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Copper Accumulation by Gizzard Shad (Dorosoma cepedianum) in the Charleston Side Channel Reservoir

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Copper Accumulation by Gizzard Shad (*Dorosoma cepedianum*)

in the Charleston Side Channel Reservoir

(TITLE)

BY

Mark T. Christ

THESIS

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ABSTRACT

Christ, Mark T. M.S., Eastern Illinois University. July 1990. Copper Accumulation by Gizzard Shad (*Dorosoma cepedianum*) in the Charleston Side Channel Reservoir.

Exposure to trace heavy metals is being extensively studied in relation to their adverse and toxic effects on organisms. Processes affecting compartmentalization, biogeochemical cycling, and transport of trace metals in ecological systems must be elucidated if the impact of trace metal releases on biota are to be understood. Because elemental cycles are often closely linked to hydrologic processes, freshwater and marine systems serve as both vectors and sinks for trace metals in the environment. The regular addition of a heavy metal into an aquatic ecosystem will undoubtedly increase its persistance for availability to the existing biota. Once in the environment, the contaminant may be bioaccumulated so that the concentration within the tissues of an organism is greater than that in the environment. Biomagnification may result when a consumer further concentrates the contaminant with resulting increases in "body burden" at successive trophic levels.

The Charleston Side Channel Reservoir is a 346-acre impoundment in East Central Illinois which serves as the City of Charleston's water supply. Unfavorable taste and odor problems have occasionally occurred since construction in 1981, presumably due to algal blooms. The reservoir has been treated with one to two thousand pounds of copper sulfate each summer as an algicide. The impoundment is also used for sport fishing, prompting concern about possible copper accumulation in fish in that reservoir. Gizzard shad (*Dorosoma cepedianum*) were chosen for this study to determine

relative condition and copper accumulation in whole body portions. This species is greatest in abundance in the reservoir and undergoes ontogenetic diet shifts, consuming detritus at some period in its life cycle. Copper is known to accumulate in detrital sediments, and thus should be directly passed on to detritivorous organisms.

Gizzard shad were collected from the Charleston Side Channel Reservoir and Lake Shelbyville and processed for physical and chemical measurements. Copper accumulation in whole fish portions was determined by atomic absorption spectroscopy. Measurements of whole body heavy metal content are critical to the study of biomagnification because predators consume entire prey, not selected organs. Furthermore, concentrations for whole organisms are required for investigations of trace metal compartmentalization and flux in populations and ecosystems.

No correlation between copper concentration and coefficient of condition for 0- and 1-yr fish from Charleston was observed. Although no significant difference was found for the 0-yr coefficient of condition between Charleston and Shelbyville shad, there was a highly significant difference between the coefficient of condition of 0- and 1-yr Charleston fish. It appears that 1-yr fish are in poorer condition than 0-yr fish. There was no significant difference in copper concentration between 0- and 1-yr fish from Charleston, but there is a significant difference in copper concentration for 0-yr fish from Shelbyville and Charleston. Shelbyville fish appear to have a greater copper concentration even though copper sulfate is not used in this impoundment. Since there is no correlation between coefficient of condition and copper concentration in Charleston gizzard shad, copper appears to have no deleterious effect in these fish at this time.

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INTRODUCTION

Environmental contaminants such as heavy metals that are introduced into aquatic environments are readily available for uptake by invertebrate and vertebrate organisms through the water column, food, and sediments. Metal accumulation in the gills, organs and tissues of these organisms can produce adverse physiological effects depending on the amount and rate of uptake. The most direct approach to estimating bioavailability is measurement of contaminant uptake by aquatic organisms exposed to the sediments of concern. Detrital feeding fish would therefore be expected to contain increased contaminant levels by direct sediment consumption.

A food chain in an aquatic ecosystem is one of the most important transfer routes of heavy metals by aquatic organisms. Piscivore fish being at the highest trophic level are the most likely recipients of biomagnification. These fish may have major roles in elemental compartmentalization in certain aquatic systems, and because of their importance in the human diet, can comprise a significant pathway for transport of toxic substances to man.

The Charleston Side Channel Reservoir is a 346-acre impoundment located in Coles County of east-central Illinois (Appendix C). The reservoir serves as a retention basin and municipal water supply for water pumped from the Embarras River. Since construction in 1981, algal blooms have frequently occurred contributing to the hypereutrophic state of the reservoir presumably influencing taste and odor. Reservoir management has included copper sulfate treatments and the addition of an aeration unit. Copper sulfate has been added annually to the reservoir since 1981. Dosage rates range from 1000 lbs. per treatment in 1983 to 2000 lbs. per treatment in 1989. Citric acid is added as a chelating agent at the rate of 1000 lbs. per treatment. The aeration unit was installed in 1988 near the raw water intake, an area having the greatest depth hence most susceptible to anoxic conditions. The average depth of the reservoir is 8.4 ft. with a maximum depth of 16 ft. The pH ranges from 8.1-8.9; the average hardness (as $CaCO_3$) is 173 ppm; and the average total alkalinity is 122 ppm.

The impoundment is used for sportfishing, prompting concern about a possible biomagnification route of copper to piscivores by consuming forage fish. Copper is known to accumulate in detrital sediments, and thus should be directly passed on to detritivorous organisms (Allison, 1989). Gizzard shad (*Dorosoma cepedianum*) were chosen as the test species for three reasons: 1) they are the most abundant fish in the reservoir (Dufford, 1988); 2) young gizzard shad are planktivorous feeders which undergo an ontogenetic diet shift to detrital feeders; and 3) this species is the main forage fish in the reservoir whereby contaminants may biomagnify to higher piscivorous fish.

Lake Shelbyville is an impoundment in which no copper sulfate is applied and was chosen as a control. Gizzard shad were compared between Lake Charleston and Lake Shelbyville in relation to relative condition and copper concentration in whole body portions.

LITERATURE REVIEW

The gizzard shad *(Dorosoma cepedianum)* is in the order Clupeiformes, family Clupeidae. Its native habitat once extended from Minnesota to the St. Lawrence River and from New Jersey to the Gulf of Mexico but introduction as a forage fish has spread the gizzard shad throughout the southwestern states as well. It resides predominantly in freshwater but may be found in estuaries and brackish water bays (Smith, 1979).

The gizzard shad spawns in April, May, and June. Females lay several thousand eggs which hatch in three days at 25°C (Smith, 1979). Larval shad are cylindrical with a terminal mouth which contains teeth in both jaws. Juvenile shad feed on zooplankton until they reach about 25mm long, when they lose their teeth and become detritus feeders. The first growing season is rapid, the juvenile reaching about 100mm in length by the end of this period. Adulthood occurs in the second year with average life expectancy at 5-6 years (Smith, 1979).

Usually the most abundant forage fish in reservoirs, gizzard shad are well adapted for detritivore feeding with a gizzard-like stomach, long intestine, and long, fine gill rakers for straining plankton (Smith, 1979). The majority of research on this species over the past 10 years has focused on feeding habits and growth with minimal accounts of toxicity and morphology.

Feeding Habits of the Gizzard Shad

Heinrichs (1982) described the digestive anatomy of young planktivorous gizzard shad and development of the digestive system through the adult stage. The mouth changes from a supra-terminal to a sub-terminal position to facilitate benthic feeding and the pharynx is modified for straining microscopic particles. Also, pharyngeal organs possessing both goblet cells emerge, intestinal length increases and taste buds develop, and esophageal and gastric glands appear. The gizzard acquires a thick muscular wall which facilitates its presumed function as a crushing organ in the adult.

Food habits were determined for juvenile gizzard shad from open-water and near shore habitats in Melvern Reservoir, Kansas during mid-July of 1984 (Todd and Willis, 1985). In the open-water habitat, filamentous algae steadily increased in importance for shad from 16-40mm in length. In addition, diatoms were taken by 16-35mm fish, but decreased in importance as length increased. Sand grains were absent in all shad from this habitat. In the near shore habitat, the portion of the diet composed of filamentous algae generally decreased as the length of the shad increased. Diatoms and cladocerans never comprised more than 5% of the diet for any length group. Organic detritus steadily increased in importance for fish from 21-75mm in length.

Algal-clay flocculation is a potentially important mechanism directly linking smaller primary producers and filter feeding fish. McIntyre (1983) induced flocculation by adding bentonite clay to a tank of water containing algae. Gizzard shad placed in the tank collected twenty four times as much food material as in a tank where flocculation was not induced. No significant differences were found between gill raker spacings of the gill arches between fish in the control and clay treated tanks. Plankton for both treatments were examined for differences in size. *Lagerheima* sp., *Keratella* sp., and

Astramoeba sp. were dominant in both with *Lagerheima* sp. being the most abundant organism. *Lagerheima* sp. represents a major potential food source, but because of its small size it probably could not be efficiently collected by the fish without flocculation. The other species were large enough to be collected as individuals.

Mummert and Drenner (1986) developed a model for predicting the filtering efficiency of different sizes of gizzard shad. The model is based on the cumulative frequencies of interraker distances of the gill rakers. This was demonstrated by feeding trials in which shad ingested different sizes of suspended plastic microspheres and plankton. The proportion of particles removed by fish increased as a function of particle size. The same relationship was observed by Drenner et al. (1982) and Drenner and Mummert (1984) in which ingestion rate increased as a function of particle size. According to selectivity index values computed from the filtering efficiency model, shad of 5, 15, and 25cm standard length would selectively feed on particles larger than 19, 40, and 63um, respectively. The growth-related changes in filtering efficiency and selective particle ingestion may explain why phytoplankton declines in the diet of gizzard shad and why its feeding niche is shifted as length increases.

Mundahl and Wissing (1987) found that gizzard shad ingested foods of varying nutritional quality during the 1981-1983 growing seasons in Acton Lake, Ohio. Adult fish (ages 2-4) were fed on a mixed diet of zooplankton and organic detritus in the early summer growing season and on detrital materials during the remainder of the growing season. Age-0 fish (<35mm) ingested only detrital materials.

As condition factors of ages-3 and 4 gizzard shad improved in mid-summer, muscle C:N ratios of these fish also increased, suggesting a direct relationship between lipid depostion in muscles of adult shad. A similar relationship was not evident in age-0 fish, suggesting that ingestion of poor-qualtiy detritus can reduce the growth and condition of gizzard shad, whereas seasonal inclusion of high-quality zooplankton in the diet can result in rapid growth and improved condition.

Seasonal changes in body dry weight and caloric and lipid contents were studied in ages-0 and 1 and ages- 1 and 2 shad in 1973 and 1974, respectively (Pierce and Wissing, 1980). Data on body caloric and lipid contents indicated a period of rapid storage in late summer and early autumn. There was a decline in body dry weight, body calories, and body lipids during the winter months. The intensity of the fattening process was greater in age-1 fish than in ages-0 and 2 fish. Strange and Pelton (1987) also found that in general, fat content increased through summer, peaked during fall, and declined during winter.

Salvatore et al. (1987) conducted a study to evaluate the influence of water temperature on food evacuation and feeding activity in age-0 gizzard shad fed pulverized food. Fish kept at 5°C exhibited little feeding activity. At temperatures of 10, 15, and 20°C, food was evacuated through the digestive tract in 8.5, 4.9, and 4.4 hrs., respectively. Frequency of feeding acts also increased significantly with increased water temperatures. It is apparent from this study that small, age-0 shad can ingest and process substantial amounts of detritus daily.

It is evident from the above studies that gizzard shad undergo digestive morphological developments into the adult stage to facilitate detrital feeding. Consequently, ingestion of poor-quality detritus can reduce growth and condition although substantial amounts are consumed. Detrital contaminants would therefore be directly consumed through the alimentary tract with bioaccumulation occurring in the organs and tissues.

Gizzard Shad as a Forage Fish

Concern for forage fishes in impoundments developed quickly in response to the proliferation of ponds and reservoirs which occurred about 40 years ago, but research has been slow to produce results applicable to effective fisheries management. The key problem in forage fish management for ponds and reservoirs is to maintain adequate stocks of prey within size ranges vulnerable to predators. In spite of its major contribution to food of predator species, the gizzard shad remains problematic due to its propensity to develop a high biomass of large individuals with the potential to interfere with production of other species. In southern reservoirs, young gizzard shad frequently are the principal prey, but they reach lengths of over 400mm and rapidly grow too large to be eaten by most predators (Noble, 1981). Storck (1986) investigated the changes in food habits of largemouth bass compared with seasonal and annual changes in size of gizzard shad in Lake Shelbyville. The stomachs of 5,283 largemouth bass were examined during spring, summer, and fall from 1978 to 1981. Gizzard shad was the most important species in the diet of age-1 and older bass in all years. However, the age and size composition and the volume contributed by this species varied substantially among years and seasons. Gizzard shad were only important to the diet of age-0 largemouth bass in 1981. This was due to gizzard shad growing more slowly that year resulting in a larger fraction remaining vulnerable to predation.

As a forage fish, gizzard shad prey availability to piscivores depends on growth rate. Slow growing shad remain vulnerable to young and older piscivores whereas faster growing shad would be only vulnerable to older piscivores. Biomagnification in piscivores from consuming shad would appear to have the greatest effect on older piscivores because older gizzard shad would be available for consumption resulting in a larger increase of contaminant uptake through shad tissue.

Trophic Interactions

Recent research on manipulating trophic interactions among aquatic organisms has suggested a "natural" means of controlling phytoplankton by the addition of piscivorous fish. Cascading trophic interactions is the effect whereby a rise in piscivore biomass brings decreased zooplanktivore biomass, increased herbivore biomass, and decreased phytoplankton biomass. Potential productivity at all trophic levels is set by nutrient supply (Carpenter et al., 1985). Consequently, when piscivores consume zooplankton dependent fish the zooplankton population increases thereby consuming increasingly more phytoplankton (Carpenter, 1988). Short-term enclosure experiments have demonstrated this strategy but the full scope and dynamics of these implications are presently unknown in whole-lake ecosystems and will be forthcoming.

Gizzard shad may be having an indirect effect on phytoplankton stimulation. Lammarra (1975) and Brabrand et al. (1990) demonstrated that soluble phosphorus was significantly increased by benthophagous fish presumably due to detritus digestion, and was readily available for algal growth. The high density of gizzard shad in Lake Charleston may be effecting internal nutrient loading, and thus contributing to the hypereutrophic state of the reservoir.

A recent control measure implemented at Lake Charleston in 1988 and 1989 involved stocking non-vulnerable, piscivorous hybrid-striped bass to feed on the shad population. This strategy will hopefully in the future have one or both of the above mentioned effects in improving the reservoirs trophic state resulting in reduction or elimination of copper sulfate treatment. Elimination of copper sulfate will reduce the further concentration of copper in the sediment, thereby reducing its availability to gizzard shad and other aquatic organisms.

Copper Accumulation in Sediments

Several experiments have demonstrated that copper concentrates in the sediment fraction of aquatic environments. Jackson (1977) found a good positive correlation between the organic carbon content and the copper (also Hg, Zn, Cd, and Fe) concentration in sediments from Clay Lake and Ball Lake (Ontario). Also, Loring (1976) found that most of the copper (79 to 86%), and also zinc and lead, in the sediments of the Saquenay Fjord was found in the detrital fraction (acid soluble). Strong, positive correlations were found between the metal (Cu, Zn, Pb, both detrital and non-detrital) and the mud (<53um) contents of the sediment.

The treatment of a California aqueduct with copper sulfate (as an algal control) at a rate of approximately 823 kg/km over a 4-yr. period did not lead to an increase in the copper concentration in the water. The mean concentration of copper in the <20um sediment fraction in the untreated reach was 0.0298 mg/kg compared to 0.0597 mg/kg in the treated reach (Fuller and Averett, 1975). In a similar study, Symmes (1975) showed that 98% of the total copper input to Indian Lake (Massachusetts) (consisting of 90% of soluble copper from copper sulfate treatment) was retained in the system, most of it (98%) in the sediments and the rest in the water column as soluble copper or as suspended sediments. The conversion of the soluble copper to particulate forms and their deposition on the lake bottom was approximately 90% completed by the 10th day.

Cline and Upchurch (1973) showed that copper in lake sediments has a very significant tendency (exponential increase through time) to migrate towards the upper 5cm of the sediment, possibly by a bacterial mechanism. Diks and Allen (1983) related geochemistry to bioaccumulation by comparing the distribution of copper in the sediment and the water column of four freshwater systems to the amount of copper available for bioaccumulation. Data determined that there was an increase in copper uptake by tubificid worms as a function of time in the sediment-water system as compared to the water column.

The preceeding studies demonstrate that the copper ion readily accumulates in detrital sediments rather than remaining suspended in the water column. Thereby enabling detrital feeders to directly ingest copper through uptake of detrital sediments.

Copper Accumulation in Fish

Many studies have revealed that the copper ion is readily concentrated in the organs and tissues of various fish species by uptake through the water column and sediments. Green sunfish were exposed to copper in a pond with 3 mg/l copper as copper sulfate. The aqueous concentration declined as the copper accumulated in the mud and remained there for the duration of the 4-month study. After one week, the fish had accumulated an equilibrium concentration of 3 mg/kg Cu (wet weight) in their tissues which decreased to background levels in 79 days. The copper remained in the fish for 5 weeks longer than in the pond water (McIntosh, 1975).

Northcote et al. (1975) found that copper levels in the muscle of a number of fish species from the Fraser River, British Columbia were generally low, when compared to other areas, and ranged from below the limit of detection to 1.53 mg/kg (wet weight) with an average of 0.54 mg/kg. The concentration of copper in the muscle decreased as the size of the fish increased.

Chronic Copper Toxicity and Sublethal Effects in Fish

The frequent introduction of copper into the aquatic environment and the high degree of susceptibility of certain forms of aquatic life to copper has promoted considerable toxicological research. The absolute copper concentration that induces toxicity or antagonistic physiological responses in fish depends upon characteristics of the water such as hardness, pH, and alkalinity. Copper is highly complexed by carbonate and hydroxide ions in natural waters and this complexation will determine the speciation and concentration of copper ions in solution. Table 1 contains some sublethal and toxic effects of copper to fish species mentioned in this review.

Reproduction

Mount (1969) found that survival, growth, and reproduction of fathead minnows *(Pimephales promelas)* were similar in soft water (30 mg/l as $CaCO_3$) with copper concentrations of 0.0044 to 0.0106 mg/l. Fry from parents exposed to these concentrations or unexposed parents grew equally well. The survival of fry was zero at 0.0184 mg/l Cu, half the adult fish were killed, and the growth and sexual development of the survivors was retarded. In hard water, it appeared that 0.033 to 0.0145 mg/l Cu had similar effects to 0.0184 to 0.0106 mg/l Cu in soft water.

The maximum acceptable toxicant concentration (MATC), by definition, is the highest toxicant concentration that has no adverse effect on survival, growth and reproduction. Sauter et al. (1975) studied the effects of low levels of copper on the hatching success, survival, and growth of brook trout eggs and concluded that the MATC for brook trout in soft water (37.5 mg/l as $CaCO_3$) lay between 0.003 and 0.005 mg/l. The MATC in hard water (187 mg/l as $CaCO_3$) was estimated to be between 0.005 and 0.008 mg/l Cu.

Osmoregulation and Condition

Changes in blood osmolarity in fish exposed to zinc or copper stress were observed by Lewis and Lewis (1976). Freshwater fish displayed lower serum sodium ion, a decrease in blood osmolarity and an increase in wet weight.

Copper was shown to affect the growth, survival, and condition of juvenile coho salmon at a level of 0.03 mg/l. Salmon survival in 0.02 mg/l Cu and less was similar to that of the control fish.

Feeding ceased at the beginning of exposure to 0.02 and 0.03 mg/l; it was resumed after a while but only slowly at 0.03 mg/l Cu, while 0.01 mg/l Cu caused slight inhibition of feeding. The coefficient of condition (Kn) of the fish was the parameter which correlated most closely with copper concentration. The coefficient of condition fell rapidly in 0.03 mg/l Cu for 10 weeks, leveled off, and then increased slightly with resumption of feeding (Lorz and McPherson, 1976).

Respiration and Viral Antigens

Antagonistic physiological responses in fish have been attributed to copper exposure. In bluegill, oxygen consumption rates were determined for juveniles subjected to various copper concentrations. Respiratory responses were postively correlated with copper concentrations up to the 96-hr TLm (median tolerance limit) of 2.4 mg/l of copper. Higher concentrations produced greater increases in respiraton. The concentration that caused no alteration in the rate of oxygen consumption was found to be 0.1 mg/l of copper (O'Hara, 1971). Exposure of blue goumaris to methylmercury, copper, or methylmercury and copper in combination resulted in an appreciable decrease in the production of antibodies against viral and bacterial antigens (Roales and Perlmutter, 1977).

Hematology and Endocrinology

Investigations of the effects of sublethal exposure to copper on various blood parameters using brook trout were carried out by McKim et al. (1970). Changes were related to survival and reproductive success. Short-term physiological changes occurred in the red blood cell count, hematocrit, hemoglobin, plasma chloride, plasma glutamic oxaloacetic transaminase (PGOT), osmolarity, and total protein on exposure to 0.038 to 0.069 mg/l Cu for 21 and 337 days. During the 21-day experiment, a 10% mortality occurred at 0.0692 mg/l and during the 337 day exposure, a 60% mortality occurred at the 0.0325 mg/l level.

A response to stress in fish, the increased secretion of corticosteroids, has been suggested as a possible parameter for a fairly rapid method of detecting the presence of pollutants (Donaldson and Dye, 1975). Elevated levels of serum cortisol were caused by exposure to sublethal levels of copper but not cadmium. The increase in cortisol on exposure to copper was dose dependent. The level returned to more basal levels within 8 to 24 hours, as the fish acclimated to the copper (e.g., 0.015 mg/l Cu). The cortisol levels returned to prestress conditions even though this stress was still present. The cortisol level in fish exposed to 0.06 and 0.09 mg/l Cu increased significantly after 24 hrs. and remained elevated until the end of the experiment.

Previous studies have demonstrated the toxic and adverse effects of copper exposure on various species of fish. Factors such as survival, growth, and reproduction can all be potentially affected in the presence of elevated copper levels. Furthermore, antagonistic physiological responses may be induced where presently no sublethal effects are occurring but may arise in the future with continued copper exposure.

TABLE 1

Sublethal and Toxic Effects of Copper to Fish

Species	Effect	[Cu] (mg Cu/l)	Reference
Fathead Minnow *Pimephales promelas*	Adult & Fry mortality	0.0184	Mount and Stephan (1969)
Brook Trout *Salvelinus fontinalis*	MATC	0.003-0.005	Sauter et. al. (1976)
	10% Mortality	0.0692	McKim et. al. (1970)
	60% Mortality	0.0325	"
Coho Salmon *Oncorhynchus kisutch*	Feeding Inhibited	0.02-0.03	Lewis and Lewis (1976)
Bluegill *Lepomis machrochirus*	Increased Respiration	2.4	O'Hara (1971)

MATERIALS AND METHODS

Collection

Gizzard shad were obtained by cast net on September 14, 1989 from Lake Charleston. Sampling was conducted at Site 1 (Appendix C) and at the north shore of the reservoir. An aeration unit at Site 1 causes shad to maintain a position at the surface of the water column. The north shore is shallow and sparsely populated by emergent aquatic vegetation. Gizzard shad were obtained by electroshocking from Lake Shelbyville on the same day. Specimens were taken by routine shocks at the base of the dam by personnel of the Illinois Natural History Survey. Specimens from both lakes were stored in ice filled coolers during transportation to the laboratory.

Standard Length and Weight

Standard length and weight were determined on the day of capture. Standard length (mm) was measured on a measuring board and weight (g) was measured on a Mettler E200 Balance to the nearest centigram. All specimens were then frozen in sterile 6 ounce Whirl-Pak bags for subsequent analyses.

Age

All specimens were aged by annuli enumeration in sagittae otoliths. Otoliths were removed, cracked laterally, mounted in Crit-O-Sealant and examined using a dissecting microscope (10-40X) and light source equipped with a fiber optic strand.

Relative Condition

The relative condition of each gizzard shad was calculated for both Lake Shelbyville and Lake Charleston fish by the following equation (Moyle and Cech, 1988):

$$Kn = W/aL^{b}$$

Where Kn is the relative condition factor, W is the weight in grams, L is the length in millimeters, and a and b are constants derived from the power equation for the regression of weight on length. For the pooled gizzard shad populations, this equation is:

$$Kn = W/(0.00001119 \times L^{2.9522})$$

A second condition factor, known as relative weight (Wr), was calculated. This is the percentage of an individual gizzard shad weight calculated with a constant length-weight formula computed by the Kansas Department of Conservation (Schonhoff, 1990). The length-weight equation for gizzard shad is:

$$\log Ws = -5.376 + 3.170 \log L$$

The relative weight (Wr) is calculated as follows:

$$Wr = W/Ws \times 100$$

where L is the length in millimeters, W is the weight of an individual in grams, and Ws is a length specific standard weight.

Homogenization

To prevent sediment copper contribution, viscera were removed from all specimens with nitric acid rinsed stainless steel surgical tools (Schmitt and Finger, 1987). Wet weight was again measured on a Mettler E200 Balance. Depending on weight, stainless steel blenders of various sizes were used for homogenization (Schmitt and Finger, 1987). Reagent grade water (Milli-Q system) was filtered through membrane filters (0.45um pore diameter) to eliminate copper contribution by bacteria. Filtered water was then added in a proportion (ml) of three times the wet weight of the fish.

After the homogenate was liquified (usually about 3 minutes), a 10ml portion was pipetted into a nitric acid rinsed crucible. Crucibles were ignited in a muffle furnace at 600°C overnight in an exhaust hood (Clesceri et al., 1985). Homogenate ash was rehydrated in the crucible with 5ml of 6N HNO_3 and filtered through a 25mm glass fiber filter to remove ash residue (Perkin-Elmer, 1982). The liquid was collected and brought to a 10ml final volume with 6N HNO_3 in a calibrated centrifuge tube. The samples were then transferred to and stored in acid rinsed glass screw-cap vials.

Copper Quantitation

All samples were suitable for direct determination on the Perkin Elmer 2300 Atomic Absorption Spectrophotometer. The wavelength selected for copper was 327.5nm and the instrument was equipped with a deuterium background corrector. Approximately 4ml was directly aspirated into an air/acetylene flame. Sample uptake occurred every 0.2 seconds and absorption readings were averaged for 50 replicate burns (Perkin-Elmer, 1982).

Standards were prepared from a stock solution of 100 mg/l (as Cu) cupric sulfate in 6N HNO_3. Standards included 0.0, 0.1, 0.5, 1.0, and 5.0 mg/l. A standard curve was constructed having a correlation coefficient of 0.999. The copper concentration for each sample was estimated from the standard curve.

A reagent blank (6N HNO_3) was read after every five samples and all readings were 0.000 ± 0.0002-0.0004 mg/l. Copper contamination from all equipment used was checked and found to be negligible.

RESULTS AND DISCUSSION

Copper in Sediments and Water Column

Sediment samples taken from Lake Charleston in June 1988 and August 1989 were analyzed for a number of metals including copper (Table 2). Sites 4 and 5, located at the mouths of the ravines feeding into the reservoir (Appendix C), showed a significant increase in copper levels from 1988 to 1989. Site 3 showed no such increase. This lack of increase may be attributed to the bottom composition being largely inorganic (i.e. sand and stone). Organics, in general, are lacking, presumably due to the scouring action of the water being pumped from the river at this site. Results at Site 3 are in accordance with McDuffie et al. (1976) who found that the total copper concentrations in silt and clay were 2.9 to 10.6 times greater, respectively, than the total copper concentration in sand. In the same study, it was found that sand transported only 1% of the "available" copper.

Comparable data for copper concentrations of sediments in Lake Shelbyville are currently unavailable, although, the upper and lower reaches of the Kaskaskia River basin have been found to contain sediment copper. These reaches of the Kaskaskia River basin border Lake Shelbyville to the north and south (Appendix D). For instance, in 1982-1983, 24 sites were sampled for copper sediment load with a resulting range of 4-22 mg/kg and having a mean of 14.75 mg/kg (Kelly et al., 1989). This mean value is considerably lower than the copper sediment concentrations in Lake Charleston. These mean values do not classify these sediments as having "elevated sediment copper" (>38 mg/kg) established by Kelly and Hite for Illinois streams (1984).

The longevity of the copper ion in the water column at Lake Charleston is relatively short. Copper analyses before, immediately after, and 24 hours after copper sulfate application indicate that copper levels recede 90-95% 24 hours after application (Table 3). These results reinforce the fact that copper is precipitating to the bottom sediments at a rapid rate rather than remaining suspended in the water column.

Weight and Length of Gizzard Shad

According to data for Lake Charleston (Table 4), gizzard shad are increasing in weight and length with age. The 0-yr shad from Lake Charleston are heavier and longer than the same age class for Lake Shelbyville. The 0-yr shad from Lake Shelbyville have a much lower mean weight (3.0g) than those from Lake Charleston (11.5g). Although a small sample size exists for 1-yr shad from Lake Shelbyville, they also show a lower mean weight (11.8g) than do Lake Charleston shad (16.4g). It is not until the third growing season (2-yr age class) that the mean weight of Shelbyville gizzard shad (49.1g) exceeds that of Lake Charleston shad (30.0g).

Copper in Gizzard Shad

Linear regressions for copper concentration with weight of gizzard shad for Lake Charleston and Lake Shelbyville are presented in Figures 1 and 2, respectively. Neither lake shows a significant correlation of copper concentration to weight. It appears that the copper concentration in whole body portions is not increasing or decreasing with increasing weight. Cross et al. (1973) suggested that a primary reason for a decrease or no change in metal concentration with size is related to new tissues being incorporated at a greater rate than metals can be actively transported into the tissues to establish a steady-state concentration (dilution by growth). O'Rear (1971) observed a negative correlation for Zn and Cu in small striped bass due to rapid growth which allowed a steady state of metal uptake and elimination from tissues to be established. Furthermore, Cross et al. (1973) indicated Zn and Cu exchange rates between muscle and blood are constant, or decrease, over the life-span of an adult fish.

Discrepancies in bioconcentration patterns observed by researchers may be due to inconsistencies in analyzing either whole body or various tissues. For instance, several studies have revealed that the highest copper levels occur in the liver tissues of both marine and freshwater fish species (Wright, 1976; Wharfe and Van Den Broek, 1977; Cowx, 1982; Richard and Dulley, 1983). The highest levels of copper may occur in the internal tissues, which were discarded in this research, while lower levels, depending on metabolic rate may concentrate in bone and flesh.

Another explanation by Weiner and Giesy (1979) suggested that trace metal concentrations in fish apparently may be related to factors other than food habits. They found that interspecific variation between body size and Cu, Mn, and Zn concentrations was related to homeostatic control of these metals in fish. Copper is required metabolically by living organisms and these fish may be actively regulating concentrations of this essential metal in their tissues.

The assumption that Lake Shelbyville gizzard shad would not be exposed to elevated copper levels and hence would have less copper in their tissues than similar aged fish in Lake Charleston appears to have been incorrect. Figures 3 and 4 show copper concentration within gizzard shad age classes for Lake Charleston and Lake Shelbyville, respectively. Total copper concentrations in Lake Shelbyville gizzard shad are greater than those for Lake Charleston. Copper concentrations are greatest in the 0-yr (n=20) followed by the 2-yr (n=2) and 1-yr (n=14) age classes in Lake Charleston. In Lake Shelbyville, the greatest concentration occurs in the 3-yr (n=7) gizzard shad followed by the 2-yr (n=7), 0-yr (n=10), 4-yr (n=1) and 1-yr (n=3) age classes. In both lakes, the 1-yr age class has the lowest copper concentration. Theoretically, the 1-yr gizzard shad should have a greater copper concentration than the 0-yr shad due to ontogeny occuring in the diet. By age-1, completion of ontogeny in gizzard shad would allow increased copper ingestion through detrital consumption. However, gizzard shad may have already undergone ontogeny during the first growing season (age-0) and since these fish are growing at such a rapid rate the metal is consumed and eliminated before accumulation occurs.

Analysis of Variance and Duncan's multiple range test was used to determine significance differences between copper concentrations in 0-yr and 1-yr Lake Charleston and 0-yr Lake Shelbyville gizzard shad (Table 6). Fish with copper levels below the limit of detection were eliminated from this analysis. Lake Shelbyville 0-yr shad contain significantly more copper than the same age class from Lake Charleston (F=4.90, P=0.013). Furthermore, it appears that the longer exposure to copper by the Charleston 1-yr shad does not cause these fish to accumulate more copper.

A wide variability of copper conentrations exists within all age classes from each lake. Bohn and Fallis (1978) also found great variability of copper concentrations within Arctic char liver and muscle. The mean, standard deviation, and range were 2.4 ± 0.9 mg/kg (1.1-4.2) and 87 ± 56 mg/kg (22-220) for liver and muscle, respectively. They also found no correlation between body weights and copper concentrations in either tissue group. As mentioned earlier, homeostatic control is believed to play a major role in individual distribution of trace heavy metals in tissues and this factor may attribute to the wide variability of copper concentrations in gizzard shad.

Relative Condition of Gizzard Shad

Correlations of relative condition with copper concentraion for Lake Charleston and Lake Shelbyville gizzard shad are presented in Figures 5 and 6, respectively. Since there is a lack of significant correlation for gizzard shad from each lake, it appears that copper concentration is not a factor contributing to condition. Other factors such as diet, food availability, and variable spawning times are the most probable determining factors affecting condition in these fish. Although copper levels apparently have no effect on condition in gizzard shad, these fish may be passing on copper to piscivores wherein induced physiological responses are going unnoticed. Additional research will need to be completed to determine the full effects on copper transfer from gizzard shad to higher carnivores.

Analysis of Variance and Duncan´s multiple range test were used to compare relative condition (Kn) of 0-yr gizzard shad from Lake

Charleston and Lake Shelbyville and 1-yr shad from Lake Charleston (Table 7). Although there was no significant difference in relative condition between 0-yr Charleston shad and 0-yr Shelbyville shad, a significant difference ($F=4.65$, $P=0.016$) was observed between Charleston 0-yr and 1-yr shad, indicating that 1-yr shad are in significantly poorer condition than are 0-yr shad. It appears that the 0-yr shad from Lake Charleston and Lake Shelbyville are feeding upon the abundant plankton communities in each lake at the beginning of the first growing season which would account for their good condition (Surratt, 1990). However, the 1-yr shad in Lake Charleston are feeding more poorly than the 0-yr shad possibly due to the onset of ontogeny in the diet. Ontogeny would allow a reduction in consumption of highly nutritous plankton and an increase in consumption of detritus of low nutritional quality (Mundahl and Wissing, 1987).

Relative weight (Wr) is the percentage of length-specific standard weight represented by the observed weight. Values less than 100 represent "stunted" fish (less available/nutritious food) while values greater than 100 represent fish of superior condition (abundant/nutritious food). As with the Kn values, the high Wr values for Lake Charleston (Appendix A) and Lake Shelbyville (Appendix B) gizzard shad also reflects the good condition of the 0-yr age class. One reason attributed to the good health of the 0-yr gizzard shad may be due to a lack of competition for food items. These fish have greater filter feeding efficiency during the first growing season and therefore would be able to feed on a wide range of plankton of varying sizes (Drenner _et al._, 1982).

Relative weights were correlated with copper concentration for Lake Charleston and Lake Shelbyville gizzard shad. Lake Charleston (Fig. 7) shows a significant positive correlation, possibly due to data outliers causing a positive relationship. Lake Shelbyville (Fig. 8) shows no significant correlation. It appears, as with the Kn values, that copper concentration is having no noticeable adverse effect on relative weights of gizzard shad in Lake Charleston and Lake Shelbyville.

CONCLUSION

The sediment copper load in Lake Charleston is increasing with copper sulfate treatments. Within a 24 hour period, 90-95% of the copper ion dissipates from the water column to the sediments where the ion remains and is readily available for consumption by detrital feeders. Although the observed sediment copper levels are not considered "elevated" by the Illinois Environmental Protection Agency, additional treatments in the near future may excede this level.

The reason Lake Shelbyville gizzard shad would contain a higher copper concentration than Lake Charleston shad remains unresolved. Allocthanous metals may be entering Lake Shelbyville at some point and gizzard shad are aquiring copper through food uptake. The high variability of copper concentrations within gizzard shad age classes for each lake may be attributerd to individual homeostatic control of the copper ion within whole body tissue.

It appears that copper in Lake Charleston gizzard shad populations 0- and 1-yr age classes is not a contributing factor to their condition, although further copper sulfate treatments will undoubtedly increase sediment copper load and in the future may affect these detritus feeders. Even though condition does not seem to be affected in gizzard shad, accumulated copper may be inducing unnoticed and unstudied adverse physiological responses in higher piscivores.

Gizzard shad are undoubtedly the main forage fish for these piscivores in Lake Charleston and are considered the main link between trace metal biomagnification for piscivores in this impoundment. Further research on piscivore copper accumulation needs to be completed to determine the full scope of copper magnification in these higher carnivores.

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Table 2. Mean sediment copper levels (mg/kg) in Lake Charleston, 1988-1989 (Allison, 1989).

SITE	1988 (n)	1989 (n)
River	*NA	7.8 (3)
01	NA	21.7 (2)
02	NA	16.1 (1)
03	4.6 (1)	4.3 (2)
04	8.8 (1)	21.7 (3)
05	8.7 (1)	22.6 (3)

*Not Available

Table 3. Water column copper concentrations before and after copper sulfate ($CuSO_4$) application, 1987-1989.

Date of Application	Amount CuSO4 Applied	Water Column Before CuSO4	Water Column immediately after CuSO4	Water Column 24 hrs. after Application
Aug 1989	2,000 lbs.	< 0.01 ppm	0.68 ppm	0.04 ppm
Jun 1988	2,000 lbs.	0.01 ppm	2.00 ppm	0.05 ppm
Sep 1987	2,000 lbs.	0.04 ppm	1.00 ppm	0.10 ppm

Table 4. Mean length and weight (± standard deviation) of gizzard shad from Lake Charleston and Lake Shelbyville.

		Lake Charleston			Lake Shelbyville	
AGE	n	LENGTH (mm)	WEIGHT (g)	n	LENGTH (mm)	WEIGHT (g)
0+	20	107.0 (14.86)	11.52 (4.61)	10	67.9 (4.21)	2.97 (0.51)
1+	14	122.3 (14.76)	16.36 (5.51)	3	97.3 (41.56)	11.84 (12.78)
2+	2	148.0 (15.00)	30.01 (9.01)	7	174.1 (19.45)	49.12 (17.30)
3+	NC	----	----	7	186.1 (14.30)	62.57 (16.09)
4+	NC	----	----	1	217.0 (0.00)	69.91 (0.00)

n = number of samples
NC = not collected

Table 5. Mean and standard error for relative condition and copper concentration (mg/kg) for Charleston 0- and 1-yr and Shelbyville 0-yr gizzard shad.

SOURCE/ AGE CLASS	n	Kn MEAN	Kn STD. ERROR	[Cu] mg/kg MEAN	[Cu] mg/kg STD. ERROR
CHAR. 0+	16	1.006	0.0119	1.262	0.3737
CHAR. 1+	14	0.967	0.0116	0.694	0.1999
SHELBY. 0+	10	1.030	0.0208	3.259	1.1218

n = number of samples with detectable metal

Table 6. Analysis of variance and Duncan's multiple range test for copper concentrations (mg/kg) in Charleston 0-and 1-yr and Shelbyville 0-yr gizzard shad.

	Sum Squares	Deg. Freedom	Mean Square
Between Age Classes	40.79868	2	20.39934
Error	154.037	37	4.163161
Total	194.8356		

F-Test Ratio = 4.899964
Significance = 0.0129

Duncan´s Multiple Range Test for Copper Data

Age Class vs.	Age Class	Difference	Sig .05	Sig .01
Char. 1+	Char. 0+	0.5678928	-	-
Char. 1+	Shelby. 0+	2.565343	*	*
Char. 1+	Shelby. 0+	1.99745	*	-

Standard Error of Treatment Means = 0.5697609

Table 7. Analysis of variance and Duncan's multiple range test for relative condition (Kn) in Charleston 0-and 1-yr and Shelbyville 0-yr gizzard shad.

	Sum Squares	Deg. Freedom	Mean Square
Between Age Classes	0.0245549	2	0.0122775
Error	0.0976741	37	0.0026398
Total	0.122229		

F-Test Ratio = 4.650835
Significance = 0.0158

Duncan´s Multiple Range Test for Relative Condition

Age Class vs.	Age Class	Difference	Sig .05	Sig .01
Char. 1+	Char. 0+	0.0391964	-	-
Char. 1+	Shelby. 0+	0.0626714	*	*
Char. 0+	Shelby. 0+	0.0234749	-	-

Standard Error of Treatment Means = 0.014347

Figure 1. Linear regression of copper concentration (mg/kg) wet weight vs. body weight (g) for gizzard shad in Lake Charleston.

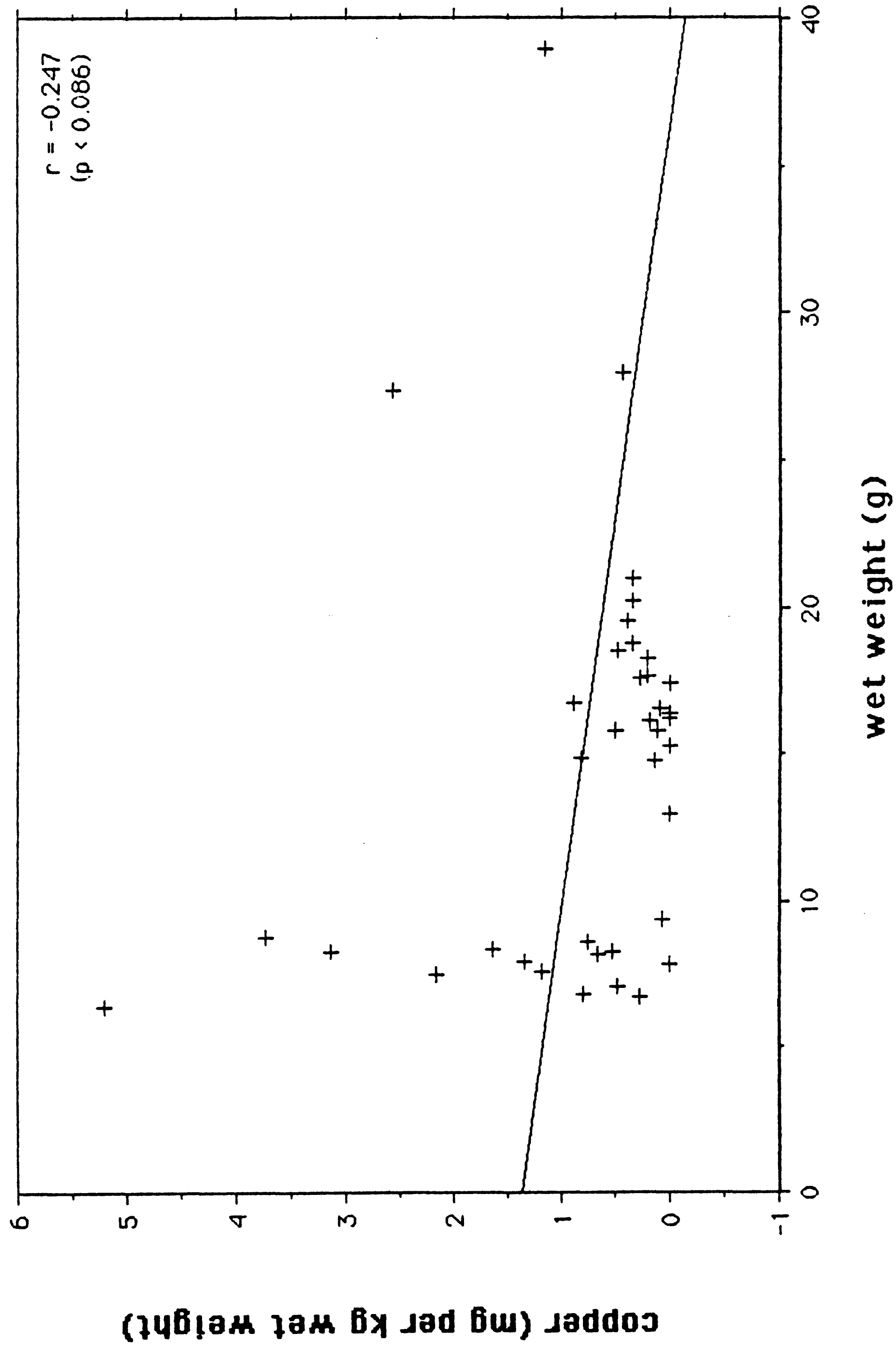

r = −0.247
(p < 0.086)
wet weight (g)
copper (mg per kg wet weight)
0
10
20
30
40
-1
0
1
2
3
4
5
6

Figure 2. Linear regression of copper concentration (mg/kg) wet weight vs. body weight (g) for gizzard shad in Lake Shelbyville.

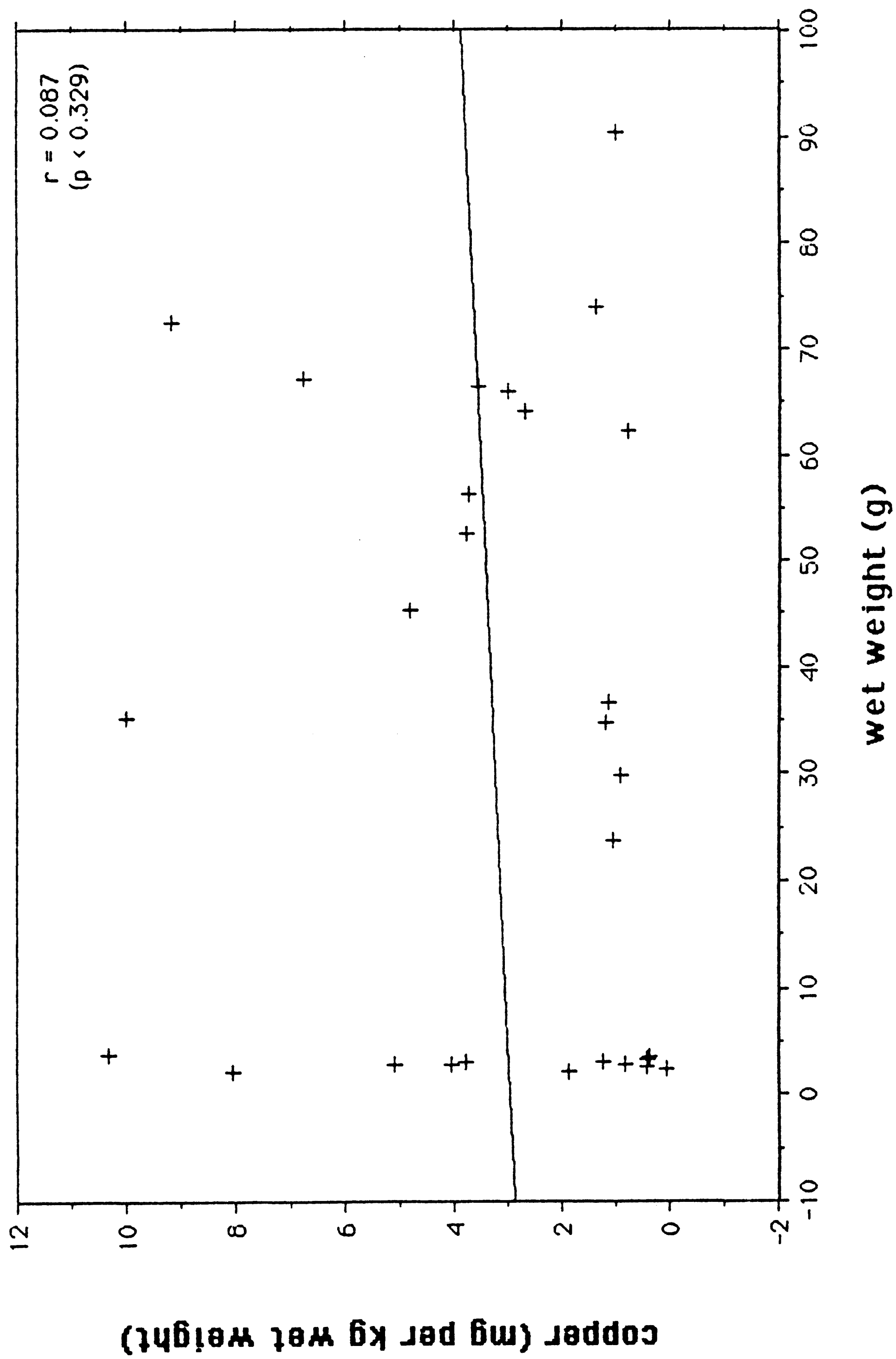

r = 0.087
(p < 0.329)
wet weight (g)
copper (mg per kg wet weight)
-10
0
10
20
30
40
50
60
70
80
90
100
-2
0
2
4
6
8
10
12

Figure 3. Histogram of mean copper concentration (mg/kg) wet weight for gizzard shad age classes in Lake Charleston. Error bars indicate ± standard deviation. Sample sizes are indicated by N.

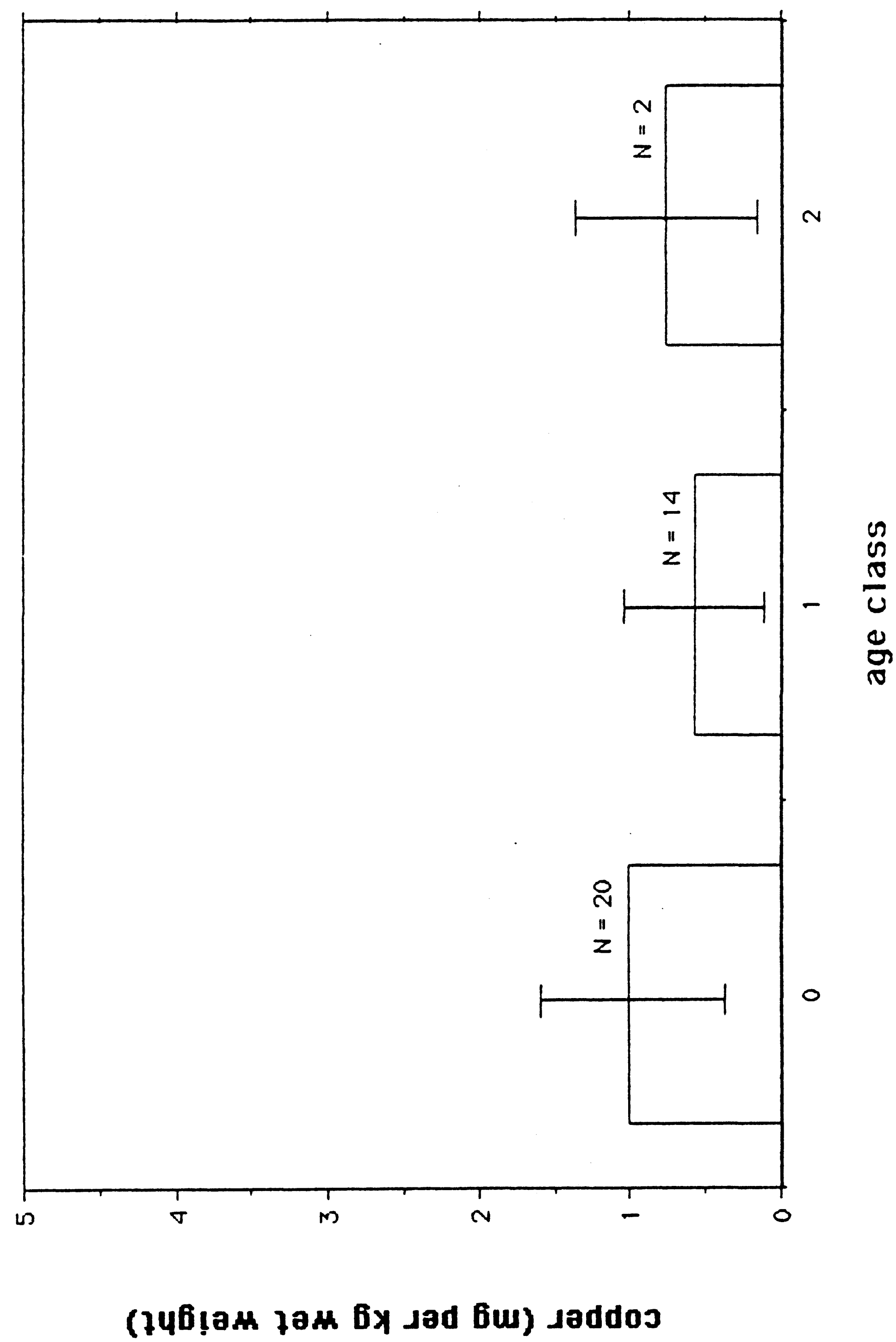

copper (mg per kg wet weight)
0
1
2
3
4
5
N = 20
N = 14
N = 2
0
1
2
age class

Figure 4. Histogram of mean copper concentrations (mg/kg) wet weight for gizzard shad age classes in Lake Shelbyville. Error bars indiacte ± standard deviation. Sample sizes are indicated by N.

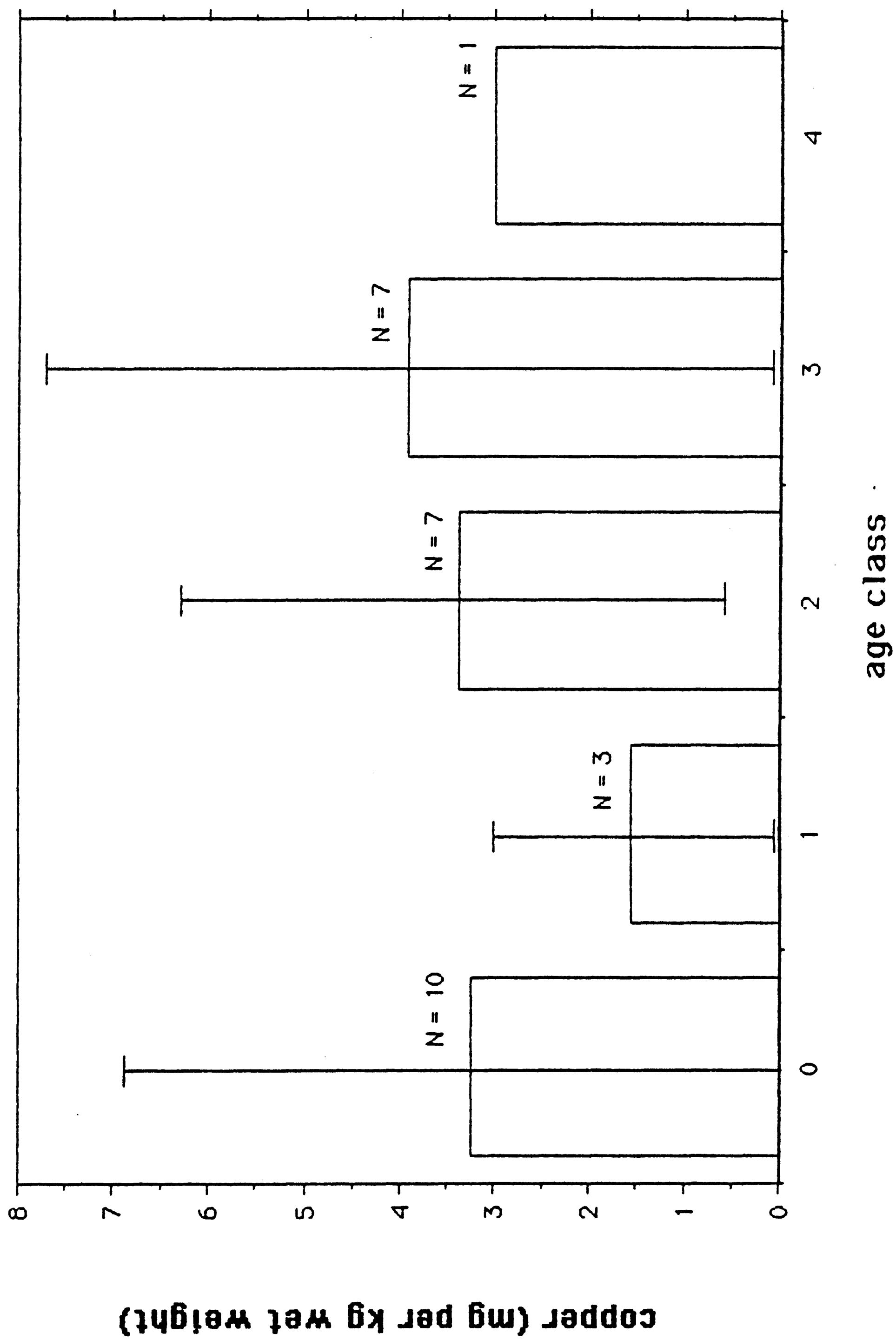

copper (mg per kg wet weight)
age class
0
1
2
3
4
5
6
7
8
N = 10
N = 3
N = 7
N = 7
N = 1

Figure 5. Correlation of copper concentration (mg/kg) wet weight vs. relative condition (Kn) for gizzard shad in Lake Charleston.

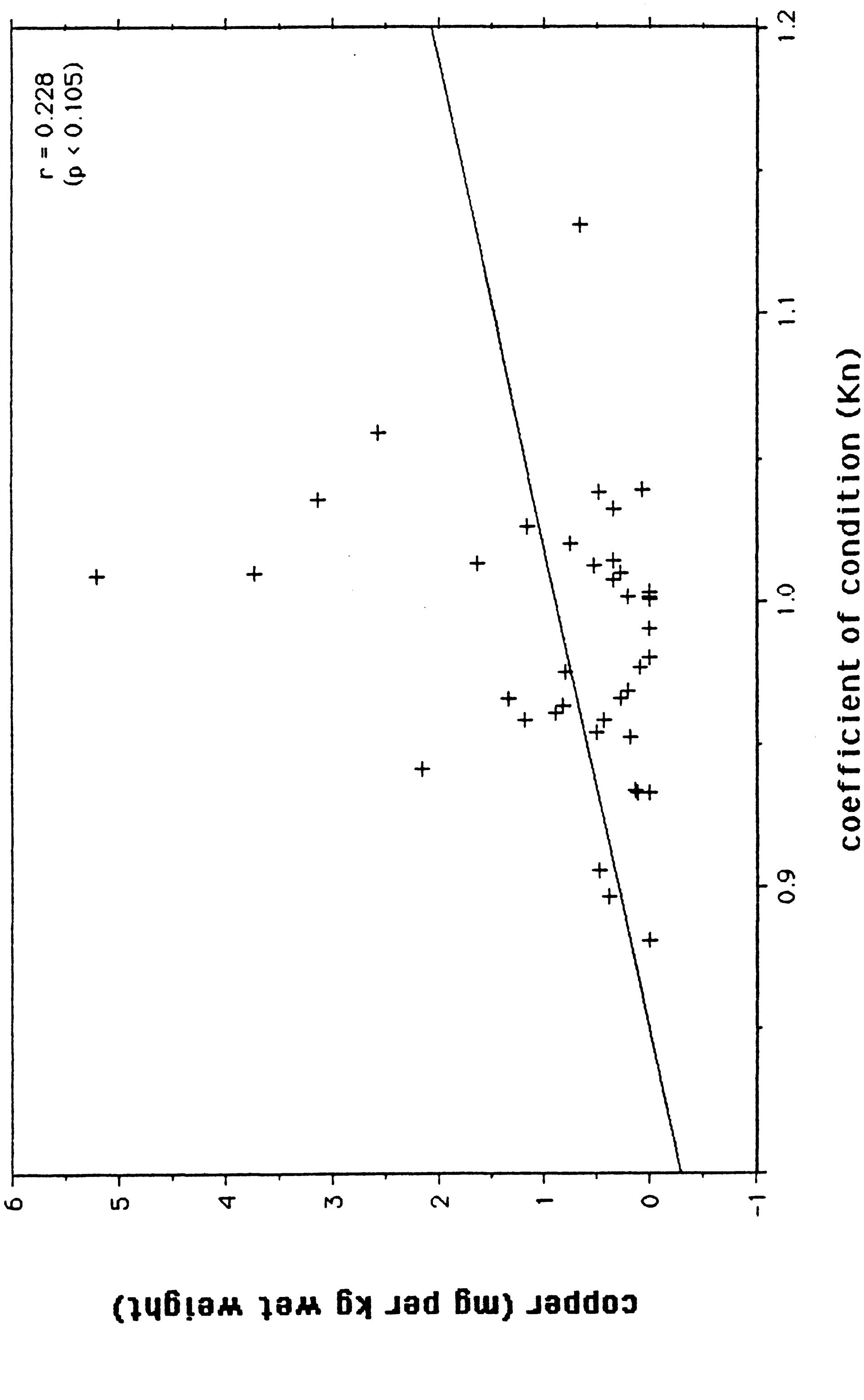

r = 0.228
(p < 0.105)
coefficient of condition (Kn)
copper (mg per kg wet weight)
0.9
1.0
1.1
1.2
-1
0
1
2
3
4
5
6

Figure 6. Correlation of copper concentration (mg/kg) wet weight vs. relative condition (Kn) for gizzard shad in Lake Shelbyville.

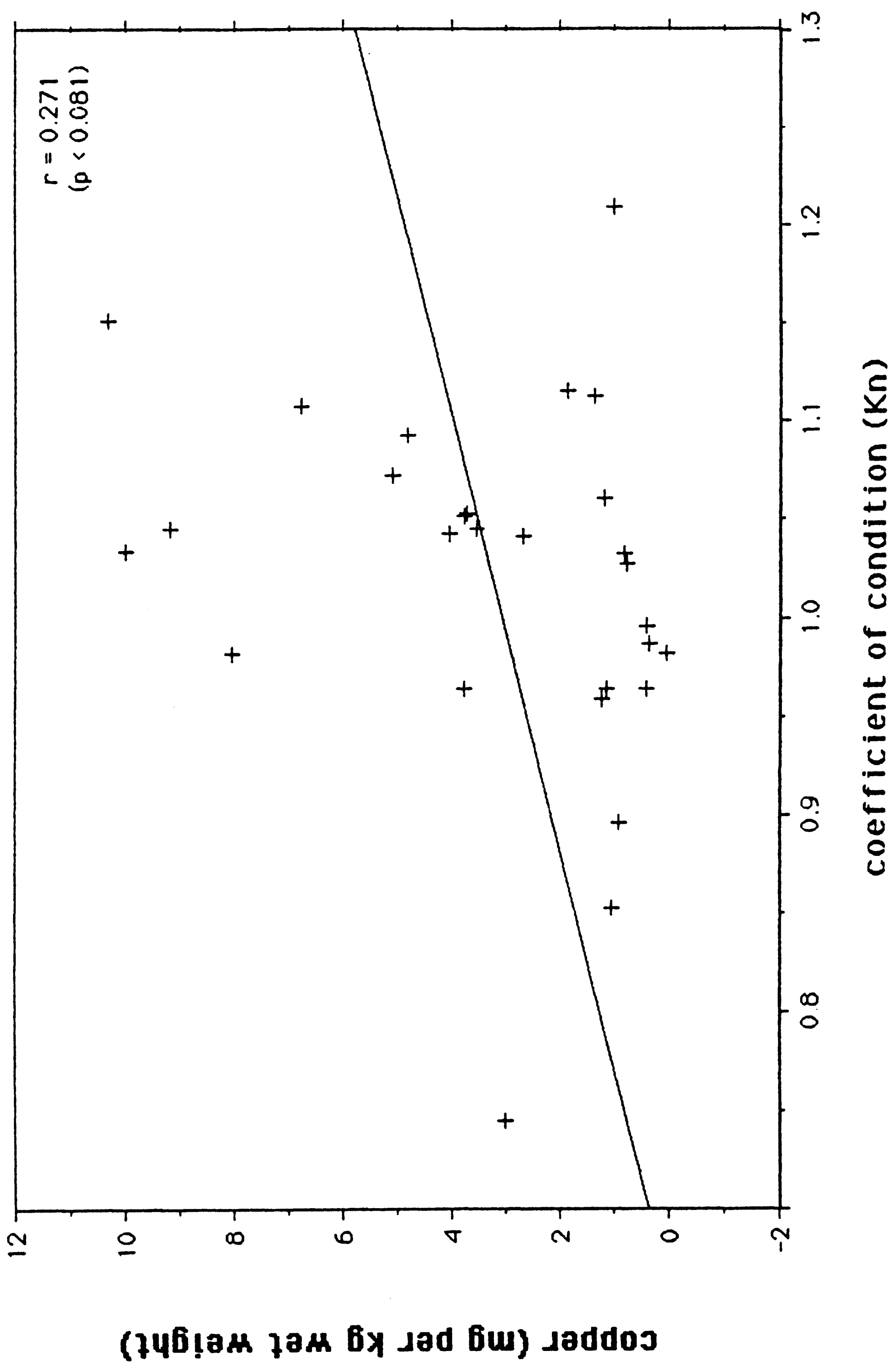

r = 0.271
(p < 0.081)
copper (mg per kg wet weight)
coefficient of condition (Kn)
-2
0
2
4
6
8
10
12
0.8
0.9
1.0
1.1
1.2
1.3

Figure 7. Correlation of relative weight (Wr) vs. copper concentration (mg/kg) for gizzard shad from Lake Charleston.

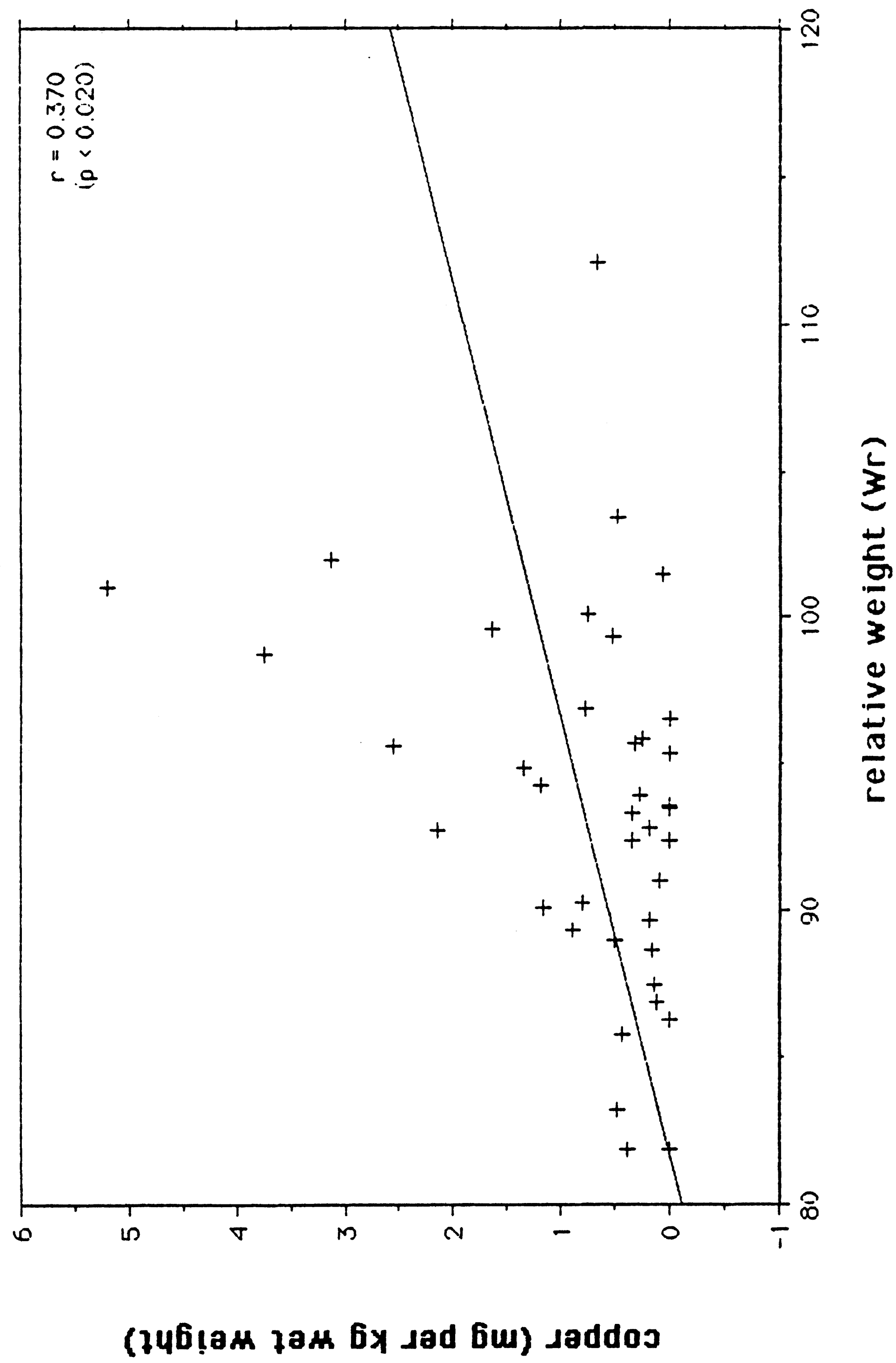
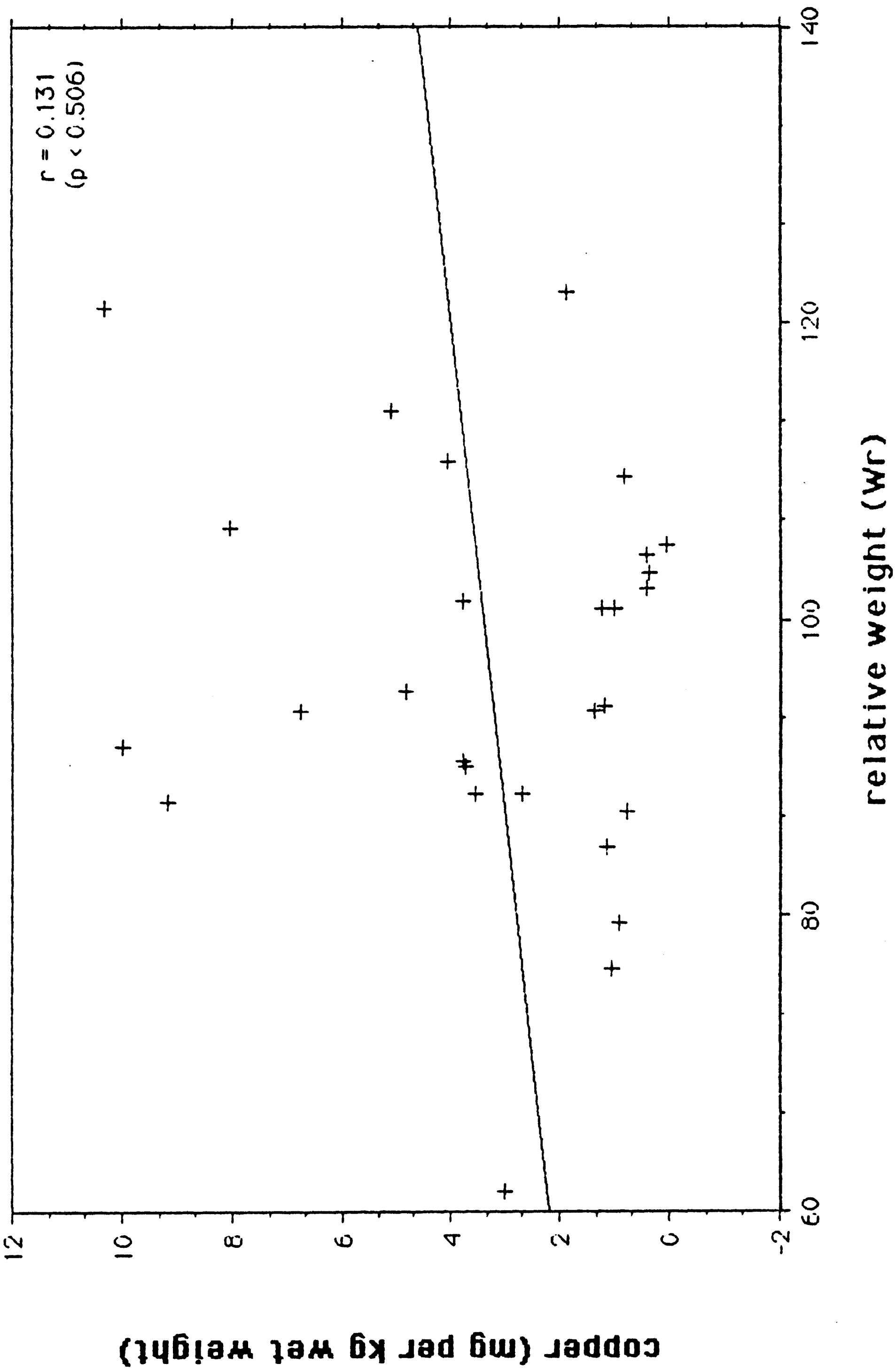

r = 0.370
(p < 0.020)
relative weight (Wr)
copper (mg per kg wet weight)
80
90
100
110
120
-1
0
1
2
3
4
5
6

Figure 8. Correlation of relative weight (Wr) vs. copper concentration (mg/kg) for gizzard shad from Lake Shelbyville.


r = 0.131
(p < 0.506)
relative weight (Wr)
copper (mg per kg wet weight)
60
80
100
120
140
-2
0
2
4
6
8
10
12

APPENDICIES

APPENDIX A: LAKE CHARLESTON GIZZARD SHAD, RAW DATA

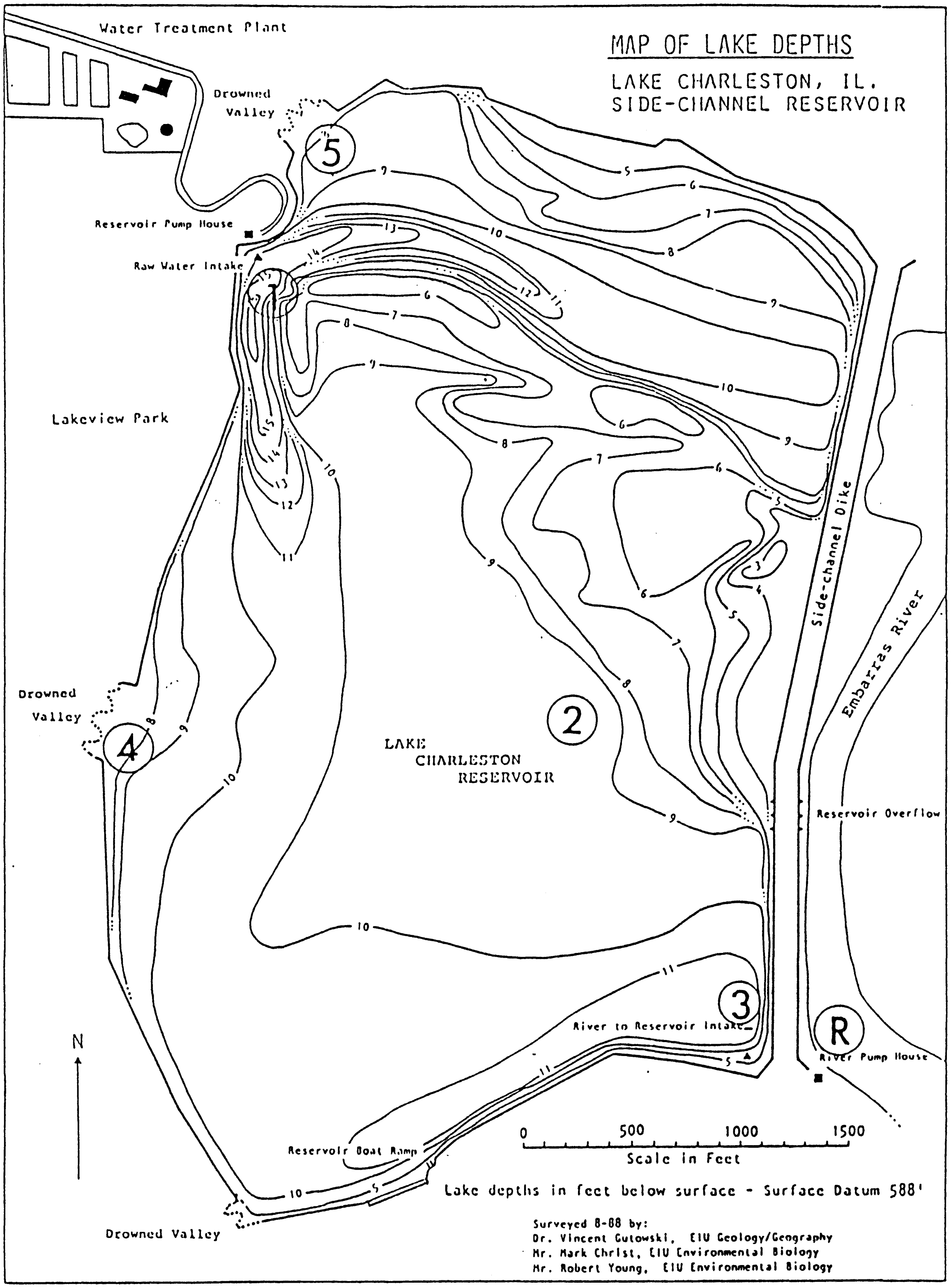
	LENGTH (mm)	WEIGHT (g)	Wr	Kn	[Cu] (mg/kg)
AGE-0	89	6.43	101.1	1.010	5.205
	91	7.06	103.4	1.039	0.480
	92	6.85	96.9	0.976	0.788
	93	8.20	112.1	1.131	0.659
	96	7.63	94.3	0.959	1.180
	96	8.25	102.0	1.036	3.133
	96	7.81	96.6	0.981	LD*
	97	7.93	94.9	0.966	1.349
	97	8.32	99.6	1.014	1.638
	98	8.64	100.1	1.021	0.748
	99	8.81	98.8	1.011	3.738
	100	9.34	101.5	1.040	0.065
	113	12.93	95.4	1.004	LD
	122	16.17	93.5	1.001	LD
	124	16.15	88.7	0.953	0.174
	124	15.81	86.9	0.933	0.120
	127	17.62	89.7	0.969	0.195
	127	18.79	95.7	1.033	0.332
	128	17.37	86.3	0.933	LD
	131	20.22	93.3	1.015	0.348
X	107.0	11.517	96.5	1.0013	1.008
S.D.	14.86	4.611		0.0446	1.390
AGE-1	92	6.78	95.9	0.966	0.264
	96	7.50	92.7	0.942	2.157
	97	8.31	99.4	1.013	0.525
	120	14.82	90.3	0.964	0.809
	121	14.73	87.5	0.934	0.129
	122	16.19	93.6	1.002	LD
	123	15.79	89.0	0.954	0.508
	123	16.39	92.4	0.991	LD
	124	16.56	91.0	0.977	0.086
	125	16.68	89.3	0.961	0.895
	125	17.54	93.9	1.011	0.284
	125	15.29	81.9	0.881	LD
	127	18.22	92.8	1.002	0.195
	132	18.47	83.2	0.906	0.475
	135	19.51	81.9	0.896	0.392
	143	27.34	95.6	1.059	2.568
	149	27.95	85.8	0.959	0.427
X	122.3	16.357	90.4	0.9659	0.571
S.D.	14.76	5.509		0.0445	0.705
AGE-2	133	21.00	92.4	1.008	0.352
	163	39.01	90.1	1.027	1.169
X	148.0	30.005	91.2	1.0173	0.761
S.D.	15.00	9.005		0.0096	0.409

LD* BELOW THE LIMITS OF DETECTION

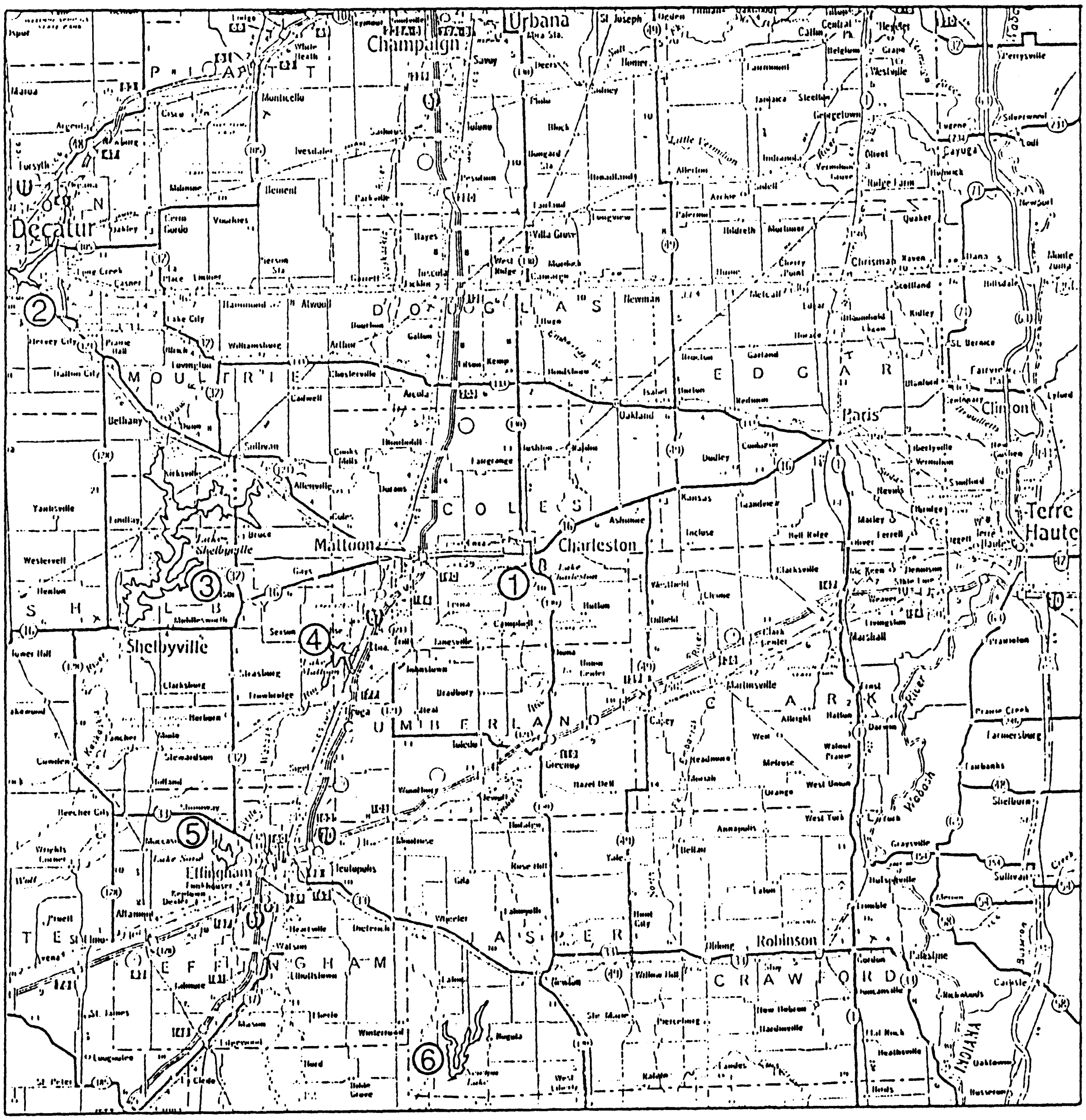
APPENDIX B. LAKE SHELBYVILLE GIZZARD SHAD, RAW DATA

	LENGTH (mm)	WEIGHT (g)	Wr	Kn	[Cu] (mg/kg)
AGE-0	59	2.11	122.1	1.116	1.846
	62	2.15	106.3	0.982	8.045
	67	2.95	114.1	1.072	5.093
	68	3.00	110.7	1.043	4.047
	68	2.77	102.2	0.963	0.411
	68	2.97	109.6	1.033	0.811
	71	3.76	121.0	1.151	10.312
	71	3.13	100.7	0.958	1.249
	72	3.39	104.3	0.996	0.418
	73	3.50	103.1	0.987	0.360
X	67.9	2.973	109.4	1.0300	3.259
S.D.	4.21	0.506		0.0623	3.365
AGE-1	65	2.47	105.1	0.981	0.032
	71	3.15	101.3	0.964	3.781
	156	29.91	79.4	0.896	0.895
X	97.3	11.843	95.3	0.9473	1.569
S.D.	41.56	12.778		0.0367	1.603
AGE 2	147	23.85	76.4	0.852	1.057
	155	34.75	94.1	1.061	1.192
	163	36.61	84.5	0.964	1.120
	168	45.39	95.2	1.093	4.822
	192	64.20	88.2	1.042	2.698
	194	66.37	88.2	1.045	3.524
	200	72.65	87.7	1.045	9.204
X	174.1	49.117	87.8	1.0146	3.374
S.D.	19.45	17.296		0.0756	2.720
AGE-3	157	35.17	91.4	1.034	10.020
	179	52.70	90.4	1.052	3.779
	183	56.28	90.0	1.053	3.708
	191	62.32	87.0	1.027	0.758
	191	67.19	93.8	1.108	6.776
	197	73.99	93.7	1.113	1.363
	205	90.35	100.8	1.209	1.015
X	186.1	62.571	92.5	1.0852	3.917
S.D.	14.30	16.086		0.0593	3.172
AGE-4	217	65.91	61.4	0.745	3.012

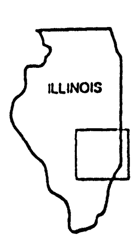
APPENDIX C. DEPTH PROFILE MAP OF LAKE CHARLESTON



APPENDIX D. MAP OF EAST-CENTRAL ILLINOIS



MILES 5 10 20 30 40 50



N

1 = CHARLESTON SIDE-CHANNEL RESERVOIR

2 = LAKE DECATUR

3 = LAKE SHELBYVILLE

4 = LAKE MATTOON

5 = LAKE SARA

6 = NEWTON LAKE