrations relative to tissue screening values;
- Comparison of calculated COPC dosages for upper trophic level receptors to toxicity reference values;
- Assessment of bulk sediment toxicity test results and comparison of these results with the reference location results; and
- Analysis of the benthic community structure and comparison to the benthic community structure of the reference location.

6.5.1 Comparison of COPC Concentrations with Sediment Benchmarks

Concentrations of COPCs in sediment were compared to effects-based sediment screening benchmarks in Section 6.3.2 (Tables 6-2 through 6-8, and 6-10). The results of the sediment screening for each outfall were ranked, as described below, in relation to the reference site to produce a Weight of Evidence for exposure and effect assessment for the benthic community. The results of this ranking are used in the Risk Characterization phase of the ERA.

<table>
<thead>
<tr>
<th>Risk Ranking</th>
<th>Criteria</th>
</tr>
</thead>
<tbody>
<tr>
<td>Baseline</td>
<td>Chemical concentrations similar to the reference location.</td>
</tr>
<tr>
<td>Low</td>
<td>Minor exceedences of benchmarks (i.e., few mean BQs greater than 5), but subsequent tests showed no effects.</td>
</tr>
<tr>
<td>Intermediate</td>
<td>Several minor exceedences of benchmarks, and subsequent tests showed adverse effects.</td>
</tr>
<tr>
<td>High</td>
<td>Substantial exceedences of benchmarks (i.e., several mean BQs greater than 5), and subsequent tests showed adverse effects.</td>
</tr>
</tbody>
</table>
Outfall 005/006
For Outfall 005/006, the potential ecological risk relative to the reference site was ranked as low based on the results of the sediment screening. Although toxicity testing was recommended for station MY06SD04, the exceedences were relatively minor and there were few benchmark exceedences at the other sampling locations.

Outfall 008
The potential ecological risk relative to the reference site at Outfall 008 was ranked as baseline, based on the results of the sediment screening. No further sampling or testing was recommended for sediment at Outfall 008. Except for metals with higher concentrations at the reference site, no chemicals exceeded benchmarks at this outfall.

Outfall 009
Based upon the results of the sediment screening, toxicity testing was recommended for Station MY06SD16. Toxicity testing was limited to this station, which is located directly in front of the outfall, because the sediment screening indicated little to no risk was present at the other two subtidal sampling locations. However, the magnitude of the benchmark exceedences at this station warranted a risk ranking of high for the outfall, relative to the reference site.

Outfall 010
Potential ecological risk at Outfall 010 was ranked as low, relative to the reference site. This ranking was based on the sediment screening, which determined that toxicity testing was warranted for Station MY06SD20. Toxicity testing was limited to this station, which is located directly in front of the outfall, because the sediment screening indicated little to no risk was present at the other sampling locations.

Outfall 011
The potential ecological risk relative to the reference site at Outfall 011 was ranked as baseline, based on the results of the sediment screening. No further sampling or testing was recommended for sediment at Outfall 011.

Outfall 012
The potential ecological risk relative to the reference site at Outfall 012 was ranked as baseline, based on the results of the sediment screening. No further sampling or testing was recommended for sediment at Outfall 012.

6.5.2 Evaluation of Chemical Residues in Tissue of Target Receptors
This section describes how chemical residues in the tissue of soft-shell clams, blue mussels, and mummichog were used as indicators of COPC exposure. COPC exposure was assessed by evaluating outfall vs. reference tissue concentration ratios, or TCRs, as outlined in the work plan (Appendix E, Stratex, 2001d). The maximum TCR for each chemical at each outfall was used for this evaluation. To focus the risk characterization on the chemical of most concern, only those chemicals that were identified in the tissue screening as having a
TCR > 2.0 and a HQ > 1.0 in the same samples were carried forward into the risk characterization phase and are summarized in Tables 6-18 and 6-19.

The TCRs were ranked as follows to allow a weight of evidence approach for the Risk Characterization phase of the ERA:

```
-  Target species tissue concentration is less than or equal to reference location tissue concentration (TCR ≤ 1);
+  TCR > 1;
++  TCR > 10; and
+++ TCR > 20.
```

The HQs were ranked as follows to allow a weight of evidence approach for the Risk Characterization phase of the ERA:

```
-  Target species tissue concentration is less than or equal to the tissue screening value (HQ ≤ 1);
+  HQ > 1;
++  HQ > 10; and
+++ HQ > 20.
```

**6.5.3 Exposure Estimation for Upper Trophic Level Receptors**

Chemical exposure to upper trophic level fish receptors was evaluated by comparing estimated tissue residues in shortnose sturgeon with critical residue values from the literature associated with adverse ecological effects. Whole body chemical concentrations in shortnose sturgeon were estimated by taking a weighted average of prey item tissue concentrations as described below, and multiply by bioaccumulation factor of ten. A factor of ten was used for a conservative estimate of potential long-term exposure and the high likelihood of exposure due to the life span and foraging behavior of shortnose sturgeon. The average chemical concentrations in mummichog, soft-shell clam, and blue mussel tissues from each side of the facility and the reference area were used in the exposure estimation. Only bioaccumulative chemicals as described in USEPA (2000a) were used in the evaluation.

Since shortnose sturgeon feed on primarily on crustaceans, insect larvae, worms, molluskus, and small fishes (NMFS, 1998; Gilbert, 1989), a dietary input of 40 percent soft-shell clam, 40 percent blue mussel, and 20 percent mummichog was assumed in calculating predicted tissue chemical concentrations. No blue mussels were collected from the west side of the facility (Outfall 005/006); therefore, a dietary input of 80 percent soft-shell clam and 20 percent mummichog was used for this area. Since shortnose sturgeon are long-lived and potentially highly exposed to sediment associated contaminants, a multiplier of 10 was used to conservatively predict tissue chemical concentrations in the sturgeon.

Chemical exposure to aquatic bird receptors was evaluated by estimating daily dosages based on dietary composition and food ingestion rates for each receptor. Only bioaccumulating chemicals, as described in USEPA (2000a), were evaluated for potential exposure to upper trophic level avian receptors.

COPC concentrations from clam, mussel, and mummichog tissue samples were used to calculate the dose to the herring gull. Incidental ingestion of sediment was also included.
when calculating the total level of exposure for the herring gull. For the kingfisher, mummichog tissue chemical concentrations were used to calculate dose, since the kingfisher feeds primarily on small fishes. For the osprey, estimated fish tissue concentrations for the shortnose sturgeon were used in calculating daily chemical exposure dosages, since osprey feed on larger fishes. The body weights and food ingestion rates shown in Table 6-20 were used to develop exposure estimates for the avian receptor species.

Dietary intakes for each receptor species were calculated using the following formula (modified from USEPA [1993b]):

$$DI_x = \frac{\left[ \sum_i (FIR_i)(FC_{xi})(PDF_i) \right] + [(FIR)(SC_x)(PDS)]}{BW}$$

where:

- $DI_x$ = Dietary intake for chemical x (mg chemical/kg body wt./day)
- $FIR$ = Food ingestion rate (kg/day, dry-weight)
- $FC_{xi}$ = Concentration of chemical x in food item i (mg/kg, dry weight)
- $PDF_i$ = Proportion of diet composed of food item i (dry weight basis)
- $SC_x$ = Concentration of chemical x in sediment (mg/kg, dry weight)
- $PDS$ = Proportion of diet composed of sediment (dry weight basis)
- $BW$ = Body weight (kg, wet weight)

For carnivorous wading birds, represented by the herring gull, which feeds on prey in the sediments, potential risk was evaluated for each individual outfall area, since these receptors could potentially be exposed to localized contamination at each outfall. However, for osprey, belted kingfisher, and shortnose sturgeon, potential risk was evaluated for each side of the facility (east and west), since these receptors likely forage over much larger areas and are potentially exposed to chemicals from each outfall area.

### 6.5.4 Effects Assessment for Upper Trophic Level Receptors

The purpose of the effects evaluation is to establish chemical exposure levels (screening values) that represent conservative thresholds for adverse ecological effects. For fish receptors, tissue chemical residues were estimated (shortnose sturgeon) and compared with critical tissue residue values from the literature associated with adverse effects. Literature values for sturgeon were typically not available, so values for other fish species were used. Critical residue values from studies using dietary exposures were used preferentially over other types of exposure routes. Residue values for whole body concentrations were used preferentially over tissue residues from individual organs. Lowest Observed Adverse Effect Levels (LOAELs) were used as screening values. If LOAEL values were not available, then No Observed Adverse Effect Levels (NOAELs) were converted to LOAELs using an uncertainty factor of ten. The screening values derived from the critical residue values for mummichog and shortnose sturgeon are presented in Tables 6-21 and 6-22, respectively.
Ingestion screening values for dietary exposures were derived for each avian receptor species and chemical evaluated. Toxicological information from the literature for wildlife species most closely related to the receptor species was used, where available, but was supplemented by laboratory studies of non-wildlife species (e.g., chicken) where necessary. The ingestion screening values are expressed as milligrams of the chemical per kilogram body weight (wet) of the receptor per day (mg/kg-BW/day).

NOAELs and LOAELs from chronic studies with endpoints of growth or reproduction were selected preferentially. When chronic values were unavailable, estimates were derived or extrapolated from acute values as follows:

- When values for chronic toxicity were not available, the median lethal dose (LD$_{50}$) was used. An uncertainty factor of 100 was used to convert the acute LD$_{50}$ to a chronic NOAEL (i.e., the LD$_{50}$ was multiplied by 0.01 to obtain the chronic NOAEL).
- An uncertainty factor of 10 was used to convert a reported LOAEL to a NOAEL.

Ingestion screening values for avian receptors are summarized in Table 6-23.

A comparison of exposure with effect levels is used to characterize potential ecological risk. The comparison is expressed as a HQ or the ratio of the exposure to the effect level. If the two are equal, the HQ is 1.0. If the concentration at the point of exposure is greater than the effect level, the quotient is greater than one, suggesting a potential risk.

HQs for ecological receptors were calculated as:

$$ HQ = \frac{Dose_{total}}{LOAEL \text{ or } NOAEL} $$

Chemicals with HQs in excess of 1 for the LOAEL HQ calculation were selected as COPCs. HQs were ranked according to the following method for use in Risk Characterization (Tables 6-24 through 6-27):

- “-”  Dose is less than or equal to the TRV (HQ ≤ 1);
- “+”  Dose exceeds the TRV (HQ > 1);
- “++”  Dose exceeds the TRV by a factor of 10 (HQ > 10); and
- “+++”  Dose exceeds the TRV by a factor of 20 (HQ > 20).

This hazard ranking scheme, while evaluating potential ecological effects to individual organisms, also attempts to evaluate potential population-wide effects. Exposure to COPCs may cause population reductions by affecting birth and mortality rates, immigration, and emigration (USEPA, 1989). In many circumstances, lethal or sub-lethal effects may occur to individual organisms with little population or community level impacts; however, as the number of individual organisms experiencing toxic effects increases, the probability that population effects will occur also increases.

### 6.5.5 Risk to Upper Trophic Level Receptors

**Carnivorous Fish**

Risk was evaluated for carnivorous fishes, represented by the shortnose sturgeon, by conservatively estimating chemical residues in whole body tissue and comparing these estimates with adverse effect screening values (Table 6-24). This evaluation revealed that
several bioaccumulative metals (arsenic, cadmium, copper, lead, selenium, and silver) are present in the prey of benthic-feeding carnivorous fishes at concentrations that may pose a potential ecological risk. However, all of these metals were also predicted to pose a potential risk at the reference area and predicted HQs were similar between the facility and the reference area, with one exception (arsenic). Therefore, if there is any elevated risk, it does not appear to be associated with the Maine Yankee facility. The one possible exception is for arsenic on the west side of the facility (Bailey’s Cove). The predicted HQ for arsenic was 12.18 here, compared with HQs of 6.51 at the reference area and 4.76 on the east side of the facility (Back River). Therefore, a potential risk to shortnose sturgeon and similar fishes from arsenic cannot be dismissed for this area based upon the data collected; however, there is uncertainty in this conclusion because of the assumptions used in the food web calculations (e.g., multiplier of ten to estimate long-term exposure and accumulation may not be appropriate for arsenic).

Although several pesticides, two PCBs, and several PAHs were detected in food items from each of the areas, none of these chemicals were present at concentrations predicted to pose a risk to shortnose sturgeon and similar fishes (Table 6-24).

Carnivorous Wading Birds

The herring gull was selected as a representative species for evaluating potential risk to carnivorous wading birds. Daily dosages of each bioaccumulative chemical were estimated and compared with NOAEL and LOAEL values for each chemical and each outfall (Table 6-25). This evaluation showed that although many bioaccumulative chemicals were detected in mummichog, clams, and mussels, only the two PCBs (Aroclors 1254 and 1260) and one pesticide (Endrin ketone) detected in these prey items, were predicted to exceed daily NOAEL values (HQs ranging from 1.9 to 3.8 at the site and from 4.7 to 7.5 at the reference area). However, none of the estimated dosages of these chemicals exceeded their daily LOAEL values. Therefore, there is little to no elevated risk to carnivorous wading birds that may forage in the outfall areas around Maine Yankee. In addition, relative to the reference area in Brookings Bay, there is no elevated potential risk from any of the chemicals detected, since the two Aroclors and the pesticide were present at higher concentrations in prey items from the reference area, and thus represent a pervasive presence throughout Montsweag Bay.

Piscivorous Birds

The potential risk to two groups of piscivorous birds was evaluated, birds that feed primarily on small estuarine fishes, such as the belted kingfisher, and birds that feed on larger predaceous fishes, such as the osprey. The potential risk to these groups of birds was evaluated by comparing estimated daily dosages of the bioaccumulative chemicals detected in prey items with NOAEL and LOAEL values for the chemicals (Tables 6-26 and 6-27, for the kingfisher and osprey, respectively).

This evaluation revealed that for the kingfisher and similar birds, the dosages of only two chemicals, mercury and zinc, exceeded their NOAEL values at both the site and the reference area. The estimated daily dosages of all the other bioaccumulative chemicals were below their NOAEL values. The dosages of these metals did not exceed their LOAEL values. Since the NOAEL exceedences were similar between the site and the reference area for both mercury and zinc, and the dosages did not exceed their LOAEL values, it is unlikely that
either of these metals poses an elevated site-related risk to piscivorous birds such as the belted kingfisher.

The evaluation for piscivorous birds, such as the osprey, that feed on larger predaceous fishes, revealed that the dosages of five bioaccumulative metals (arsenic, chromium, mercury, selenium, and zinc) exceeded their NOAEL values on each side of the facility and at the reference site (Table 6-27). The daily dosage of arsenic was also predicted to exceed the NOAEL value at only one location, the west side of the facility (Bailey’s Cove). However, the dosage was only slightly above the NOAEL value (HQ of 1.49) and the LOAEL value was not exceeded. Although the NOAELs were exceeded for chromium, selenium, and zinc at all the areas, none of the estimated dosages of these metals exceeded their LOAEL values and the exceedences were similar to those at the reference site. Therefore, it is likely that these metals do not pose an elevated, site-related risk to piscivorous birds. The one metal that exceeded both the NOAEL and LOAEL values at all of the locations was mercury. However, the HQs were similar between the site (east side HQ of 1.59, west side HQ of 1.09) and the reference area (HQ of 1.54). Therefore, although a potential risk from mercury cannot be dismissed for piscivorous birds that feed on larger predaceous fishes, the potential risk appears to be pervasive throughout Montsweag Bay and thus is likely unrelated to activities at the Maine Yankee facility. No other bioaccumulative chemicals were predicted to pose potential ecological risk, since all calculated dosages were below their respective NOAEL values.

6.5.6 Toxicity of Site Sediment Versus Reference Sediment

Bulk sediment toxicity tests were conducted with sediment collected in November, 2001 from Station 16 (Outfall 009), Station 4 (Outfall 005/006), Station 20 (Outfall 010), and Reference Station 2. These stations were identified as locations where further study was needed to characterize the potential ecological risk based on the comparison to sediment screening values. The sediment screening results alone were deemed sufficient to assess the potential risk at the other outfall locations (CH2M HILL, 2001b). An intertidal reference station was included to compare test results for site sediments with results from comparable reference sediments. Sediment chemical analyses were performed, as well, to check for comparability with earlier sediment chemistry results. Toxicity tests included a 10-day test (growth and survival) with the polychaete Neanthes arenaceodentata and a chronic 28-day test (growth, survival, and reproduction) with the amphipod Leptocheirus plumulosus. Neanthes arenaceodentata is widely distributed throughout the world (ASTM, 1994) while Leptocheirus plumulosus is an Atlantic coast estuarine species, found from Cape Cod, Massachusetts to northern Florida (USEPA, 2001). Results from each of these tests were compared to the reference location. The test results were statistically analyzed according to ASTM (1994) and USEPA guidance (USEPA, 2001d) to determine if any of the three test sediments were significantly different (alpha level of 0.05) from the reference sediment, with respect to survival or growth (N. arenaceodentata and L. plumulosus) or reproduction (L. plumulosus). The results of these tests are summarized in Table 6-28. The raw data and laboratory report is provided in Appendix I

As shown in Table 6-28, for the 10-day test with Neanthes arenaceodentata there were no statistical differences in survival or growth of this organism in any of the outfall sediments,
compared with the reference sediments. The control sample exhibited 100 percent survival for this test, which met test performance criteria.

The results of the 28-day test with Leptocheirus plumulosus showed similar results for Outfalls 009 and 010, with no statistical differences in survival, growth, or reproduction of this organism in the outfall sediments, compared with the reference sediments. However, for the Outfall 005/006 sediment, significantly less survival (43%) was measured compared with the reference sediment (67%). However, only 75% of the organisms survived in the control sample for this test, which is slightly less than the 80% performance criteria for this test (USEPA, 2001d). Therefore, there is uncertainty associated with these results. There were no significant differences measured for growth or reproduction of this organism in the Outfall 005/006 sediments, in comparison to the reference sediments.

Overall the results of the sediment toxicity testing suggest that there is no risk of toxicity to benthic invertebrates from the sediments at Outfalls 009 and 010. The results for Outfall 005/006 suggest some amphipod toxicity in the laboratory test with L. plumulosus. There is uncertainty in this conclusion, however, since there was also mortality in the controls and there were no statistically significant differences measured in L. plumulosus growth and reproduction.

These measures, one from each toxicity test, were ranked as follows for use in the Risk Characterization phase of the ERA:

- “-“ no effects for all tests;
- “+” low (+) effects observed for one or more tests or intermediate (++) effects for one test;
- “++” intermediate (++) effects observed for two or more tests or high (+++) effects for one test; and
- “+++” intermediate (++) or higher effects observed for two or more tests, one of which indicates high (+++) effects.

### Ranking of Toxicity Test Results

<table>
<thead>
<tr>
<th>Sediment Toxicity Testing</th>
<th>Outfall 005/006</th>
<th>Outfall 008</th>
<th>Outfall 009</th>
<th>Outfall 010</th>
<th>Outfall 011</th>
<th>Outfall 012</th>
</tr>
</thead>
<tbody>
<tr>
<td>Low (+)</td>
<td>–</td>
<td>No Effects (-)</td>
<td>No Effects (-)</td>
<td>–</td>
<td>–</td>
<td></td>
</tr>
</tbody>
</table>

### Outfall 005/006

Based on the results of the sediment screening, toxicity testing was recommended for station MY06SD04 (mid-intertidal, in drainage channel). The results of the toxicity testing revealed low potential toxicity at this location, but these results were uncertain.

The toxicity testing showed possible toxicity from the sediment at this outfall. BCSA was performed for this sampling location to gather more information on the health of the benthic community at this outfall.
Outfall 009

Based on the results of the sediment screening, toxicity testing was recommended for Station MY06SD16 (subtidal). Toxicity testing was limited to this station, which is located directly in front of the outfall, because the sediment screening indicated little to no risk was present at the other two subtidal sampling locations.

The toxicity testing showed no apparent toxicity from the sediment at this outfall. However, several SVOCs were detected above sediment screening values, so BCSA was performed for this sampling location to gather more information on the health of the benthic community at this outfall.

Outfall 010

Based on the results of the sediment screening, toxicity testing was recommended for Station MY06SD20 (middle intertidal station). Toxicity testing was limited to this station, which is located directly in front of the outfall, because the sediment screening indicated little to no risk was present at the other sampling locations.

Toxicity testing indicated no toxicity from this sediment. However, since there were several exceedences of sediment screening values at this location, BCSA was performed to aid in assessing whether the chemicals present may have impaired the benthic community and thus pose a potential risk.

6.5.7 Benthic Community Structure

The benthic community structure was analyzed at the outfall locations where toxicity testing was performed:

- Station 16 - Outfall 009, subtidal
- Station 4 - Outfall 005/006, intertidal
- Station 20 - Outfall 010, center intertidal
- Intertidal Reference Station 2 (for comparison)

These stations were identified as locations where further study was needed to characterize the potential ecological risk. The sediment screening results alone were deemed sufficient to assess the potential risk at the other outfall locations (CH2M HILL, 2001b). The BCSA was performed to gather additional information to assist in evaluating the health of the benthic community. BCSA was also conducted for stations MY06SD01, MY06SD02, and MY06SD03 at Outfall 005/006 because the results of the toxicity testing suggested potential toxicity at location MY06SD04. Additionally, one intertidal and one subtidal station at the reference location was analyzed for comparative purposes.

Four petite Ponar grab samples (6” x 6”) were collected in September, 2001 at each sampling location. The samples were sieved (0.5mm), preserved with formalin, and archived pending the results of the sediment chemical screening and toxicity testing. Macroinvertebrates were identified to the lowest practical taxonomic level (generally, species). Abundance was identified by the total number of individuals at each station (Table 6-29). Diversity was measured by the number of taxa at each station and by calculating the Shannon-Weiner Index (H’), calculated as follows (Pielou, 1977):
Shannon-Weiner Index (H')

\[ H' = \{-\sum (n_i/N \log(n_i/N))\} \]

- \( N_i \) = number of the i\textsuperscript{th} taxon individuals in the sample
- \( N \) = total number of individuals in the sample

Since natural physical and chemical factors can affect community structure in addition to chemical stress from contamination, total organic carbon (TOC) and sediment grain size was considered in evaluating the benthic community structure at each location. TOC and percent sand/silt/clay/gravel for the sampling locations are presented in Table 6-30.

The benthic community data revealed that overall abundance was generally lower in samples collected at the site, than it was in reference site samples. However, the average number of taxa in the samples was consistent among all the locations and diversity, as measured by the Shannon-Weiner Index, was generally higher at the outfall locations (Table 6-31). The raw BCSA data and laboratory report is provided in Appendix J.

The sediment where the BCSA samples were collected varied somewhat in TOC and grain size composition. The reference samples contained 2.6 and 3.2 percent TOC (reference station 2 and reference station 5, respectively), whereas the samples from Outfalls 009 and 010 contained only about 1.4 percent TOC. The Outfall 005/006 samples were closer to the reference samples, with values ranging from 1.4 to 2.6 percent TOC. The reference site samples were dominated by silt and clay, with less than 10 percent sand. Samples 3 and 4 collected at Outfall 005/006 were the closest in physical composition to the reference samples. The substrate at Outfall 009 is considerably different than the substrate at the reference site, being characterized by mostly sand and gravel. Stations 1 and 2 at Outfall 005/006 and Station 20 at Outfall 010 also contained much more sand than the reference samples.

To aid in interpreting the benthic community structure, the six overall most abundant species were identified, among both the intertidal and subtidal stations combined. The six species comprised 80 percent of the total number of individuals found and included, in order of overall abundance, *Streblospio benedicti*, *Heteromastus filiformis*, *Neanthes virens*, *Tharyx acutus*, Tubificidae (individuals identified only to family), and *Gemma gemma*. The first four species are polychaetes, segmented marine worms that comprised the bulk of the individuals collected. The family Tubificidae are oligochaetes and *Gemma gemma* are small marine clams. The rank of most abundant species varied somewhat between intertidal (Table 6-31) and subtidal stations (Table 6-32), with Tubificid worms found in greater abundance in the subtidal samples than in the intertidal samples.

*Streblospio benedicti* was the most abundant species at five of the eight stations (both reference stations and at three of the four locations at Outfall 005/006). At Outfall 009, Tubificidae were the most abundant organisms, followed closely by *Streblospio benedicti*, while at Outfall 010 *Neanthes virens* was the most abundant species, followed by Tubificidae, with *Streblospio benedicti* being the fourth in abundance. *Heteromastus filiformis* was also relatively abundant (second or third in abundance) at all the stations except Station 16 at Outfall 009, where this species was sixth in abundance. *Tharyx acutus*
was second in abundance at both reference stations, but relatively scarce at all the other stations.

Since *Streblospio benedicti* was the numerically dominant species (approximately 42 percent of all individuals) in both the site and reference samples, and the reference site was verified to be relatively free of pollution, this species was not particularly useful as a potential pollution-tolerant indicator species. Therefore, the abundance of the six most abundant species was compared between the reference and site samples. The abundance of these species are compared for the intertidal locations in Figures 6-2 through 6-7 and for the subtidal stations in Figures 6-8 through 6-13.

The results of this evaluation suggest that although there are differences in the abundance of *Streblospio benedicti* (Figures 6-2 and 6-8) and *Tharyx acutus* (Figures 6-5 and 6-11) between the reference location and the outfall locations, this difference can possibly be explained by the abundance of the predatory species *Neanthes virens* at the outfall locations and its relative scarcity at the reference location (Figures 6-3 and 6-9). *Neanthes virens* is a relatively large invertebrate that feeds on smaller relatively immobile invertebrates, such as *Streblospio benedicti* and *Tharyx acutus*. The relative scarcity of *Neanthes virens* at the reference site may be explained by worm harvesting. It is likely that the outfall locations receive little or no worm harvesting pressure, because while the general area is harvested, the areas immediately in front of the outfalls generally are not. In contrast, the reference site is an active worm harvesting area (based on field observations and discussions with the landowner).

*Heteromastus filiformis* abundance was similar among both the reference and outfall intertidal stations (Figure 6-4). At Outfall 009, this species was considerably less abundant than at the reference subtidal station (Figure 6-10). This difference is likely related to some degree by the differences in substrates between the reference site and Outfall 009. The sediment at the Outfall 009 location was dominated by sand and gravel, whereas the reference subtidal area was dominated by silt and clay, which is a more favorable habitat for this species.

The relative abundance of Tubificid worms at Outfalls 009 and 010 (Figures 6-6 and 6-12) compared with the reference site locations and the Outfall 005/006 locations, combined with the presence of *Capitella capitata* (Table 6-32) suggests possible impairment of the benthic community at Outfall 009 and, to a lesser extent, at Outfall 010. *Capitella capitata* is perhaps the best known benthic indicator species for pollution tolerance, and an abundance of Tubificid worms is generally an indication of an organically enriched or oxygen-deficient environment. However, some genera such as *Spiroserpma* and *Varichaetradrilus* are contaminant sensitive (Engle and Summers, 1998). Polychaetes of the genus *Eteone* are often present in contaminated sediments and can often be found along with *Capitella* (Engle and Summers, 1998).

These results were used in the overall ranking of benthic community structure to produce a Weight of Evidence used in the Risk Characterization phase of the ERA:

```
... no effects for all indicators;
“+” low (+) effects observed for one or more indicators or intermediate (++) effects for one indicator;
```
“++” intermediate (+++) effects observed for two or more indicators or high (+++) effects for one indicator; and
“+++” intermediate (++) or higher effects observed for two or more indicators, one of which indicates high (+++) effects.

Ranking of the Benthic Community Structure Analysis

<table>
<thead>
<tr>
<th>Benthic Community Structure Analysis</th>
<th>Outfall 005/006</th>
<th>Outfall 008</th>
<th>Outfall 009</th>
<th>Outfall 010</th>
<th>Outfall 011</th>
<th>Outfall 012</th>
</tr>
</thead>
<tbody>
<tr>
<td>No Effects (-)</td>
<td>−</td>
<td>Intermediate (+++)</td>
<td>Low (+)</td>
<td>−</td>
<td>−</td>
<td></td>
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</tbody>
</table>

6.6 RISK CHARACTERIZATION

This section characterizes the ecological risk posed by chemicals in the sediments at Maine Yankee based upon the information presented in previous sections. Risk is characterized for each outfall. A weight of evidence approach was used to combine the results from each endpoint and present an integrated risk characterization. The lines of evidence used in the risk characterization are described below for each assessment endpoint and outfall.

6.6.1 Ecological Risk at Each Outfall Based on the Weight of Evidence

A weight of evidence approach was used to characterize offshore ecological risk associated with Maine Yankee. The weight of evidence was based on the analysis of exposures and effects. The weights of evidence for exposure were namely tissue concentration ratios and sediment benchmark quotients; likewise, the weights of evidence for effects were namely laboratory toxicity testing, benthic community structure, and receptor food web modeling.

Each line of evidence was evaluated and ranked, according to the following criteria, for use in the characterization of overall potential risk at each outfall.

- Baseline: No elevated risk (-) relative to the reference site;
- Low: Low (+) ranking for one or more indicators relative to the reference site, or intermediate (++) ranking for only one indicator;
- Intermediate: Intermediate (++) ranking relative to the reference site in two or more indicators, or high (+++) for only one indicator;
- High: High (+++) ranking for two or more indicators.

Important to the interpretation of risk is the extent to which elevated exposure relative to reference conditions and adverse effects occur concurrently. Where this concurrence exists, there is strong evidence that there is a complete exposure pathway between the contaminants and the receptors of concern. The joint probability of exposure and effects will be used to presume the probability of risk for each outfall, as follows:

- Baseline Risk: No greater than Baseline ranking for both exposure or effects, relative to the reference site ranking;
• Low Risk: No greater than Low ranking for both exposure and effects relative to the reference site;

• Intermediate Risk: Intermediate ranking for both exposure and effects, or High or intermediate ranking for one and no greater than Low ranking for the other, relative to the reference site; and

• High Risk: High ranking for either exposure or effects, and Intermediate or High ranking for the other, relative to the reference site.

Summary of the Weight of Evidence and the Overall Potential Ecological Risk Relative to the Reference Site for each Outfall

<table>
<thead>
<tr>
<th></th>
<th>Outfall 005/006</th>
<th>Outfall 008</th>
<th>Outfall 009</th>
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<td>Baseline</td>
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<td>Benthic Community Structure Analysis</td>
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<td>–</td>
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<tr>
<td>Clam / Mussel Tissue Screening*</td>
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<tr>
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<td>Intermediate</td>
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<td>Low</td>
</tr>
</tbody>
</table>

* Risk determinations based on the location of maximum concentrations at each outfall.

6.6.2 Benthic Community

Outfall 005/006

Based on the lines of evidence gathered, there appears to be some effect on the benthic community near Outfall 005/006. This is indicated by the apparent toxicity of the sediment at Station MY06SD04 to the amphipod *Leptocheirus plumulosus* and the occurrence of several chemicals above reference levels in clam tissue at some of the Outfall 005/006 stations. However, there is some uncertainty regarding the toxicity test results, and the BCSA and sediment chemical screening suggest minimal to no impairment or risk in the
overall area. Therefore, although the potential for risk to the benthic community cannot be dismissed entirely, the potential risk appears to be limited spatially and in magnitude, and will likely diminish once the outfalls are removed and future tidal flushing and sedimentation occur. Based on the bulk sediment chemistry (including review of the chromatographs for non-target compounds) any impairment to the benthic community that might exist does not appear to be related to any chemical stressor.

**Outfall 008**

Since no organic chemicals exceeded the sediment benchmarks, metals concentrations were consistent with the concentrations at the reference site, and clam and mussel tissue residues were generally similar to those at the reference location, there is likely no risk to the benthic community from chemicals in the sediments at Outfall 008.

**Outfall 009**

The results of the BCSA revealed that the benthic community is likely impaired to some degree, as the pollution indicator species *Capitella capitata* was found here, as well as an abundance of Tubificid worms. This is in contrast to the reference site and Outfall 005/006 where these organisms were absent or scarce. Additional evidence suggesting impairment included substantial exceedence of the screening benchmarks and the results of the mussel tissue analysis. Several SVOCs were detected in the mussel tissue at concentrations in excess of those observed in the reference tissue. These results suggest substantial bioaccumulation of organic chemicals from the sediment at this outfall.

**Outfall 010**

Overall, there appears to be a very localized area in the drainage channel immediately in front of Outfall 010, where there are potential effects to the benthic community. No direct toxicity was apparent from the toxicity tests. However, the sediment screening, BCSA, and clam tissue analysis all suggest some SVOC contamination at this location. Once the outfall is removed, it is likely that the concentrations will diminish substantially as the area is subjected to tidal flushing and no further chemical loading.

**Outfall 011**

The results of the clam and mussel tissue analyses showed no elevated risk from bioaccumulated chemicals in either type of tissue in relation to the reference site. Based upon the results of the sediment screening and the clam and mussel tissue analyses, it can be concluded that there is minimal to no risk to the benthic community from chemicals in the sediment near Outfall 011.

**Outfall 012**

The results of the clam and mussel tissue analyses showed no elevated risk from bioaccumulated chemicals in either type of tissue in relation to the reference site. Based upon the results of the sediment screening and clam and mussel tissue analyses, there appears to be minimal risk to the benthic community from chemicals in the sediments at Outfall 012.
6.6.3 Fish

This section characterizes potential risk to fishes from chemicals in the sediments at Maine Yankee.

**Small Benthic Fishes (mummichog)**

The concentrations of two metals (copper and zinc) in the mummichog tissue slightly exceeded screening values. However, since the concentrations in the reference tissue were similar, it is unlikely that these metals pose an elevated risk to mummichog and similar fishes in the area. None of the pesticides or PCBs detected in mummichog tissue exceeded screening values and thus are unlikely to pose a risk to mummichog and similar fishes. Screening values were not available for most of the PAHs detected, but tissue concentrations were below screening values for those that were available. Therefore, PAHs in the sediments likely do not pose a significant risk to mummichog and similar fishes. However, given the carcinogenic properties of these chemicals and their rapid depuration in fishes, further discussion of the potential risk posed by PAHs to fish is presented below.

**Carnivorous Fishes (shortnose sturgeon)**

Although several bioaccumulative metals (arsenic, cadmium, copper, lead, selenium, and silver) are present in the prey of benthic-feeding carnivorous fishes at concentrations that may pose a potential risk, these metals were also present at similar concentrations in prey from the reference area as well, with one exception (arsenic). Arsenic on the west side of the facility (Bailey’s Cove) was identified as a COPC for carnivorous fishes. The predicted HQ for arsenic was 12.18, compared with HQs of 6.51 at the reference area and 4.76 on the east side of the facility (Back River). Therefore, a potential risk to shortnose sturgeon and similar fishes from arsenic cannot be dismissed for this area based upon the data collected; however, there is uncertainty in this conclusion because of the assumptions used in the food web calculations (e.g., multiplier of ten to estimate long-term exposure and accumulation may not be appropriate for arsenic).

Although several pesticides, two PCBs, and several PAHs were detected in food items from each of the areas, none of these chemicals were present at concentrations predicted to pose a risk to shortnose sturgeon or similar fishes.

**Evaluation of Potential Effects from PAH Exposure to Fish**

Since PAHs are rapidly metabolized by fishes, additional evaluation was undertaken to assess potential risk to fishes from these chemicals that might not be identified by tissue chemical residues. A discussion of sediment PAH concentrations found to be linked to mutagenic and carcinogenic effects in fishes is presented below.

Baumann and Harshbarger (1998) published the results of approximately 20 years of monitoring the effects of PAH contamination on brown bullhead in the Black River, Ohio. Their study presents evidence linking PAH sediment concentrations with varying rates of cancerous liver lesions in brown bullhead. The sediment concentrations linked with causing cancer in brown bullhead are presented in Table 6-33 along with the maximum PAH concentrations detected in sediments at the outfalls identified for further study in the screening phase (Table 6-10). As the table shows, a total PAH concentration of 4,850 µg/kg...
was associated with a 7 percent liver cancer rate in brown bullhead. However, a higher concentration of 10,669 µg/kg total PAH in 1994, following a remedial dredging operation in 1990, was associated with no liver cancer in brown bullhead.

Another study of cancerous liver lesions in mummichog was published by Vogelbein et al. (1990). This study found that 93 percent (56 of 60) of the individuals from a highly PAH contaminated site (total PAHs 2,200,000 µg/kg) had grossly visible hepatic lesions. However, they also reported that no hepatic lesions were found in individuals from two less contaminated sites. Mummichog from a site with 61,000 µg/kg total PAHs exhibited no hepatic lesions, but did show some changes in liver histology. Individuals from a relatively uncontaminated site (3,000 µg/kg total PAHs) were histologically normal.

The maximum total PAH concentration initially measured at Outfall 005/006, 009, and 010 were 4,671, 91,950, and 43,900 µg/kg, respectively (Table 6-33). Subsequent sampling at Outfall 9 revealed maximum total PAH concentrations of 177,570 µg/kg. These data suggest that there is no cancer risk to fish from PAH exposure at Outfall 005/006, since the concentration is less than the concentration associated with the zero percent cancer rate presented in Baughmann and Harshbarger (1998) and close to the no effect concentration of 3,000 µg/kg presented in Vogelbein et al. (1990).

The total PAH concentration at Outfall 010 (43,900 µg/kg) suggests a possibility of risk to fish. However, since the concentration is less than the 61,000 µg/kg associated with no hepatic lesions in mummichog (Vogelbein et al., 1990), it is unlikely that this concentration represents a significant risk to fish. This conclusion is also supported by the nature and extent of PAH contamination at the outfall. PAHs were mostly limited to the center, intertidal sampling location. The substrate at Outfall 010 consists of shallow sediments interspersed with and overlaying rock outcrops. Therefore, the contamination is limited in spatial and vertical extent and thus is unlikely to pose a significant risk to mobile receptors, such as fish.

The maximum total PAH concentration detected at Outfall 009 was 177,570 µg/kg. Although this concentration is less than the sediment concentrations in the literature associated with carcinogenic effects in fish (i.e., 1,226,400 and 2,200,000 µg/kg) the concentration is higher than the 61,000 µg/kg reported to affect liver histology in mummichog (Volgelbein et al., 1990). Therefore, a potential risk to fish from PAHs in the sediment at Outfall 009 cannot be ruled out.

6.6.4 Aquatic Birds

This section characterizes the potential ecological risk posed by bioaccumulative chemicals in the sediments at Maine Yankee to aquatic birds in the area.

Carnivorous Wading Birds

Although many bioaccumulative chemicals were detected in mummichog, clams, and mussels, only the two PCBs and one pesticide detected in these prey items, Aroclors 1254 and 1260 and Endrin ketone, were predicted to exceed daily NOAEL values. However, none of the estimated dosages of these chemicals exceeded their daily LOAEL values. Therefore, there is little to no elevated risk to carnivorous wading birds that may forage in the outfall.
areas around Maine Yankee. In addition, relative to the reference area in Brookings Bay, there is no elevated potential risk from any of the chemicals detected, since the chemicals were present at higher concentrations in prey items from the reference area, and thus represent a pervasive presence throughout Montsweag Bay.

**Piscivorous Birds**

The potential risk to two groups of piscivorous birds was evaluated, birds that feed primarily on small estuarine fishes, such as the belted kingfisher, and birds that feed on larger predaceous fishes, such as the osprey.

This evaluation revealed that for the kingfisher and similar birds, the dosages of only two chemicals, mercury and zinc, pose a potential risk. Although mercury and zinc were found to potentially pose a risk, the concentrations of these metals in mummichog tissue were similar between the site and the reference area. Therefore, it is unlikely that either of these metals poses an elevated risk to piscivorous birds such as the belted kingfisher.

The evaluation for piscivorous birds that feed on larger predaceous fishes revealed that only one chemical (mercury) may pose a potential risk. Mercury was the only chemical that exceeded both the NOAEL and LOAEL values. However, the HQs were similar between the site (east side HQ of 1.59, west side HQ of 1.09) and the reference area (HQ of 1.54). Therefore, although a potential risk from mercury cannot be dismissed for piscivorous birds that feed on larger fishes, the potential risk appears to be pervasive throughout Montsweag Bay and thus is likely unrelated to activities at the Maine Yankee facility.

6.6.5 Uncertainty

Uncertainty is inherent in risk assessment and limits the applicability of the results. The lack of site-, species-, and COPC-specific data is common, leading to situations where assumptions and substitutions must be made in order to arrive at a reasonable estimate of risk. Use of these assumptions and substitutions leads to uncertainties in the conclusions of the risk assessment. Where uncertainty cannot be avoided, best professional judgment was used to guide decisions, and reasonable levels of conservatism were applied to risk estimates to ensure that risk was neither underestimated nor grossly overestimated. The uncertainties inherent in the risk assessment process, and therefore with the outfall-specific ERAs, were discussed and quantified, in terms of overestimating or underestimating risk, to the extent possible in the ERA report.

Uncertainties are present in all risk assessments because of the limitations of the available data and the need to make certain assumptions and extrapolations based on incomplete information. The uncertainty in this risk evaluation is mainly attributable to the following factors:

- **Ingestion Screening Values** - Data on the toxicity of many chemicals to the receptor species were sparse or lacking, requiring the extrapolation of data from other wildlife species or from laboratory studies with non-wildlife species. This is a typical limitation and extrapolation for ecological risk assessments because so few wildlife species have been tested directly for most chemicals. The uncertainties associated with toxicity extrapolation were minimized through the selection of the most appropriate test species for which suitable toxicity data were available. The factors considered in selecting a test
species to represent a receptor species included taxonomic relatedness, trophic level, foraging method, and similarity of diet.

A second uncertainty related to the derivation of ingestion screening values applies to metals. Most of the toxicological studies on which the ingestion screening values for metals were based used forms of the metal (such as salts) that have high water solubility and high bioavailability to receptors. Since the analytical samples on which site-specific exposure estimates were based measured total metal concentrations, regardless of form, and these highly bioavailable forms are expected to compose only a fraction of the total metal concentration, this is likely to result in an overestimation of potential risks for these chemicals.

A third source of uncertainty associated with the derivation of ingestion screening values concerns the use of uncertainty factors. For example, NOAELs were extrapolated to LOAELs using an uncertainty factor of ten. This approach is likely to be conservative since Dourson and Stara (1983) determined that 96 percent of the chemicals included in a data review had LOAEL/NOAEL ratios of five or less. The use of an uncertainty factor of 10, although potentially conservative, also serves to counter some of the uncertainty associated with interspecies extrapolations, for which a specific uncertainty factor was not used.

- **Critical Tissue Residue Screening Values** – Data on the toxicity of many chemicals to the fish species evaluated were sparse or lacking, requiring the extrapolation of data from other species. This is a typical limitation and extrapolation for ecological risk assessments because so few species have been tested directly for most chemicals. The uncertainties associated with toxicity extrapolation were minimized through the selection of the most appropriate test species for which suitable toxicity data were available. Critical residue values from whole body were used as much as possible over data collected from individual organs, and data from studies using dietary exposure were used over other routes of exposure (e.g., water or direct injection). Studies involving long-term, chronic exposures were used preferentially over short-term, acute studies where possible. If chronic exposure data were not available, then uncertainty factors were used to convert from acute to chronic. Similar species or guilds were used to select surrogate species as much as possible in the selection of toxicity information. However, there remains some uncertainty in extrapolating toxicity data between species.

- **Chemical Mixtures** - Information on the ecotoxicological effects of chemical interactions is generally lacking, which required (as is standard for ecological risk assessments) that the chemicals be evaluated on a compound-by-compound basis during the comparison to screening values. This could result in an underestimation of risk (if there are additive or synergistic effects among chemicals) or an overestimation of risks (if there are antagonistic effects among chemicals). However, exposure to chemical mixtures was accounted for in several of the lines of evidence collected, sediment toxicity testing and benthic community analyses.

- **Mean Versus Maximum Media Concentrations** - As is typical in an ERA, a finite number of samples of environmental media are used to develop the exposure estimates. The most realistic exposure estimates for mobile species with relatively large home ranges and for
species populations (even those that are immobile or have limited home ranges) are those based on the mean chemical concentrations in each medium to which these receptors are exposed. This is reflected in the wildlife dietary exposure models contained in the Wildlife Exposure Factors Handbook (USEPA 1993b), which specify the use of average media concentrations.

- **Control Survival in the Bulk Sediment Toxicity Tests** – The percent survival of the control organisms in the 28-day bulk sediment toxicity tests with the amphipod *Leptocheirus plumulosus* did not achieve the 80 percent survival test criterion. Although this introduces some uncertainty in the results of the tests, the control survival was 75 percent, only slightly less than the criterion. Although there is uncertainty with the test, the control survival for the 10-day test with *Neanthes arenaceodentata* was 100 percent. Therefore, the results of the 10-day test provided additional information with little uncertainty on the potential toxicity of the sediments tested.

### 6.7 CONCLUSIONS

Based on the weight of evidence from the various studies and evaluations conducted for the ecological risk assessment, there are potentially significant risks to fish and benthic invertebrates from site-related chemicals in the sediments at Outfall 009. Although site-related chemicals were detected in the sediments at some of the other outfall locations, the weight of evidence suggests that the potential ecological risk at the other outfalls is minimal.