1995 Annual fish and shellfish report

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1995 Annual Fish and Shellfish Report

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EXECUTIVE SUMMARY

In 1995, the Massachusetts Water Resources Authority (MWRA) conducted a biomonitoring program for fish and shellfish. The 1995 activities represent the latest year in a continuing biomonitoring program which supports evaluation of the future MWRA effluent outfall in Massachusetts Bay. The goal of the biomonitoring program is to obtain baseline data that may be used to assess the potential environmental impact of the effluent discharge on Massachusetts Bay, and to evaluate the facility's compliance against the NPDES effluent discharge permit.

The specific objective of the 1995 fish and shellfish monitoring program was to define the baseline condition of three indicator species: winter flounder (*Pseudopleuronectes americanus*), Northern lobster (*Homarus americanus*), and blue mussel (*Mytilus edulis*). Specimens were collected from sites in Boston Harbor (Deer Island Flats (DIF), off *Discovery*), Massachusetts Bay (Future Outfall Site (FOS), Nantasket Beach (NB), Broad Sound (BS)), and Eastern Cape Cod Bay (ECCB). Baseline conditions were characterized in terms of biological parameters (length, weight, biological condition); the presence/absence of disease (both internal and external); and concentrations of organic and inorganic compounds in various tissues. These tissues included: for the winter flounder - liver and filet; for the northern lobster - hepatopancreas and tail meat; and for the blue mussel - soft tissue. The monitored parameters were examined for spatial trends between stations in 1995 and interannual variations from previous monitoring data. In addition, body burdens of certain pesticides (DDT, aldrin/dieldrin, chlordane, heptachlor, etc.), PCBs, lead, and mercury were compared to FDA Action limits and monitoring program warning limits to evaluate potential risk or trends. Finally, the results were evaluated for their ability to answer the underlying monitoring hypotheses.

Flounder

Fifty winter flounder were collected at five sites (DIF, NB, BS, FOS, ECCB) for the 1995 monitoring program. At DIF, 16 fish were collected in the April survey and the remaining 34 obtained through the New England Aquarium's "Fish Day 3". The mean age of fish collected at DIF was significantly lower than for the FOS, NB, and BS. In general, the mean flounder age at DIF and BS has declined since 1989, but has shown little change at other stations.

The external condition of the collected fish indicated few abnormalities. Fin erosion varied between stations (range = 0.2 to 0.5), with the FOS fish significantly lower than other stations. The amount of fin erosion observed throughout was considered at the low end of the range and well below that observed in the late 1980s.

The flounder liver histology results indicated that DIF and BS fish exhibited the greatest prevalence of lesion hydropic vacuolation (focal, tubular, centrotubular). Interannual comparison of the lesion prevalence at DIF indicated a statistically significant decrease over the period 1987 to 1995. Neoplasia was absent from fish collected in 1995 except in one specimen from BS. Neoplasm prevalence in DIF winter flounder has fallen from elevated levels in the 1980s to undetectable levels during the 1992-1995 period.

Fifteen winter flounder were collected at three sites (DIF, FOS, ECCB) for analysis of tissue concentration of organic and inorganic in filet and liver. The spatial patterns of tissue contaminant levels in winter flounder were examined. Mean 1995 concentrations of organic compounds in filets and liver tissue were generally highest at DIF and lowest at ECCB. This pattern was also true for mercury; but other metals (Ag, Cr, Cu, Ni, Pb, Zn) detected in liver tissue showed station-to-station variation.

Interannual comparisons of tissue organic contaminant levels for the period 1992-1995 displayed some station-to-station variability, but two general trends observed were for elevated concentrations of DDT and PCBs and reduced chlordane levels in 1995 fish relative to earlier years. For metals, the observed tissue levels were comparable to or somewhat lower than 1994 and earlier years.

The relationship between contaminant levels and liver histopathology was explored. Regressing the prevalence of centrotubular hydropic vacuolation (CHV) against winter flounder tissue levels from fish collected across the monitoring area during 1992-1995 indicated statistically significant relationships ($p<0.05$) for selected organic compounds and mercury in filet, and chlordanes in liver, but for other organic compounds or metals, no relationship exists. These relationships will be further examined during future monitoring years.

Comparison was made between flounder edible tissue contaminant levels and FDA regulatory action limits. The 1995 levels, like those detected in previous monitoring years (1992-1994), were well below the federal legal limits.

Lobster

Fifteen northern lobster were collected at three sites (DIF, FOS, ECCB) for the 1995 monitoring program. Three legal size lobsters were collected by direct trapping, 42 others were obtained from commercial traps located at or within 2 km of sampling stations.

The size, sex, and external appearance (i.e., black gill disease, shell erosion, external tumors, etc.) were determined for the collected lobsters. Little difference in length, weight, or sex ratios was observed between stations with the exception of a preponderance of male lobsters at FOS. With the exception of incidence of shell erosion at FOS, no deleterious external conditions were noted.

The spatial patterns of tissue contaminant levels in northern lobster were examined. Mean 1995 concentrations of organic compounds in edible tail meat tissue were generally highest at DIF and lowest at ECCB. In the hepatopancreas, the organic compounds were generally highest at FOS and lowest at ECCB. Mean mercury concentrations in tail meat were highest at FOS, while mercury concentrations in the hepatopancreas were similar among stations. The highest concentrations of other metals in hepatopancreas were at DIF (Ag, Cr, Cu) or FOS (Ni, Pb, Zn).

Comparison of 1995 data with previous years (1992-1994) indicates that most tissue contaminant levels were comparable to the range previously observed. The exceptions were higher levels of PCBs and pesticides in the hepatopancreas and lower levels of chlordane in tail meat.

Comparison was made between lobster edible tissue contaminant levels and FDA regulatory action limits for pesticides, PCBs, and mercury. The 1995 levels, like other monitoring years (1992-1994), were well below the federal legal limits and indicate no risk for human consumption.

Blue Mussel

Mussels were collected at two reference sites (Gloucester, Sandwich) and deployed for 49 days in arrays at three sites (DIF, FOS, and off *Discovery* in Boston Harbor). Sandwich mussels were used as reference material to better assess the potential bioaccumulation of mercury. Arrays were successfully retrieved at DIF and *Discovery* sites, but not at FOS (note: loss was apparently due to entanglement with fishing gear). Mussel survival within the deployed arrays was high ($\geq$94%). Sex determination of the mussels indicated a high proportion of immature males, continuing an unexplained trend observed in 1994.

Although the 1995 PAH results were slightly confounded by differences in analytical technique, the spatial and temporal trends observed were consistent with those observed in 1992-1994. The LMW-PAHs were highest in the DIF mussels, while HMW-PAHs were highest in *Discovery* site mussels. NOAA total PAH body burdens for all sites were comparable to 1991-1994 levels. Pesticide (DDT, chlordane, dieldrin) levels were also comparable to earlier data.

Lead tissue concentrations were statistically greater in the *Discovery* site mussels. Mercury was not significantly different among the two deployment locations and reference site and mussel tissue concentrations were uniformly low. These low concentrations, specifically in DIF mussels, contrasted with other studies, and further evaluation of mercury body burdens may be necessary to fully evaluate the bioavailability of mercury to deployed mussels at the three locations.

Evaluation of Monitoring Hypotheses

An integral part of the MWRA fish and shellfish monitoring is a periodic re-evaluation of the adequacy of the current program to fulfill the overall goals of the monitoring program. In particular, this means a re-examination of the adequacy and effectiveness of the underlying monitoring hypotheses to answer questions regarding the potential effects of the relocated MWRA effluent.

Of the six monitoring hypotheses, four are associated with the potential for edible tissue (flounder, lobster, mussel) to exceed warning levels for mercury, lead, or PCBs. These four hypotheses appear to be sufficient. Current tissue concentrations are generally an order of magnitude or more below warning and FDA regulatory levels. Values approaching the warning or action levels should be readily detectable in the program. Similarly, the monitoring hypothesis regarding future increases of the prevalance of flounder liver CHV at FOS relative to baseline levels measured in outer Boston Harbor also appears to be sufficiently sensitive to detect trends, based on current data.

The remaining monitoring hypothesis, detection of a three-year trend in elevated bioaccumulative lipophilic contaminants, was further evaluated due to greater-than-anticipated levels of some organic contaminants (pesticides, PCBs) in flounder liver and lobster hepatopancreas. A review of the magnitude of interlaboratory analytical performance was conducted and concluded that differences in analytical data between laboratories are negligible. Since interlaboratory variation does not appear responsible for the observed increases in some organic compounds, this monitoring hypothesis will require further study and evaluation.

1.0 INTRODUCTION

In 1995, the Massachusetts Water Resources Authority (MWRA) conducted a biomonitoring program for fish and shellfish as one part of a multi-faceted environmental monitoring program (MWRA, 1991). This program is the latest year in a continuing biomonitoring program (MWRA, 1991) which supports evaluation of the future MWRA effluent outfall, located approximately 9½ miles offshore of Deer Island in Massachusetts Bay (as shown in Figure 1-1). The goal of Phase I of the biomonitoring program is to obtain baseline data that may be used to assess the potential environmental impact of the effluent discharge on Massachusetts Bay, and to evaluate the facility's compliance against the NPDES effluent discharge permit.

The specific objective of the fish and shellfish monitoring is to define the baseline condition of three indicator species [i.e., winter flounder (*Pseudopleuronectes americanus*), Northern lobster (*Homarus americanus*), blue mussel (*Mytilus edulis*)] in terms of biological parameters (length, weight, biological condition), the presence of disease (both internal and external), and the concentrations of organic and inorganic compounds in various tissues, including the liver (winter flounder), hepatopancreas (lobster), and edible tissues (winter flounder filet, blue mussel soft tissue, and lobster tail). This baseline characterization of the health of winter flounder, lobster, and mussel in Boston Harbor, Massachusetts Bay, and Eastern Cape Cod Bay (i.e. Boston Harbor and the Bays) forms the basis for assessing potential changes resulting from the relocation of the outfall discharge.

One survey per indicator species was conducted in 1995 in Boston Harbor and the Bays to determine the body burden of toxic compounds of these three species, and to assess the physiological status of flounder and lobster, following the procedures and protocols described in greater detail in Section 2. Section 3 presents the results of the 1995 surveys, and discusses the recent data, as well as comparisons with historical data. Section 4 presents the conclusions drawn from these 1995 survey results and historical trends. Section 5 includes recommendations for the biomonitoring program, while Section 6 lists the references cited in this document.

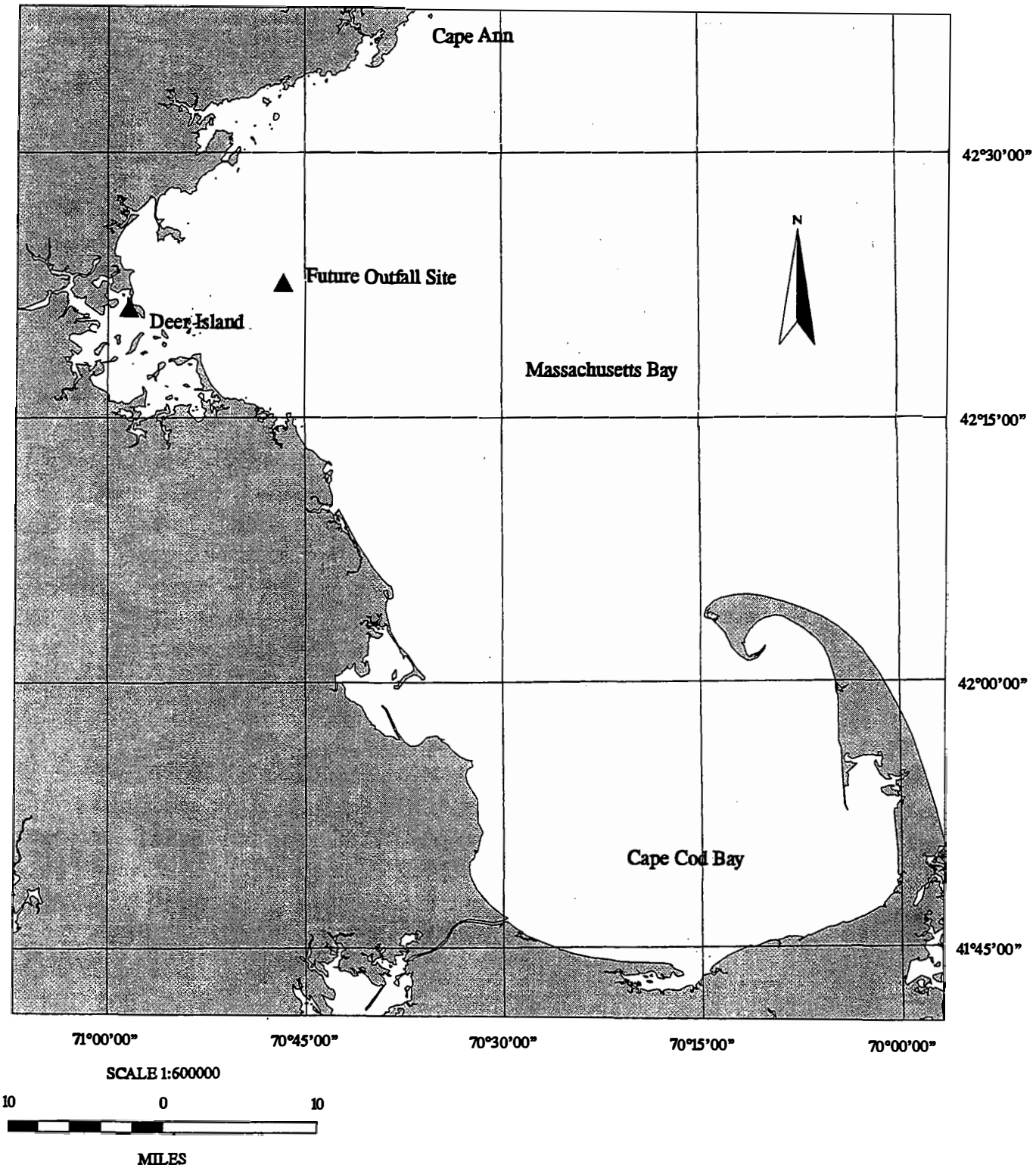

FIGURE 1-1

1995 Biomonitoring Program Study Area

2.0 METHODS

This section provides an overview of the methods and protocols used in the three surveys. The definitive descriptions of the methods are contained in: *Combined Work/Quality Assurance Project Plan (CW/QAPP) for the Fish and Shellfish Monitoring: 1995-97 (Fish and Shellfish Monitoring CW/QAPP)* (Mitchell et al., 1995).

2.1 Winter Flounder Monitoring

Winter flounder (*P. americanus*) were obtained from five locations in Boston Harbor and the Bays for gross examination, histology, aging, and chemical analyses of liver and filet tissue. Figure 2-1 depicts the five sampling locations for winter flounder monitoring during 1995. It should be noted that while fish were collected at all five sites, tissues were analyzed for chemicals of concern only for samples from Deer Island Flats, the Future Outfall Site, and Eastern Cape Cod Bay. Tissues from the remaining two sites, Broad Sound and Nantasket Beach, were labeled, frozen, and archived for potential future use.

2.1.1 Stations and Sampling

Fish were collected at Deer Island Flats (i.e., Boston Harbor), off Nantasket Beach, Broad Sound, the Future effluent outfall site (Massachusetts Bay), and Eastern Cape Cod Bay (Figure 2-1, Stations 1-5) from April 10 through April 12, 1995. The F/V *Odessa*, captained by William Crossen, was used to conduct otter trawls at each of the five sites. Table 2-1 presents the trawl data for the flounder surveys including stations, positions, etc. Tows were conducted until 50 sexually mature (4- to 5-year old) winter flounder of 30 cm or more in length were obtained. Only 16 were obtained at Deer Island Flats during the collection survey in April, however. Therefore, 34 fish from this area were obtained during "Fish Day 3" (May 6, 1995), a study of harbor flounder coordinated by the new England Aquarium, involving area scientists and recreational fisherman, to supplement the earlier catch.

Fifteen of the 50 sexually mature fish collected at each station were selected randomly, identified with a fin clip, and held in static, aerated, ambient seawater prior to dissection for chemical and histopathological analysis at a clean lab. Fish from Stations 1, 2, 4 and 5 were processed at the Northeastern University Marine Station at Nahant, while those from Station 3 were processed on board (see Section 2.1.3). The remaining 35 fish were killed on board by cervical section and used for histological processing (see Section 2.1.4).

All 50 specimens from each site were visually examined and external conditions such as for fin rot and other abnormalities (e.g., lymphocystis) were noted. Each fish was weighed and measured for total and standard lengths.

Fish were also assessed for sexual maturity. After each fish was killed by cervical section, an oval incision was made in the ventral body wall overlying the liver and anterior ventral gonad. The gonads were examined, and their color and sex recorded. Sexual maturity was determined on the basis of the following criteria: gonads are blue-gray in immature fish; pink and elongated in mature females; and white and triangular in mature males. This examination was conducted in the clean lab for the 15 fish held in the live tank, and in the field for the remaining 35 fish.

2.1.2 Age Determination

Scale samples were collected from each fish for age determination. Mucus, debris, and epidermis was removed from the dorsum of the caudal peduncle prior to obtaining scale samples. Scales were collected by wiping in the direction of the tail with a blunt-edged knife by applying quick, firm, scraping motions in the direction of the head. Scales were scraped into envelopes labelled for each individual fish. Age was determined by enumerating the annuli on a scale sample taken from each fish.

2.1.3 Dissection of Fish

The fifteen fish held on board in the live tank at each station were delivered to the clean lab established for the project at the Northeastern University Marine Station at Nahant for dissection (fish from Stations 1,2,4 and 5 were processed here, while those from Station 3 were processed on board). Each fish was assigned a sample identification number indicating the date and site of collection. All fish were killed by cervical section. Fish were dissected in a laminar flow hood, using a precleaned titanium knife (precleaning consisted of rinsing with 10% HCl, Milli-Q water, acetone, dichloromethane, and hexane). The filets (muscle tissue) were removed from the flounder, and the skin removed from the filet.

2.1.4 Histological Processing

Livers were removed from the 50 fish collected at each site. Each liver was examined for color and gross abnormalities, placed in a separate clearly labeled sample container, and preserved in 10% buffered formalin for histological processing.

Histological processing was conducted in a clean room. Livers were removed from the preservative after a minimum of 24 hours and rinsed overnight in running tap water. Sections approximately 4-6 mm thick were cut transversely from 3 separate areas of the liver with a precleaned scalpel, embedded in paraffin, and placed in labelled cassettes. Two 5 μm sections were cut from each of these three sections, and

stained with hematoxylin and eosin according to standard methods (Hillman *et al.*, 1994). The remaining liver tissue from the 15 flounder designated for chemical analysis was retained. The liver tissue from the remaining 35 fish from each station designated for histological analysis only were placed in labeled containers, frozen, and archived.

2.1.5 Histological Analysis

The 250 liver samples set aside for histological analysis were initially examined for the prevalence and severity of the following lesions, which were described in Moore, 1991 and Hillman *et al.*, 1994:

- Vacuolation (including centrotubular, tubular hydropic, and focal hydropic);
- Macrophage aggregation;
- Biliary duct proliferation; and
- Neoplasia.

Slides were prepared for each fish, including three liver sections per slide. Each slide was examined under bright-field illumination at 25x, 100x, and 200x. The severity of each lesion was rated on a scale from 0 to 4, as follows:

- 0 = absent
- 1 = minor
- 2 = moderate
- 3 = severe
- 4 = extreme.

For each lesion and each fish, a histopathological index was calculated as the mean of scores from three liver slices on one slide. A lesion index was also calculated for each site, based on the mean scores for each particular lesion at each site.

2.1.6 Tissue Processing

Chemical analyses were performed on composite samples of flounder filet and liver tissues obtained from the 15 fish set aside in the live well at each of 3 sites: Deer Island Flats, the future outfall site, and Eastern Cape Cod Bay (samples from off Nantasket Beach and Broad Sound were labelled, frozen and archived). Liver and filet tissues from individual fish were homogenized separately; the liquid from each sample was stored with the homogenate. Using a random number generator, three composites were made of 5 randomly-selected fish per composite; liver and filet composites were generated from the same 5 fish. Filet composites contained equal portions of dorsal and caudal tissue; equal amounts of liver (or filet) were used from each of the five fish. Wet weights were recorded for each liver and filet sample. After

compositing, the tissues were re-homogenized prior to shipment for chemical analysis. Remaining liver and filet tissues were placed in labeled containers, frozen, and archived.

2.1.7 Chemical Analyses

Chemistry analyses performed on composited flounder tissues are listed in Table 2-2. Flounder filets were analyzed for mercury, polychlorinated biphenyls (PCBs), chlorinated pesticides, and lipids. Flounder livers were analyzed for trace metals, PCBs, polycyclic aromatic hydrocarbons (PAHs), chlorinated pesticides, and lipids. The composited samples were split for organic and metals analyses at separate laboratories.

2.1.7.1 Organic Compound Analyses

Tissue samples were serially extracted for PAH, chlorinated pesticides, and PCBs. An aliquot of homogenized tissue was extracted with dichloromethane (DCM) and sodium sulfate using a Tekmar tissuemizer. An aliquot of the original sample was retained for dry weight determination. The sample was weighed in a Teflon extraction jar, spiked with the appropriate surrogate internal standards. Sodium sulfate and solvent were added, the samples were macerated for 2 minutes, and centrifuged. The solvent extract was decanted into an Erlenmeyer flask. After each extraction (two homogenizations and a third shake by hand), the centrifuged solvent was combined in the flask. A 10-ml aliquot of the combined extracts was removed for lipid weight determination, and sodium sulfate was added to the extract remaining in the flask. After approximately 30 minutes, the contents of the flask was processed through an alumina column. The elutriate from the column was concentrated to 900 µl (via a Kuderna-Danish apparatus and nitrogen evaporation techniques). The concentrated extract was further cleaned using a high performance liquid chromatographic (HPLC) gel-permeation technique (which removes common contaminants that interfere with the instrument, including lipids). The post-HPLC extract was concentrated to approximately 500 µl under nitrogen gas, and the recovery internal standards were added to quantify extraction efficiency. The tissue final extract was split for analysis, with one half remaining in DCM for PAH analysis, and one half solvent-exchanged with isooctane for PCB/pesticide analysis.

It should be noted that co-planar PCB congeners 77 and 126 were reported in the MWRA Fish and Shellfish Monitoring Program data for 1995. The methods used to analyze for these two compounds did not include a carbon column clean-up technique which would isolate only co-planar PCB congeners and require a separate GC/ECD analysis. However, the methods used were comparable to those used throughout the history of the MWRA program by other contractors. Thus, the data generated for these compounds should be consistent with historical MWRA data reported for these compounds.

As noted in the letter to Mr. Ken Keay by Dr. Jack Schwartz (dated 12/26/96), other PCB congeners have the potential to co-elute with congeners 77 and 126. While the dual column confirmation method utilized

by the laboratory contractor in 1995 served to minimize this issue, there is greater uncertainty associated with the PCB congener 77 and 126 results due to the potential interferences from other PCB congeners.

Sample extracts were analyzed for PAH compounds in the selected-ion monitoring (SIM) mode by gas chromatography/mass spectrometry (GC/MS) using a modification of EPA Method 8270. The modifications include operating the mass spectrometer in the SIM mode, and tuning the mass spectrometer with PFTBA. Pesticides and PCB congeners were analyzed and quantified using gas chromatography/electron capture detection (GC/ECD) using EPA Method 8080, modified to include additional analytes and a second column for qualitative confirmation.

2.1.7.2 Metals Analyses

Tissues were analyzed for the metals indicated in Table 2-2. Approximately 0.75 g of wet tissue (0.3 g dry tissue) were weighed into the Teflon inserts of Parr Bombs. The sample was turned into a slurry upon the addition of 3 ml of superpure aqua-regia. The bombs were sealed and heated for three minutes. The bombs were then cooled, another 2 ml of superpure aqua-regia added, and heated for two minutes. The digestate was diluted to a final volume of 50 ml and transferred to a precleaned 125 ml polyethylene bottle. The digestate was analyzed for silver, cadmium, chromium, nickel, copper, lead, and zinc by inductively coupled plasma-mass spectrometry (ICP-MS). In order to achieve the low detection limits required for mercury analysis, the EPA method for total recoverable mercury (EPA Method 245.1) was modified. Mercury was analyzed using a flow injection cold vapor technique with atomic absorption detection following preconcentration on gold amalgam (McIntosh, 1993).

2.1.8 Data Reduction and Statistical Analyses

Data reduction was conducted as described in the Fish and Shellfish Monitoring CW/QAPP (Mitchell *et al.*, 1995). Histopathological indices and prevalence of lesions were compared between classes of fish by differences in station, age, sex, and length. Chemical constituents were presented graphically. Analyses of variance were used to compare lesions from site to site and from interannual data.

Histopathological observations of the livers of the winter flounder from all sites were conducted and, where possible, comparisons of the results with those of previous years were made. Possible relationships between observed lesions and contaminant body burdens were also investigated.

In addition to reporting the prevalence and lesion index of hydropic vacuolation, historical data has included several other lesions, including macrophage aggregates, biliary proliferation, neoplasia, and a lesion unreported before 1993, referred to as "balloon hepatocytes".

The levels of contaminants measured in edible tissues were compared to Food and Drug Administration (FDA) Action Levels (U.S. EPA, 1989) for those contaminants.

2.1.9 Deviations from CW/QAPP

At Deer Island Flats, only 16 fish were collected during the April survey due to fish availability limitations. The remaining 34 fish from this station were purchased from participants of "Fish Day 3," an angling event sponsored by the New England Aquarium held every three years. This event was held on May 6, 1995 and provided sufficient documentation of the fish collection to allow pooling of these samples with the previous fish.

Chemistry samples from fish of Broad Sound were taken aboard the vessel without the use of a laminar flow hood. The CW/QAPP stated that the samples were to be collected under a hood aboard the vessel, at dockside. This proved to be impracticable in the field. Therefore, fish from all other stations were processed at a clean dissection bench established at the Nahant Marine Lab.

The balance used at sea for weighing the fish collected for histology data only was a Normark digital fish scale, with an accuracy of ±60 g. On arrival at the first station (Broad Sound), it was discovered that the balances battery had discharged during transit from Woods Hole. Therefore, the weights for the Broad Sound fish could not be measured directly. Instead, fish weight was estimated using a length-weight regression from all of the other fish collected during the 1995 survey.

2.2 Northern Lobster Monitoring

Northern lobster (*H. americanus*) were collected from three locations for gross examination and chemical analyses of hepatopancreas and edible (tail) tissue. Figure 2-1 presents the sampling locations for the 1995 survey.

2.2.1 Stations and Sampling

Lobster were collected at three stations for gross examination and analysis of chemical contaminants in the tail and hepatopancreas. Sampled stations included Deer Island Flats (Boston Harbor), the future effluent outfall site (Massachusetts Bay), and Eastern Cape Cod Bay (Figure 2-1, Stations 1, 4, and 5). The R/V *QT*, captained by Phil Romano, was used to set and collect 20 lobster traps at each of the three stations. Table 2-3 contains the trap data for the 1995 surveys. Traps were set for 3 days to collect 15 legal-size non-berried individuals; however, few lobster were caught (see Section 3.2.1; Mounce and Mitchell, 1995), and the desired number of individuals was supplemented through the purchase of lobsters from commercial lobstermen observed to have lobster pots in the area of the designated sampling sites.

2.2.2 Size and Sex Determination

Immediately upon removal from the traps, lobsters were measured with a commercial lobster gauge to determine if the lobster met the legal size limit. Lobsters which did not meet the legal size limit were enumerated and immediately returned to the sea. Lobsters that met the legal size limit were banded with a claw band color-coded per site: green for the future outfall site; blue and white bands for Deer Island Flats; and yellow bands for Eastern Cape Cod Bay. Lobsters were then measured for carapace length and width, and weighed on a Sartorius balance (accuracy ± 1 gram).

In addition to length, width, and weight measurements, the sex of individual lobsters was recorded on the field data sheets. External conditions were noted, including the presence or absence of black gill disease, shell erosion, parasites, external tumors, and other external abnormalities. Upon completion of the field observations, lobsters were assigned a specific identification number, placed in labelled, doubled plastic bags, and preserved on dry ice. Lobsters were transported to the analytical laboratory and kept frozen until dissected and the tissue composited.

2.2.3 Dissection of Lobster

Because clean conditions could not be satisfied in the field, lobster dissection was conducted in the laboratory setting. Each lobster was assigned a unique laboratory identification number. Lobsters were thawed prior to dissection. The hepatopancreas and edible meat (tail only) was removed from each lobster via ventral incisions in the carapace and tail.

2.2.4 Tissue Processing

Once the hepatopancreas and edible tissues were removed from the carapace, each was homogenized separately. Composite samples were generated by homogenizing the edible tissue of five randomly-selected lobsters, with an equal amount (±5% by weight) of tissue from each lobster. Equal volumes of hepatopancreas homogenate (±5%) from the same five lobsters were composited and rehomogenized prior to chemical analysis. Three composite samples of hepatopancreas and edible tissue were analyzed per station.

2.2.5 Chemical Analyses

The chemical analyses conducted on lobster hepatopancreas and edible tissue samples followed the same procedures used for analysis of flounder tissue described in Section 2.1.7. A list of the chemical analytes is given in Table 2-2. Lobster hepatopancreas tissues were analyzed for trace metals and organic compounds. Edible tissues were analyzed for mercury and organic compounds.

2.2.6 Data Reduction and Statistical Analyses

Data reduction was conducted as described in the Fish and Shellfish Monitoring CW/QAPP (Mitchell *et al.*, 1995). Spatial and temporal trends of contaminants in composites of edible lobster tissue and hepatopancreas tissue were evaluated through available data from 1985 through 1995. Analyses of variance were used to compare chemical constituents from site to site and from 1994 to 1995 data. Comparisons were made to the FDA Legal Limits and other appropriate levels of regulatory concern.

2.2.7 Deviations From CW/QAPP

Only 3 legal-sized non-berried lobsters were caught in the 1995 survey. The CW/QAPP suggests that the traps be set repeatedly until the 15 lobsters are obtained. Due to the low level of trapping success, the remaining 42 were purchased from lobstermen with traps either adjacent to ENSR's (Eastern Cape Cod Bay, Deer Island Flats) or within a 2 km radius (Future Outfall Site). This procedure is in accordance with contingency provisions contained in the *Fish and Shellfish CW/QAPP* (Mitchell *et al.*, 1995).

Lobsters were not killed and dissected in the field, as the requisite clean conditions could not be guaranteed. Lobsters were immediately assessed for external conditions, placed on dry ice, and delivered to the laboratory frozen, under chain-of-custody for later thawing and analysis.

2.3 Mussel Bioaccumulation Monitoring

Mussel condition (growth and reproduction) is used to assess the current health of mussel populations in the Boston Harbor and Bay areas prior to the relocation of the existing effluent discharge and to provide a common yardstick to compare biomonitoring results to other areas covered by the National Oceanographic and Atmospheric Administration's (NOAA) National Status and Trends (NST) program. Blue mussels (*M. edulis*) were obtained from a reference station, deployed at 3 stations, and retrieved for determination of biological contamination. The station locations used in the 1995 survey are shown in Figure 2-2. Short-term soft tissue accumulation of station-specific contaminants was monitored over a period ranging from 40 to 60 days. Specimens were collected from reference sites and deployed during a June survey, and the mussels were collected after incubation in August, 1995.

2.3.1 Stations and Reference Areas

Mussels were deployed and retrieved at three sites, including a baseline reference site. Figure 2-1 illustrates the sampling locations in Boston Harbor and Massachusetts Bay. Table 3-12 presents the 1995 mussel survey sampling design.

Mussels were deployed on June 23, 1995 from the R/V *Profile* in replicate arrays at the sites described below:

- Approximately 75 meters east of Deer Island Light, within the zone of initial dilution (ZID) of the Deer Island effluent discharge. This site was selected for monitoring the potential bioaccumulation associated with the Deer Island effluent discharge;

- The stern of the *Discovery*, anchored at the New England Aquarium in Boston Inner Harbor. This site serves as a "dirty" control, allowing the evaluation of ambient contamination within Boston Inner Harbor; and

- The Large Navigation Buoy (LNB) located approximately one nautical mile(NM) south of the project MWRA offshore discharge installation. This site is monitored for predischarge baseline conditions.

2.3.2 Mussel Collection

Approximately 950 mussels were collected from the University of Massachusetts Research Station at Hodgkins Cove, Gloucester, MA on June 22, 1995. Mussels from this location have been shown to be relatively free of contamination, and have been used in previous bioaccumulation studies conducted throughout the Massachusetts coastline (Camp, Dresser and McKee, 1988; MRI, 1989; Downey and Young, 1992; Downey *et al.*, 1993; Downey, 1994a and 1994b). However, concerns have been raised relative to the concentrations of mercury in Gloucester mussels, therefore additional mussels were collected at an alternate control site in Sandwich, MA. Accordingly, only tissue derived from Sandwich-harvested mussels was used for mercury analyses. All other analyses were performed on Gloucester-harvested mussels.

Mussels were harvested during low tide. Only mussels between 55 and 70 mm in total length were used. Total length was recorded on a subset of 200 of the 950 mussels collected in Gloucester. Approximately 50 Gloucester mussels were randomly distributed to each of 19 plastic cages (40 mussels for cages deployed to the Large Navigation Buoy site), and suspended overnight in seawater along the seawall adjacent to the UMass Research Station in Gloucester. Approximately 40 Sandwich mussels were distributed to separate cages (50 mussels for the LNB site) and suspended overnight in seawater at the Sandwich collection site. In addition, approximately 85 Gloucester mussels and 35 Sandwich mussels were sent on ice to the Aquatec facility on June 23, 1995 for initial biological analyses, including total length, sex and sexual maturity, and tissue weights. Samples of Gloucester and Sandwich mussels were stored frozen for chemical analyses.

2.3.3 Mussel Deployment

At the Deer Island Light, three arrays were deployed on June 23, 1995. Each deployment array consisted on two replicate cages containing approximately 50 Gloucester-harvested mussels per cage (= 100 Gloucester mussels) and one cage containing 40 Sandwich-harvested mussels for a total of approximately 140 mussels per array. The cages were attached to polypropylene line with nylon cable ties. The arrays were positioned within the water column by steel mooring weights and a styrofoam subsurface buoy. The subsurface buoy for each array was located approximately 3 meters above the bottom, and the cages were fixed approximately 1 meter below the buoy (or 2 meters above the bottom). At Deer Island Light, the arrays were deployed in approximately 4 to 6 meters mean low water (MLW), approximately 75 meters east of the light. Deployment positions were confirmed with an on-board Loran-C instrument, and latitude and longitude measurements were recorded.

Two arrays were deployed at the *Discovery* on June 23, 1995. Each deployment array consisted of two replicate cages containing approximately 50 Gloucester-harvested mussels per cage (= 100 Gloucester mussels) and one cage containing 40 Sandwich-harvested mussels for a total of approximately 140 mussels per array. The arrays were suspended on a nylon line off the stern of the vessel. The four cages were attached in pairs (0.2 meters between pairs); each pair was considered to be a deployment array. The arrays were anchored approximately 2 to 2½ meters above the bottom in approximately 7 to 9 meters of water.

Three arrays were deployed at the LNB station on June 23, 1995. Each array consisted of three cages, with a total of approximately 140 Gloucester and Sandwich mussels, as previously described. The arrays were deployed near the LNB with a mooring and suspension system similar to that deployed at the Deer Island Light. In addition, a surface buoy was attached to the subsurface buoy, which was deployed approximately 13 meters below the surface.

2.3.4 Mussel Retrieval

On August 2, 1995, mid-deployment period retrieval activities were conducted. However, mussels were collected only at the *Discovery*, for an exposure time of 40 days. Only one array remained at the Deer Island Light and the LNB; these were left in place for the full deployment period. The mussels retrieved at the *Discovery* at this time were checked in the field for gross abnormalities and apparent survival, frozen, and archived (they were not used during the remainder of the study). No abnormalities were observed and survival was estimated to be 98 percent. The rages recovered at the *Discovery* were moderately to extensively covered with sea squirts, resulting in some occlusion of spaces between the bars of the cage.

Mussels were retrieved on August 11, 1995, for a total deployment period of 49 days (see Section 3.3.1 for details). Mussels were retrieved from the *Discovery* and the Deer Island Light; no arrays were remaining at the LNB station. At the *Discovery*, one array was recovered, containing 40 Sandwich mussels and 101 Gloucester mussels. At the Deer Island Light, one array was recovered, containing 104 Gloucester mussels and 40 Sandwich mussels. Random subsamples of mussels from each station were set aside for biological and chemical analyses. Mussels were stored in separate labeled plastic bags and preserved on ice for transport. All mussels were transported on ice to Inchcape/Aquatec on August 11, 1995, and stored frozen (for chemical analyses) or refrigerated (for biological analyses).

Deer Island cages retrieved after 49 days were tangled in a large brown macroalgae, *Laminaria*, and both cages and mussels were covered with a fine silt-like material. Barnacles present on several of the mussels (less than 10 percent) were believed to be predominantly carryover organisms resulting from incomplete removal during the initial harvest at Hodgkins Cove. Seed mussels and a few amphipods were also observed inside the cages.

The *Discovery* cages recovered after 49 days exhibited extensive fouling, predominantly by sea squirts, which covered the cage bars. A silty material was also found on surfaces not occupied by sea squirts, including mussel shells. Spaces between mussels also commonly contained large quantities of this silt-like material, which had accumulated in layers up to several millimeters thick on mussel shells. Numerous crabs and amphipods were also present.

2.3.5 Determination of Biological Condition

Of the mussels collected in Gloucester, the total shell lengths (in mm from umbo to distal gape) were recorded for a subsample of 200 mussels. Measurements were made in the field with Vernier calipers to 0.1 mm to obtain an average size estimate for the entire sample population of approximately 950 mussels. Random subsamples of 30 mussels harvested at Gloucester were selected from the predeployment mussels and from the mussels retrieved at the full deployment period from Deer Island Light and the *Discovery*. Mussels from this subsample were processed for biological analyses. These included observations of viability at recovery, shell weight, shell length, total tissue weight (both wet and dry), and gonadal tissue weight (wet and dry). Each mussel was opened by slicing the adductor muscles between the valves with a microtome blade. The gill tissue was drawn back to expose the gonad. A small aliquot of macerated gonadal tissue was transferred to a slide and examined under a compound microscope for sex and sexual maturity. Sexually mature males were identified by sperm motility and immature males by lack of sperm motility. Sexually mature females were identified by the presence of eggs of a diameter greater than 60 microns, while those with egg diameters less than 60 microns were considered to be sexually immature.

Approximately 300 mussels were collected from the alternate control site in Sandwich, MA. At the time of collection, the sizes of a subsample of 60 of these mussels were measured and recorded. Ten mussels

were retained for sex determinations and measurements of gonadal and non-gonadal wet weights. These alternate control mussels were harvested at Deer Island and from the *Discovery* after a 49-day deployment. Biological analyses were conducted as described above for the Gloucester mussels, although mussel dry weights were not measured on the alternate control mussels.

2.3.6 Tissue Processing

A random subsample of Gloucester derived mussels (50 mussels per station) were selected from pre-deployment mussels and from each of the two stations' 49-day deployment harvest. Replicate samples for chemical analysis were prepared as composites of ten mussels, for a total of 5 replicate composite samples per station. Each individual mussel was cleaned of attached material, all byssal threads removed, and all soft tissue including fluids placed directly into an amber 500-ml I-Chem Certified clean bottle. Mussel composite samples were prepared for chemical analyses by dissection of each of the 10 mussels using disposable Teflon-coated stainless steel blades rinsed with methanol and deionized water prior to use.

A random subsample of Sandwich derived mussels (25 mussels per station) were selected from pre-deployment mussels, Deer Island Light and *Discovery* full deployment mussels. These were prepared as described above for the Gloucester mussels, although composite samples consisted of 5 mussels per composite, for a total of 5 replicate composite samples per station.

2.3.7 Chemical Analyses

Table 2-2 summarizes the chemical analyses conducted on mussel tissues. Five composites of mussel soft tissue per station were analyzed for these parameters. Samples assigned to each specific pool were homogenized together prior to conducting analyses. Organic and inorganic analyses were conducted according to the procedures described for flounder or lobster chemical analyses (Section 2.1.7).

2.3.8 Data Reduction and Statistical Analyses

The extent of bioaccumulation of contaminants in blue mussels was evaluated, and compared to initial contaminant levels in control mussels. Evaluation focused on spatial and temporal trends in contaminant accumulation. Data were compared to mussel body burdens of contaminants in other studies, including the NOAA Status and Trends Mussel Watch monitoring program, and other available studies. The relationship(s) between biological condition and tissue contamination was also assessed.

2.3.9 Deviations from the CW/QAPP

In 1995, the major deviation from the CW/QAPP was the length of the mussell incubation period and the loss of the arrays at the Future Outfall Site. The 60-day harvest occurred after 49 days of deployment at Deer Island and the *Discovery* (with the concurrence of the MWAA Task manager) due to concerns about vulnerability of the arrays to boat traffic and/or vandalism. The loss of the caged arrays at the Future Outfall Site prevented collection of data at this location. Finally, the use of mussels collected from Sandwich to evaluate mercury bioaccumulation was a supplementary task not described by the CW/QAPP.

2.4 General Data Treatment and Reduction

This section describes data reduction performed on 1995 Fish and Shellfish data as part of the 1995 MWRA Harbor and Outfall Monitoring Project. Samples were analyzed for trace metals, polynuclear aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs) and/or pesticides. Morphological data were collected for all samples and histopathology data were collected for flounder liver samples. Data were presented to ENSR from the analytical laboratories conducting the various analyses in both paper and electronic format.

The data received were not in the required MWRA HOM Oracle® Database format and it was necessary to reformat all data, including reassignment of Sample IDs according to the MWRA Fish and Shellfish CW/QAPP. All operations were performed in Microsoft Excel®. This problem was addressed and rectified for future data. All data were entered into the Database in the required format. Laboratory replicate samples (samples that were diluted and run a second time if instrument detection levels were exceeded) were combined and treated as one sample. Duplicate samples were treated as a separate entry in the Database. All analytical data entered into the Database were in dry weight units.

An audit was conducted on the 1995 Fish and Shellfish Database before finalizing results for this document. A random statistical procedure (Hoover and Baldwin, 1984) that ensures 99 percent accuracy with 95 percent confidence was used. During this procedure, a subset of the total data is checked for accuracy and then compared to a pre-established allowable error rate. The data were found to be acceptable.

For this document all data were retrieved from the MWRA Database for the years 1992-1995. Data from other years were obtained from reports as referenced in individual sections. Flounder and Lobster data from the Database were summarized into averages per station per analyte. All Non Detects were treated as zero. Duplicate samples were averaged and the result treated as an individual sample. Standard error was calculated by arithmetic formula in Microsoft Excel® when there were enough results to support analysis. Mussel data were summarized using the lowest detection limit value for all Non Detects. Wet weight data were calculated by multiplying the percentage dry weight of a sample with the analytical result

for that sample. Mussel morphology data were summarized by averaging the biological data reported for each group of mussels (i.e. data from mussels harvested from Gloucester and deployed at Deer Island were averaged together).

TABLE 2-1

Winter Flounder Survey Data

Station[1]	Trawl Date	Start Time	Latitude	Longitude	End Time	Latitude	Longitude	Bottom Time (min.)	# Fish >300 mm
BS	4/10/95	0930	42°24.508'	70°57.636'	1011	42°24.381'	70°57.446'	41	9
		1020	42°24.579'	70°57.601'	1120	42°24.762'	70°57.547'	60	11
		1158	42°26.725'	70°53.572'	1312	42°26.521'	70°52.659'	74	30
DIF	4/10/95	1420	42°20.751'	70°58.000'	1515	42°20.583'	70°58.385'	55	9
		1559	42°20.918'	70°58.155'	1619	42°20.851'	70°58.638'	20	0
	4/11/95	0849	42°21.850'	70°59.416'	0856	42°21.847'	70°59.214'	5	0
		0905	42°21.415'	70°58.755'	0932	42°20.934'	70°58.803'	27	2
		0948	42°20.830'	70°58.923'	1010	42°21.142'	70°58.272'	22	1
		1019	42°28.918'	70°58.075'	1048	42°21.346'	70°58.632'	29	4
NB	4/11/95	1200	42°17.459'	70°51.752'	1257	42°17.457'	70°51.428'	57	49
		1314	42°17.538'	70°51.245'	1327	42°17.183'	70°51.617'	10	10
FOS	4/11/95	1414	42°23.131'	70°49.378'	1545	42°23.601'	70°49.713'	91	55
CCB	4/12/95	1012	41°57.776'	70°07.156'	1120	41°55.734'	70°07.582'	68	34
		1131	41°55.435'	70°07.501'	1220	41°57.151'	70°07.853'	49	25

[1] Station Locations Include:
BS = Broad Sound
DIF = Deer Island Flats
NB = Nantasket Beach
FOS = Future Outfall Site
CCB = Cape Cod Bay

TABLE 2-2

Chemistry Analyses for Fish and Shellfish Monitoring

Organism	Number/ Type of Samples	Parameters
Flounder	9/fillet*	Mercury PCB Chlorinated pesticides Lipids
Flounder	9/liver*	Trace metals PCB PAH Chlorinated pesticides Lipids
Lobster	9/meat	Mercury PCB Chlorinated pesticides Lipids
Lobster	9/hepatopancreas	Trace metals PCB PAH Chlorinated pesticides Lipids
Mussel	23/soft tissue	Mercury Lead PCB PAH Chlorinated pesticides Lipids
*15 samples during 1996 survey.		

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TABLE 2-3

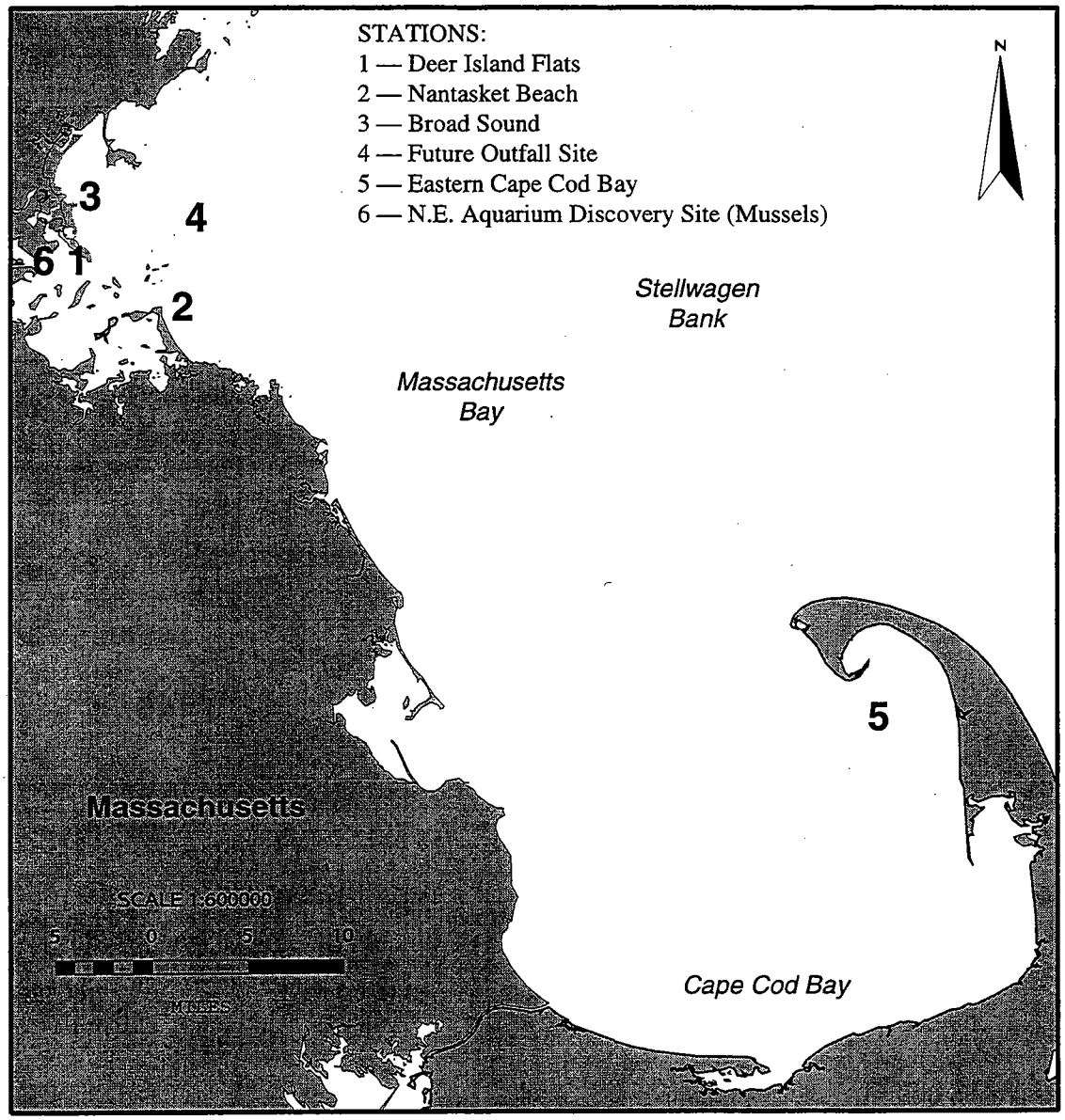
Lobster Survey Collection Data

Station[1]	Date	Start Time	First Trap		Last Trap[2]		End Time
			Latitude	Longitude	Latitude	Longitude	
CCB	7/18/95	1137	41°55.89'	70°05.90'	41°57.17'	70°06.10'	1220
	7/21/95	1103					1200
DIF	7/25/95	0930	42°20.72'	70°58.10'	42°20.70'	70°58.46'	0946
	7/28/95	1040					1059
FOS	7/25/95	1101	42°23.11'	70°49.31'	42°23.07'	70°49.71'	1113
	7/28/95	0905					0939

[1]Station Locations Include:

CCB = Eastern Cape Cod Bay
DIF = Deer Island Flats
FOS = Future Outfall Site

[2]Twenty traps were set at each station, with approximately 100 feet between traps.



M960345

FIGURE 2-1
Winter Flounder, Lobster and Mussel Sampling Locations

3.0 RESULTS AND DISCUSSIONS

The results of the three 1995 biomonitoring surveys are reported in this section. The results of the Flounder Survey are given in Section 3.1; the results of the Lobster Survey are given in Section 3.2; and the results of the Mussel Bioaccumulation Survey are presented in Section 3.3.

3.1 Winter Flounder

3.1.1 Fish Collected

Fifty winter flounder, each a minimum 30 cm in length, were collected between April 10 and 12, 1995 at four stations in the study area: Nantasket Beach, Broad Sound, the Future Outfall Site, and Eastern Cape Cod Bay. For Deer Island Flats, 16 fish were obtained in the original survey, and supplemented with 34 fish collected during "Fish Day 3" (see Section 2.1.9 for details).

The catch per unit effort (CPU), defined as the number of fish obtained per minute of bottom trawling time, was highest at Nantasket Beach in 1995, as it had been during the last three years (Table 3-1). The lowest CPU of 1995 was observed at Deer Island Flats, as is true for many of the years of the fish biomonitoring study. The CPU is determined in order to make a relative comparison of catch within the study between years.

3.1.2 Age/Length Parameters

The physical characteristics (i.e., mean age, length, weight) of the winter founder collected in 1995 are given in Table 3-2, and the interannual trends at the five stations are depicted in Figures 3-1a through 3-1e. The total length and weights were similar for all stations in 1995; there was less than 10% difference between average station values. The average age at Deer Island Flats (3-6 years) was significantly lower than those at three other stations (Nantasket Beach, Future Outfall Site, Eastern Cape Cod Bay). As shown in Figures 3-1a and 3-1b, the mean ages of Deer Island Flats and Broad Sound flounder declined slightly in 1995 as compared to previous years. In general, the mean age has been decreasing at these stations since 1989. At the other stations, little change in the mean age has been observed throughout the study (Figures 3-1c,d,e). Mean fish length varied little between years at most stations and reflects the minimum size requirement (i.e., minimum 30 cm required for the study).

3.1.3 External Condition

The physical characteristics and external conditions (i.e., fin erosion, gross abnormalities) of winter flounder collected in 1995 are presented as averages per station in Table 3-2. As described in Section 2.1.1, each of the individual winter flounder collected were assessed for external conditions, and rated on a scale of 0 to 4 (no units), with 0 indicating the absence of the condition, and 4 indicating extreme abnormalities (or erosion). As shown in Table 3-2, only a few fish at each station exhibited gross physical abnormalities. Fin erosion varied between station (0.2 to 0.5), with the Future Outfall Site being significantly lower than most other stations. This amount of fin erosion would be considered in the low end of the range (Murchelano, 1975) and, with regard to the DIF station, apparently well below the fin erosion observed (but not quantified) in the later 1980's (pers. comm. M. Moore).

3.1.4 Interstation Comparison of Lesion Prevalence

The prevalence of histological changes in winter flounder liver from the 1995 survey is shown in Table 3-3. Neoplasia was absent from all flounder with the exception of one individual from Broad Sound. Fish from Deer Island Flats and Broad Sound showed the greatest prevalence of the chemically-associated lesion hydropic vacuolation (i.e., focal, tubular, and centrotubular hydropic vacuolation). Other lesions that were recorded include macrophage aggregation, and biliary, hepatocytic and pancreatic lesions. These lesions did not show a trend that related to the apparent gradient of chemical contamination. They may be part of an undescribed series of pathological process or processes in these fish. The balloon cells appeared much as described previously (Hillman et al., 1994; Hillman and Peven, 1995). They are probably apoptotic cells (i.e., cells that are dying from any of a number of causes). The other interesting observation was the occasional presence of intrabiliary and subcapsular metazoan hepatic parasites. The potential effect of these parasites on fish health is not known.

3.1.5 Temporal Comparison of Lesion Prevalence

The incidence of toxic chemical-associated liver lesions in winter flounder from Deer Island Flats was noted by Murchelano and Wolke (1985) in a 1984 study. Annual monitoring of the lesions in winter flounder from the Harbor has been ongoing since 1987 (Moore, 1991; Moore *et al.*, 1992; Moore and Stegeman, 1993; Hillman *et al.*, 1994). Four additional locations in Massachusetts and Eastern Cape Cod Bays were added in 1991 (Moore *et al.*, 1992). These studies have provided an internally consistent baseline data set on winter flounder liver pathology for the Deer Island Flats and Future Outfall sites, in addition to other sites in the region. This has been critical because of the existing Deer Island outfall's apparent biological effects, such as the hepatocellular hydropic vacuolation and other toxicopathic lesions, and the need to understand and document the change in biological impact on this ecosystem of recent and projected changes in sewage management by the MWRA. These changes include cessation of ocean

dumping of sludge in December 1991, planned initiation of primary and secondary treatment, and the relocation of the effluent outfall to the future site, now scheduled to occur in 1998.

The rationale and necessary background information on the biology and toxicology of winter flounder have been reported previously (Moore *et al.*, 1992; Moore and Stegeman, 1993; Hillman *et al.*, 1994). In these previous studies, hydropic vacuolation in the liver of winter flounder was detectable at all stations sampled, but was substantially more prevalent at the contaminated near-urban sites. Moore (1991) has shown a close association between hydropic vacuolation and liver neoplasms in winter flounder, and Johnson *et al.* (1992) have demonstrated that hydropic vacuolation was closely correlated with a suite of chemical contaminants, particularly chlorinated hydrocarbons. Hydropic vacuolation can be regarded as a harbinger of neoplastic risk, given adequate duration and level of exposure to carcinogens. Hydropic vacuolation appears to be an irreversible change. This is in distinct contrast to hydropic degeneration. The irreversibility of the former was shown convincingly when flounder from Deer Island Flats were maintained for 5 months on clean water and clean food, with no reduction in vacuolation prevalence over that time (Moore et al., 1996). Therefore, given age-specific analysis and between-year consistency in histopathological interpretation, observation of the prevalence of hydropic vacuolation is an appropriate long-term monitoring parameter for the effects of benthic contaminants on winter flounder in the Boston Harbor/Massachusetts Bay area. Thus, hydropic vacuolation is one of the principal lesions emphasized in this report.

Neoplasm prevalence in winter flounder from Deer Island Flats has fallen from a persistently elevated level in the 1980's to undetectable levels in the last four years of this study. This trend has been accompanied by a reduction in the prevalence of the three stages of hydropic vacuolation. These temporal trends are summarized in Figures 3-1a through 3-1e. To investigate the potential confounding variables in this temporal trend, we correlated lesion prevalences with average age, total length, and year of sampling for Deer Island Flats winter flounder by calculating Spearman rank correlation coefficients (as shown in Table 3-4). These coefficients represent comparisons between variables: coefficients from -1 to 0 indicate high ranks of one variable occur with low ranks of another (or a negative correlation); 0 indicates no correlation; and 0 to +1.0 indicates high ranks of one variable occur with high ranks of another (or a positive correlation). Results of this analysis indicate that the year of collection was significantly correlated ($p<0.05$) with each lesion type, with lesion prevalence decreasing in more recent years. In other words, this is a statistically significant decrease in lesion prevalence over the study period 1987-1995.

3.1.6 Relationships Between Age, Length and Lesion Prevalence

As shown in Table 3-4, there was a highly significant relationship ($p<0.01$) between flounder length and centrotubular hydropic vacuolation (CHV). However, no statistically significant relationship was observed between CHV and flounder age. This lack of a significant relationship between age and vacuolation, in

contrast to length and vacuolation, results from the lack of age data in 1987 and 1988. The high disease prevalence in these two years drives the significant temporal trend seen for length vs. vacuolation. Significant relationships were shown between focal hydropic vacuolations and fish length. Neoplasia and tubular hydropic vacuolation correlated significantly with age, and all lesions correlated significantly with length. Thus, not surprisingly, chronic contaminant-associated disease in these fish increase with growth but in addition, it seems that the persistent trend of decreasing lesion prevalence in more recent years is of statistical significance (as shown in Figures 3-1a-e). These analyses attempt to discover whether recent reductions in mean age of the fish are sufficient to account for the reduction in lesion prevalence. Within the 1991 to 1995 time frame this may be true, but between 1987 and 1995 this seems insufficient reason given the above analyses.

To tease out underlying trends obscured by the lack of age data in 1987 and 1988, we made a retrospective cohort analysis, by back-projecting the age class of each fish on the basis of its age at sampling, for Deer Island Fish. As shown in Table 3-5, the prevalence of CHV in winter flounder of Deer Island Flats consistently increased with age, although across the entire study area, no significant relationship was observed between CHV and age. The average CHV was lowest in Deer Island flounder aged 3, and highest in fish aged 6 to 9 years old. Figure 3-2 further illustrates the relationship at the Deer Island Flats study area, in fish spawned from 1980 through 1992. The lines in the figure represent the prevalence of lesions in successive years for a specific age class year. As can be seen, there is only a general tendency for increase over years. One of the potential confounding factors for this type of analysis is the widely-varying number of samples collected between years.

3.1.7 Interstation Comparison of Tissue Contaminant Levels

The patterns of tissue contaminant levels (i.e., body burdens) were examined for winter flounder collected in the 1995 survey. The mean tissue contaminant levels for winter flounder are given in Table 3-6 as unit of mass contaminant per mass dry weight. The two tissue types of concern were edible tissue (filets) and liver tissue. Results for filets are presented in Figure 3-3, while those for flounder liver are shown in Figures 3-4 and 3-5. The associated lines on the bar graphs represent one standard error.

3.1.7.1 Edible Tissue

The levels of target analytes (total DDT, dieldrin, total chlordane, total PCBs, and mercury) in winter flounder filets from the three 1995 survey sites (Deer Island Flats, Future Outfall Site, and Eastern Cape Cod Bay) are shown in Figure 3-3. Comparisons of the 1995 mean concentrations of organic compounds in filets across the study area indicate that chlordane, dieldrin and PCBs were lowest in Eastern Cape Cod Bay flounder, while DDT was lowest at the Future Outfall Site. Deer Island Flats flounder filets consistently contained the highest concentrations of these four organics. Total PCB concentrations were approximately three times (3x) higher at Deer Island Flats than at the Future Outfall Site. In 1995,

mercury was highest in Deer Island Flats filets, while mean tissue concentrations of mercury were lowest at Eastern Cape Cod Bay.

3.1.7.2 Liver

The levels of organic target analytes (total DDT, dieldrin, total chlordane, total PCBs, total PAHs) and mercury in winter flounder liver from the 1995 survey are shown in Figure 3-4. As with the edible tissues, an interstation comparison of concentrations of organic compounds in flounder livers indicates that the highest mean tissue concentrations were consistently found at Deer Island Flats, while the lowest typically occurred in Eastern Cape Cod Bay flounder. The only exception was dieldrin, where tissue concentrations were below the method detection limit in all three liver composite samples from the Future Outfall Site.

The levels of inorganic target analytes (Ag, Cr, Cu, Ni, Pb, Zn) in winter flounder liver from the 1995 survey are shown in Figure 3-5. An interstation comparison of inorganics indicates that the distribution of metals did not follow the gradient of tissue burdens established by the organic contaminants. For example, copper, lead, silver and zinc were highest at the Future Outfall Site in 1995. Nickel was highest at the Eastern Cape Cod Bay station, while chromium was highest at Deer Island Flats. It is interesting to note that, even though the metals exhibit considerable interstation variability, that the tissue levels and trends are comparable to 1994 and earlier years (see below).

3.1.8 Temporal Comparison of Contaminant Levels

The temporal or interannual variation of tissue contaminant levels was examined. The pattern of temporal variation can be seen for flounder filet in Figure 3-3, while that for flounder liver is given in Figures 3-4 and 3-5. Each of these figures show data from 1992 to 1995 surveys, grouped according to sampling station. The associated line represents one standard error.

3.1.8.1 Edible Tissue

Annual tissue concentrations of organic compounds from 1992 to 1995 were analyzed in winter flounder of Deer Island Flats, the Future Outfall Site, and Eastern Cape Cod Bay (Figure 3-3). At Deer Island Flats, concentrations of total DDT and dieldrin were within the range of previously recorded values (1992 through 1994 studies); total chlordane was lowest in 1995 for the four-year period, while total PCBs were highest. At the Future Outfall Site, chlordane in 1995 was the lowest of record, while tissue concentrations of DDT, dieldrin and PCBs all fell within the 1992 to 1994 range. At Eastern Cape Cod Bay, levels of DDT and PCBs in 1995 were the highest of the record period, dieldrin was the lowest of record, and chlordane was within the 1992 to 1994 range.

Mercury was assessed in flounder filets collected in 1995 from these three stations. Although tissues of Deer Island Flats and Eastern Cape Cod Bay were within the range of previously observed values, the concentration of mercury at the Future Outfall Site in 1995 was the lowest.

3.1.8.2 Liver

Annual tissue concentrations of organic compounds in flounder livers from 1992 to 1995 were measured for Deer Island Flats, the Future Outfall Site, and Eastern Cape Cod Bay winter flounder (Figures 3-4, 3-5). Mean concentrations of total DDT and total PCBs were higher in 1995 in flounder sampled throughout the study area than in any of the previous years (1992 through 1994), while the mean concentration of total PAHs was lower than any measured during 1993 through 1994. [Note: the 1992 flounder liver PAH values are considered suspect due to potential to contamination introduced at the time of dissection and were deleted from consideration.] The mean concentration of total chlordane was also higher in 1995 than previous years for Deer Island Flats and Eastern Cape Cod Bay flounder, while the chlordane concentration at the Future Outfall Site was lower in 1995 than in 1994. At Deer Island Flats, dieldrin concentrations were higher than in previous years; at the Future Outfall Site, dieldrin was non-detectable for the first time, and at Eastern Cape Cod Bay, the concentration increased somewhat.

Tissue concentrations of inorganic metals in livers were also measured at these three stations in 1995. At Deer Island Flats, mean flounder liver concentrations of lead, nickel, silver, chromium, and mercury followed a continuing trend of decreasing annual levels. Zinc concentrations were slightly lower than in 1994, but within the previously observed range. Copper concentrations were slightly elevated in comparison to the 1994 mean value. At the Future Outfall Site, nickel levels in 1995 were also the lowest of record. Silver and zinc concentrations varied little-to-none; lead and mercury were lower than observed in 1994; and copper concentrations were slightly elevated over the 1994 value. At Eastern Cape Cod Bay, flounder liver mean concentrations increased slightly over the 1994 mean value for lead, nickel, and mercury. The tissue means of silver, chromium, copper and zinc were lower than those measured in 1994, but within the previously observed range.

3.1.9 Relationship of Contaminant Levels to Histopathology

The relationship between tissue contaminants and indices of histopathological effects was investigated. This relationship provides the linkage between changes in chemical bioavailability and human seafood consumption risk, as indicated by fish body burdens, and potential ecotoxicological impacts, as indicated by fish histopathology. It also tests how one set of measures might predict the other. The prevalence of hydropic vacuolation was compared with chemical contaminant concentrations in fish collected from monitoring stations. Comparable analyses for the greater number of samples analyzed in 1992, for instance, have shown significant correlations with halogenated organic compounds such as the DDT and chlordane groups (Moore et al., in press). Organic contaminant and histology data were compared

between all stations for the years 1992-1995 (Moore et al., 1992; Shea, 1993; Hillman et al., 1994; Hillman and Peven, 1995). (Please note that these are Modell II-type regressions (i.e., exploring relationships in the data and) rather than Model I-type regression (i.e., predicting CHV as a function of tissue concentration)).

These relationships are illustrated in Figures 3-6a to 3-6e for filet tissue concentrations of selected organic compounds and mercury; and in Figures 3-7a through 3-7d for liver tissue concentrations of selected organic compounds and silver. Each data point in each figure compares a specific chemical or group of chemicals in either the liver or filet, with the prevalence of hydropic vacuolation at the same station in the same year. The regressions which were statistically significant ($p<0.05$) are plotted. Examination of these plots reveals a persistent relationship for total PCBs, chlordanes and DDTs, which support the notion that histopathological data is a reasonable predictor for tissue burden of these compound classes. Significant correlations were seen for all of the major organic classes and mercury in the filet, but only for chlordanes in the liver. This may reflect the greater short-term variability in liver burden, whereas the skeletal muscle is a more stable longer-term store. The plots of liver tPCB and tDDT vs. CHV (Figure 3-7a, b) are suggestive but non-significant; the influence of the 1995 data points may bear further scrutiny. For some liver tissue contaminants, such as silver (Figure 3-7d), no relationship between tissue contaminant levels and CHV exists.

3.1.10 Relationship of Contaminant Levels to FDA Legal Limits

Comparison was made between tissue contaminant levels and regulatory action limits. The U.S. Food and Drug Administration (FDA) has set legal limits for the maximum tissue concentrations of specific organic compounds and pesticides in the edible portions of fish and fishery products. The 1995 mean concentrations of target analytes per station were compared to the FDA's Legal Limits. These are presented below in Table 3-7.

The available historical data as well as data gathered during this program (1992-1995), were compared to the FDA Legal Limits for mercury and PCBs in Figure 3-7. Note that the concentrations in this figure are expressed as mass contaminants per mass wet weight. As both Table 3-7 and Figure 3-11 indicate, the tissue concentrations in winter flounder edible tissues are well below the federal legal limit for fish and shellfish.

3.2 Northern Lobster

3.2.1 Lobster Collected

The 1995 lobster survey was conducted according to the CW/QAPP (Mitchell et al., 1995), but few northern lobster (*H. americanus*) were collected in the twenty commercial lobster traps set at each station.

Most of the lobsters retained in the traps were below the legal size limit (as measured with a commercial lobster gauge). At the Future Outfall Site (FOS), three legal-sized nonberried lobsters were obtained. No legal-sized nonberried lobsters were obtained at either the Eastern Cape Cod Bay station or the Deer Island Station. The remaining 42 lobsters were purchased from commercial lobstermen with traps either adjacent to the traps set by ENSR (Eastern Cape Cod Bay, Deer Island Flats) or within 2 km of the designated sampling stations (Future Outfall Site).

3.2.2 Size, Sex, and External Conditions

The size, sex and external conditions (i.e., black gill disease, shell erosion, parasites, external tumors, etc.) were determined for the lobsters collected in the 1995 survey. The mean length and weight of lobsters collected in 1995 are presented in Table 3-8 below.

As shown in Table 3-8, little difference in lobster length or weight was observed between the three sampling sites, although there was a slight trend of increasing length and weight proceeding from the Deer Island Flats to Eastern Cape Cod Bay. Although the ratio of males to females was about equal at Deer Island Flats and Eastern Cape Cod Bay, nearly three times more males than females were trapped at the Future Outfall Site.

Table 3-9 presents the average values for general external observations made for the 15 lobsters collected at each station in the 1995 survey. With the exception of a single observation of shell erosion in a lobster at the Future Outfall Site, no deleterious external conditions were noted.

3.2.3 Spatial Comparison of Tissue Contaminant Levels

A summary of the lobster mean tissue contaminant levels by station is presented in Table 3-10. The spatial pattern of tissue contaminant levels in lobsters collected during the 1995 survey was examined. Figure 3-9 presents a graphic picture of the spatial and temporal trends in lobster edible tissue concentrations of selected organic compounds and mercury on a station-by-station basis. Figure 3-10 shows annual concentrations of target organic analytes and mercury in lobster hepatopancreas tissues, and Figure 3-11 portrays changes in hepatopancreas metals concentrations. Each of the figures show data from 1992 to 1995 surveys, grouped according to sampling station. The associated line represents one standard error. If no line is shown, then sample size was insufficient (i.e., <3) to calculate the standard deviation.

3.2.3.1 Edible Tissue

The spatial pattern of tissue contaminant levels of organic compounds in edible tissue was generally uniform for lobsters collected during the 1995 survey (Figure 3-9). In general, the 1995 tissue concentrations of organics were lower than those observed in 1994. Concentrations of organic compounds

were generally lowest in lobsters collected in Eastern Cape Cod Bay, with the exception of total DDT, where little variation was observed between stations in the study area. Little variation was observed between concentrations of organics measured at Deer Island Flats and the Future Outfall Site.

Mean mercury concentrations were similar in Deer Island Flats and Eastern Cape Cod Bay lobsters in 1995, and somewhat elevated at the Future Outfall Site. A similar trend in mean mercury concentrations was also seen in 1993 and 1994 lobster edible tissues.

3.2.3.2 Hepatopancreas

The spatial pattern of tissue contaminant levels of organic compounds in the hepatopancreas varied somewhat from the edible tissue patterns. In 1995, concentrations of organic compounds in lobster hepatopancreas tissues were consistently highest at the Future Outfall Site. The lowest concentrations of dieldrin, total PCBs, and total PAHs were seen in Eastern Cape Cod Bay, while the lowest concentrations of total DDT and total chlordane were seen at Deer Island Flats.

The highest concentrations of metals in study area lobster hepatopancreas were seen at Deer Island Flats (silver, chromium, copper) and the Future Outfall Site (lead, nickel, zinc). Mercury concentrations were approximately the same in hepatopancreas tissues at all three stations. The lowest concentrations of metals in lobster hepatopancreas tissues were usually seen at Eastern Cape Cod Bay. For example, lead concentrations were 87% lower than at the Future Outfall Site, and 85% lower than at Deer Island Flats. Copper, mercury, silver and chromium concentrations were also much lower in Eastern Cape Cod Bay than the other stations, while nickel concentrations were similar throughout the study area. Zinc was the only tissue metal highest in Eastern Cape Cod Bay lobsters.

3.2.4 Temporal Comparison of Tissue Contaminant Levels

The temporal trends of tissue contaminants in lobster edible tissue and hepatopancreas are shown in Figures 3-9 through 3-11. Figure 3-12 summarizes both historical and recent lobster edible tissue data for PCBs and mercury and compares them to FDA Legal Limits.

3.2.4.1 Edible Tissues

In 1995, concentrations of organic compounds in lobster edible tissues at Deer Island Flats and the Future Outfall Site were consistently lower than concentrations measured in 1994. Concentrations in Eastern Cape Cod Bay lobster edible tissues were lower in 1995 than 1994 for total chlordane, but slightly higher for total DDT, dieldrin and total PCBs.

Total DDT, total chlordane, dieldrin, and total PCBs decreased by approximately 48% (DDT) to 93% (chlordane) from concentrations observed in the edible tissues of lobster of Deer Island Flats in 1994. Although the concentrations of dieldrin and total PCBs were within the range of values observed from 1992 to 1994, the concentrations of DDT and chlordane were well below the 1992-1994 range. Concentrations of these organics decreased from 1994 to 1995 by approximately 10% (dieldrin) to 89% (chlordane) at the Future Outfall Site. Chlordane was outside the range of concentrations observed here from 1992 through 1994, although concentrations of the other organics were within this observed time period. Sample collections were performed in August of 1992, 1994, and 1995, although 1993 samples were collected in April. Concentrations of these organics in Eastern Cape Cod Bay lobsters collected in 1995 were generally within the range of observations made from 1992 to 1994.

Concentrations of mercury decreased from 1994 to 1995 by 36% in the edible tissues of lobsters collected at Deer Island Flats; this mean concentration (0.53 µg/g dry weight) was also below values observed in 1992 and 1993. The concentrations of mercury in lobsters of the Future Outfall Site and Eastern Cape Cod Bay were generally within the range of those observed from 1992 to 1995.

3.2.4.2 Hepatopancreas

Concentrations of organic compounds in lobster throughout the study area were generally observed to be outside the range of values observed from 1992 to 1994. At Deer Island Flats, lobster concentrations of chlordane, and total PAHs were below the range previously measured, and below 1994 mean values by 69% (PAHs); total PCB concentrations exceeded the 1994 mean concentration by 82%. The mean concentration of total DDT was higher than the 1994 value, but within the range observed previously. At the Future Outfall Site, the mean concentration of dieldrin was the highest of the record period. Concentrations of all other organics also increased in comparison to all previously mean concentrations. Similar trends were observed at the Eastern Cape Cod Bay station.

Mean concentrations of inorganics in lobster hepatopancreas tissues fluctuated in comparison to earlier values throughout the study area, although mean mercury concentrations remained within the 1992 to 1994 range at all stations. At Deer Island Flats, the mean tissue concentration of silver increased by 157% over the highest previously measured mean (10.74 µg/g dry weight, 1994); zinc and lead mean concentrations were lower than all previously measured values; nickel, chromium, copper, and mercury were within the 1992 to 1994 range. At the Future Outfall Site, mean concentrations of lead, nickel, chromium, and zinc were all below the 1992 to 1994 values; silver was 194% higher than the highest previously measured mean (7.47 µg/g, 1994); and copper and mercury were within the 1992 to 1994 range. At the Eastern Cape Cod Bay station, concentrations of all inorganics decreased in comparison to 1994 mean values (except mercury, which was approximately the same); lead, nickel, chromium, and copper were below all 1992 to 1994 means.

3.2.5 Relationship of Contaminant Levels to FDA Legal Limits

The U.S. Food and Drug Administration has set legal limits for maximum concentrations of numerous organic compounds and mercury in the edible portions of fish and fishery products (U.S. EPA, 1989). These organic compounds include: total PCBs (2.0 ppm); aldrin/dieldrin (0.3 ppm); chlordane (0.3 ppm); DDT (5.0 ppm); DDE (5.0 ppm); DDD (5.0 ppm); DDI (5.0 ppm); endrin (0.3 ppm); heptachlor/ heptachlor-epoxide (0.3 ppm); kepone (0.3-0.4 ppm); mirex (0.1 ppm); and toxaphene (5.0 ppm).

The following table (Table 3-11) compares the mean wet weight concentrations of these organic compounds and mercury in lobster edible tissues collected throughout the study area in 1995 to the applicable FDA Legal Limits. As the table indicates, all organic tissue contaminant levels were far below the applicable FDA Legal Limit, normally by 2 or more orders of magnitude. Mercury levels in lobster were also well below the FDA Legal Limit.

3.3 Blue Mussel

3.3.1 Mussels Collected

The 1995 Mussel Bioaccumulation Survey was hampered by incomplete retrieval of deployed mussels due to apparent disturbance of the mooring stations (as described in Section 2.3.4). On August 2, 1995, mussels were collected from one of the three sampling locations, the New England Aquarium's *Discovery* (exposure time of 40 days). Only one array was found at both the Deer Island Light and the Large Navigation Buoy (LNB); these arrays were left in place for subsequent retrieval.

The final mussel retrieval for 1995 was conducted on August 11 (49 days after deployment). This earlier-than-planned retrieval was selected due to concerns over the deployment security. Mussels were retrieved from only two stations (*Discovery* and Deer Island); no intact arrays were found at the LNB location. Two LNB deployment structures were obtained but cages were missing on both of these arrays. Further details on mussel retrieval and cage condition are given in Section 2.3.4. The mussels/arrays deployed and recovered at each sampling location are quantified in Table 3-12.

3.3.2 Biological Condition Indices

As part of the 1995 Mussel Bioaccumulation Survey, data were collected on the survival, sexual maturity, size, and weights for pre-deployment and recovered mussels. The results of survival and sexual maturity analyses of the mussels retrieved at each station are summarized in Table 3-13.

3.3.2.1 Survival

As shown in Table 3-13, the percent survival observed in the cages was high (i.e., ≥94%) for both the forty- and forty-nine day harvested mussels. Survival at the forty-day harvest of the *Discovery* mussels was 98 percent for both the Gloucester and Sandwich mussels. Slightly lower survival was observed at 49 days, including 95% for the Sandwich mussels at both stations, 98% for Gloucester mussels at Deer Island, and 94% for Gloucester mussels at the *Discovery*.

3.3.2.2 Sexual Maturity

A representative sample of randomly selected mussels was examined from three locations (Gloucester pre-deployment, Deer Island, and the *Discovery*) to determine the sex ratio and stage of gametogenesis of mussels (Table 3-13). Sex was determined visually with the female gonads generally appearing orange, while the males were more yellow in color following methods described by Downey et al. (1995).

Of the 30 pre-deployment mussels from Gloucester which were examined in June, all ten females collected were mature, while the remaining 20 mussels consisted of immature males. The proportion of mature females in the Gloucester pre-deployment mussels is consistent with observations made earlier in this study (i.e., 1991-93 surveys). However, the high proportion of immature males observed in 1994 was again evident in 1995. It is not clear why the proportion of immature males was dramatically higher in 1994 and 1995 than in the previous three years. Of the 60 random mussels examined from the 49 day harvest, 9 mature females and 0 mature males were collected at Deer Island, and 17 mature female and 2 mature males at the *Discovery* (as shown in Table 3-13), while the remainder was comprised almost exclusively of immature males.

Ten pre-deployment mussels from Sandwich, MA were also examined. Six of the 10 were mature females and the remaining 4 consisted of immature males. The 20 random Sandwich mussels examined at the forty-nine day harvest yielded 4 mature females and 0 mature males at the Deer Island station, and 5 mature females and 1 mature male at the *Discovery*. The 49-day harvested mussels were also examined for stage of gametogenesis and abnormalities, such as lesions or parasites in the soft tissue. No lesions or parasites were observed on any of the mussels originating from Gloucester or Sandwich mussel beds.

3.3.2.3 Growth and Condition

The size and growth of the mussels at the various stations were statistically analyzed using paired t-tests (Snedecor and Cochran, 1973) (Table 3-14). Mean shell length increased slightly in Gloucester mussels deployed at Deer Island (0.7 mm) and decreased slightly at the *Discovery* (1.4 mm); however, the observed changes were not statistically significant ($P>0.05$). The mean shell wet weight of Deer Island mussels (17.9g) was significantly higher ($P<0.05$) than pre-deployment mussels (16.0 g). The mean shell

wet weight of *Discovery* mussels (16.2 g) was not significantly different from Gloucester pre-deployment mussels.

The mean shell length of Sandwich mussels deployed at Deer Island and the *Discovery* was not statistically different from Sandwich pre-deployment mussels (P>0.05). The mean shell lengths for Deer Island and *Discovery* Sandwich mussels were also not significantly different from each other with respect to this parameter.

The mean non-gonadal soft tissue wet weights of Deer Island and *Discovery* mussels were not significantly different from the Gloucester pre-deployment mussels (P>0.05). However, the gonadal mean wet weights for Deer Island and *Discovery* mussels were significantly larger than the Gloucester pre-deployment mussels. Deer Island and *Discovery* mussels were not significantly different from each other.

The mean non-gonadal wet weights of Sandwich mussels deployed at Deer Island and the *Discovery* were significantly smaller than Sandwich pre-deployment non-gonadal weights (P<0.05). There were no statistically significant differences between Deer Island and *Discovery* mean non-gonadal wet weights.

3.3.3 Spatial Comparison of Tissue Contaminant Levels

The differences in mussel tissue contaminant levels were examined across the various sampling and deployment locations. For purposes of comparison with historical data (Downey et al., 1992; 1993; Downey, 1994a; 1994b), non-detect values were treated as values with levels equal to the detection limit.

3.3.3.1 PAH compounds

The compound list of Low Molecular Weight PAHs, defined as those target 2- and 3-ringed compounds, (LMW-PAH) and High Molecular Weight PAHs, defined as 4-, 5- and 6-ringed compounds, (HMW-PAH) analyzed in 1995 differed slightly as compared to previous years (Table 3-15). In order to examine trends in body burdens of mussels harvested with previous studies, the total PAHs (t-PAH), LMW-PAHs, and the HMW-PAHs, were calculated two ways. The sums of all LMW-PAHs and HMW-PAHs were calculated by adding all of the LMW and HMW PAHs listed in Table 3-16, respectively.

Study area PAH mussel burdens were also calculated to include similar compounds reported by the National Oceanic and Atmospheric Administration's (NOAA) National Status and Trends Program. Cumulative NOAA LMW-PAHs and HMW-PAHs were calculated using the individual or comparable PAHs comprising the target compounds of the Status and Trends Program. The NOAA LMW-PAHs included the following 1995 measured compounds: naphthalene, C1-naphthalene (comparable to 1-methylnaphthalene and 2-methylnaphthalene), C2-naphthalene (similar to 2,5-dimethylnaphthalene), C3-naphthalenes (similar to 1,3,5-trimethyl-naphthalene), acenaphthalene, acenaphthene, fluorene,

phenanthrene, anthracene, and C1-phenanthrenes/anthracene (similar to 1-methylphenanthrene). The HMW-PAHs include: fluoranthene, pyrene, benzo(a)anthracene, chrysene, benzo(b)fluoranthene, benzo(k)fluoranthene, benzo(e)pyrene, benzo(a)pyrene, perylene, indeno(1,2,3,-c,d)pyrene, dibenz(a,h)anthracene, and benzo(g,h,i)perylene. The NOAA comparable parameters were used to evaluate both spatial and temporal trends.

A total of five samples from each of the three locations (Gloucester pre-deployment, *Discovery* and Deer Island) were analyzed for 41 individual PAHs (Table 3-16). Mussels deployed at both the *Discovery* and the Deer Island locations contained body burdens of PAHs that were significantly higher than the Gloucester pre-deployment mussels ($P<0.05$) (Table 3-17). Although the body burdens of individual PAH compounds were quantitatively different for the entire analyte list and the NOAA list of PAHs, the spatial trends were similar. Mussel body burdens of NOAA comparable t-PAHs from both the *Discovery* deployed mussels (1444 ug/kg dry wt) and the Deer Island deployed mussels (761 ug/kg dry wt) were significantly higher than the Gloucester pre-deployment mussels (214 ug/kg dry wt) ($P<0.05$). The tPAH *Discovery* mussel body burdens were significantly higher than the tPAH Deer Island mussel body burdens ($P<0.05$).

The highest tissue concentrations of LMW-PAHs were found in the Deer Island mussels ($P<0.05$). The NOAA LMW-PAH Deer Island deployed mussel burdens (340 ug/kg dry wt) were approximately 50 percent higher than the *Discovery*-deployed mussel burdens (206 ug/kg dry wt ($P<0.05$)). The NOAA LMW-PAH body burdens from mussels deployed at both of these stations was significantly higher than the Gloucester-predeployed mussel body burdens (105 ug/kg dry wt) ($P<0.05$). Of the comparable NOAA analytes, the C1-, C2-, C3-naphthalenes, phenanthrene, and C1-phenanthrene/anthracenes were significantly higher in Deer Island- deployed mussels compared to the *Discovery*-deployed mussels ($P<0.05$). Acenaphthalene was detected in *Discovery*-deployed mussels but not in either the Gloucester pre-deployment or Deer Island deployed mussels.

The NOAA HMW-PAHs were detected in the highest concentrations in the *Discovery*-deployed mussels (1238 ug/kg dry wt) ($P<0.05$). Seventeen of the nineteen HMW-PAHs analyzed were detected in significantly higher concentrations in *Discovery* mussels when compared to Deer Island deployed mussels (dibenz(a,h)anthracene and C4-chrysene were found in similar concentrations) ($P<0.05$). The NOAA comparable HMW-PAHs for Deer Island deployed mussels (421 ug/kg dry wt) were significantly higher than the pre-deployment mussels (109 ug/kg dry wt) ($P<0.05$).

NOAA tPAH body burdens for all three locations were comparable to body burdens of mussels deployed from 1991-1994 (Figures 3-13). The 1995 Deer Island mussel body burdens of NOAA tPAHs of 761 ug/kg dry wt was comparable to mussel body burdens detected in 1993 and 1994 of 665 ug/kg and 848 ug/kg dry, respectively. The NOAA LMW-PAH Deer Island mussel body burdens of 340 ug/kg dry wt was approximately 50 percent higher than comparable mussel body burdens reported for 1993 and 1994

(169 ug/kg dry wt and 217 ug/kg dry wt). The NOAA HMW-PAHs body burdens were numerically the lowest reported for Deer Island mussels during the five years (1991-1995) of study.

The 1995 NOAA tPAH for Discovery deployed mussels (1444 ug/kg dry wt) was comparable to previous years which ranged from 1321 ug/kg dry wt in 1993 to 3546 ug/kg dry wt in 1992. The Discovery NOAA LMW-PAH body burdens of 206 ug/kg dry wt was nearly twice that found in 1993 and 1994 (110 ug/kg dry wt and 79 ug/kg dry wt).

Benzothiazole, a potential marker for urban runoff, was also measured in mussel tissue. Tissue levels of benzothiazole reported from the 1995 survey show a maximum at Deer Island (11.0 ng/g dry weight), and lesser values at *Discovery* (4.5 ng/g dry weight) and for the Gloucester pre-deployment mussels (7.6 ng/g dry weight).

3.3.3.2 Pesticides

Eight pesticides including: hexachlorobenzene, heptachlor, aldrin, trans-nonachlor, heptachlor epoxide, lindane (gamma-BHC), 2,4'-DDE, and Mirex were found at or near detection levels at all stations (Tables 3-18 and 3-19). Five compounds, including dieldrin, alpha-chlordane, trans-nonachlor, 2,4-DDD, and 4,4-DDE, were found at significantly higher levels in Deer Island and *Discovery* mussels, after 49 days of exposure as compared to the pre-deployment mussels ($P<0.05$). The tissue concentrations for these same five compounds were numerically higher for the *Discovery* mussels (<0.05). Two compounds, 2,4-DDT and 4,4-DDT, were found in significantly higher tissue concentrations in the Deer Island deployed mussels (3.6 and 6.6 µg/kg dry wt, respectively) as compared to the pre-deployment mussels (0.8 and 2.5 µg/kg dry wt, respectively). Another pesticide, DDMU, was only detected in the *Discovery* mussels (23.5 ng/g dry wt). The Deer island 2,4'DDT body burdens were similar to *Discovery* mussels (2.4 ug/kg dry wet), while the Deer island 4,4'DDT burdens were higher than the *Discovery* 4,4'DDT body burdens (0.6 ug/kg dry wt) ($P<0.05$).

3.3.3.3 Polychlorinated Biphenyls

Mussel tissues were analyzed for twenty polychlorinated biphenyl (PCB) congeners (Tables 3-20 and 3-21). Twelve of the twenty PCB congeners were found in significantly higher tissue concentrations in Deer Island and *Discovery* mussels when compared to the Gloucester pre-deployment mussels ($P<0.05$). All twelve PCB congeners were highest in *Discovery* mussels ($P<0.05$). Eight congeners (BZ #5, 8, 77, 126, 170, 180, 195, 205, 209) were not routinely detected in body burdens from all three stations.

3.3.3.4 Mercury and Lead

Mercury tissue concentrations from mussels harvested during Sandwich pre-deployment, the *Discovery* and Deer Island deployments (0.065 mg/kg, 0.068 mg/kg and 0.061 mg/kg, respectively) were statistically similar for all three locations (P>0.05) (Table 3-22). Average lead tissue concentrations (8.5 mg/kg) in *Discovery* deployed mussels was significantly higher than the Gloucester pre-deployment concentrations which averaged 6.1 mg/kg (P<0.05). Deer Island deployed mussels body burdens of 8.0 mg/kg were not statistically different from either the Gloucester pre-deployment mussels or the *Discovery* deployed mussels (P>0.05).

3.3.4 Temporal Trends in Tissue Contaminants

The differences in mussel tissue contaminant levels were also examined across the various study years. These comparison are discussed below for each of the analytical groups.

3.3.4.1 PAH Compounds

In 1995, there were several differences in the study design compared to previous years which may have contributed to annual variability observed. The nominal 60-day harvest occurred after 49 days of deployment at Deer Island and Discovery due to concerns about vulnerability of the arrays to boat traffic and/or vandalism. The effect of this early harvest is not known but was expected to be minimal (Peven et al., 1996). If early harvesting was to have an effect on the body burdens, it would be expected that body burden estimates for 1995 deployed mussels might be conservative (lower) since steady state concentrations might not have been attained.

The 1995 analytical detection levels for individual PAHs of 10 ug/kg dry wt were twice the detection levels of 5 ug/kg dry wt reported in previous (1991-1994) studies. To calculate the 1995 tPAHs, LMW-PAHs, and HMW-PAHs, for mussels harvested from all stations, detection levels were used as estimates of body burdens for those analytes below detection levels. This had the effect of inflating the calculated tissue concentrations for the 1995 NOAA tPAH, LMW/PAH, and HMW-PAH averages relative to previous years.

The estimated average NOAA LMW-PAH concentrations of 105 ug/kg dry wt for the Gloucester pre-deployment mussels may be high due to the higher detection levels reported in 1995. If the non-detectable analytes were able to be reported (i.e., truly not present) at the lower detection levels of previous years, this would have resulted in the average concentrations being 35 ug/kg dry wt lower for pre-deployment mussels. It is equally likely that the NOAA LMW-PAHs in Gloucester mussels were just below 10 ug/kg, and thus, the average body burdens may have been closely approximated by the 1995 average values reported for pre-deployment mussels.

The average NOAA LMW-PAH tissue concentrations in Deer Island and *Discovery* deployed mussels were higher than the last several years (1993 and 1994) but within the range observed from 1991-1994. Analytical variability between 1995 and previous years may have resulted in slightly higher estimates for this year relative to 1993 and 1994. The bias associated with the detection levels (discussed in the previous paragraph) may have resulted in a proportionally small (about 15 ug/kg dry wt) inflation of actual concentrations in mussels deployed for 49 days in 1995.

Another confounding factor which may have affected annual comparability was the expanded analyte list which may have inflated 1995 average concentrations relative to previous years. The C2-naphthalene (2,6-dimethylnaphthalenes and other dimethylnaphthalenes) and C3-naphthalene (2,3,5-trimethylnaphthalene and other trimethylnaphthalenes) concentration reported in the 1995 Deer island deployed mussels averaged 174 ug/kg dry wt. In 1994 the NOAA protocols used reported 2,6-dimethylnaphthalene and 2,3,5-trimethylnaphthalene body burdens which averaged a total of 78 ug/kg dry wt. This represents methylphenanthrene concentrations in 1995 of about double (nearly 100 ug/kg dry wt) the tissue concentrations reported in 1994. The 1995 C1-phenanthrenes/anthracene (1methylphenanthrene and other C1-methylphenanthene/anthracene compounds) Deer Island body burdens also averaged about twice the tissue concentrations observed in 1994 for Deer Island mussels. These analytical differences in reporting dimethylnaphthalenes and trimethylnapthalenes and methylphenanthrene in 1995 relative to previous year's averages account for a substantial portion of the higher LMW-PAH body burdens reported in 1995 to 1994 and previously reported LMW-PAH body burden.

Notwithstanding of the differences in study design in 1995, general trends in annual body burdens from the three stations can be gleaned. With the differences in analytical techniques for PAHs taken into account, 1995 appears to be comparable to previous years. The LMW-PAHs, particularly the methylnaphthalenes, phenanthrene and methylphenanthrenes, were highest in Deer Island deployed mussels suggesting that these compounds are present in the Deer Island POTW effluent.

The tPAHs and HMW-PAHs were highest in the Discovery deployed mussels. High concentrations were also found in Deer Island deployed mussels. These results suggest that other source(s) are relatively more important in making HMW-PAHs available to mussels deployed in the inner Boston Harbor (*Discovery* station) than the Deer Island POTW effluent.

Mussel body burdens harvested from the *in situ* cages deployed at Deer Island from 1991-1995 continue the trend of reduced body burdens relative to the mussels deployed and harvested during the 1987 study. The available data on annual variability from 1987 to 1995 suggests that the availability of PAHs, particularly the methylnaphthalenes and phenanthrenes, have declined in the 1990's (compared to 1987). Since the Deer Island POTW effluent is implicated as a source of these LMW-PAHs, this trend indicates that LMW-PAH concentrations in the effluent were lower during 1991-1995 compared to 1987. Although

some annual variability and laboratory data complications exists for PAH concentrations in Deer Island mussel body burdens during the 1991-1995, there were no discernable trends during this five year period.

3.3.4.2 Pesticides

The 1995 t-DDT tissue concentrations for *Discovery* and Gloucester deployed mussels remained at a similar level as observed in 1994 suggesting a leveling off from the peak levels observed in 1993. The 1995 t-DDT concentrations at *Discovery* were similar to those observed in 1991 and 1992. Analytical difficulties encountered in 1993 may have obscured the temporal comparisons of body burdens for mussels deployed at the three sites. These data support the general observation (Downey et al., 1995) that the t-DDT concentrations at Deer Island have been relatively stable from 1987 to 1995.

Hexachlorobenzene (HCB) tissue concentrations were reported at lower levels for mussels collected in 1994 and 1995 from Deer Island, and *Discovery* stations as compared to the 1993 concentrations. The 1993 results may have been unreliable due to possible blank contamination with HCB during sample processing and analysis.

The mussel body burdens of total chlordane (trans-nonachlor, alpha-chlordane, heptachlor epoxide, and lindane) and tDDTs at the Deer Island and *Discovery* stations have varied yearly since 1991, but trends have remained generally the same (Figure 3-14). The total-chlordane levels at Deer Island and *Discovery* were numerically lower during 1995 as compared to 1994. However, the shortened exposure period in 1995 may have resulted in artificially lower tissue concentrations. The pre-deployment Gloucester mussels generally displayed similar concentrations of these three groups of pesticides with the exception of 1993 when high burdens of DDT and total chlordane were observed. The alpha-chlordane concentrations of Deer Island mussels were similar among the year (Figure 3-14). The 1995 Deer Island total chlordane levels (12.8 µg/kg) were significantly lower as compared to the 1994 concentrations. Dieldrin/aldrin concentrations in Deer Island mussels were also significantly lower in 1995 (3.3 µg/kg) as compared to 1994 levels (11.1 µg/kg), but were comparable to those levels reported in 1991-1993.

Deer Island mussel tissue concentrations for 2,4'-DDE and 4,4'-DDD were significantly lower in 1995 as compared to 1994 tissue concentrations. The concentration of 2,4-DDD was found in significantly higher (approximately 5 to 10 times higher) in Deer Island and *Discovery* mussels in 1995 than 1994. However, the tDDT concentrations observed in Deer Island mussels during the past five years suggest that tDDT concentrations at Deer Island are relatively stable (Figure 3-14). Yearly variations in the observed t-DDT concentrations may be partly attributable to analytical variations resulting from the use of a different capillary column configuration in 1993 and 1994 as compared to 1991-1992 (see Downey, 1994b for discussion).

3.3.4.3 Polychlorinated Biphenyls

Although the Deer Island tPCB tissue concentrations increased numerically from 1994 to 1995, tPCB burdens have been relatively stable from 1991-1995. Body burdens during 1991-1995 have been consistently lower than 1987 burdens suggesting a decrease in tPCBs exposure since 1987. the tPCB tissue concentrations for *Discovery* deployed mussels were numerically lowest in 1995 as compared to 1991-1994 levels. The pattern of tPCBs in *Discovery* mussel burdens have steadily declined in average concentrations since 1992 suggesting lower exposure concentrations for mussels deployed at this station. Body burdens of tPCBs in pre-deployment Gloucester mussels have remained relatively stable during 1991-1995.

3.3.4.4 Mercury and Lead

Metal body burdens in indigenous mussels harvested throughout North America have been well described in the Mussel Watch program. Average concentrations and one standard deviation above the average (on a log normal scale) referred to as "high" values, are available for numerous sites throughout the U.S. (O'Conner 1992, O'Conner and Beliaeff 1995). There are several sites, including Boston Harbor (one at Deer Island) and Cape Ann, where indigenous mussels are routinely analyzed. Burdens reported at these sites are grossly comparable to this study's stations. If the concentrations are above the "high" value, the site is generally considered to have elevated contaminant levels.

The 1995 body burdens of lead for mussels deployed at the *Discovery* and Deer Island stations were consistent with previous years (1991 - 1994). Since *Discovery*-and Deer Island-deployed mussels were comparable statistically, it is likely that there may be more significant sources of available lead to Boston Inner Harbor relative to the Deer Island POTW effluent discharge. However, it should be noted that lead tissue concentrations from mussels harvested from the three locations, including the pre-deployment mussel tissues, exceeded the NOAA "high" lead concentration of 4.3 mg/kg (O'Connor, 1992).

Mercury concentrations were low (ranging from 0.061 to 0.068 mg/kg) and were not statistically different for mussels harvested in 1995 from the *Discovery*, Deer Island and Sandwich (control) pre-deployment locations (Table 3-22). The 1995 Deer Island and *Discovery* mussel mercury body burdens were lower than comparable body burdens reported in previous years (ranging from 0.18 to 0.21 mg/kg for Deer Island mussels and averaging 0.16 mg/kg for *Discovery*-deployed mussels (Downey et al., 1995).

Lower mercury body burdens in deployed mussels were anticipated in 1995 due to the use of mussels obtained from the Sandwich control site for deployment (prior deployments utilized Gloucester mussels, which were later found to exhibit mercury contamination). The pre-deployment mussel mercury concentration of Sandwich mussels was 0.065 mg/kg, which was lower than the 1993 and 1994 pre-deployment Gloucester mussel body burdens of 0.39 and 0.18 mg/kg, respectively. The mussel mercury

concentrations were generally less in mussels harvested from the Inner Harbor than in pre-deployment mussels. Under these circumstances it would be difficult to assess the bioavailability of mercury in the Inner Harbor.

The 1995 results using Sandwich mussels with reported low pre-deployment tissue concentrations suggest that there was no statistically significant bioaccumulation of mercury by the *Discovery* and Deer Island deployed mussels. These data could be interpreted that there are no high concentrations of bioavailable mercury present in the Inner Harbor. However, these results are contrary to trends reported by NOAA for the National Status and Trends Program Mussel Watch Project (O'Connor and Beliaeff, 1995). Since 1990, three out of four annual samplings of indigenous mussels from the Deer Island location contained average Hg body burdens which exceeded the Mussel Watch "high" concentration of 0.24 mg/kg.

There are several possible explanations for the apparent differences between the mussel tissue concentrations observed in the Mussel Watch Project and this study. The mussels collected and used in the Mussel Watch project are exposed year round to the waters near Deer Island, while the deployed mussels were exposed for 49 days. It may be that the 49 day exposure was inadequate for steady state tissue concentrations to be reached by the deployed mussels. Analytical differences between the Mussel Watch Project and this study may also contribute to these apparent differences. At this time, there is no definitive answer to explain the apparent differences.

Table 3-1

Catch per unit effort (# of fish per minute of bottom time) for winter flounder trawled in April 1991-1995

	1991	1992	1993	1994	1995
Deer Island	0.38	0.23	0.15	0.39	0.1
Nantasket Beach	0.48	1.29	1.52	0.76	0.88
Broad Sound		2.8	0.49	0.42	0.29
Future Outfall	0.13	0.48	0.62	0.24	0.6
Eastern Cape Cod Bay	0.67	0.49	0.77	0.42	0.21

The same vessel (F/V *Odessa*) and fishing gear was used consistently through the study period.

Table 3-2

Summary of physical characteristics of winter flounder sampled in 1995 from Massachusetts and Cape Cod Bays.

	Station	1 Deer Island Flats	2 Nantasket Beach	3 Broad Sound	4 Future Outfall Site	5 Eastern Cape Cod Bay
	Number of fish	50	50	50	50	50
Total Length (mm)	Mean	343	334	338	331	330
	Std. Dev.	35	24	28	34	22
	ANOVA[a]	5				
Weight (gm)	Mean	448	454	469	433	437
	Std. Dev.	116	111	114	130	94
	ANOVA					
Age (yrs)	Mean	3.6	4.4	4.2	4.1	4
	Std. Dev.	0.8	0.8	0.9	1	0.8
	ANOVA	2,3,4	5			
Fin Erosion (0 to 4)	Mean	0.5	0.4	0.4	0.2	0.3
	Std. Dev.	0.6	0.5	0.8	0.4	0.5
	ANOVA	4	4	4		
Gross Score (0 to 4)	Mean	0.02	0.02	0.04	0	0.04
	Std. Dev.	0.14	0.14	0.2	0	0.2
	ANOVA					

[a] ANOVA was performed for comparison of individual physical characteristics by station. Fishers Probability of Least Squares Difference was the criteria used for significant difference between stations. The level of significance was $p<0.05$

Table 3-3

Prevalence (%) of histological changes in winter flounder liver from 5 stations in Massachusetts and Cape Cod Bays - 1995

Station	Deer Island Flats	Nantasket Beach	Broad Sound	Future Outfall Site	Eastern Cape Cod Bay
Number	50	50	50	50	50
Neoplasm	0	0	2	0	0
Focal HV[a]	2	0	0	0	0
Tubular HV	24	10	20	6	0
Centrotubular HV	38	26	28	14	4
Balloons	6	12	18	6	14
Macrophage aggregation	62	66	60	70	58
Biliary proliferation	18	18	28	18	24
Necrosis	22	22	6	14	22
Pancreatic hyperplasia	0	2	0	0	2
Pancreatic necrosis	0	4	0	0	4
Eosinophilic focus	0	2	0	0	0
Basophilic focus	0	0	0	0	0
Hepatocye regeneration	16	36	14	12	6

[a] HV = Hydropic Vacuolation

Table 3-4

Spearman rank correlation of histology with year of collection, age and body length for winter flounder from Deer Island Flats, Boston Harbor - 1987 to 1995[a].

	Year		Age		Length	
	rho[b]	p	rho	p	rho	p
Centrobular hydropic vacuolation	-0.721	0.0052	0.31	0.269	0.74	0.004
Tubular hydropic vacuolation	-0.76	0.0031	0.48	0.0848	0.82	0.0016
Focal hydropic vacuolation	-0.64	0.0138	0.05	0.852	0.65	0.0114
Neoplasms	-0.58	0.0238	0.64	0.0207	0.79	0.0022

[a] 1989 to 1993 only for age

[b] rho = coefficient value of -1.0 for a perfect inverse relationship, and +1.0 for a perfect direct relationship.

Table 3-5

Age class analysis of prevalence (%) of centrotubular vacuolation in winter flounder from Deer Island Flats, Boston Harbor, collected in 1989-1995.

	All fish	3 years old	4 years old	5 years old	to 9 years old
1987	77 (51)	NA	NA	NA	NA
1988	65 (52)	NA	NA	NA	NA
1989	51 (25)	0 (2)	29 (7)	33 (9)	86 (7)
1990	39 (99)	0 (4)	38 (29)	48 (31)	41 (32)
1991	47 (167)	33 (12)	32 (69)	55 (40)	66 (41)
1992	48 (56)	45 (20)	55 (20)	50 (12)	25 (4)
1993	55 (29)	0 (1)	47 (15)	50 (8)	100 (5)
1994	54 (50)	46 (13)	61 (18)	67 (9)	40 (10)
1995	38 (50)	42 (24)	45 (20)	67 (3)	100 (2)
Mean	47.4	23.70%	43.90%	52.90%	63.40%

Prevalence given as a % of sample affected, with sample size in parentheses

Table 3-6
Summary of Mean Fish Tissue Contaminant Levels from 1995 MWRA Flounder Survey (dry wt.)

Class	Fraction	Parameter Code	Deer Island Mean	Deer Island S.E.	Outfall Mean	Outfall S.E.	Cape Cod Mean	Cape Cod S.E.	Units
				Pesticides					
DDD	Filet	2,4'-DDD	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
DDD	Filet	4,4'-DDD	7.13	1.43	1.63	0.23	1.80	0.46	ng/g
DDE	Filet	2,4'-DDE	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
DDE	Filet	4,4'-DDE	35.00	14.11	19.67	2.73	25.67	1.45	ng/g
DDT	Filet	2,4'-DDT	1.10	1.10	1.50	0.12	0.00	0.00	ng/g
DDT	Filet	4,4'-DDT	0.00	0.00	0.33	0.33	0.00	0.00	ng/g
Total			**43.23**	**16.63**	**23.13**	**2.66**	**27.47**	**1.88**	ng/g
DDD	Liver	2,4'-DDD	0.00	0.00	6.00	6.00	0.00	0.00	ng/g
DDD	Liver	4,4'-DDD	143.67	35.03	27.67	3.76	16.97	5.71	ng/g
DDE	Liver	2,4'-DDE	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
DDE	Liver	4,4'-DDE	710.00	83.86	420.00	101.49	143.33	18.56	ng/g
DDT	Liver	2,4'-DDT	12.67	6.44	0.00	0.00	0.00	0.00	ng/g
DDT	Liver	4,4'-DDT	0.00	0.00	1.57	1.57	0.00	0.00	ng/g
Total			**866.33**	**76.78**	**455.23**	**100.00**	**160.30**	**17.05**	ng/g
PEST	Filet	ALDRIN	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
PEST	Filet	DIELDRIN	3.00	1.61	1.10	0.55	0.00	0.00	ng/g
PEST	Filet	ENDRIN	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
PEST	Filet	GAMMA-BHC	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
PEST	Filet	HCB	0.71	0.04	0.52	0.04	0.55	0.01	ng/g
PEST	Filet	MIREX	0.36	0.05	0.28	0.03	0.16	0.08	ng/g
PEST	Liver	ALDRIN	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
PEST	Liver	DIELDRIN	52.67	4.48	0.00	0.00	7.00	7.00	ng/g
PEST	Liver	ENDRIN	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
PEST	Liver	GAMMA-BHC	1.40	0.15	0.97	0.49	0.00	0.00	ng/g
PEST	Liver	HCB	6.70	0.44	3.63	1.86	2.27	2.27	ng/g
PEST	Liver	MIREX	4.90	0.29	3.80	0.35	1.39	0.89	ng/g
Chlordane	Filet	ALPHA-	5.57	1.27	2.03	0.19	1.37	0.09	ng/g
Chlordane	Filet	TRANS-	9.90	2.06	3.73	0.28	3.27	0.18	ng/g
Chlordane	Filet	HEPT	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
Chlordane	Filet	HEPT-EPOX	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
Total			**15.47**	**3.32**	**5.77**	**0.47**	**4.63**	**0.19**	ng/g
Chlordane	Liver	ALPHA-	86.33	13.67	30.33	4.18	13.07	5.48	ng/g
Chlordane	Liver	TRANS-	196.67	29.63	65.67	12.71	26.67	11.26	ng/g
Chlordane	Liver	HEPT	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
Chlordane	Liver	HEPT-EPOX	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
Total			**283.00**	**42.88**	**96.00**	**16.29**	**39.73**	**16.74**	ng/g
				PCBs					
PCB	Filet	101	19.67	5.17	10.30	0.35	5.87	0.22	ng/g
PCB	Filet	105	18.67	4.70	7.07	0.87	3.40	0.00	ng/g
PCB	Filet	118	77.33	31.34	31.00	2.52	17.00	0.58	ng/g
PCB	Filet	126	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
PCB	Filet	128	6.57	1.44	4.27	0.59	2.40	0.00	ng/g
PCB	Filet	138	94.33	32.87	45.67	2.03	23.00	1.53	ng/g

Table 3-6
Summary of Mean Fish Tissue Contaminant Levels from 1995 MWRA Flounder Survey (dry wt.)

Class	Fraction	Parameter Code	Deer Island Mean	Deer Island S.E.	Outfall Mean	Outfall S.E.	Cape Cod Mean	Cape Cod S.E.	Units
PCB	Filet	153	151.33	40.35	52.33	1.20	29.33	2.03	ng/g
PCB	Filet	170	14.33	3.33	8.00	0.46	2.53	0.15	ng/g
PCB	Filet	18	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
PCB	Filet	180	189.00	55.90	47.67	18.02	8.67	0.59	ng/g
PCB	Filet	187	18.00	4.51	17.67	3.71	8.40	0.75	ng/g
PCB	Filet	195	2.17	0.29	1.73	0.58	0.33	0.03	ng/g
PCB	Filet	206	2.23	0.22	2.60	0.70	0.77	0.02	ng/g
PCB	Filet	209	0.85	0.08	0.97	0.08	0.35	0.02	ng/g
PCB	Filet	28	4.33	0.87	1.20	0.12	1.23	0.07	ng/g
PCB	Filet	44	0.33	0.33	0.06	0.06	0.00	0.00	ng/g
PCB	Filet	52	3.57	0.72	1.47	0.12	0.92	0.11	ng/g
PCB	Filet	66	11.17	2.92	5.17	0.78	3.40	0.31	ng/g
PCB	Filet	77	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
PCB	Filet	8	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
Total			**613.88**	**176.48**	**237.16**	**15.01**	**107.61**	**5.60**	ng/g
PCB	Liver	101	336.67	75.13	126.67	8.82	56.67	17.90	ng/g
PCB	Liver	105	343.33	39.30	96.67	48.42	45.33	20.41	ng/g
PCB	Liver	118	1300.00	173.21	666.67	375.65	173.33	53.64	ng/g
PCB	Liver	126	0.00	0.00	1.77	1.77	0.00	0.00	ng/g
PCB	Liver	128	92.67	11.46	69.00	16.77	18.00	5.51	ng/g
PCB	Liver	138	1666.67	88.19	1233.33	284.80	233.33	78.39	ng/g
PCB	Liver	153	2100.00	152.75	1600.00	404.15	286.67	96.84	ng/g
PCB	Liver	170	286.67	31.80	140.00	15.28	35.33	17.33	ng/g
PCB	Liver	18	5.80	0.25	3.53	1.97	0.00	0.00	ng/g
PCB	Liver	180	2366.67	260.34	1600.00	503.32	253.67	188.17	ng/g
PCB	Liver	187	380.00	72.11	370.00	170.10	70.00	12.77	ng/g
PCB	Liver	195	30.33	1.86	27.33	6.98	6.70	4.15	ng/g
PCB	Liver	206	27.33	2.03	36.00	10.02	9.80	3.62	ng/g
PCB	Liver	209	8.23	0.91	10.40	1.40	3.76	1.82	ng/g
PCB	Liver	28	60.33	4.18	15.00	2.31	12.43	4.31	ng/g
PCB	Liver	44	9.03	0.22	2.93	0.12	0.00	0.00	ng/g
PCB	Liver	52	38.33	1.76	17.67	0.67	12.00	7.57	ng/g
PCB	Liver	66	190.00	55.08	73.67	7.80	32.33	9.33	ng/g
PCB	Liver	77	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
PCB	Liver	8	0.91	0.55	0.00	0.00	0.00	0.00	ng/g
Total			**9242.98**	**839.25**	**6090.63**	**1747.82**	**1249.36**	**520.59**	ng/g
				PAHs					
PAH	Liver	ACE	3.67	3.67	0.00	0.00	0.00	0.00	ng/g
PAH	Liver	ACEY	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
PAH	Liver	BAA	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
PAH	Liver	BAP	0.37	0.37	0.00	0.00	0.00	0.00	ng/g
PAH	Liver	BBF	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
PAH	Liver	BEP	0.00	0.00	0.67	0.67	0.00	0.00	ng/g
PAH	Liver	BGP	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
PAH	Liver	BIP	12.67	0.33	1.93	1.93	1.53	1.53	ng/g
PAH	Liver	BKF	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
PAH	Liver	C0A	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
PAH	Liver	C0C	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
PAH	Liver	C0D	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
PAH	Liver	C0F	8.33	2.93	0.00	0.00	0.00	0.00	ng/g
PAH	Liver	C0N	87.67	11.20	32.00	0.58	32.00	0.82	ng/g
PAH	Liver	C0P	7.60	4.99	2.60	1.38	10.47	1.27	ng/g

Table 3-6
Summary of Mean Fish Tissue Contaminant Levels from 1995 MWRA Flounder Survey (dry wt.)

Class	Fraction	Parameter Code	Deer Island Mean	Deer Island S.E.	Outfall Mean	Outfall S.E.	Cape Cod Mean	Cape Cod S.E.	Units
PAH	Liver	C1C	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
PAH	Liver	C1D	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
PAH	Liver	C1F	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
PAH	Liver	C1F/P	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
PAH	Liver	C1N	38.00	2.52	15.00	0.58	11.33	2.33	ng/g
PAH	Liver	C1P/A	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
PAH	Liver	C2C	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
PAH	Liver	C2D	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
PAH	Liver	C2F	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
PAH	Liver	C2N	53.67	12.44	7.00	7.00	0.00	0.00	ng/g
PAH	Liver	C2P/A	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
PAH	Liver	C3C	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
PAH	Liver	C3D	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
PAH	Liver	C3F	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
PAH	Liver	C3N	20.00	20.00	0.00	0.00	0.00	0.00	ng/g
PAH	Liver	C3P/A	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
PAH	Liver	C4C	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
PAH	Liver	C4N	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
PAH	Liver	C4P/A	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
PAH	Liver	DAH	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
PAH	Liver	DBF	5.47	3.18	1.57	1.57	3.97	0.13	ng/g
PAH	Liver	FLANT	1.67	1.67	0.00	0.00	0.00	0.00	ng/g
PAH	Liver	IND	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
PAH	Liver	PER	1.13	0.64	0.77	0.39	0.00	0.00	ng/g
PAH	Liver	PYR	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
Total			**240.23**	**40.11**	**61.53**	**11.62**	**48.63**	**7.96**	ng/g
				Metals					
Metal	Liver	Lead	0.84	0.16	5.94	1.69	5.22	1.17	µg/g
Metal	Liver	Nickel	0.14	0.02	0.44	0.11	0.46	0.05	µg/g
Metal	Liver	Silver	3.42	1.88	9.89	2.60	4.55	0.39	µg/g
Metal	Liver	Cadmium	0.44	0.07	1.42	0.09	0.66	0.01	µg/g
Metal	Liver	Chromium	0.14	0.01	0.09	0.02	0.09	0.02	µg/g
Metal	Liver	Copper	55.86	22.31	121.40	12.89	64.52	4.16	µg/g
Metal	Liver	Zinc	105.68	1.38	151.65	6.49	138.12	11.65	µg/g
Metal	Liver	Mercury	0.25	0.05	0.39	0.02	0.30	0.05	µg/g
Metal	Filet	Mercury	0.40	0.02	0.31	0.04	0.10	0.01	µg/g
				Other					
PAH	Liver	BTHOL	6.00	6.00	6.33	6.33	31.67	6.17	ng/g

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TABLE 3-7

Comparison of FDA Legal Limits to Mean Concentrations of Select Organic Compounds and Mercury In Winter Flounder Edible Tissues - 1995[1]

Compound/Analyte	Deer Island Flats	Future Outfall Site	Eastern Cape Cod Bay	FDA Legal Limit[2]
Total DDT (ppb)	7.43 (5.1)	3.97 (0.91)	4.89 (0.48)	5,000[3]
Total Chlordane (ppb)	2.64 (1.04)	1.00 (0.15)	0.83 (0.004)	300
Dieldrin (ppb)	0.52 (0.48)	0.19 (0.16)	ND	300
Total PCBs (ppm)	105.34 (54.6)	41.28 (4.91)	19.9 (1.80)	2,000
Total Mercury (ppm)	0.07 (0.005)	0.06 (0.016)	0.03 (0.011)	1.0

1 All values given are on a wet-weight basis. The numbers in parenthesis indicate the standard deviation.

2 U.S. EPA. 1989. Assessing Human Health Risks from Chemically Contaminated Fish and Shellfish. EPA Document No. EPA-503/8-89-002. Office of Marine and Estuarine Protection (WH-556F) and the Office of Water Regulations and Standards (WH-552). Washington, D.C.

3 The value is the FDA Legal Limits for DDT (5.0 ppm), DDE (5.0 ppm), and DDD (5.0 ppm), which comprise the mean total DDT tissue concentration, total DDT value below 5 ppm assumes that all DDT derivations are not exceeded.

ND Not detected

TABLE 3-8

Mean Length, Weight, and Sex of Lobsters Collected in 1995

Station	Mean Carapace Length (mm)	Mean Weight (gm)	Sex Ratio (M/F)
Deer Island Flats	87.6 (2.8)	35.3 (1.9)	7/8
Future Outfall Site	88.7 (3.4)	35.4 (2.1)	11/4
Eastern Cape Cod Bay	89.4 (3.3)	37.3 (2.2)	8/7

NOTES: Each mean is based on a sample size of 15 individual lobsters. The number in parenthesis indicates the standard deviation.

TABLE 3-9

Mean External Conditions of Lobsters Collected in 1995

Station	Black Gill[1]	Shell Erosion	Parasites	External Tumors
Deer Island Flats	0 (0)	0 (0)	0 (0)	0 (0)
Future Outfall Site	0 (0)	0.13 (0.50)	0 (0)	0 (0)
Eastern Cape Cod Bay	0 (0)	0 (0)	0 (0)	0 (0)

NOTES:

Each mean is based on a sample size of 15 individual lobsters. The number in parenthesis indicates the standard deviation.

1 Observed values were based on visual examination of individual lobsters. The values range from 0 (absent) to 4 (extreme).

Table 3-10
Summary of Mean Lobster Tissue Contaminant Levels from 1995 MWRA Lobster Survey (dry wt.)

Class	Fraction	Parameter Code	Deer Island Mean	Deer Island S.E.	Outfall Mean	Outfall S.E.	Cape Cod Bay Mean	Cape Cod Bay S.E.	Units
				Pesticides					
DDD	Hepatopanc	2,4'-DDD	0.00	0.00	13.67	6.89	5.00	5.00	ng/g
DDD	Hepatopanc	4,4'-DDD	107.17	31.56	123.00	18.77	70.00	20.01	ng/g
DDE	Hepatopanc	2,4'-DDE	6.00	6.00	2.23	2.23	0.00	0.00	ng/g
DDE	Hepatopanc	4,4'-DDE	546.67	124.41	763.33	27.28	656.67	67.41	ng/g
DDT	Hepatopanc	2,4'-DDT	0.00	0.00	10.67	10.67	8.00	8.00	ng/g
DDT	Hepatopanc	4,4'-DDT	10.67	5.33	17.00	1.53	6.27	3.13	ng/g
Total			**670.50**	**155.45**	**929.90**	**29.58**	**745.93**	**92.05**	**ng/g**
DDD	Tissue	2,4'-DDD	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
DDD	Tissue	4,4'-DDD	1.00	0.64	1.12	0.18	0.29	0.15	ng/g
DDE	Tissue	2,4'-DDE	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
DDE	Tissue	4,4'-DDE	12.33	1.45	13.00	0.58	12.93	2.70	ng/g
DDT	Tissue	2,4'-DDT	0.29	0.29	0.21	0.21	0.00	0.00	ng/g
DDT	Tissue	4,4'-DDT	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
Total			**13.62**	**2.36**	**14.34**	**0.79**	**13.22**	**2.55**	**ng/g**
Chlordane	Hepatopanc	ALPHA-	24.33	7.42	18.67	9.40	8.00	4.04	ng/g
Chlordane	Hepatopanc	TRANS-	14.33	14.33	55.00	27.93	57.00	26.63	ng/g
Chlordane	Hepatopanc	HEPT	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
Chlordane	Hepatopanc	HEPT-EPOX	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
Total			**38.67**	**13.57**	**73.67**	**37.02**	**65.00**	**22.72**	**ng/g**
Chlordane	Tissue	ALPHA-	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
Chlordane	Tissue	TRANS-	0.39	0.20	0.59	0.21	0.06	0.05	ng/g
Chlordane	Tissue	HEPT	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
Chlordane	Tissue	HEPT-EPOX	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
Total			**0.39**	**0.20**	**0.59**	**0.21**	**0.06**	**0.05**	**ng/g**
PEST	Hepatopanc	ALDRIN	0.00	0.00	2.80	1.49	0.00	0.00	ng/g
PEST	Hepatopanc	DIELDRIN	52.67	26.84	106.67	11.79	30.00	15.04	ng/g
PEST	Hepatopanc	ENDRIN	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
PAH	Hepatopanc	GAMMA-BHC	5.50	0.26	5.13	0.90	2.67	0.32	ng/g
PEST	Hepatopanc	HCB	10.13	0.75	11.67	0.33	8.80	0.90	ng/g
PEST	Hepatopanc	MIREX	7.05	0.73	8.57	0.59	6.10	0.51	ng/g
PEST	Tissue	ALDRIN	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
PEST	Tissue	DIELDRIN	6.50	0.15	5.77	0.23	3.93	0.33	ng/g
PEST	Tissue	ENDRIN	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
PEST	Tissue	GAMMA-BHC	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
PEST	Tissue	HCB	0.00	0.00	0.21	0.21	0.25	0.25	ng/g
PEST	Tissue	MIREX	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
				PCBs					
PCB	Hepatopanc	101	179.33	85.36	230.00	40.00	53.67	26.87	ng/g
PCB	Hepatopanc	105	281.67	105.37	296.67	28.48	130.00	11.55	ng/g

Table 3-10
Summary of Mean Lobster Tissue Contaminant Levels from 1995 MWRA Lobster Survey (dry wt.)

Class	Fraction	Parameter Code	Deer Island Mean	Deer Island S.E.	Outfall Mean	Outfall S.E.	Cape Cod Bay Mean	Cape Cod Bay S.E.	Units
PCB	Hepatopanc	118	916.67	345.17	906.67	96.84	483.33	49.78	ng/g
PCB	Hepatopanc	126	0.00	0.00	1.50	1.50	30.00	15.53	ng/g
PCB	Hepatopanc	128	170.00	45.83	176.67	3.33	92.33	7.67	ng/g
PCB	Hepatopanc	138	866.67	274.12	1026.67	73.33	566.67	73.11	ng/g
PCB	Hepatopanc	153	980.00	269.07	1150.00	104.08	720.00	109.70	ng/g
PCB	Hepatopanc	170	125.67	26.93	143.33	23.33	80.33	8.29	ng/g
PCB	Hepatopanc	18	1.53	1.53	1.77	1.77	0.43	0.43	ng/g
PCB	Hepatopanc	180	350.00	81.85	486.67	33.83	216.67	37.56	ng/g
PCB	Hepatopanc	187	270.00	52.92	356.67	31.80	223.33	32.83	ng/g
PCB	Hepatopanc	195	17.83	2.92	25.00	6.00	11.30	1.48	ng/g
PCB	Hepatopanc	206	31.17	3.09	46.67	7.45	27.00	3.21	ng/g
PCB	Hepatopanc	209	14.00	1.00	15.67	1.20	10.27	0.91	ng/g
PCB	Hepatopanc	28	76.33	17.25	82.33	8.95	25.00	1.53	ng/g
PCB	Hepatopanc	44	9.75	1.67	11.07	2.45	5.17	1.71	ng/g
PCB	Hepatopanc	52	19.33	9.94	40.00	3.51	11.67	6.17	ng/g
PCB	Hepatopanc	66	215.00	62.92	236.67	21.86	92.00	4.73	ng/g
PCB	Hepatopanc	77	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
PCB	Hepatopanc	8	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
Total			**4524.95**	**1354.20**	**5234.00**	**342.50**	**2779.17**	**305.36**	ng/g
PCB	Tissue	101	4.20	0.64	3.93	0.23	3.17	0.28	ng/g
PCB	Tissue	105	8.40	1.81	7.40	1.00	3.87	0.80	ng/g
PCB	Tissue	118	29.00	7.02	25.67	2.96	16.67	2.91	ng/g
PCB	Tissue	126	0.67	0.67	0.60	0.60	0.00	0.00	ng/g
PCB	Tissue	128	6.30	1.33	5.80	0.20	2.87	0.82	ng/g
PCB	Tissue	138	22.00	4.16	21.67	1.76	15.00	2.52	ng/g
PCB	Tissue	153	23.67	3.28	23.33	1.45	17.67	2.73	ng/g
PCB	Tissue	170	2.50	0.38	2.53	0.17	1.87	0.37	ng/g
PCB	Tissue	18	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
PCB	Tissue	180	7.90	0.95	9.17	0.94	4.67	0.68	ng/g
PCB	Tissue	187	7.37	0.42	7.60	0.36	5.63	0.94	ng/g
PCB	Tissue	195	0.28	0.04	0.38	0.03	0.21	0.03	ng/g
PCB	Tissue	206	0.62	0.09	0.58	0.10	0.51	0.06	ng/g
PCB	Tissue	209	0.12	0.02	0.13	0.03	0.03	0.03	ng/g
PCB	Tissue	28	2.13	0.38	1.90	0.17	0.86	0.10	ng/g
PCB	Tissue	44	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
PCB	Tissue	52	0.63	0.63	1.63	1.63	0.00	0.00	ng/g
PCB	Tissue	66	6.53	0.88	5.80	0.47	3.07	0.43	ng/g
PCB	Tissue	77	0.00	0.00	0.00	0.00	0.00	0.00	ng/g
PCB	Tissue	8	0.00	0.00	0.63	0.63	0.00	0.00	ng/g
Total			**122.31**	**22.28**	**118.76**	**9.56**	**76.08**	**12.45**	ng/g
				PAHs					
PAH	Hepatopanc	ACE	18.33	3.28	10.00	5.03	16.67	0.88	ng/g
PAH	Hepatopanc	ACEY	6.70	0.59	2.30	2.30	6.63	3.76	ng/g
PAH	Hepatopanc	BAA	63.67	16.25	83.00	18.88	101.33	33.49	ng/g
PAH	Hepatopanc	BAP	63.17	14.42	101.33	23.84	88.67	22.10	ng/g
PAH	Hepatopanc	BBF	115.67	19.55	163.33	20.28	148.33	33.71	ng/g
PAH	Hepatopanc	BEP	109.83	11.69	176.67	29.63	83.67	11.92	ng/g

Table 3-10
Summary of Mean Lobster Tissue Contaminant Levels from 1995 MWRA Lobster Survey (dry wt.)

Class	Fraction	Parameter Code	Deer Island Mean	Deer Island S.E.	Outfall Mean	Outfall S.E.	Cape Cod Bay Mean	Cape Cod Bay S.E.	Units
PAH	Hepatopanc	BGP	47.00	7.09	78.00	14.73	45.67	2.33	ng/g
PAH	Hepatopanc	BIP	7.42	1.41	8.47	1.27	6.13	0.64	ng/g
PAH	Hepatopanc	BKF	54.17	9.34	92.67	17.33	68.67	19.13	ng/g
PAH	Hepatopanc	C0A	41.50	19.53	21.00	0.58	26.67	6.17	ng/g
PAH	Hepatopanc	C0C	225.00	36.17	316.67	26.67	193.33	46.67	ng/g
PAH	Hepatopanc	C0D	16.42	3.98	10.60	0.95	8.70	0.68	ng/g
PAH	Hepatopanc	C0F	27.33	1.76	23.00	1.53	35.67	2.67	ng/g
PAH	Hepatopanc	C0N	28.17	3.61	30.33	8.35	34.67	4.67	ng/g
PAH	Hepatopanc	C0P	162.33	45.04	84.33	2.91	108.67	11.33	ng/g
PAH	Hepatopanc	C1C	78.67	12.41	115.33	17.37	64.67	14.08	ng/g
PAH	Hepatopanc	C1D	53.00	2.08	58.00	5.20	26.67	3.18	ng/g
PAH	Hepatopanc	C1F	61.83	5.53	67.00	3.06	50.00	7.37	ng/g
PAH	Hepatopanc	C1F/P	255.00	39.69	353.33	39.30	280.00	85.05	ng/g
PAH	Hepatopanc	C1N	47.67	1.86	48.67	8.67	56.00	5.20	ng/g
PAH	Hepatopanc	C1P/A	251.67	55.70	250.00	32.15	220.00	51.96	ng/g
PAH	Hepatopanc	C2C	66.33	10.35	88.33	10.87	39.67	5.33	ng/g
PAH	Hepatopanc	C2D	153.33	24.04	193.33	31.80	67.67	14.75	ng/g
PAH	Hepatopanc	C2F	186.67	17.64	243.33	23.33	113.00	18.77	ng/g
PAH	Hepatopanc	C2N	110.00	5.77	107.00	11.79	80.33	8.09	ng/g
PAH	Hepatopanc	C2P/A	421.67	74.29	496.67	88.76	253.33	60.09	ng/g
PAH	Hepatopanc	C3C	38.33	4.98	50.33	4.33	0.00	0.00	ng/g
PAH	Hepatopanc	C3D	128.33	15.90	166.67	35.28	42.33	7.69	ng/g
PAH	Hepatopanc	C3F	330.00	88.88	343.33	75.13	243.33	110.20	ng/g
PAH	Hepatopanc	C3N	228.33	13.02	250.00	17.32	130.00	10.00	ng/g
PAH	Hepatopanc	C3P/A	301.67	59.61	390.00	70.95	150.00	11.55	ng/g
PAH	Hepatopanc	C4C	0.00	0.00	14.33	7.84	0.00	0.00	ng/g
PAH	Hepatopanc	C4N	205.00	18.93	313.33	59.25	166.00	67.00	ng/g
PAH	Hepatopanc	C4P/A	161.67	30.05	203.33	37.12	104.33	18.66	ng/g
PAH	Hepatopanc	DAH	6.42	1.23	12.63	2.99	5.63	2.84	ng/g
PAH	Hepatopanc	DBF	23.33	0.33	20.33	1.20	23.67	1.45	ng/g
PAH	Hepatopanc	FLANT	611.67	102.48	703.33	66.92	586.67	177.04	ng/g
PAH	Hepatopanc	IND	46.00	9.07	76.67	17.85	59.67	9.49	ng/g
PAH	Hepatopanc	PER	17.83	4.15	34.67	8.88	28.00	2.00	ng/g
PAH	Hepatopanc	PYR	391.67	64.96	463.33	34.80	420.00	113.72	ng/g
Total			**5162.78**	**687.90**	**6265.00**	**630.73**	**4184.43**	**809.13**	**ng/g**
Metal	Hepatopanc	Lead	0.26	0.04	0.30	0.04	0.04	0.01	ug/g
Metal	Hepatopanc	Nickel	0.42	0.09	0.43	0.04	0.45	0.04	ug/g
Metal	Hepatopanc	Silver	27.55	1.95	21.99	3.37	8.10	2.35	ug/g
Metal	Hepatopanc	Cadmium	5.29	0.25	5.32	0.59	7.94	0.22	ug/g
Metal	Hepatopanc	Chromium	0.24	0.04	0.18	0.03	0.09	0.03	ug/g
Metal	Hepatopanc	Copper	324.73	60.19	314.35	35.15	125.24	33.84	ug/g
Metal	Hepatopanc	Zinc	43.94	2.24	51.60	3.18	54.44	1.30	ug/g
Metal	Hepatopanc	Mercury	0.35	0.03	0.34	0.05	0.27	0.07	ug/g
Metal	Tissue	Mercury	0.61	0.30	1.09	0.26	0.54	0.05	ug/g

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TABLE 3-11

Comparison of FDA Legal Limits to Mean Concentrations of Select Organic Compounds and Mercury In Lobster Edible Tissues - 1995[1]

Compound/Analyte	Deer Island Flats	Future Outfall Site	Eastern Cape Cod Bay	FDA Legal Limit[2]
Total DDT (ppb)	1.63 (0.43)	1.79 (0.07)	1.91 (0.46)	5,000[3]
Total Chlordane (ppb)	0.05 (0.03)	0.07 (0.04)	0.01 (0.01)	300
Dieldrin (ppb)	0.78 (0.04)	0.73 (0.09)	0.57 (0.06)	300
Total PCBs (ppm)	0.015 (0.004)	0.015 (0.001)	0.011 (0.002)	2.0
Total Mercury (ppm)	0.07 (0.05)	0.13 (0.04)	0.08 (0.01)	1.0

1 All values given are on a wet-weight basis. The number in parenthesis indicates the standard deviation.

2 U.S. EPA. 1989. Assessing Human Health Risks from Chemically Contaminated Fish and Shellfish. EPA Document No. EPA-503/8-89-002. Office of Marine and Estuarine Protection (WH-556F) and the Office of Water Regulations and Standards (WH-552): Washington, D.C.

3 This value is the FDA Legal Limits for DDT (5.0 ppm), DDD (5.0 ppm), and DDE (5.0 ppm), which comprise the total DDT tissue concentration. Total DDT values below 5 ppm assume that all DDT derivations are not exceeded.

Table 3-12
Mussel harvest and analysis experimental design summary for 1995. Square brackets represent the number of cages and parentheses represent the number of alternate clean site (Sandwich) mussels.

	Initial Deployment		Analysis		Forty-Day Harvest		Forty-Nine Day Harvest Analyses	
Station	**Mussels**	**Arrays**	**Biology[1]**	**Arrays**	**Mussels**	**Arrays**	**Mussels**	**Biology[1]**
Gloucester	85	NA	30	NA	NA	NA	NA	NA
Sandwich	35	NA	10	NA	NA	NA	NA	NA
LNB	360(150)	3	NA	1 [3]1 [1]	NR	NR	NR	NR
Deer Island	300(120)	3	NA	1 [2]1 [1]	0(0)	0(0)	104(40)	30(10)
Discovery	201(80)	1	NA	1 [2]1 [1]	100(40)	1 [2]1 [1]	101(40)	30(10)
Total Mussels:	**946(385)**							

NA Not applicable
NR No mussels recovered.
[1]Biological analyses included sex, sexual maturity, wet/dry weight of gonad mantle and non gonadal soft tissue, shell weight, and total shell length determinations. Biological analyses on alternate clean site (Sandwich) mussels included sexual maturity determinations, total shell length, total wet weight, and gonadal and non gonadal wet weights.

Table 3-13
Survival and stage of gametogenesis of mussels for following predeployment (Gloucester), forty day and forty nine day collections at specific stations. Values in parentheses represent the alternate clean site (Sandwich) mussels.

				Number of			
		Percent	Sample	Females		Males	
Station	**Number**	**Survival**	**Size**	**Mature**	**Immature**	**Mature**	**Immature**
Predeployment							
Gloucester	NA	NA	30	10	0	0	20
Sandwich	NA	NA	10	6	0	0	4
Forty Day							
LNB	NR	NA	NA	NA	NA	NA	NA
Deer Island	NR	NA	NA	NA	NA	NA	NA
Discovery	100(40)	98(98)	NA(NA)	NA(NA)	NA(NA)	NA(NA)	NA(NA)
Forty nine Day							
LNB	NR	NA	NA	NA	NA	NA	NA
Deer Island	104(40)	98(95)	30(10)	9(4)	1(0)	0(0)	20(6)
Discovery	101(40)	94(95)	30(10)	17(5)	0(0)	2(1)	11(4)

NA Not applicable
NR No mussels recovered.

TABLE 3-14

Summary of Various Biological Measurements Expressed as Mean Values for Mussels. "Predeployment" Represents Measurements on Mussels at the Beginning of the Study and "Sixty" - Day Retrieval Represent Values at the End of the Study. All Values Reported on a Wet Basis.

Station	Mean Shell Length (mm)	Mean Wet Weight (g)			
		Total Organism	Gonad - Mantle	Non - gonadal Soft Tissue	Shell
Predeployment:					
Gloucester	64.9	32.7	2.1	5.6	16.0
Sandwich	62.3	30.3	3.3	7.1	14.5
"Sixty" - day:					
Deer Island[1]	65.6	33.7	3.5	6.0	17.9
Deer Island[2]	61.0	28.8	3.0	5.2	15.6
Discovery[1]	63.5	29.6	3.8	5.6	16.2
Discovery[2]	63.3	29.7	3.8	5.8	15.9

[1] Gloucester mussels, n = 30.

[2] Sandwich mussels, n = 10.

TABLE 3-15

Summary of Low Molecular Weight (LMW) and High Molecular Weight (HMW) PAH Analytes Examined in 1995 and the Comparable Analytes from Previous Studies (1991-1994).

1995 PAH Analytes	1995 NOAA PAH Analytes	Comparable NOAA PAHs Analyzed Prior to 1995
LMW PAH's:		
Naphthalene	Naphthalene	Naphthalene
C1-naphthalenes	C1-Naphthalene	1-methylnaphthalene and 2-methylnaphthalene
C2-naphthalenes	C2-Naphthalene	2,5-dimethylnaphthalene
C3-naphthalenes	C3-Naphthalene	1,3,5-trimethylnaphthalene
C4-naphthalenes		
Acenapthhylene	Acenaphthalene	Acenaphthalene
Acenaphthene	Acenaphthene	Acenaphthene
Biphenyl	Biphenyl	1,1-biphenyl
Dibenzofuran		
Fluorene	Fluorene	Fluorene
C1-fluorene		
C2-fluorene		
C3-fluorene		
Phenanthrene	Phenanthrene	
Anthracene	Anthracene	
C1-Phenanthrenes/anthracene	C1-phenanthrene/anthracene	1-methylphenanthrene
C2-Phenanthrenes/anthracene		
C3-Phenanthrenes/anthracene		
C4-Phenanthrenes/anthracene		
Dibenzothlophene		
C1-dibenzothlophene		
C2-dibenzothlophene		
C3-dibenzothlophene		
HMW PAH's:		
Fluoranthene	Fluoranthene	Fluoranthene
Pyrene	Pyrene	Pyrene
C1-fluoranthenes/pyrene		
C2-fluoranthenes/pyrene		
C3-fluoranthenes/pyrene		
Benzo[a]anthracene	Benzo[a]anthracene	Benzo[a]anthracene
Chrysene	Chrysene	Chrysene
C1-chrysene		
C2-chrysene		
C3-chrysene		
C4-'chrysene		
Benzo[b]fluoranthene	Benzo[b]fluoranthene	Benzo[b] and Benzo[k] fluoranthene
Benzo[k]fluoranthene	Benzo[k]fluoranthene	
Benzo[e]pyrene	Benzo[e]pyrene	Benzo[e]pyrene
Benzo[a]pyrene	Benzo[a]pyrene	Benzo[a]pyrene
Perylene	Perylene	Perylene
Indeno[1,2,3-c,d]pyrene	Indeno[1,2,3-c,d]pyrene	Indeno[1,2,3-c,d]pyrene
Dibenzo[a,h]anthracene	Dibenzo[a,h]anthracene	Dibenzo[a,h]anthracene
Benzo[g,h,i]perylene	Benzo[g,h,i]perylene	Benzo[g,h,i]perylene

TABLE 3-16

PAH (µg/kg dry weight) Concentrations in Mussels Exposed at Two Stations. Total PAHs were Calculated Using Detection Limit Values as an Estimated Concentration when Individual Analytes were not Detected.

	Gloucester Predeployment, June 1995					Discovery, August 1995					Deer Island, August 1995					
Parameter	M9511G7 000TC1	M9511G7 000TC2	M9511G7 000TC3	M9511G7 000TC4	M9511G7 000TC5	M9511G6 000TC1	M9511G6 000TC2	M9511G6 000TC3	M9511G6 000TC4	M9511G6 000TC5	M9511G1 000TC1	M9511G1 000TC2	M9511G1 000TC3	M9511G1 000TC4	M9511G1 000TC5	M9511G1 000TC5-dup
Naphthalene	7.6	8	7	9.1	14	13	8.4	7.7	10	15	8.1	9.3	9.6	12	10	10
C1-naphthalenes	9.8	10	6.7	9	18	14	8.9	12	10	12	14	16	17	18	19	25
C2-naphthalenes	10	10	10	10	10	10	12	10	10	14	46	63	50	58	59	80
C3-naphthalenes	10	10	10	10	10	10	23	17	9.6	10	87	110	100	120	110	160
C4-naphthalenes	10	10	10	10	10	10	10	10	10	10	95	150	160	150	190	220
acenapthhylene	5.3	3.3	10	2.9	10	29	29	32	30	26	10	10	5.9	10	10	8.5
acenaphthene	10	10	10	10	10	10	14	14	11	11	10	10	10	10	10	10
biphenyl	10	10	10	10	10	10	10	10	10	4.2	10	10	10	10	10	10
dibenzofuran	10	10	10	10	10	7.7	6.1	5.9	5.4	5.4	3.5	3.9	4.4	4.9	5.3	6.4
fluorene	10	10	10	10	10	10	6.6	6.6	4	7	7	10	10	8.1	10	10
C1-fluorene	14	9.3	12	12	16	27	18	16	15	18	37	40	37	38	55	52
C2-fluorene	11	11	10	10	10	10	69	68	63	43	62	72	78	70	75	84
C3-fluorene	10	10	10	10	10	10	130	140	110	95	120	160	120	130	140	200
phenanthrene	9.8	8.9	8.2	8.2	11	17	17	16	14	18	22	29	25	28	32	32
anthracene	6.2	5.6	5.1	5.5	4.8	55	54	54	55	48	10	17	12	14	13	15
C1-Phenanthrenes/anthracene	15	13	13	14	16	28	36	36	31	31	50	56	55	58	58	76
C2-Phenanthrenes/anthracene	24	20	23	18	24	90	97	91	78	42	130	160	150	150	170	190
C3-Phenanthrenes/anthracene	10	12	17	8.7	11	180	180	180	160	130	140	160	140	150	150	180
C4-Phenanthrenes/anthracene	10	10	10	10	10	280	250	240	240	190	98	120	110	100	100	110
dibenzothiophene	10	10	10	10	10	10	10	10	10	10	5	10	3.9	5.7	5.5	10
C1-dibenzothiophene	10	10	10	10	10	10	10	10	10	10	15	21	19	21	22	31
C2-dibenzothiophene	10	10	10	10	10	59	50	58	46	49	62	88	73	76	89	100
C3-dibenzothiophene	10	10	10	10	10	150	140	10	120	130	92	110	89	110	110	140
Total LMW PAH's	242.7	231.1	242	227.4	264.8	1049.7	1189	1054.2	1062	928.6	1131.6	1435.2	1298.8	1349.7	1450.8	1759.9
total NOAA LMW-PAH's	103.7	98.8	100	98.7	123.8	206	218.9	215.3	194.6	198.2	274.1	340.3	304.5	344.1	339	436.5
fluoranthene	27	29	23	23	23	170	160	160	150	130	62	80	68	71	75	85
pyrene	19	20	16	18	16	250	230	240	220	200	73	87	77	82	82	97
C1-fluoranthenes/pyrene	16	23	18	14	<10	250	210	210	190	180	78	88	74	85	78	95
C2-fluoranthenes/pyrene	<10	<10	<10	<10	<10	220	180	210	160	150	72	82	64	74	70	85
C3-fluoranthenes/pyrene	<10	<10	<10	<10	<10	140	110	120	130	110	39	37	37	41	34	42
benzo[a]anthracene	8.4	12	7.9	6.2	8.1	86	80	84	72	75	30	39	30	31	33	37
chrysene	14	16	10	9.1	12	170	160	160	150	140	56	66	55	58	58	68
C1-chrysene	4.9	7	6.4	<10	<10	120	130	130	120	120	35	48	36	36	36	44
C2-chrysene	<10	<10	<10	<10	<10	110	97	93	91	94	28	37	28	24	24	30
C3-chrysene	<10	<10	<10	<10	<10	54	43	53	42	47	18	16	9.4	10	8.9	<10
C4-chrysene	<10	<10	<10	<10	<10	17	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10
benzo[b]fluoranthene	7.6	9	7.9	5.6	6.6	200	190	180	170	180	43	60	37	47	47	55
benzo[k]fluoranthene	2	3.4	1.5	2.2	1.5	45	48	56	52	55	16	18	13	14	17	18
benzo[e]pyrene	7.7	8.1	6.6	6.2	6.1	210	220	220	200	190	45	53	43	47	50	55
benzo[a]pyrene	2.7	6.2	3.6	2.9	2.5	61	55	58	53	66	15	21	13	16	16	20
perylene	<10	1.7	<10	<10	<10	18	17	16	16	16	5.4	6	4.1	5.7	6.6	5.6
indeno[1,2,3-c,d]pyrene	3.1	4.1	2.9	2.2	2.3	28	27	26	28	36	10	14	9.6	12	11	13
dibenzo[a,h]anthracene	<10	1.1	<10	<10	<10	8.6	8.6	8.5	8.9	11	3.4	3.9	2.4	3	3.9	3.3
benzo[g,h,i]perylene	4.9	4.7	9.2	4.6	6.2	56	55	52	54	59	23	21	16	16	19	22
Total HMW-PAH's	189.3	195.3	183.2	174.4	174.5	2213.6	2050.8	2106.5	1936.9	1871	659.6	788.9	626.7	686.7	679.6	794.9
Total NOAA HMW-PAH's	116.4	115.3	108.8	100.4	104.5	1302.6	1270.6	1282.5	1173.9	1160	381.6	470.9	368.3	406.7	416.7	478.9
Total PAH's	432	426.4	425.2	401.6	439.3	3263.5	3239.6	3162.7	2998.9	2799.6	1791.4	2224.1	1925.5	2036.4	2130.4	2554.6
Total NOAA PAH's	220.1	214.1	208.6	199.1	228.3	1508.6	1489.7	1497.6	1368.5	1358.2	655.9	811.2	672.6	750.6	757.7	915.4

TABLE 3-17

Summary PAH (µg/kg dry weight) Concentrations in Mussels Exposed at Two Stations. Total PAHs were Calculated Using Detection Limit Values as an Estimated Concentration when Individual Analytes were not Detected.

	Gloucester Predeployment, June 1995 (Sample size = 5)				Discovery, August 1995 (Sample size = 5)				Deer Island, August 1995 (Sample size = 6)			
Parameter	Avg.	SD	Range		Avg.	SD	Range		Avg.	SD	Range	
Naphthalene	9.1	2.8	7.0	14.0	10.8	3.1	7.7	15.0	9.8	1.3	8.1	12.0
C1-naphthalenes	10.7	4.3	6.7	18.0	11.4	2.0	8.9	14.0	18.2	3.8	14.0	25.0
C2-naphthalenes	10.0	0.0	10.0	10.0	11.2	1.8	10.0	14.0	59.3	11.9	46.0	80.0
C3-naphthalenes	10.0	0.0	10.0	10.0	13.9	5.9	9.6	23.0	114.5	24.9	87.0	160.0
C4-naphthalenes	10.0	0.0	10.0	10.0	10.0	0.0	10.0	10.0	160.8	42.2	95.0	220.0
acenapththylene	6.3	3.5	2.9	10.0	29.2	2.2	26.0	32.0	9.1	1.7	5.9	10.0
acenaphthene	10.0	0.0	10.0	10.0	12.0	1.9	10.0	14.0	10.0	0.0	10.0	10.0
biphenyl	10.0	0.0	10.0	10.0	6.8	2.6	4.2	10.0	10.0	0.0	10.0	10.0
dibenzofuran	10.0	0.0	10.0	10.0	6.1	0.9	5.4	7.7	4.7	1.0	3.5	6.4
fluorene	10.0	0.0	10.0	10.0	6.8	2.1	4.0	10.0	9.2	1.3	7.0	10.0
C1-fluorene	12.7	2.5	9.3	16.0	18.8	4.8	15.0	27.0	43.2	8.1	37.0	55.0
C2-fluorene	10.4	0.5	10.0	11.0	50.6	25.0	10.0	69.0	73.5	7.5	62.0	84.0
C3-fluorene	10.0	0.0	10.0	10.0	97.0	51.7	10.0	140.0	145.0	30.8	120.0	200.0
phenanthrene	9.2	1.2	8.2	11.0	16.4	1.5	14.0	18.0	28.0	3.9	22.0	32.0
anthracene	5.4	0.5	4.8	6.2	53.2	2.9	48.0	55.0	13.5	2.4	10.0	17.0
C1-Phenanthrenes/anthracene	14.2	1.3	13.0	16.0	32.4	3.5	28.0	36.0	58.2	9.0	50.0	76.0
C2-Phenanthrenes/anthracene	21.8	2.7	18.0	24.0	79.6	22.1	42.0	97.0	158.3	20.4	130.0	190.0
C3-Phenanthrenes/anthracene	11.7	3.2	8.7	17.0	166.0	21.9	130.0	180.0	153.3	15.1	140.0	180.0
C4-Phenanthrenes/anthracene	10.0	0.0	10.0	10.0	240.0	32.4	190.0	280.0	108.0	8.9	96.0	120.0
dibenzothiophene	10.0	0.0	10.0	10.0	10.0	0.0	10.0	10.0	6.7	2.6	3.9	10.0
C1-dibenzothiophene	10.0	0.0	10.0	10.0	10.0	0.0	10.0	10.0	21.5	5.3	15.0	31.0
C2-dibenzothiophene	10.0	0.0	10.0	10.0	52.4	5.8	46.0	59.0	81.3	13.6	62.0	100.0
C3-dibenzothiophene	10.0	0.0	10.0	10.0	110.0	57.0	10.0	150.0	110.2	16.4	92.0	140.0
Total LMW PAH's	241.6	14.6	227.4	264.8	1056.7	92.2	928.6	1189.0	1404.3	208.8	1131.6	1759.9
Total NOAA LMW-PAH's	105.0	10.7	98.7	123.8	206.2	10.9	194.6	218.9	339.8	54.6	274.1	436.5
fluoranthene	25.0	2.8	23.0	29.0	154.0	15.2	130.0	170.0	73.5	8.3	62.0	85.0
pyrene	17.8	1.8	16.0	20.0	228.0	19.2	200.0	250.0	83.0	8.4	73.0	97.0
C1-fluoranthenes/pyrene	16.6	4.9	10.0	23.0	208.0	26.8	180.0	250.0	82.7	8.1	74.0	95.0
C2-fluoranthenes/pyrene	10.0	0.0	10.0	10.0	186.0	27.7	150.0	220.0	74.5	7.8	64.0	85.0
C3-fluoranthenes/pyrene	10.0	0.0	10.0	10.0	122.0	13.0	110.0	140.0	38.3	2.9	34.0	42.0
benzo[a]anthracene	8.5	2.1	6.2	12.0	79.4	5.9	72.0	86.0	33.3	3.8	30.0	39.0
chrysene	12.2	2.8	9.1	16.0	164.0	18.2	140.0	180.0	60.5	5.9	55.0	68.0
C1-chrysene	7.7	2.3	4.9	10.0	124.0	5.5	120.0	130.0	39.2	5.5	35.0	48.0
C2-chrysene	10.0	0.0	10.0	10.0	97.0	7.6	91.0	110.0	28.5	4.8	24.0	37.0
C3-chrysene	10.0	0.0	10.0	10.0	47.8	5.5	42.0	54.0	12.1	3.9	8.9	18.0
C4-chrysene	10.0	0.0	10.0	10.0	11.4	3.1	10.0	17.0	10.0	0.0	10.0	10.0
benzo[b]fluoranthene	7.4	1.2	5.8	9.0	184.0	11.4	170.0	200.0	48.2	8.3	37.0	60.0
benzo[k]fluoranthene	2.1	0.8	1.5	3.4	51.2	4.7	45.0	56.0	16.0	2.1	13.0	18.0
benzo[e]pyrene	7.0	0.9	6.1	8.1	208.0	13.0	190.0	220.0	48.8	4.7	43.0	55.0
benzo[a]pyrene	3.6	1.5	2.5	6.2	58.6	5.1	53.0	66.0	17.2	3.1	13.0	21.0
perylene	8.3	3.7	1.7	10.0	17.4	0.9	16.0	18.0	5.8	0.9	4.1	6.8
indeno[1,2,3-c,d]pyrene	2.9	0.8	2.2	4.1	28.0	4.0	26.0	36.0	11.8	1.7	9.8	14.0
dibenzo[a,h]anthracene	8.2	4.0	1.1	10.0	9.2	1.0	8.5	11.0	3.3	0.6	2.4	3.9
benzo[g,h,i]perylene	6.0	1.9	4.7	9.2	55.2	2.6	52.0	59.0	19.8	2.8	16.0	23.0
Total HMW-PAH's	183.3	9.2	174.4	195.3	2036.2	136.1	1871.0	2213.8	706.1	69.7	626.7	794.9
Total NOAH HMW-PAH's	109.1	6.9	100.4	115.3	1238.0	66.0	1160.0	1302.8	420.9	45.5	368.3	478.9
Total PAH's	424.9	14.1	401.8	439.3	3092.9	193.9	2799.6	3263.5	2110.4	265.4	1791.4	2554.8
Total NOAA PAH's	214.1	11.1	199.1	228.3	1444.2	75.2	1356.2	1508.8	760.6	95.2	655.9	915.4

TABLE 3-18

Pesticide Concentrations (µg/kg dry weight) in Mussels Exposed at Two Stations. Total Pesticides were Calculated Using Detection Limit Values as an Estimated Concentration when Individual Analytes were not Detected.

Parameter	Gloucester Predeployment, June 1995					Discovery, August 1995					Deer Island, August 1995					
	M9511G7 000TC1	M9511G7 000TC2	M9511G7 000TC3	M9511G7 000TC4	M9511G7 000TC5	M9511G6 000TC1	M9511G6 000TC2	M9511G6 000TC3	M9511G6 000TC4	M9511G6 000TC5	M9511G1 000TC1	M9511G1 000TC2	M9511G1 000TC3	M9511G1 000TC4	M9511G1 000TC5	M9511G1 000TC5 (dup)
2,4'-DDD	4.5	4.3	4.4	4.4	4.7	13	12	14	12	10	5.7	6.4	6	6.4	6.4	6.8
2,4'-DDE	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
2,4'-DDT	0.91	0.74	0.86	0.93	0.67	2.8	4.2	0.22	4.6	0.22	3.2	3.7	3.4	3.5	3.7	3.9
4,4'-DDD	11	10	10	11	11	34	37	46	35	29	10	12	11	12	11	12
4,4'-DDE	10	9.8	9.9	10	11	41	40	46	42	33	17	18	17	18	17	16
4,4'-DDT	2.5	2.3	2.3	2.9	2.7	1.8	0.25	0.25	0.25	0.25	5.7	7.2	6.4	6.8	5.9	6.5
ALDRIN	0.12	0.12	0.12	0.12	0.12	0.12	0.12	0.12	0.12	0.12	0.12	0.12	0.12	0.12	0.12	0.12
ALPHA-CHLORDANE	2.9	2.7	2.1	2.6	2.6	13	12	13	11	9.7	6.7	7.7	7.2	7.6	7.4	8.1
TRANS-NONACHLOR	0.49	0.67	0.37	0.87	0.6	8.8	9.2	11	9.4	6.8	4.4	4.4	4.4	4.7	3.6	3.7
DIELDRIN	1.6	1.4	1.4	1.7	1.6	7.4	7.1	7.8	6.6	5.8	2.8	3.3	3.1	3.3	3.2	3.3
ENDRIN	0.14	0.14	0.14	0.14	0.14	0.14	0.14	0.14	0.14	0.14	0.14	0.14	0.14	0.14	0.14	0.14
GAMMA-BHC (Lindane)	0.57	0.8	0.77	0.53	0.59	0.87	0.89	0.92	1	0.73	1	0.92	1.1	0.96	1.1	1
HCB (Hexachlorobenzene)	0.23	0.56	0.1	0.23	0.2	1	0.52	0.82	0.56	0.82	0.35	0.59	0.63	0.65	0.87	0.98
HEPT (Heptachlor)	0.16	0.16	0.16	0.16	0.16	0.16	0.16	0.16	0.16	0.16	0.16	0.16	0.16	0.16	0.16	0.16
HEPT EPOX (Heptachlor epoxide)	0.14	0.14	0.14	0.14	0.14	0.14	0.14	0.14	0.14	0.14	0.14	0.14	0.14	0.14	0.14	0.14
MIREX	0.08	0.08	0.08	0.08	0.08	0.08	0.08	0.35	0.36	0.32	0.17	0.08	0.065	0.095	0.08	0.08
Dieldrin / Aldrin Group	1.72	1.52	1.52	1.82	1.72	7.52	7.22	7.92	6.72	5.92	2.92	3.42	3.22	3.42	3.32	3.42
Chlordane group	4.1	4.31	3.38	4.14	3.93	22.81	22.23	25.06	21.54	17.37	12.24	13.16	12.84	13.4	12.24	12.94
Total DDT's	29.11	27.34	27.66	29.43	30.27	92.8	93.65	106.67	94.05	72.67	41.8	47.5	44	46.9	44.2	45.4
Total Pesticides	35.54	34.11	33.04	36.0	36.5	124.51	124	141.12	123.53	97.4	57.78	65.05	61.055	64.765	61.01	63.12

TABLE 3-19

Summary Pesticide Concentrations (µg/kg dry weight) in Mussels Exposed at Two Stations. Total Pesticides were Calculated Using Detection Limit Values as an Estimated Concentration when Individual Analytes were not Detected.

Parameter	Gloucester Predeployment, June 1995 (Sample size = 5)				Discovery, August 1995 (Sample size = 5)				Deer Island , August 1995 (Sample size = 6)			
	Avg.	SD	Range		Avg.	SD	Range		Avg.	SD	Range	
2,4'-DDD	4.46	0.15	4.30	4.70	12.20	1.48	10.00	14.00	6.28	0.38	5.70	6.80
2,4'-DDE	0.20	0.00	0.20	0.20	0.20	0.00	0.20	0.20	0.20	0.00	0.20	0.20
2,4'-DDT	0.82	0.11	0.67	0.93	2.41	2.11	0.22	4.60	3.57	0.25	3.20	3.90
4,4'-DDD	10.60	0.55	10.00	11.00	36.20	6.22	29.00	46.00	11.33	0.82	10.00	12.00
4,4'-DDE	10.14	0.49	9.80	11.00	40.40	4.72	33.00	46.00	17.17	0.75	16.00	18.00
4,4'-DDT	2.54	0.26	2.30	2.90	0.56	0.69	0.25	1.80	6.42	0.56	5.70	7.20
ALDRIN	0.12	0.00	0.12	0.12	0.12	0.00	0.12	0.12	0.12	0.00	0.12	0.12
ALPHA-CHLORDANE	2.58	0.29	2.10	2.90	11.74	1.41	9.70	13.00	7.45	0.48	6.70	8.10
TRANS-NONACHLOR	0.60	0.19	0.37	0.87	9.04	1.51	6.80	11.00	4.20	0.44	3.60	4.70
DIELDRIN	1.54	0.13	1.40	1.70	6.94	0.77	5.80	7.80	3.17	0.20	2.80	3.30
ENDRIN	0.14	0.00	0.14	0.14	0.14	0.00	0.14	0.14	0.14	0.00	0.14	0.14
GAMMA-BHC (Lindane)	0.65	0.12	0.53	0.80	0.88	0.10	0.73	1.00	1.01	0.07	0.92	1.10
HCB (Hexachlorobenzene)	0.26	0.17	0.10	0.56	0.74	0.20	0.52	1.00	0.68	0.22	0.35	0.98
HEPT (Heptachlor)	0.16	0.00	0.16	0.16	0.16	0.00	0.16	0.16	0.16	0.00	0.16	0.16
HEPT EPOX (Heptachlor epoxide)	0.14	0.00	0.14	0.14	0.14	0.00	0.14	0.14	0.14	0.00	0.14	0.14
MIREX	0.08	0.00	0.08	0.08	0.24	0.14	0.08	0.36	0.10	0.04	0.07	0.17
Dieldrin / Aldrin Group	1.66	0.13	1.52	1.82	7.06	0.77	5.92	7.92	3.29	0.20	2.92	3.42
Chlordane group	3.97	0.36	3.38	4.31	21.80	2.81	17.37	25.06	12.80	0.48	12.24	13.40
Total DDT's	28.76	1.23	27.34	30.27	91.97	12.21	72.67	106.67	44.97	2.09	41.80	47.50
Total Pesticides	35.04	1.43	33.04	36.50	122.11	15.68	97.40	141.12	62.13	2.75	57.78	65.05

TABLE 3-20

Polychlorinated Biphenyl (µg/kg dry weight) Concentrations in Mussels Exposed at Two Stations. Total PCBs were Calculated Using Detection Limit Values as an Estimated Concentration when Individual Analytes were not Detected.

	Gloucester Predeployment, June 1995					Discovery, August 1995					Deer Island, August 1995					
Parameter	M9511G70 00TC1	M9511G70 00TC2	M9511G70 00TC3	M9511G70 00TC4	M9511G70 00TC5	M9511G60 00TC1	M9511G60 00TC2	M9511G60 00TC3	M9511G60 00TC4	M9511G60 00TC5	M9511G10 00TC1	M9511G10 00TC2	M9511G10 00TC3	M9511G10 00TC4	M9511G10 00TC5	M9511G10 00TC5 (dup)
CL2-PCB(8)	0.86	0.86	0.86	0.86	0.86	0.86	0.86	0.86	0.86	0.86	0.86	0.86	0.86	0.86	0.86	0.86
CL3-PCB(18)	2.8	5.9	4.5	4.7	2.6	14	9.2	8.3	11	10	5.3	5.3	5.4	5.8	8.7	7.8
CL3-PCB(28)	1.2	1.3	1	1.1	1.5	13	12	12	12	9.6	4.2	4.4	4.3	4.2	5.7	5.3
CL4-PCB(44)	4.6	3.8	4.2	4	3.8	20	18	18	18	15	5	6.5	5.9	6.2	6.8	7.5
CL4-PCB(52)	5.2	5.6	5	5.8	5.3	33	31	32	32	25	8.8	10	10	10	11	12
CL4-PCB(66)	5.5	4.6	4.9	4.9	5.2	14	11	11	16	15	7.3	7.8	8.8	8.1	7.2	8.3
CL4-PCB(77)	0.49	0.49	0.49	0.49	0.49	0.49	0.49	0.49	0.49	0.49	0.49	0.49	0.49	0.49	0.49	0.49
CL5-PCB(101)	12	11	11	11	11	72	62	66	65	57	20	22	21	21	22	25
CL5-PCB(105)	4.3	4	4.1	4.1	4.3	26	25	28	27	22	7.2	8.7	7.7	8.1	7.9	8.2
CL5-PCB(118)	13	12	12	12	13	78	70	75	75	63	24	27	25	26	26	30
CL5-PCB(126)	0.49	0.49	0.93	0.94	0.49	0.49	0.49	0.49	0.49	0.49	0.49	0.49	0.49	0.49	1.9	1.9
CL6-PCB(128)	2	1.7	2	2.2	2.2	13	12	14	13	11	4.2	4.4	4.2	4.4	4.4	4.9
CL6-PCB(153)	19	17	18	19	19	88	77	85	87	76	32	34	32	32	34	34
CL7-PCB(170)	1.1	1.1	1.1	1.1	1.1	1.1	1.1	1.1	1.1	1.1	1.1	1.1	1.1	1.1	1.1	1.1
CL7-PCB(180)	0.46	0.46	0.46	0.46	0.46	0.46	0.46	0.46	0.46	0.46	0.46	0.46	0.46	0.46	0.46	0.46
CL7-PCB(187)	6.3	5.4	5.8	6	6	23	20	23	23	21	9.5	9.6	9.3	9.9	10	11
CL8-PCB(195)	0.48	0.48	0.48	0.48	0.48	0.48	0.48	0.48	0.48	0.48	0.48	0.48	0.48	0.48	0.48	0.48
CL9-PCB(206)	0.38	0.38	0.38	0.38	0.38	0.38	0.38	0.38	0.38	0.38	0.38	0.38	0.38	0.38	0.38	0.38
CL10-PCB(209)	0.82	0.82	0.82	0.82	0.82	0.82	0.82	0.82	0.82	0.82	0.82	0.82	0.82	0.82	0.82	0.82
CL6-PCB(138)	15	13	14	14	16	76	68	76	77	66	26	27	27	27	30	29
Total PCBs	95.98	90.38	92.02	94.33	94.98	475.08	420.28	453.38	461.08	395.68	158.58	171.78	165.68	167.78	180.19	189.49

TABLE 3-21

Summary Polychlorinated Biphenyl (µg/kg dry weight) Concentrations in Mussels Exposed at Two Stations. Total PCBs were Calculated Using Detection Limit Values as an Estimated Concentration when Individual Analytes were not Detected.

	Gloucester Predeployment, June 1995 (Sample size = 5)				Discovery, August 1995 (Sample size = 5)				Deer Island, August 1995 (Sample size = 6)			
Parameter	Avg.	SD	Range		Avg.	SD	Range		Avg.	SD	Range	
CL2-PCB(8)	0.86	0.00	0.86	0.86	0.86	0.00	0.86	0.86	0.86	0.00	0.86	0.86
CL3-PCB(18)	4.10	1.39	2.60	5.90	10.50	2.20	8.30	14.00	6.38	1.49	5.30	8.70
CL3-PCB(28)	1.22	0.19	1.00	1.50	11.72	1.26	9.60	13.00	4.68	0.65	4.20	5.70
CL4-PCB(44)	4.08	0.33	3.80	4.60	17.80	1.79	15.00	20.00	6.32	0.85	5.00	7.50
CL4-PCB(52)	5.38	0.32	5.00	5.80	30.60	3.21	25.00	33.00	10.30	1.09	8.80	12.00
CL4-PCB(66)	5.02	0.34	4.60	5.50	13.40	2.30	11.00	16.00	7.92	0.61	7.20	8.80
CL4-PCB(77)	0.49	0.00	0.49	0.49	0.49	0.00	0.49	0.49	0.49	0.00	0.49	0.49
CL5-PCB(101)	11.20	0.45	11.00	12.00	64.40	5.50	57.00	72.00	21.83	1.72	20.00	25.00
CL5-PCB(105)	4.16	0.13	4.00	4.30	25.60	2.30	22.00	28.00	7.97	0.50	7.20	8.70
CL5-PCB(118)	12.40	0.55	12.00	13.00	72.20	5.89	63.00	78.00	26.33	2.07	24.00	30.00
CL5-PCB(126)	0.67	0.24	0.49	0.94	0.49	0.00	0.49	0.49	0.96	0.73	0.49	1.90
CL6-PCB(128)	2.02	0.20	1.70	2.20	12.60	1.14	11.00	14.00	4.42	0.26	4.20	4.90
CL6-PCB(153)	18.40	0.89	17.00	19.00	82.60	5.68	76.00	88.00	33.00	1.10	32.00	34.00
CL7-PCB(170)	1.10	0.00	1.10	1.10	1.10	0.00	1.10	1.10	1.10	0.00	1.10	1.10
CL7-PCB(180)	0.46	0.00	0.46	0.46	0.46	0.00	0.46	0.46	0.46	0.00	0.46	0.46
CL7-PCB(187)	5.90	0.33	5.40	6.30	22.00	1.41	20.00	23.00	9.88	0.60	9.30	11.00
CL8-PCB(195)	0.48	0.00	0.48	0.48	0.48	0.00	0.48	0.48	0.48	0.00	0.48	0.48
CL9-PCB(206)	0.38	0.00	0.38	0.38	0.38	0.00	0.38	0.38	0.38	0.00	0.38	0.38
CL10-PCB(209)	0.82	0.00	0.82	0.82	0.82	0.00	0.82	0.82	0.82	0.00	0.82	0.82
CL6-PCB(138)	14.40	1.14	13.00	16.00	72.60	5.18	66.00	77.00	27.67	1.51	26.00	30.00
Total PCBs	93.54	5.23	0.38	19.00	441.10	27.37	0.38	88.00	172.25	10.07	0.38	34.00

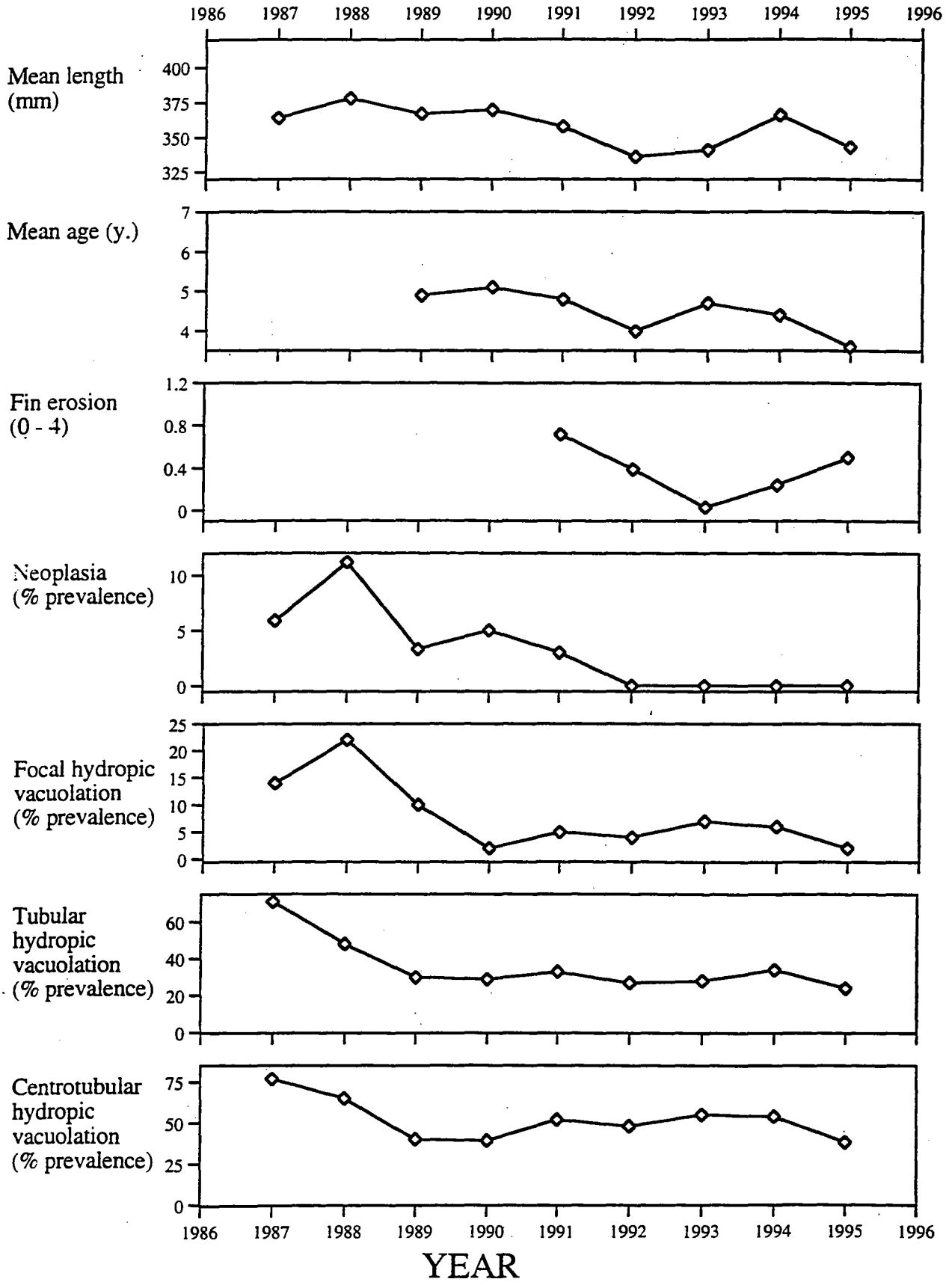
TABLE 3-22

Mercury and Lead Concentrations (µg/kg dry weight) in 1995 Predeployment, Deer Island and Discovery Mussels.

Sandwich Predeployment Mussels	Concentration (µg/g)	Discovery Mussels	Concentration (µg/g)	Deer Island Mussels	Concentration (µg/g)
Mercury:					
M9511S8000TC1	0.045	M9511S6000TC1	0.229	M9511S1000TC1	0
M9511S8000TC2	0.084	M9511S6000TC2	0.031	M9511S1000TC2	0.02
M9511S8000TC3	0.063	M9511S6000TC3	0.058	M9511S1000TC3	0.07
M9511S8000TC4	0.073	M9511S6000TC4	0.069	M9511S1000TC4	0.11
M9511S8000TC5	0.059	M9511S6000TC5	0.008	M9511S1000TC5	0.08
		M9511S6000TC5	0.01		
Mean	0.064	Mean	0.068	Mean	0.056
Standard deviation	0.01	Standard deviation	0.080	Standard deviation	0.040
Min. concentration	0.045	Min. concentration	0.008	Min. concentration	0.000
Max. concentration	0.084	Max. concentration	0.229	Max. concentration	0.110

Gloucester Predeployment Mussels	Concentration (µg/g)	Discovery Mussels	Concentration (µg/g)	Deer Island Mussels	Concentration (µg/g)
Lead:					
M9511G7000TC1	5.61	M9511G6000TC1	6.93	M9511G1000TC1	8.35
M9511G7000TC2	6.25	M9511G6000TC2	8.12	M9511G1000TC2	10.70
M9511G7000TC2	5.97	M9511G6000TC3	8.73	M9511G1000TC3	5.93
M9511G7000TC3	7.31	M9511G6000TC4	8.80	M9511G1000TC3	5.96
M9511G7000TC4	5.17	M9511G6000TC5	10.10	M9511G1000TC4	8.51
M9511G7000TC5	6.04			M9511G1000TC5	8.49
Mean	6.06	Mean	8.54	Mean	7.99
Standard deviation	0.72	Standard deviation	1.15	Standard deviation	1.81
Min. concentration	5.17	Min. concentration	6.93	Min. concentration	5.93
Max. concentration	7.31	Max. concentration	10.1	Max. concentration	10.7

DEER ISLAND

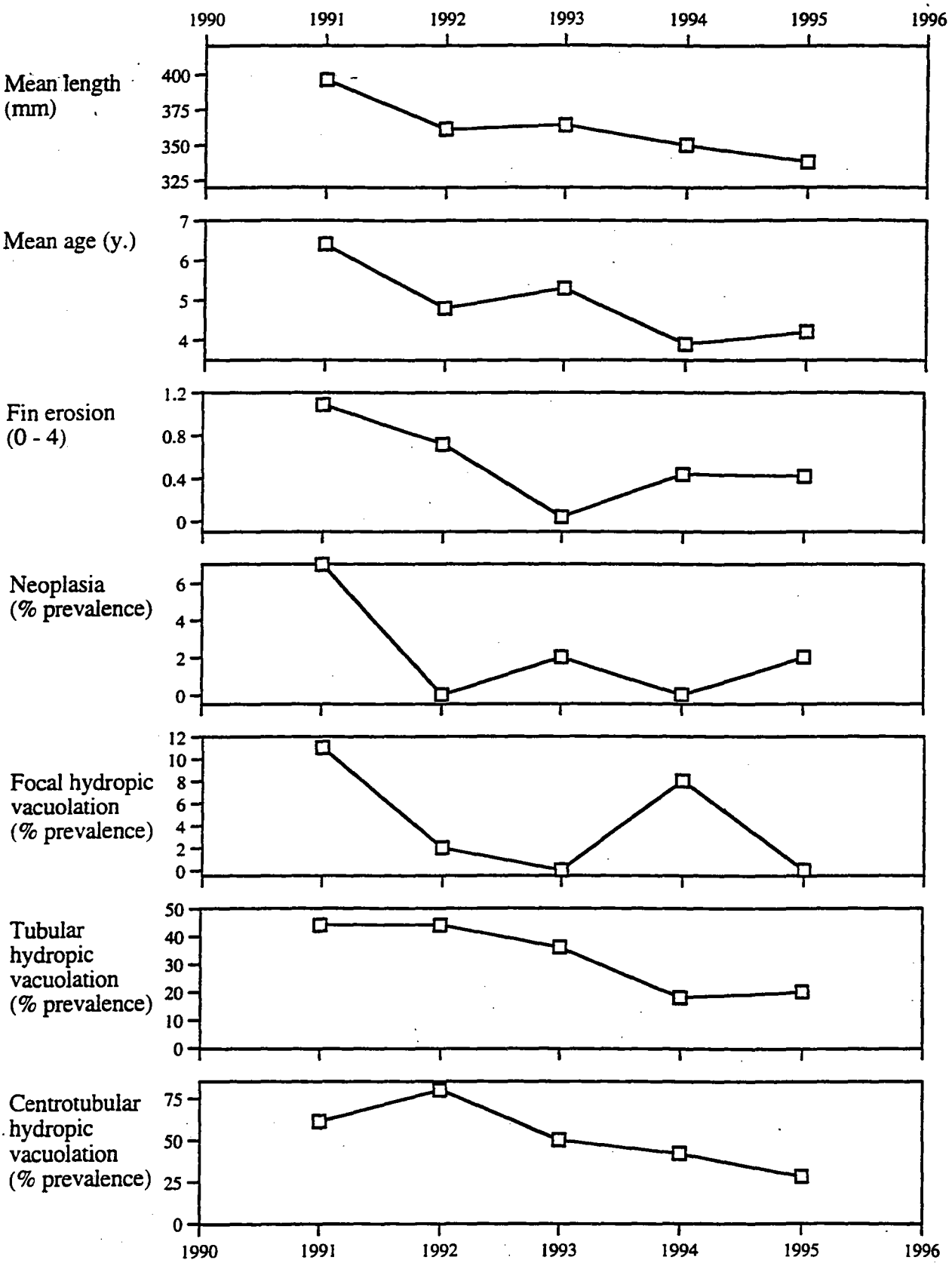


Liver slices examined per fish: 1 for 1987 to 1990, 3 for 1991 to 1993

FIGURE 3-1a

Temporal Changes in Lesion Prevalence and Physical Characteristics in Deer Island Flats Winter Flounder

BROAD SOUND

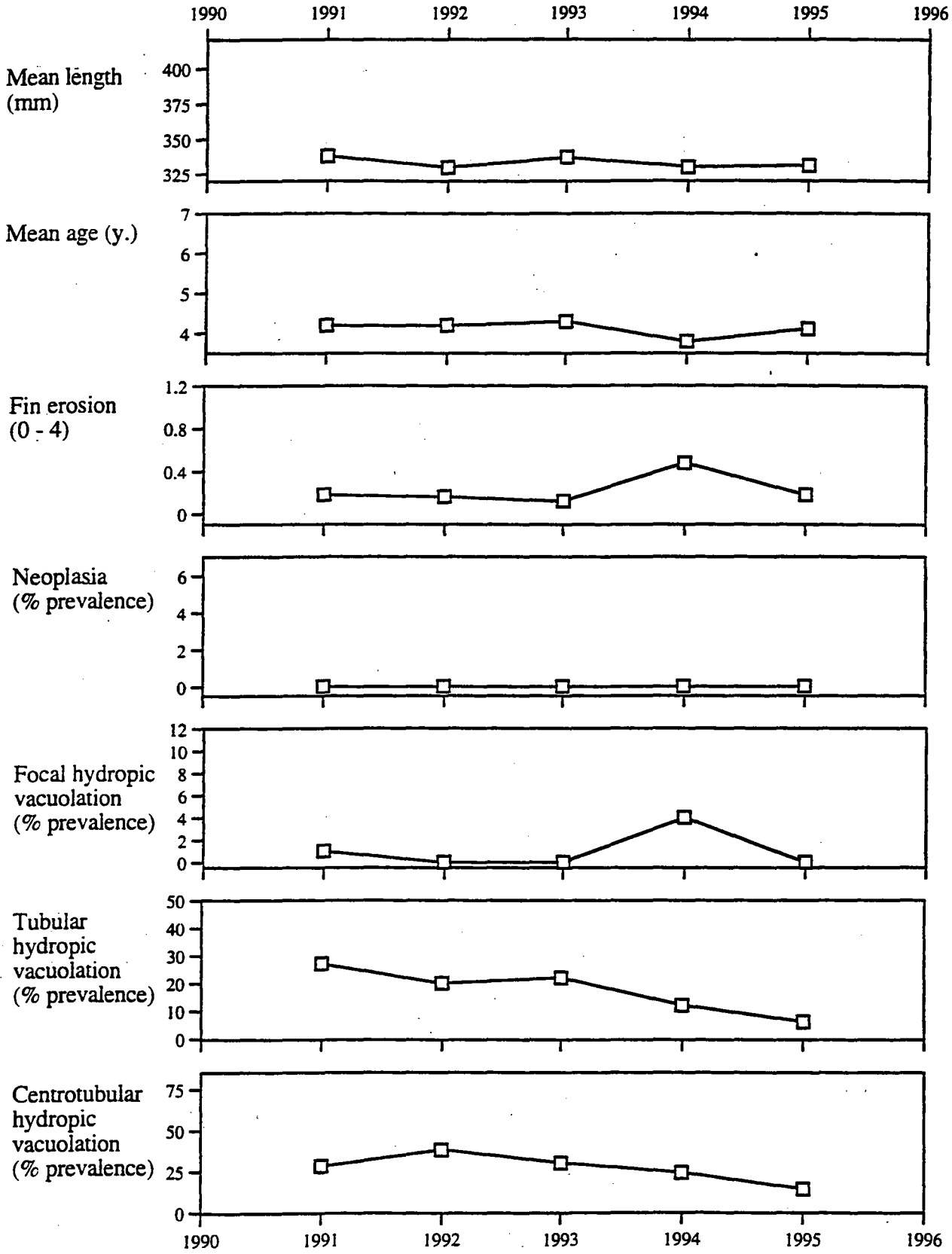


YEAR

FIGURE 3-1b

Temporal Changes in Lesion Prevalence and Physical Characteristics
Broad Sound Winter Flounder

FUTURE OUTFALL SITE

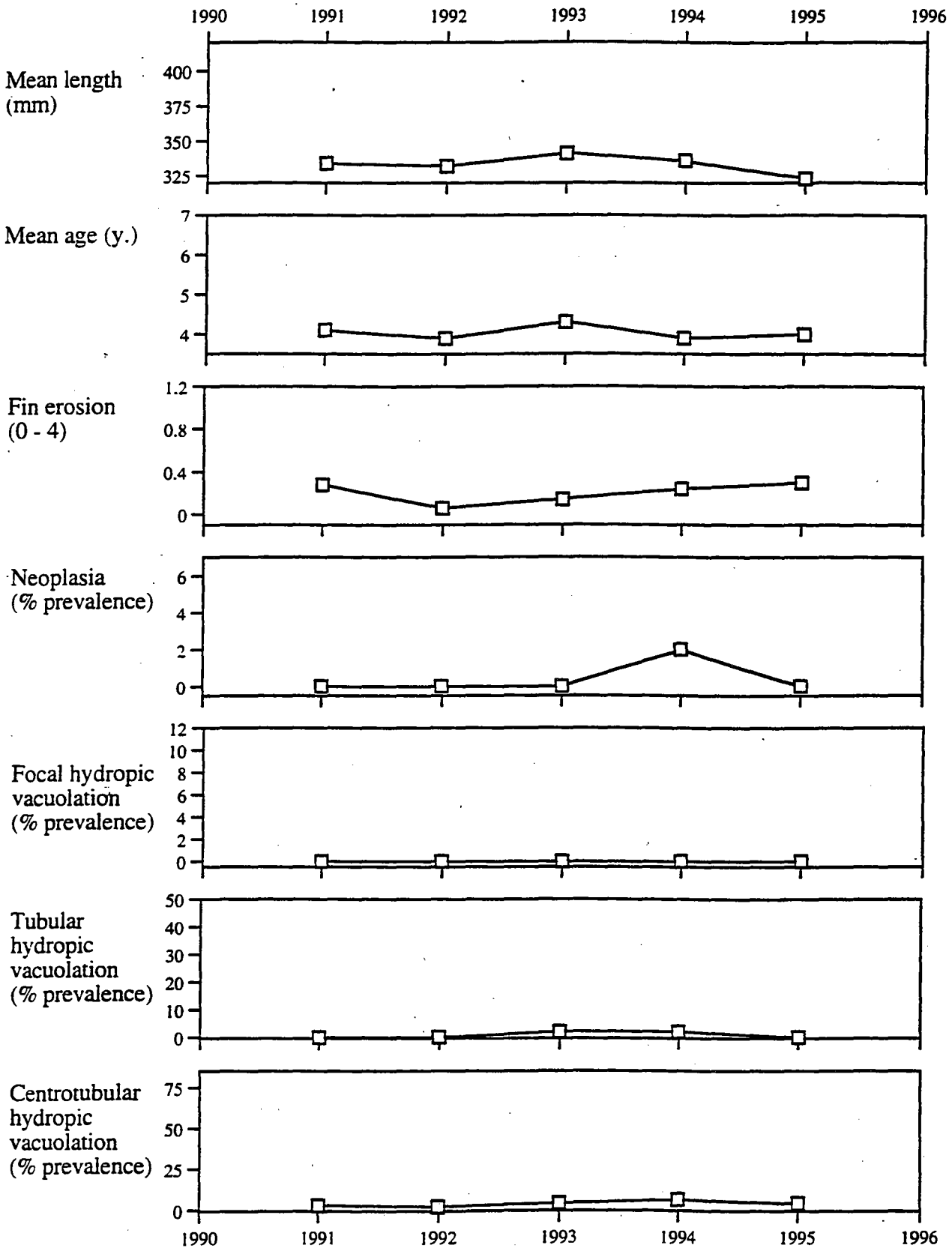


YEAR

FIGURE 3-1c

Temporal Changes in Lesion Prevalence and Physical Characteristics in Future Outfall Site Winter Flounder

EASTERN CAPE COD BAY

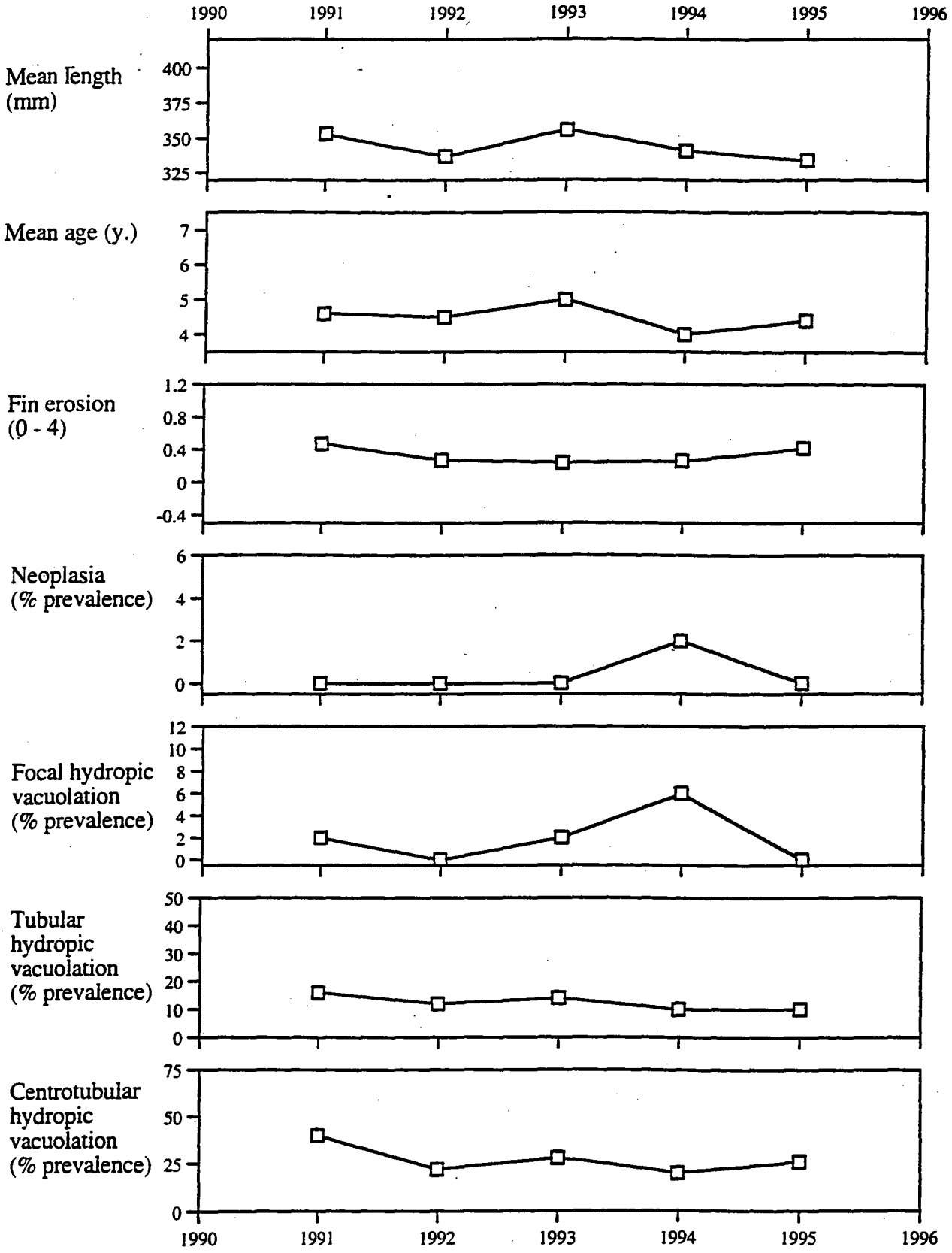


YEAR

FIGURE 3-1d

Temporal Changes in Lesion Prevalence and Physical Characteristics
Eastern Cape Cod Bay Winter Flounder

NANTASKET BEACH



YEAR

FIGURE 3-1e

'Temporal Changes in Lesion Prevalence and Physical Characteristics in Nantasket Beach Winter Flounder

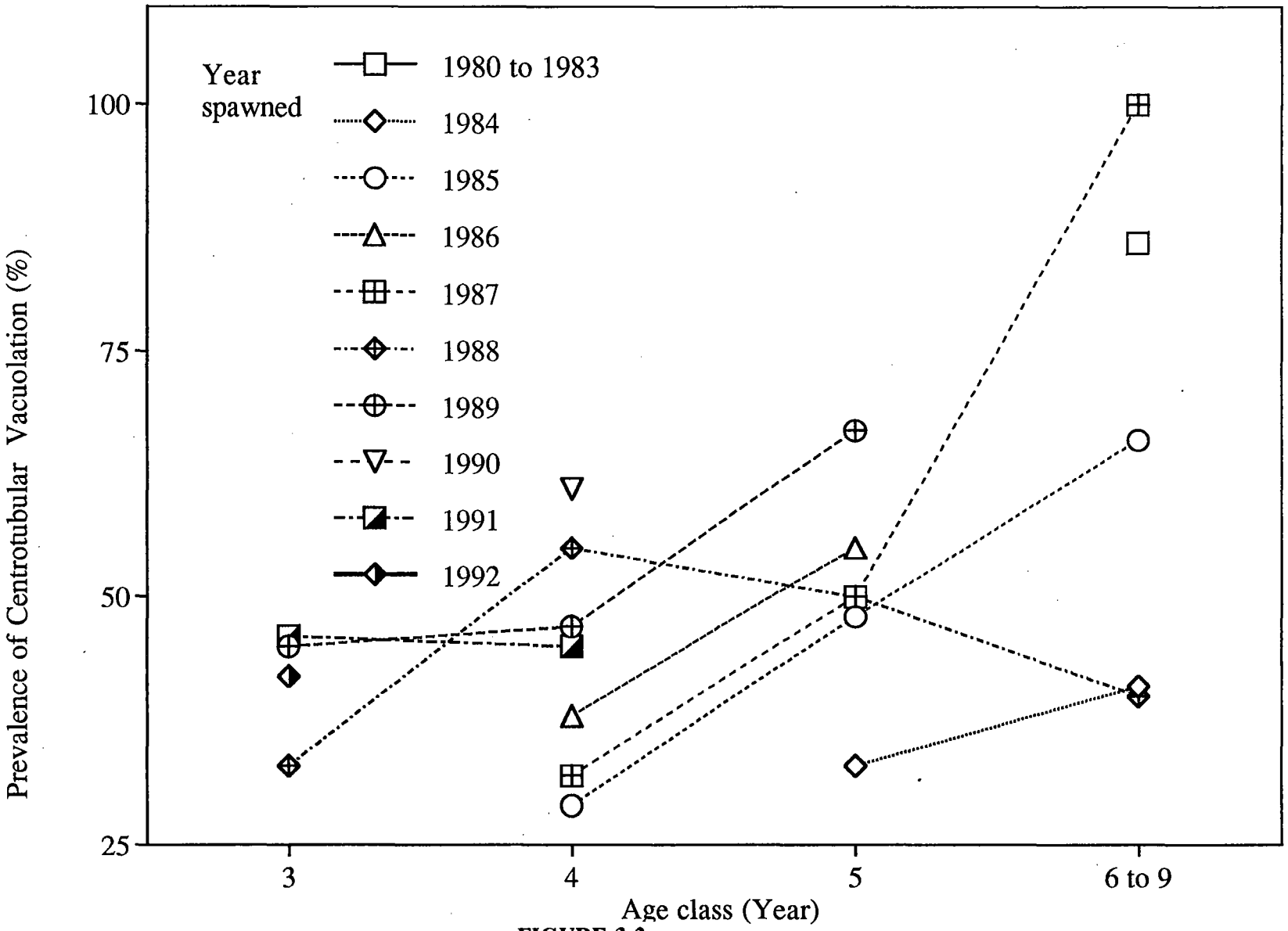


FIGURE 3-2

Progression of hydropic vacuolation through sucessive cohorts of winter flounder. Data points with N<5 omitted.

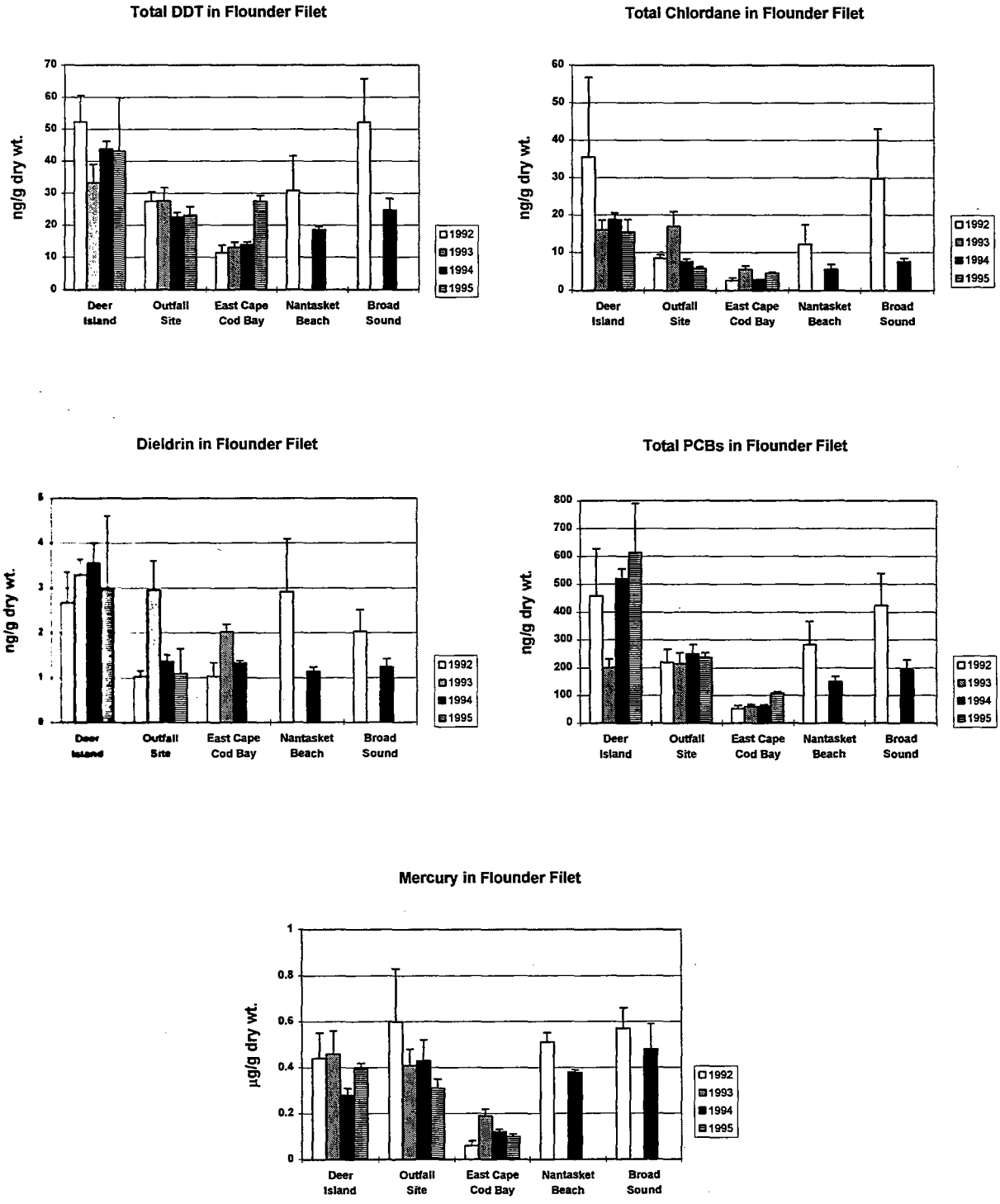


Figure 3-3
Comparison of Target Analytes in Flounder Filet, 1992-1995

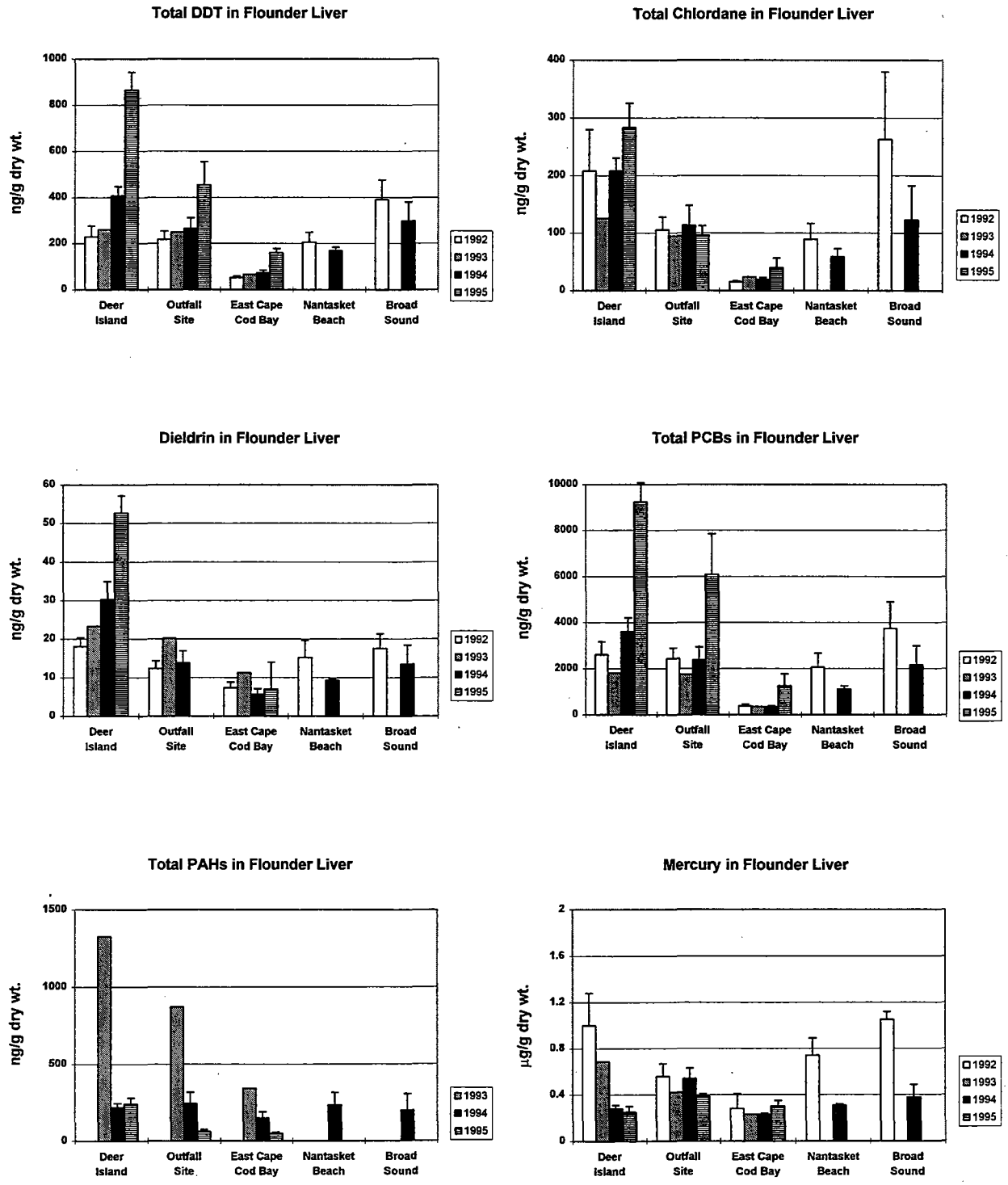


Figure 3-4
Comparison of Target Analytes in Flounder Liver, 1992-1995

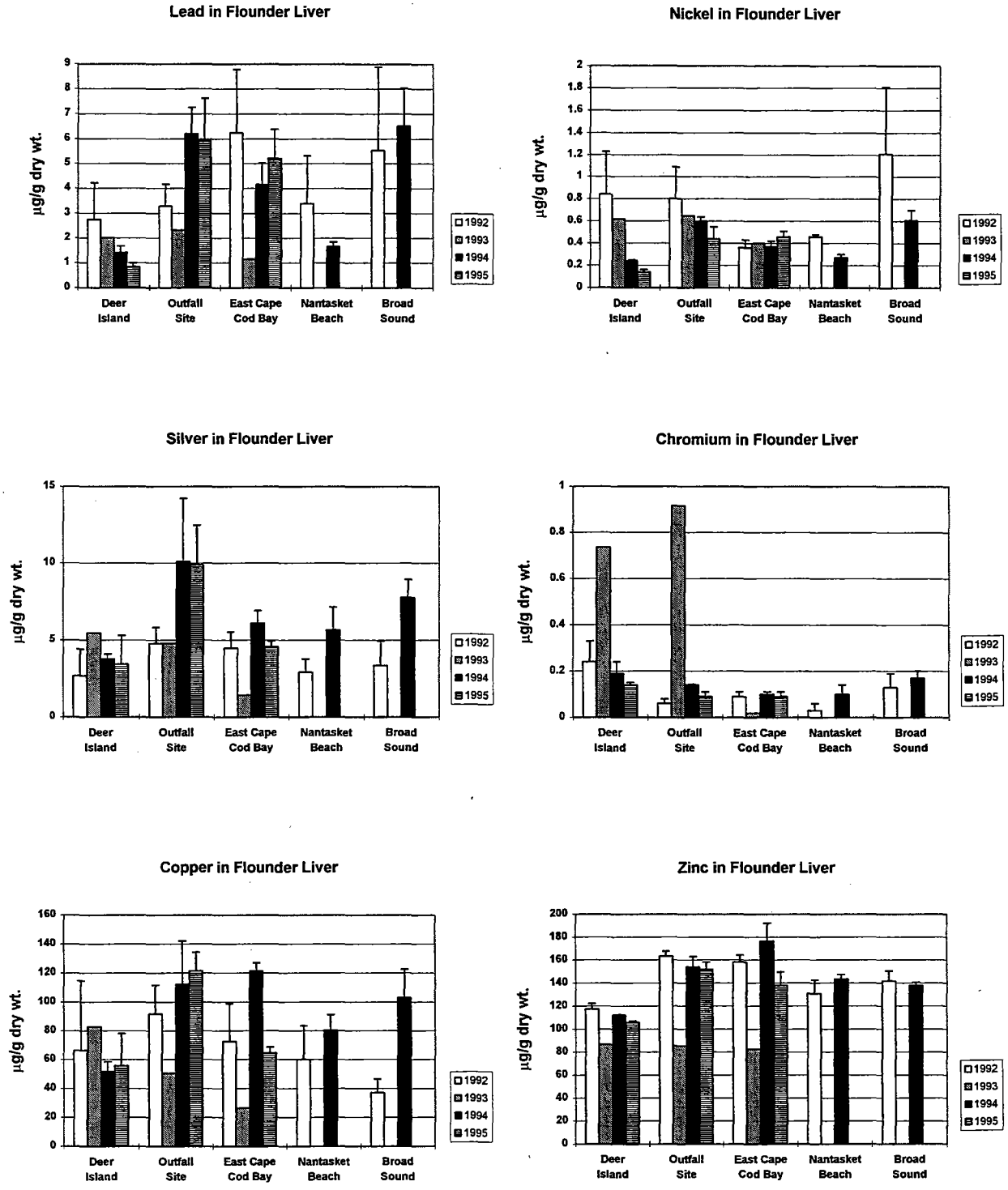


Figure 3-5
Comparison of Trace Metals in Flounder Liver, 1992-1995

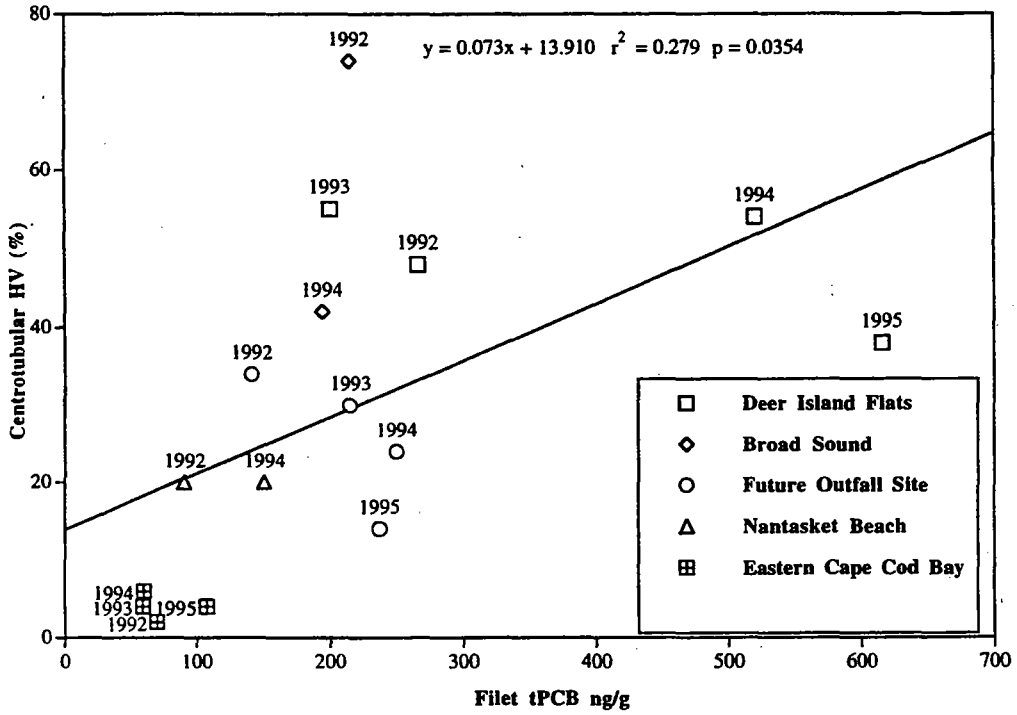


FIGURE 3-6a
Prevalence of Centrotubular HV Versus Winter Flounder Filet Concentrations of PCBs: 1992-1995

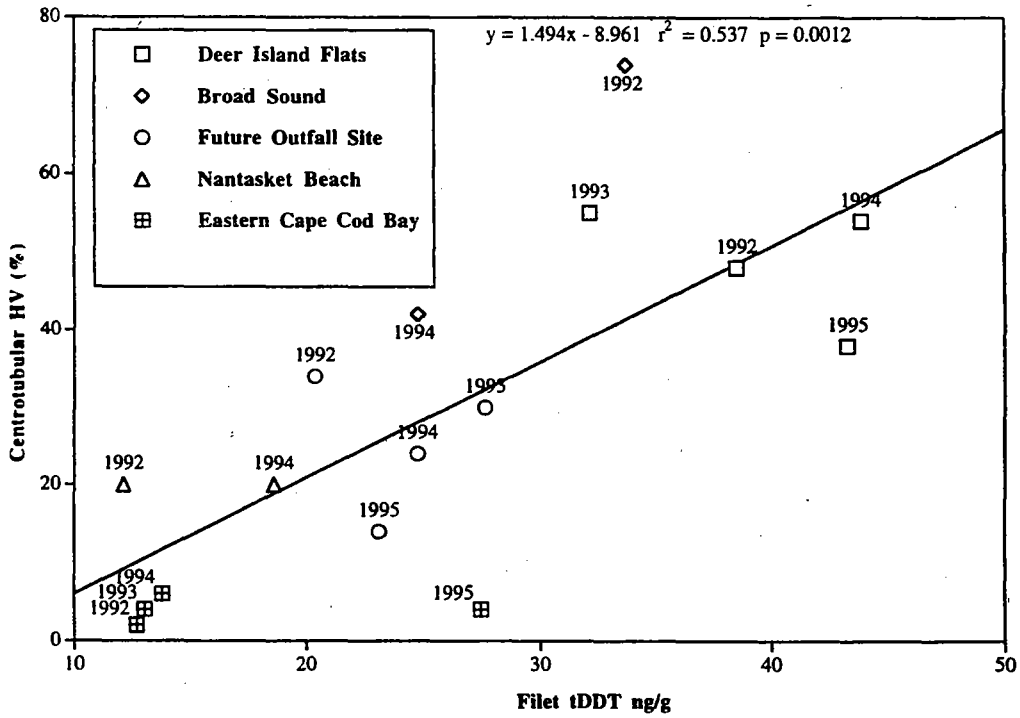


FIGURE 3-6b
Prevalence of Centrotubular HV Versus Winter Flounder Filet Concentrations of DDT: 1992-1995

Pooled available data for all stations from 1992 to 1995. Each data point compares, for a single station in a single year, vacuolation prevalence from 29 to 50 fish with a mean chemical concentration in individual or pools of 7 to 15 fish. Linear regression for all data points. (Data from this study, Moore & Stegeman 1993; Shea 1993; Hillman et al. 1994; Hillman & Peven 1995).

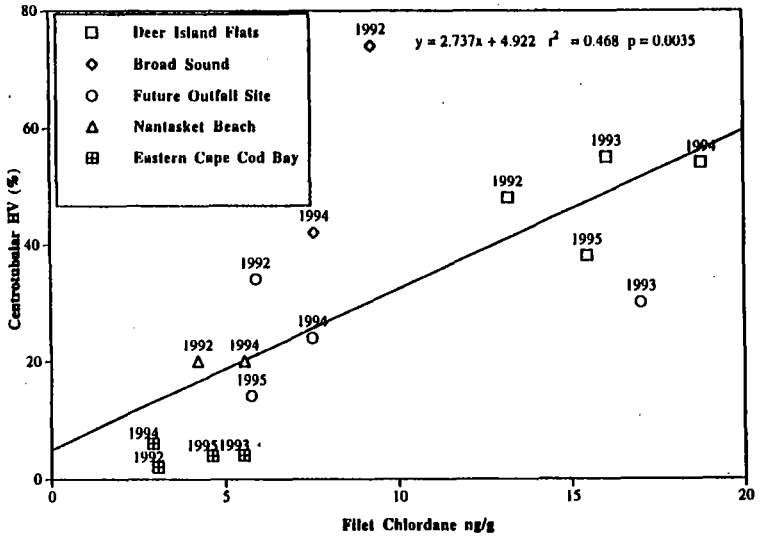


FIGURE 3-6c
Prevalence of Centrotubular HV Versus Winter Flounder Filet Concentrations of Chlordane: 1992-1995

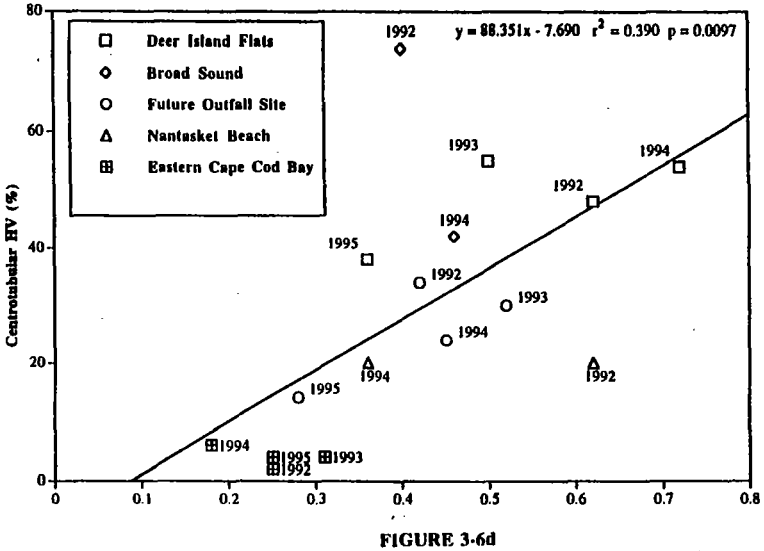


FIGURE 3-6d
Prevalence of Centrotubular HV Versus Winter Flounder Filet Concentrations of Mirex: 1992-1995

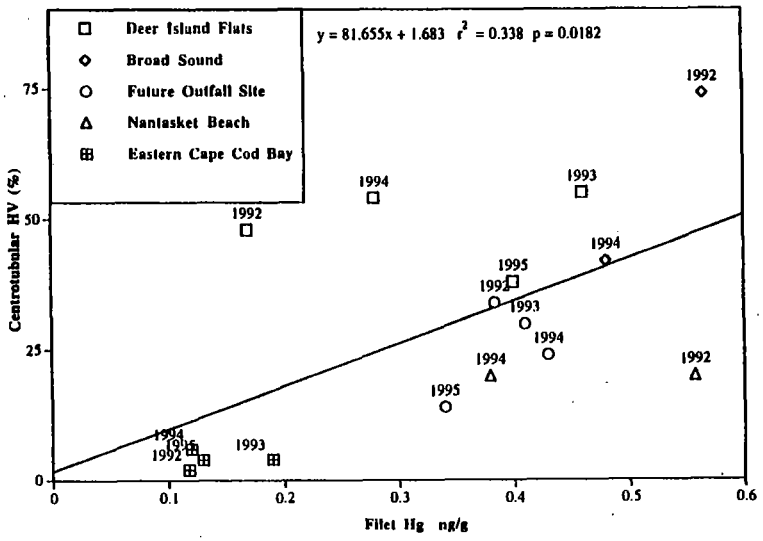


FIGURE 3-6e
Prevalence of Centrotubular HV Versus Winter Flounder Filet Concentrations of Mercury 1992 - 1995

Pooled available data for all stations from 1992 to 1995. Each data point compares, for a single station in a single year, vacuolation prevalence from 29 to 50 fish with a mean chemical concentration in individual or pools of 7 to 15 fish. Linear regression for all data points. (Data from this study, Moore & Stegeman 1993; Shea 1993; Hillman et al. 1994; Hillman & Peven 1995).

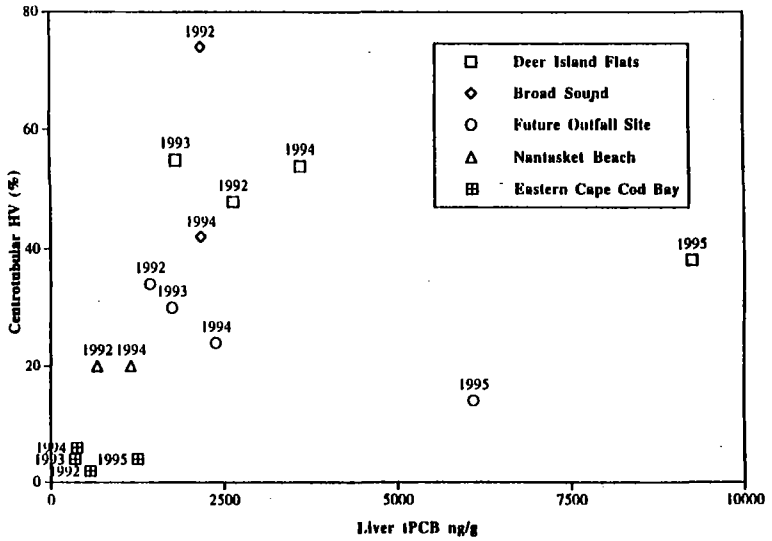


FIGURE 3-7a
Prevalence of Centrotubular HV Versus Winter Flounder Liver Concentrations of PCBs: 1992-1995

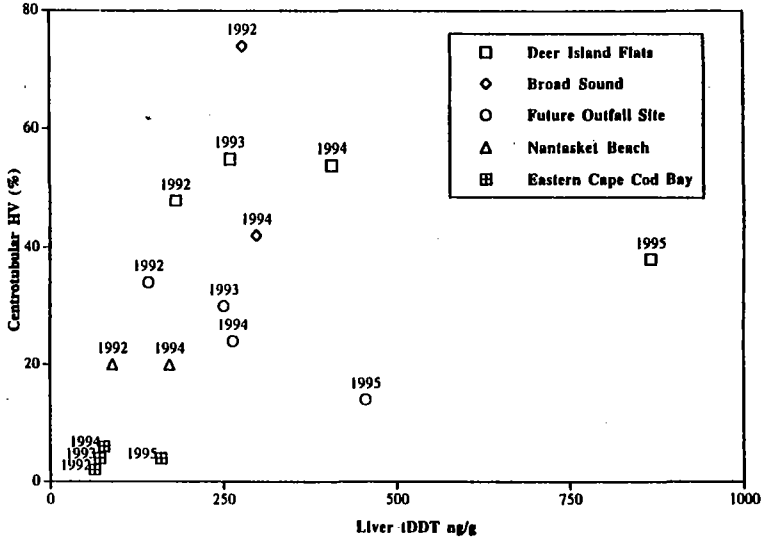


FIGURE 3-7b
Prevalence of Centrotubular HV Versus
Winter Flounder Liver Concentrations of Total DDT 1992 --1995

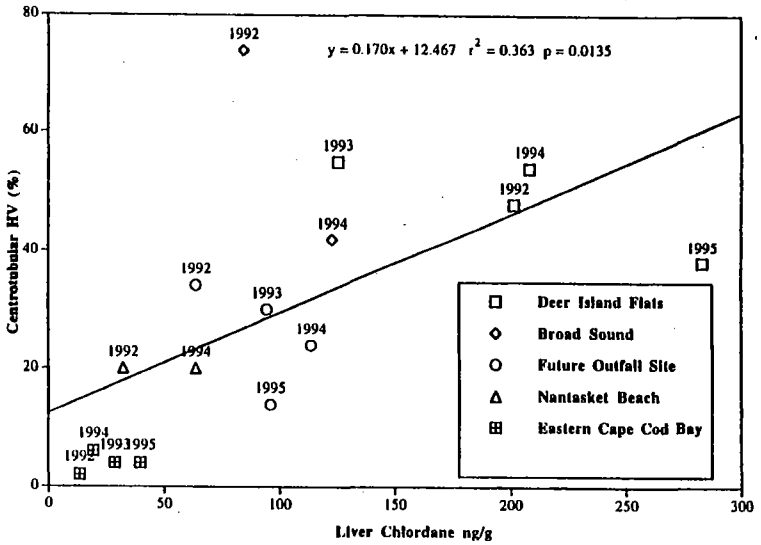


FIGURE 3-7c
Prevalence of Centrotubular HV Versus
Winter Flounder Liver Concentrations of Chlordene 1992 - 1995

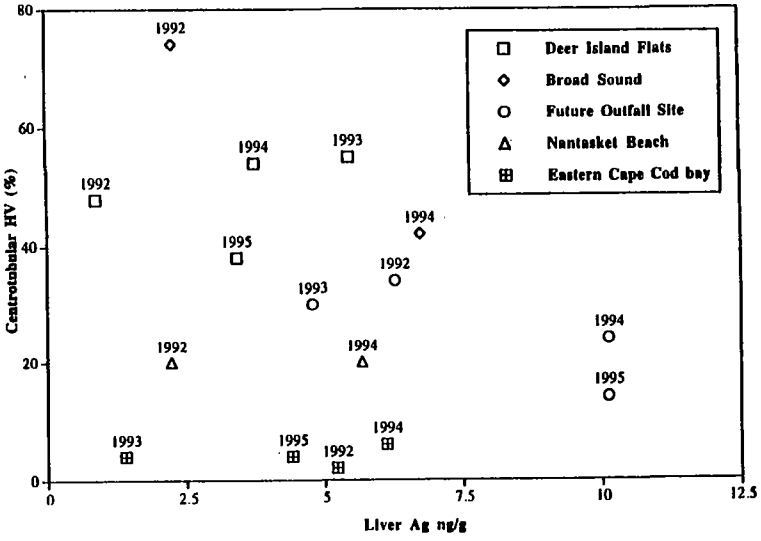
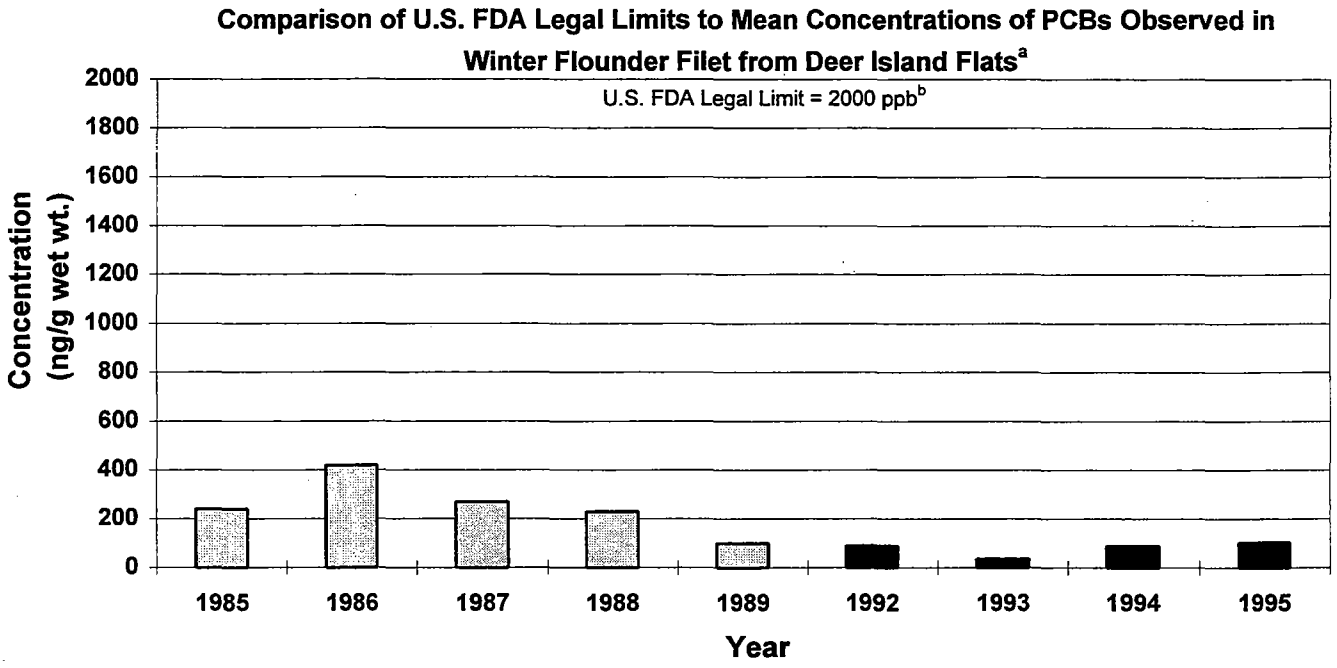
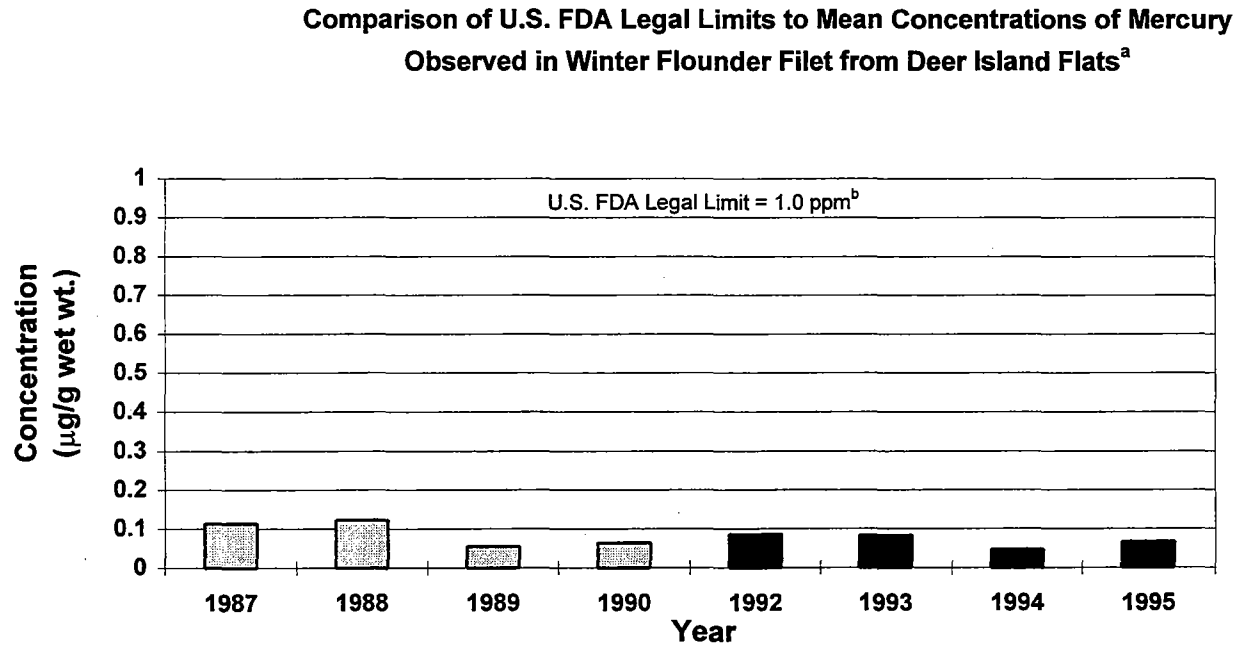


FIGURE 3-7d
Prevalence of Centrotubular HV Versus
Winter Flounder Liver Concentrations of Silver 1992 - 1995

Pooled available data for all stations from 1992 to 1995. Each data point compares, for a single station in a single year, vacuolation prevalence from 29 to 50 fish with a mean chemical concentration in individual or pools of 7 to 15 fish. Linear regression for all data points. (Data from this study, Moore & Stegeman 1993; Shea 1993; Hillman et al. 1994; Hillman & Peven 1995).





[a] 1985-1991 data is from Schwartz et al., *PCBs in Winter Flounder, American Lobster, and Bivalve Molluscs from Boston Harbor, Salem Harbor and Coastal Massachusetts:1984-1989* or *Metal Concentrations in Winter Flounder, American Lobster, and Bivalve Molluscs from Boston Harbor, Salem Harbor and Coastal Massachusetts.*, 1991. 1992-1994 data are from Hillman and Peven, *1994 Annual Fish and Shellfish Report*, 1995. 1995 data are presented in this report.

[b] U.S.EPA, *Assessing Human Health Risks From Chemically Contaminated Fish and Shellfish*, 1989.

Figure 3-8
Comparison of Winter Flounder Tissue Concentrations of PCBs and Mercury to U.S. FDA Legal Limits

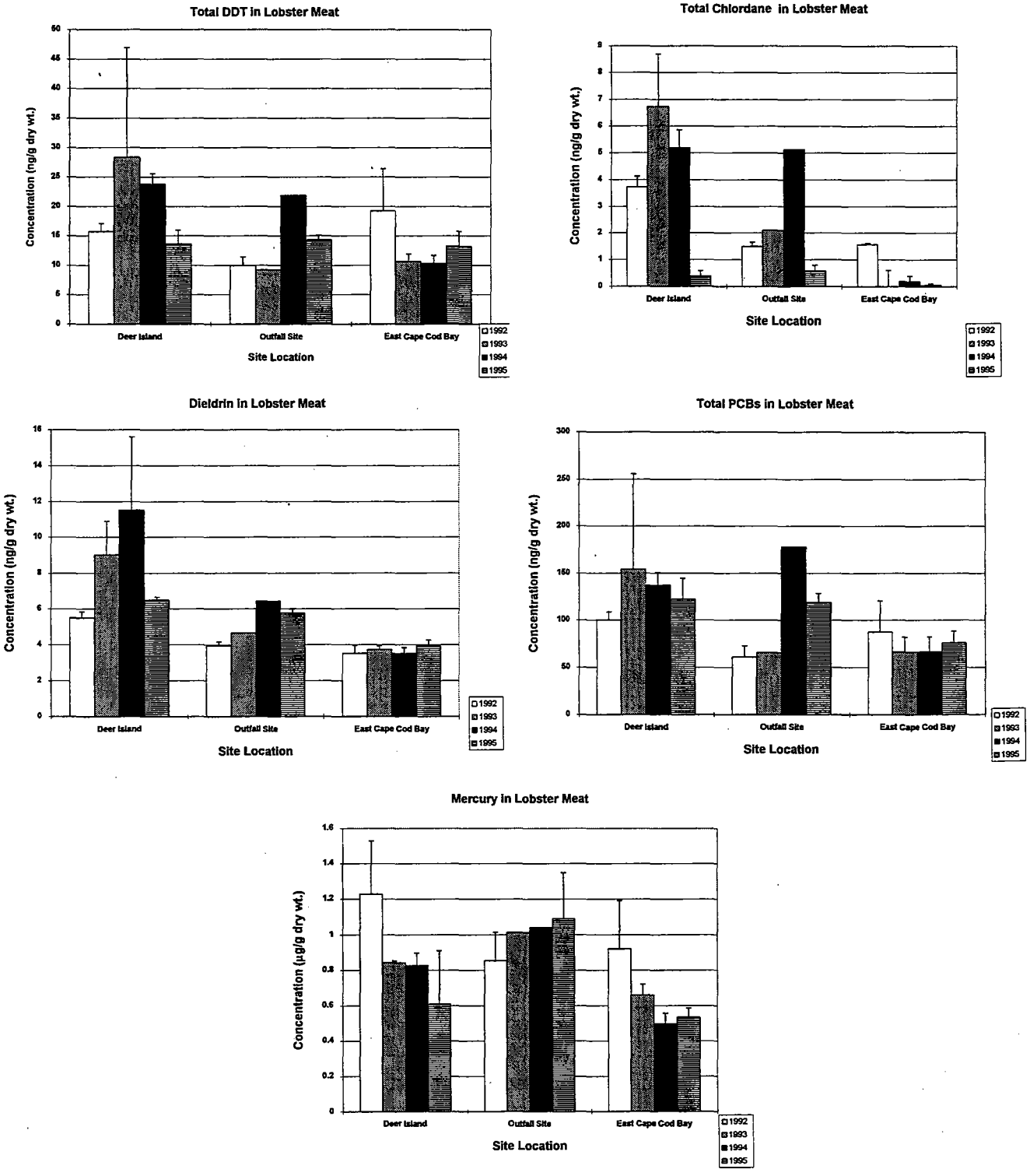


Figure 3-9
Comparison of Target Analytes in Lobster Meat 1992-1995

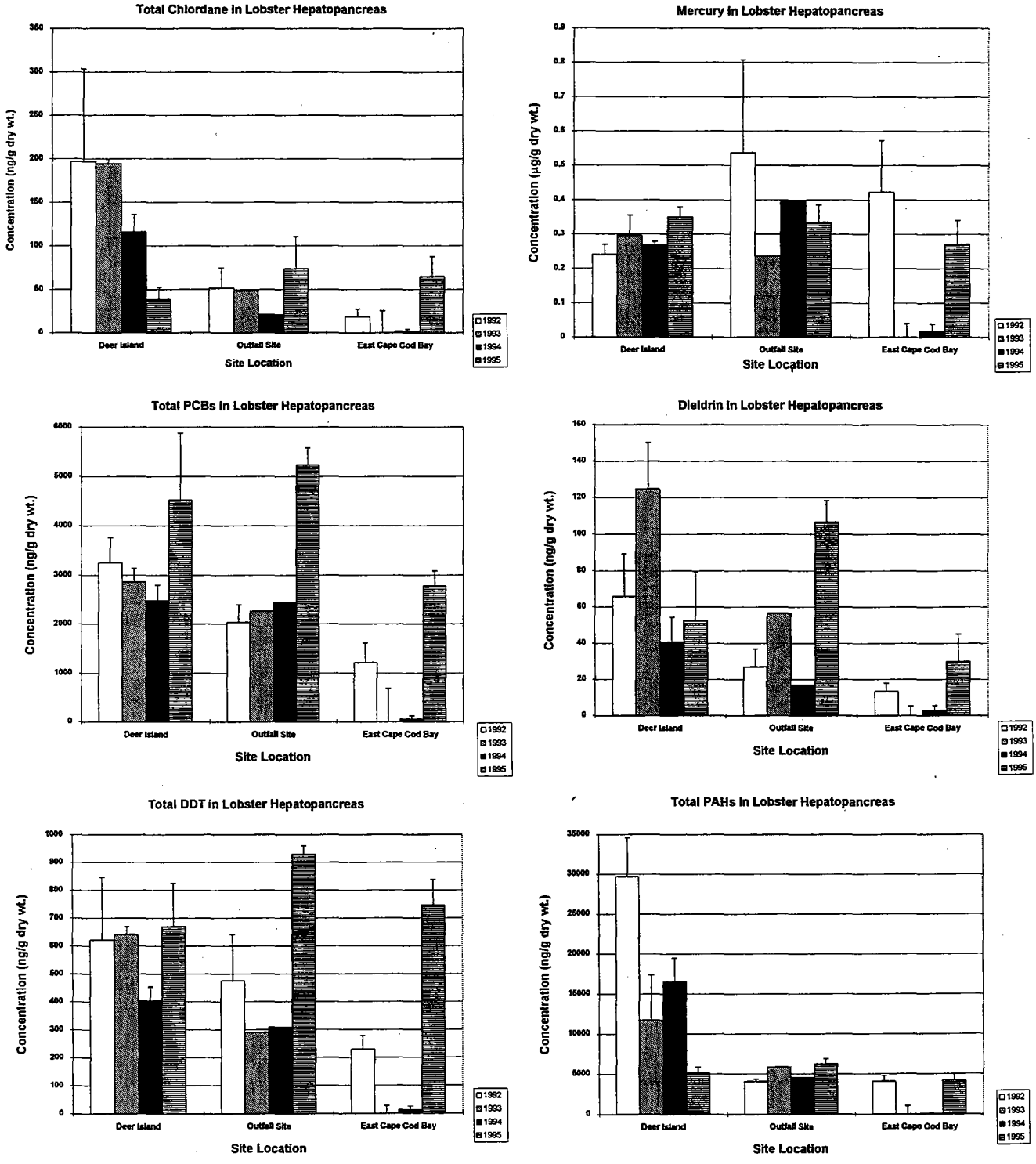


Figure 3-10
Comparison of Target Analytes in Lobster Hepatopancreas, 1992-1995

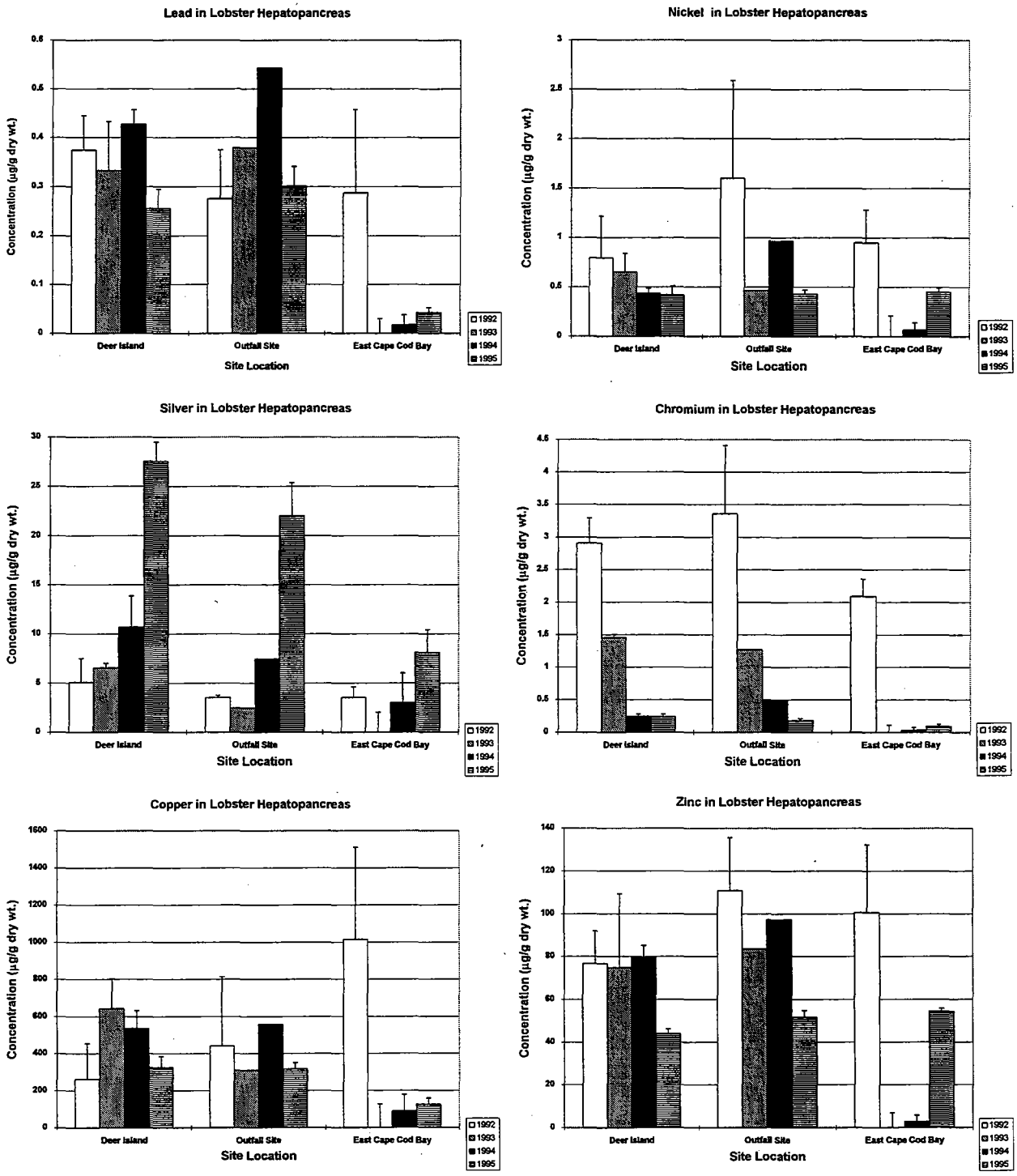
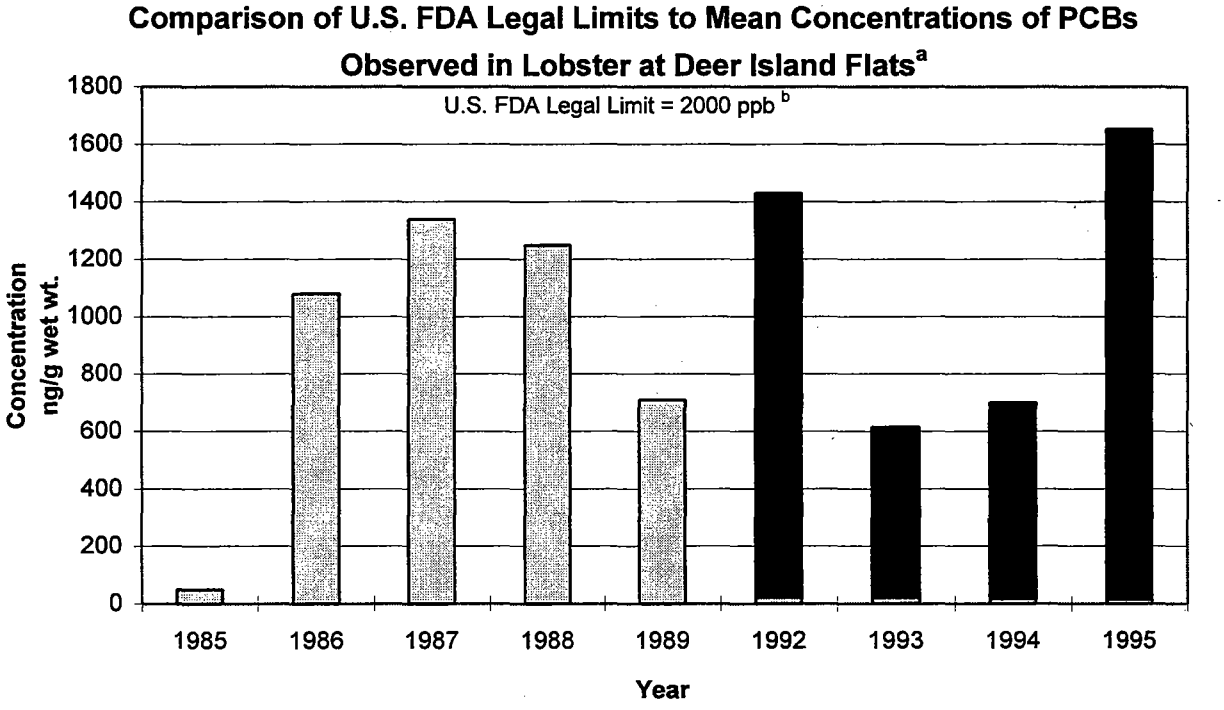
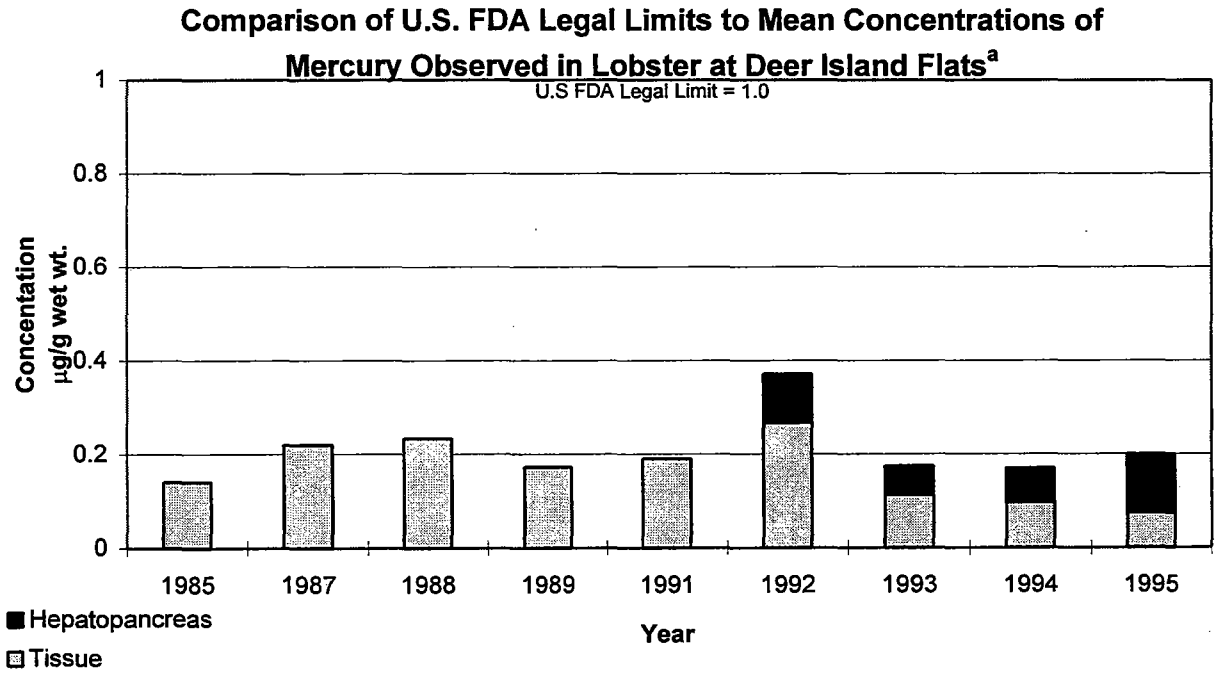


Figure 3-11
Comparison of Trace Metals in Lobster Hepatopancreas, 1992-1995





[a] 1985 data (Wallace et al., *Analysis of Contaminants in Marine Resources*, 1988) include analyses for lobster claws and tail. 1987 to 1991 data (Schwartz et al., *Metal Concentrations in Winter Flounder, American Lobster, and Bivalve Molluscs from Boston Harbor, Salem Harbor and Coastal Massachusetts* or *PCBs in Winter Flounder, American Lobster, and Bivalve Molluscs from Boston Harbor, Salem Harbor and Coastal Massachusetts:1984-1989*. 1991) include tails, claws, hepatopancreas and gonads. 1992-1994 data (Hillman and Peven, *1994 Annual Fish and Shellfish Report*, 1995) as well as 1995 data (presented in this report), include tail meat only.

[b] U.S.EPA, *Assessing Human Health Risks From Chemically Contaminated Fish and Shellfish*, 1989.

Figure 3-12
Comparison of Available Data to U.S. FDA Legal Limits for PCBs and Mercury

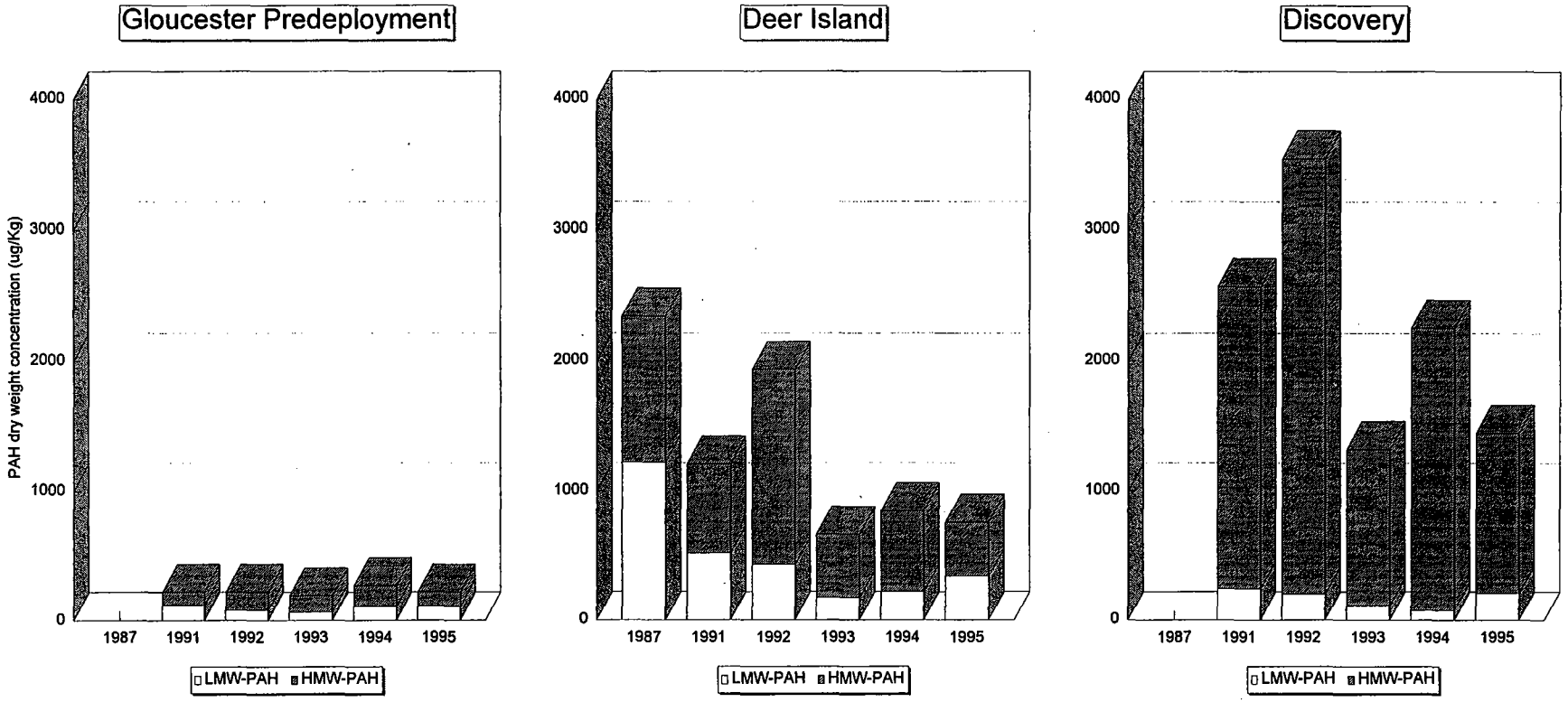


FIGURE 3-13 Annual average PAH body burdens for 1995 deployed mussels.

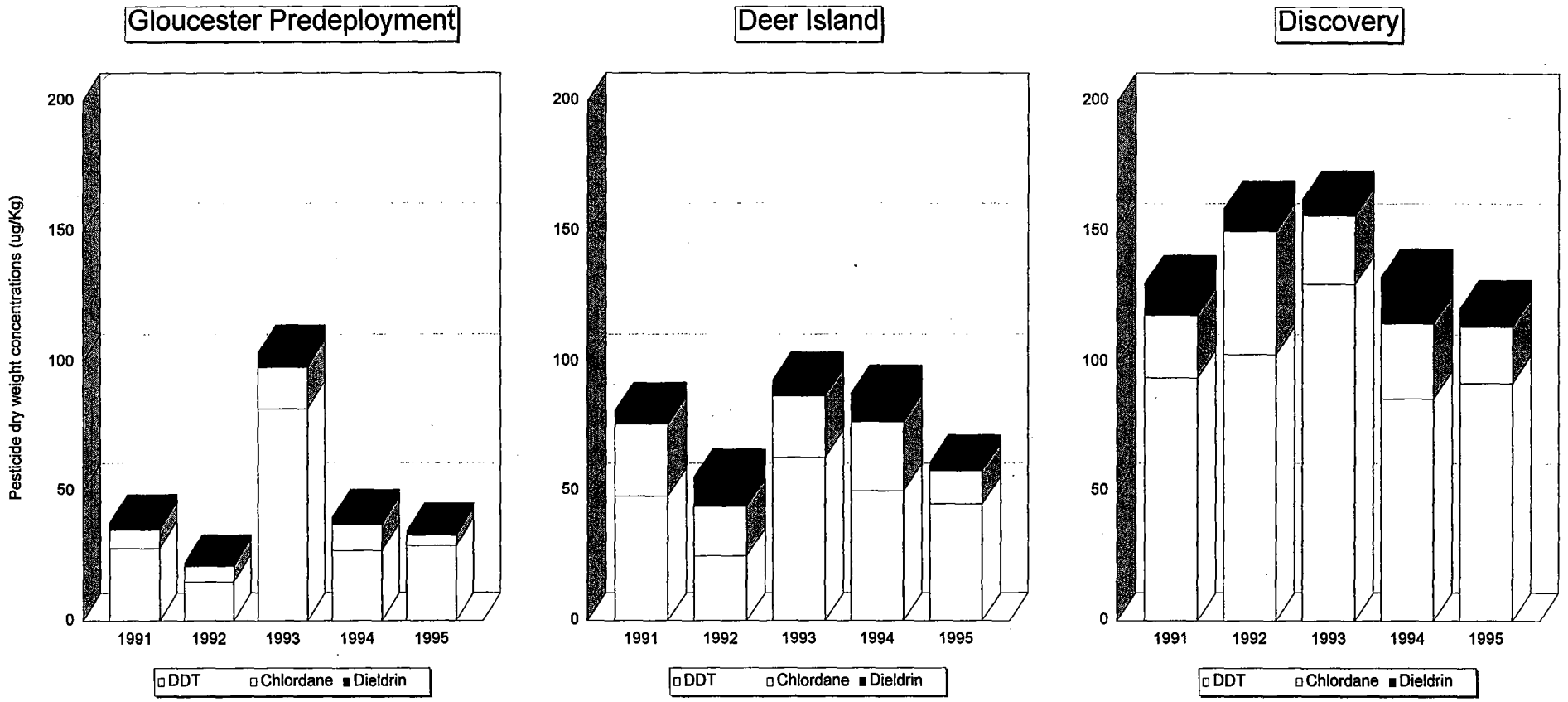


FIGURE 3-14 Annual average pesticide body burdens for 1995 deployed mussels.

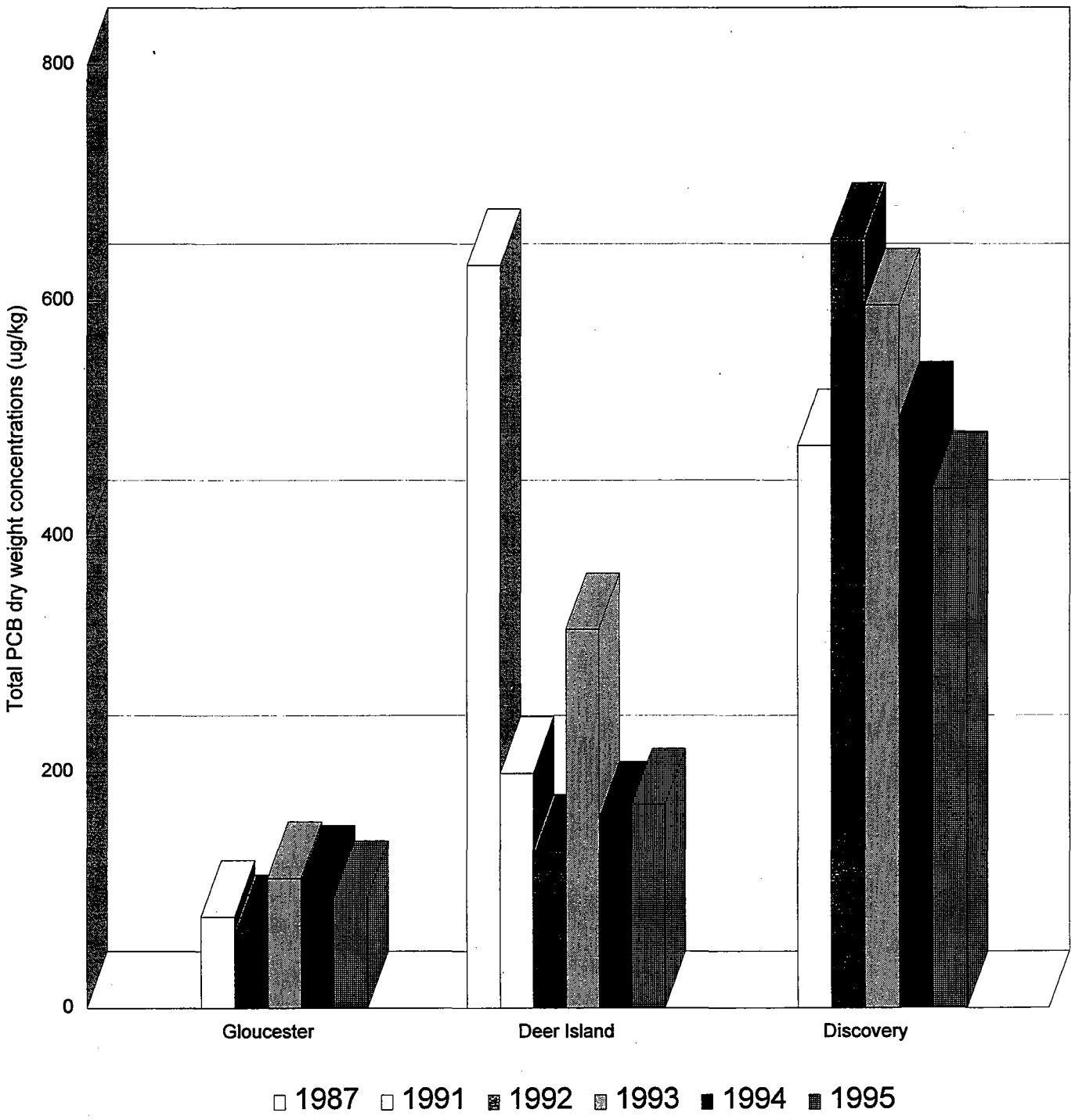


FIGURE 3-15 Average annual PCB body burdens for mussels harvested from three stations.

4.0 CONCLUSIONS

The 1995 Fish and Shellfish Monitoring program was successful in providing another year's worth of data to better document the existing, pre-effluent diversion conditions. Compared to previous years, most measured parameters were comparable to past efforts. There were some exceptions to this trend, including pesticides and PCBs in flounder liver and lobster hepatopancreas, that warranted further investigation of analytical variation. More importantly, the tissue contaminant levels in edible portions of flounder and lobster were detected at levels well below the FDA legal limits nor did they exceed the warning or action levels. Further comments on the surveys are given below.

4.1 Winter Flounder

The 1995 Flounder Survey was conducted consistent with previous years' surveys. However, the difficulty of obtaining flounder at Deer Island Flat may be of concern for future surveys (since no augmentation by Fish Day is available). A statistically significant decreasing relationship between prevalence of neoplasm and survey year was noted in the years 1987-1995. Whether this trend is linked to improvement in environmental conditions is an important monitoring issue.

The levels of tissue contaminant levels were comparable to the previous three years of data, with the already noted exceptions of pesticides and PCBs. Fillet tissue contaminant concentrations were below the FDA legal limits for pesticides, PCBs and mercury (as well as action and warning limits), thus indicating no risk for human consumption.

Analyses of the last four years of data (1992-1995) histology and tissue chemistry indicate a series of statistically significant relationships between centrotubular hydropic vacuolation and tissue contaminant levels for selected analytes. These suspected causal relationships between contaminant exposure (body burdens) and detectable precursors of neoplasm should be further evaluated in following years to check any corresponding changes in prevalence and body burden following effluent diversion at FOS.

4.2 Northern Lobster

The 1995 Lobster Survey collected specimens from the three sampling locations through a combination of trapping and direct shipboard collection from lobstermen. Analyses of the lobster tissue contaminant levels found that body burdens were generally greatest at Deer Island Flat and lowest at the Eastern Cape Cod reference station. Comparison of 1995 data with previous years (1992-1994) indicates the most tissue levels were comparable to the range previously observed. The exceptions were higher levels of PCBs and pesticides in the hepatopancreas and lower levels of chlordane in the lobster edible tissue. Lobster edible

tissue contaminant concentrations were below the FDA legal limits and action and warning limits for pesticides, PCBs and mercury, thus indicating no risk for human consumption.

4.3 Blue Mussel

The 1995 Mussel Bioaccumulation study provided data on the Discovery and Deer Island site, but not at the Future Outfall file due to accidental loss of caged arrays. Although the 1995 PAH results were slightly confounded by differences in analytical technique, the spatial and temporal trends observed were consistent with previous 1990's studies. The LMW-PAHs were highest in the Deer Island mussels while HMW-PAHs were highest in *Discovery* mussels. NOAA total PAH body burden for all sites were comparable to 1991-1994 levels. Pesticide (DDT, chlordane, dieldrin) levels were also comparable to earlier data.

Lead tissue concentrations were statistically greater in the *Discovery* mussels. Mercury was not significantly different among the three locations and mussel tissue concentrations were uniformly low. These low concentrations, specifically in Deer Island mussels, contrasted with other studies, particularly the NOAA Mussel Watch Project for indigenous Deer Island mussels. Further evaluation of mercury body burdens may be necessary to fully evaluate the bioavailability of mercury to deployed mussels at the three locations.

4.4 Evaluation of Monitoring Hypotheses

An integral part of the MWRA fish and shellfish monitoring is a periodic re-evaluation of the adequacy of the current program to fulfill the overall goals of the monitoring program. In particular, this means a re-examination of the adequacy and effectiveness of the underlying monitoring hypotheses to answer questions regarding the potential effects of the relocated MWRA effluent.

Of the six monitoring hypotheses (MWRA, 1995), four are associated with the potential for edible tissue (flounder, lobster, mussel) to exceed warning levels for mercury, lead, or PCBs. These four hypotheses appear to be sufficient. Current tissue concentrations are generally an order of magnitude or more below warning and FDA regulatory levels. Values approaching the warning or action levels should be readily detectable in the program. Similarly, the monitoring hypothesis regarding future increases of the prevalence of flounder liver CHV at FOS relative to baseline levels measured in outer Boston Harbor also appears to be sufficiently sensitive to detect trends, based on current data.

The remaining monitoring hypothesis, detection of a three-year trend in elevated bioaccumulative lipophilic contaminants, was further evaluated due to greater-than-anticipated levels of some organic contaminants (pesticides, PCBs) in flounder liver and liver hepatopancreas. A review of the magnitude of interlaboratory analytical performance was conducted and concluded that differences in analytical data

between laboratories are negligible (see Appendix A). Since interlaboratory variation does not appear responsible for the observed increases in some organic compounds, this monitoring hypothesis will require further study and evaluation but for the present is considered sufficient.

Overall, it appears that the six monitoring hypotheses associated with the fish and shellfish monitoring program are sufficient and do not require replacement. However, these hypotheses should be revisited on a annual basis as data becomes available to check their effectiveness (e.g., if additional data indicate large amounts of interannual variability ascribed to natural variation).

5.0 RECOMMENDATIONS

Evaluation of the 1995 Fish and Shellfish tasks indicates that the program is achieving its monitoring goals. However, refinements to the program may be warranted. Based on the 1995 results, several recommendations for future effort are suggested:

- To distinguish between natural variability and potential inter-laboratory variability in measurement of organic compounds, comparison between past and current laboratories using standardized reference material or archived tissues (Note: this issue has been addressed through comparisons described in Appendix A. Further work may be warranted.);

- Continuation of the use of mussels collected from the Sandwich reference site to evaluate the bioaccumulation of mercury and possibly, lead;

- Collection of lobster should be revisited to evaluate effectiveness of trapping vs. other means of collecting geographically-documented specimens; and

- All monitoring tasks should be reviewed annually to evaluate whether their scope and effort are commensurate with the HOM goals and ability to address monitoring hypotheses.

These recommendations will be reviewed for inclusion in the 1996 Fish and Shellfish Monitoring Program.

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APPENDIX A

INTERLABORATORY COMPARISON OF ORGANIC CONTAMINANT ANALYSES

Intercomparison of organic contaminant analyses

Background.

In 1992 - 1994, samples from MWRA's Outfall Monitoring Program for organic contaminant analyses on flounder and lobster tissues were analyzed by Battelle Ocean Sciences (Batelle). Inchcape/Aquatec carried out tissue analyses on samples from the caged mussel bioaccumulation studies in those years. Data from these analyses are contained in MWRA's EM&MS database, and are detailed in several reports, for example Hillman and Peven (1995) and Downey *et al.* (1995) (References are detailed in the report to which this is forms an appendix).

In 1995, a new team began carrying out the MWRA outfall monitoring program, including all the fish and shellfish tissue chemistry analyses. As a result of this, Arthur D. Little (ADL) became responsible for organic contaminant analyses in tissue samples.

As detailed in the attached report, substantial increases were seen for several tissues and compounds betwen 1994 and 1995 (for example, summed PCB congeners in lobster hepatopancreas samples). During MWRA's science review meetings on May 23-24, 1996, questions were raised by some reviewers as to whether these changes actually represented inter-annual variability in the body burdens, or if the change in analytical laboratories may itself partially explain the apparent changes. One reccomendation made was that results of recent national intercomparison exercises in which the laboratories participated should be looked at.

Following discussions between MWRA, ENSR, and ADL, it was determined that the most recent exercise in which all 3 laboratories were likely to have participated was the *1994 NIST/NOAA NS&T EPA EMAP Intercomparison Exercise Program for Organic Contaminants in the Marine Environment* MWRA therefore contacted all 3 laboratories, asking for the results of their participation in that exercise. Battelle and ADL provided copies of the data resulting from that exercise (Attachment A), along with their laboratory codes from the exercise. Discussions with the laboratories confirmed that these results were generated using the NS&T program protocols incorporated in MWRA's monitoring program.

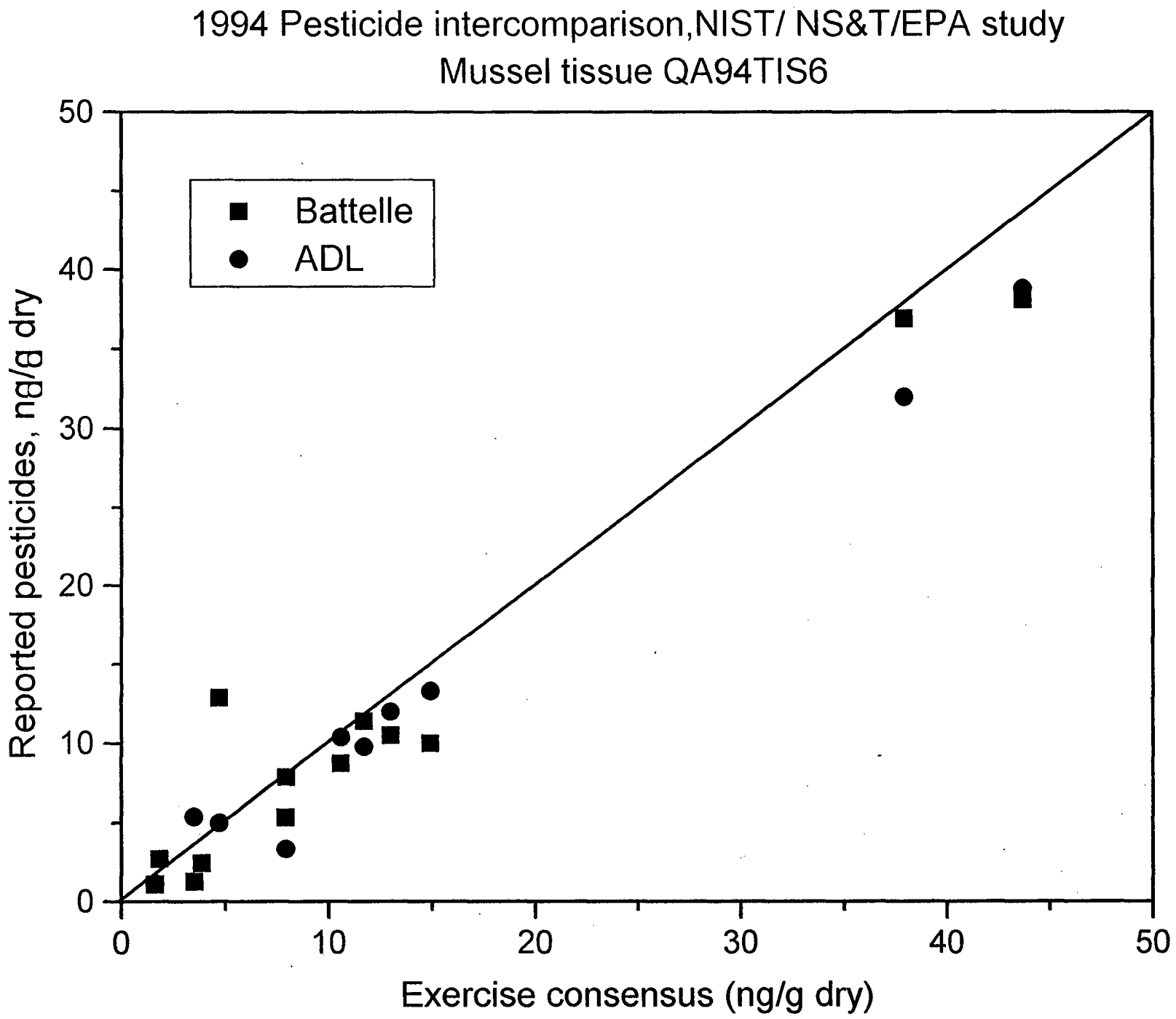
Unfortunately, Inchcape/Aquatec had not participated in that exercise, and therefore cannot be included in this comparison. However, there were few major apparent changes in contaminant body burdens in the caged mussel study between 1994 and 1995, so the lack does not overly constrain this comparison.

Data treatment Organic contaminant data from this exercise generated by Battelle and ADL were keypunched into spreadsheets, as were the results of the exercise consensus (Mean and Standard Deviation). The laboratory results represent the means of 3 replicate analyses. Different spreadsheets were compiled for PCB congeners, PAHs, and for organochlorine pesticides. Keypunching was restricted to those compounds quantified by both Battelle and ADL; if one of those labs identified a compound as below a detection level or as ND, the compound was not included in this comparision. Accuracy of keypunching was verified by double-keypunching all data, followed by an electronic comparison. Following that check a manual 100% check of the

data was carried out against the original tables.

Data were subjected to exploratory analyses. First, for PCBs, pesticides, and PAHs, barcharts were constructed (Figure 1a - 1c), comparing the results of the 2 laboratories to the exercise consensus. Second, scatterplots were constructed (Figure 2a - 2c), with the exercise consensus on the x-axis and the individual laboratories' results on the y-axis. Third, Pearson product-moment correlation coefficients were calculated between the laboratories, and between each lab and the exercise consensus (Table 1).

Results The congener by congener results for Battelle and ADL closely track each other, and are very consistent with the exercise consensus (fig. 1, Fig. 2, Table 1). The differences between the labs in this exercise were substantially smaller than the scale of the differences seen for contaminants in some tissue types between 1994 and 1995, suggesting that the monitoring results reflect actual interannual variability.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Battelle
ADL
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Exercise consensus (ng/g dry)
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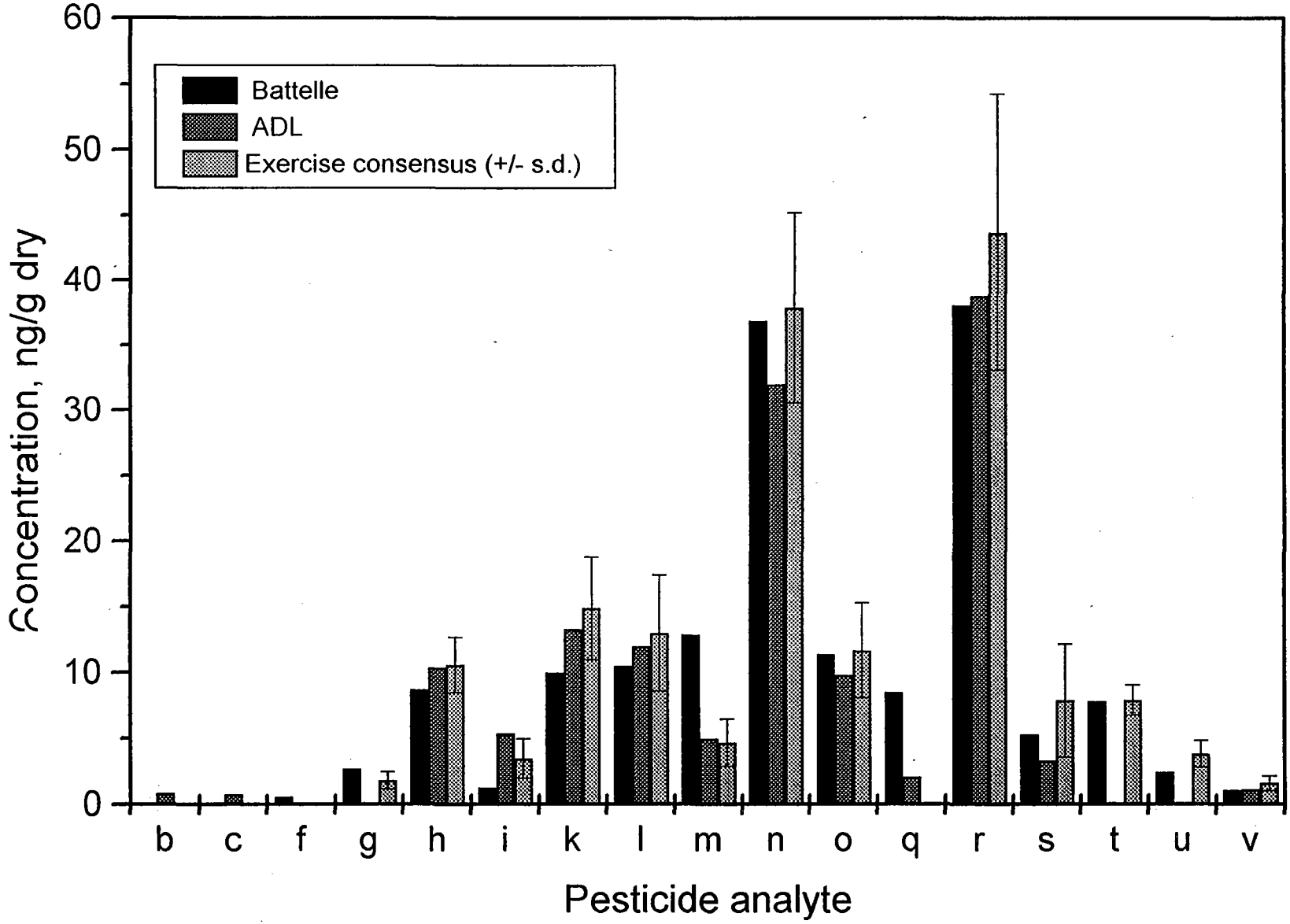
10
20
30
40
50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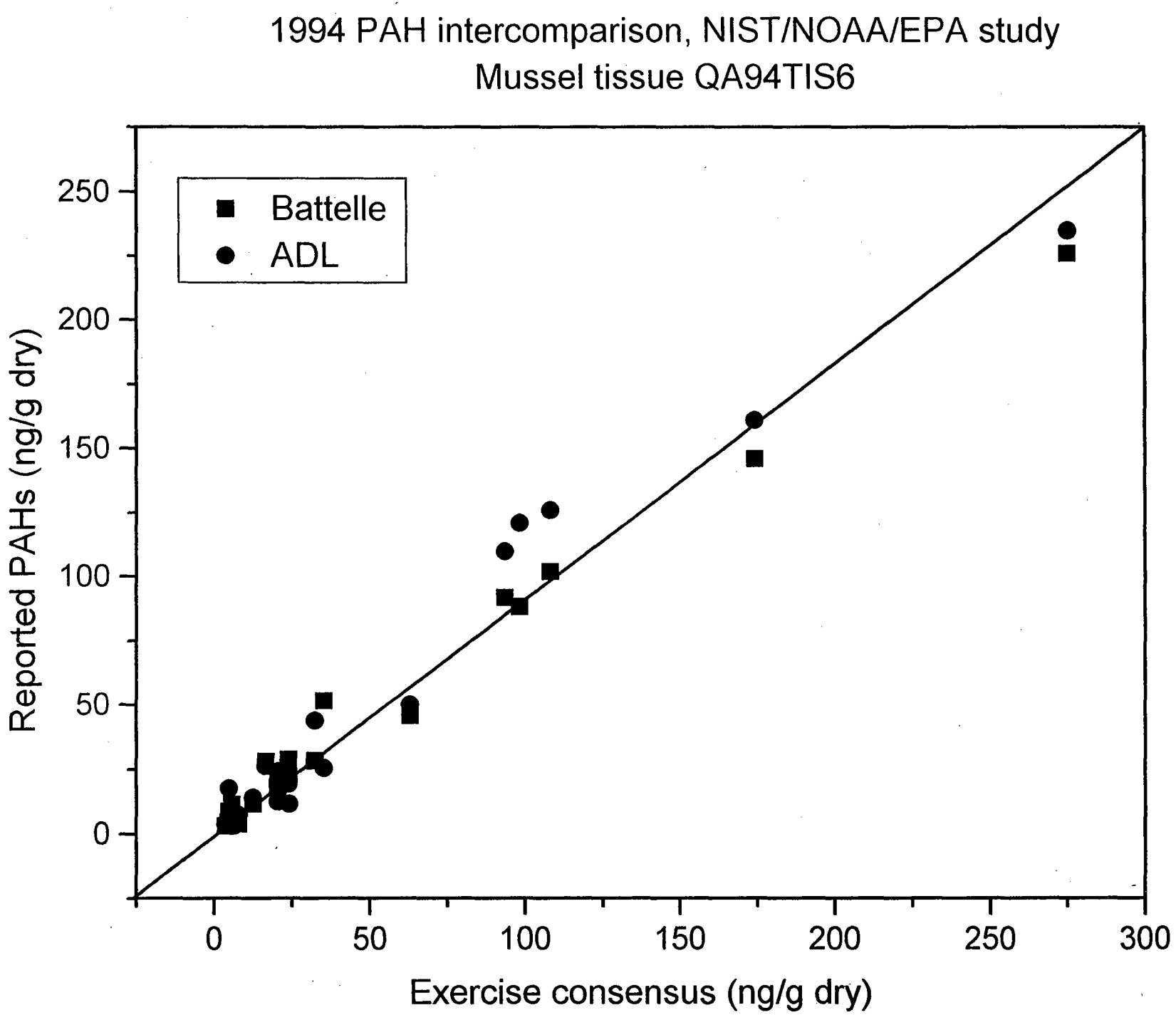
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ADL
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Exercise consensus (ng/g dry)
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50
100
150
200
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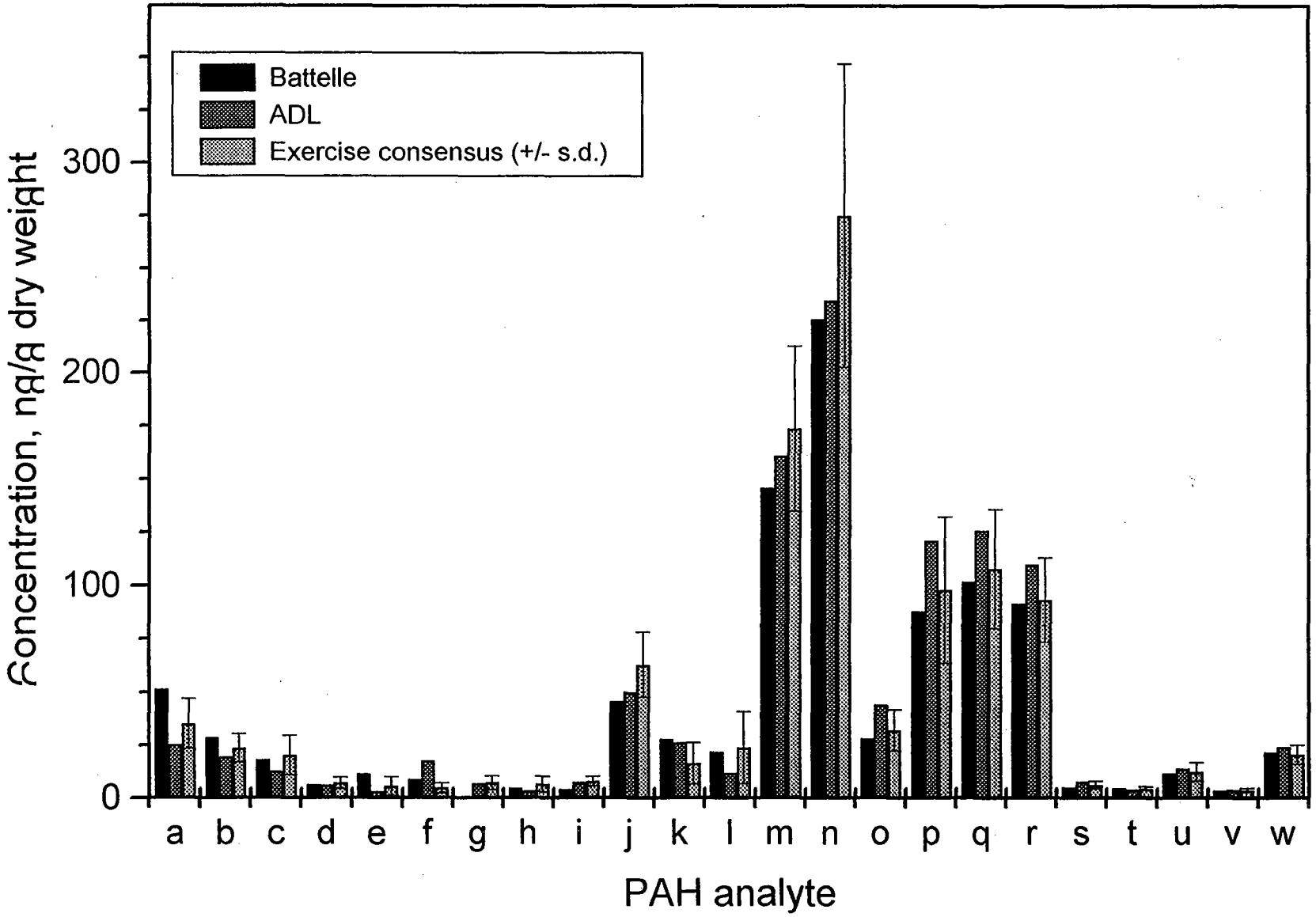

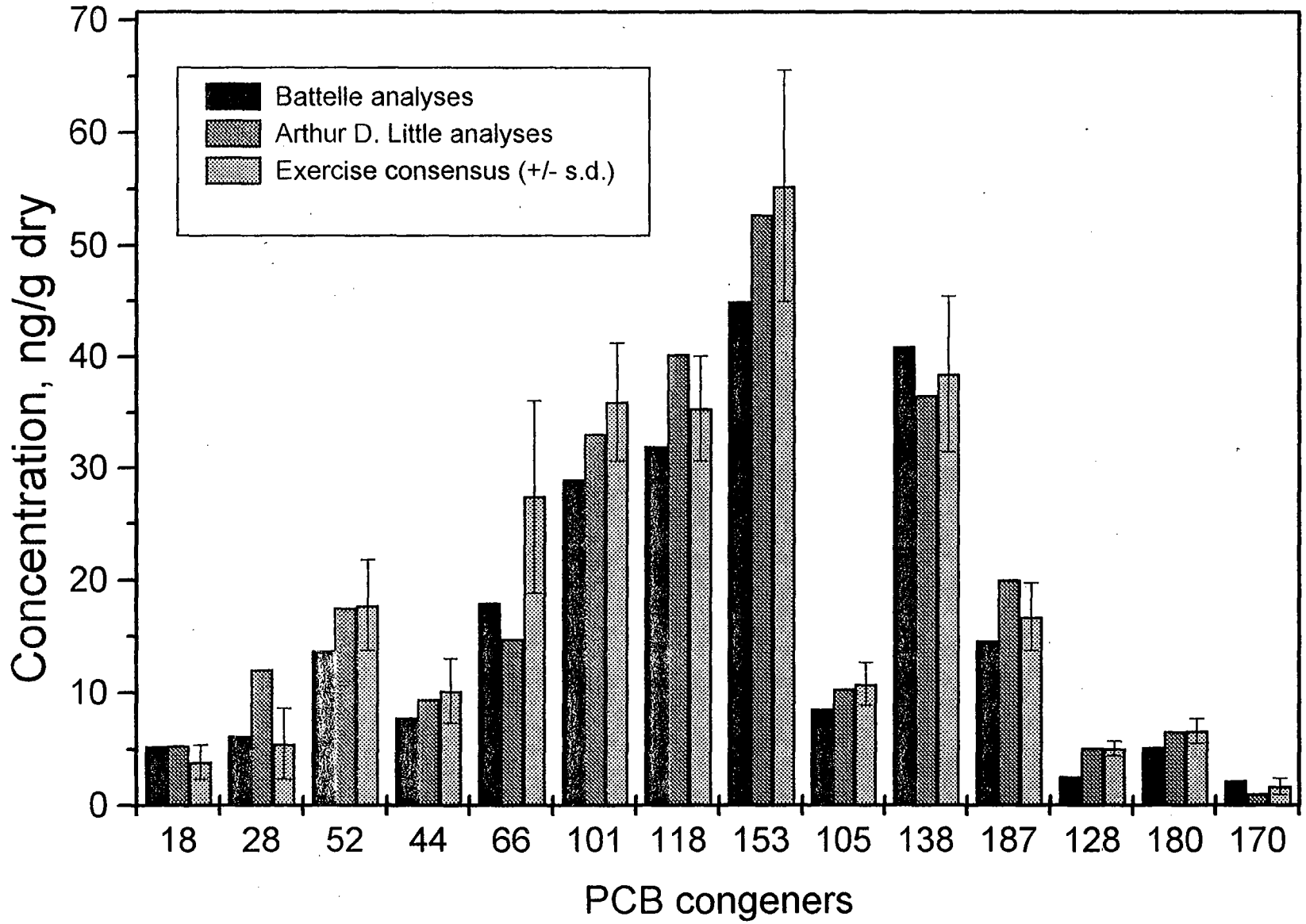
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Mussel tissue QA94TIS6
Battelle analyses
Arthur D. Little analyses
Exercise consensus (+/- s.d.)
Concentration, ng/g dry
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10
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138
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128
180
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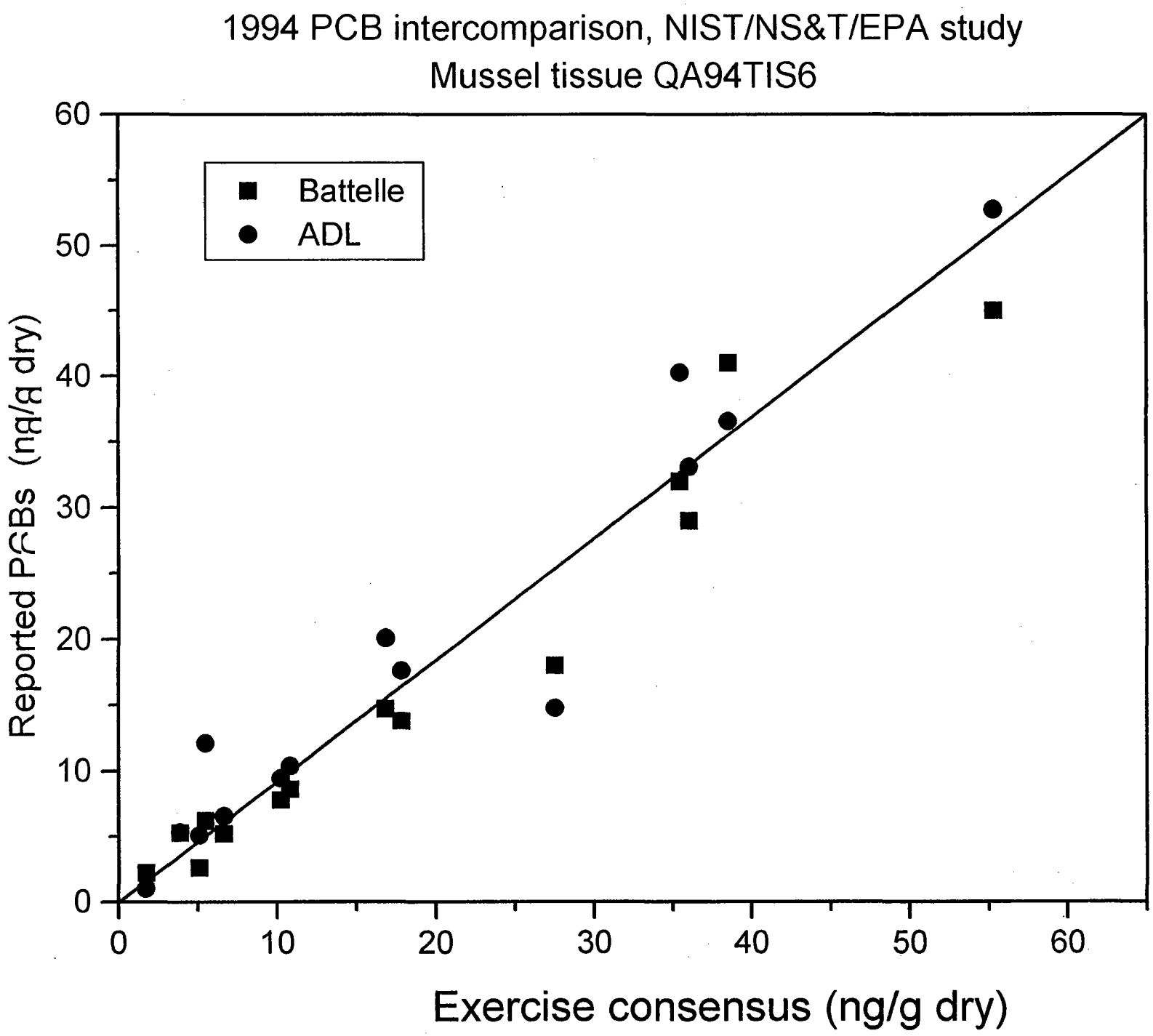
PCB congeners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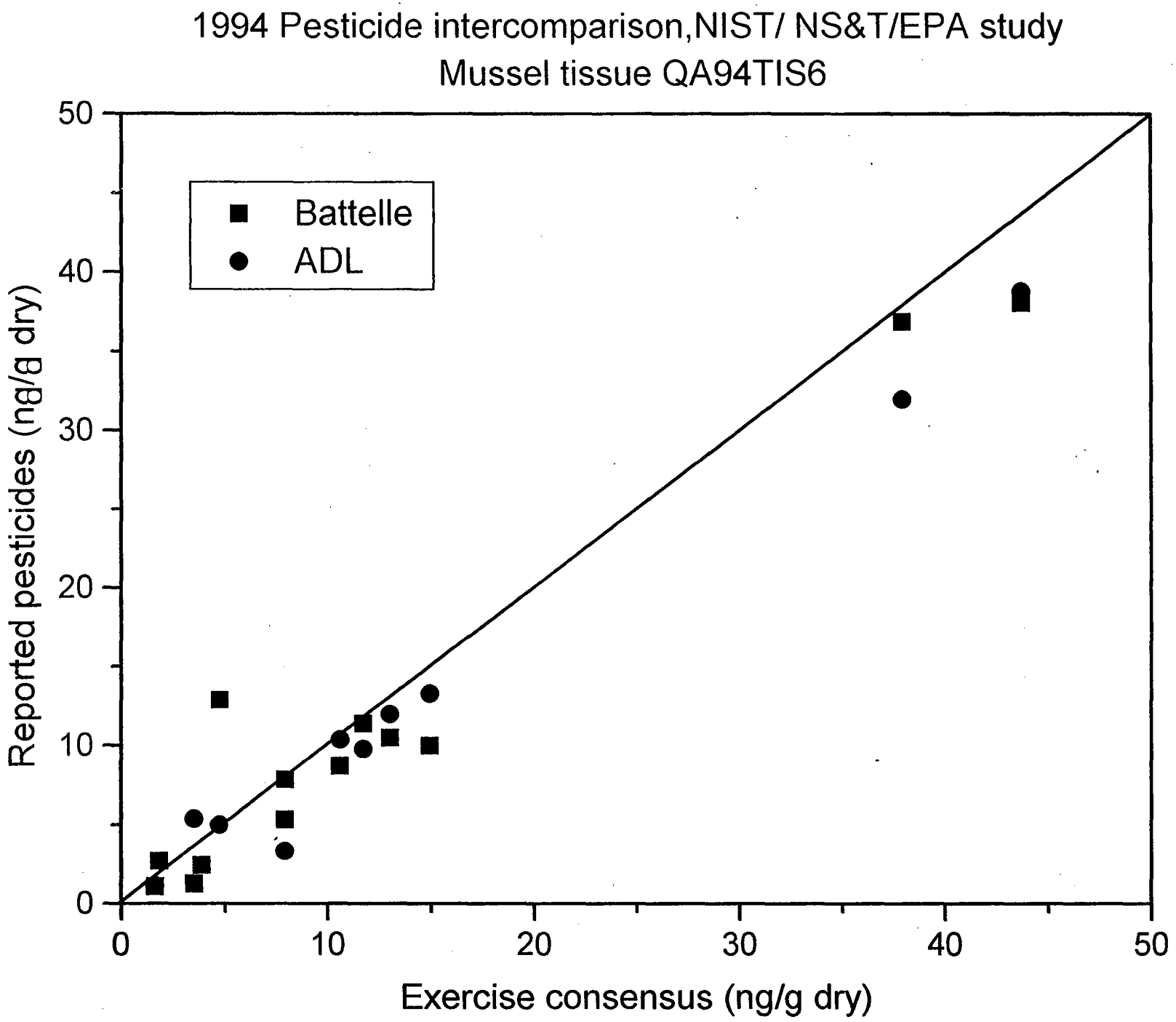
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Mussel tissue QA94TIS6
Battelle
ADL
Reported pesticides (ng/g dry)
Exercise consensus (ng/g dry)
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