THE RELATIONSHIP OF FISH CONSUMPTION AND MERCURY CONTAMINATION IN THE ST. JOHNS RIVER

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INTRODUCTION

Mercury contamination of aquatic environments is a widespread problem. Much of the mercury in water can be attributed to industrial wastes from pre-1975 chloralkali operations and pre-1972 paper pulping operations (May and McKinney 1977). It is estimated that up to 230 metric tons of mercury are released annually into water by weathering processes including erosion of rocks and sediments. Coal combustion may release as much as 3,000 metric tons of mercury and mercury compounds per year which can also contaminate the aquatic environment through atmospheric deposition (F.D.E.R., 1984).

Mercury contamination of the environment has declined sharply since the 1960's when the adverse health impacts of mercury became widely known. Contaminated sediments are still a prominent mercury source (May and McKinney 1977). High concentrations of mercury were found in sediments over 100 miles downstream from a synthetic fiber plant which had discontinued mercury use 27 years earlier (May and McKinney 1977).

Swedish scientists found that microorganisms could transform relatively harmless elemental mercury and inorganic mercury salts into the highly toxic methyl mercury (Miller 1975).

Aryl organic mercury, C_6H_5Hg^+ and (C_6H_5)_2Hg, is moderately toxic and exhibits a short retention time in tissue, but it is rapidly transformed by the body, or in the environment, into inorganic mercury, Hg and Hg^{2+}. Inorganic mercury also exhibits moderate toxicity and a short retention time in the body (Miller 1975). The threat of mercury poisoning begins when inorganic mercury salts are converted by microorganisms into aryl organic mercury. These methyl and dimethyl compounds are highly toxic and are readily concentrated within the food web (Miller 1975).

Symptoms of mercury poisoning in humans include headaches, fatigue, irritability, and tumors along with other nervous disorders. These symptoms may surface weeks or months after initial exposure to toxic levels of mercury. Inorganic mercury damages the kidneys, liver and brain by concentrating in these organs (Ehrlich et al. 1977). Methyl mercury also concentrates in the membranes of red blood cells and the nervous system. Biological

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half-life in humans is approximately 70 days (Berlin in Ehrlich et al. 1977).

Accumulation of methyl mercury within the food web is influenced by many factors. Mobility characteristics of an organism may limit the amount of time spent in a contaminated region. The temperature of the water in a contaminated zone will influence the amount of time the organism spends in that area. The extent of mercury contamination of the area is another essential element, as is the size and life stage of the organism. Species trophically higher in the food web are likely to have higher concentrations of mercury.

The current Florida Department of Environmental Regulation (F.D.E.R.) maximum standard for total mercury in the marine environment is 0.10 ug/l (Table 1). While inorganic mercury compounds have been proven to cause mortality in fish larvae at levels of 0.10 ppb, methyl mercury at 0.06 ppb inhibits reproductive capacities (Anonymous 1984).
Table 1. Water quality criteria for mercury as established by the Florida Department of Environmental Regulations, Chapter 17, Paragraph 3.

<table>
<thead>
<tr>
<th>CLASS</th>
<th>Activity</th>
<th>Mercury Limit (ug/l)</th>
</tr>
</thead>
<tbody>
<tr>
<td>CLASS I</td>
<td>Potable Water</td>
<td>0.20</td>
</tr>
<tr>
<td>CLASS II</td>
<td>Shellfish Harvesting</td>
<td>0.10</td>
</tr>
<tr>
<td>CLASS III</td>
<td>Recreational Waters</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Marine</td>
<td>0.10</td>
</tr>
<tr>
<td></td>
<td>Fresh</td>
<td>0.20</td>
</tr>
</tbody>
</table>

Public susceptibility to mercury contamination is largely determined by their fish consumption patterns. Fish consumption is a function of the seasonality of a species and the dominance of one species over another. Weather and water conditions influence the extent of fishing and the amount of available fish. Dietary or economic restrictions of individuals could also increase the rate of fish consumption by excluding other sources of protein from the diet.

In this study, the high-risk population is defined as those individuals who use fish from the St. Johns River as a primary source of protein. In some instances, the target population purchases fresh fish from local vendors. In most cases, however, the high-risk individuals fish regularly and consume their own catch. Therefore, the individuals at highest risk are those who consume the greatest quantities of fish from the St. Johns.

Methods

To determine the high-risk population, interviews of local fishermen were conducted from September 1983 through May 1984 at 15 sites along the lower St. Johns River (Figure 1). The main objective of the survey was to determine the number of pounds of fresh fish, by species, that were consumed per person per week in a given household. Other objectives included determination of the extent of fishing in a given region, the distance traveled to
Figure 1. Map of the St. Johns River showing the 15 interview sites.

Location
1. Bert Maxwell Park
2. South Main Street
3. North Main Street
4. Broward River Bridge
5. Dunns Creek Bridge
6. Dunns Creek Bulkhead
7. JEA Northside Gen. Station
8. San Carlos Creek Bridge
9. Blount Island Old Bridge
10. Browns Creek Bridge
11. Clapboard Creek Bridge
12. Milton Drive
13. Sisters Creek Bridge
14. Haulover Creek Bridge
15. Huguenot Park
the survey site, occupation and age bracket of the individual, and the uses of the fish caught. Collections of organisms were taken from the river for analysis of mercury levels in their edible portions. The species targeted for sampling are listed in Table 2.
Table 2. Species sampled for mercury contamination and criteria for selection of each species.

<table>
<thead>
<tr>
<th>SPECIES</th>
<th>REASON FOR SELECTION</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cynoscion nebulosus (Spotted Sea Trout)</td>
<td>8th highest commercial tonnage; Carnivore - feeds on smaller fish and shrimp; potentially high bioaccumulation; significant human consumption.</td>
</tr>
<tr>
<td>Mugil cephalus (Striped Mullet)</td>
<td>Highest commercial tonnage; Omnivore - feeds on detritus and deposits; significant human consumption.</td>
</tr>
<tr>
<td>Micropogon undulatus (Croaker)</td>
<td>20% of spring harvest; Carnivores - feed on small fish and shrimp; use estuary as nursery; pool dwellers; significant human consumption.</td>
</tr>
<tr>
<td>Leiostomus xanthurus (Spot)</td>
<td></td>
</tr>
<tr>
<td>Callinectes sapidus (Blue Crab)</td>
<td>Year round resident crustacean; Omnivore - scavenger - feeds on plant and animal material; significant human consumption from commercial and recreational sources.</td>
</tr>
</tbody>
</table>

Each sample consisted of approximately 20 to 30 relatively mature individuals of each species. Immediately after capture, each individual was wrapped in aluminum foil, placed on ice, and later frozen in the laboratory until they could be processed for analysis. The samples were sent to the U.S. Geological Survey (U.S.G.S.) Central Laboratory where the edible portions of each organism were removed and homogenized into bulk samples for mercury determination.

Results

A total of 807 interviews were conducted at the 15 sites on the north side of the St. Johns River. The participant profile shown in Table 3 summarizes the results of the interviews. The
distribution of meals per week is shown in the histogram in Figure 2. Meals ranged from one per month to one per day.

The interview analysis in Table 4 shows for each location: the number of trips made to that site, the number of interviews conducted, the number of fishermen present and the number of repeat interviews. These data were used in estimating the total fishing population of 7,162 for the study sites. This calculated total fishing population value was based on capture-recapture principles. The initial capture is represented by all of the fishermen interviewed less those questioned during the last sample since those individuals cannot be reinterviewed (807-26). The recapture effect is seen in the total number of fishermen which were interviewed an additional time (88). The fishermen reinterviewed represent the dilution factor in the population, hence, the number interviewed less the last sample multiplied by the total number interviewed divided by the number of repeat interviews (807/88) represents the total fishing population (Figure 3.)

An analysis of fish caught (Table 5) shows the distribution, by species, at each site. The dominant fish reported was croaker (Micropogon undulatus), with catfish (Ictalurus sp.), spot (Leiostomus xanthurus) and spotted seatrout (Cynoscion nebulosus) also being taken in relatively large numbers.

Mercury analysis (Tables 6 and 7) showed a worst-case of 1.00 ug/g in catfish. The reported worst-case for water quality in the St. Johns River was 0.50 ug/l (U.S.G.S. data).
Table 3. Profile of individuals interviewed, and meals per week at which river fish/shellfish were consumed.

<table>
<thead>
<tr>
<th>PARTICIPANT PROFILE</th>
<th>Average</th>
<th>Median</th>
<th>Range</th>
</tr>
</thead>
<tbody>
<tr>
<td>Age</td>
<td>40.20 ± 12.20</td>
<td>36.00</td>
<td>12.00 - 79.00</td>
</tr>
<tr>
<td>Family Size</td>
<td>3.40 ± 1.50</td>
<td>3.00</td>
<td>1.00 - 10.00</td>
</tr>
<tr>
<td>No. of Meals/Week</td>
<td>1.40 ± 0.69</td>
<td>1.00</td>
<td>0.25 - 7.00</td>
</tr>
</tbody>
</table>
Figure 2. A comparison of the population interviewed and the number of meals of fish consumed per week.
Table 4. Distribution analysis of interview information.

<table>
<thead>
<tr>
<th>Location</th>
<th>No. of Trips</th>
<th>No. of Interviews</th>
<th>No. of Fishermen</th>
<th>No. of Repeat Interviews</th>
</tr>
</thead>
<tbody>
<tr>
<td>1. Bert Maxwell Park</td>
<td>20</td>
<td>42</td>
<td>60</td>
<td>2</td>
</tr>
<tr>
<td>2. S. Main Street</td>
<td>27</td>
<td>14</td>
<td>68</td>
<td>0</td>
</tr>
<tr>
<td>3. N. Main Street</td>
<td>29</td>
<td>108</td>
<td>125</td>
<td>19</td>
</tr>
<tr>
<td>4. Broward River Bridge</td>
<td>35</td>
<td>28</td>
<td>46</td>
<td>2</td>
</tr>
<tr>
<td>5. Dunns Creek Bridge</td>
<td>34</td>
<td>48</td>
<td>65</td>
<td>2</td>
</tr>
<tr>
<td>6. Dunns Creek Bulkhead</td>
<td>16</td>
<td>8</td>
<td>28</td>
<td>0</td>
</tr>
<tr>
<td>7. JEA Northside Generating Station</td>
<td>18</td>
<td>7</td>
<td>31</td>
<td>0</td>
</tr>
<tr>
<td>8. San Carlos Creek Bridge</td>
<td>28</td>
<td>7</td>
<td>22</td>
<td>1</td>
</tr>
<tr>
<td>9. Blount Island Old Bridge</td>
<td>46</td>
<td>305</td>
<td>444</td>
<td>36</td>
</tr>
<tr>
<td>10. Browns Creek Bridge</td>
<td>36</td>
<td>121</td>
<td>141</td>
<td>20</td>
</tr>
<tr>
<td>11. Clapboard Creek Bridge</td>
<td>29</td>
<td>45</td>
<td>64</td>
<td>5</td>
</tr>
<tr>
<td>12. Milton Drive</td>
<td>7</td>
<td>7</td>
<td>22</td>
<td>0</td>
</tr>
<tr>
<td>13. Sisters Creek Bridge</td>
<td>26</td>
<td>51</td>
<td>73</td>
<td>0</td>
</tr>
<tr>
<td>14. Haulover Creek Bridge</td>
<td>20</td>
<td>8</td>
<td>6</td>
<td>1</td>
</tr>
<tr>
<td>15. Huguenot Park</td>
<td>8</td>
<td>8</td>
<td>8</td>
<td>0</td>
</tr>
<tr>
<td><strong>TOTAL</strong></td>
<td><strong>379</strong></td>
<td><strong>807</strong></td>
<td><strong>1,203</strong></td>
<td><strong>88</strong></td>
</tr>
</tbody>
</table>
Figure 3. Fishing population estimate.

\[
\text{Population} = \frac{(\text{No. Interviewed}) (\text{No. Interviewed}-\text{last Sample})}{(\text{No. of Repeat Interviews})}
\]

\[
= \frac{(807)}{88} (807-26)
\]

\[
= 7,162
\]
Table 5. The distribution of species caught at each of the 15 interview sites.

<table>
<thead>
<tr>
<th>Organism</th>
<th>1</th>
<th>2</th>
<th>3</th>
<th>4</th>
<th>5</th>
<th>6</th>
<th>7</th>
<th>8</th>
<th>9</th>
<th>10</th>
<th>11</th>
<th>12</th>
<th>13</th>
<th>14</th>
<th>15</th>
<th>TOTAL</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bass</td>
<td>31</td>
<td>0</td>
<td>103</td>
<td>26</td>
<td>38</td>
<td>7</td>
<td>1</td>
<td>2</td>
<td>110</td>
<td>143</td>
<td>28</td>
<td>35</td>
<td>12</td>
<td>16</td>
<td>10</td>
<td>562</td>
</tr>
<tr>
<td>Blue</td>
<td>4</td>
<td>0</td>
<td>6</td>
<td>0</td>
<td>9</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>44</td>
<td>80</td>
<td>6</td>
<td>2</td>
<td>17</td>
<td>11</td>
<td>2</td>
<td>181</td>
</tr>
<tr>
<td>Brim</td>
<td>5</td>
<td>0</td>
<td>61</td>
<td>12</td>
<td>6</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>65</td>
<td>58</td>
<td>60</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>267</td>
</tr>
<tr>
<td>Catfish</td>
<td>46</td>
<td>25</td>
<td>204</td>
<td>74</td>
<td>77</td>
<td>10</td>
<td>5</td>
<td>8</td>
<td>988</td>
<td>241</td>
<td>86</td>
<td>5</td>
<td>172</td>
<td>17</td>
<td>10</td>
<td>1,968</td>
</tr>
<tr>
<td>Croaker</td>
<td>165</td>
<td>44</td>
<td>665</td>
<td>95</td>
<td>183</td>
<td>16</td>
<td>10</td>
<td>7</td>
<td>2518</td>
<td>941</td>
<td>367</td>
<td>0</td>
<td>192</td>
<td>0</td>
<td>0</td>
<td>23,320</td>
</tr>
<tr>
<td>Drum</td>
<td>4</td>
<td>3</td>
<td>33</td>
<td>10</td>
<td>16</td>
<td>7</td>
<td>0</td>
<td>0</td>
<td>98</td>
<td>60</td>
<td>13</td>
<td>5</td>
<td>20</td>
<td>1</td>
<td>0</td>
<td>270</td>
</tr>
<tr>
<td>Flounder</td>
<td>13</td>
<td>1</td>
<td>44</td>
<td>0</td>
<td>9</td>
<td>1</td>
<td>100</td>
<td>0</td>
<td>23</td>
<td>40</td>
<td>8</td>
<td>22</td>
<td>0</td>
<td>1</td>
<td>4</td>
<td>266</td>
</tr>
<tr>
<td>Mullett</td>
<td>0</td>
<td>12</td>
<td>0</td>
<td>13</td>
<td>1</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>7</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>33</td>
</tr>
<tr>
<td>Sheepshead</td>
<td>1</td>
<td>1</td>
<td>82</td>
<td>4</td>
<td>3</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>41</td>
<td>97</td>
<td>7</td>
<td>2</td>
<td>3</td>
<td>0</td>
<td>0</td>
<td>241</td>
</tr>
<tr>
<td>Spot</td>
<td>8</td>
<td>18</td>
<td>98</td>
<td>19</td>
<td>85</td>
<td>10</td>
<td>1</td>
<td>0</td>
<td>676</td>
<td>562</td>
<td>219</td>
<td>0</td>
<td>139</td>
<td>0</td>
<td>0</td>
<td>1,835</td>
</tr>
<tr>
<td>Spotted Trout</td>
<td>16</td>
<td>2</td>
<td>69</td>
<td>30</td>
<td>75</td>
<td>1</td>
<td>0</td>
<td>8</td>
<td>151</td>
<td>359</td>
<td>75</td>
<td>30</td>
<td>23</td>
<td>26</td>
<td>54</td>
<td>1,705</td>
</tr>
<tr>
<td>Whiting</td>
<td>0</td>
<td>3</td>
<td>121</td>
<td>1</td>
<td>11</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>245</td>
<td>37</td>
<td>30</td>
<td>2</td>
<td>36</td>
<td>0</td>
<td>0</td>
<td>486</td>
</tr>
<tr>
<td>Yellow Mouth Trt.</td>
<td>9</td>
<td>3</td>
<td>55</td>
<td>16</td>
<td>6</td>
<td>6</td>
<td>0</td>
<td>1</td>
<td>72</td>
<td>31</td>
<td>18</td>
<td>20</td>
<td>19</td>
<td>0</td>
<td>0</td>
<td>256</td>
</tr>
<tr>
<td>Yellow Tail</td>
<td>0</td>
<td>6</td>
<td>81</td>
<td>16</td>
<td>31</td>
<td>15</td>
<td>0</td>
<td>4</td>
<td>90</td>
<td>97</td>
<td>79</td>
<td>0</td>
<td>309</td>
<td>0</td>
<td>0</td>
<td>728</td>
</tr>
<tr>
<td>Crabs</td>
<td>98</td>
<td>6</td>
<td>155</td>
<td>12</td>
<td>35</td>
<td>0</td>
<td>88</td>
<td>0</td>
<td>243</td>
<td>150</td>
<td>14</td>
<td>0</td>
<td>15</td>
<td>15</td>
<td>0</td>
<td>831</td>
</tr>
<tr>
<td>Unidentified Spp.</td>
<td>2</td>
<td>4</td>
<td>144</td>
<td>25</td>
<td>19</td>
<td>4</td>
<td>8</td>
<td>7</td>
<td>75</td>
<td>76</td>
<td>20</td>
<td>18</td>
<td>20</td>
<td>1</td>
<td>1</td>
<td>424</td>
</tr>
</tbody>
</table>
Table 6. 1983 Mercury concentrations in prominent species.

<table>
<thead>
<tr>
<th></th>
<th>Maximum Concentration (ug/g)</th>
<th>Average Concentration (ug/g)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Croaker</td>
<td>4</td>
<td>&lt;0.01</td>
</tr>
<tr>
<td>Spot</td>
<td>51</td>
<td>&lt;0.01</td>
</tr>
<tr>
<td>Yellow Trout</td>
<td>11</td>
<td>0.14</td>
</tr>
<tr>
<td>Spotted Trout</td>
<td>5</td>
<td>&lt;0.01</td>
</tr>
<tr>
<td>Crab</td>
<td>11</td>
<td>0.03</td>
</tr>
</tbody>
</table>
Table 7. 1984 Mercury concentrations in prominent species.

<table>
<thead>
<tr>
<th>Species</th>
<th>No.</th>
<th>Maximum Concentration (ug/g)</th>
<th>Average Concentration (ug/g)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Blue Crab</td>
<td>116</td>
<td>0.23</td>
<td>0.12</td>
</tr>
<tr>
<td>Jacks</td>
<td>17</td>
<td>0.76</td>
<td>0.46</td>
</tr>
<tr>
<td>Yellow Trout</td>
<td>19</td>
<td>0.37</td>
<td>0.17</td>
</tr>
<tr>
<td>Spotted Trout</td>
<td>13</td>
<td>0.23</td>
<td>0.13</td>
</tr>
<tr>
<td>Catfish</td>
<td>20</td>
<td>1.00</td>
<td>0.52</td>
</tr>
<tr>
<td>Bluefish</td>
<td>3</td>
<td>0.57</td>
<td>0.50</td>
</tr>
<tr>
<td>Flounder</td>
<td>2</td>
<td>0.10</td>
<td>0.09</td>
</tr>
<tr>
<td>Spot</td>
<td>83</td>
<td>0.36</td>
<td>0.10</td>
</tr>
<tr>
<td>Red Bass</td>
<td>2</td>
<td>0.24</td>
<td>0.19</td>
</tr>
<tr>
<td>Mullet</td>
<td>27</td>
<td>0.31</td>
<td>0.21</td>
</tr>
</tbody>
</table>
**Discussion**

Mercury poisoning in humans may result from ingestion of contaminated aquatic organisms. The maximum safe limit of total mercury in water from which aquatic organisms are caught and eaten was determined to be 0.10 ug/l (Sittig 1980). Mercury levels of 0.50 ug/l were found in water samples from the St. Johns River. The U.S. Food and Drug Administration (U.S.F.D.A.) standard for mercury in fish flesh is 1.00 ug/g (Sittig 1980); the St. Johns fish have reached this level with a maximum concentration of 1.00 ug/g in catfish and an average concentration of 0.18 ug/g for all species analyzed.

The worst-case contamination, calculated from test and interview results, was ingestion of 200 ug Hg/day (Figure 4.C). This figure is based on the highest mercury content found in collected fish and the maximum amount of fish consumed per person per day as determined by interviews. The potential worst-case is based on water contamination, potential bioaccumulation, and maximum fish consumption per day which results in a value of 620 ug/day (Figure 4.D). The documented Lowest Observable Effect Level (LOEL) ranges from 200 to 500 ug/day (Sittig 1980).

Although the mercury concentration in some areas of the St. Johns exceeds the Florida D.E.R. limit of 0.10 ug/l, only one species of the fish tested has met the F.D.A. standard of 1.00 ug/g. Yet, the observed worst-case contamination of 200 ug Hg/day for individuals in this study is within the LOEL range. The possibility also exists for an individual to regularly consume 620 ug of Hg/day based on the potential bioaccumulation factor of 6,200 and water quality data (maximum of 0.50 ugHg/l) (Figure 4.D). These levels of consumption introduce the potential for adverse health effects in humans.

The current mercury levels in the St. Johns suggest complications for individuals consuming fish from the river; therefore monitoring should continue. This point may be illustrated by the apparent increase in mercury concentrations in the fish from 1983 to 1984 (Tables 6 and 7). The possibility of the mercury levels dissipating over time exists only if no further contamination occurs. This is unlikely because of natural weathering processes and industrial activities. Continual efforts should be made to avoid additional mercury contamination of the river.
Figure 4. Bioaccumulation and contamination calculations based on the 1984 experimental data.

A. Observed Bioaccumulation Factor = \[
\frac{\text{Mercury Concentration in Fish}}{\text{Mercury Concentration in Water}}
\]

\[
= \frac{1.00 \text{ ug/g}}{0.5 \text{ ug/l}}
= \frac{1,000 \text{ ug/kg}}{0.5 \text{ ug/l}}
= 2,000
\]

B. Potential Bioaccumulation Factor:

Published Range = 900 - 63,000 (Sittig 1980)
Adopted Average = 6,200

C. Observed Worst-Case Contamination = (Highest Mercury Concentration in Fish) (Highest Fish Consumption/day)

\[
= (1.00 \text{ ug/g}) (200 \text{ g/day})
= 200 \text{ ug/day}
\]

D. Potential Worst Case Contamination = Water Concentration (Potential Bioaccumulation Factor \times Highest Fish Consumption/day)

\[
= 0.5 \text{ ug/l} (6,200 \times 0.2 \text{kg/day})
= 620 \text{ ug/day}
\]

*The bioaccumulation factor for fish and shellfish has an estimated value of 6,200 (Sittig 1980).*
BIBLIOGRAPHY


COLONIZATION AND UTILIZATION OF A CONCRETE ARTIFICIAL REEF OFF THE GEORGIA COAST BY INVERTEBRATES AND FISHES

Matthew R. Gilligan

Introduction

The efficacy of using artificial reefs to attract fish and as a fishery management tool is well known (Bohnsack and Sutherland 1985), especially in Japan where they are intensively and extensively used in commercial fisheries (Vik 1982). In the United States, they have been developed by all levels of government and private organizations principally to enhance recreational and sport fishing (Steimle and Stone 1973, Artificial Reef Development Center 1985). Artificial reefs have been used as a mitigation measure to offset the loss of marine habitats by coastal development (Bair and Feigenbaum 1984), and methods to measure the economic value of artificial reefs in the U.S. have been proposed (Sport Fishing Institute 1985).

Artificial reefs serve to attract and concentrate fishes by 1) functioning as orienting devices for adult midwater and migratory schooling fishes (e.g. jacks, mackerels, barracudas, Atlantic spadefish, cobia), 2) providing sites for colonization of the planktonic larvae for reef fishes (e.g. snappers, groupers, tropicales) and, 3) providing habitats for resident reef fishes. The spatial heterogeneity of reefs in general and reef height diversity in particular (Molles 1978) seem to contribute to higher reef-fish species diversity while higher productivity of some target reef fish species (e.g. black sea bass) seems to be a result of low reef profile and wide spatial distribution of reefs (Feigenbaum 1984). Floating structures suspended in the water column (Fish Attracting Devices of FADs) attract baitfish schools and larger pelagic fishes offshore, but improvement of fishing nearshore (near fishing piers) has not been unequivocally shown (Murray et al. 1985).

It is evident to both researchers and fishermen that artificial reefs increase finfish productivity when placed in reef-poor localities. In 1983, no less than seven Georgia saltwater gamefish records were fishes taken at artificial reefs in Georgia (Savannah Sport Fishing Club 1984). It has not been shown, however, that artificial reefs actually increase the net production of the shelf. In other words, would the primary productivity which ends up as reef fish biomass have ended up as just as much fish biomasses in a benthic or pelagic ecosystem? If it does, then it would be improper to say that artificial reefs increase shelf production. Davis et al. (1982) showed that artificial reefs can change the composition and density of

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adjacent benthic communities up to 200 m away. Testing a shelf-wide hypothesis may be nearly impossible.

It is also difficult to compare the production of fishes on natural reefs to that of artificial reefs. Natural reefs show wide geographically and topographically-related variation in fish community structure (Hiatt and Strasburg 1960, Jones and Chase 1975, Gilligan 1980, Thomson and Gilligan 1983). Comparisons of artificial to natural reefs have been made and, in general, artificial reefs seem to attract higher densities of fish than natural reefs. However, most of these studies (see Bohnsack and Sutherland 1985) have not standardized, quantified, or replicated, reef structural complexity and distribution, factors that may significantly affect their utilization by fishes and have not addressed questions of production, productivity, and reef fish community structure.

For the most part, Georgia's broad sandy shelf is reef-poor. The only significant natural reefs (also called live bottoms) are located 35 miles offshore in the 30–40 m isobath range (e.g. Snapper Banks) and 18 miles offshore in the 18–21 m isobath range (e.g. Gray's Reef National Marine Sanctuary). Since 1970, the Coastal Resources Division of the Georgia Department of Natural Resources and local sport fishing organizations have undertaken the deployment of artificial reefs in the near-shelf zone (<20 m depth) to enhance recreational and sport fishing resources of the coast. To date, the effort totals 130,000 scrap automobile tires and 14 surplus vessels ranging in length from 33 to 444 feet at eight buoy-marked locations along the coast. The selection of materials to be used in these reefs was usually based upon material and transport availability (donations), and considerations regarding the persistence of the material once in place (will it move or break up causing a hazard/nuisance to shipping or commercial fishing?; Ansley, pers. comm. 1983).

On March 31, 1983 through efforts of the Savannah Sport Fishing Club, Inc. and other contributing organization and individuals, 407 tons of surplus precast concrete culvert and pipe material comprising 249 pieces was offloaded from a barge in the area of an already established artificial reef (reef 'L'; buoy at 31° 45.4 N., 80° 36.5 W.) establishing Georgia's first concrete reef. The purpose of the effort was to enhance recreational and sport fishing.

In view of 1) increasing coastal populations and demand for fishing resources, 2) the increasing role of private sport fishing groups in artificial reef development and 3) recent federal legislation designed to make it easier for private organizations to build fishing reefs, there is a need for information on artificial reef type effectiveness and persistence in the South Atlantic Bight. This report summarizes an effort to develop techniques for monitoring the colonization of the reef by invertebrates and fishes.
Methods

The concrete material used for the reef consisted of 249 commercial, iron-reinforced precast culvert and pipe sections. The cylindrical and box-like structures were up to 2.4 m in longest dimension with one or two open sides and smaller openings. Some smaller pieces and rubble were created incidental to the loading and dumping process.

The material was unloaded from an unanchored barge over an area of approximately 100 by 200 meters (depth 18-20 m) in the vicinity (100 m at 120-130 degrees magnetic from the maintenance buoy) of the Georgia Department of Natural Resources artificial reef 'L' which is located 20 nmi (nautical miles) ESE of Wassaw Island, Georgia (Figure 1). Materials already at the site, which was established in March 1977, include two steel barges, 2,000 tire units and a 150-foot dredge.

Monthly sampling, visual observation and photography programs were conducted for five months after the material was dumped (table 1). Each month, during one 50-minute SCUBA dive at the reef site (depth 18-20m), small, loose pieces of exposed concrete with marine growth were collected, a rapid visual assessment of fishes was done, and close-up underwater photographs of concrete surfaces were taken. Following the five monthly visits, close-up photography and visual assessment was done three times over the next 13 months.

Concrete samples were returned to the laboratory where they were analyzed for the size, density and diversity of encrusting growth. Each rapid visual assessment of fishes was done near at least three large concrete pieces and recorded on water-resistant paper and underwater clipboard. No predetermined swimming pattern, transect, or search pattern was followed for the visual assessment. Each species seen on or around the concrete and estimates of the average sizes of individuals were recorded. Underwater close-up photography of outside, inside and edges of concrete structures was done using a Nikonos IV-A 35 mm camera, SB-101 electronic flash, 1:3 close-up extension tube, 35 mm lens and either Kodachrome 64 or Ektachrome 64 (ASA 64) transparency film. Water temperature, salinity and an estimate of water clarity (visibility) were also noted during each trip to the reef.

Results

Invertebrates

Colonization of the concrete surfaces by barnacles, bryozoan colonies, serpulid worms, hydroid colonies, algae, ascidian colonies, anemones and sponges was rapid. Barnacle settlement and growth was most dramatic during the first five months. Underwater photographs showed nearly 100 percent cover of concrete surfaces by barnacles on some concrete surfaces after only
one month. Densities and barnacle sizes are given in Table 1. Maximum barnacle densities were observed on outside concrete surfaces particularly near upper edges of the structures while inside surfaces remained relatively clean (<0.01 barnacles/cm2).

After the first month, hydroid colonies had grown to 1 cm in height, and the calcareous tubes of serpulid worms were detected on concrete surfaces. After the second month, the serpulid worms were well established and two kinds of bryozoans (flat colonies in calcified radiating rows and branched stalk colonies) were established. After the third month, some of the flat bryozoan colonies had overgrown serpulids and barnacles or met other colonies in a stand-off, ascidian colonies had become established, and many surfaces were still dominated by barnacles. After the fourth and fifth months, the increasingly complex encrusted surfaces had acquired anemones, filamentous red algae, crustaceans (amphipods and decapods), gastropod mollusks, sponges, octocorals, and asteroidian echinoderms.

**Fishes**

Results of the visual observations of fishes are summarized in Table 2. All 20 of the fish species seen on or near the concrete during the study are common near natural and artificial reefs of the inner and midshelf of the South Atlantic Bight (Gilligan ms).

Adult black sea bass were common in and around the concrete after only one month. Most of the fishes utilizing the reef during the study period were adults of resident reef species that have planktonic (drifting) eggs and larvae rather than those that deposit their eggs on the reef (e.g. blennies, gobies, damsel-fishes). Small juveniles of the black and rock sea bass, sand perch, scup, gray triggerfish, squirrelfish, and whitespotted soapfish were observed at the reef indicating that not all colonization was by the migration of adults from nearby established reefs. Pelagic fishes (e.g. jacks, mackerels, cobia) were not easily observable due to poor visibility (<8 m) during most of the visits.

Characteristic of inner and midshelf reefs of the Georgia Coast, the density and diversity of fishes decreased dramatically during the winter months (data not included).

**Discussion**

Higher densities and growth of barnacles on the outer surfaces and edges of the concrete structures suggests that either the rate of planktonic propagule (egg or larva) arrival is higher in these areas or that, once settled, feeding rates, hence growth rates were higher in these areas. Since this area of the shelf is characterized by significant currents and since propagule and food-bearing water currents near exposed surfaces would likely be higher than near protected inner reef surfaces, this might
explain both higher immigration and faster growth on exposed surfaces. The barnacle data shows that colonization is a constant process since small size barnacles (newer colonists) were present in all areas throughout the study. Growth inhibition is not suspected because mean barnacle size was not significantly lower in high density area.

The physical appearance of the encrusting growth persisted while fish diversity declined during the winter. Close-up underwater-photographs revealed that the growth and colonial proliferation among colonizing sessile species, which began within months of placement, were characterized by inter- and intra-specific competition for space resulting in a mosaic of encrusting growth. More importantly, the added dimensionality and topographic complexity of the reef created by the encrusting growth (e.g. barnacles, bryozoan colonies, serpulid worms, hydroid colonies, ascidian colonies, anemones and sponges) appeared to permit a succession of other organisms to establish themselves on the reef (e.g. small mollusks, crustaceans, annelids, echinoderms and fishes). The resulting wider resource spectrum of the reef. (i.e. food and microhabitats) could, in turn, further increase the species diversity since, in general, a wide resource spectrum in a habitat permits more species to coexist.

Lowered fish diversity and abundance at the reef during the winter is most likely the result of low water temperature. Winter sea temperatures at this site, which is near the inner/midshelf margin, average less than 15° C (Atkinson, pers. comm.). These temperatures are near if not below the lethal thermal minimum of many tropical species which are common on natural and artificial reefs in this area during the summer and fall.

It appeared that the shape of the concrete structures, while large and stable, did not initially provide the smaller holes and interstices necessary as refuge for smaller fishes especially the post larvae and juveniles of reef species. During the first months, juveniles and smaller reef fishes were normally seen only in broken concrete rubble. It appeared that appropriate refuges for juvenile reef fishes were in short supply on most of the structures during this early period. This was particularly evident on occasions during visual observations when juvenile reef fishes (e.g. whitespotted soapfish) were observed lying flat against partially encrusted concrete surfaces—no other cover (e.g. crevice or hole) being available.

Dumping of the 249 concrete units individually from an unanchored barge resulted in a scattered pattern on the bottom, although some aggregations of 8 to 13 units each were found. The inability to survey the entire reef prohibited making overall assessments of fish population at the reef. Poor visibility during visits to the reef necessarily decreases the accuracy/repeatability of visual assessment of fishes. The data from the visual assessment must also be viewed in light of a significant
but unquantified harvest of fishes from the reef by recreational fishermen, divers, and commercial trap fishing. Nonetheless, the methods developed and employed in this study may be useful in future efforts to identify the important structural and ecological factors that determine the productivity of artificial reefs of the South Atlantic Bight.

Biological studies of artificial reefs coupled with oceanographic, hydrographic, and climatological information are necessary to predict and understand the processes that regulate the productivity of both artificial and natural reefs. Improved quantitative assessment of artificial reefs, reef communities and biotic changes that occur on them are necessary to effectively evaluate artificial reef type effectiveness (Bohnsack and Sutherland 1985). The information derived from artificial reef studies will be directly or indirectly useful to those who develop, manage, and use them.

This study was funded in part by Savannah Sport Fishing Club, Inc. Savannah, Georgia.
Table 1. Summary of barnacle growth and density on the concrete artificial reef at the 'L' buoy during the first four months after placement. md = mean barnacle density in individuals per cm² in the most dense areas of barnacle growth, mxd = maximum barnacle density in individuals per cm², ms = mean barnacle size in millimeters (longest basal dimension), mxs = maximum barnacle size, and mns = minimum barnacle size.

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<tr>
<th>Date</th>
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<td>6.84</td>
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Table 2. Results of visual observations fishes at the concrete artificial reef at 'L' buoy during five monthly visits to the reef after its establishment. j=juveniles, a=adults, ( ) = estimated size or size range of individuals in centimeters.

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<tbody>
<tr>
<td>1. Centropristis striata</td>
<td>a(30)</td>
<td>j(4-5)</td>
<td>j(5-6)</td>
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<tr>
<td>Black sea bass</td>
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<tr>
<td>2. Haemulon Aurolineatum</td>
<td>a(18)</td>
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<td>Tomtate</td>
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<td>3. Stenotomus caprinus</td>
<td>a(12)</td>
<td>a</td>
<td>j(4-5)</td>
<td>a(15-20)</td>
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<tr>
<td>Longspine porgy</td>
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<td>4. Decapterus punctatus</td>
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<td>Round scad</td>
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<td>5. Hemipteronotus novacula</td>
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<tr>
<td>Pearly razorfish</td>
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<td>6. Centropristis philadelphica</td>
<td>a</td>
<td>j(6-7)</td>
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<tr>
<td>Rock sea bass</td>
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<td>7. Urophysis earli</td>
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<td>Caolina hake</td>
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<td>8. Chaetodipterus faber</td>
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<tr>
<td>Atlantic spadefish</td>
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<td>9. Archosargus probatocephalus</td>
<td>a(35)</td>
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<tr>
<td>Sheephead</td>
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<td>10. Balistes capriscus</td>
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<td>Gray triggerfish</td>
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<td>11. Diplodus holbrooki</td>
<td>a(15-20)</td>
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<td>Spottail pinfish</td>
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<td>12. Rypicus maculatus</td>
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<td>(6-7)</td>
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<tr>
<td>Whitespotted soapfish</td>
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<td>13. Holocentrus sp.</td>
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<td>Squirrelfish</td>
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<td>14. Calamus leucosteus</td>
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<tr>
<td>Whitebone porgy</td>
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<td>15. Halichoeres bivittatus</td>
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<td>a</td>
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<tr>
<td>Slippery dink</td>
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Table 2 continued:

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<td>sand perch</td>
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<td><strong>17.</strong> <em>Stenomus chrysops</em></td>
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<td><strong>18.</strong> <em>Hypeurochilus geminatus</em></td>
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<td>crested blenny</td>
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<td><strong>19.</strong> <em>Monocanthus hispidus</em></td>
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<td>planehead filefish</td>
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<td><strong>20.</strong> <em>Sphyraena barracuda</em></td>
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<tr>
<td>great barracuda</td>
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Randall, J.E. 1963. An analysis of the fish population of arti-


Figure 1.

Map of the Georgia coast showing the location of Georgia Department of Natural Resources Artificial Reef 'L' at 31 45.4N and 80 36.5 W near the 20m isobath.
ONTGENETIC AND SEASONAL PATTERNS OF OSMOTIC AND IONIC REGULATION
AND THE ENERGETIC COST OF A VARIABLE SALINITY ENVIRONMENT
IN THE STRIPED MULLET, MUGIL CEPHALUS L.

Frank G. Nordlie

Introductions

No teleost fish group more fully typifies the euryhaline
condition than the mullets (family Mugilidae) which are ubiqui-
tous in coastal marine and brackish waters, and frequently in
connected fresh waters, throughout temperate, subtropical and
tropical zones of the world (Nelson 1976). One of the most
abundant of these fishes and the best known in this part of the
world is the striped mullet, Mugil cephalus L. The present paper
will explore some of the environmental problems facing this
species through its life history in coastal waters and salt
marshes of southeastern United States.

Based on information contained in reviews of Breder and
Rosen (1966), DeSilva (1980), and supported by the extensive work
of Kilby (1949, 1955), Mugil cephalus populations spawn offshore
in marine waters where the zygotes develop and hatch. The devel-
oping larvae/juveniles migrate toward shore, and the successful
individuals move into estuarine and tide-marsh channels and some
ultimately into the shallow pools of tide marshes. Here they
continue to develop for a varying portion of their first year of
life, following which they may move to deeper channels of the
tidal system, go into fresh water tributaries, or return to sea
for varying periods (including spawning migrations after sexual
maturation). The factors determining the directions to be taken
remain unknown. Obviously a species with such a complex life
history faces extreme variability in several key environmental
conditions in its life, but especially in ambient salinity and
temperature. This paper, while not pretending to be all-
inclusive, will explore some aspects of physiological responses
to these environmental conditions.

Since the goal of this symposium is to assess potential
environmental impacts of coastal development in Northeastern
Florida and Southeastern Georgia, Mugil cephalus is certainly a
species whose environment stands to be altered by such cultural
activities.

Department of Zoology, University of Florida, Gainesville,
Florida 32611
Methods

Ontogenesis of Osmotic Regulatory Capacity

Juvenile individuals of Mugil cephalus were obtained by seining in Gulf coast salt marshes near Cedar Key, Florida, through the winter and spring months. They were returned to the laboratory in Gainesville, and subjected, in laboratory acclimations, to a series of experimental salinities ranging from fresh water to full sea water, at an ambient temperature of 22 ± 2°C and on a photoperiod of 12L:12D. All acclimations were continued for a period of 14 days before individuals were transferred to the next salinity in the acclimation series or until blood sampling was carried out.

Osmotic concentrations were determined using a Wescor model 5100B vapor pressure osmometer on 5 Hl plasma samples. Even so, it was generally necessary to pool plasma from two or three individuals in order to obtain the necessary 5 Hl of plasma. Blood was taken by blind sternal puncture using heparinized (ammonium heparinate) microhematocrit tubes that had been drawn out to fine tips and broken off to the appropriate diameter. The procedures were described by Nordlie et al. (1982). Lengths are expressed as standard lengths (SL).

Osmotic Regulation as Influenced by Environmental Temperature

Groups of individuals averaging ca. 18 gm in body mass (roughly 10 cm SL) were collected, using a cast net, from salt marshes adjacent to the Matanzas River near the Whitney Marine Laboratory on the Florida east coast. These individuals were returned to the laboratory in Gainesville, and subjected to laboratory acclimations. Groups of individuals were acclimated to a series of ambient temperatures, controlling the ambient salinity (34.5 ± 1°/oo) and the photoperiod (12L:12D). Again, all acclimations to a given set of conditions extended for a period of 14 days. Plasma samples were taken from individuals of each acclimation group following a 14-day acclimation to the ultimate acclimation temperature (starting from 20°C and working in 5°C alterations upward or downward until the desired acclimation was achieved. Blood was taken from the caudal artery by severing the caudal peduncle and allowing arterial blood to flow into heparinized (ammonium heparinate) collecting tubes. Plasma osmotic concentrations were determined on 200 Hl samples pooled from the several individuals in an acclimation group. Determinations of osmotic concentrations were made on a Precision Systems osmometer. Other aspects of the techniques were described in Nordlie (1976).

Respiratory Demands as Influenced by Variations in Environmental Salinities

Individuals ranging in body mass from 10 to 37 gm were used in these studies and were captured by means of seines and cast
nets from Gulf coast salt marshes near Cedar Key, Florida. They were returned to the laboratory in Gainesville and subjected to controlled acclimations. Ambient temperature was maintained at 25 ± 1°C with a photoperiod of 12L:12D. Individuals were acclimated to the test salinities for a period of 8 days before metabolic determinations were carried out. Rates of oxygen consumption of intact individuals were measured in closed systems, using opaqued Erlenmeyer flasks of appropriate volumes as respiration vessels and following the decrease in O₂ pressures in the vessels. Determinations of O₂ pressure were made using a Radiometer pO₂ electrode. The techniques were applied as described by Nordlie and Leffler (1975).

Individuals utilized in all parts of the work were fed Tetramin R and all statistical procedures utilized followed Sokal and Rohlf (1969).

Results

Osmotic Regulatory Capability as a Function of Body Size

Determinations of osmotic regulatory capacity of juvenile Mugil cephalus were carried out over a range of sizes from 25 mm to 105 mm (SL) after laboratory acclimation to a series of ambient salinities. This study was carried out to determine whether or not the smallest (youngest) individuals found in the salt marshes showed any qualitative differences in osmotic regulatory characteristics from individuals of progressively larger sizes. Also, the salinity tolerances of these individuals were investigated over the ambient range from fresh water to normal sea water (Nordlie et al. 1982). Results of these analyses are represented in Fig. 1 (taken from Nordlie et al., 1982). First, it was found that the smaller juvenile mullet (25 and 35 mm SL groups) did not survive the acclimation to fresh water, so their osmotic regulatory characteristics are represented only for salinities of 5% sea water through 100% sea water. Individuals of larger sizes (55 mm SL and larger) survived acclimation over the entire salinity range from fresh water through full sea water. Second, it was also determined that the overall osmotic regulatory capability gradually improved with development to a length of ca. 55 mm. Beyond this size, the responses at a series of ambient salinities from fresh water to full sea water are roughly those of small (and presumably larger adult individuals of the species (>105mm) (Nordlie et al. 1982).

Osmotic Regulation as Influenced by Environmental Temperature Variations

The range of ambient temperature encountered by individuals of this species among the various habitats potentially occupied by developing individuals or adults in a year’s time in coastal waters of this part of the world can easily extend from a low of 10°C to a high of 30°C. The pattern of responses in plasma osmotic concentrations to a series of environmental temperatures
Figure 1.

Plasma osmotic concentrations (mOsm/Kg) as functions of ambient salinities (% of full sea water) for juvenile Mugil cephalus of a series of lengths (SL in mm) (From Nordlie et al., 1982).
within this range in an environment of normal sea water was investigated in laboratory acclimation studies involving juveniles of a relatively uniform size (ca. 18 gm--10 cm SL). Results of this analysis are found in Table 1. It can be seen from these data that there are significant alterations in the plasma osmotic and ionic concentrations with alterations in ambient temperature.

The data in Table 1 show that there were consistent reductions in the concentrations of Na⁺ and Cl⁻ and in the total plasma osmotic concentration as the acclimation temperature was sequentially reduced in 5°C steps from a high of 30°C to a low of 10°C. One can also look at these alterations in the opposite way, following the increases in Na⁺ and Cl⁻ and plasma osmotic concentrations with increases in acclimation temperature. No significant alteration was found in K⁺ concentration with alterations in ambient temperature.

Respiratory Demands as Influenced by Variations in Environmental Salinities

As respiratory oxygen demands of organisms are altered through variations in ambient temperatures, they may also be altered by variation in ambient salinities and by temperature x salinity interactions. There have been relatively few studies of any complexity attempted to determine possible implications of an euryhaline mode of existence on respiratory oxygen demands in any teleost species. Some of the host of questions that can be asked about such interactions were posed in reference to Mugil cephalus (Nordlie and Leffler 1975). In that study we assumed that at least one likely pathway by which metabolic demands of the organism might be altered as salinity varied was through the ionic and osmotic regulatory processes. A summary of the thermodynamically minimal energy demands of the ionic transport process is as follows (Lehninger 1971):

$$G^O = RT \ln \frac{C_2}{C_1}$$

where R is the molar gas constant (1.984 cal/°-mol), T is the absolute temperature, and C₂ and C₁ are the molar concentrations of a specific ion being transported from compartment 1 to compartment 2.

We hypothesized that in the ideal compensation system this cost would be minimal where environmental and blood concentrations were equal and would rise progressively in more and less dilute environments (that is, where there are progressively greater differences between environmental and blood ionic and/or osmotic concentrations.) This is what we found to be the case in Mugil cephalus with respect to the overall osmotic concentration as can be seen from Figure 2. However, this is by no means a universal pattern of response in respiratory oxygen demands to
Table 1

Ionic and Osmotic Concentrations of Mullet Plasma as Functions of Acclimation Temperatures
(From Nordlie, 1976)

<table>
<thead>
<tr>
<th>Acclimation Temperature (°C)</th>
<th>Plasma Ion Concentrations</th>
<th>Plasma Osmotic Concentration mOsm/kg</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Na⁺</td>
<td>K⁺</td>
</tr>
<tr>
<td></td>
<td>X</td>
<td>S.E.</td>
</tr>
<tr>
<td>10</td>
<td>158.8</td>
<td>10.7</td>
</tr>
<tr>
<td>15</td>
<td>164.2</td>
<td>7.7</td>
</tr>
<tr>
<td>20</td>
<td>166.9</td>
<td>10.3</td>
</tr>
<tr>
<td>25</td>
<td>176.3</td>
<td>6.8</td>
</tr>
<tr>
<td>30</td>
<td>178.0</td>
<td>10.5</td>
</tr>
</tbody>
</table>

N = sample size
Figure 2.

Oxygen uptake (ml of O$_2$/gram/hour) as functions of ambient salinity (mOsm/l) in juvenile Mugil cephalus acclimated to the test salinities (25 $\pm$ 1°C) (redrawn from Nordlie and Leffler, 1975).
altered ambient salinities as was later pointed out by Nordlie (1978). Based on further work and a search through the literature I suggested that there are at least four basic patterns of response in respiratory oxygen demands to variations in ambient salinity as seen in Table 2.

Discussion

Neither the zygotes nor newly-hatched fry of Mugil cephalus are truly euryhaline. Sylvester et al. (1975) demonstrated that the optimal salinity for development of embryos of striped mullet (incubated at 19.5-20.5°C) was 30-32°C/oo, with less than 50% survival at salinities below ca. 29°C/oo. No upper salinity limit was set in that work. Lee and Menu (1981) looked at both ends of the salinity spectrum and demonstrated better than 50% hatching success at salinities ranging from 15 to 55°C/oo. However, the optimal salinity range (22-22.5°C) was found to be from 30 to 40°C/oo with the peak at 35°C/oo. While there was good hatching success outside of this range, many anatomical anomalies were noted in the developing embryos.

Sylvester et al. (1975) found that the optimal salinity range for the survival of newly-hatched larvae was lower than that for egg development, at 26-28°C/oo. The range of salinities within which there was 50% or greater larval survival was roughly from 25 to 35°C/oo. This was interpreted to indicate an adaptation to an estuarine environment in the larvae.

Obviously salinity tolerances of M. cephalus extend over a wider range (at least toward the more dilute end of the scale) in the older and larger juvenile stages evaluated here (>25 mm SL). Kilby (1949) reported that the smallest juveniles that he took in Gulf shore waters were of lengths of 16 mm, or appreciably smaller than the smallest individuals used in this study. It is not yet known when in the developmental sequence the extensions of salinity tolerances and improvements in osmotic regulatory capacity begin to develop. From the results cited here (from Nordlie et al. 1982), it is obvious that these capacities are at least not in their definitive state by the time the juveniles enter brackish waters. Certainly it appears that individuals of small sizes (<55 mm) are vulnerable to very low salinities. It is suggested here that the juveniles migrating in salt water and newly-arrived in brackish waters are most vulnerable to sudden reductions in salinity and this vulnerability is reduced as the individuals develop. By the time they have reached a length of ca. 55 mm they achieved salinity tolerances and osmotic regulatory capacities (at least under the experimental environmental conditions investigated) roughly similar to those of small adults. Since it is known that larger individuals of this species can tolerate salinities in excess of 2X that of sea water (McFarland 1965; Gunter 1967), and that individuals of ca. 10-15 cm SL are already capable of tolerating salinities of at least 1.6 X sea water (ca. 55°C/oo) (Nordlie and Leffler 1975), this
Table 2
METABOLIC RESPONSES TO ALTERED SALINITY IN TELEOST FISHES
(After Nordlie, 1978)

<table>
<thead>
<tr>
<th>Response Type</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>I.</td>
<td>The rate of oxygen consumption is not significantly altered over a wide range of environmental salinities (freshwater to normal sea water).</td>
</tr>
<tr>
<td>II.</td>
<td>The rate of oxygen consumption is minimal at an ambient salinity that is isosmotic with the blood. Rates of oxygen consumption increase at lower and higher salinities.</td>
</tr>
<tr>
<td>III.</td>
<td>The rate of oxygen consumption is minimal at the optimal salinity for the form and is at other than the isosmotic level. Again, rates of oxygen consumption increase as functions of the osmotic gradient in more and less saline waters.</td>
</tr>
<tr>
<td>IV.</td>
<td>Rates of oxygen consumption are maximal at the optimal salinity for the form and rates of oxygen consumption are depressed at salinities higher and lower than the optimal level.</td>
</tr>
</tbody>
</table>
tolerance of hypersaline waters must also develop relatively early. Salinities in excess of those of normal sea water are frequently produced by evaporation in the shallow salt marshes at low tide during hot, dry weather or on a longer term basis in more or less isolated lagoons. Again, we predict that the migrating and newly-arrived juveniles are potentially most vulnerable to the stresses at increased salinities.

A few workers have focused their attention over the years on the influences of environmental temperature on osmotic regulatory capacities of various species of ectotherms. Wikgren (1953) published one of the early classics in this area demonstrating that salinity tolerances were altered as osmotic regulatory ability of a species was influenced by environmental temperature. Prosser, Mackay and Kato (1970) generalized from their, and other, findings that plasma inorganic electrolyte concentrations decrease with increasing ambient temperatures and increase with decreasing ambient temperatures in marine fishes (or in euryhaline fishes in a marine environment). The reciprocal was said to be true of freshwater fishes. The basic assumption here was that at colder temperatures, the ionic/osmotic regulatory systems functioned less well than at higher temperatures and this differences between internal and external concentrations were reduced. We have ignored here the accumulation of organic "anti-freeze" substances in the bloods of fishes that occupy environments reaching subzero temperatures (e.g. Scholander et al., 1957; Umminger, 1969b). Umminger (1971) demonstrated that there was somewhat more variability in total osmotic response patterns, at least of freshwater fishes (or of euryhaline fishes in a freshwater environment) than Prosser, Mackay and Kato (1970) had suggested. However, he agreed that their conclusions with respect to the plasma inorganic electrolytes were generally correct. An unexpected result of the present work was to discover that Mugil cephalus does not respond to temperature in its plasma osmotic concentration and inorganic electrolytes as one would be led to expect of a euryhaline teleost in marine waters. Other euryhaline forms such as Fundulus heteroclitus and F. grandis (Umminger 1969a, 1970) do show expected response patterns both in fresh and in marine waters. Nordlie speculated (Nordlie 1976) that the response pattern observed in M. cephalus might, in fact, represent an adaptation (or stress response), not to reduced, but rather, to increased environmental temperatures. At high environmental temperatures saturation levels of dissolved oxygen are reduced and ambient levels would be expected to be even lower in tide marshes where there is a significant accumulation of suspended and deposited organic material. Under such potentially stressful conditions, a partial relaxation of osmotic and ionic regulation should ease the metabolic burden on the organism and thus reduce the oxygen demands in an oxygen-poor environment. While this hypothesis has not been tested directly, the data and conclusions of Cech and Wohlschlag (1982) provide indirect support. They demonstrated adjustments in respiratory, ventilatory and hematological patterns in Mugil cephalus correlated with the
problems of obtaining respiratory oxygen during the warmest part of the summer and at the onset of the fall spawning migration.

In as yet unpublished work, Nordlie found in a preliminary investigation that ionic and osmotic concentrations in Mugil cephalus acclimated to hard freshwater were not significantly altered at acclimation temperatures ranging from 30°C to 10°C. Tentative conclusions are that, while Mugil cephalus is obviously capable of tolerating a wider range of ambient temperatures than that utilized in the present study (10°C-30°C), it demonstrates a stress response within this range, to increased temperatures in marine waters but not in fresh waters. It seems logical that the ambient temperature ranges from 10°C on down toward 0°C, and from 30 to 35 or 40°C should also be investigated. Whatever additional information these extensions might provide, it is already clear that Mugil cephalus is influenced in its osmotic/ionic regulatory capacity by environmental temperature.

It has been well documented that a contributing cause of death in fishes subjected to rapid change in temperature (upward and/or downward) is the disruptive influence of such temperature changes on the osmotic/ionic regulatory processes of the individuals (Wikgren 1953; Heinicke and Houston 1965; Stanley and Colby 1971). One expects, based on the lesser salinity tolerances of smaller Mugil cephalus that these smaller individuals would be more stressed by abnormal temperatures or salinity temperature combinations than would larger individuals.

There are some data in a study by Carr and Giesel (1975) on effects of high temperature conditions in coastal salt marshes (Jacksonville area) that serve as a nursery area for a number of marine forms, including Mugil cephalus, and also serve as receiving waters for the effluent waters from the cooling systems of a nuclear power plant. They found that while there was an obvious reduction in the utilization, by juveniles of most teleost species, of creeks receiving such heated waters, that juvenile mullets (here M. cephalus and M. curema) remained roughly as abundant as in a control area, an unaffected creek. One expects that it is not such constantly extreme conditions (at least within limits) that seriously disrupt regulatory processes, but rather changes in temperature. However, a shutdown of the cooling systems and thus of warm water discharge during the colder period of the year might have devastating effects on the juvenile fishes inhabiting such affected areas. It would be expected that individuals of M. cephalus would be more vulnerable to the effects of elevated temperatures and less to those of reduced temperatures than would those of M. curema. This conclusion is based on the work of Moore (1976) who concluded that M. cephalus is more of a temperate-subtropical form and M. curema is more of a tropical form based on a comparison of ranges of temperature tolerances and of geographical distributions of the two forms. Moore agreed with the conclusion of Liao, et al. (1972) and Nash and Kuo (1975) that the optimal range of temperature, including for spawning, in M. cephalus was roughly 20 to 24°C. Obviously
the tolerable ranges, for at least some life stages, extend well beyond these limits. However, it is likely that normal development, and especially the development of reproductive readiness and its ultimate success, will occur only within fairly narrow ranges of environmental temperature and salinity and their combinations.

The possible impacts of altered environmental conditions, other than of temperature influences, singly or in concert, on metabolic demands of organisms, have been largely ignored, perhaps mainly because of the difficulty of conducting such studies even in the laboratory. The work summarized here on Mugil cephalus (Nordlie and Leffler 1975) shows this species to have lowest respiratory oxygen demands (and, we assume lowest total metabolic demands) at ambient salinities roughly equivalent to the blood plasma concentration (at 25 ± 1 °C). Metabolic demands then increase in either direction from this salinity. Thus, we suggested that such a species commits least energy to the demands of maintenance of osmotic/ionic concentrations in this ambient zone. This would allow greater energy commitments to growth and reproduction at such salinities. This hypothesis was supported by findings of DeSilva and Perera (1976) who found juvenile Mugil cephalus to have lower food intakes, greater growth rates and higher conversion efficiencies at brackish salinities. However, while food intake was least and conversion efficiency was highest at an ambient salinity of 10⁰/oo, highest growth rate in their study was attained at an ambient salinity of 20⁰/oo, but only with an unlimited food supply. On a restricted ration only the individuals in an ambient salinity of 10⁰/oo survived. An ambient salinity of 10⁰/oo is roughly equivalent to the average plasma osmotic concentration of this species. The results of a limited investigation by Marais (1978) of the influences of a series of ambient temperatures on the routine metabolic rate of juvenile Mugil cephalus at salinities of 1⁰/oo and 35⁰/oo showed results similar to ours with respect to salinity effects for this limited salinity series. He also showed that metabolic rates increased with temperature increases from 13 to 23°C in both 1⁰/oo and 35⁰/oo media, but at ambient temperatures of 28 and 33°C, metabolic rates were depressed at both environmental salinities. This again suggests that Mugil cephalus, while tolerating high ambient temperatures, shows stress responses at ambient temperatures well below the maximum tolerable levels. These data again support the hypothesis that the increased osmotic/ionic concentrations at elevated temperatures represent an adaptation in Mugil cephalus to reduce metabolic demands at such temperatures.

Speculating on the possible impacts of altered environmental conditions on species (Table 2) showing other types of responses in metabolic rates to altered salinities leads to the conclusion that all forms obviously have limits within which they can respond to salinity alterations and that most species show some response in metabolic demands to salinity alterations. I have
found that these various types of responses are at least corre-
lated with the ionic (and osmotic?) regulatory patterns of the
various species studied as they respond to altered salinities.
Every species, be it a teleost fish, alga, crustacean, or bac-
terium, has a set of environmental tolerances for each stage in
its life history. In some instances the tolerable ranges of
environmental conditions do not vary much over the life-time of
the individual. However, for some forms there are dramatic
ontogenetic alterations in certain tolerance patterns and thus in
the need of predictability of environmental conditions of certain
habitats for survival and/or successful completion of a life
cycle. Mugil cephalus is a species showing the latter pattern of
ontogenetic variations in environmental tolerances accompanying
ontogenetic shifts in habitats and thus in requirements for
predictable ranges of conditions—here temperature and salinity—
in the specific habitats utilized at various times in the
developmental sequence. It seems reasonable to conclude from the
information now available that the most environmentally sensitive
stages in the life cycle of this species are in embryonic and
larval development through the time that the juveniles reach a
size (or age equivalent) of ca. 55 mm SL. Various workers have
also demonstrated that another period of special sensitivity to
environmental conditions is that of the development of reproduc-
tive readiness. While adult individuals may tolerate wide ranges
in salinity and/or temperature conditions, the ranges known to
allow the successful development of gametes are very restricted.
Also information from the literature suggests that rapid changes
in environmental temperature/salinity conditions may be more
destructive than more gradual chances (within the tolerable
ranges).

While a fair amount is known about environmental require-
ments of Mugil cephalus, perhaps because of the actual and
potential economic importance of this species, much yet remains
to be done. Also most species that have little obvious or direct
economic value have been largely ignored, despite their possible
trophic importance in coastal and/or estuarine ecosystems. We
must make a plea to carry out studies of environmental effects of
individual species and their ecosystems long before proposals for
alterations of specific ecosystems are to be evaluated. Only
then can there be time to make systematic and detailed analyses
that will provide predictive power, useful in the assessment of
possible biological alterations that might be produced by such
cultural alterations.

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Bibliography


