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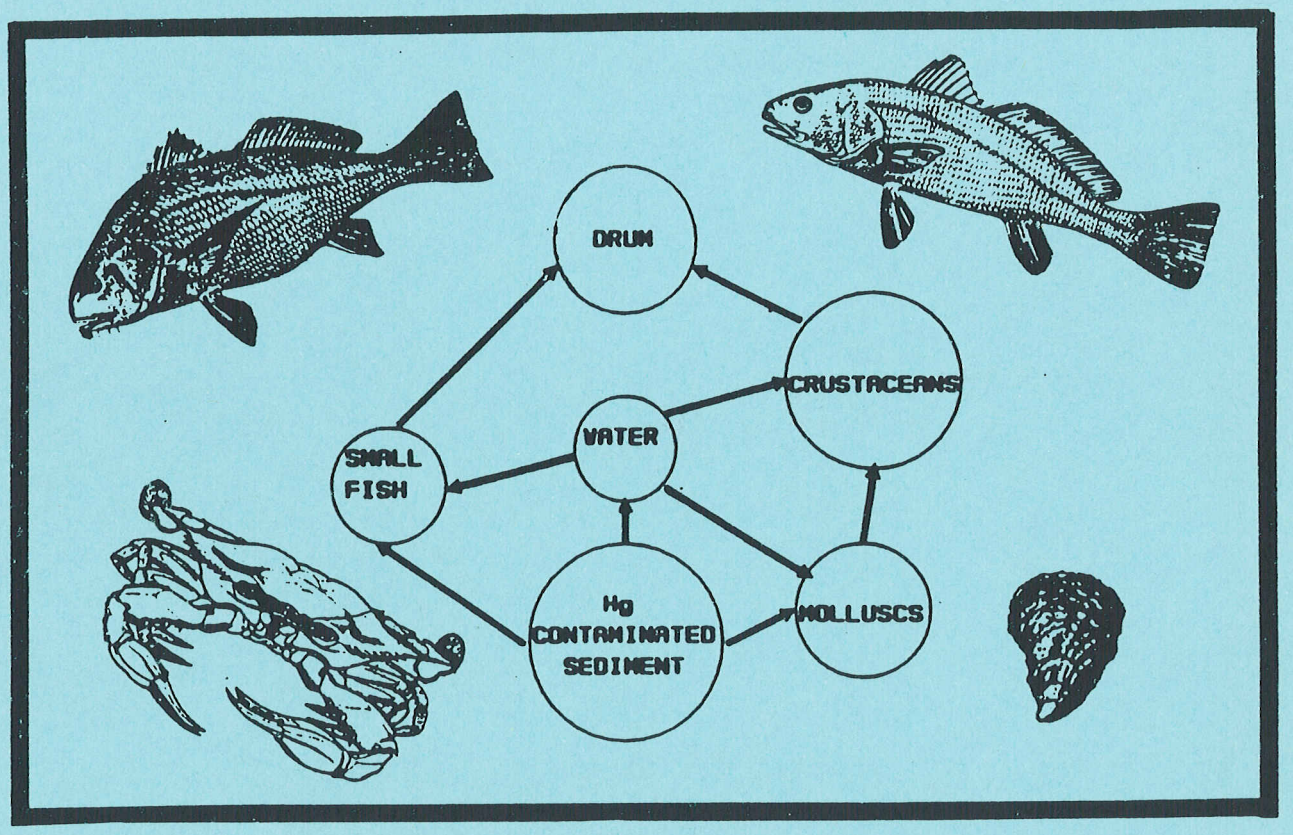
MERCURY BIOACCUMULATION IN FINFISH AND SHELLFISH FROM LAVACA BAY, TEXAS: DESCRIPTIVE MODELS AND ANNOTATED BIBLIOGRAPHY

by

DAVID W. EVANS and DAVID W. ENGEL



U.S. DEPARTMENT OF COMMERCE
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DAVID W. EVANS AND DAVID W. ENGEL

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MAY 1994

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EXECUTIVE SUMMARY

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From 1966 to 1970 a chlor-alkali plant operated by ALCOA released mercury enriched wastewater into Lavaca Bay, Texas. A reservoir of mercury enriched sediment has persisted in parts of the bay. In 1988 the Texas Department of Health closed a portion of Lavaca Bay adjacent to the chlor-alkali plant to recreational and commercial taking of finfish and crabs, because the edible flesh of many of these organisms exceeded the FDA guideline for methylmercury of 1.0 ppm wet weight.

The purpose of our investigation was to search the literature for information on the partitioning, bioconcentration, and bioaccumulation of inorganic mercury and methylmercury in aquatic ecosystems which could be used to define the connection between mercury contamination in the sediment and elevated mercury levels in Lavaca Bay biota. We have not considered other potential sources of continuing mercury contamination such as agricultural or urban runoff or industrial emissions. Additional information was sought on the life histories of fish and invertebrates found in bays and estuaries along the Texas coast that could explain this connection. Three target species, red drum, black drum and blue crab were chosen for development of a conceptual model of mercury bioaccumulation. Descriptive models of food web relationships were constructed for red drum, black drum, and blue crab which could be used to evaluate the possible pathways of mercury to these target species. In addition, models also were constructed for the major classes of primary food items: crustaceans, molluscs, and small fish. Models were conceptualized by boxes representing compartments of mercury accumulation and storage(e.g. sediments, water, fish, crabs, and molluscs) and arrows representing pathways of transfer of mercury among compartments.

Our approach in constructing a descriptive model of mercury transfer and partitioning in the food web in Lavaca Bay was to use multiple sub-models. These consisted of three models addressing the target organisms, red drum, *Sciaenops ocellatus*, black drum, *Pogonias cromis*, and blue crab, *Callinectes sapidus*, and three models concerned with the major food items: crustaceans, molluscs and small fish. Each of these latter models are for species aggregates rather than a specific crab, clam, or fish species. Each of the six models is discussed individually in a descriptive sense.

In aquatic systems the form of mercury most readily bioaccumulated and which causes greatest toxic effects is the organic form, methylmercury. Methylmercury is emphasized in our model development for these reasons. While the chemistry of

mercury in saline or estuarine waters is reasonably well understood qualitatively, actual measurements of mercury in estuarine waters are few, because the concentrations are very low and sample analysis is difficult. No data are available for dissolved methylmercury in Lavaca Bay. A water compartment is included in all of the descriptive models presented because of the potential role of water in transferring mercury from sediments to biota.

We assume that sediments are the primary reservoir and potential source of mercury in Lavaca Bay. Past surveys have shown that a plume of mercury-enriched sediments emanated from the vicinity of the ALCOA chlor-alkali plant at Point Comfort, where more recent surveys have continued to show elevated concentrations of mercury in the sediments. The pathway of methylmercury from the sediments to target species is thought to be through feeding on sediment dwelling organisms such as crustaceans, molluscs, and worms.

The two species of fish, red drum and black drum have life histories that help explain the observed differences in mercury concentrations measured in fish from different estuaries and bays on the Texas coast, specifically the elevated mercury levels found in Lavaca Bay. Both species of fish tend to feed and grow in relatively well defined locations throughout their life spans. The result of these two life history strategies is that fish which reside in a location contaminated with mercury should have body burdens that reflect the environmental concentrations of mercury.

Red drum spend their first 3 to 5 years in coastal bays and estuaries where they grow rapidly. They stay relatively close to the area to which they originally recruited, reaching a total length of 600 to 700 mm. Based on our modelling, they receive their major input of methylmercury from food items rather than directly from either the sediment or the water. Their primary foods are crustaceans(crabs and shrimp). Small fish are also a food source of the red drum , but are probably not as important in the cycling of mercury, since the methylmercury concentration in small prey fish is low relative to crustaceans.

Black drum have not been studied as extensively as red drum, but have a similar growth pattern, and migrate even less. They recruit into an estuary where they tend to remain throughout their life. When the fish become sexually mature they spawn in the estuary in the vicinity of inlets or passes and then return to the same embayment. Lack of movement by the black drum reduces the uncertainty about where they acquire their food. This information is critical in limiting the uncertainty of their exposure to methylmercury in Lavaca Bay. The food chain supporting the black drum is simpler than the one that supports the red drum. The two main source terms for mercury are molluscs and crustaceans. The black drum is primarily a mollusc predator, and is well equipped anatomically to grind and crush mollusc shells.

The blue crab has a well defined life history, but a less specific habitat selection process than the two species of fish. It is more mobile than the fish. Blue crabs(along with stone crabs) are also among those species exceeding the FDA guideline for methylmercury of 1.0 ppm wet weight in parts of Lavaca Bay.

Blue crabs are opportunistic feeders with a diet that includes everything from dead red drum to juvenile oysters. They have been classified as detritivores, omnivores, and carnivores. Detritus, animal/plant material, sediment, and molluscs, are likely more important mercury sources than direct accumulation from water.

The molt cycle affects some metal concentrations in blue crab hemolymph and digestive gland but not in the muscle. It is not known whether methylmercury in the edible tissues of the crabs is lost during molting.

Modelling mercury accumulation in blue crabs is the most complicated of the three target species (numerous pathways exist). Causes for the complications are as follows: feeding patterns are not well defined; growth is discontinuous and life span is short; and growth can directly affect the turnover and metabolism of accumulated mercury.

Major components of the food webs for red drum, black drum and blue crabs consists of assemblages of organisms (crustaceans, molluscs, and small fish). The descriptive models for these groups are more generic than the single species food webs.

Primary food sources for crustaceans are animal and plant detritus, epiphytes and benthic algae, and benthic invertebrates which include both micro- and macro-fauna. The choice of particular organisms is dictated by their relative abundance and the developmental stage and size of the crab or shrimp. All species with the exception of xanthid crabs are found primarily in seagrass meadows and salt marshes where detritus is abundant. All tend to be omnivores and detritivores.

Food web relationships for the molluscs are simpler than for crustaceans. Filter feeding molluscs can accumulate metals and nutrients both through the consumption of phytoplankton and directly from the water. Sediment may be a dietary source of mercury for deposit feeding molluscs and an indirect one for filter feeders.

Small fish are secondary to other more important food web pathways. Of all the descriptive models that have been constructed, the small fish model has the highest level of uncertainty. The uncertainties are derived from the large number of species and trophic diversities. There is a notable lack of information on their mercury concentrations in Lavaca Bay.

Red drum was used in an example of model implementation. Values assigned to the boxes and arrows of the model were taken from the world-wide mercury literature and compared to measured values from Lavaca Bay when available. Calculations were made of the fluxes of mercury through the food web to the red drum. Results of this implementation exercise with the red drum model were realistic and consistent with observed methylmercury concentrations in Lavaca Bay red drum. The results demonstrated the utility of the model and also demonstrated vividly the large areas of uncertainty in model parameter estimates and gaps in the mercury databases for Lavaca Bay biota and geochemistry.

In summary, the conceptual modelling effort identified five critical factors that have led to the localized elevations of mercury concentrations in certain finfish and crustaceans in Lavaca Bay

1. The persistence of high mercury concentrations in sediments in the Point Comfort area of Lavaca Bay.

2. Low organic matter concentrations in Lavaca Bay and Matagorda Bay sediments enhance mercury bioavailability to sediment inhabiting invertebrates.

3. A simple food web in which red drum, black drum, and blue crabs feed predominantly on these sediment inhabiting invertebrates.

4. Limited movement of red drum, black drum, and blue crabs within Lavaca Bay which allows them to be exposed to the elevated mercury concentrations in these invertebrates throughout most of their early lives.

5. Highly efficient and rapid assimilation of methylmercury from food by the target species. In combination with very slow excretion of methylmercury, this allows the target species to accumulate methylmercury to high levels during their residence in Lavaca Bay.

I. INTRODUCTION

Until about 1970, mercury (Hg) was widely used as the working electrode in the production of chlorine and sodium hydroxide from sodium chloride in the chlor-alkali process. About one quarter of the mercury used in the United States in 1970 went to this process (D'Itri, 1972). Substantial amounts of the mercury used in the process escaped recycling and reuse efforts and entered waste streams where it could enter aquatic, terrestrial, and atmospheric environments. In 1970, alerted by serious public health problems with mercury in Japan and Sweden, intense public concern developed around mercury pollution in the United States and Canada. Elevated levels of mercury in the environment of Lake St. Clair were traced largely to two chlor-alkali plants in Ottawa operated by the Dow Chemical Company. Of particular concern were high levels of mercury in fish and their subsequent consumption by humans. Chlor-alkali plants throughout the U.S. and Canada were quickly identified as major sources of mercury contaminating rivers, lakes and estuaries. Among the mercury contaminated water sources identified by the Federal Water Quality Administration and the Food and Drug Administration was Lavaca Bay, Texas (D'Itri, 1972)

From 1966 to 1970 a chlor-alkali plant operated by ALCOA released mercury enriched wastewater into Lavaca Bay near Point Comfort, Texas (Figure 1). Discharges of mercury to the bay and to an offshore spoil island lagoon were estimated to reach as much as 67 lbs Hg/day (Trebatoski and Gooris, 1990). Monitoring of mercury concentrations in seafood by EPA and Texas Department of Health (TDH) in 1970 found elevated mercury concentrations in oysters and crabs in Lavaca Bay. Consequently, parts of the bay were closed to oystering by TDH later that year. The closure was lifted in 1971 when mercury concentrations dropped below the Food and Drug Administration's action level of 0.5 ppm Hg wet weight in response to ALCOA's diversion of wastewater to onshore disposal sites the previous year. TDH issued public health warnings concerning mercury in fish and shellfish in the area, but lacked authority to close these fisheries.

Periodic monitoring by TDH, continued to find elevated mercury concentrations in some fish and shellfish in Lavaca Bay in the vicinity of ALCOA's facility even after the chlor-alkali plant was dismantled in 1979. Finding mercury concentrations in excess of the new FDA limit of 1 ppm Hg wet weight (as methylmercury) in some finfish and blue crabs, TDH closed portions of Lavaca Bay to their recreational and commercial harvest in 1988.

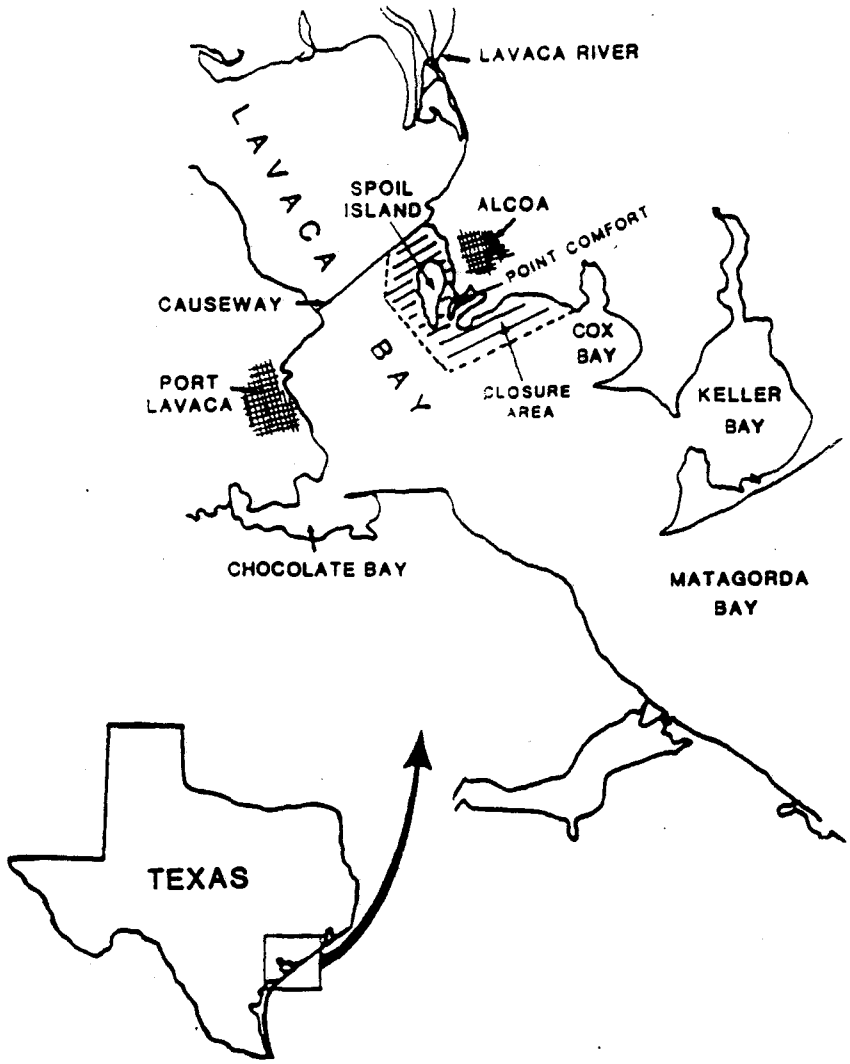
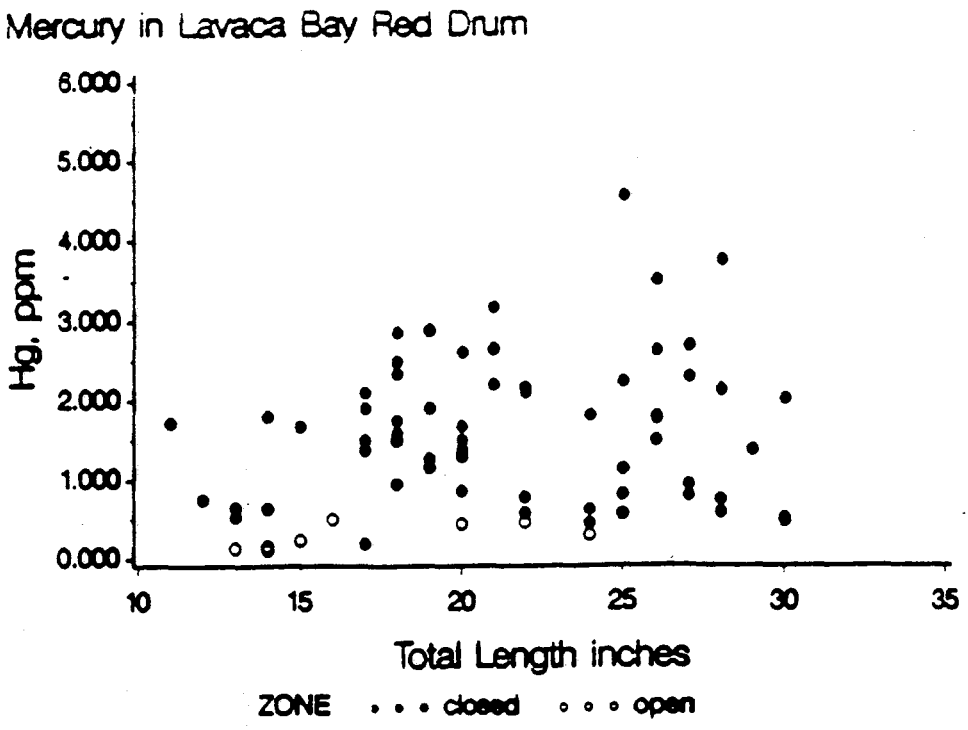


Figure 1. Generalized map of the Matagorda/Lavaca Bay complex and its relationship to the rest of the Texas, Gulf of Mexico coast. The locations of important sites are shown on the map. The portion of Lavaca Bay closed to recreational and commercial fin and shellfishing due to elevated concentrations of mercury in the biota is highlighted.

Recognizing the possible damage to natural resources in Lavaca Bay, a group of natural resource trustees (representing Texas Department of Health, Texas Water Commission, Texas General Land Office, Texas Parks and Wildlife Department, U.S. Fish and Wildlife Service, and the National Oceanic and Atmospheric Administration) initiated damage assessment strategy development on mercury in Lavaca Bay in 1990. A memorandum of agreement was reached with ALCOA to develop and fund studies to determine the present distribution and partitioning of mercury in the Lavaca Bay environment. It sought to provide basic information concerning the significance of mercury contamination needed to support restoration planning in Lavaca Bay. One product of this effort was an initial survey of mercury concentrations in sediments in Lavaca Bay. This would be used in planning more detailed evaluation of the extent of mercury contamination. A second product was a report, based on existing information, describing the sources, food chain pathways and rates through which mercury in the sediment reaches upper level carnivores [including humans] in Lavaca Bay. This report to the Technical Management Team of the Lavaca Bay Study was submitted in final form December 15, 1992 under the title, "Mercury Bioaccumulation Study: Stage 1. Descriptive Model and Literature Search". It is the basis, with modifications, of this technical memorandum.

This technical memorandum contains two major sections: 1. A narrative report that develops a food web model of mercury accumulation in selected fisheries organisms in Lavaca Bay and 2. an annotated bibliography of literature references used in developing the model.

Two species of fish, red drum (Sciaenops ocellatus) and black drum (Pogonias cromis) and the blue crab (Callinectes sapidus) were selected for specific modelling effort because they are important intermediate and top predators in the Lavaca Bay food web, they are important commercial and recreational fisheries resources, and they are observed to have among the highest mercury concentrations of any fish and shellfish in the closure area of Lavaca Bay (Table A.). During the period 1981 to 1992, mercury concentrations measured in black drum in the closed area of Lavaca bay averaged 1.45 ug Hg/g wet weight. This can be compared to an average of 0.19 ug Hg/g in the open area of Lavaca Bay during the same period and an average of 0.20 ug Hg/g in other Texas estuaries measured in 1976-1977 (TDH, 1988). For red drum, comparable averages were 1.60 ug Hg/g (closed area of Lavaca Bay), 0.28 ug Hg/g (open area of Lavaca Bay), and 0.20 ug Hg/g (other Texas estuaries). For blue crabs, the comparable numbers were 0.73, 0.24, and 0.16 ug Hg/g wet weight. Several other fish species (sheepshead and flounder) showed elevated total mercury concentrations in the closure zone of Lavaca Bay. This suggests that during the past decade, elevated mercury concentrations in these species have been largely restricted to the closure area in Lavaca Bay. Individual fish outside of the closure area may have elevated mercury concentrations (Fig. 2A and B).



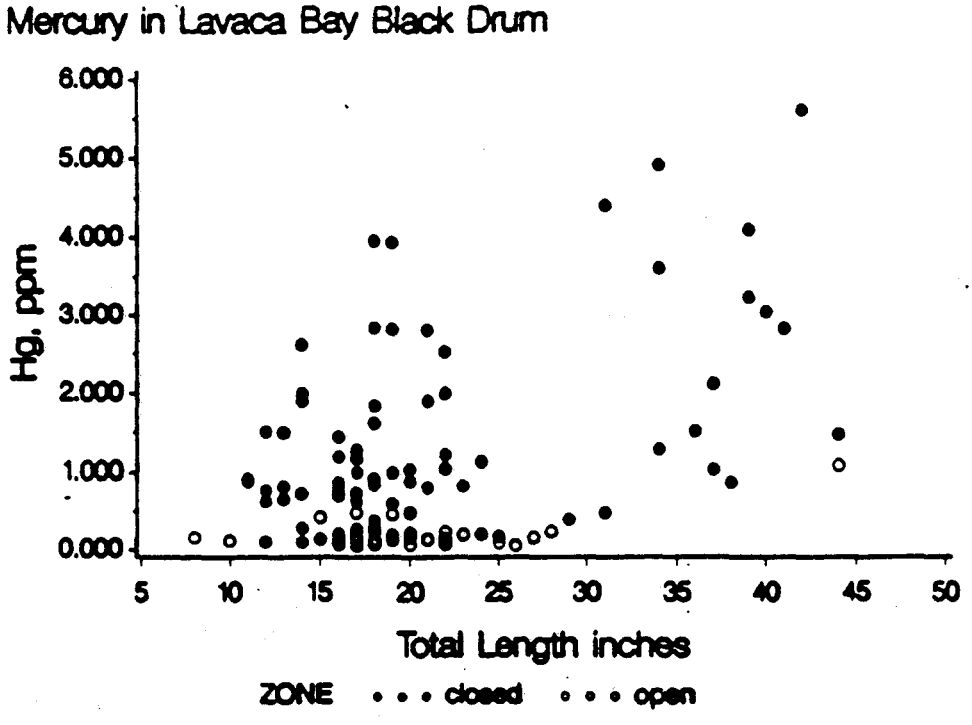


Figure 2. Relationship between the total lengths of black drum (A.) and red drum (B.) and the mercury concentration in the muscle among fish captured within and outside of the "closed area" of Lavaca Bay. Each point represents the mercury concentration in an individual fish. Data from Texas Department of Health surveys [1981-1991] (TDH, 1988)

Table I. TOTAL MERCURY CONCENTRATIONS IN FISH AND INVERTEBRATES FROM LAVACA BAY, 1981-1991 (Texas Department of Health, 1988)

Species	Mean Hg Concentration (ug Hg/g wet weight)		
	Lavaca Bay		Other Texas Estuaries
	Closed Area	Open Area	(1976-1977)
Black Drum	1.45	0.19	0.20
Red Drum	1.60	0.28	0.20
Speckled Trout	0.56	0.38	0.42
Sheepshead	1.36	0.11	0.08
Flounder	0.74	0.17	0.10
Atlantic croaker	----	0.07	0.05
Shad	0.21	----	----
Gafftopsail catfish	0.74	----	0.70
Tripletail catfish	0.47	----	----
Blue crab	0.73	0.24	0.16
Stone crab	1.99	----	----
Oyster	0.08	----	----

which may be the result of emigration from the closure area. On average, however, mercury concentrations of fish outside the closure area are little different from those measured in other Texas estuaries. Some fish species captured within the closure area (e.g. speckled trout and gafftopsail catfish) showed mercury concentrations only marginally elevated relative to those captured in other Texas estuaries. Different feeding patterns or habitat preferences may be the explanation.

Model development required integrating information on the biology, ecology, and physiology of important estuarine organisms, the biogeochemistry of mercury, and the mathematical and data requirements of numerical models. Information was sought in the following areas which could explain the observed high concentrations of mercury observed in Lavaca Bay fish and shellfish.

1. Identification of the feeding habits of the three target species. This includes quantitative estimates of the food items consumed, and their variations in response to season, life history and age, and food availability.

2. Resident histories of the three species in the Lavaca Bay area that would influence their exposure to elevated mercury concentrations. Short term, seasonal, and age dependent (spawning) migration are relevant.

3. Existing information on the mercury concentrations in the target species and their food items. This includes the chemical form of mercury (inorganic or methylmercury), the relationship to mercury concentrations in water and sediments, the spatial distribution of mercury in the Lavaca Bay and Matagorda Bay systems. Comparable information from other contaminated and non-contaminated areas was also sought in order to place the mercury problem in Lavaca Bay in context.

4. Models of mercury accumulation in aquatic food webs, especially those linked to the benthos because of the presumed sediment source for mercury in the Lavaca Bay system. Models of accumulation of other chemical contaminants were also considered since such models might have relevant structures representing similar transfer processes. This would allow their application to mercury bioaccumulation, albeit with different parameterization.

5. Information on the parameter values for such models: assimilation efficiencies for mercury in different chemical forms from different foods, from water, and from sediments; excretion kinetics of mercury from fish and invertebrates, especially the target species and their prey; growth rates and bioenergetics of the same or similar species which are important inputs in many of the more complex models. Bioconcentration factors of mercury under steady state conditions were also sought.

FRAMEWORK FOR CONCEPTUAL MODEL

Models are simplified abstract representations of the real world, which serve specific purposes. In our case, we would like a model of bioaccumulation of mercury in the Lavaca Bay food web to serve three purposes: 1. Describe the pathways of mercury through the food chain from mercury contaminated sediments to the target species (red drum, black drum, and blue crab); 2. Predict mercury concentrations in the target species based on measured concentrations in contaminated sediments and the physical and biological characteristics of the Lavaca Bay system; 3. Allow probabilistic inference that contaminated sediment is the source of elevated mercury concentrations in the target species.

We have chosen box and arrow models to represent parts of the food web in Lavaca Bay. Boxes represent reservoirs of mercury accumulation in the system (e.g. sediment, water, benthic infauna,

benthic epifauna, pelagic fish, target species such as red drum). Boxes are connected by arrows pointing the direction of mercury transfer between the reservoirs, (i.e. between sources and receptors of mercury).

To introduce the modelling strategy, we are presenting a series of box and arrow models that are progressively more complex and hopefully more realistic. Boxes representing additional components of the food chain will be added as needed. Eventually it will be possible to assign quantitative values to the arrows reflecting the transfers between boxes, with full knowledge that large uncertainties can be associated with those values. A trial quantitative implementation is attempted for red drum as part of our conceptual modelling effort.

POSSIBLE FORMATS

The simplest model would include only one box, the target species, and one arrow, the inputs of mercury added to the Lavaca Bay system (Fig. 3). Such an overly simplified model was used in the United Kingdom to develop a critical pathways analysis for mercury pollution control in contaminated estuaries (Preston and Portmann, 1981). The relationship between mercury inputs and observed mercury concentrations in fish must be calibrated by repeated measurements of inputs and concentrations in fish over time or over space. Predictions of mercury concentrations in fish consequent to alteration in inputs is then possible by application of the calibration relationship. This is unlikely to be very useful in Lavaca Bay, if as we assume, major inputs of anthropogenic mercury ceased more than twenty years ago, and where we assume current elevated mercury concentrations in fish are probably due to residual mercury in bed sediments.

A slightly more complex model might include two boxes and a single arrow (Fig. 4). The boxes would represent sediments and a target fish species. The arrow would represent the relationship between their respective mercury concentrations. In a typical steady state situation, this relationship would represent a fish/sediment concentration factor (BIOTA SEDIMENT FACTOR or BSF), a simple ratio of concentrations. The BSF should not be confused with "bioconcentration factor" or BCF, which is generally applied to the accumulation of a contaminant from water, and is the organism to water ratio of concentrations. This approach has been applied to organic contaminants in closed systems such as lakes. It makes no assumptions about the details of the mercury transfer pathway (i.e. What are the intervening food chain steps?) but does assume that the concentration factor is constant over time and space.

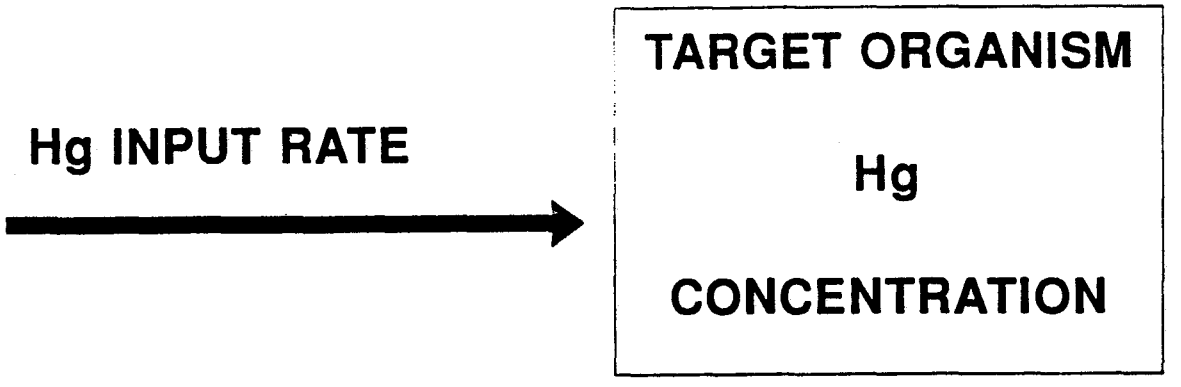
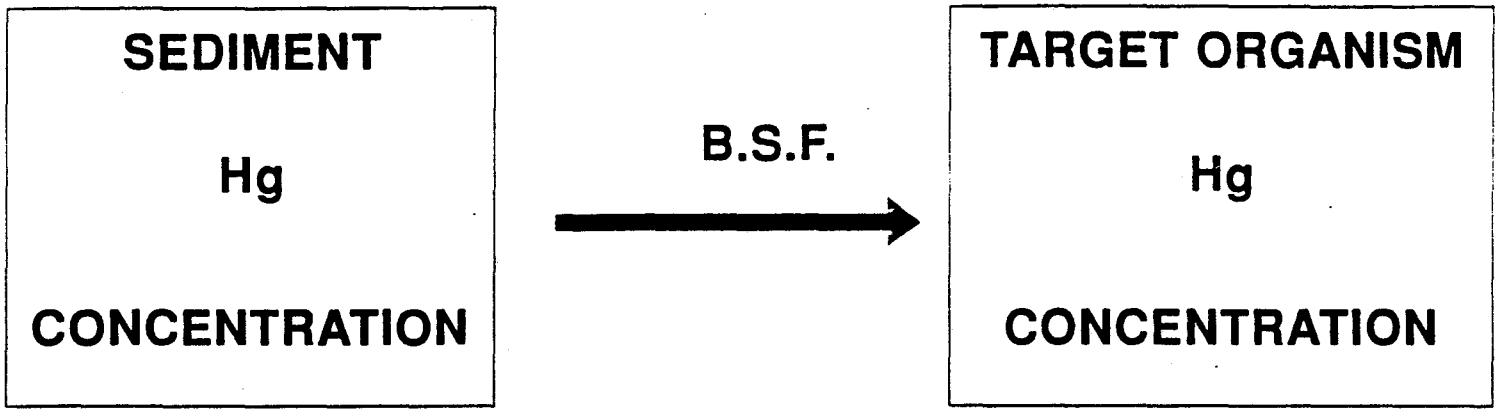


FIGURE 3. Simple one step transfer model with a single input source of mercury to an estuarine ecosystem and a single receptor organism. The arrow denotes the transfer from source to receptor (e.g. fish) compartment.



B.S.F. = BIOTA SEDIMENT FACTOR

B.S.F. = [BIOTA] / [SEDIMENT]

FIGURE 4. A two compartment transfer model showing the transfer of mercury from a sediment compartment to the receptor organism. The BSF [Biota Sediment Factor] is a simple ratio of the concentration of mercury in the sediment and the organism (e.g. fish).

To make the model more descriptive, a third box might be added, representing water, through which mercury could be transferred between sediments and fish (Fig. 5). The two transfer arrows could then represent the steady state ratio of mercury in water to that in sediments (i.e. a partition coefficient) and the water to fish bioconcentration factor.

Additional boxes could be added to link water or sediment to boxes representing biota on which the target fish species feeds. The pathways could branch as multiple prey species or intermediate predators are introduced. Such models are shown in Figs. 6 - 11. The arrows of the models indicate the transfer of Hg between the different components of the food webs. All models of this sort are time invariant (steady state); they suppress the information that mercury transfers between boxes are not instantaneous, but have characteristic time constants. Moreover, fish and organisms in their food chain can migrate over time between areas of different mercury concentrations within the primary sediment source. Diet shifts also can occur over time, changing the mercury exposure. Seasonal cycles in partition coefficients and bioconcentration factors can exist. Fish growth over time can be associated with sexual maturity, spawning, migration, and altered metabolic rates, any of which can change both the boxes and the quantification of the transfers associated with the arrows.

Including the element of time adds both realism and complexity to the model. Significant effort has been made to link organism bioenergetics (metabolism and growth) to the kinetics of mercury accumulation and excretion in the organism (e.g. Norstrom et al. 1976; Rodgers and Beamish, 1981). A model for Kepone accumulation in striped bass nicely adds the features of temporally changing source terms, multispecies food chains, diet shifts with age, altered contaminant exposure during estuarine migration and altered metabolism with growth (Connolly and Tonelli, 1985). Such a model is beyond our immediate means to develop. However, a simplified model suggested by the work of Fordham and Reagan (1991) seems feasible. It adds to the steady state model the features of kinetics of mercury accumulation and excretion to the already existing multiple pathways of accumulation by the target species. In reality, it is still a steady state model because the time dependence introduced by the feeding rate and mercury excretion rate cancel upon integration. The advantage is that explicit estimates of these rate constants are available from the literature or from field data (adding greater realism to the model) whereas bioconcentration factors may be unavailable. Such a steady state version can in theory be enhanced to a time varying mode for greater realism and detail in predictive capability if this is warranted.

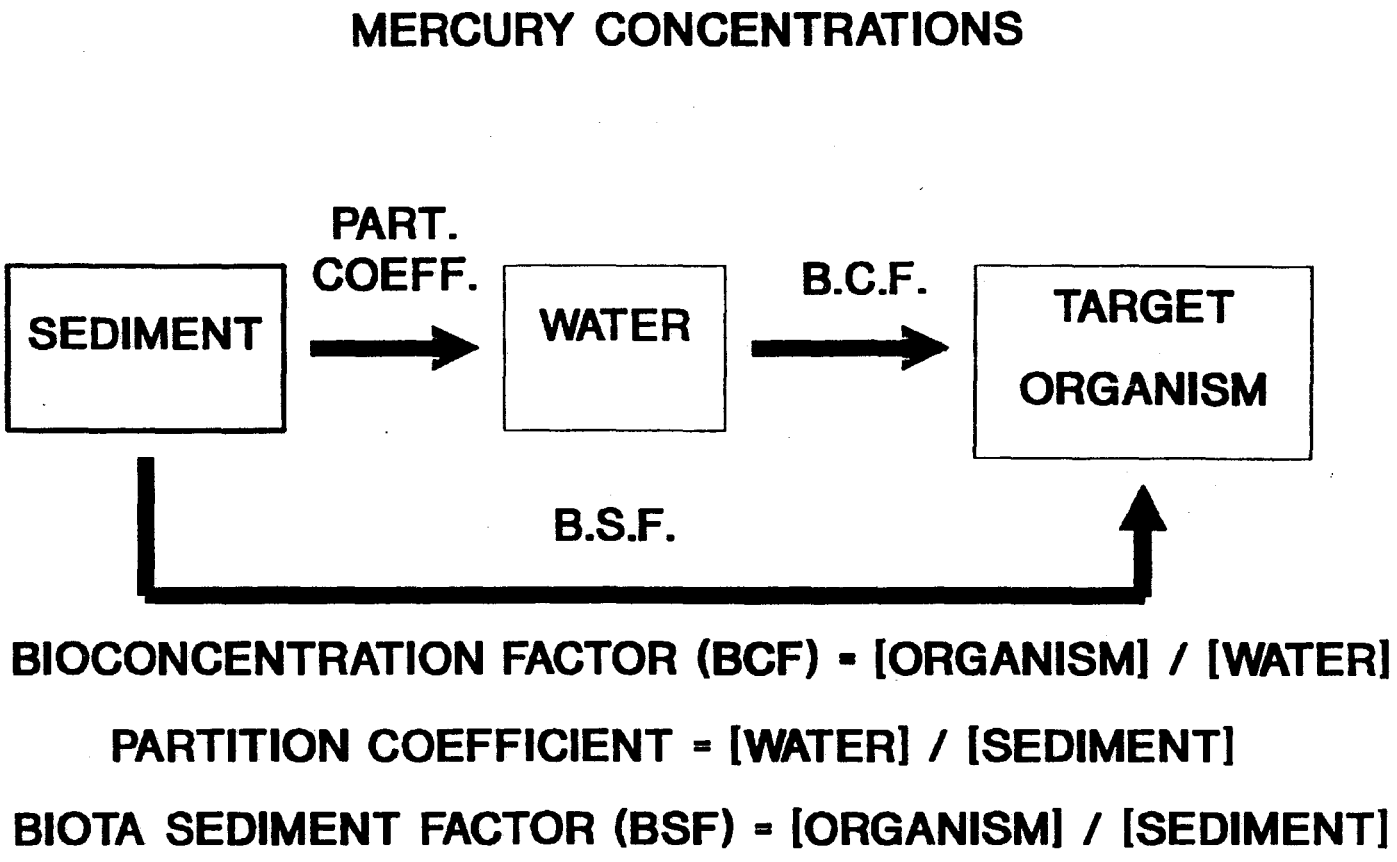


FIGURE 5. A three step transfer model showing the movement of mercury from the sediment compartment through the water to the receptor organism. There are also three transfer pathways (1.) from sediment to water [Partition Coefficient], (2.) from water to fish [Bioconcentration Factor (BCF)], and (3.) from sediment to fish [Biota Sediment Factor (BSF)]

II. DESCRIPTIVE MODELS

Our approach in constructing a descriptive model of mercury transfer and partitioning in different components of the food web in Lavaca Bay was to use multiple models. They consist of three models addressing the target organisms, red drum, Sciaenops ocellatus, black drum, Pogonias cromis, and blue crab, Callinectes sapidus, and then three models concerned with the major food items, crustaceans, molluscs and small fish. Each of these latter models are for multiple species rather than a single species of crab, clam, or fish. Each of the six models will be discussed individually in a descriptive sense, then, using red drum as an example, values will be assigned to the boxes and arrows (e.g. the numbers will come from the world-wide mercury literature) and calculations made of the fluxes of mercury through the food web to the red drum.

In considering the transfers of mercury through the food web, it is first necessary to examine the inputs from water and sediment. We assume for this modelling effort that currently the only source of mercury is that which is bound in the sediment and cycles in and out of the water column. For the most part this cycling is governed by geochemical processes, but bioturbation may be contributing to the turnover.

Water is potentially an important source term which should be included in any type of descriptive model of mercury partitioning in the estuarine environment. The speciation of mercury in saline waters will control its availability to organisms in the water column. In saline waters inorganic ionic Hg^{+2} is complexed actively by chloride, which limits its bioavailability, bioaccumulation, and toxic responses. In our laboratory we have shown that it is the free ionic forms of other trace metals, Cd, Cu, Zn, and Ag, that are toxic and accumulated rather than the complexed or ligated forms of the metals. In aquatic systems the form of mercury that is most readily accumulated and causes toxic effects is the organic form, methylmercury. This organic form of mercury is hydrophobic in its chloro-complexed form. This means that its transit across cell membranes is enhanced. Inorganic mercury can be transformed to methylmercury, probably by bacteria in sediments or in the water column (Compeau and Bartha, 1985). Released into the interstitial water, it is available to sediment dwelling organisms. Methylmercury that is not accumulated or demethylated to inorganic mercury can move into overlying waters where it can be accumulated by organisms or dispersed through the estuary. While the chemistry of mercury in estuarine waters is qualitatively known to some extent, quantitative measurements of methylmercury concentrations and rates and locations of production are little known. Measurement of mercury concentrations in natural waters is

extremely difficult, because concentrations are very low, and sample contamination is a major problem. Though few data are available for dissolved mercury in Lavaca Bay, a water term is included in all of the descriptive models presented.

Sediments are assumed to be the primary source term for mercury in Lavaca Bay. Surveys conducted since 1970 by USGS (Holmes, 1977), the Texas Water Commission (Bowman, 1988), Texas Department of Health (TDH, 1988), and Texas A&M University (Riegel, 1990) have shown that elevated concentrations of mercury are found in sediments in the vicinity of the ALCOA chlor-alkali plant at Point Comfort. A sediment survey being conducted by USGS will shed further light on the current depth distribution and variation of mercury in Lavaca Bay. Since the form of mercury most probably accumulated by the target species is methylmercury, it is important to learn the relative abundance of methylmercury vs. total mercury in the sediment. The relative partitioning of these forms among abiotic sediment particles, non-living organic detritus, and living microbes (bacteria, fungi, and algae) in the sediments will influence mercury accumulation by the smaller macroinvertebrates such as polychaete worms, small crustaceans, and molluscs through which methylmercury is transferred to the larger invertebrate foods of red drum, black drum and blue crabs.

The three target species that will be considered in the development of the food web model are: the red drum, Sciaenops ocellatus, the black drum, Pogonias cromis, and the blue crab, Callinectes sapidus. These species are of importance in Lavaca Bay, because some individuals have been shown to exceed the U.S. Food and Drug Administration action level of 1.0 ppm wet weight methylmercury (FDA, Compliance Guides [Guide-708.07] Chapter 8. Fish and Seafood), and therefore, present a potential health hazard to man through recreational and commercial fishing (TDH, 1988). The purpose of the model is to provide a descriptive framework that can be used in policy making discussions concerning the future uses of Lavaca Bay and in assessing the possible impacts of the chronic mercury contamination on the fishery resources of the Bay.

The two species of fish, red drum, Sciaenops ocellatus, and black drum, Pogonias cromis, have life histories that partially explain the observed differences in mercury concentration measured in fish from different estuaries and bays on the Texas coast (Simmons and Breuer, 1962; Mercer, 1984; Reagan, 1985; Sutter et al., 1986; Monaco et al. 1989). Red drum spend their first 3 to 5 years in coastal bays and estuaries. During this period of rapid growth they stay relatively close to the area to which they originally recruited, and reach a total length of 600 to 700 mm (Miles, 1950; Simmons and Breuer, 1962; Perret, et al. 1980; Matlock, 1992). The fishes' lack of movement was demonstrated in a tagging and recapture study that showed during the three year growth period the fish remained within a few kilometers of where they were tagged (Osburn, et al., 1982). After the fish reach a total length of greater than 600 to 700 mm in 3 - 5 years they

reach sexually mature and move out of the estuary into the Gulf of Mexico where they join schools of adult drum that move up and down the coast (Simmons and Breuer, 1962; Wilder, 1986). Spawning also occurs in the offshore waters. Black drum have not been studied as extensively, but follow the same general growth strategies, and tend to move even less (Green, 1986). The black drum recruits into an estuary where it tends to remain throughout its life (Sutter, et al. 1986). When the fish become sexually mature they spawn in the estuary in the vicinity of inlets or passes and then return to the same embayment (Simmons and Breuer, 1962; Murphy and Taylor, 1989). Thus, both species of fish tend to feed and grow in relatively well defined locations throughout their juvenile life spans. The result of these two life history strategies is that fish which reside in a location contaminated with mercury should have body burdens that reflect the local exposure concentrations of mercury.

The blue crab has a well defined life history, but a much less specific habitat selection process than the two species of fish, and also tends to be more mobile than the fish (Williams, 1965; Millikin and Williams, 1984; van Den Avyle and Fowler, 1984). Females shed their larvae in the coastal waters of the Gulf of Mexico, and the early life stages, zoeal stages, are part of the pelagic planktonic community. After the last zoeal stage, they transform into megalopa and enter the estuary. During the megalop stage they move up the estuary toward lower salinity water where they transform into the first crab stage. While they are small they grow, molt rapidly and by the end of the first year can attain a carapace width of 80-100 mm. In the spring of the second year the immature females undergo their maturity terminal molt and mate. After mating adult females migrate toward higher salinity water where the eggs are produced. Males and immature females are excellent osmoregulators (Engel, 1977) and tend to stay in the more brackish portions of the estuary throughout their lives. They will move extensively within the estuary depending on food supply.

To explain the relationships that exist between the different components of the food web, we have constructed a series of descriptive models that describe some of the possible pathways of Hg transfer. The relationships described in these models are derived from the literature and from conversations with colleagues active in the field of estuarine ecology. It must be stressed from the outset that the degree of uncertainty concerning absolute pathways of mercury transfer and the rates of mercury transfer or feeding in this type of modelling effort is relatively large. We have constructed the relationships depicted in these models through an iterative process and therefore the degree of uncertainty concerning food web relationships is reduced.

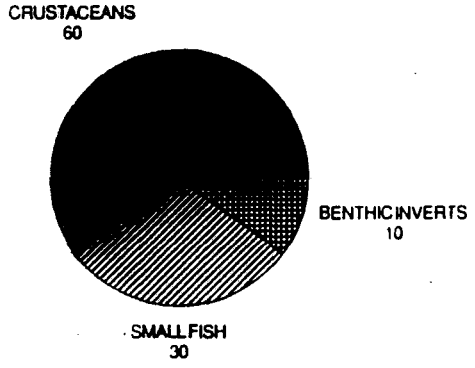
RED DRUM MODEL

The descriptive model of mercury accumulation pathways to red drum concentrates on the food web utilized by pre-adult fish which are less than 3 to 5 years old and less than 600 mm standard length. This age and size class is important because it roughly corresponds to the legal size range of recreationally caught fish (20 to 28 inches or 500 to 700 mm) in Texas. In sampling programs of the Texas Department of Health, measurements of mercury in red drum from the Point Comfort area of Lavaca Bay have frequently found fish of this size with methylmercury in the edible flesh in excess of the FDA guideline of 1.0 ppm wet weight. It is also important to recognize that pre-adult red drum do not move far from the area to which they recruited as early juveniles (Osburn, et al., 1982; Green, 1986). The lack of movement of the juvenile red drum is typical of the species, whether the fish are from Texas or North Carolina (Mercer, 1984). This lack of movement reduces the uncertainties as to the origin of food the fishes consume. This information is critical for the determination of the sources of methylmercury measured in the fish. Wilder (1986) also has shown that red drum along the Texas coast can be considered one genetically homogeneous population. While the juveniles may be geographically isolated, adults occupy the open water of the Gulf where they form a single reproductively active stock. These data mean that the high mercury concentrations among fish from Lavaca Bay cannot be attributed to a genetically unique stock.

The descriptive model for red drum is shown in Figure 6. As stated earlier, the arrows presented in this model are derived from the literature, and for the purpose of the descriptive model have not yet been assigned quantifiable amounts of mercury.

The pie diagram in Figure 6 depicts the relative percentages of food items in the diet of red drum comprised of crustaceans (blue crab and shrimps), small fish, and other invertebrates (Miles, 1949; Miles, 1950; Darnell, 1958; Bass and Avault, 1975; Overstreet and Heard, 1978; Peters and McMichael, 1987). These percentages (i.e. crustaceans, 60%; small fish, 30%; and other invertebrates, 10%) are estimates for the pre-adult life history of the fish. The uncertainties that underlie these estimates involve the size and age of the fish, the availability of food items, and diel cycles. It has been shown that as red drum grow their food preferences change, and that these changes are dictated to some extent by the size of the fish (Bass and Avault, 1975). For example, Miles (1949 and 1950) described the food habits of red drum and indicated that larger fish, greater than 350 mm, preferred shrimp and crabs. Small fish were also a major portion of the diet. It has been shown that food preferences for juvenile red drum from

RED DRUM FOOD PREFERENCES



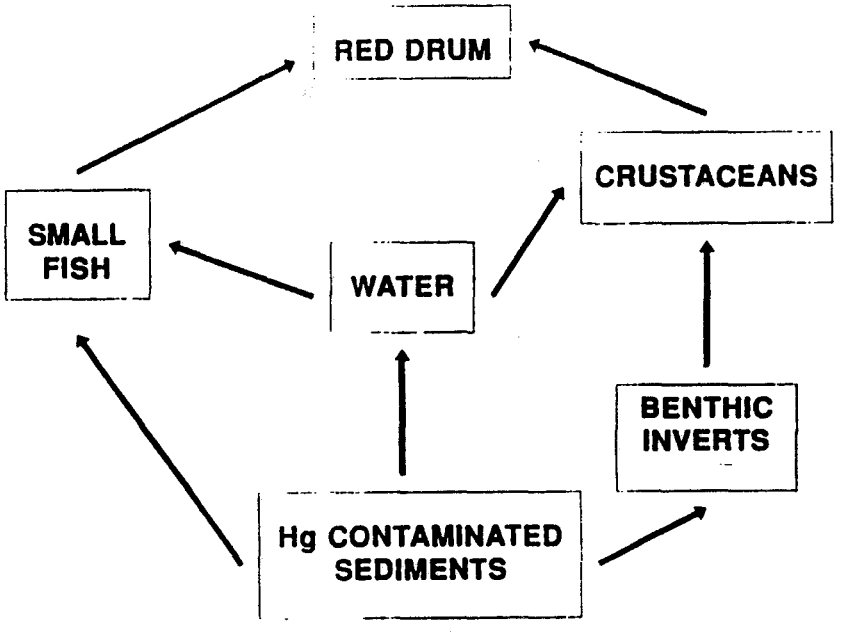


FIGURE 6. A descriptive food web model for red drum that describes the relationships between the various components of the web. The pie diagram represents the relative composition of the diet of two year old red drum in a Texas estuary. The components shown as molluscs, crustaceans, and small fish are generic designations for a number of species within each taxonomic grouping.

Louisiana change depending on time of day. They preferred grass shrimp during the day and small fish at night (Bass and Avault, 1975). Other invertebrates are consumed primarily when the fish are less than 80 mm standard length. To determine more accurately how food intake and mercury are linked, studies of stomach contents of red drum should be conducted in Lavaca Bay. Mercury analyses of organisms identified in the stomachs would allow more precise quantification of the food web pathway of mercury to red drum. For the purpose of this model we are assuming that the red drum are greater than 350 mm and in their second year of growth. Such assumptions fit the diet depicted in the pie diagram.

In the model, red drum are shown to receive their input of mercury from food rather than directly from either sediment or water (Fig. 6). Direct uptake of mercury from water is thought to be trivially small, and therefore, it is not shown in the figure. Sediment contributes to the accumulation of mercury by red drum indirectly by supplying mercury to the lower trophic level food items utilized by crustaceans and molluscs (e.g. detritus, phytoplankton, amphipods, and polychaete worms). In this scenario, the primary pathway of mercury to the red drum remains through crustaceans (crabs and shrimp), I-3,5 (Darnell, 1958; Miles, 1950). This box could be subdivided into separate crab and shrimp compartments (or even into individual species compartments) which changed in importance with the season of the year, relative species abundance, and age of the drum. Small fish also are an important input in the nutrition of red drum (Miles, 1949). They are probably less important in the accumulation of mercury, since mercury concentrations in small fish prey of red drum are thought to be much less than in crustaceans. The input of mercury from molluscs (the main "other invertebrates"), I-2,5, is predicted to be small, because red drum are active predators rather than sediment grubbers (Mercer, 1984; Perret, et al., 1980; Reagan, 1985).

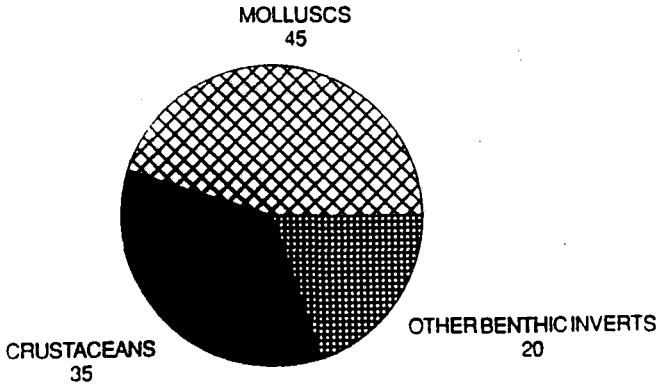
BLACK DRUM MODEL

The descriptive model of mercury accumulation for black drum concentrates on the food web utilized by the pre-adult and adult fish which range from 300-1,000 mm standard length. The commercial and recreational size limit for black drum in Texas is 14 to 30 inches (350 mm to 750 mm). Mercury analyses by the Texas Department of Health (TDH, 1988) have frequently found black drum in Lavaca Bay with concentrations of methylmercury in the edible flesh in excess of the FDA guideline of 1.0 ppm wet weight. Unlike red drum, black drum can inhabit the same estuarine habitat throughout their entire life cycle, juvenile, pre-adult and adult (Simmons and Breuer, 1962; Osburn and Matlock, 1984; Green, 1986). Of importance, black drum do not move far from where they recruited as early juveniles. This is typical of this species whether from Texas or North Carolina (Sutter, et al., 1986). Limited movement by the black drum greatly reduces the uncertainty associated with the source of their food. This information is critical in inferring the source of mercury measured in the fish.

The pie diagram in Figure 7 describes the food preferences of black drum. The black drum is primarily a mollusc predator, and is well equipped anatomically to grind and crush mollusc shells (Cave and Cake, 1980). The drum tend to grub in the surficial sediments where they turn up clams, snails, and small crustaceans (Miles, 1949; Simmons and Breuer, 1962; Sutter, et al., 1986). They also have been shown to be capable of feeding directly on oysters on an oyster reef (Cave and Cake, 1980). In Texas waters their preferred mollusc is probably Mulinia lateralis (Sutter, et al., 1986) which has been shown to contain high concentrations of mercury in Lavaca Bay, as much as 2 ppm Hg dry weight (Palmer, 1992). Black drum probably feed opportunistically on sediment inhabiting invertebrates other than molluscs, using their chin barbels to locate prey items. Their ability to turnover sediment has made them a pest in areas with extensive seagrass beds, causing considerable damage to grass beds in the search for food.

The food chain supporting the black drum is simpler than that supporting the red drum (Fig. 7). The two main sources of mercury are molluscs and crustaceans, with a strong preferences for molluscs (Miles, 1949; Simmons and Breuer, 1962; Sutter, et al., 1986) . Therefore, I-3,5 is much larger than I-4,5. The importance of an indirect input from water, is potentially greater than for red drum, because water is the most probable direct pathway of methylmercury to phytoplankton and thence to filter feeding molluscs (Fig. 10). An additional input arrow going directly from sediments to black drum might be added, because black drum are bound to consume sediment in grubbing for buried prey. It has not been added because we lack any information on the assimilation efficiency of sediment-bound methylmercury and on the rates at which sediment is ingested.

BLACK DRUM FOOD PREFERENCES



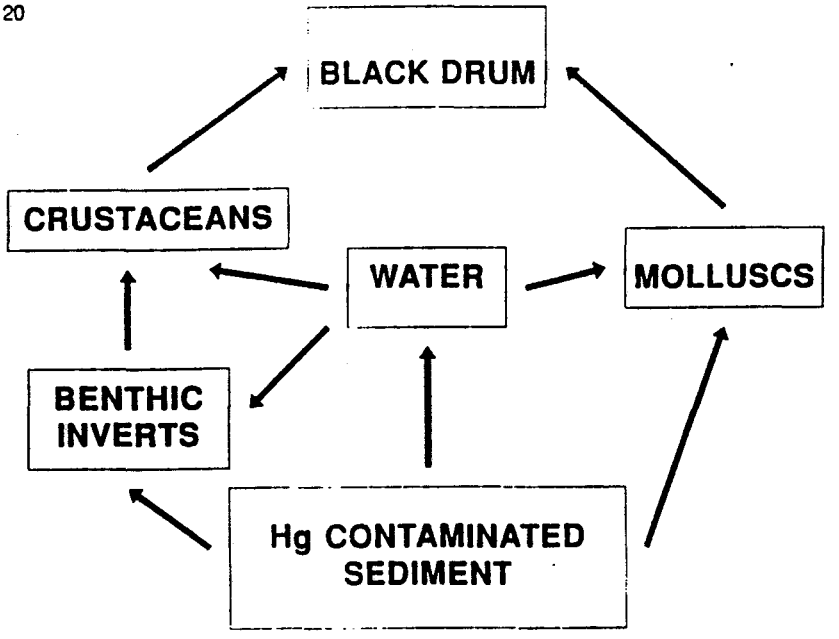


FIGURE 7. A descriptive food web model for black drum that describes the relationships between the various components of the web. The pie diagram describes the food habits of two year old black drum in Texas estuary. The components shown as molluscs, crustaceans, small fish, and phytoplankton are generic designations for a number of species within each taxonomic grouping.

BLUE CRAB MODEL

The descriptive model for blue crabs is in many ways the most complicated of the three target species. Causes for the complications are as follows: feeding patterns are not well defined; growth is discontinuous in crabs and their life span is short; growth can directly affect the turnover and metabolism of accumulated metals.

Blue crabs along with stone crabs are the only harvested invertebrate species from the Lavaca Bay/Matagorda Bay complex frequently exceeding the FDA guideline for methylmercury of 1.0 ppm wet weight. For red drum and black drum an extensive literature exists detailing feeding patterns, but feeding patterns are not as well defined for blue crabs. The blue crab tends to be an opportunistic feeder and its diet can include everything from dead red drum to juvenile oysters (Williams, et al., 1990). The types and variety of food items that blue crabs will consume, however, depends on the size and age of the crab. For example, crabs less than 31 mm feed on bivalves, ostracods, and plant detritus, crabs 31 to 60 mm feed mostly on fish, gastropods, and xanthid crabs, and crabs greater than 60 mm (adults) feed primarily on bivalves, fishes, xanthid crabs, and smaller blue crabs (Laughlin, 1982; Alexander, 1986). Adult crabs, 100 to 150 mm carapace width, feeding on a bivalve/fish/crab diet will be considered in our model. This size class is fished commercially and recreationally for human consumption. The majority of the crabs caught will be males, since the large adult females will migrate to the Gulf to spawn. Sex should not be a significant factor in mercury accumulation.

During the growth of crustaceans, the molt cycle is a discontinuous or stepped process, rather than continuous as in fish and other vertebrates. Life spans are relatively short, 3-4 years (Millikin and Williams, 1984). Both factors can influence the accumulation and retention of mercury. During the molt cycle there are extensive physiological and biochemical changes that affect both the general metabolism and metal metabolism of the animals (Mangum, 1992). Laboratory and field investigations have shown that there are significant changes in both nutritional and contaminant metal concentrations in crabs immediately after molting (Engel, 1987; Engel and Brouwer, 1987; Engel and Brouwer, 1991; Brouwer, et al., 1992). Apparently, crabs have the capability to excrete metals through the feces to reduce body burdens after molting. During molting, changes in some metal concentrations have been measured in the hemolymph and digestive gland but not in the muscle; it is not known whether methylmercury in the edible tissues of the crabs would be affected by molting. To further complicate the situation, both temperature and salinity affect the duration of

the molt cycle. In general the duration of the molt cycle increases with the size of the crab and the amount of growth appears to be correlated with the nutrition of the animals.

In the descriptive model of mercury accumulation in blue crabs (Fig. 8) there are numerous pathways that can be used, because crabs have been classified as detritivores, omnivores, and carnivores (Alexander, 1986). All of these sources of mercury , detritus, animal/plant material, sediment, and molluscs, are most likely more important than water. In the laboratory it has been shown that accumulation of the metals, copper and cadmium, is more efficient, by orders of magnitude, from food than from water (Engel, 1983). However, the water input and its importance in the model, could be dependent on the concentrations of methylmercury at the sediment water interface where the crabs live. Sediments also are a source of uncertainty in the accumulation of mercury by crabs in the Lavaca Bay estuary, because the amount of sediment ingested by crabs is unknown. In a cadmium contaminated estuary in the northeast, however, blue crabs were collected that had significant amounts of cadmium contaminated sediment in their guts (Engel, unpublished data). One of the most complicated aspects of this model is that blue crabs are highly mobile, and therefore, can move considerable distances in search of food. The primary food source for the adult crabs will be molluscs (Eggleston, 1977; Laughlin, 1982; Alexander, 1986; Bisker and Castagna, 1987), but fish and other crustaceans also will be important. The "Animal/Plant Material" component of the model may be one of the more important components of the model, but it is the most poorly defined. For that reason it is treated as a randomly linked combination of compartments. These compartments include: live fish and other crustaceans, dead and decaying fish and crustaceans, and live and dead plant material. This component also reflects the opportunistic nature of the blue crab's feeding behavior. The detritus link, I-4,5, is a logical extension of the previously mentioned component.

CRUSTACEANS, MOLLUSCS, AND SMALL FISH MODELS

The following descriptive models are for the non-target species that are major components of the food webs for red and black drum and blue crabs. Since the target organisms feed on more than one species of each of these groupings, the descriptive models are more generic than the single species food webs. The models represent the generic pathways of feeding and mercury transfer rather than specific ones.

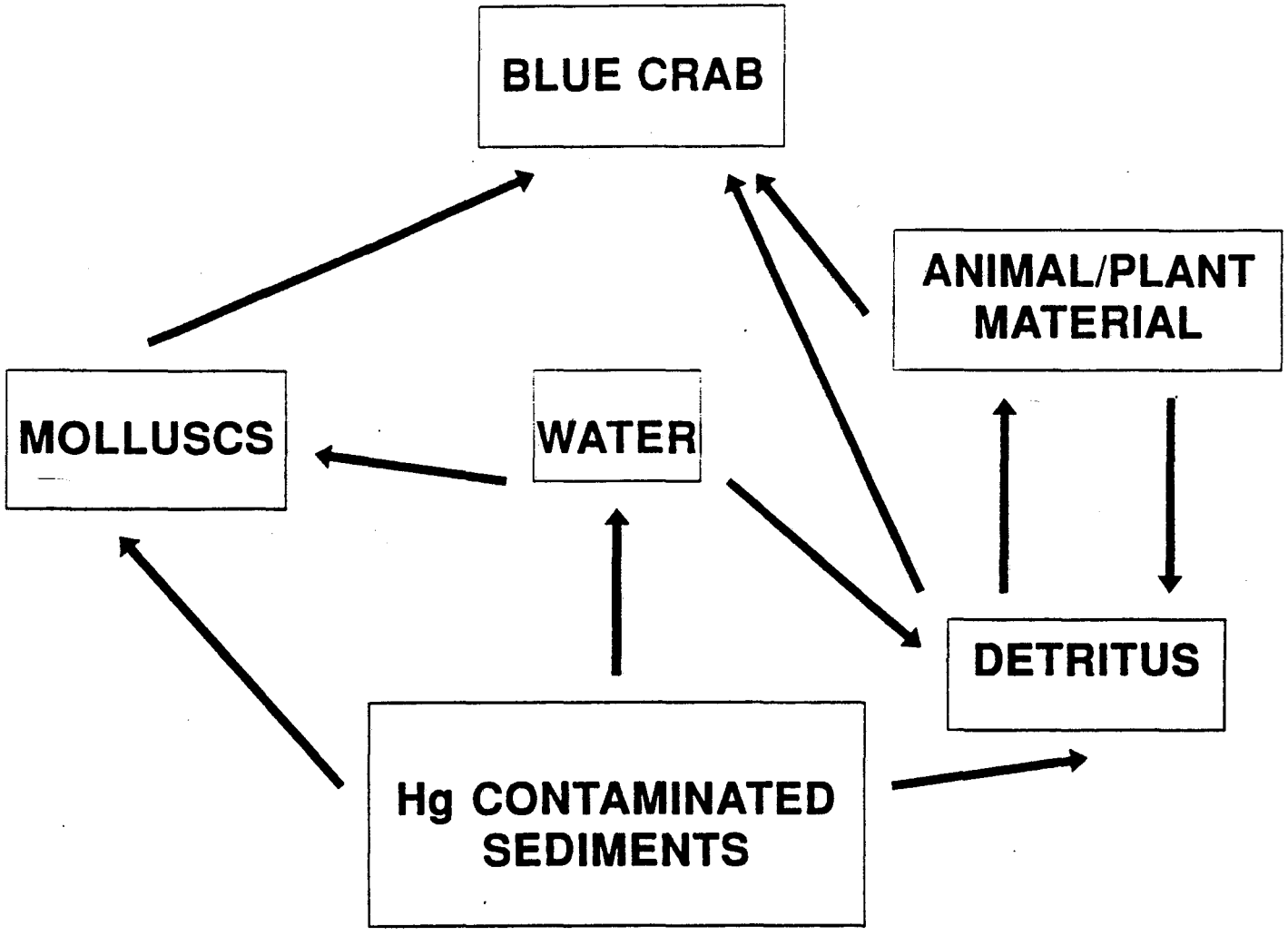


FIGURE 8. A descriptive food web model for blue crabs that describes the generic relationships between the different components of the web. The two compartments, detritus and plant/animal material, refer to non living material in different stages of decomposition, and molluscs refers to a generic taxonomic grouping.

The CRUSTACEANS grouping is comprised of members of the families Penaeidae (includes commercial shrimp), Palaemonidae (includes grass shrimp), Portunidae (includes blue crabs), and Xanthidae (includes stone crabs), all of which are macro-benthic crustaceans. During their early life stages, the three target species are heavily dependent upon micro-crustaceans (e.g. copepods, ostracods, and amphipods) as major food sources. As stated previously, in this modelling effort we are confining our efforts to those life stages that are commercially and recreationally harvestable. We will be concerned, therefore, only with the larger crustaceans. With these restrictions, it is impossible to be precise about exact food web relationships, since each family has its own unique feeding strategies and requirements. The relationships are complex (Figure 9).

The species comprising the model crustacean food web are the following: brown shrimp, Penaeus aztecus, white shrimp P. setiferus; grass shrimp, Palaemonetes vulgaris and P. pugio; mud crabs, Panopeus herbstii and Rithropanopeus harrisii; and juvenile blue crabs, Callinectes sapidus. These species are known to serve as forage organisms for red and black drum and blue crabs (Miles, 1949; Darnell, 1958; Simmons and Breuer, 1962; Bass and Avault, 1975; Overstreet and Heard, 1978; Laughlin, 1982; Minello and Zimmerman, 1983; Minello, et al. 1989).

Primary food sources for crustaceans are animal and plant detritus, epiphytic and benthic algae, and benthic invertebrates (including both micro- and macro-crustaceans) (Fig. 9). The choice of particular organisms is determined by their relative availability as a food item and the developmental stage or size of the feeding crab or shrimp. All species but xanthid crabs are found primarily in seagrass meadows and salt marshes where detritus is abundant. All are omnivores and detritivores (Alexander, 1986; McTigue and Zimmerman, 1991). Shrimp feed heavily on plant material and benthic algae as well as on epiphytes of seagrass (Gleason and Zimmerman, 1984; Gleason, 1986). Both juvenile blue crabs and mud crabs feed on small molluscs, such as juvenile oysters (Bisker and Castagna, 1987; Eggleston, 1990), but the particular species is dependent upon availability (Laughlin, 1982). Other benthic invertebrates, such as polychaete worms, are available as food, but little field information is available for soft bodied animals, because they are difficult to identify as part of the stomach contents. The transfer of mercury from the sediment is mediated through the water to benthic algae and possibly through the seagrasses and marsh plants.

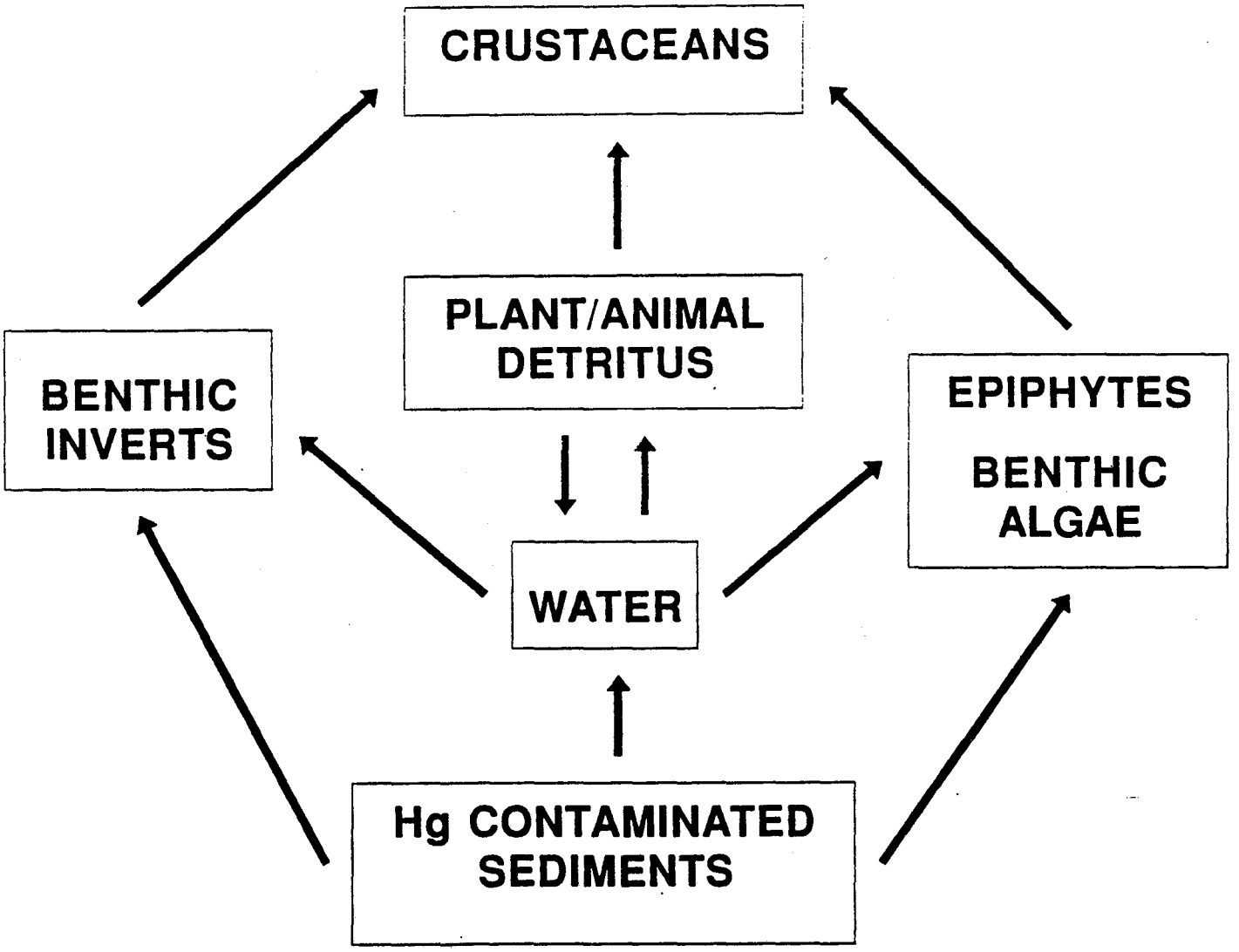


FIGURE 9. A descriptive food web model for "crustaceans" which includes a large number of different species of organisms with different food habits and requirements. This model is an effort to integrate all important estuarine crustaceans into one generic food web relationship.

The MOLLUSC component of the food web contains a large variety of bivalves and gastropods too numerous to list. Molluscs are the main food of black drum and an important one for blue crabs and red drum (Miles, 1949; Simmons and Breuer, 1962; Cave and Cake, 1980; Laughlin, 1982; Millikin and Williams, 1984; Sutter, et al., 1986; Bisker and Castagna, 1987; Eggleston, 1990; Williams, et al., 1990). The mollusc species that has received the most study is the American oyster, Crassostrea virginica. Its life history is typical of most bivalve molluscs (Galtsoff, 1964) except that upon settling it becomes entirely sessile. Depending on where settlement occurs, oysters can serve as indicators of local discharges of and exposure to contaminants, particularly metals (NOAA, National Status and Trends, Mussel Watch Program). Their ability to bioconcentrate contaminants also makes oysters excellent vectors of mercury and other metals to the organisms that consume them.

The food web relationships for molluscs are simpler than for crustaceans (Figure 10). Molluscs accumulate metals and nutrients both through the consumption of phytoplankton and directly from the water. Resuspended sediment may also be a source of mercury to filter feeding molluscs. Few deposit (i.e. sediment) feeding molluscs such as Macoma sp. are abundant in Lavaca Bay (White et al, 1989). In Texas estuaries the dominant food for black drum is Mulinia lateralis which inhabits muddy substrates (Sutter, et al., 1986). This species is very common in Lavaca Bay (White et al, 1989) in muddy and muddy sand substrates. High concentrations of total mercury have been found in Mulinia lateralis from Lavaca Bay (Palmer, 1992). Oysters, which have been widely monitored for mercury in Lavaca Bay (TDH, 1988), could be an important secondary source of mercury.

The SMALL FISH component of the food web models for red drum and blue crabs is extremely difficult to delineate (Figure 11), because of the number of different species of fish that comprise this group. Some of the fish species that may be included are: Gulf menhaden, Brevoortia patronus; pinfish, Lagodon rhomboides; Atlantic croaker, Micropogonias undulatus; striped mullet, Mugil cephalus; Gulf killifish, Fundulus grandis; bay anchovy, Anchoa mitchilli; and spot, Leiostomus xanthurus. One of the main difficulties is that all of these fish use different feeding strategies, and feed on different components of the estuarine food web. In the cases of the three target species, the small fish components of the models are all secondary to other more important trophic pathways. Of all the descriptive models that have been constructed, the small fish model has the highest level of uncertainty. The uncertainties are derived from the large number of species and trophic diversities. Also, there is limited information on mercury concentrations in potential small fish prey in Lavaca Bay. However, Atlantic croaker, Micropogonias undulatus from Lavaca Bay had higher concentrations of mercury in their flesh and liver than the same species from other Texas estuaries (TDH, 1988; NBSP, 1988).

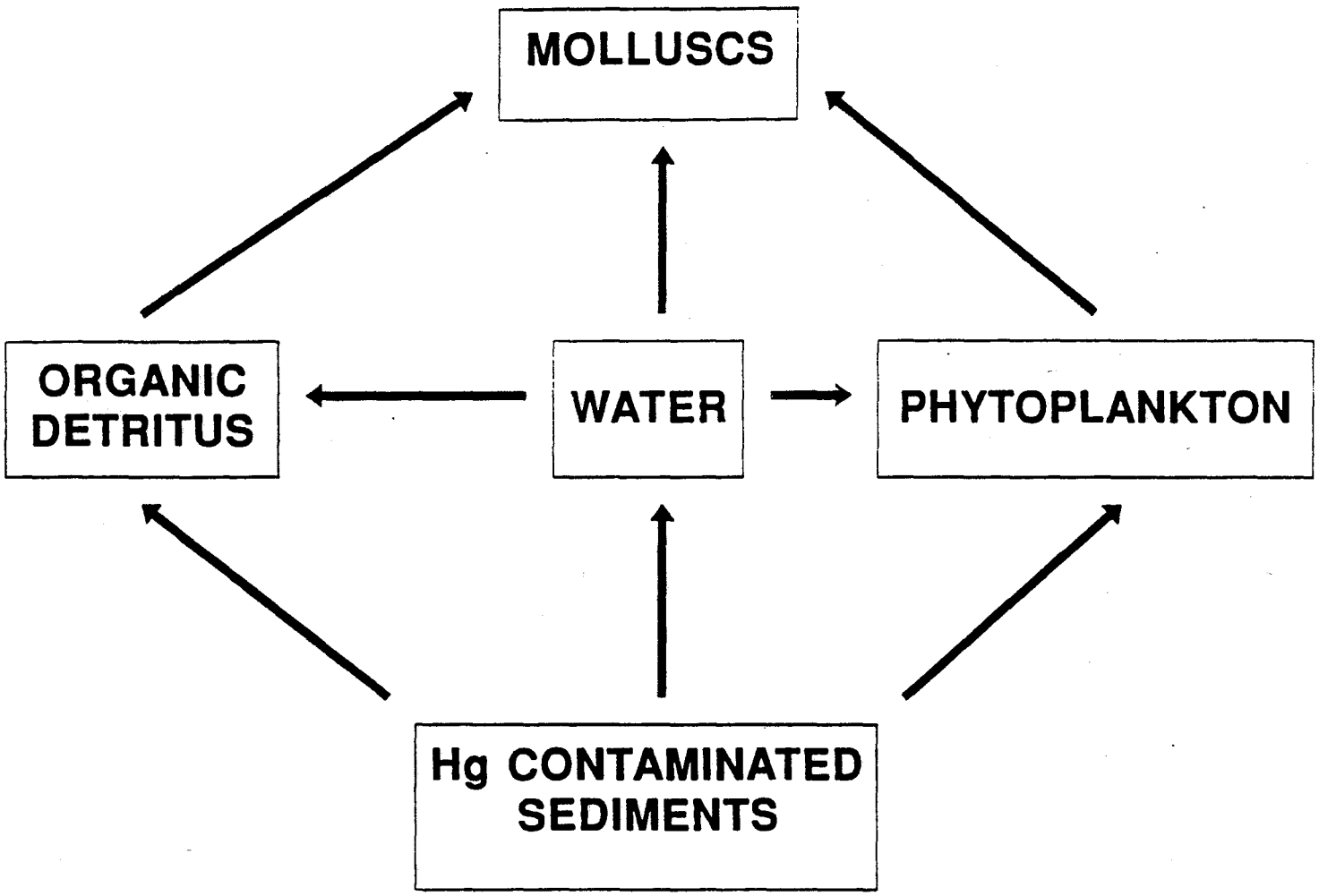


FIGURE 10. A descriptive food web model for "molluscs" which includes a large number of different species of organisms with different food habits and requirements. This model is an effort to integrate all important estuarine molluscs in Texas into one generic food web relationship.

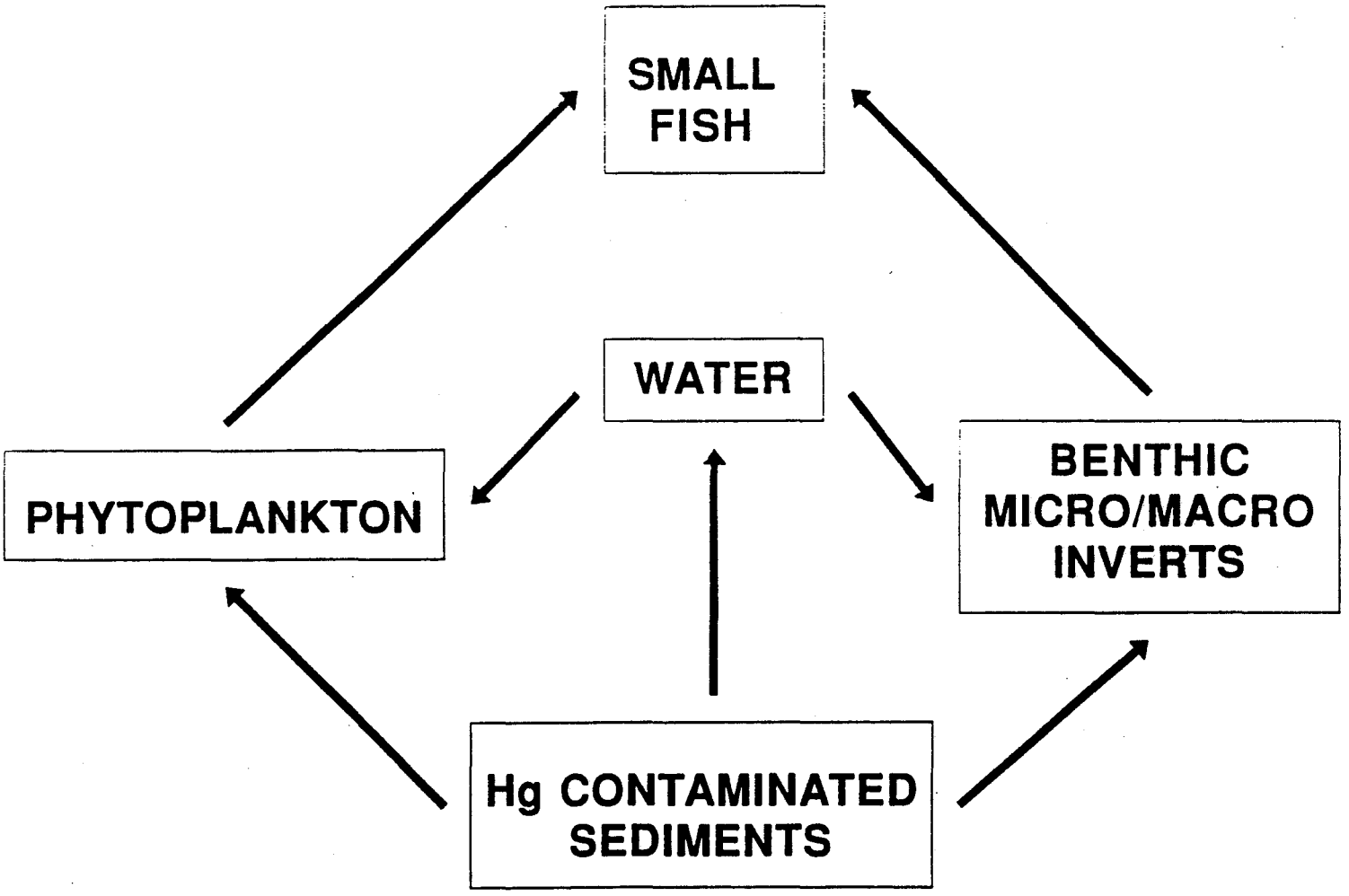


FIGURE 11. A descriptive food web model for "small fish" which includes a large number of different species of organisms with different food habits and requirements. This model is an effort to integrate all important small estuarine forage fish in Texas into one generic food web relationship.

III. MODEL IMPLEMENTATION FOR RED DRUM

It is possible to take a small step beyond development of a conceptual model of mercury bioaccumulation and attempt an initial quantification of the red drum model. We do this as a mechanism to demonstrate its implementation, provide initial predictions of mercury concentrations, identify data needs and uncertainties, and identify model limitations. In this exercise, we will assume a two year old red drum of 500 mm length and 2000 g weight which has been exposed for its entire lifetime to a constant total mercury concentration in sediments of 1 ppm Hg. This total mercury concentration is typical of that measured in the area of Lavaca Bay closed to fishing(Table II).

Table II. Total Mercury concentrations in surface sediments in the zone closed to fishing in Lavaca Bay (ug Hg/g).

Source	range	mean	median
Palmer, 1992	0.43 to 1.35	0.82	0.67
Riegel, 1990	0.29 to 1.86	0.80	0.64
Bowman, 1988	0.11 to 7.10	1.37	0.80
Woodward Clyde, 1992	0.02 to 25.2	1.17	0.63

CONSENSUS VALUE 1.0 ug Hg/g

The model implementation will seek to describe methylmercury concentrations in red drum and lower trophic components. There are two reasons for this. First, since 1984, FDA action limits on mercury contamination of seafood specify methylmercury rather than total or inorganic mercury as the substance of concern (FDA, 1984). Second, the greater assimilation efficiency, and longer retention time of methylmercury as compared to inorganic mercury in biota means that methylmercury is the predominant form of mercury in most biota. This is true despite the fact that concentrations of inorganic mercury are many times higher than methylmercury in sediments. Thus, a separate modeling effort for the accumulation of inorganic mercury by biota is not needed.

TROPHIC TRANSFER OF INORGANIC MERCURY IN LAVACA BAY FOOD WEBS

The transfer of inorganic mercury is largely ignored in the following discussion for several reasons. Compared to methylmercury, inorganic mercury is preferentially selected against in partitioning from sediments into the more bioavailable aqueous phase. The sediment to water partition coefficient may be many times greater for inorganic mercury than for methylmercury (Hudson

et al., 1992). The assimilation of methylmercury from water across the gill surfaces of fish and invertebrates may be many times more efficient than that for inorganic mercury, perhaps 2 to 20 fold more efficient (Pentreath, 1976a,b,c; Fowler and La Rosa, 1978). Assimilation efficiencies of methylmercury from food are also greater by a factor of 2 to 20 (Pentreath, 1976a,b,c; Riisgard and Famme, 1986) Conversely, methylmercury is also less rapidly excreted from organisms than inorganic mercury by a similar factor (Pentreath, 1976a,b,c; Fowler and La Rosa, 1978) resulting in greater relative methylmercury retention. One might predict a 10 fold preference for partitioning of methylmercury relative to inorganic mercury from sediments into water, a 4 fold preference in partitioning from water into a sediment inhabiting invertebrate and a further four fold preference in retention in the invertebrate because of the longer excretion half life. It is easy to see how methylmercury that constitutes 0.5% of total mercury in estuarine sediments could become as much as 40% in invertebrates or small fish prey of red drum(i.e. (0.5%)(5)(4)(4)=40%). Operation of the same processes of relative retention of methylmercury in red drum feeding on these prey would result in the observation that nearly all of the mercury found in edible tissues of red drum and other top predators is found as methylmercury. This process of preferential bioaccumulation of methylmercury has been demonstrated experimentally in a food chain leading to flounder (Riisgard and Hansen, 1990).

Red drum are postulated to accumulate methylmercury from four sources: directly from water and by eating three taxa of mercury containing prey (60% crustaceans, 30% small fish, and 10% infaunal benthic invertebrates such as molluscs and worms). However, we will show in a subsequent discussion of methylmercury accumulation by small fish (which is also applicable to red drum and their other prey), that direct uptake of methylmercury from water is likely to be an inconsequential pathway of accumulation. Small fish are assumed to accumulate methylmercury indirectly from water via phytoplankton (especially menhaden and anchovies) or indirectly from sediments via benthic invertebrates, pathways not explicitly shown in Figure 11. Crustaceans and infaunal invertebrates are assumed to accumulate methylmercury directly from mercury containing sediments. These sources are assumed to be additive inputs to red drum. Although red drum may ingest sediment when feeding, it is unlikely to be an important source of methylmercury because of the low concentrations and assimilation efficiency in sediment relative to living prey. Thus, there is no direct sediment to red drum pathway. Red drum are assumed to excrete methylmercury with a characteristic rate that balances the input of methylmercury at a steady state equilibrium.

With this background, it is possible to attempt an initial parameterization of the model for methylmercury in red drum along the lines of that taken by Fordham and Reagan (1991). This approach bases model parameterization on an assumed equilibrium concentration of methylmercury in water to which red drum and their

prey are exposed. This model is supplemented by terms describing direct uptake of methylmercury from sediments by crustaceans and benthic invertebrates along the lines of Thomann et al.(1992). Unfortunately, at this time, we do not know either the methylmercury concentrations in water or in sediments from Lavaca Bay to which food web organisms are exposed. To develop the model implementation, we must estimate methylmercury concentrations indirectly from estimates of total mercury concentrations and relationships between methylmercury and total mercury concentrations derived from the literature.

RELATIONSHIPS THAT CONTROL THE STEADY STATE DYNAMICS OF THE DESCRIPTIVE MODEL

I = Input rate of Hg to a target species such as red drum

a = Assimilation efficiency of mercury from food

R = Feeding rate of target species

C_o = Hg concentration in food item

E = Excretion rate of Hg by target species

C_1 = Hg concentration in target species

K = Hg excretion rate constant at steady state in target species

.....

1. $I = (a)x(R)x(C_o)$

2. $E = (C_1)x(K)$

3. I - E = rate of change of mercury concentration in target species over time

At steady state, input and excretion of mercury are in balance, and mercury in the target species does not change over time.

4. $I - E = 0$ or

5. $(a)x(R)x(C_0) = (C_1)x(K)$ Rearranging

6. $C_1 = [(a)x(R)x(C_0]/(K)$

7. B.A.F. = Hg Bioaccumulation Factor = $[C_1/C_o] = (a)x(R)/(K)$

SEDIMENT TO WATER TRANSFER OF MERCURY

In marine and estuarine sediments, methylmercury (MeHg) is typically in the range of 0.01% to 1% of the total mercury (Bartlett and Craig, 1981; D'Itri, 1990). We will assume a value of 0.5%. The sediment to water partition coefficient of methylmercury is not well known, estimates ranging from 10^4 L/kG to 10^6 L/kG (Table III). We will assume a value of 10^5. This would imply a methylmercury concentration of 0.00005 ppb MeHg in pore waters in equilibrium with sediments containing 1 ppm total mercury (Equations 9 and 10, pg. 36). Methylmercury dissolved in estuarine waters overlying contaminated sediments would have lower concentrations due to dilution of sediment pore water that was the source of methylmercury. Measured methylmercury concentrations in estuarine and marine waters have ranged from 0 and 0.00016 ppb (Table IV). The value 0.00005 ppb MeHg predicted from the 1 ppm total mercury concentration in sediment seems to be a reasonable estimate that can be used as an input in the transfer of methylmercury to Lavaca Bay biota.

Table III. Partition coefficients for methylmercury between water and suspended sediments (K_d)

Source	K_d (L/Kg)
Mason and Fitzgerald, 1990	10^6
Hudson et al., 1992	10^4
Evans et al., 1984	2×10^4

CONSENSUS VALUE 10^5 L/Kg

Table IV. Reported estuarine, coastal and open ocean seawater methylmercury concentrations

Source	Range of concentrations (ppb=ug/L)
Mason and Fitzgerald, 1990	0 to 0.000060
Yamamoto et al., 1983	0.000040 to 0.000160
Bloom, 1989	0.000016
Davies et al., 1979	0.000060(indirect estimate)

CONSENSUS VALUE 0.000050 ppb

The only reliable measurements of total mercury concentrations in Lavaca Bay are those of Gary Gill, reported in Palmer(1992); 0.0045 ppb Hg was found near Point Comfort and 0.0005 ppb Hg was found in Keller Bay. Methylmercury concentrations in open ocean and coastal waters have been measured at from 1% to 10% of total mercury concentrations (Mason and Fitzgerald, 1990; Bloom, 1989). If similar ratios hold for Lavaca Bay, one would predict methylmercury concentrations in the range of 0.000050 ppb to 0.00050 ppb MeHg, similar to the 0.000050 ppb MeHg predicted from sediment partitioning calculated above.

MERCURY TRANSFERS FROM WATER TO ORGANISMS(BCFs)

Few bioconcentration factors(BCFs) are available for methylmercury accumulation in aquatic organisms, which is not surprising given the long time required to reach equilibrium in controlled laboratory studies, and the difficulty in measuring methylmercury concentrations in natural waters where a BCF might be estimated under field conditions. Pentreath (1976a,b,c) determined a methylmercury BCF of 3.3×10^3 for small flounder. Kopfler's (1974) data suggests a BCF of 4×10^4 for methylmercury in oysters. Zubaric and O'Conner (1978) reported a BCF of 10^3 for gammarid amphipods from freshwater. At an exposure concentration of 0.00005 ppb MeHg, these BCF's would yield equilibrium methylmercury concentrations of 0.00016 ppm wet weight MeHg for flounder, 0.002 ppm wet weight MeHg for oysters, and 0.00005 ppm wet weight MeHg for amphipods. These predicted methylmercury concentrations are all very much less than those reported in fish, molluscs, and crustaceans collected in the vicinity of mercury contaminated sediments of Lavaca Bay (Table I). This suggests that direct uptake from water is a trivial source of methylmercury to higher organisms of this area. It is for this reason that we do not develop the model pathway directly from water to red drum; it is likely to be negligibly small. For benthic invertebrates, direct sediment ingestion and indirect bioaccumulation through micro- and meio-flora and fauna must represent the main routes of methylmercury transfer. Accumulation of methylmercury by small fish preyed upon by red drum is also likely to be through consumption of their food, whether it is phytoplankton, zooplankton, or other fish.

MERCURY IN SMALL FISH PREY OF RED DRUM

Reports of mercury concentrations in small fish in Lavaca Bay upon which red drum feed (such as mullet, menhaden, and bay anchovy) are limited. However, juvenile Atlantic croaker, *Micropogonias undulatus*, from Lavaca Bay contained an average of 0.07 ppm wet weight of total mercury and mullet averaged 0.135 ppm wet weight of total mercury (TDH, 1988). Blanton et al. (1972) reported an average total mercury concentration in Lavaca Bay mullet of 0.15 ppm wet weight of which one third was methylmercury

(i.e. methyl mercury concentrations were 0.05 ppm wet weight). We will assume a methylmercury concentration of 0.05 ppm wet weight in the fish fed upon by red drum (Table V).

Table V. Mercury and methylmercury in Lavaca Bay forage fish (ppm wet weight)

Source	Fish	Total Hg	Methyl Hg
Blanton et al. 1972	mullet	0.15	0.05
TDH, 1988	mullet	0.135	
	Atlantic croaker	0.07	

CONSENSUS VALUE = 0.05 ug Hg/g wet weight as methylmercury

MERCURY ACCUMULATION FROM SEDIMENTS BY INVERTEBRATE FOOD ORGANISMS OF RED DRUM

Empirical relationships, called Biota Sediment Factors (BSF), ratio contaminant concentrations in benthic organisms to concentrations in the sediments they inhabit. BSF values for total mercury are available in the literature (Table VI). BSF values for methylmercury are rare, however, and better estimates might be made in predicting methylmercury concentrations from published BSF values for total mercury and estimating the percentage of total mercury in the organisms that is methylmercury (Table VI).

TABLE VI. Biota Sediment Factors (BSF) ratioing total mercury concentrations (dry weight) in benthic organisms to total mercury concentrations in habitat sediments. Reported values are from multiple sources in the literature.

ORGANISM TYPE	BSF			
	MEDIAN	MINIMUM	MAXIMUM	INTERQUARTILE RANGE
WORMS	2.7	0.12	22.2	0.56 - 6.8
CLAMS	1.5	0.28	27.6	0.9 - 3.1
MUSSELS	0.75	0.08	21	0.24 - 2.47
OYSTERS	1.9	1.2	5.7	----------
GASTROPODS	2.68	0.05	75	1.25 - 5.7
SHRIMP	0.78	0.00	8.6	0.1 - 1.6
CRABS	0.95	0.07	40	0.33 - 5.3

CONSENSUS OVER ALL BENTHIC ORGANISMS: BSF = 2

Values of BSF for total mercury in other estuarine and coastal systems seem to center around 2 on a dry weight basis but values range from <0.1 to >10 depending on species and other factors (Table VI). We will use a BSF value for total mercury of 2 for all species of molluscs, crustaceans and other benthic fauna that serve as prey of red drum.

A BSF value of 2 would predict a total mercury concentration of 2 ppm Hg (dry weight) in benthic organisms exposed to our model Lavaca Bay sediments of 1 ppm Hg concentration. Palmer (1992) reported mean total mercury concentrations in benthic fauna from Lavaca bay sites (with sediment mercury concentrations of about 1 ppm Hg) of 1.4 to 10.1 ppm Hg dry weight for various species of polychaete worms, 0.9 to 4.3 ppm Hg dry weight for various bivalve molluscs, and 0.2 to 1.5 ppm Hg dry weight for various crustaceans, including 1.2 ppm Hg dry weight for blue crabs and 1.5 ppm Hg dry weight for penaeid shrimp. Both are major food items of red drum. The Texas Department of Health has measured total mercury concentrations in edible flesh of blue crabs in Lavaca Bay. During the period 1981 to 1991, they report a median total mercury concentration in blue crabs from the Point Comfort area of 0.43 ppm Hg wet weight (TDH, 1988). This is equivalent to 2.15 ppm Hg dry weight assuming a wet weight to dry weight ratio of 5. It would appear that measured total mercury concentrations in benthic organisms of Lavaca Bay are roughly consistent with concentrations predicted from BSF values reported in other contaminated coastal ecosystems. Mobile epibenthic organisms such as shrimp and crabs may have lower total mercury concentrations less than some infaunal organisms. This seems consistent with lower BSF values reported elsewhere. It is unknown whether this results from their ability to move into and out of mercury contaminated habitats during migrations or from differences in feeding habits and reduced mercury exposure as epibenthic dwellers. Nevertheless, published BSF values seem capable of roughly predicting the total mercury concentrations in benthic invertebrate prey of red drum.

SEDIMENT ORGANIC CARBON (TOC) AS A FACTOR LIMITING MERCURY BIOAVAILABILITY FROM SEDIMENTS

Sediment mercury concentrations and BSFs are usually normalized for organic carbon (TOC) content to adjust for its presumed influence on mercury bioavailability (Langston, 1982). We assume a TOC of 1% carbon (equivalent to about 2% organic matter) which is typical of fine grained sediments in the Point Comfort area of Lavaca Bay (White et al. 1989). Langston (1982) reported the following relationships between mercury in bivalves and in sediments:

Equation 8. $C_o = (A) x (C_s / \%\ \text{organic matter}) + B$

where

C_o = total Hg in bivalve in ppm dry weight

C_s = total Hg in sediment in ppm dry weight

A and B are fitted constants

with A=1.95 and B=0.22 for Scrobicularia plana and A=2.72 and B=0.34 for Macoma baltica, both are deposit feeing bivalves

This would yield 1.2 ppm and 1.7 ppm dry weight for the total mercury content of these respective species exposed to sediment with a total mercury concentration of 1 ppm Hg and 1% TOC. Several other investigators have found inverse relationships between total mercury or methylmercury accumulation in benthic fauna and the TOC or organic carbon content of their sediment habitat (Eganhouse and Young, 1978; Breteler et al., 1981; Langston, 1986).

METHYLMERCURY AS A PERCENTAGE OF TOTAL MERCURY IN LAVACA BAY BIOTA

With the exception of a few measurements on blue crabs, the proportion of total mercury that is in the methyl form is not known for benthic invertebrates in Lavaca Bay. Ward et al.(1979) found an average of 35% of the total mercury was methylmercury in blue crabs from the contaminated area of Lavaca Bay. In contrast, the Texas Department of Health found the percentage of methylmercury among blue crabs with greater than 1 ppm total mercury was 112% (standard deviation of 35%) in Lavaca Bay (TDH,1988). The difference, if real, may reflect the 20 year difference in sampling times.

TABLE VII. Methylmercury as a percentage of total mercury in various groups of marine organisms. Reported values are taken from the literatrue much of which is referenced herein.

ORGANISM GROUP	PERCENTAGE METHYLMERCURY			
	MEDIAN	MINIMUM	MAXIMUM	INTERQUARTILE RANGE
FISH	80	21	112	65 - 93
WORMS	32	4	43	25 - 36
CLAMS	20	1	60	6 - 30
MUSSELS	20	4	36	15 - 24
OYSTERS	27	17	29	-------
GASTROPODS	60	7	72	43 - 68
SHRIMP	50	23	100	46 - 60
BLUE CRABS	63	35	112	57 - 69
CRABS (other)	85	45	102	81 - 89
MARSH PLANTS	2	2	2	-------

Differences in sampling location might also be important; Hornung et al. (1984) found that the proportion of methylmercury increased in mussels sampled at increasing distances from the outfall of a chlor-alkali plant. Epibenthic crustaceans are like fish in seeming to have a high percentage of their mercury body burden as methylmercury (Table VII). Non-predatory molluscs and worms seem to have methylmercury as a much smaller proportion of their total mercury (Table VII), although filter feeding cockles are reported to have a range from 30% to 90% methylmercury (Mohlenberg and Riisgard, 1988). We will assume that methylmercury is 70% of total mercury in crustaceans, and 25% of total mercury in infaunal invertebrates (molluscs and worms) in Lavaca Bay.

MERCURY ACCUMULATION BY RED DRUM FROM PREY ORGANISMS

The last information required to implement the red drum model concern red drum itself. Growth rates for red drum have been reported to be about 0.1%/day for total length (Matlock, 1992). This is equivalent to about 0.3%/day on a weight basis. For pond cultured red drum of greater than 100 g initial weight, production to consumption ratios are about 0.14 (Trimble, 1979), which is similar to those observed for benthic fish such as cod and haddock in the western Atlantic (Grosslein, et al., 1980). From this ratio, it is possible to estimate the food consumption rate necessary to support measured growth rates in free living red drum; this is about 2.1%/day (=0.3%/day/0.14). We will assume a feeding rate of 2%/day (R=0.02g/g/day) for juvenile red drum. From the relationship developed by Sharpe et al. (1977), the excretion rate constant for methylmercury for a juvenile 2000 g fish would be K=0.00035/day. The assimilation efficiency [a] of methylmercury in food by fish is generally found to be about 0.8 (deFreitas et al., 1977).

Thus at steady state equilibrium we have the following relationships for methylmercury concentrations in the simplified food web of the red drum. A dry weight to wet weight ratio of 0.2 is assumed for crustaceans and other invertebrates.

9. Sediment
   $C_0=(0.005)(C_T)=0.005$ ppm MeHg
   where C_T=total sediment mercury concentration = 1.0 ppm Hg

10. Water
   $C_1=(k_p)(C_0)$ = (0.005 ppm MeHg)/10^5 L/Kg = 0.00005 ppb MeHg

11. Small Fish
   C_4=0.05 ppm MeHg wet weight

12. Crustaceans

$C_3=(BSF_3)(C_T)(P_3)=(2)(1\text{ ppm Hg})(0.7)=1.4$ ppm MeHg dry weight
$=0.28$ ppm MeHg wet weight

where P_3=proportion of total mercury in crustaceans that is methylmercury = 0.7

13. Other Invertebrates

$C_2=(BSF_2)(C_T)(P_2)=(2)(1\text{ ppm Hg})(0.25)=0.5$ ppm MeHg dry weight
$=0.1$ ppm MeHg wet weight

where P_2 = Proportion of total mercury in invertebrates that is methylmercury = 0.25

14. Red Drum

$C_5=[(a)(R)/(K)][(\%_4)(C_4)+(\%_3)(C_3)+(\%_2)(C_2)]$
$=[(0.8)(0.02/\text{day})/(0.00035/\text{day})]\times$
$[(0.3)(0.05\text{ ppm MeHg wet weight})+(0.6)(0.28\text{ ppm MeHg wet weight})+(0.1)(0.10\text{ ppm MeHg wet weight})]$

$=[46][0.015 + 0.168 + 0.010] = 8.9$ ppm MeHg wet weight

The model prediction of 8.9 ppm MeHg wet weight in red drum is about twice the highest measured value of 4.55 ppm MeHg measured in Lavaca Bay red drum since 1981 (TDH, 1988). Model predictions might be much lower if one accounts for the rapid growth of red drum and the effect growth has in diluting methylmercury concentrations. In Equation 6 the effect of growth on methylmercury concentrations in red drum can be approximated by substituting (g + K) for (K), where g is the growth rate coefficient for red drum. We earlier estimated g=0.003/day. Where excretion is slow (small K) and growth is fast (large g), the term (g +K) is dominated by g. Thus (g + K)=(0.003 + 0.00035)=(0.00335) for red drum. The consequence of growth dilution would be to reduce the predicted concentration of methylmercury in red drum (C_6) by about a factor of 10, to 0.9 ppm wet weight. The role of growth dilution needs to be more fully developed.

An important result of the model parameterization is to identify crustaceans as the dominant source of methylmercury to red drum. This is a consequence not only of their high proportion in the diet of red drum, but also their relatively high total mercury concentration and the relatively high proportion of total mercury that is methylmercury in crustaceans compared to either invertebrates or small fish.

IV. CONCLUSIONS

The bioaccumulation literature search and modelling exercise has answered many of the qualitative questions concerning the accumulation of mercury in red drum, black drum, and blue crab food webs. This effort also has identified data gaps about life histories and food web relationships that need to be filled before a truly predictive model of mercury food web dynamics can be constructed.

The following conclusions can be drawn from the literature search and modelling effort:

* Available data indicates that some mercury concentrations in the sediments of Lavaca Bay are in excess of 1.0 ppm. Published sediment distributions of mercury indicate higher concentrations adjacent to the ALCOA Point Comfort plant.

* Data from the Texas Department of Health, Texas Parks and Wildlife Department, Texas Water Commission, and academic institutions in Texas indicate that elevated levels of mercury in the target species have persisted. In a restricted portion of the bay methylmercury concentrations in edible flesh of some fish and shellfish species exceed the FDA guideline of 1 ppm wet weight.

* The relatively limited intra-estuarine movements of red and black drum during their first three years of life allows them to be exposed for extended periods of time to locally elevated concentrations of mercury in the sediment. This aspect of their life history may partly explain the high mercury concentrations in these fish caught in Lavaca Bay.

* Preferred food of red drum (crustaceans) and black drum (molluscs) are benthic organisms which can acquire high mercury concentrations from mercury contaminated sediments. This short direct pathway helps explain high mercury concentrations observed in fish flesh.

* Blue crabs also have elevated mercury concentrations. Their food web pathways are more complicated than red drum or black drum, but direct feeding on sediment inhabiting invertebrates and detritus are likely mechanisms of accumulation.

* The descriptive models for the red drum, black drum, and the blue crab identify critical but poorly defined steps between sediment and primary consumers that control the availability of mercury to the target species. Methylation of inorganic mercury to methylmercury and methylmercury accumulation by benthic organisms are likely steps.

* Low organic matter concentrations in Lavaca Bay sediments probably contributes to enhanced bioavailability of mercury to benthic invertebrate prey of the target species.

* Using red drum as an example, the implementation of the descriptive model has shown that using literature values for partition coefficients, growth, bioaccumulation, and excretion rates, and concentration factors, it was possible to predict with some accuracy concentrations of methylmercury accumulated by red drum in Lavaca Bay.

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VI. ANNOTATED BIOACCUMULATION REFERENCES

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RED DRUM - Sciaenops ocellatus
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Arnold, C.R., G.J. Holt, and P. Thomas (eds.). 1988. Red drum aquaculture. Proceedings of a symposium on the culture of red drum and other warm water fishes. Contrib. Mar. Sci. 30: 197 pp.

A collection of 36 individual papers concerning the following topics: spawning technology; fingerling production technology; biological, engineering and regulatory aspects; growout technology; and an annotated bibliography.

Bass, R.J. and J.W. Avault, Jr. 1975. Food habits, length-weight relationship, condition factor, and growth of juvenile red drum, Sciaenops ocellata, in Louisiana. Trans. Am. Fish. Soc. 104(1): 35-45.

Juvenile red drum (8-183 mm SL) displayed a fair degree of size dependent selectivity in feeding. Foods selected according to fish length were:1. <15 mm, zooplankton; 2. 15-75 mm, bottom invertebrates and small fish; 3. >75 mm, decapods (crabs and shrimp) and fish. Day/night food differences for fish 65-85 mm were grass shrimp during the day and fish at night. The length-weight relationship, average coefficient of condition, and average growth rate were: 1> log W = -7.2052 + (4.1913); 2> 1.969; and 3> range 13.8 and 25.6 mm/mo.

Beckman, D.W., C.W. Wilson and A.L. Stanley 1989. Age and growth of red drum, Sciaenops ocellatus, from offshore water of the northern Gulf of Mexico. Fish. Bull. U.S. 87: 17-28.

The offshore populations of red drum in the Gulf of Mexico ranged in age from 1 to 37 years. The vast majority of the fish captured in this study were over 10 years of age. Aging was done using otolith sagittae.

Buckley, J. 1984. Habitat suitability index models: larval and juvenile red drum. U.S. Fish Wildl. Service FWS/OBS-82/10.74, 15 pp.

Chamberlain, G.W., R.J. Miget, and M.G. Haby (comps.). 1990. Red drum aquaculture. Rept. Tex. A&M Univ. Sea Grant Program, 236 pp. See: Arnold, et al. 1988. This is the same symposium.

Darnell, R.M. 1958. Food habits of fishes and larger invertebrates of Lake Pontchartrain Louisiana, an estuarine community. Publ. Inst. Mar. Sci. Univ. Texas 5: 354-416.

Small Texas redfish 40 mm consumed small crustaceans (schizopods and amphipods). Large fish consumed penaeids, blue crabs, and small fish. Lake Pontchartrain fish (181-625 mm) consumed blue crabs and mud crabs (62%), fish remains (17%), and unidentified organic material.

Doerzbacher, J.F., A.W. Green, G.C. Matlock and H.R. Osburn. 1988. A temperature compensated von Bertalanffy model for tagged red drum and black drum in Texas bays. Fish. Res. 6(2): 135-152.

Red drum were tagged and recaptured in Texas bays from 11/75-6/85, and estimates of von Bertalanffy growth model (VBGM) parameters were made. An annual temperature model was used to examine the growth model when periods of cold temperatures were excluded. The best fit of the VBGM for red drum was based on time at large expressed in day-degrees with the coldest 60 days of the year excluded and yielded parameter estimates (SE) of K=0.422(0.023) and L sub(infinity) = 918 (21) mm.

Green, L. 1986. Fish tagging on the Texas coast, 1950-1975. Manage. Data Ser. 99, Tex. Parks Wildl. Dep., Coastal Fish. Br., Austin, 206 p.

This study was a longterm tagging and recapture study of a number of different species of fish, but the fish of concern, red and black drum were among the fish tagged. A total of 73,926 fish were tagged and of that 12,449 red drum were tagged and 1,485 were recaptured and 28,423 black drum tagged and 850 recaptured. Of the fish that were recaptured the majority moved ≤10 km from the tagging locations. The average period of time at liberty was ≤180 days. Fish in Lavaca Bay showed the same pattern of restricted movement as fish in other Texas Bays.

Heffernan, T.L. 1973. Survey of adult red drum (Sciaenops ocellatus). Texas Parks Wildl. Dep., Coastal Fish Proj. Rep.: 37-66.

This particular reference is not very useful, but it does strengthen the observations that red drum spawn in the fall through early winter in the coastal waters around passes on the Texas coast. The reference is concerned primarily with catch per unit effort statistics for different types of gear.

Hoese, H.D., D.W. Beckman, R.H. Blanchet, D. Drullinger and D.L. Nieland. In Press. A biological and fisheries profile of Louisiana red drum, Sciaenops ocellatus. Louisiana Department of Wildlife and Fisheries, Fishery Management Plan Series, No. 4, Part 1.

An overview of red drum life history and biology for the stock in Louisiana waters. Female fish reach sexual maturity at 3-7 years and the males at 1-3 years. The fish are opportunistic feeders with the larger fish feeding primarily on blue crabs, penaeid shrimp, menhaden and anchovy. Red drum are estuarine dependent and the young inhabit a wide range of salinities from low to high, while the older individuals occur along the beaches and the offshore Gulf continental shelf.

Holt, J., C.L. Kitting and C.R. Arnold. 1983. Distribution of young red drum among different seagrass meadows. Trans. Am. Fish. Soc. 112: 267-271.

Seagrass meadows are primary habitats for young red drum Sciaenops ocellatus in south-Texas estuaries. The abundance of small red drum (6-27 mm, SL) in different meadows averaged 0.1 - 0.8/m^2. Small red drum were not found on large (>5 m across) nonvegetated sites; however, the ecotone between sea grass and nonvegetated bottom had significantly more red drum than did homogeneously vegetated sites. Heterogeneous sea-grass meadows, therefore, may support more young red drums than homogeneous ones.

Lee, W.I., G.J. Holt and C.R. Arnold. 1984. Growth of red drum larvae in the laboratory. Trans. Am. Fish. Soc. 113: 243-246.

Weight decreased through yolk absorption and then increased rapidly after feeding began. There were significant differences in the rate of fish growth at temperatures of 24 and 28C. Weight were 186.3 ug vs. 363.7 ug respectively.

Linton, T.L., J.H. Clark and J.M. Boslet. 1990. Annotated bibliography of the red drum (Sciaenops ocellata). In: C.R. Arnold, G.J. Holt, and P. Thomas (eds.) Red Drum Aquaculture. 214-225 pp.

An annotated bibliography of red drum that comprise selected references concerned with the life history, distribution, and aquaculture of the species.

Matlock, G.C. 1987. Maximum and total length and age of red drum off Texas. Northeast Gulf Sci. 9(1): 49-52.

Red drum (8,519) were captured in trammel nets in Texas bays from October 1976-1980. The fish were measured and weighed and some were tagged and released. The average maximum length of red drum for Texas bays was typically 755 mm. Of the fish

captured between 1976-1980 99.5% of the fish were less than 755 mm total length.

Matlock, G.C. 1992. Growth of five fishes in Texas bays in the 1960s. Fish. Bull. 90: 407-411.

These data are of fish (i.e. black drum, red drum, sheepshead, Southern flounder, and spotted seatrout) caught, tagged and recaptured at different locations along the Texas coast in the period between 1950-1975. The average daily growth rates for the five species were: black drum, 0.187 mm; red drum, 0.133 to 0.395 mm; sheepshead, 0.167 mm; southern flounder, 0.233 mm; and spotted seatrout, 0.171 mm.

Mercer, L.P. 1984. Fishery management plan for the red drum (Sciaenops ocellatus) fishery. Fishery Management Report No. 5 of the Atlantic States Marine Fisheries Commission. 107 pp.

This report is a thorough discussion of the life history, distribution, and fisheries information for red drum. The data used to compile the report contains many references to the Gulf of Mexico and Texas with respect to this species of fish even though the emphasis is on North Carolina and the Atlantic coast.

Miles, D.W. 1949. Study of the food habits of the fish of the Aransas Bay area. Texas Game, Fish, and Oyster Comm., Mar. Lab. Ann. Rep. 1948-1949: 126-169.

Redfish stomachs were examined over a nine month period. A total of 1,549 stomachs were analyzed. The majority of the food items that the fish consumed, that were identifiable to species or unidentifiable, were crustaceans and fish. The most prominent crustaceans were shrimps. Small fish did comprise a major portion of the diet.

Miles, D.W. 1950. The life histories of the spotted seatrout Cynoscion nebulosus and the redfish Sciaenops ocellata. Texas Game Fish Oyster Comm., Mar. Lab. Ann. Rep. (1949-1950): 66-103.

Redfish spawn near the passes on the Texas coast in the fall and the juveniles show up inside the bays in October and November. During the first year the redfish grow to about 350 mm, in the second year to about 550 mm, and in the third year to about 650. Fish under 700 mm were not sexually mature. The food habits of redfish in this study indicate that they are particularly partial to grass shrimp, penaeids, and crabs, primarily blue crabs.

Monaco, M.E., T.E. Czapla, D.M. Nelson and M.E. Pattillo. 1989. Distribution of fishes and invertebrates in Texas estuaries. In: NOAA's Estuarine Living Marine Resources Project. 107 pp. NOAA, Rockville, MD.

The publication reviews the distribution and abundance data for 40 species of marine invertebrates and vertebrates from the area of the Texas coast. Three detailed species profiles are presented for the bay anchovy, brown shrimp, and red drum.

Osburn, H.R., G.C. Matlock and A.W. Green. 1982. Red drum (Sciaenops ocellatus) movement in Texas bays. Contrib. Mar. Sci. 25: 85-97.

Red drum movement in Texas bays was studied by means of fish tagged with internal abdominal anchor tags and subsequently released. Interbay movement was minimal with the majority (>71%) of the returned tagged red drum moving ≤10 km.

PERCENT OF RETURNED TAGGED RED DRUM IN RELATION TO DISTANCE TRAVELED FROM SITE OF TAGGING DURING 11/75 - 9/78

BAY	#	Minimum Distance Traveled (km) 0-5	6-10	11-15	16-20	21-25	>26
Matagorda	41	58.5	14.6	9.8	7.3	0	9.8
Galveston	237	66.7	12.2	5.9	3.0	3.8	8.5
Corpus Christi	741	63.3	12.4	7.7	5.3	3.0	8.4

There was no significant difference between the minimum distance moved and season or size of fish. Texas bay systems can be considered closed systems for fish ~305-625 mm TL when managing red drum stocks. Due to the restricted movement patterns, local populations of red drum could be vulnerable to intense fishing pressure. [EDITORIAL NOTE: This also could mean that fish populations in bays with point sources of contaminants would reflect the contaminants that were introduced.]

Overstreet, R.M. and R.W. Heard. 1978. Food of the red drum, Sciaenops ocellata, from Mississippi Sound. Gulf Res. Rep. 6(2): 131-135.

The food habits of red drum from Mississippi Sound were studied by examining the stomach contents of 107 fish. The digestive tracts of these fish contained primarily crustaceans, fish, and polychaetes (i.e. occurrences of 99%, 43%, and 15% respectively). Of the crustaceans consumed a

large majority were portunid and grapsid crabs, followed by penaeid and palaemoneid shrimps. These results agree with studies done in other locations along the Gulf coast.

Pafford, J.M. 1981. Seasonal movement and migration of red drum (Sciaenops ocellatus) in Georgia's coastal waters. Estuaries 4:279-280.

Seasonal and/or migratory patterns of red drum were studied using tag and recapture techniques in Georgia coastal waters. Over 200 red drum were tagged in the Altamaha, St. Simons and St. Andrew estuarine systems and at reporting time 30% of the tags had been recovered. The recovery data indicated that fish <7kg were dispersed randomly throughout the estuary and fish >7kg were found primarily on the beaches and shoals during the warmer months and then offshore during the colder months. Most of the recovered fish remained in the tagging area, 10% were captured outside the area. Those fish showed a definite northerly movement with a maximum distance traveled of 161 km and an average of 14.5 km.

Perret, W.S., J.E. Weaver, R.O. Williams, P.L. Johansen, T.D. McIlawain, R.C. Raulenson and W.M. Tatum. 1980. Fishery profiles of red drum and spotted seatrout. Gulf States Mar. Fish. Comm. No. 6, Ocean Springs, MS, 60 pp.

The data presented in the profile of red drum give a broad overview of the life history, biology, growth, and migratory patterns of the fish in the Gulf of Mexico.

Peters, K.M. and R.H. McMichael, Jr. 1987. Early life history of the red drum, Sciaenops ocellatus (Pisces: Sciaenidae), in Tampa Bay, Florida. Estuaries 10: 92-107.

The red drum recruit into Tampa Bay as larvae and remain there through their juvenile growth period. The food habits of the juveniles change as they increase in size. The predominant foods: 45% of 8-15 mm fish ate mysids [56% by volume and 20% by number]; in 75-90 mm fish, shrimp was 56% of the volume; and the larger juveniles, 90-105 mm, ate crabs and small fish. The fish grew to a total length of >300 mm in the first year.

Reagan, R.E. 1985. Species profiles: Life histories and environmental requirements of coastal fishes and invertebrates (Gulf of Mexico) Red Drum. U. S. Fish Wildl. Serv., Biol. Rep. 82(11.36), TR EL-82-4, 16 p.

A synopsis of life history characteristics, length-weight relationships for fish from Louisiana and Texas, fishery catch statistics, and environmental requirements.

Ross, J.L., J.S. Pavella and M.E. Chittenden, Jr. 1983. Seasonal occurrence of black drum, Pogonias cromis, and red drum, Sciaenops ocellatus, off Texas. Northeast Gulf Sci. 6: 67-70.

Simmons, E.G. and J.P. Breuer. 1962. A study of redfish, Sciaenops ocellata Linnaeus and black drum, Pogonias cromis Linnaeus. Publ. Inst. Mar. Sci. Univ. Texas 8: 184-211.

Red drum spawn offshore in the Gulf of Mexico during the fall and winter. Larvae/juveniles enter the bays and nursery grounds where they remain for six month to three or four years. They grow rapidly during the first three years 320, 530, and 700 mm SL. The fish begin spawning at 700-800 mm. The movements of redfish within the bays is rather restricted during the first three years of growth and interbay movement is relatively rare. The fish prefer salinities of 20-40% seawater and temperatures of from 3-33°C. Their preferred food is crabs, but they do feed on small fish and shrimp.

Topp, R. 1963. The tagging of fishes of Florida, 1962 program. Fla. Board Conserv. Mar. Res. Lab. Prof. Pap. Ser. 5. 76pp.

Wilder, W.R. 1986. An analysis of Texas Gulf coast red drum (Sciaenops ocellata): Identification of possible stocks and implications for fisheries management. Ph.D. dissertation, Rice University, Houston, Texas, 124p.

An electrophoretic analysis of Texas coastal populations of red drum (Sciaenops ocellata) was made on fish collected from embayments; Galveston to Lower Laguna Madre. A total of 30 genetic loci were screened and population divergence calculated statistically. The result of the investigations shows significant differences between the different embayments. While juveniles are geographically isolated, adults occupy the open waters of the Gulf where they form a single reproductively active stock.

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BLACK DRUM - Pogonias cromis

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Beckman, D.W., A.L. Stanley, J.H. Render and C.A. Wilson. 1990. Age and growth of black drum in Louisiana waters of the Gulf of Mexico. Trans. Am. Fish. Soc. 119(3): 537-544.

Morphometric measurements and otoliths (sagittae) were collected from black drum Pogonias cromis caught in estuarine and coastal waters off LA. One annulus was formed per year during winter and early spring months in all age classes during each of four years of sampling. Von Bertalanffy growth models did not describe growth well for all age-classes,

perhaps due to an extreme decrease in growth rate at about four years of age and continuous, nonasymptotic growth thereafter.

Cave, R.N. and E.W. Cake, Jr. 1980. Observations on the predation of oysters by black drum Pogonias cromis (Linnaeus) (Sciaenidae). Proc. Natl. Shellfish Assoc. 70(1):121.

Laboratory studies have shown that black drum are capable of crushing and consuming any oyster that will fit into their pharyngeal apparatus. The tensile strength of the shell controls whether the drum will crush or reject the oyster.

Cody, T.J., K.W. Rice and C.E. Bryan. 1985. Distribution and gonadal development of black drum in Texas Gulf waters. Texas Parks Wildl. Dep., Coastal Fish. Branch, Manage. Data Ser.No. 72. 16pp.

Cornelius, S.E. 1984. Contribution to the life history of black drum and analysis of the commercial fishery of Baffin Bay, Vol. II. Tech. Rep. 6, Caesar Kleberg Wildl. Res. Inst. Kingsville, TX, 241 p.

Doerzbacher, J.F., A.W. Green, G.C. Matlock and H.R. Osburn. 1988. A temperature compensated von Bertalanffy growth model for tagged red drum and black drum in Texas bays. Fish. Res. 6(2): 135-152.

Black drum were tagged and recaptured in Texas bays from 11/75-6/85 and estimates of the von Bertalanffy growth model were calculated. An annual temperature model was used to examine the growth model when periods of cold temperature were excluded. Black drum were best fitted with time at large in days with the coldest 120 days excluded and the yielded VBGM parameters of K = 0.219 and L sub(infinity) = 798 mm.

Green, L. 1986. Fish tagging on the Texas coast, 1950-1975. Manage. Data Ser. 99, Tex. Parks Wildl. Dep., Coastal Fish. Br., Austin, 206 p.

The black drum tagged in the Matagorda Bay system showed very little movement from the location of tagging. About 54% of the fish tagged moved <10 km from the site of tagging.

Hoese, H.D. and R.H. Moore. 1977. Fishes of the Gulf of Mexico. Texas A&M University Press, College Station. 327 pp.

A key to the fishes of the Gulf of Mexico which gives useful information on the life histories of a large number of species of fish.

Matlock, G.C. 1992. Growth of five fishes in Texas bays in the 1960s. U.S. Natl. Mar. Fish. Serv., Fish. Bull. 90: 407-411.

SEE: Same reference under red drum.

Matlock, G.C. 1990. Maximum total length and age of black drum, Pogonias cromis (Osteichthyes: Sciaenidae) off Texas. Northeast Gulf Sci. 11(2): 171-174.

Largest black drum in Texas bays are typically at least 755 mm TL as determined from 7,244 fish captured in trammel nets and this estimate agree well with the calculated value of 798±42 mm. Among the 4,329 fish caught by sport anglers, however, 99.5% of the fish were less than 1,015 mm which suggests that the value calculated is an underestimate. The largest fish tend to be caught on the Atlantic coast in the colder waters north of Cape Hatteras.

Miles, D.W. 1949. A study of the food habits of the fishes of the Aransas Bay area. Texas Game, Fish, and Oyster Comm., Marine Lab Ann. Rep. (1948-1949): 126-169.

As part of the study of food habits of fish, 288 black drum stomachs were examined. The black drum is anatomically adapted to feed on molluscan shellfish, and the study verified the fact that molluscs of a number of different species comprise a major portion of the diet. Crustaceans and small fish also were found in the stomachs.

Minello, T.J., R.J. Zimmerman and T.E. Czapla. 1989. Habitat related differences in diets of small fishs in Lavaca Bay, Texas, 1985-1986. NOAA Tech. Memo. SEFC-NMFS-236, 16 p.

A study was conducted of the stomach contents of nine species of small fish in Lavaca Bay. The primary shrimp predators were the southern flounder, spotted seatrout, and the sand seatrout. Also the quantity of food eaten by fish from the coastal areas in the Bay was larger than in fish from the delta area.

Murphy, M.D. and R.G. Taylor. 1989. Reproduction and growth of black drum, Pogonias cromis, in northeast Florida. Northeast Gulf Sci. 10: 127-137.

Maturation of black drum in northeast Florida occurred for males at ages 4-6 years and 590-679 mm SL and at 5-6 years and 650-699 mm SL for females, but in Texas "granular gonads" suggesting maturity occurred at 275-320 mm at year 2. Growth rates for drum were 100 mm/yr, for ages 1-3 and gradually slowed to 10-30 mm/yr for ages 15-20. Texas fish were similar and there was no sex difference. Apparent maximum age is 50-60 years.

Osburn, H.R. and G.C. Matlock. 1984. Black drum movement in Texas bays. N. Am. J. Fish. Manage. 4: 523-530.

Tagging experiments between 11/75 and 8/80 showed that overall 44% of the black drum tagged moved less than 10 km, and of those that left the bay 75% were captured in an adjacent bay. In east Matagorda and Matagorda Bays 53% & 41% moved 0-5 km and 20% & 18% moved 6-10 km, which means that fish in these embayments moved less than the overall group tested. No mass migrations of fish in the winter or during the spring spawning period were noted.

Ross, J.L., J.S. Pavella and M.E. Chittenden, Jr. 1983. Seasonal occurrence of black drum, Pogonias cromis, and red drum, Sciaenops ocellatus, off Texas. Northeast Gulf Sci. 6: 67-70.

Simmons, E.G. and J.P. Breuer. 1962. A study of redfish, Sciaenops ocellata Linnaeus and black drum, Pogonias cromis linnaeus. Publ. Inst. Mar. Sci. Univ. Texas 8: 184-211.

Black drum, unlike the red drum, spawn in coastal embayments in the Gulf and around passes over different bottom types. Primary spawning occurs in February or March and there is a secondary spawning in May or June. Black drum has a slower growth rate than red drum for the first three years (160, 310, and 415 mm SL), and then grows at a rate of 50 mm/yr. as determined from tagging studies. They are euryhaline and prefer salinities of 0-80% seawater. Their feeding activity consists of grubbing in the sediment, and their primary food items are small molluscs, but they do consume vegetation, small fish, polychaetes, and shrimp.

Sutter, F.C., R.S. Waller and T.D. McLlwain. 1986. Species profiles: Life histories and environmental requirements of coastal fishes and invertebrates (Gulf of Mexico) Black Drum. Biol. Rep. U.S. Fish Wildl. Serv., 22 pp.

A synopsis of the life history characteristics, ecological role, and environmental requirements of the black drum in the western Gulf of Mexico. The migratory patterns of the black drum are very limited and inter-bay movements are slight (i.e. 60% of tagged adult black drum collected in given locale were tagged in the same area).

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ESTUARINE INVERTEBRATES

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Alexander, S.K. 1986. Diet of the blue crab, Callinectes sapidus Rathbun, from nearshore habitats of Galveston Island, Texas. Texas J. Sci. 38: 85-89.

In this study the larger crabs (carapace width >60 mm) utilized, in order of importance, molluscs, fish, and crustaceans. Due to their feeding habits the blue crab can be considered a detritivore, omnivore, and primary carnivore.

Bisker, R. and M. Castagna 1987. Predation on single spat oysters, Crassostrea virginica Gmelin, by blue crabs, Callinectes sapidus Rathbun, and mud crabs, Panopeus herbstii Milne-Edwards. J. Shellfish Res. 6: 37-40.

Blue crabs from 9.3 to 85.5 mm carapace width caused significant mortalities of oyster spat in controlled laboratory experiments.

Brouwer, M., D.W. Engel, J. Bonaventura and G.A. Johnson. 1992. In Vivo magnetic resonance imaging of the blue crab, Callinectes sapidus: effect of cadmium accumulation in tissues on proton relaxation properties. J. Exp. Zool. 261: 32-40.

MRI imaging of live blue crabs that have been dosed with cadmium prior to examination. An outgrowth of the study was the observation that following molting the concentrations of cadmium in the digestive gland of crabs decreased about five to ten fold. This suggests that cadmium in the digestive gland is excreted along with copper and zinc at ecdysis.

Eggleston, D.B. 1990. Foraging behavior of the blue crab, Callinectes sapidus, on juvenile oysters, Crassostrea virginica: effects of prey density and size. Bull. Mar. Sci. 46: 62-83.

Blue crabs have been shown to be major predators on juvenile oysters in Chesapeake Bay. The selection of the prey is size dependent and the success is dictated by the density of the oysters.

Engel, D.W. 1977. Comparison of the osmoregulatory capabilities of two portunid crabs, Callinectes sapidus and C. similis. Mar. Biol. 41: 275-279.

This investigation was designed to determine the similarities and differences that exist between the two closely related species of the genus Callinectes. The data show that C. similis does not osmoregulate as well as C. sapidus even though the ranges of the two species overlap extensively.

Engel, D.W. 1983. The intracellular partitioning of trace metals in marine shellfish, p. 129-140. In R.E. Wildung and E.A. Jenne (ed.), Biological Availability of Trace Metals. Elsevier, Amsterdam.

Discussion of experiments dealing with the partitioning and accumulation of metals by blue crabs and oysters. The data on blue crabs shows that the accumulation of cadmium is more efficient through food than via water. The intracellular partitioning of the metal is identical.

Engel, D.W. 1987. Metal regulation and molting in the blue crab, Callinectes sapidus: copper, zinc and metallothionein. Biol. Bull. 172: 69-82.

This is the first demonstration that the metal metabolism of the blue crab is linked directly with the molt cycle of the crab. At molt it was shown that the crab lost about 60% of the hemolymph hemocyanin concentration and an appreciable amount of the digestive gland copper and zinc.

Engel, D.W. and M. Brouwer. 1987. Metal regulation and molting in the blue crab, Callinectes sapidus: metallothionein function in metal metabolism. Biol. Bull. 173: 239-251.

The changes in trace metals as a function of the molt cycle was further explored and the decreases shown to be correlated with decreases in hemocyanin. Evidence was also presented that metallothionein could act as a donor of copper in the resynthesis of hemocyanin.

Engel, D.W. and M. Brouwer. 1991. Short-term metallothionein and copper changes in blue crabs at ecdysis. Biol. Bull. 180: 447-452.

The time course of changes in trace metal concentrations in the digestive gland is very rapid. There are significant changes in hemocyanin and in copper and zinc in the digestive gland within 60 minutes of ecdysis. A descriptive model of metallothionein and copper/zinc turnover was constructed to explain the changes that occurred at ecdysis.

Galtsoff, P.S. 1964. The American oyster, Crassostrea virginica Gmelin. U.S. Fish Wildl. Serv. Fish. Bull. 64: 1-480.

This is the classic treatise on the American oyster. The information contained in this volume includes detailed descriptions of the life history, larval and juvenile development, food habits, and culturing techniques.

Gleason, D.F. 1986. Utilization of salt marsh plants by postlarval brown shrimp: carbon assimilation rates and food preferences. Mar. Ecol. Prog. Ser. 31: 151-158.

Postlarval brown shrimp raised on different types of plant material showed differences in growth. Combinations of carbon sources gave the best growth rather than a single source. The study did show, however, that certain organisms common to

Spartina salt marshes such as Skeletonema costatum are important to the growth of shrimp.

Gleason, D.F. and R.J. Zimmerman. 1984. Herbivory potential of postlarval brown shrimp associated with salt marshes. J. Exp. Mar. Biol. Ecol. 84: 235-246.

The results of this laboratory study indicate that certain planktonic diatoms and the epiphytes on Spartina in salt marshes are nutritionally necessary for the survival and growth of postlarval brown shrimp.

Laughlin, R.A. 1982. Feeding habits of the blue crab, Callinectes sapidus Rathbun, in Apalachicola Estuary, Florida. Bull. Mar. Sci. 32: 807-822.

A one year study showed that while blue crabs are detritivores and omnivores, they had significant shifts in diet with season of the year, but there were no diel differences. A summary of all crabs examined (N=3,200) indicates a preference for bivalves 35.6%. Both fish and xanthid crabs were also valued food items.

Mangum, C. 1992. Physiological aspects of molting in the blue crab Callinectes sapidus. Am. Zool. 32: 459-469.

This paper is a collection of recent research findings on the physiological changes in blue crabs during the molt cycle.

McTigue, T.A. and R.J. Zimmerman. 1991. Carnivory vs. herbivory in juvenile Penaeus setiferus (Linnaeus) and Penaeus aztecus (Ives). J. Exp. Mar. Biol. Ecol. 151: 1-16.

Brown and white shrimp were raised under laboratory conditions for 24 days, and fed a diet that consisted of either vegetal or animal material or a combination of the two. The two species did not do well and did not survive through the 24 days of the experiment on a diet of either animal or vegetal material. Both species grew very well on a diet that consisted of a combination of the animal and vegetal material. This indicates that shrimp require a diverse dietary source.

Messick, G.A. and C.J. Sindermann. 1992. Synopsis of principal diseases of the blue crab, Callinectes sapidus. NOAA Tech. Memo. NMFS-F/NEC-88, 24 p.

An overview of the diseases of blue crabs throughout their range from Delaware Bay to Texas. Gives descriptions of the principal diseases that affect blue crabs which will be useful in determining if chemical contaminants are causing observable pathologies.

Millikin, M.R. and A.B. Williams. 1984. Synopsis of biological data on the blue crab, Callinectes sapidus Rathbun. NOAA Tech. Rep. NMFS 1 / FAO Fish. Synop. 138, 39p.

This publication is an excellent overview of the life history of the blue crab and includes detailed descriptions of growth, reproduction, feeding, etc. The literature citations are also very complete.

Minello, T.J. and R.J. Zimmerman. 1983. Fish predation on juvenile brown shrimp, Penaeus aztecus Ives: The effect of simulated Spartina structure on predation rates. J. Exp. Mar. Biol. Ecol. 72: 211-231.

Laboratory experiments were conducted to determine the effects of artificial Spartina structure on the predation efficiencies on juvenile brown shrimp, Penaeus aztecus, by four species of fish: pinfish, Atlantic croaker, red drum and speckled trout. The structures reduced the capture efficiencies of croaker and pinfish, but did not affect the red drum or speckled trout. This result would make sense since trout and drum use seagrass beds as nursery areas.

Minello, T.J., R.J. Zimmerman and E.X. Martinez. 1989. Mortality of young brown shrimp Penaeus aztecus in estuarine nurseries. Trans. Am. Fish. Soc. 118: 693-708.

A study of the predation of brown shrimp by estuarine fishes. The primary fish predator on juvenile and larval brown shrimp is the southern flounder, but other fish such as the red drum are also effective predators.

Orth, R.J. and J. van Montfrans. 1990. Utilization of marsh and seagrass habitats by early stages of Callinectes sapidus: a latitudinal perspective. Bull. Mar. Sci. 46: 126-144.

The relationship between vegetated (total marsh and seagrass) and landings within the Gulf region emphasizes the importance of this habitat type for blue crabs. Reticulated marshes of Texas and Louisiana resulting from factors such as tidal amplitude and subsidence causes longer periods of inundation which may be a prime factor in the importance of these marshes to megalopa in the Gulf area. The abundance of seagrass and marsh habitat on the Gulf coast would suggest that the blue crab population size should be large, but the landings do not reflect that hypothesis.

Thomas, J.L., R.J. Zimmerman and T.J. Minello. 1990. Abundance patterns of juvenile blue crabs (Callinectes sapidus) in nursery habitats of two Texas bays. Bull. Mar. Sci. 46: 115-125.

The lowest abundances were seen in the early spring among the overwintering crabs and the highest abundances were seen in the late summer and fall corresponding to seasonal recruitment. Crabs from marsh habitat were larger than crabs from seagrass meadows or unvegetated habitat. The two bays studied were Christmas and West Bays near Galveston.

Van Den Avyle, M.J. and D.L. Fowler. 1984. Species profiles: Life histories and environmental requirements of coastal fishes and invertebrates (South Atlantic). Blue Crab. U.S. Fish. Wildl. Serv. FWS/OBS-82/11.19. 16 p.

Blue crabs occur in the estuarine systems and coastal waters of the Gulf of Mexico and the Atlantic seaboard. Their growth is strongly influenced by salinity and temperature, and the males continue to grow throughout their life and the female stops growing at her maturity molt. The eggs and early larval stages require high salinity, >20 ppt., while the megalops and juvenile stages are euryhaline. Crabs generally mature at 1-2 yr of age. The blue crab is an opportunistic and omnivorous feeder and utilizes all available sources. The blue crab is also a favored food for many important estuarine fishes, including red drum.

Williams, A.B. 1965. Marine decapod crustaceans of the Carolinas. U.S. Fish Wildl. Serv. Fish. Bull. 65: 1-298.

A complete treatment of marine decapod crustaceans of the Carolinas which includes the target species in the Gulf of Mexico: brown shrimp, white shrimp, and blue crab.

Williams, A.B. 1974. The swimming crabs of the genus CALLINECTES (DECAPODA: PORTUNIDAE). U.S. Natl. Mar. Fish. Serv. Fish. Bull. 72: 685-798.

An excellent treatise on the genus Callinectes which of course includes the blue crabs. The data provided in this publication covers the distribution and speciation of the crabs of this genus. For example, along the Texas coast there are two species, C. similis and C. sapidus.

Williams, C.D., D.M. Nelson, M.E. Monaco, S.L. Stone, C. Iancu, L. Coston-Clements, L.R. Settle and E.A. Irlandi. 1990. Distribution and abundance of fishes and invertebrates in eastern Gulf of Mexico estuaries. NOAA ELMR Report #6, 105 p.

A listing of the distribution and abundance of fishes and invertebrates in the Gulf of Mexico which includes a detailed description of the life history and biology of the blue crab. It gives a good overview of the blue crab.

Zimmerman, R.J., T.J. Minello, D.L. Smith and J. Kostera. 1990. The use of Juncus and Spartina marshes by fisheries species in Lavaca Bay, Texas, with reference to effects of floods. NOAA Tech. Memo. NMFS-SEFC-251, 40 p.

The usage of deltic and coastal marshes within Lavaca Bay was similar. The marshes were used primarily by penaeid shrimp, blue crabs, and economically important fishes (i.e. red drum, spotted seatrout, and southern flounder). Flooding did not have any appreciable effect on the marshes even though the salinity in the 1987 flood went to 0 ppt. for two weeks.

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MERCURY BIOACCUMULATION MODELS

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Aoyama, I., Y. Inoue and Y. Inoue. 1978. Simulation analysis of the concentration of trace heavy metals by aquatic organisms from the viewpoint of nutrition ecology. Water Res. 12:837-842.

Simulation model of prey to predator transfers of metals. Emphasizes feeding rate and food availability as determining parameters in metal accumulation. Requires estimates of assimilation efficiencies and assumes a logistic model of predator growth. Can predict growth dilution of metal concentrations during trophic transfers. Theoretical approach.

Barber, M.C., L.A. Suarez and R.R. Lassiter. 1991. Modelling bioaccumulation of organic pollutants in fish with an application to PCBs in Lake Ontario salmonids. Can. J. Fish. Aquat. Sci. 48:318-337.

A comprehensive single trophic step contaminant transfer model exemplified by freshwater fish species. Used to predict bioaccumulation of lipid soluble organic contaminants, but could be generalized to other contaminants such as methyl mercury. Includes uptake from both water (across gills) and food (across intestinal mucosa, emphasizing equilibrium partitioning. Among the many required model input parameters are feeding, metabolic and growth rates of fish, assimilation efficiency of food, octanol/water partition coefficients of contaminants, gill morphology, contaminant diffusion rates, lipid content of fish. Model outputs are time dependent, thus allowing response to changing exposure concentration. Model predicts size/age dependent differences in contaminant concentrations in fish.

Borgmann, U. and D.M. Whittle. 1992. Bioenergetics and PCB,DDT, and mercury dynamics in Lake Ontario lake trout (Salvelinus namaycush): A model based on surveillance data. Can. J. Fish. Aquat. Sci. 49:1086-1096.

Considers two related bioenergetic models of contaminant uptake by fish. Both models ignore contaminant uptake from water, and differ in how they treat the dynamics of digestion. Necessary model parameters include fish growth rate, ingestion rate, contaminant assimilation efficiency and excretion rate, and lipid content of food and fish tissue. Mercury (methylmercury) is explicitly considered, and estimated parameters are provided. Feeding on multiple prey species with different contaminant concentrations is included in the model. Time dependency and size/age dependency of contaminant concentrations in fish are predicted.

Braun, B.M. 1987. Mercury accumulation in relation to size and age of Atlantic herring (Clupea harengus harengus) from the southwestern Bay of Fundy, Canada. Arch. Environ. Contam. Toxicol. 16(3):311-320.

A bioenergetics-based model is applied to observed mercury concentrations in free living herring. Increases in mercury concentrations with fish age/size are predicted within one standard deviation of measured values.

Connolly, J.P. and R. Tonelli. 1985. Modelling kepone in the striped bass food chain of the James River estuary. Est. Coastal Shelf Sci. 20:349-366.

Mathematical model of multiple trophic step transfer of an organic contaminant to a top predator estuarine fish species. Both contaminated water and sediments are considered as ultimate sources to both the top predator and intermediate prey species. Requires feeding, respiration, and growth rate parameter estimates for all species in the food chain, as well as assimilation efficiencies and excretion rate constants for the contaminant. The model is calibrated with real field data and is used to make predictions of future concentrations of the contaminant. The model could easily be applied to methylmercury bioaccumulation with appropriate parameterization.

Curtis, E.H., J.J. Beauchamp and B.G. Blaylock. 1977. Application of various mathematical models to data from the uptake of methylmercury in bluegill sunfish (Lepomis macrochirus). Ecol. Model. 3:273-284.

Mathematical model of methylmercury uptake from water only. Emphasizes different models of internal partitioning and excretion in the fish. The different models were fitted to laboratory uptake data.

Fagerstrom, T. and A. Jernelov. 1972. Some aspects of the quantitative ecology of mercury. Wat. Res. 6:1193-1202.

Fagerstrom, T. and B. Asell. 1973. Methyl mercury accumulation in an aquatic food chain. A model and implications for research planning. Ambio 2:164-171.

Fagerstrom, T. 1974. Model for accumulation of methyl mercury in northern pike, Esox lucius. Oikos 25(1):14-20.

Fordham, C.L. and D.P. Reagan. 1991. Pathways analysis method for estimating water and sediment criteria at hazardous waste sites. Environ. Toxicol. Chem. 10:949-960.

An equilibrium model of contaminant bioaccumulation in which a top predator is identified as the critical contaminant receptor. From criteria selected to protect this receptor, a reverse pathway leading back to the contaminant source is developed. The model parameterization leads to predictions of allowable contaminant concentrations in the sources (water and sediment) which would conservatively protect the top predator by maintaining its contaminant concentration below the protection criterion. The model requires estimates of food habits, food chain structure, feeding rates, contaminant assimilation efficiency and excretion rates, and bioconcentration factors from water. Dieldrin accumulation in eagles is used to exemplify the modelling approach, but it could be applied to other contaminants including mercury.

Gramme, P.E., G.Norheim, B.Boe, B.Underdal and O.C. Bockman. 1984. Detection of cod (Gadus morhua) subpopulations by chemical and statistical analysis of pollutants. Arch. Environ. Contam. Toxicol. 13(4):433-440.

Utilizes a statistical model of mercury (and octachlorostyrene) concentrations in fish to distinguish different subpopulations subject to differences in mercury exposure. Adjusts for differences in age/size/sex of fish and considers diet differences and time of exposure.

Hartung, R. 1976. Pharmicokinetic approaches to the evaluation of methylmercury in fish. In R.W. Andrews, P.V. Hodson and D.E. Konasewich, (Eds.) Toxicity to biota of metal forms in water. Great Lakes Research Advisory Board Stand. Comm. Res. Adv. Bd. Windsor Ontario, pp. 233-248.

Herrick, C.J., E.D. Goodman, C.A. Guthrie, R.H. Blythe, G.A. Hendrix, R.L. Smith, and J.E. Galloway. 1982. A model of mercury contamination in a woodland stream. Ecol. Model. 15:1-26.

Niimi, A.J. 1983. Physiological effects of contaminant dynamics in fish. In J.O. Nriagu (ed.) Aquatic Toxicology. Wiley, New York, pp. 206-246.

Reviews the basis for the bioenergetic modelling approach to contaminant accumulation in fish and provides parameter estimates for accumulation efficiency and excretion kinetics for some contaminants, including mercury.

Norstrom, R.J., A.E. McKinnon and A.S.W. deFreitas. 1976. A bioenergetics-based model for pollutant accumulation by fish, simulation of PCB and methylmercury residue levels in Ottawa River yellow perch (Perca flavescens). J. Fish. Res. Board Can. 33:248-267.

One of the earliest papers coupling fish bioenergetics and contaminant biokinetics in a model to predict contaminant concentrations in fish. The model is applied to methylmercury. It is time dependent and includes seasonal and growth dependent terms. It is a single step contaminant transfer model including uptake from water and food by fish.

Preston, A. and J.E. Portmann. 1981. Critical path analysis applied to the control of mercury inputs to United Kingdom coastal waters. Environ. Pollut. (series B) 2:451-464.

An empirical model that relates mercury inputs into coastal waters to mercury concentrations in fish identified as critical target organisms. The model is then used to predict the degree of mercury input reduction necessary to protect the critical resource by limiting mercury concentrations to some predetermined level.

Ribeyre, F. 1985. Problems and methodologies in ecotoxicology: biological models and experimental plans. Ecotoxicol. Environ. Safety 9(3):346-363.

Design principles in the laboratory measurement of mercury transfers in aquatic food chains.

Ribeyre, F., A. Delarche and A. Boudou. 1980. Transfer of methylmercury in an experimental freshwater trophic chain-temperature effects. Environ. Pollut. (series B) 1(4):259-268.

A laboratory experimental system is used develop an empirical model of methylmercury transfer along a four step food chain. Transfer rate parameters are determined.

Roberts, J.R., A.S.W. deFreitas and M.A.J. Gidney. 1979. Control factors on uptake and clearance of xenobiotic chemicals by fish. Animals as monitors of environmental pollutants. Symposium on Pathobiology of Environmental Pollutants: Animal Models and Wildlife as Monitors. Storrs, Conn., pp. 3-14.

A bioenergetics model of contaminant (including methylmercury) uptake from both food and water by fish. Assimilation efficiency and excretion rate data are required as well as bioenergetic parameterization for the fish.

Rodgers, D.W. and F.W.H. Beamish. 1981. Uptake of waterborne methylmercury by rainbow trout (Salmo gairdneri) in relation to oxygen consumption and methylmercury concentration. Can. J. Fish. Aquat. Sci. 38:1309-1315.

A single step model of mercury uptake from water to a freshwater fish in relation to metabolism. Model is applied to laboratory measured uptake of methylmercury.

Rodgers, D.W. and S.U. Qadri. 1982. Growth and mercury accumulation in yearling yellow perch, Perca flavescens, in the Ottawa River, Ontario. Environ. Biol. Fish. 7(4):377-383.

Application of the bioenergetic model of Nordstrom et al. (1976) to field observations of methylmercury accumulation in a freshwater fish.

Terhaar, C.J., W.S. Ewell, S.P. Dziuba, W.W. White and P.J. Murphy. 1977. A laboratory model for evaluating the behavior of heavy metals in an aquatic environment. Water Res. 11:101-110.

A physical model system (microcosm) to measure food chain bioaccumulation of metals including mercury in the laboratory.

Thomann, R.V. 1981. Equilibrium model of fate of microcontaminants in aquatic food chains. Can. J. Fish. Aquat. Sci. 38:280-296.

The use of literature derived bioconcentration (from water) factors and bioaccumulation (from food) factors to predict contaminant concentrations in components of aquatic food chains. Contaminant assimilation efficiencies and excretion rates are used along with feeding rate and growth rates in organisms used in developing model predictions . The contaminants considered do not include mercury, but the approach is general and now widely used.

Thomann, R.V. and J.P. Connolly. 1984. Model of PCB in the Lake Michigan lake trout food chain. Environ. Sci. Technol. 18(2):65-71.

A multiple step food chain bioenergetics model of PCB bioaccumulation in freshwater that could be applied to mercury in estuarine systems. Growth, respiration, and feeding habits of fish and other organisms are integral to the model. Contaminant assimilation efficiency and excretion rates are needed to parameterize the model. Age/size dependencies in contaminant concentrations are predicted. Contaminant accumulation from water as well as prey is included in this equilibrium model.

Thomann, R.V., J.P. Connolly and T.F. Parkerton. 1992. An equilibrium model of organic chemical accumulation in aquatic food webs with sediment interaction. Environ. Toxicol. Chem. 11:615-629.

An extension of the modelling approach of Thomann and Connolly (1984) that explicitly includes contaminant loading of sediments as an ultimate source of contaminants to multi-step aquatic food webs. Organic contaminants are modelled, but extension to methylmercury should be possible. The relative importance of sediments, water, and food as contaminant sources is predicted.

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MERCURY ASSIMILATION AND EXCRETION BY FISH

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Beuhringer, H. 1981. Uptake and excretion of radioactive mercury by rainbow trout (Salmo gairdneri Rich). Inf. Fischwirtsch 28(2):72-74.

Hartman, A.M. 1978. Mercury feeding schedules: effects on accumulation, retention, and behavior in trout. Trans. Am. Fish. Soc. 107:369-375.

Jernelov, A. 1968. Laboratory experiments on the change of mercury compounds from one into another. Vatten 24(4):360-362.

Jarvenpaa, T., M. Tillander and J.K. Miettinen. 1970. Methylmercury: half-life of elimination in flounder, pike and eel. Suomen Kemistilehti B43 439-442.

Found turnover half-times of 640 to 780 days in pike, 400 to 700 days in flounder, and 900 to 1000 days in eels.

Keckes, S. and J.K. Miettinen. 1972. Mercury as a marine pollutant. In M. Ruivo (ed.) Marine Pollution and Sea Life. Fishing News (Books) Ltd., London, pp. 276-289.

An early general summary that includes methylmercury turnover half-times in fish ranging from 267 to 1000 days.

Lock, R.A.C. 1975. Uptake of methylmercury by aquatic organisms from water and food. In U.H. Koeman and J.J.T.W.A. Strik (ed.) Sublethal Effects of Toxic Chemicals on Aquatic Animals. Elsevier, Netherlands, p. 61-79.

Lockhart, W.L., J.F. Uthe, A.R. Kenney and P.M. Mehrle. 1972. Methylmercury in northern pike (Esox lucius) : distribution, elimination, and some biochemical characteristics of contaminated fish. J. Fish. Res. Board Can. 29:1519-1523.

Cross transplants of fish between mercury contaminated and uncontaminated lakes suggests a turnover half-time of about 2 years for methylmercury in large pike (4-19 kg).

Miettinen, J.K. 1974. The accumulation and excretion of heavy metals in organisms. Prog. Water Technol. 7: 215-229.

Summarizes turnover times of inorganic and organic mercury in man and some aquatic animals. Depending on species, half times ranged from 200 to 1200 days in fish, and they are temperature dependent.

Miettinen, J.K., M. Heyraud, and S. Keckes. 1972. Mercury as a hydrospheric pollutant. II. Biological half-time of methyl mercury in four Mediterranean species: a fish, a crab and two molluscs. In M. Ruivo (ed.) Marine Pollution and Sea Life. Fishing News (Books) Ltd., London, pp. 295-298.

Determined the excretion rate of methylmercury in a small(7-12 cm) marine fish(Serranus scriba) to have a half time of 267 days. Half times for the invertebrates were, 400 days for the crab, Carcinus maenus, 481 days for the clam Tapes decussatus, and 1000 days for the mussel, Mytilus galloprovincialis.

Nagashima, Y., T. Kikuchi and M. Chiba. 1984. Toxicity and accumulation of mercury in fish, the medaka Oryzias latipes. Bull. Jpn. Soc. Sci. Fish. 50:95-99.

Newman, M.C. and D.K. Doubet. 1989. Size-dependence of mercury (II) accumulation kinetics in the mosquitofish, Gambusia affinis (Baird and Girard). Arch. Environ. Contam. Toxicol. 18(6):819-825.

Niimi, A.J. 1987. Biological half-lives of chemicals in fishes. Rev. Environ. Contam. Toxicol. 99:1-46.

Summarizes the biological half-life information for contaminants, including mercury, in fish. Half-lives range from 11 to 242 days for inorganic mercury and from 53 to 780 days for methylmercury. The effects of biological variables such as body weight and environmental variables such as temperature are discussed.

Pentreath, R.J. 1976a. The accumulation of organic mercury from sea water by the plaice, Pleuronectes platessa L. J. Exp. Mar. Biol. Ecol. 24:121-132.

Rapid accumulation of methylmercury from water was observed. Excretion rate half-time of about 275 days was measured. Effects of body size and growth were evaluated.

Pentreath, R.J. 1976b. The accumulation of mercury from food by the plaice, Pleuronectes platessa L. J. Exp. Mar. Biol. Ecol. 25:51-65.

In contrast to inorganic mercury, methylmercury was found to be readily absorbed and only slowly eliminated. Assimilation efficiencies from food were 4 to 28% for inorganic mercury, and 81 to 93% for methylmercury. Retention half-times were 26 to 43 days for inorganic mercury and 100 to 257 days for methylmercury. These data were incorporated in a bioenergetic model to predict time dependent mercury concentrations in fish. Growth and size/age dependence were explicitly incorporated in the model.

Pentreath, R.J. 1976c. The accumulation of inorganic mercury from sea water by the plaice, Pleuronectes platessa L. J. Exp. Mar. Biol. Ecol. 24:103-119.

Although inorganic mercury is rapidly taken up from water in laboratory tracer studies by the fish, its tissue distribution is much different from the mercury distribution observed in field captured fish in which methylmercury is the dominant form. The observed biological half-time of inorganic mercury was 103 days. Accumulation from water of inorganic mercury does not appear to account for the observed methylmercury observed in free living fish;. Size/age dependence of mercury accumulation was measured in the laboratory studies and incorporated in a model of inorganic mercury bioaccumulation and turnover.

Pentreath, R.J. 1976d. The accumulation of mercury by the thornback ray Raja clavata L. J. Exp. Mar. Biol. Ecol. 25:131-140.

Assimilation efficiencies of 9 to 20% was found for inorganic mercury in food and 94 to 100% for methylmercury. Respective turnover times were 53 to 72 days for inorganic mercury and 278 to 417 for methylmercury.

Phillips, G.R. and R.W. Gregory. 1979. Assimilation efficiency of dietary methylmercury in northern Pike(Esox lucius). J. Fish. Res. Board Can. 36-1516-1519.

An assimilation efficiency of methylmercury of only 19% (6-31%) was observed in pike feeding on young carp. A review of methylmercury assimilation efficiencies for other species of fish ranged from 10% to 89% with a median value of about 60%.

Phillips, G.R. and D.R. Buhler. 1978. The relative contributions of methylmercury from food or water to rainbow trout (Salmo gairdneri) in a controlled laboratory environment. Trans. Am. Fish. Soc. 107:853-861.

Rodgers, D.W. and F.W.H. Beamish. 1983. Dynamics of dietary methylmercury in rainbow trout, Salmo gairdneri. Aquat. Toxicol. 2:271-290.

Rodgers, D.W. and F.W.H. Beamish. 1981. Uptake of waterborne methylmercury by rainbow trout (Salmo gairdneri) in relation to oxygen consumption and methylmercury concentration. Can. J. Fish. Aquat. Sci. 38:1309-1315.

Ruohtula, M. and J.K. Miettinen. 1975. Retention and elimination of 203Hg-labelled methylmercury in rainbow trout. Oikos 26:385-390.

Sharpe, M.S., A.S.W. deFreitas and A.E. McKinnon. 1977. The effect of body size on methylmercury clearance by goldfish (Carassius auratus). Environ. Biol. Fish. 2:177-183.

Stary, J., B. Havlik, K. Kratzer, J. Prasilva, and J. Hanusova. 1981. Mercury circulation in aquatic environment. Part 4. The accumulation of inorganic mercury and phenylmercury by fish (Poecilia reticulata (Peters)). Acta Hydrochim. Hydrobiol. 9:545-553.

Suzuki, T. and M. Hatanaka. 1975. Experimental investigation on the biological concentration of mercury--II. On the origin of mercury found in the body of young yellowtail. Bull. Jpn. Soc. Sci. Fish. 41:225-231

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MERCURY IN AQUATIC FOOD CHAINS

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Bernhard, M. and M.O. Andreae. 1985. Transport of trace metals in marine food chains. In J.O. Nriagu (ed.) Changing metal cycles and human health. Springer-Verlag, Berlin, pp. 143-167.

Boudou, A. and F. Ribeyre. 1985. Experimental study of trophic contamination of Salmo gairdneri by two mercury compounds-$HgCl_2$ and CH_3HgCl-analysis at the organism and organ level. Water Air Soil Pollut. 26:137-148.

Assimilation efficiencies from food ranged from 6 to 14% for inorganic mercury and from 76 to 86% for methylmercury. Much of the food associated inorganic mercury was fixed to intestinal walls from which it was not assimilated but released back to feces by desorption or intestinal wall sloughing and excreted.

Buffoni, G., M. Bernhard and A. Renzoni. 1982. Mercury in the Mediterranean tuna. Why is their level higher than in the Atlantic tuna? A model. Thalassia Jugoslavica 18:231-243.

Gardner, W.S., D.R. Kendall, R.R. Odom, H.L. Windom and J.A. Stephens. 1978. The distribution of methyl mercury in a contaminated salt marsh ecosystem. Environ. Pollut. 15:243-251.

Mercury contamination from a chlor-alkali plant was studied in a salt marsh ecosystem. Mercury concentrations in biota of the salt marsh were measured. Snails, echinoderms, annelid worms, blue crabs and other crustaceans as well as 11 species of fish were analyzed for both total and methylmercury. Birds and mammals were also analyzed, and had the highest concentrations suggesting biomagnification of mercury. Fish (generally small) had somewhat higher mercury concentrations than their food; predaceous fish had higher mercury concentrations than herbivorous fish, also suggesting food chain biomagnification. Plant materials were low in mercury which was predominantly in an inorganic form in contrast to animals which contained mercury mostly as methylmercury. The importance of mercury methylation in the salt marsh was emphasized. Sediments and biota both reflected the elevated mercury concentrations derived from the chlor-alkali plant.

Hamdy, M.K. and N.V. Prabhu. 1979. Behavior of mercury in biosystems III. Biotransference of mercury through food chains. Bull. Environ. Contam. Toxicol. 21:170-178.

An experimental food chain, bacteria-insect larvae-freshwater fish was used to study the accumulation and trophic transfer of a radioactive mercury tracer.

Jernelov, A. and H. Lann. 1971. Mercury accumulation in food chains. Oikos 22:403-406.

The authors argue that mercury transfers between benthos (sediment inhabitants) and fish is small because the proportion of methylmercury in benthos is small compared to fish and concentrations are variable. The argument is not well supported.

Knauer, G.A. and J.H. Martin. 1972. Mercury in a marine pelagic food chain. Limnol. Oceanogr. 17(6):868-876.

Mercury concentrations in a phytoplankton-zooplankton-anchovy food chain were low and similar. This suggested lack of food chain amplification in this particular food chain. This conclusion should not be generalized to other food chains with more trophic steps and piscivorous predators.

Mikac, N, M. Picer, P. Stegnar and M. Tused-Znidaric. 1985. Mercury distribution in a polluted marine area, ratio of total mercury, methyl mercury and selenium in sediments, mussels and fish.

Mercury contamination from a chlor-alkali plant was the inferred source to sediments, mussels, and fish. Patterns of mercury concentrations in sediments and mussels were similar, suggesting transfers of mercury between sediments and mussels. A negative correlation between concentrations of total mercury and the percentage of methylmercury in mussels suggested that the normal pattern of dominant uptake of methylmercury in low (near natural) contamination environments is overwhelmed by the accumulation of inorganic mercury in the vicinity of massive inorganic mercury contamination. Most mercury in fish is methylmercury. Sediments contaminated with inorganic mercury may not show a proportional influence on mercury concentrations in the food chain terminated by fish because of the difference in the chemical form of mercury in sediments.

Paasivirta, J., J. Sarkka, K. Surma-Aho, T. Humppi, T. Kuokkanen and M. Martinen. 1983. Food chain enrichment of organochlorine compounds and mercury in clean and polluted lakes of Finland. Chemosphere 12:239-252.

Prabhu, N.V. and M.K. Hamdy. Behavior of mercury in biosystems I. Uptake and concentration in food chains. Bull. Environ. Contam. Toxicol. 18(4):409-417.

Ratkowsky, D.A., T.G. Dix and K.C. Wilson. 1975. Mercury in fish in the Derwent Estuary, Tasmania, and its relation to the position of the fish in the food chain. Aust. J. Mar. Freshwat. Res. 26:223-231.

Fish position in the food chain appeared to be an important factor in determining its mercury content; piscivorous fish had generally higher mercury concentrations than those fish feeding on invertebrates or plant material. Gradients of mercury contamination in the sediments were paralleled by mercury concentrations in fish.

Riisgard, H.U. and S. Hansen. 1990. Biomagnification of mercury in a marine grazing food-chain: algal cells Phaeodactylum tricornutum, mussels Mytilus edulis and flounders Platicthys flesus studied by means of a stepwise-reduction-CVAA method. Mar. Ecol. Prog. Ser. 62:259-270.

Both laboratory studies and field transfers to mercury contaminated environments were employed to document transfers of mercury along an algal, mussel, fish food chain. Assimilation efficiencies of methyl mercury by fish are high (minimum 34%) and elimination is slow, in partial contrast to inorganic mercury. Analyses of feces indicated assimilation efficiencies of 63% for inorganic mercury, and 86% for methylmercury.

Zauke, G.P. 1977. Mercury in benthic invertebrates of the Elbe estuary. Helgol. Wiss. Meeresunter. 29:358-374.

Field sampling along a mercury contamination gradient in the Elbe estuary shows that mercury concentrations in benthic invertebrates are roughly proportional to mercury concentrations in sediments. Polychaetes, bivalve and gastropod molluscs, and crustaceans were among the taxa sampled. Limited fish sampling occurred; mercury concentrations in fish were similar to those found in the invertebrates on which they fed, suggesting little food chain biomagnification.

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MERCURY TRANSFERS BETWEEN SEDIMENTS
AND BENTHIC INVERTEBRATES

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Bacci, E. 1989. Mercury in the Mediterranean. Mar. Pollut. Bull. 20(2):59-63.

High levels of mercury in Mediterranean deepwater biota is attributed to high rates of methylation in sediments.

Breteler, R.J. 1981. Bioavailability of mercury in several north-eastern U.S. Spartina ecosystems. Estuarine Coastal Shelf Sci. 12:155-166.

Estuarine marshes were treated with mercury contaminated sewage sludge. Mercury concentrations did not increase in marsh grasses, mussels, and crabs of the treated marshes. Mercury concentrations are highest in organisms collected from marshes whose sediments have the lowest organic carbon content. This may explain the absence of elevated mercury concentrations in sludge treated marshes since sludge is very high in organic carbon content, and its associated mercury would be unavailable.

Eganhouse, R.P. and D.R. Young. 1978. Total and organic mercury in benthic organisms near a major submarine wastewater outfall system. Bull. Environ. Contam. Toxicol. 19:758-766.

Mercury concentrations in fish and benthic invertebrates sampled at mercury contaminated sites near a wastewater

outfall were lower than concentrations in organisms sampled at a site remote from such contamination. The strong association of mercury with sediment organic matter at the outfall sites was thought to reduce its availability to benthic organisms despite the much higher mercury concentrations in the sediments there. Organic mercury was a high percentage (>70%) of total mercury in muscle tissue of fish, crabs and shrimp but a lower percentage (<50%) in tissues of snails, urchins and nudibranchs.

Frithsen, J.B. 1984. Metal incorporation by benthic fauna: Relationships to sediment inventory. Estuarine Coastal Shelf Sci. 19:523-539.

Mercury bioconcentration factors (BCF) from sediment to benthic organisms were calculated in a mesocosm study. Radiotracers of various other metals were also used. Mercury BCFs of 24 were measured for meiofaunal harpactacoids and kinorhynchs. Among macrofauna, the BCFs were 1 to 8 for polychaetes, 20 for bivalves and 2 for gastropods 89 days after addition of radiotracers to the water.

Guthrie, R.K., E.M. Davis, D.S. Cherry and H.E. Murray. 1977. Biomagnification of heavy metals by organisms in a marine microcosm. Bull. Environ. Contam. Toxicol. 21:53-61.

Khan, A.T., J.S. Weis and L. D'Andrea. 1989. Bioaccumulation of four heavy metals in two populations of grass shrimp, Paleomonetes pugio. Bull. Environ. Contam. Toxicol. 42(3):339-343.

Langston, W.J. 1986. Metals in sediments and benthic organisms in the Mersey estuary. Estuarine Coastal Shelf Sci. 23:239-261.

Highest concentrations of mercury in British estuarine sediments are found near sites of industrial inputs, especially from chlor-alkali plants. Mercury concentrations in benthic organisms are predicted best if mercury concentrations in sediments are normalized for organic matter concentrations. About 20% of the mercury in the deposit feeding mollusc, Scobicularia plana was found as methylmercury. Selective uptake from sediments of this form was thought to explain the 20 to 50 fold higher bioconcentration factor from sediments of methyl-mercury compared to inorganic mercury.

Langston, W.J. 1982. The distribution of mercury in British estuarine sediments and its availability to deposit-feeding bivalves. J. Mar. Biol. Asso. U.K. 62:667-684.

Mercury concentrations in deposit feeding bivalve molluscs and in worms are positively related to sediment mercury concentration and negatively related to sediment organic

carbon concentrations. Sediment organic matter seems to reduce mercury bioavailability. Organisms' mercury concentrations are predictable from sediment mercury and organic matter concentrations in the Mersey River estuary. Reductions in sediment mercury concentrations subsequent to pollution control are reflected in reductions in mercury concentrations in benthic invertebrates. Sediment to organism biological concentration factors are about 1 to 3/% organic matter.

Neff, J.W., R.S. Foster and J.F. Slowey. 1978. Availability of sediment-adsorbed heavy metals to benthos with particular emphasis on deposit-feeding infauna. Tech. Memo. D-78-42, Dredge Material Research Program. Environmental Laboratory, U.S Army Corps of Engineer Waterways Experiment Station, Vicksburg Miss.

Clams, shrimp, and worms of five different species were used to test the uptake and bioavailability of the metals Cd, Cr, Cu, Fe, Mn, Pb, Hg, and V from sediments collected from contaminated harbor sites subject to routine channel dredging. Patterns of mercury accumulation were not clear in both short and long term exposure studies. Evidence of mercury uptake from contaminated sediments was most clearly demonstrated from freshwater sediments.

Pelletier, E. 1986. Modification de la bioaccumulation du selenium chez Mytilus edulis en presence du mercure organique et inorganique. Can. J. Fish. Aquat. Sci. 43:203-210.

Exposure of mussels to inorganic or methylmercury increased the availability of selenium to them in microcosm studies using TiO_2 as an artificial sediment substrate. However, selenium exposure did not influence the bioavailability of either form of mercury to the mussels.

Porcu, M. and M.L. Tagliasacchi. 1983. Trophic ecology of crustaceans in a brackish littoral pond of Sardinia Island polluted by mercury. Cah. Biol. Mar. 24(2):159-175.

Rubenstein, N.I., E. Lores and N.R. Gregory. 1983. Accumulation of PCB's, mercury and cadmium by Nereis virens, Mercenaria mercenaria, and Paleomonetes pugio from contaminated harbor sediments. Aquat. Toxicol. 3(3):249-260.

Mercury in contaminated New York Harbor sediments was not accumulated above background concentrations in sandworms, hard clams and grass shrimp. PCB's were accumulated above background.

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MERCURY ASSIMILATION AND TURNOVER IN BENTHIC INVERTEBRATES

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Cossa, D. and J.C. Rondeau. 1985. Seasonal, geographical and size-induced variability in mercury content of Mytilus edulis in an estuarine environment: a re-assessment of mercury pollution level in the Estuary and Gulf of St. Lawrence. Mar. Biol. 88:43-49.

Seasonal variations in mercury concentrations of free living mussels show changes of as much as two fold over periods of as little as 2 months. This suggests that the turnover time of mercury is rapid in mussels, on the order of a few months or less.

Cunningham, P.A. and M.R. Tripp. 1975a. Factors affecting the accumulation and removal of mercury from tissues of the American oyster Crassostrea virginica. Mar. Biol. 31:311-319.

Accumulation and depuration of inorganic mercury by oysters was determined in laboratory experiments under conditions of changing and constant temperature. Biological half-lives ranged from 9 to 35 days in experiments of 80 days duration. Half-lives were shorter after exposure to higher mercury concentrations in water (100 ppb) than in lower concentration (10 ppb) and longer under conditions of declining temperature.

Cunningham, P.A. and M.R. Tripp. 1975b. Accumulation, tissue distribution and elimination of $^{203}HgCl$ and $CH_3{}^{203}HgCl$ in the tissues of the American oyster Crassostrea virginica. Marine Biology 31:321-334.

Inter-tissue transfers of inorganic and methylmercury were inferred from changes in tissue distributions in laboratory exposed animals during periods of uptake and depuration. Transfers of methylmercury to muscle and gonad tissue from superficial tissues (e.g. gills) occurred even during depuration. Such transfers are a likely source of the multiple component depuration rate constants observed in such studies.

Denton, G.R.W. and C. Burdon-Jones. 1981. Influence of temperature and salinity on the uptake, distribution and depuration of mercury, cadmium and lead by the black-lip oyster Saccostrea echinata. Mar. Biol. 64:317-326.

Dillon, T.M. and J.M. Neff. 1978. Mercury and the estuarine marsh clam, Rangia cuneata Gray. II. Uptake, tissue distribution and depuration. Mar. Environ. Res. 1:67-77.

Eganhouse, R.P. and D.R. Young. 1978. In situ uptake of mercury by the intertidal mussel, Mytilus californianus. Mar. Pollut. Bull. 9:214-217.

Fowler, S.W. and J. La Rosa. 1978. Factors affecting methyl and inorganic mercury dynamics in mussels and shrimp. Marine Biology 46:267-276.

In radiotracer experiments, methylmercury was accumulated to a greater degree than inorganic mercury from both food and water. Methylmercury was eliminated more slowly than inorganic mercury. In the mussel, turnover of both forms was more rapid at higher temperatures and more rapid in field held animals, presumably because of greater growth under field conditions. Biological half times in days were as follows: methylmercury- 63 (field mussels), 377 (lab mussels), 529 (lab and field shrimp); inorganic mercury- 82 (field mussels), 140 (lab mussels), 112 (lab and field shrimp). Tissue distributions of the isotope in mussels and shrimp are described. Biological half lives in other invertebrates reported in the literature are reviewed.

Kopfler, F. 1974. The accumulation of organic and inorganic mercury compounds by the eastern oyster (Crassostrea virginica). Bull. Environ. Contam. Toxicol. 11:275-280.

Luoma, S.N. 1977. The dynamics of biologically available mercury in a small estuary. Estuarine Coastal Mar. Sci. 5:643-652.

Total mercury concentrations in shrimp (Paleomonetes debilis) and polychaete worms (Nereis succinea) in a Hawaiian estuary fluctuated by more than two orders of magnitude over a year. Rapid accumulation of mercury carried into the estuary during rainy periods was thought to be the main cause of this variation. Half-times of loss measured in laboratory radiotracer studies with inorganic mercury were 17 days for the crab and 22 days for the worm, supporting the argument of rapid response to changing conditions of mercury exposure in these invertebrates.

Luoma, S.N. 1976. The uptake and interorgan distribution of mercury in a carnivorous crab. Bull. Environ. Contam. Toxicol. 14(6):719-723.

The tissue distribution of mercury in the crab, Thalamita crenata, differed for animals collected in the field and those labelled with radioactive inorganic mercury via food in the laboratory over a 13 day period. Field collected crabs were thought to accumulate mercury rapidly from a biologically available pool in the water via gills which was then translocated more slowly to other organs.

Mohlenberg, F. and H.U. Riisgard. 1988. Partitioning of inorganic and organic mercury in cockles Cardium edule (L.) and C glaucum (Bruguiere) from a chronically polluted area: influence of size and age. Environ. Pollut. 55:137-148.

Organic mercury increased as a percentage of total mercury in cockles as a function of age, increasing from 30% in two year olds to 90% in four year olds. This is attributed to the more rapid accumulation and slower depuration of methylmercury relative to inorganic mercury.

Riisgard, H.U., T. Kiorboe, F. Mohlenberg, I Drabaek and P. Pheiffer Madsen. 1985. Accumulation, elimination and chemical speciation of mercury in the bivalves Mytilus edulis and Macoma baltica. Mar. Biol. 86:55-62.

Cross transplant experiments with mussels between mercury contaminated and clean sites showed depuration half-times of 293 days for mussels transferred from a chronically contaminated area to a clean area, but only 53 days in mussels transferred from a temporary massively contaminated site. In both cases, 75% of the mercury in the mussels was inorganic, but phenylmercury rather than methylmercury was the dominant organic form. Mercury concentrations in deposit feeding clams were much lower than in the filter feeding mussels. The depuration half-time for mercury in the clams probably exceeded a year.

Riisgard, H.U. and P Famme. 1986. Accumulation of inorganic and organic mercury in shrimp Crangon crangon. Mar. Pollut. Bull. 17(6):255-257.

Shrimp fed on mercury labeled clams retained 4% of the ingested inorganic mercury and 75% of the organic (mixed phenylmercury and methylmercury).

Seymour, A.H. and V.A. Nelson. 1971. Biological half-lives for zinc and mercury in the Pacific oyster, Crassostrea gigas. D.A. Nelson (ed.) Proc. Natl. Symp. Radioecology. Oak Ridge National Laboratory, Oak Ridge, Tenn. pp.849-856.

The biological loss rate of inorganic mercury in oysters was measured in laboratory radiotracer studies. Turnover half-times ranged from 5 to 44 days in a 19 week experiment. Longest turnover times were measured late in the depuration period, suggesting several pools of mercury in the oysters with different kinetics of mercury loss.

Sloan, A.L., J.A.J. Thompson and P.A. Larkin. 1974. The biological half-life of inorganic mercury in the Dungeness Crab (Cancer magister). J. Fish. Res Board Can. 31:1571-1582.

Smith, A.L., R.H. Green and A. Lutz. 1975. Uptake of mercury by freshwater clams (family Unionidae). J. Fish. Res. Board Can. 32:1297-1303.

Wrench, J.J. 1978. Biochemical correlates of dissolved mercury uptake by the oyster Ostrea edulis. Mar. Biol. 47:79-86.

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BACKGROUND DATA ON MERCURY CONCENTRATIONS
IN THE LAVACA BAY ENVIRONMENT

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Blanton W.G. and C.J. Blanton. 1971. A study of the mercury concentrations in the edible tissues of selected animals from Lavaca Bay, Texas. Final Report to the Texas Water Quality Board, Austin, Texas. 118 pp.

Data are more fully reported in Blanton et al.(1972). Some biological samples may have been preserved in formalin which might compromise their integrity. However, oyster samples are similar in mercury concentration to those from the TDH monitoring program collected at the same time and location.

Blanton, W.G., C.J. Blanton and M.C. Robinson. 1972. The ecological Impact of mercury discharge on an enclosed secondary bay. Final Report to the Aluminum Company of America, Point Comfort, Texas. 231 pp.

Mercury concentrations were measured in surface sediments of Lavaca Bay during the period immediately following cessation of mercury containing wastewater discharges from ALCOA's chloralkali facility in April 1970. Sediment was sampled in July, August and October of 1970 and April, July, and October of 1971. Initial sampling in July 1970 showed highest mercury concentrations adjacent to the ALCOA facility, reaching as much as 85 ppm Hg near the wastewater outfalls. A plume of mercury enriched sediment extended northwest into upper Lavaca Bay at this time. Later sampling indicated a redistribution of mercury throughout much of upper and lower Lavaca Bay and into Cox Bay and Keller Bay with maximum concentrations oriented along the ship channel suggesting the effects of tidal mixing. Concentrations adjacent to the ALCOA facility diminished somewhat during the year and a half following wastewater diversion.

Sediment cores collected near the ALCOA facility showed subsurface maxima in mercury concentrations(reaching 103 ppm Hg) thought to reflect relict inputs from the period of active mercury inputs from wastewater discharges. Subsequent dilution with low mercury sediments and surface resuspension and redistribution were thought to explain these observations, echoing the conclusions of Riegel(1990) nearly twenty years later.

Laboratory microcosm studies of mercury methylation indicated that rates of methylation were too low to significantly affect total mercury distributions in sediments which are thought to be controlled largely by strong binding of mercury to sediments and physical redistribution of sediment particles. Methylmercury could barely be detected in either natural sediments or microcosm samples spiked with inorganic mercury. The role of mercury methylation in enhancing its bioavailability was suggested but not experimentally documented.

Mercury concentrations were measured in fish from the general area. Unfortunately, most were sampled from seafood markets and represented mostly pelagic fish caught throughout the western Gulf of Mexico. Methylmercury was the dominant form of mercury in these fish. Mullet caught near Point Comfort, however, had only about a third of their total mercury as methylmercury, and averaged about 0.15 ppm Hg wet weight in total mercury concentration.

Mercury concentrations in oysters(apparently collected and analyzed by TDH, TWC and ALCOA) reveal highest concentrations (as high as 8 ppm Hg wet weight) at Mitchell's Reef, just south of Point Comfort. Here, as at other sites, mercury concentrations in oysters declined rapidly during the year immediately following cessation of wastewater inputs.

A brief summary figure reports mercury concentrations in other species of biota from Lavaca Bay. Penaeid shrimp, sheepshead, southern flounder, spotted seatrout, Atlantic croaker, bay anchovy, banded drum, black drum, Spanish sardine, and gafftopsail catfish were analyzed. Some of the species are possible intermediate prey of red and black drum. Mercury concentrations in them would be useful in delineating the importance of the water-to pelagic fish to drum pathway of mercury transfer.

Bowman, J.W. 1988. Mercury in Lavaca Bay. Memorandum to Texas Water Commission, Water Quality Division, dated April 29,1988.

A short narrative summary of the Lavaca Bay mercury problem is followed by data on total mercury concentrations in water and sediments sampled at 4 sites in Lavaca Bay: at State Highway 35, at the junction of the Port Lavaca and Matagorda Ship channels, at the ALCOA ship channel dock, and at Cox Bay. Mercury concentrations in water are only rarely above the methodological detection limit(0.2 to 1.0 ppb Hg); the methods utilized are not appropriately sensitive enough for the purposes of mercury bioaccumulation studies. Mercury concentrations in sediments were highest at the site nearest the ALCOA facility, with a maximum of 7.1 ppm Hg reported in 1977. Mercury concentrations in sediments appear to have declined several fold in sediments at this and the other sites over the period 1977-1988, perhaps in response to dredging

operations which could either allow mercury enriched sediments to settle from peripheral areas into deepened channels or by burial under less contaminated sediments.

Mercury concentrations in Lavaca Bay biota for the period 1981 to 1988 derived from TDH (1988) are included.

Mercury concentrations in influent and effluent waters of an offshore dredge spoil lagoon are reported for the period 1975 to 1987. The lagoon received spoil from the dredging of the ALCOA ship channel and perhaps other areas. The data were provided by ALCOA. Mercury concentrations in the water were only infrequently in excess of the methodological detection limit of 1 ppb Hg. However, on a dry weight basis (which probably means the concentration on suspended dredge spoil sediments), concentrations of mercury were as high as 342 ppm. This is nearly 50 times higher than the maximum mercury concentration in bottom sediments. It could reflect a preferential enrichment of mercury in fine grained, suspendable sediments.

Hall, R.A., E.G. Zook and G.M. Meaburn. 1978. National Marine Fisheries Service Survey of trace elements in the fishery resource. NOAA Tech. Rep. NMFS SSRF-721. 313 pp.

A massive survey of metal concentrations in fish, molluscs and crustaceans caught for food in U.S. coastal and adjacent waters. Mercury was measured in many of the species of concern to this study, including red drum, black drum, and blue crabs. Unfortunately none appear to have been sampled from Lavaca Bay. The data are therefore largely useful to define a baseline of near natural mercury concentrations in these species against which to compare those in Lavaca Bay. Only regional ranges and mean concentrations along with similar statistics for length and weight are reported. The regions are not precisely located (e.g. "Hawaii", "Gulf", or "North Atlantic").

Nationwide mean concentration ranges for mercury are as follows (ppm Hg wet weight): red drum (0.1-0.2), black drum (0.1-0.2), blue crab (0.1-0.2), oyster (<0.1), shrimp (<0.1), gafftopsail catfish (0.5-0.6), Atlantic croaker (<0.1), southern flounder (<0.1), mullet (<0.1), menhaden (<0.1), and spotted seatrout (0.2-0.3).

Holmes, C.W. 1977. Effects of dredged channels on trace metal migration in an estuary. J. Res., U.S. Geol. Survey 5:243-251.

Holmes, C.W. 1986. Trace metal seasonal variations in Texas marine sediments. Mar. Chem. 20:13-27.

Mercury concentrations in Lavaca Bay sediments measured in 1970 are summarized from the report of Holmes (1977). More

than 800 bottom sediment samples were analyzed. A lens of elevated mercury concentrations (above 1 ppm Hg) was found to emanate from the vicinity of the ALCOA facility at Point Comfort and orient southeast along the eastern side of the Matagorda ship channel. Mercury concentrations ranged from a high of about 10 ppm Hg to background levels of about 0.02 ppm Hg. A secondary maximum in mercury concentrations was found further south in Matagorda Bay in a zone of fine grain sediment deposition. Some movement of mercury enriched sediments north into upper Lavaca Bay was observed. The distribution patterns and retention of mercury in Lavaca Bay sediments was attributed to a combination of estuarine circulation which retained mercury containing sediments near the turbidity maximum and reduced oxygen in bottom waters which acted to precipitate dissolved mercury as sulfides and limit its flushing from the estuary.

Mercury concentrations in suspended matter exceeded 100 ppm Hg during flood tide.

National Benthic Surveillence Project (NBSP). 1988. Unpublished data, National Benthic Surveillance Project, National Status and Trends Program, NOAA, NMFS, Southeast Fishery Science Center, Beaufort Laboratory, Beaufort, North Carolina.

Total mercury concentrations were measured in the livers of 10 Atlantic croaker, 7 black drum, 8 red drum, and 1 speckled trout. Liver mercury concentrations were as high as 30 ppm Hg wet weight in a large black drum. Mean mercury concentrations in livers of Atlantic croaker were about 4 times as much as the mean measured previously in croaker from other estuaries in the southeastern U.S., suggesting elevated mercury concentrations in Lavaca Bay croaker.

TDH measured mercury concentrations in edible flesh of the same fish. Concentrations of mercury in edible flesh averaged only about 20% of those in livers.

Cadmium concentrations in the livers of black drum were also extraordinarily high compared to other species. Lead, copper, and zinc concentrations were not unusual. This suggests an unusual sequestration process for mercury and cadmium in black drum.

National Oceanographic and Atmospheric Administration (NOAA). 1989. A summary of data on tissue contamination from the first three years (1986-1988) of the Mussel Watch Project. NOAA Tech. Memo. NOS OMA 49. 22 pp. and appendices.

Mercury and other metal and organic contaminant concentrations are reported from the Mussel Watch Project of NOAA's Status and Trends Program. Mercury concentrations are reported for mussels or oysters at 177 coastal sites in the U.S. Among 97

sites where mercury was measured in oysters (Crassostrea virginica), the third highest three year mean mercury concentration was reported for the Matagorda Bay site at Gallnipper Point, on the western shore of Lavaca Bay opposite the ALCOA facility at Point Comfort. Total mercury concentrations averaged 0.38 ppm Hg dry weight, which should be equivalent to about 0.08 ppm Hg wet weight. This is similar to concentrations reported by TDH (1988) for oysters near Point Comfort. Sites at the Lavaca River mouth and seaward in Matagorda Bay proper had lower mercury concentrations. This reinforces the conclusion of elevated mercury concentrations in the Point Comfort area of Lavaca Bay.

National Ocean Service (NOS/NOAA). 1991. Second summary of data on chemical contaminants in sediments from the National Status and Trends Program. NOAA Tech. Memo. NOS OMA 59. 29 pp. and appendices.

Mercury and other metal and organic contaminant concentrations in sediments are reported for 205 coastal sites from NOAA's Status and Trends Program. Among Lavaca/Matagorda Bay sites, that at Gallnipper Point had the highest mean total mercury concentration after normalizing for grain size, 0.29 ppm Hg. This emphasizes, like the Status and Trends oyster data (NOAA, 1989), the elevated mercury concentrations in the vicinity of the ALCOA facility at Point Comfort. However, in contrast to measurements of mercury in oysters, sediment mercury at Gallnipper Point is only slightly elevated in comparison to most Status and Trends sites, ranking 60th among 205 sites nationwide. The differences in the ranking of mercury concentrations in oysters and sediments at the Gallnipper Point site, may reflect enhanced biological availability of sediment bound mercury there. The reported total organic carbon concentration in sediments at this site is low, only 0.1%. Mercury binding to sediment organic carbon is thought to reduce its biological availability.

Palmer, S.J. 1992. Mercury bioaccumulation in Lavaca Bay, Texas. Masters Thesis, Texas A.& M. University, College Station, Texas. 139 pp.

A wealth of relevant information focused on the issue of mercury bioaccumulation in Lavaca Bay biota. Of particular importance are measurements of mercury concentrations in benthic infauna and epifauna that are at the base of the food web of red and black drum and blue crabs, information otherwise lacking in the Lavaca Bay data base.

Also important are results of experimental cross transplants of shrimp, crabs and oysters between the area adjacent to the Point Comfort and less contaminated areas. Shrimp and oysters

transplanted to the mercury contaminated area rapidly accumulated mercury, roughly in proportion to concentrations in local sediments. Oysters transplanted from the contaminated area to one of lower mercury concentrations, rapidly lost mercury. These results strongly implicate the sediments near the ALCOA facility as a major source of bioavailable mercury to resident invertebrates. However, blue crabs transplanted to the mercury contaminated area did not show statistically significant increases in mercury concentrations. Variability was high. Homogenized whole bodies were analyzed, which may have obscured trends in edible tissues. The turnover time of mercury in blue crabs could be much longer than in oysters and shrimp, and the 35 day period of exposure to elevated mercury concentrations may have been inadequate to allow any significant approach to equilibrium.

Total mercury concentrations are reported for 7 species of benthic polychaetes, 5 species of bivalve molluscs, and 4 species of crustaceans from Lavaca (Point Comfort) and Keller Bays. Algae, nemerteans, chironimids, detritus and wood, and a single fish and Littorina snail were also analyzed for mercury. In general, Lavaca Bay (Point Comfort) samples had 5 to 10 times the mercury concentrations of Keller Bay samples of the same species or type. Sediments in the two bays had mercury concentrations in roughly this same ratio, as did shrimp and oysters from the cross transplant experiments. Mercury concentrations in biota seem to reflect mercury concentrations in coexisting sediments. Infaunal polychaete worms seemed to have higher mercury concentrations than infaunal bivalves which were higher in turn than epifaunal crustaceans. Within these broad taxonomic divisions, however, there were large interspecific differences in mercury concentrations and large variations among individuals within a species. Among worms of the species Laeonereis culveri, for example, four individuals from the Point Comfort area had measured mercury concentrations ranging from 0.6 to 16.0 ppm Hg dry weight.

Median mercury concentrations (ppm Hg dry weight) in Lavaca Bay biota were algae (0.57), crustaceans (0.89), polychaete worms (2.79), bivalve molluscs (1.58), oysters (2.14), whole shrimp (>1.17), and organic detritus (3.55).

Sediments in the vicinity of Point Comfort had total mercury concentrations measured at as much as 1.35 ppm Hg.

Two analyses for total mercury concentrations in bay water are reported: 0.0045 ppb in Lavaca Bay and 0.0005 ppb in Keller Bay, a ratio of 9 which is consistent with the ratio of mercury concentrations in biota and sediments between the two areas. These concentrations are about 1000 times lower than those reported in Bowman (1988) and may be the only reliable measurements of mercury concentrations in water from the area.

Mercury concentrations in net plankton ranged from 0.1 to 1.0 ppm Hg dry weight in Lavaca Bay samples which were higher in mercury content than Keller Bay samples. The biological association of the measured mercury was uncertain because of the confounding presence of suspended sediments also collected in these samples. Samples were taken during the years 1991 and 1992.

Reigel, D.V. 1990. The distribution and behavior of mercury in sediments and marine organisms of Lavaca Bay, Texas. Masters Thesis, Texas A.& M. University, College Station, Texas. 113 pp.

Mercury concentrations were measured in sediments and biota in Lavaca Bay during the period 1988 to 1990. A gradient of surface sediment mercury concentrations declining away from the ALCOA facility was observed, with concentrations nearest the facility being as high as 1.86 ppm Hg. Sediments with as much as 0.40 ppm Hg were found as far as 6 km south of the facility. Sediment cores often showed subsurface maxima in mercury concentrations suggesting either burial under less contaminated sediments or resuspension and redistribution of contaminated sediments to other areas of the bay. Cores with surface mercury maxima and uniform mercury distributions were also observed, suggesting a complex pattern of sediment movements. The very smallest size fraction (<0.03mm) of sediments tended to have the highest mercury concentrations. This fraction is most readily resuspended from the bottom into the water column. Highest mercury concentrations were not always found in sediments of finest grain size, however. Sites with coarse sediments but very high mercury concentrations at the sediment surface were found near the ALCOA facility suggesting that sediment resuspension and scouring had not removed remnant mercury contamination.

Dissolved mercury concentrations in the water column and in sediment pore waters were always less than the detection limit of 0.01 ppb Hg. Laboratory experiments showed that dissolved mercury added to sediment suspensions was rapidly adsorbed and tightly bound to sediments.

Limited sampling of non-predatory benthic invertebrates near Point Comfort found the following average total mercury concentrations (ppm Hg dry weight): snails (0.60), whole blue crabs (0.97), razor clams (0.58), worms (0.64), barnacles (0.32), mussels (1.32), stone crabs (2.27), hermit crabs (0.30), and oysters (1.55). Oysters collected from the spoil island and from Gallnipper Point, sites progressively more distant from Point Comfort, had mean mercury concentrations of 0.53 and 0.32 ppm Hg dry weight, respectively.

Sager, D.R. 1992a. Memorandum from David R. Sager, Texas Parks and Wildlife Department to Jim Jeansonne, NOAA Damage Assessment Center, St. Petersburg, Florida, dated April 2, 1992.

Mercury concentrations in Lavaca Bay fish measured by the Texas Parks and Wildlife Department in 1978 are reported. Edible flesh samples were analyzed for mercury in individual fish: 20 red drum, 20 black drum, and 21 spotted seatrout collected at 10 sites in the Lavaca/Matagorda Bay system. Not all species were collected at all sites. The sites included the ALCOA spoil dump, Venado Creek, Lavaca fishing pier, Chocolate Bay, Alamo beach, Swan Lake, Redfish Lake, C.P.L. shoreline, Keller Bay, and upper northeast Lavaca Bay. Fish length and weight were measured. Highest mercury concentrations were found at the sites nearest Point Comfort. Only one fish (black drum from the Lavaca fishing pier) exceeded 1 ppm Hg wet weight.

Sager, D.R. 1992b. Memorandum from David R. Sager, Texas Parks and Wildlife Department to David Engel, NOAA, NMFS, Southeast Fisheries Science Center, Beaufort Laboratory, Beaufort, North Carolina, dated September 8, 1992.

Edible tissues of 7 seafood species collected from Christmas Bay, Texas during 1991-1992 were analyzed for mercury and 8 other metals(As, Cd, Cr, Cu, Pb, Ni, Se, and Zn). Christmas Bay is a minimally impacted subsystem of Galveston Bay. These data can provide a baseline against which to compare the mercury levels in Lavaca Bay biota. The following numbers of organisms were sampled: hardhead catfish (7), oysters (10), spotted seatrout (8), blue crabs (51), Atlantic croaker (5), southern flounder (14), red drum (30). Mercury concentrations are uniformly low, rarely exceeding 0.1 ppm Hg wet weight.

Sediments were also relatively low in mercury, being at or below the detection limit of 0.1 ppm Hg dry weight.

Texas Department of Health (TDH). 1988. Mercury concentrations in marine organisms of Lavaca Bay. A report compiled by the Division of Shellfish Sanitation Control of Texas Department of Health. 67 pp. plus addendum.

Mercury analyses of edible fish and shellfish were begun by TDH in 1970 shortly after cessation of direct release of mercury containing wastewater from the ALCOA chlor-alkali plant in Lavaca Bay. Oysters (Crassostrea virginica) in the vicinity of the plant initially showed mercury concentrations in excess of the FDA limit of 0.5 ppm Hg wet weight, but declined to below this level within months. This suggests accumulation of Hg by oysters directly from water (or indirectly via phytoplankton) since fish and shellfish more

closely associated with bottom sediments did not show comparable declines in mercury concentrations.

Blue crabs (Callinectes sapidus) and five species of finfish have also been targeted for sampling for mercury since 1970: Speckled trout (Cynoscion nebulosus), gafftopsail catfish (Bagre marina), southern flounder (Paralicthys lethostigma), red drum (Sciaenops ocellatus), and black drum (Pogonias cromis). Sheepshead (Archosargus probatocephalus) and Atlantic croaker (Micropogonias undulatus) have occasionally been sampled.

These target species collected in the area near the ALCOA wastewater outfalls frequently have mercury concentrations in excess of the FDA limit (now 1 ppm Hg wet weight). Highest concentrations of mercury reported for the period 1981 to 1991 were oysters (.093 ppm Hg wet weight), blue crabs (4.46 ppm Hg wet weight), red drum (4.55 ppm Hg wet weight), black drum (5.6 ppm Hg wet weight), sheepshead (2.92 Hg wet weight), speckled trout (0.95 ppm Hg wet weight). Crabs and fish collected outside this region but within the Lavaca/Matagorda Bay system rarely exceed the FDA limit for mercury. The localization of elevated mercury levels in crabs and fish seems to implicate the ALCOA chlor-alkali plant as the dominant source. Limited monitoring for mercury in crabs and fish in other Texas bays, has found generally low levels, substantiating this conclusion.

The Lavaca Bay monitoring for mercury in crabs, oysters, and fish has continued, with annual sampling since 1985. Annual updates to the 1988 report are available from Kirk Wiles of TDH.

Beginning in 1981, organism size has been measured, an important factor influencing mercury concentrations. Since 1987, methylmercury has also been measured in all organisms whose total mercury concentrations exceeded the 1 ppm Hg wet weight FDA limit. In these cases methylmercury was generally found to be the dominant form of mercury, typically being greater than 70% of the total mercury.

This is the most extensive data base on mercury concentrations in Lavaca Bay biota. Lacking are mercury analyses of lower trophic level organisms upon which the top predators feed. Sampling locations are specified only to the resolution of the state land tract grid (about one mile square) which is too coarse for many predictive purposes.

Comparison is made to mercury concentrations in coastal marine organisms sampled at other Texas and Gulf of Mexico sites that can serve as baseline references. The elevated mercury levels in biota in the Lavaca Bay region is clearly shown by such comparisons.

Texas Water Commission (TWC), 1992. Texas Water Commission Computer Center transmittal May 18, 1992. Selected Data Report, Statewide Monitoring Network.

Mercury concentrations in fish from several Texas bays and estuaries (including Keller Bay) are reported for the period 1974 to 1988. The data base is small. Fish species are not identified making the data of very limited utility.

Texas Water Commission (TWC). 1991. Texas Water Commission, data transmittal to J. Mayfield, March 25, 1991 of the March 1990 Lavaca Bay Toxic Assessment Study.

A limited data set of mercury analyses of whole fish and filets from hardhead catfish, gafftopsail catfish, flounder, and red drum from Lavaca Bay are reported. A total of only 28 fish were sampled from sites at Cox Bay, Lavaca Bay-Sand Point, and Lavaca Bay-ALCOA ship channel. Mercury analyses are not available on all samples, the highest of which is 2.1 ppm Hg wet weight in the filet of a hardhead catfish collected from the ALCOA ship channel site. Fish lengths and weights are reported but are unreliable.