HABITAT ASSOCIATIONS OF FISH

ASSEMBLAGES ON THE TISHOMINGO NATIONAL

WILDLIFE REFUGE

By

WILLIAM SCOTT CHAPPELL

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Thesis Approved:

William L. Fisher
Thesis Advisor

Anthony A. Echelle

Emily H. Stanley

Wayne B. Powell
Dean of the Graduate College

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CHAPTER I

INTRODUCTION

This thesis is composed of two manuscripts written in the format suitable for submission to The Southwestern Naturalist (Chapter II) and Environmental Biology of Fishes (Chapter III). Each chapter is complete without supporting materials. Chapter I is an introduction to the rest of the thesis. The manuscripts are as follows; Chapter II, "Distribution, structure, and habitat associations of fish assemblages of the Tishomingo National Wildlife Refuge," and Chapter III, "Effects of seasonal changes in habitat availability on fish assemblage structure and habitat associations in a large floodplain lake (Oklahoma, U.S.A.).

CHAPTER II

DISTRIBUTION, STRUCTURE, AND HABITAT ASSOCIATIONS OF FISH ASSEMBLAGES OF THE TISHOMINGO NATIONAL WILDLIFE REFUGE

ABSTRACT---We surveyed the ichthyofauna of the Tishomingo National Wildlife Refuge in southcentral Oklahoma to document the occurrence of species on the refuge and to describe the spatial relationship between species and habitats. We sampled 12 lentic and 4 lotic habitats using electrofishing, experimental gill nets, and minnow seines during 1996 and 1997. We collected 52 species out of a total of 84 possible for the region and report a range extension for spotted sucker (Minytrema melanops). Multivariate direct gradient analysis indicated a gradient in habitat characteristics ranging from water bodies at relatively low elevations with low water clarity, large width and depth and high conductivity to those at relatively high elevations with high water clarity, small width and depth, and low conductivity. The gradient in fish assemblage structure among lotic habitats was typical for Oklahoma. The gradient in assemblage structure among lentic habitats varied from being dominated by fishes tolerant of low water clarity to those dominated by visually oriented centrarchids. The major difference between high elevation lentic and lotic habitats was the presence of larger-sized species and absence of certain stream species in lentic habitats. The difference between lower elevation lentic and lotic habitats was less distinguishable. The mechanism behind the observed pattern of fish distribution among lentic habitats could be piscivory mediated by water clarity and/or differences in connectivity among habitats.

Extensive floodplain development in prairie river systems is generally limited by short and unpredictable flood pulses which tend to create highly

incised channels (Matthews, 1988; Junk et al., 1989). However, extensive

floodplains and associated off-channel habitats can develop in the riverine

sections of reservoirs (Kimmel and Groeger, 1986; Thornton, 1990). Although

riverine fish assemblages are not specifically adapted to reservoir or floodplain

lake environments, species-specific food and habitat requirements are likely

conserved (Gelwick & Matthews, 1990). In the Red River system in

southwestern Oklahoma, the distribution of small species was related to

conductivity, stream size, woody debris, and water clarity (Taylor et al., 1993).

Habitat specialization, physicochemical tolerances, and diffuse competition were

considered important factors responsible for these relationships (Echelle et al.,

1972; Taylor et al., 1993).

In Oklahoma, there are few natural lakes, but there are many

impoundments (Miller and Robison, 1973). The relationship between whole

lake habitat variables and fish assemblage structure in southern Great Plains

impoundments has not been identified. However, habitat-related differences in

fish assemblage structure have been evaluated within lakes of the southern

Great Plains (Summerfelt, 1971; Gelwick and Matthews, 1990) and in other

temperate (Weaver et al., 1996) and tropical (Rodriguez and Lewis, 1997) lakes.

Among tropical floodplain lakes in the dry season, local differences in fish

assemblage structure were related to transparency, depth, and surface area

(Rodriguez and Lewis, 1997). The principal mechanism driving the relationship

was the visual orientation of species (Rodriguez and Lewis, 1997). Connectivity

among habitats had little effect on fish assemblage structure in these tropical lakes (Rodriquez and Lewis, 1997). Among north temperate lakes, local differences in fish assemblage structure were related to littoral macrophyte heterogeneity, which affects habitat-mediated biotic interactions (Weaver et al. 1996). Lentic fish assemblages in the southern Great Plains may also be influenced by these factors.

My objectives were to (1) survey the fishes of the Tishomingo National Wildlife Refuge (TNWR) and (2) characterize the relationship between fish assemblage structure and habitat and fish-habitat associations among water bodies on the refuge.

STUDY AREA---The TNWR is located in Johnston and Marshall counties in southcentral Oklahoma along the Washita River arm of Lake Texoma (OK/TX). The refuge was established in 1946 following completion of the Denison Dam (Lake Texoma project) to provide refuge and breeding grounds for migratory birds and other wildlife. The refuge encompasses several lakes, creeks, ponds, sloughs, and a portion of the Washita River (Figure 1). The largest lake on TNWR (Cumberland Pool) has gradually become isolated from the main body of Lake Texoma as a result of sedimentation from the Washita River. It is now essentially a large floodplain lake. This process also created several smaller floodplain lakes (e.g. Bell Creek, Rock Creek) that were formerly coves of the Cumberland Pool. These smaller lakes are separated from the Cumberland

Pool by the forested embankments of the Washita River, although they are sometimes connected to one another during floods.

MATERIALS AND METHODS---<u>Fish collections</u>---From May to November 1996 and from March to November 1997, we used experimental gill nets, trap nets, minnow seines and electrofishing to sample the fish fauna of TNWR. Trap nets were only used in the Cumberland Pool and lower Big Sandy Creek so these collections were not included in the multivariate analyses. However, trap net data were included to document occurrences of species not collected by other sampling gears. The Cumberland Pool was sampled regularly during the entire study period; other water bodies were sampled once in the summer of 1996 or 1997.

We used experimental monofilament gill nets with mesh sizes from 1.5 to 10.2 cm, 2 to 8 15-m panels, and total lengths from 31 to 91 m. All gill nets were set on the bottom and from dusk til dawn (11-14 hour sets). We used a stratified random sampling design in the Cumberland Pool, and in the other lentic habitats we set at least a single gill net in open water habitat. In streams we set a single gill net diagonal to the flow. No gill nets were set in Dicks Pond, upper Big Sandy Creek, and the Washita River. Total effort for gill nets is reported as the total area (m^2) of netting deployed during one nocturnal set (Table 1). Catch per unit effort (CPUE) for each water body was calculated as the total number of fish divided by the total effort, multiplied by 100 (catch per 100 m^2 d^{-1}).

We used an electrofishing boat with a 3.5 GPP electrofisher (Smith RootInc.) and 6-hp generator to capture fish in some streams and ponds. All electrofishing operations used direct current at 60 pulses per second. We used a stratified systematic design for the Cumberland Pool and a simple systematic design for other water bodies. Electrofishing was not conducted in upper Big Sandy Creek and eastern Muel Lake. Total effort for electrofishing is reported as the total number of minutes electricity was conducted through the water (Table 1). CPUE for each water body was calculated as the total catch divided by the total effort.

Seining was conducted using a 4 x 0.9-m minnow seine and a 12 x 0.9-m bag seine (both 3.2 mm mesh). The Cumberland Pool received the majority of effort; we sampled at several stations around the pool. The Washita River, Pennington Creek, Dicks Pond, and upper Big Sandy Creek were also sampled with seines. Most seine hauls were pulled 15-m to the shoreline in seinable areas. Upper Big Sandy, however, was sampled using block seines and enough seine hauls to remove approximately 90% of the fish (visual estimation) in a sequence of riffle, run, and pool habitats. Effort is reported as total number of 10 m^2 areas sampled (Table 1). CPUE was calculated as total catch divided by total effort.

Voucher specimens of the smaller species were preserved in 10% formalin and later transferred to 40% isopropyl alcohol. Close-up pictures were taken of the larger species. Voucher specimens and pictures are stored in the

Oklahoma State University (Stillwater, Oklahoma 74078) fish collection at the Life Sciences West building. Scientific names for species follow Robins et al. (1991).

Habitat characteristics---Conductivity and pH were measured with a multiparameter water quality monitoring instrument (Hydrolab Scout 2) in close proximity to the fish samples. Secchi disk depths were measured at the same locations. Maximum depth of water bodies was measured with either a calibrated rod (± 0.1 m), sonar (± 0.3 m), or tape measure (± 0.05 m). Flow was classified as none (0), intermittent (1), or perennial (2) based on visual observation during the study. For lotic habitats, width was mean stream width recorded along 20 perpendicular transects following the guidelines of Simonson et al. (1994). Mean width for lentic habitats was estimated along the long axis from a Geographic Information System coverage of aquatic habitats. The approximate elevation of habitats was recorded as the closest (higher) contour interval on 1967 USGS 7.5-minute topographic quadrangles. Dam types were classified as no dam (0) or connecting pipe, low road, spillway, or water control structure (1). The last two variables were included to provide information on connectivity among habitats.

Multivariate analysis---We used cluster analysis and multivariate direct gradient analysis to determine fish-habitat relationships among the waterbodies of TNWR. Species and size categories used in the analysis were limited to those represented by ≥ 1% of the total CPUE by gear type. We used Spearman

rank correlation (Siegel, 1956) to determine if any habitat variables were significantly correlated ($p < 0.05$); as collinearity can add unnecessary redundancy to multivariate datasets (Gauch, 1982). For significantly correlated variables that were, the one with the greater range of values was used in the analysis. All species data were transformed as $\log_{10}(N+1)$. Log transformations are frequently used on fish CPUE data (Summerfelt, 1971; Taylor et al., 1993; Rodriguez and Lewis, 1997).

Cluster analysis was used to identify groupings of species and sites (i.e., water bodies). These groups were subsequently used to classify species and site scores in multivariate direct gradient analysis. We used Ward's minimum variance clustering method (Ludwig and Reynolds, 1988). For the species analysis, we used Pearson correlation matrices based on the transformed data. For the site analysis, we used the Morisita-Horn index (Krebs, 1989) because it is independent of both sample size and species diversity relative to other similarity measures (Ludwig and Reynolds, 1988). All matrices were rescaled to a range of 0 (identical) to 100 (completely different) in the analysis. The delineation of important clusters was based on the semipartial r^2 statistic which distinguished among species groups and fish assemblages in the seine data. The semipartial r^2 reflects the decrease in variance among clusters from joining two clusters to form the current cluster (SAS, 1985).

We used canonical correspondence analysis (CCA; ter Braak 1986) to visualize relationships of species and site clusters to habitat variables. CCA is

an eigenvector-based, direct gradient analysis technique frequently used in ecological investigations (Palmer 1993). Species and site scores in CCA are constrained by the environmental variables such that the highest possible fraction of variation in the dependent variable is explained by an axis (Palmer 1993). Species and site groups closely associated with the habitat variables should be close together on either end of a habitat gradient (or multiple gradients) on the multivariate ordination plots. The significance of the CCA results was determined using Monte Carlo tests (1000 permutations). CCA axes were described using the correlation of habitat variables with the axes. We arbitrarily chose $|r| = 0.50$ as the cutoff point for including a habitat variable in the description of an axis on the ordination diagrams.

RESULTS---Sampling effort---Sampling effort and sampling gear varied among water bodies at TNWR (Table 1). Cumberland Pool received the majority (77.0%) of the total sampling effort. The Washita River received the next highest sampling effort (9.6%) and all other water bodies received approximately 1% or less of the total effort. Electrofishing samples from the Washita River were not included in the analysis because of sampling inefficiency.

Habitat characteristics--Water bodies on TNWR displayed a wide range of habitat characteristics (Table 2). Widths of water bodies was lowest for upper Big Sandy Creek and highest for Cumberland Pool. The shallowest water body was upper Big Sandy Creek and the deepest was the Cumberland Pool. The

clearest water was in upper big Sandy Creek and the most turbid was in the Washita River. Increasing flow was correlated with decreasing width (r_s = -0.73, $p < 0.01$), so flow was omitted from analysis of all gear types except the seine data. The Washita River deviated from that relationship because of its greater width and flow; it was only sampled effectively with seines. Surface water conductivity was lowest in Dicks Pond and highest in the Washita River (Table 2). Because of its narrow range (7.2 - 8.6), pH was not included in the analysis. Approximate elevations ranged from 189.0 - 195.1 m MSL (above mean sea level) for a difference in elevation of 6.1 m. Presence of dams was correlated with relatively higher water clarity (r_s = 0.59, $p = 0.02$) and lower conductivity (r_s = -0.68, $p < 0.01$); thus, this variable was omitted from further analysis.

Fish abundance--We collected 3,137 fish by electrofishing, 50,618 by seining, and 2,234 from gill nets. The collections contained reprentatives from 14 families and 52 species (Table 3). The most frequently collected species by all sampling gears was gizzard shad (Dorosoma cepedianum). In the electrofishing samples bluegill sunfish (Lepomis macrochirus), inland silverside (Menidia beryllina) and warmouth (Lepomis gulosus) were among the most abundant species. Inland silversides, threadfin shad (Dorosoma petenense) and ghost shiner (Notropis buchanani) were the most abundant species in seine samples, and Ictiobus species (mostly Ictiobus bubalus), blue catfish (Ictalurus furcatus) and longnose gar (Lepisosteus osseus) were the most abundant species in gill net samples. These abundances may not reflect the most

numerically dominant species on the refuge because the fishes of the Cumberland Pool occupied such a large proportion of the catch. However, the Cumberland Pool also occupies the largest proportion of surface water on the refuge.

There were no state or federally listed endangered or threatened species (Oklahoma Biological Survey, personal communication) on TNWR. However, some species were relatively rare in our survey and one collection resulted in a range extension for spotted sucker (Minytrema melanops). Spotted sucker were not expected to occur above Denison Dam of Lake Texoma (Miller and Robison 1973) and we collected 5 juveniles in Pennington Creek. The collection of an adult american eel (Anguilla rostrata) was also unexpected because of the catodromous life history of this species and the presence of Denison Dam blocking its migration for the last 52 years. Some species that were rarely encountered in our survey (Table 3) are considered common in southern Oklahoma (Miller and Robison, 1973). These species may have been infrequently collected because of sampling biases from gear selectivity, sampling time or sampling location.

Binderim (1977) compiled a list of 81 species potentially occurring in Mill Creek, a tributary of the Washita River 25 km west of TNWR. We collected 52 of those species. These included several large river and lake fishes (Miller and Robison, 1973; Binderim, 1977) not collected in Mill Creek. We did not collect 29 of those 81 species listed in Binderim (1977). Some species that were

abundant in nearby Mill Creek (Binderim, 1977) were not in the water bodies we sampled. These species and other locally abundant species in the Red River drainage (Taylor et al., 1993) were probably not collected due to lack of suitable habitat or inadequate sampling effort in suitable habitats. Many of the species listed in Binderim (1977) are considered uncommon to rare in this area of the state. Further collections in the TNWR may reveal the presence of additional species found by Binderim (1977) and others.

Distribution and habitat associations of fish assemblages---Several species did not meet the minimum abundance requirements for inclusion in the analysis (Table 3). Redear sunfish (Lepomis microlophus) were omitted from the electrofishing analysis because the majority collected were very small and possibly confused with orangespotted sunfish (Lepomis humilis). In some cases, individuals were not identified to species (usually due to their small size). For example, in Reeves Ravine we electroshocked large numbers of atherinids (silversides) but did not identify them to species. Including certain genus or family groups in the analysis helped compensate for this deficiency.

Cluster analysis of the seine data identified four species groups (Table 3) and four site groups (Table 1) with a semipartial $r^2 < 0.20$. This classification corresponded well with our intuitive grouping of sites and species. Using $r^2 = 0.20$ as a cutoff, two species and site groups were classified from the electrofishing data (Tables 1 and 3) and two species groups and three site groups from the gill net analysis (Tables 1 and 3).

The CCA for electrofishing data was significant ($P < 0.01$; Table 4) for both the first axis and the overall analysis. The CCA on the gill net data, however, was not significant (Table 4). The seine data could not be tested because the habitat variables outnumbered the samples. The CCA of fishes collected with the three gear types indicated a general pattern in habitat characteristics that ranged from low elevation, wide, deep, and turbid habitats to high elevation, narrow, shallow, and clear habitats (Figures 2-4 and Table 5). The pattern in conductivities varied according to the analysis; for the seine data, conductivity was uncorrelated with the general habitat pattern, whereas for the electrofishing data, conductivity was correlated with increasing width and depth.

Although the CCA of seine data (Figure 2) could not be statistically evaluated, it illustrated the most distinctive associations of water bodies and habitat variables with species and fish assemblages. Fish species in groups A and B (Table 3) were associated with higher water clarity and elevation. Likewise, site groups A and B (Table 1) were also associated with higher water clarity and elevation. Within these groups, species and site group A [e.g., longear sunfish (Lepomis megalotis), orangethroat darter (Etheostoma spectabile), and central stoneroller (Campostoma anomalum), and upper Big Sandy Creek] were associated with higher flow and conductivity and species and site group B [e.g., warmouth and slough darter (Etheostoma gracile), and Dicks Pond] were associated with no flow and low conductivity. Species and site groups C and D [e.g., speckled chub (Macrhybopsis aestivalis), white bass

(Morone chrysops), river carpsucker (Carpiodes carpio), freshwater drum (Aplodinotus grunniens), inland silverside, and gizzard shad, and Pennington Creek, Washita River, and Cumberland Pool] were associated with greater width, depth, conductivity, and flow and lower elevation and water clarity.

Species and site groups based on the habitat gradients were not as clearly separated on the CCA of electrofishing data (Figure 3). Species group A (e.g., gizzard shad, white bass, and freshwater drum; Table 3) was only associated with lower elevation and higher conductivity; this group covered a wide range of other habitat variables. Species group B (e.g., Lepomis, Micropterus, and Percina species) was primarily associated with higher elevations and lower conductivity but it was also weakly associated with higher water clarity, and reduced width and depth. Site group A (e.g., Cumberland Pool, Goosepen Pond, and lower Big Sandy Creek; Table 1) was not associated with any particular habitat variable, and site group B (e.g., McAdams Pond, western Muel Lake and Dicks Pond) was most strongly associated with higher water clarity and lower width and depth, but it was also weakly associated with higher elevations. There seems to be only a weak correspondence between species group B and site group B and even less correspondence between species group A and site group A.

Few distinctions between sites, species, and fish habitat associations could be discerned from the gill-net CCA (Figure 4). Species in group A (e.g. spotted gar (Lepisosteus oculatus) and white crappie (Pomoxis annularis); Table

3) were weakly associated with increasing water clarity and declining water depth, width, and conductivity, and species in group B (e.g. goldeye (<u>Hiodon alosoides</u>), blue catfish (<u>Ictalurus furcatus</u>), and freshwater drum) were weakly associated with declining water clarity and increasing maximum depth, width, and conductivity. The pattern of site groups was only very weakly associated with elevation. These results possibly reflect the lack of variation in open water habitat, which is where the majority of gill nets were set.

DISCUSSION---Variation in fish assemblage structure in stream systems of the southern Great Plains has largely been attributed to a combination of biotic interactions, physicochemical conditions, and a few habitat variables such as stream size and habitat type (Echelle et al., 1972; Matthews, 1984; Taylor et al., 1993; Cantu and Winemiller, 1997). Several studies of fish assemblages in the southern Great Plains have documented patterns of change in composition from headwaters to tailwaters (Smith and Powell, 1971; Binderim, 1977; Gelwick and Matthews, 1990, Ashbaugh et al., 1996; Lutrell, 1996; Williams et al., 1996) that seem associated with variables such as stream size, substrate, gradient, water clarity, and depth (Taylor et al., 1993; Williams et al., 1996). Across these studies, several species have shown persistent associations with either upstream or downstream habitats. Species frequently associated with turbid downstream habitats include speckled chub, inland silverside, freshwater drum, white bass, channel catfish (<u>Ictalurus punctatus</u>), gizzard shad, threadfin shad

(Dorosoma petenense), Ictiobus species, emerald shiner (Notropis atherinoides), common carp (Cyprinus carpio), river carpsucker, and white crappie (Binderim, 1977; Taylor et al., 1993; Ashbaugh et al., 1996; Luttrell, 1996; Williams et al., 1996). Species frequently associated with clear, upstream habitats include central stoneroller, brook silverside (Labisdesdes sicculus), yellow bullhead (Ameiurus natalis), longear sunfish, green sunfish (Lepomis cyanellus), spotted bass (Micropterus punctulatus), orangethroat darter, and logperch (Percina caprodes) (Smith and Powell, 1971; Binderim, 1977; Taylor et al., 1993; Williams et al., 1996). This upstream-downstream pattern was clearly distinguishable in lotic water bodies of TNWR (Figure 2) but not in more lentic water bodies.

Floodplain lakes and impoundments are features of the TNWR landscape that provide permanent lentic habitats for native stream fishes and exotic species to colonize. These habitats vary in size, water clarity, maximum depth, macrophyte development, flow regime, and connectivity with riverine habitats. In general, species in downstream habitats were also associated with turbid, floodplain lakes on the refuge. These lakes generally had higher conductivity, greater depth and width, and lower water clarity. This close association was apparent from the similarity in species composition between the Washita River and the Cumberland Pool as indicated by the clustering of seining sites (Figure 2). However, certain species clusters distinguished between the Washita River and the Cumberland Pool. The Washita River fish assemblage (group C; Figure 2) was similar to the Pennington Creek fish assemblage. This suggests some

differentiation among floodplain-lake and lowland-stream fish assemblages. The relatively close association of lowland creeks and floodplain lakes was also discernable on the electrofishing CCA (Figure 3). For example, Rock Creek (more lake-like) and lower Big Sandy Creek (more stream-like) were close together in the ordination.

Fish assemblages in the small, high elevation, low conductivity, and clear impoundments of TNWR were different from those in the floodplain lakes and the lower reaches of creeks. The fish assemblages of these impoundments were similar to upstream areas of streams in that they had greater abundances of centrarchids. However, they were unique in having larger-sized species such as common carp, largemouth bass (Micropterus salmoides), white crappie, and Ictiobus species and lacking some upland stream residents like orangethroat darter and central stoneroller. They were also lacking some lowland stream species like freshwater drum and white bass. There was also a unique fish assemblage associated with vegetated backwaters of Dicks Pond. The fish assemblage in Dicks Pond was dominated by warmouth and included slough darters; both species are associated with dense aquatic macrophyte beds (Miller and Robison, 1973).

The effects of flooding and connectivity on fish assemblage structure in this system were only weakly inferred from our results. The results suggest some differentiation due to elevation, which plays a role in connectivity. The presence of dams is probably a major factor responsible for the observed

differences. However, connectivity in this system is complex and is based on elevation of the water bodies, their location along the river basin gradient, and limited water movement imposed by water control structures. Although the influence of connectivity on fish assemblage structure was not detected in tropical floodplain lakes (Rodriguez and Lewis, 1997) and north temperate floodplain pools (Halyk and Balon, 1983) both studies suggested that processes (i.e. piscivory mediated by water clarity, extinction) operating through prolonged isolation are probably more important than invasion during flooding. Whether this is true among floodplains lakes in the southern Great Plains is unknown and bares further study.

Seasonal isolation of tropical floodplain lakes is generally associated with low water clarity (Rodriguez and Lewis, 1997). However, unlike southern Great Plains rivers, flooding is predictable and prolonged in tropical and temperate coastal rivers, and species depend on it to move to more favorable habitats for feeding or spawning (Matthews, 1988; Junk et al., 1989). Fishes in southern Great Plains river systems may depend less on flooding that lowland river fish assemblages.

Regardless of connectivity, Rodriguez and Lewis (1997) found that fish assemblages in tropical floodplain lakes vary from being dominated by species adapted to low water clarity to those adapted to higher water clarity. The principal mechanism responsible for the difference was piscivory mediated by water clarity (Rodriguez and Lewis, 1997). Of all the centrarchid species

collected in TNWR, only orangespotted sunfish and white crappie seem adapted to lower water clarity (Miller and Robison, 1973). All centrarchid species were more abundant in the higher elevation, higher water clarity, and lower conductivity water bodies (i.e. impoundments) such as Dicks Pond, Lost lake, and McAdams Pond than in lowland water bodies. All other species, except spotted gar and some obligate upland stream species, are not usually associated with higher water clarity (Miller and Robison, 1973). Future research in reservoir-floodplain systems of the southern Great Plains should focus on the effects on isolation and piscivory, mediated by water clarity, on the distribution of fish assemblages among water bodies.

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LITERATURE CITED

ASHBAUGH, N.A., R.K. REICHERT, S.E. FRANKLIN, M.N. MERCER, H.A. LANMAN, AND B.S. MANTOOTH. 1996. A comparative study of the ichthyofauna in selected streams on the Salt Plains National Wildlife Refuge in Oklahoma. Proceedings of the Oklahoma Academy of Science 76: 17-21.

BINDERIM, G.E. 1977. Fishes of Mill Creek, a tributary of the Washita River, Johnston and Murray counties, Oklahoma. Proceeding of the Oklahoma Academy of Science 57: 1-11.

CANTU, N.E., AND K.O. WINEMILLER. 1997. Structure and habitat associations of Devils River fish assemblages. Southwestern Naturalist 42: 265-278.

ECHELLE, A.A., A.F. ECHELLE, AND L.G. HILL. 1972. Interspecific interactions and limiting factors of abundance and distribution in the Red River Pupfish, Cyprinodon rubrofluviatilis. American Midland Naturalist 88:

109-130.

GAUCH, H.G., Jr. 1982. Multivariate analysis in community ecology. Cambridge University Press, Cambridge, England.

GELWICK, F.P., AND W.J. MATTHEWS. 1990. Temporal and spatial patterns in littoral-zone fish assemblages of a reservoir (Lake Texoma, Oklahoma-Texas, U.S.A.). Environmental Biology of Fishes 27: 107-120.

HALYK, L.C., AND E.K. BALON. 1983. Structure and ecological production of the fish taxocene of a small floodplain system. Canadian Journal of Zoology 61: 2446-2464.

JUNK, W.J., P.B. BAYLEY, AND R.E. SPARKS. 1989. The flood pulse concept in river-floodplain systems. In: D.P. Dodge, D.P., editor. Proceedings of the international large river symposium. Canadian Special Publication Fisheries and Aquatic Sciences 106. Pp. 110-127.

KIMMEL, B.L., AND A.W. GROEGER. 1986. Limnological and ecological changes associated with reservoir aging. In: Hall, G.E. and M.J. Van Den Avyle, editors. Reservoir fisheries management: strategies for the 80's. Reservoir Committee, Southern Division American Fisheries Society, Bethesda, Maryland. Pp. 103-109.

KREBS, C.J. 1989. Ecological methodology. HarperCollinsPublishers, Inc., New York.

LUDWIG, J.A., AND J.F. REYNOLDS. 1988. Statistical ecology. John Wiley & Sons, Inc., New York.

LUTTRELL, G.R. 1996. The fishes of Council Creek, Payne county, Oklahoma. Proceedings of the Oklahoma Academy of Science 76: 39-42.

MATTHEWS, W.J. 1988. North American prairie streams as systems for ecological study. Journal of the North American Benthological Society 7: 387-409.

MILLER R.J., AND H.W. ROBISON. 1973. The fishes of Oklahoma. Oklahoma State University Press, Stillwater, OK.

PALMER, M.W. 1993. Putting things in even better order: the advantages of canonical correspondence analysis. Ecology 74: 2215-2230.

ROBINS C.R., R.M. BAILEY, C.E. BOND, J.R. BROOKER, E.A. LACHNER, R.N. LEA, AND W.B. SCOTT. 1991. A list of common and scientific names of fishes from the United States and Canada. American Fisheries Society Special Publication 20. Bethesda, Maryland.

RODRIGUEZ, M.A. AND W.M. LEWIS, Jr. 1997. Structure of fish assemblages along environmental gradients in floodplain lakes of the Orinoco River. Ecological Monographs 61: 109-128.

SAS (Statistical Analysis Systems). 1985. SAS user's guide: statistics, Version 5. SAS Institute, Cary, North Carolina.

SIEGEL, S. 1956. Nonparametric statistics for the behavioral sciences. McGraw-Hill, New York.

SIMONSON, T.D., J. LYONS, AND P.D. KANEHL. 1994. Quantifying habitat in streams: transect spacing, sample size, and a proposed framework. North

American Journal of Fisheries Management 14: 607-615.

SMITH, C.L., AND C.R. POWELL. 1971. The summer fish communities of Brier Creek, Marshall county, Oklahoma. American Museum Novitates 2458: 1-30.

SUMMERFELT, R.C. 1971. Factors influencing the horizontal distribution of several fishes in a Oklahoma reservoir. In: Hall, G.E., editor. Reservoir fisheries and limnology. American Fisheries Society Special Publication Number 8, Bethesda, Maryland. Pp. 425-439.

TAYLOR C.M., M.R. WINSTON, AND M.J. MATTHEWS. 1993. Fish species-environment and abundance relationships in a Great Plains river system. Ecography 16: 16-23.

TER BRAAK, C.J.F. 1986. Canonical correspondence analysis: a new eigenvector technique for multivariate direct gradient analysis. Ecology 67: 1167-1179.

THORNTON, K.W. 1990. Sedimentary processes. In: Thornton, K.W, B.L. Kimmel, and F.E. Payne, editors. Reservoir limnology: ecological perspectives. John Wiley and Sons, Inc., New York. Pp. 43-70.

WEAVER, M.J., J.J. MAGNUSON, AND M.K. CLAYTON. 1996. Habitat heterogeneity and fish community structure: inferences from north temperate lakes. American Fisheries Society Symposium 16, Bethesda, Maryland. 335-346 Pp.

WILLIAMS, L.R., C.S. TOEPFER, AND A.D. MARTINEZ. 1996. The relationship between fish assemblages and environmental gradients in an Oklahoma prairie stream. Journal of Freshwater Ecology 11: 459-468.

TABLE 1---Sampling effort by water body and gear type at Tishomingo National Wildlife Refuge, Oklahoma. Percent total effort was calculated from the sum of electrofishing, gill net, and seine effort. Letters (A, B, C, D) designate site groups from cluster analysis.

Water body	Abbrev.	Electrofishing (min)	Gill net $(100 \text{ m}^2 \text{ d}^{-1})$	Seine (10 m^2)	Total effort (%)
Bobcat Gulch	BCG	6.75 A	1.11 A	0.0	0.6
Bell Creek	BEL	9.57 A	2.79 A	0.0	1.0
Big Sandy Creek-lower	BSL	14.10 A	1.67 B	0.0	1.3
Big Sandy Creek-upper	BSU	0.00	0.00	14.0 A	1.2
Cumberland Pool	CLP	35.43 A	69.03 B	830.9 D	77.0
Dicks Pond	DKP	8.33 B	0.00	1.1 B	0.8
Goosepen Pond	GPP	8.53 A	2.97 B	0.0	0.9
Lost Lake	LOL	17.50 B	2.97 A	0.0	1.7
McAdams Pond	MAP	12.00 B	2.97 C	0.0	1.2
Muel Lake-east	MLE	0.00	1.11 A	0.0	0.1
Muel Lake-west	MLW	6.40 B	1.11 A	0.0	0.6
Pennington Creek	PEN	13.22 A	2.51 B	2.1 C	1.5
Rock Creek	RCA	7.15 A	2.79 B	0.0	0.8
Reeves Ravine	RVR	7.05 A	2.79 B	0.0	0.8
Twin Pond	TWP	9.05 A	1.11 B	0.0	0.8
Washita River	WAS	29.52	0.00	86.9 D	9.6
All water bodies		184.60	94.95	935.0	100.0

TABLE 2---Habitat characteristics of water bodies at TNWR. Water body abbreviations are shown in Table 1.

Water body	Width (m)	Max. depth (m)	Water clarity (cm)	Flow	Conductivity (μS cm^{-1})	pH	Elevation (m)	Dam
BCG	150.0	1.50	35	0	455.00	8.3	192.0	1
BEL	285.0	2.10	20	0	618.00	8.1	192.0	0
BSL	30.5	2.20	55	1	588.00	7.7	189.0	0
BSU	3.7	0.43	200 [2]	1	650.00	7.5	192.0	0
CLP	4,174.0	5.60 [1]	29	0	547.00	8.5	189.0	0
DKP	117.0	1.00	80	0	228.00	8.5	192.0	1
GPP	72.0	2.10 [1]	58	0	683.00	8.5	189.0	1
LOL	72.0	2.40	78	0	581.00	8.1	192.0	1
MAP	62.0	3.70	98	0	383.00	8.6	195.1	1
MLE	118.0	1.40	19	0	804.00	8.1	195.1	0
MLW	69.0	1.40	35	0	450.00	7.9	195.1	1
PEN	17.3	1.75	31	1	492.00	8.2	189.0	0
RCA	183.0	1.50	20 [3]	0	545.00	8.2	189.0	0
RVR	128.0	2.90	62	0	386.00	8.3	192.0	1
TWP	130.0	1.50	40	0	519.00	8.4	189.0	1
WAS	50.0	3.50	15	2	1,137.00	8.0	189.0	0

[1] Measured under flood conditions (1.58 m above normal).

[2] Estimated because water clarity greater than maximum depth.

[3] Visually estimated.

TABLE 3---Total catch, by gear type, of fish species collected in the Tishomingo National Wildlife Refuge. Letters (A, B, C, D) designate species groups from cluster analysis.

Family Scientific name	Common name	Name code	Total catch Electrofishing	Seine	Gill nets
Anguillidae	eel	-	0	0	0
Anguilla rostrata[1]	american eel	-	0	0	0
Atherinidae	silverside	-	442	34,895	0
Labidesdes sicculus	brook silverside	-	2 x	0	0
Menidia beryllina	inland silverside	tws	440 A	34,895 D	0
Catostomidae	sucker	-	218	89	309
Carpiodes carpio	river carpsucker	rcs	67 A	88 C	56 B
Ictiobus cyprinellus	bigmouth buffalo	bmb	13 B	0	74 B
Ictiobus species[2]	buffalo	ict	132 B	0	179 B
Minytrema melanops	spotted sucker	-	5 x	0	0
Moxostoma erythrurum	golden redhorse	-	1 x	1 x	0
Centrarchidae	sunfish	-	1,155	300	69
Lepomis cyanellus	green sunfish	-	4 x	1 x	0
Lepomis gulosus	warmouth	wam	155 B	5 A	0
Lepomis humilis	orangespotted sunfish	oss	106 B	235 D	0
Lepomis macrochirus	bluegill sunfish	blg	605 B	10 A	0
Lepomis megalotis	longear sunfish	lgs	81 B	21 B	0
Lepomis microlophus	redear sunfish	-	24 x	0	0

TABLE 3---Continued.

Family	Common name	Name Code	Total catch		
Scientific name			Electrofishing	Seine	Gill nets
Micropterus punctulatus	spotted bass	spb	2 x	18 B	0
Micropterus salmoides	largemouth bass	lmb	67 B	0	1 x
Pomoxis annularis	white crappie	whc	112 B	10 x	68 A
Clupeidae	shad	-	887	4,526	1,131
Dorosoma cepedianum	gizzard shad	gzs	812 A	3,062 D	1,131 A
Dorosoma petenense	threadfin shad	tfs	75 A	1,464 D	0
Cyprinidae	minnow	-	95	3,210	35
Campostoma anomalum	central stoneroller	str	0	58 B	0
Cyprinella lutrensis	red shiner	rds	8 x	402 C	0
Cyprinella venustus	blacktail shiner	bts	26 A	481 C	0
Cyprinus carpio	common carp	crp	13 A	0	35 A
Macrhybopsis aestivalis	speckled chub	spc	0	97 C	0
Macrhybopsis storeriana	silver chub	-	0	1 x	0
Notemigonus chysoleuca	golden Shiner	-	5 x	1 A	0
Notropis atherinoides	emerald shiner	-	0	5 x	0
Notropis buchanani	ghost shiner	ghs	17 A	832 D	0
Notropis stramineus	sand shiner	sds	0	67 B	0
Notropis volucellus	mimmic shiner	mms	0	412 D	0
Phenacobius mirabilis	suckermouth minnow	-	0	6 x	0

TABLE 3---Continued.

| Family | | | Total catch | | |
Scientific name	Common name	Name code	Electrofishing	Seine	Gill nets
Pimephales promelas	fathead minnow	-	0	1 x	0
Pimephales vigilax	bullhead minnow	bhm	26 A	847 C	0
Cyprinodontidae	killifish	-	0	8	0
Fundulus zebrinus	plains killifish	plk	0	8 B	0
Hiodontidae	mooneyes	-	0	0	35
Hiodon alosoides	goldeye	gde	0	0	35 B
Ictaluridae	catfish	-	21	50	254
Ameiurus natalis	yellow bullhead	-	0	2 x	1 x
Ictalurus furcatus	blue catfish	bcf	0	12 x	140 B
Ictalurus punctatus	channel catfish	ccf	18 A	34 C	108 B
Pylodictis olivaris	flathead catfish	-	4 x	2 x	5 x
Lepisosteidae	gar	-	64	2	214
Lepisosteus oculatus	spotted gar	spg	49 A	0	26 A
Lepisosteus osseus	longnose gar	lng	8 A	0	121 B
Lepisosteus platostomus	shortnose gar	sng	7 x	2 x	67 B
Percidae	perch	-	82	72	0
Etheostoma gracile	slough darter	sld	1 x	3 A	0
Etheostoma spectabile	orangethroat darter	otd	0	39 B	0
Percina macrolepida	bigscale logperch	blp	81 B	30 x	0

TABLE 3---Concluded.

Family Scientific name	Common name	Name code	Total catch		
			Electrofishing	Seine	Gill nets
Percina sciera	dusky darter	dkd	0	20 B	0
Percichthyidae	temperate bass	-	64	84	158
Morone saxatilis	striped bass	stb	0	0	45 B
Morone chrysops	white bass	whb	64 A	84 C	113 B
Poeciliidae	livebearer	-	10	484	0
Gambusia affinis	mosquitofish	mqf	10 B	484 B	0
Sciaenidae	drum	-	98	440	29
Aplodinotus grunniens	freshwater drum	fwd	98 A	440 D	29 B
Total numbers collected			3,137	50,618	2,234

x indicates species was omitted from analysis.

[1] One adult specimen was captured by trap nets in the Cumberland Pool.

[2] Mostly smallmouth buffalo, Ictiobus bubalus.

TABLE 4---Results from Monte Carlo test on dimensions of the
canonical correspondence analysis (CCA).

Analysis Dimensions	Eigenvalue	F-ratio	P-value
Electroshocking data			
Axis 1	0.21	3.10	< 0.01
Canonical (total)	0.41	2.09	< 0.01
Unconstrained (total)	0.68	NA	NA
Gill net data			
Axis 1	0.19	1.94	0.47
Canonical (total)	0.45	1.58	0.08
Unconstrained (total)	0.86	NA	NA
Seine data			
Axis 1	0.77	NT	NT
Canonical (total)	1.91	NT	NT
Unconstrained (total)	1.91	NA	NA

NT = Not testable
NA = Not applicable

TABLE 5---Correlation of environmental variables with CCA axes.

Analysis Variable	Axis 1	Axis 2	Axis 3
Electrofishing data			
Width	0.58	-0.21	0.61
Depth	0.43	-0.29	0.77
Water clarity	-0.38	0.35	0.55
Conductivity	0.70	0.19	-0.38
Elevation	-0.78	-0.19	0.29
Gill net data			
Width	0.38	-0.47	-0.43
Depth	0.46	-0.30	-0.57
Water clarity	-0.08	0.50	-0.21
Conductivity	0.37	-0.52	0.53
Elevation	0.43	0.64	0.41
Seine data			
Width	-0.47	0.12	-0.49
Depth	-0.80	0.21	-0.48
Water clarity	0.81	-0.56	-0.16
Flow	-0.24	-0.42	0.08
Conductivity	-0.43	-0.39	-0.44
Elevation	0.98	-0.09	-0.19

FIG. 1.---Location of water bodies on the Tishomingo National Wildlife Refuge (Johnston/Marshall Counties, Oklahoma).

FIG. 2.---Ordination diagram from canonical correspondence analysis (CCA) on seine data. Abbreviations for sites and species are given in Tables 1 and 3, respectively.

FIG. 3.---Ordination diagram from CCA on electrofishing data. Abbreviations for sites and species are given in Tables 1 and 3, respectively.

FIG. 4.---Ordination diagram from CCA on gill net data. Abbreviations for sites and species are given in Tables 1 and 3, respectively.

Figure 1

1=Bobcat Gulch
2=Bell Creek
3=Big Sandy Creek (lower)
4=Big Sandy Creek (upper)
5=Cumberland Pool
6=Dicks Pond
7=Goosepen Pond
8=Lost Lake
9=McAdams Pond
10=Muel Lake (east)
11=Muel Lake (west)
12=Pennington Creek
13=Rock Creek
14=Reeves Ravine
15=Twin Pond
16=Washita River

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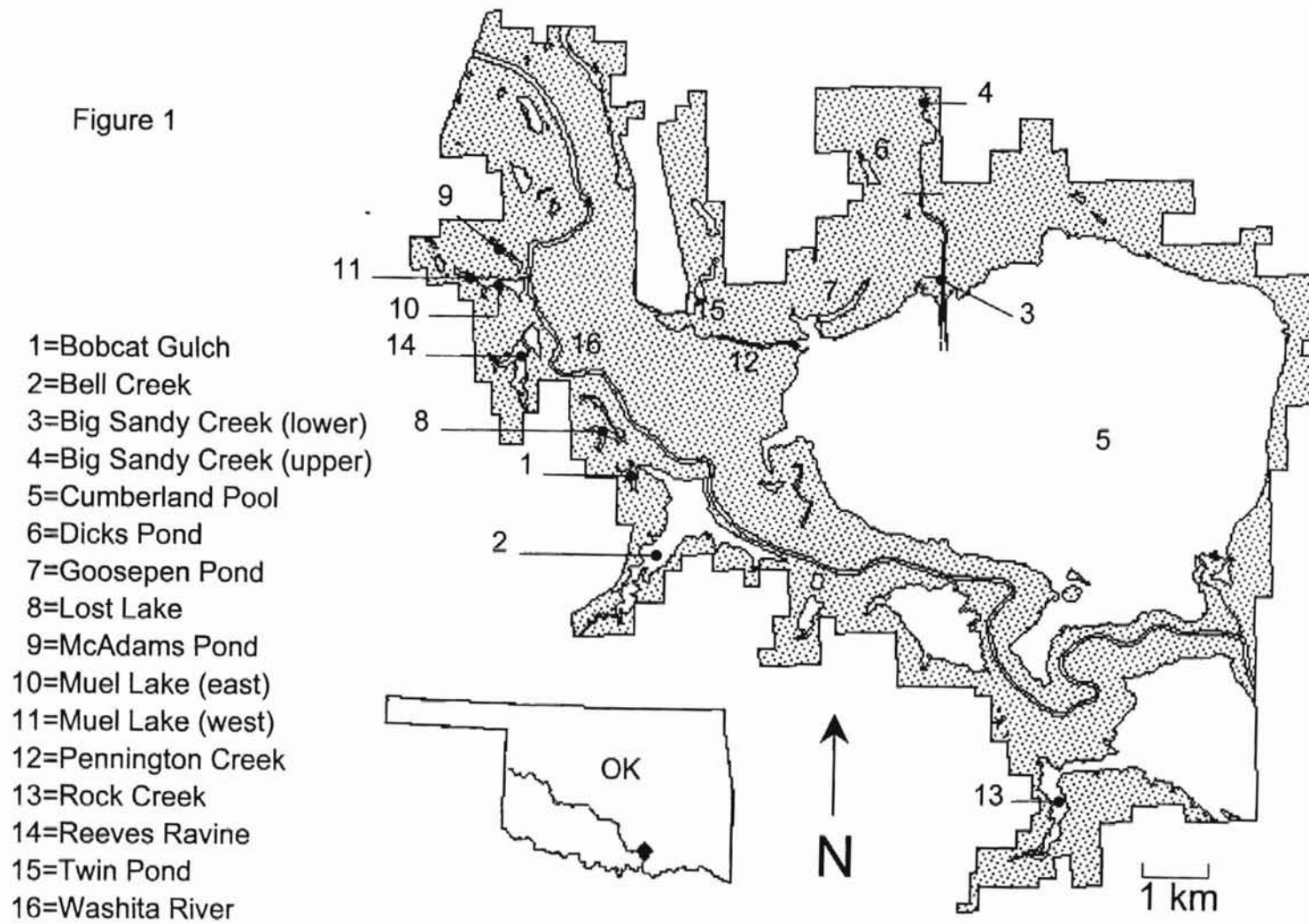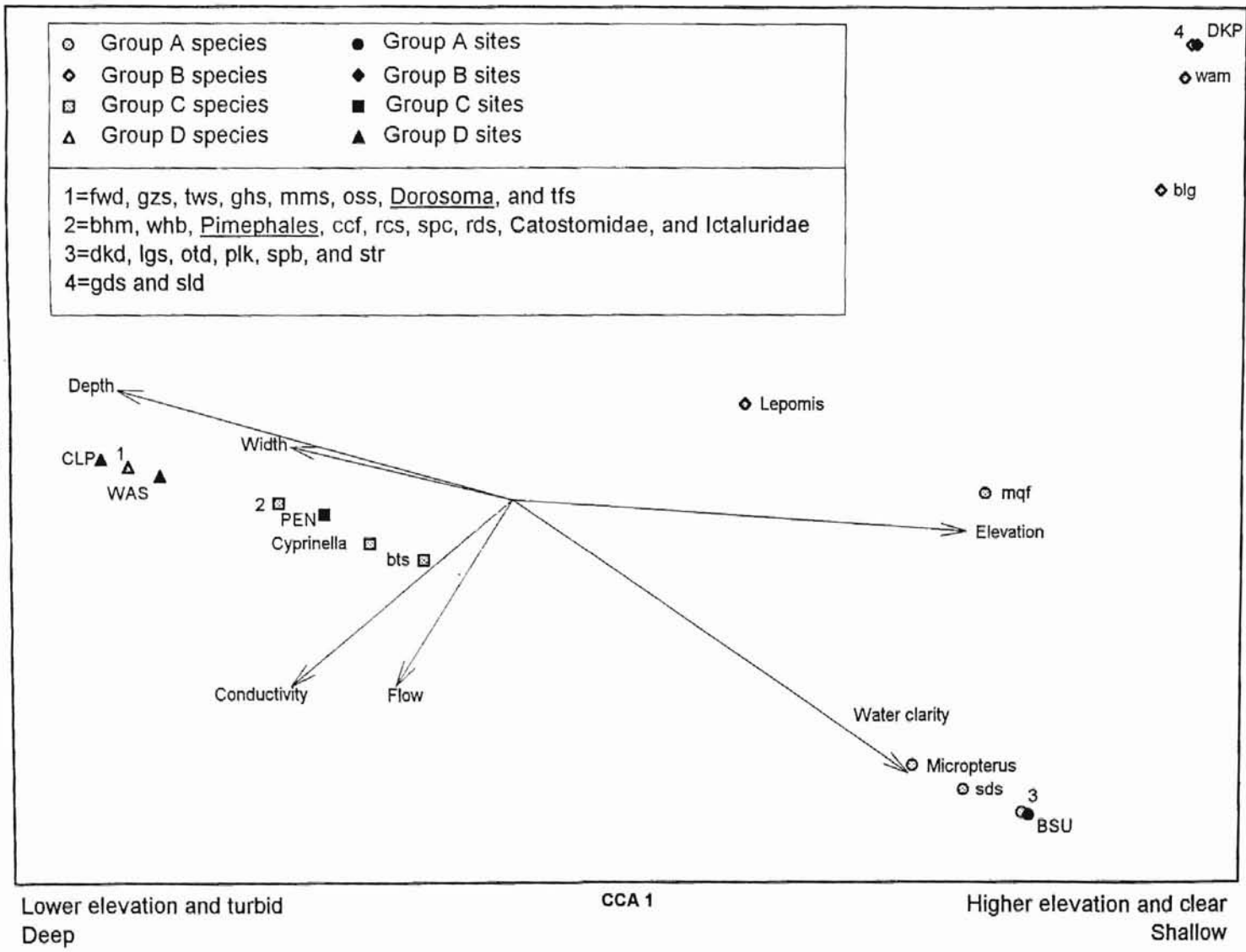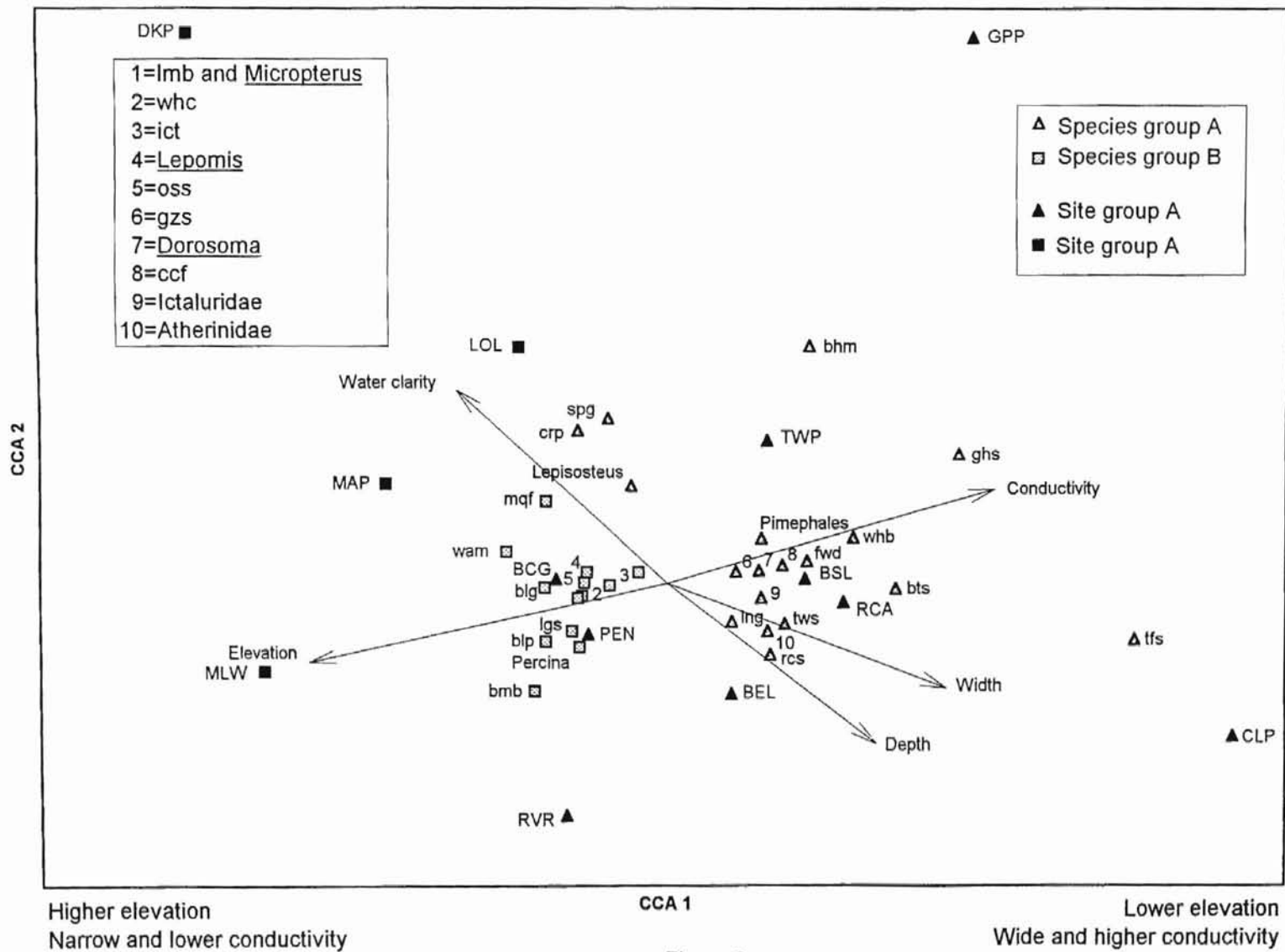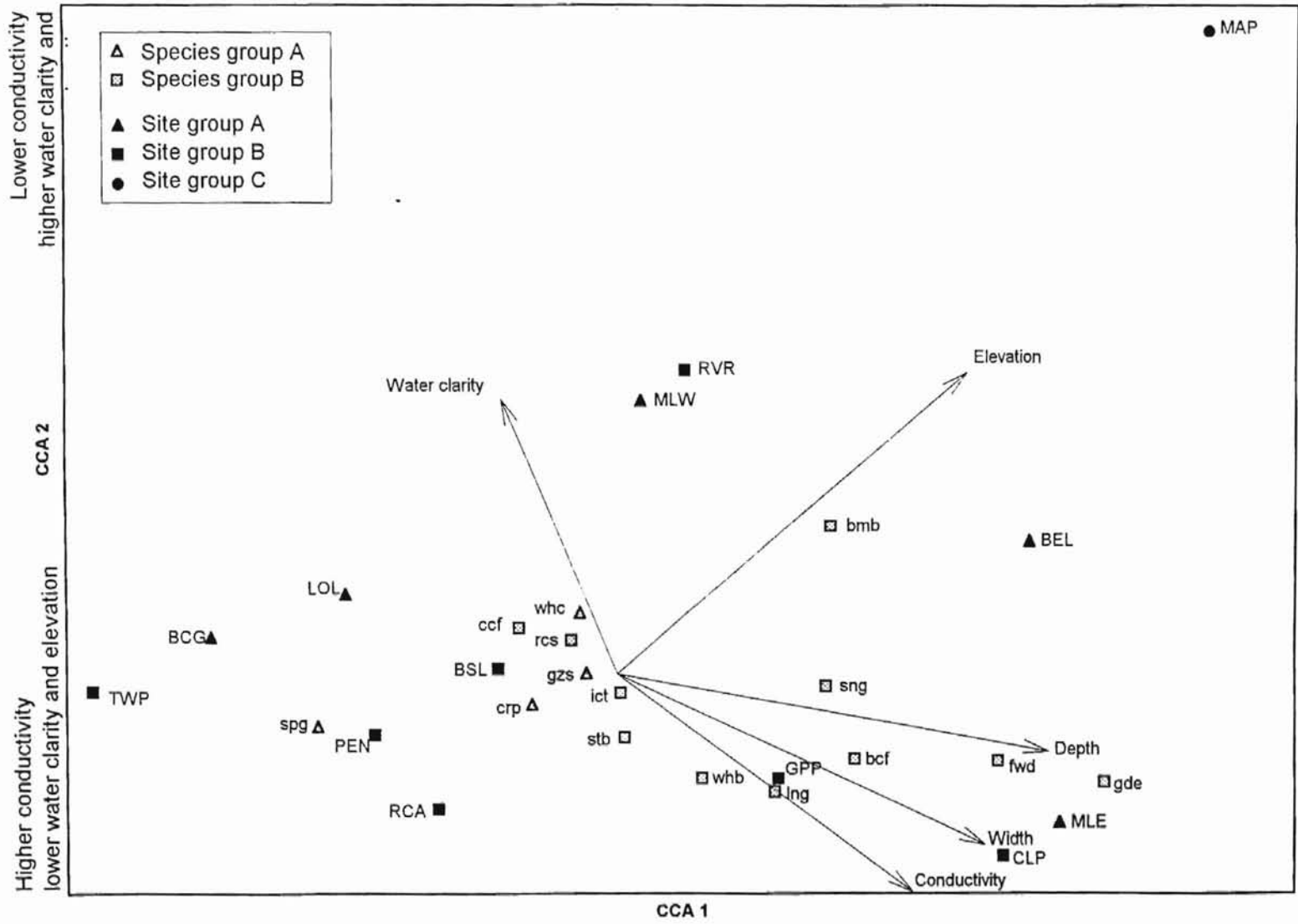
Figure 2

Figure 3

Figure 4

CHAPTER III

EFFECTS OF SEASONAL CHANGES IN HABITAT AVAILABILITY ON FISH

ASSEMBLAGE STRUCTURE AND HABITAT ASSOCIATIONS

IN A LARGE FLOODPLAIN LAKE (OKLAHOMA, U.S.A.)

Synopsis

We sampled the fish fauna of a large floodplain lake within the Lake Texoma (U.S.A.) reservoir system to examine assemblage structure and habitat associations at normal and flood water levels. Overall, adult fish assemblage structure was similar between normal and flood conditions. Within habitat types, fish assemblage structure differed significantly between broad littoral (open habitat at normal water level) and flooded lowland (vegetated habitat at flood water level) habitat types. The difference was due to an increase in vegetation spawners and/or users (primarily *Ictiobus bubalus*) and a decline in open water spawners (primarily *Hiodon alosoides*). Changes in fish-habitat associations were also related to these two ecological groups. Intra-annual variation in the spatial and temporal pattern of habitat availability led to shifts in relative abundance of vegetation users and open water users. An overriding factor structuring the fish assemblage in this lake was low water clarity, which seems to have excluded many centrarchid species from the assemblage.

Introduction

Major changes in reservoir habitats are commonly caused by seasonal water level fluctuations, which primarily affect habitat availability in the littoral zone (Ploskey 1986, Duncan & Kubecka 1995). In north temperate lakes, the structure of littoral fish assemblages can be predicted from regional differences in water temperature, turbidity, aquatic macrophytes, and substrate (Weaver et al. 1996). However, fish assemblage structure in reservoir littoral zones has been difficult to predict because of the effect of fluctuating water levels on habitat availability (Ploskey 1986, Gelwick & Matthews 1990, Weaver et al. 1996).

Changes in fish assemblage structure are most dramatic between open and vegetated littoral zones. Gelwick & Matthews (1990) found that fish assemblages in vegetated littoral zones of a large reservoir consisted of large benthic fishes and large centrarchids while assemblages in open littoral zones consisted of schooling species, small cyprinids, small centrarchids, and temperate basses (Gelwick & Matthews 1990). In most reservoirs, the relative proportion of vegetated and open littoral habitats changes with seasonal water level fluctuations (Gelwick & Matthews 1990). In Oklahoma reservoirs, peak flooding generally occurs in spring (Cone et al. 1986) resulting in an increase in vegetated littoral habitat and a decline in open littoral habitat (Gelwick & Matthews 1990). However, flood pulses in prairie river systems tend to be brief and unpredictable relative to lowland river systems (Matthews 1988, Junk et al.

1989). Consequently, many fish species have limited adaptations for direct use of the aquatic/terrestrial transition zone (ATTZ; Junk et al. 1989).

Habitat availability in aquatic ecosystems can influence fish assemblage structure by altering spawning success (Ploskey 1986) and habitat-mediated biotic interactions (Stang & Hubert 1984, Noble 1986, Wootton 1990). Thus, the spatial and temporal pattern of littoral habitat availability can influence the dynamics of fish assemblage structure in reservoirs and other aquatic ecosystems with widely fluctuating water levels. Gelwick & Matthews (1990) suggested that to understand the effects of water level fluctuations on assemblage structure, one must compare assemblage structure within habitat types at different water levels.

Our study focuses on fish assemblage structure within habitat types at normal and flood water levels in a large floodplain lake (the Cumberland Pool, Oklahoma, U.S.A.) along the Washita River arm of Lake Texoma. Habitat types in the Cumberland Pool are comparable to those described in Gelwick & Matthews (1990) for Lake Texoma. The specific questions we address in this paper are: 1) Does fish assemblage structure change within littoral zones at different water levels? 2) Do habitat associations of individual species change at different water levels? 3) Are there identifiable ecological associations that could explain the observed changes in fish assemblage structure and fish-habitat associations? 4) What are the effects of seasonal changes in habitat availability in the ATTZ on fish assemblage structure in this floodplain lake?

Methods

The Cumberland Pool is within Tishomingo National Wildlife Refuge in southcentral Oklahoma (Fig. 1). The 21.9 km^2 lake is turbid (mean mid-lake Secchi depth = 28 cm), shallow (max. depth = 4.5 m), and has a very low shoreline development index (SDI = 1.44). By comparison, Lake Texoma is 360 km^2, relatively clear (mid-lake Secchi depths 0.5 to 1.25 m), much deeper (max. depth = 28 m) and has an SDI of 13.9 (Texas Parks & Wildlife Department; unpublished data). The Cumberland Pool was created when the dam for Lake Texoma was built in 1944.

Habitat types in the Cumberland Pool were delineated from color infrared aerial photographs taken in 1991. The separation of nearshore and open water habitat types was made at the 1-m bathymetric contour line (at normal water level). The littoral zone is typically defined as the area from the waterline out to the depth of 1% light penetration (Sly & Busch 1992); however, applying this criteria to the Cumberland Pool would have resulted in areas too shallow to be sampled effectively. The decision to use a depth of 1 m was based on an estimated maximum depth of the wave impact zone (½ wavelength + ½ waveheight; Nybakken 1993).

Nearshore habitat was then separated based on distinct differences in bottom gradient and structural heterogeneity. To distinguish flooded habitats from habitats at normal water levels, we used water level data for the Cumberland Pool (1988 - 1996) to find the maximum level occupied less than

25% of the time (flood level) and the minimum level occupied greater than 75% of the time (normal level). The mean of the two values was used to separate habitats under normal and flood conditions. This roughly corresponded to the boundary between low water and land colonized by pioneering vegetation (i.e., ATTZ; Junk et al. 1989).

The nearshore habitats at normal water level (189.5 m; Fig. 2) were divided into the narrow littoral, broad littoral, and flooded timber zones (Fig. 1,3). The narrow littoral was the most heterogeneous habitat with small stands of submerged dead trees between sections of open littoral habitat with cobble, sand, or clay substrate on a moderate gradient (slope $\approx$ 8%). The broad littoral consisted of relatively homogeneous silty substrate on a low gradient (slope $\approx$ 3%) with little structural heterogeneity. The flooded timber consisted of large stands of submerged dead trees.

Habitats available under flood conditions (>189.5 m; Fig. 2) were designated as flooded highland and flooded lowland zones (Fig. 1,3). The flooded highland was the steeply sloping shoreline of the narrow littoral zone under flood conditions. The flooded lowland was located on the Washita River floodplain adjacent to the broad littoral and flooded timber habitats, and it varied in composition from flooded annual vegetation to flooded forest with sparse understory brush. Each habitat had a shoreline length of approximately 5,000 m.

To account for species abundances among habitat types, we measured

physicochemical parameters at each sample site during summer 1996. A multiparameter-water-quality monitoring instrument (Hydrolab Scout 2) was used to gather data on surface water temperature, conductivity, dissolved oxygen, and pH. Water clarity was measured with a Secchi disk.

Fish were sampled with experimental gill nets and trap nets in a stratified random design with equal sampling effort among habitats. The experimental gill nets were 91 m long with 15-m panels of 10.2-, 8.9-, 7.6-, 6.4-, 5.1-, and 3.8-cm mesh (square). Gill nets used in nearshore areas were 0.9 m deep, whereas those used in open water areas ranged from 1.8 to 2.4 m deep. All gill nets were set on the bottom in all habitats except the flooded timber and some areas of the flooded lowland. Nearshore gill nets were set in an L-shape, perpendicular from shore, with the smallest mesh in the shallowest water. Perpendicular sections were set to a maximum depth of 1.5 m or a maximum distance of 30 m (2 sections) from shore. Open-water gill nets were set in the direction of the wind. Trap nets were of the modified fyke design described in Hubert (1996) with 18-m leads. Only nearshore habitats were sampled with trap nets. Trap nets were set perpendicular to the shoreline.

Sampling was conducted approximately every two weeks from May to November 1996 and from March to November 1997. Samples were grouped by 3-day periods during which each gear was deployed with a maximum daily effort (5 trap nets or 2 gill nets) distributed among habitat types. Both gear types were set at dusk and retrieved the following morning (11-14 hour set time).

Specimens were identified to species, measured to the nearest millimeter and released. Catch per unit effort (CPUE) for trap nets was the number of fish per nocturnal set. For gill nets, CPUE was calculated as the number of fish per 100 m^2 of netting per nocturnal set.

For our analysis, we included only species represented by more than 5 individuals in each group of samples (normal and flood water levels). We also eliminated from the analysis small fish (< 80 mm total length) captured in trap nets because they were likely underestimated (Hubert 1996). We used Spearman rank correlation (r_s) to test for differences in overall assemblage structure (Gelwick & Matthews 1990) between water levels and between parallel habitats at normal and flood water level (e.g. broad littoral and flood lowland). For correlations that were not significant (indicating a difference in fish assemblage structure), we performed pairwise comparisons with a Wilcoxin 2-sample test to identify the species responsible for the difference. This test was used because the sample data were not normally distributed (Shapiro-Wilk test, $p < 0.05$).

To identify species-habitat associations, we used the Kruskall-Wallis test to determine significant differences in the rank abundance of species among habitat types at normal and flood water levels. For species that differed significantly ($p < 0.05$), we performed paired comparisons with a Wilcoxin 2-sample test ($p < 0.05$). To test for significant differences in physicochemical parameters we used oneway ANOVA. For parameters that were significantly

different (p < 0.05), we performed pairwise comparisons using Tukey-Kramer HSD test because the data were normally distributed (Shapiro-Wilk test, p < 0.05).

Results

Over the study period, a total of 59 gill nets and 71 trap nets were deployed in the Cumberland Pool. The most frequently captured species was gizzard shad, *Dorosoma cepedianum* (Table 1). Other common species (in descending order of abundance) were white crappie, *Pomoxis annularis*, shortnose gar, *Lepisosteus platostomus*, *Ictiobus* spp. (mostly smallmouth buffalo, *Ictiobus bubalus*), longnose gar, *Lepisosteus osseus*, blue catfish, *Ictalurus furcatus*, and white bass, *Morone chrysops*.

Of the physicochemical parameters measured, only Secchi depth was significantly different among habitat types at normal water levels ($n = 24$, $F = 4.88$, $p = 0.011$). Pairwise comparisons revealed that water clarity in the narrow littoral and open water was greater than in the broad littoral ($p < 0.05$). Mean Secchi depths were 29 cm in the narrow littoral and open water and 19 cm in the broad littoral.

CPUE Considerations

Inferring abundance for species caught in gill nets and trap nets rests on the assumption that CPUE is proportional to stock density. However, CPUE from passive sampling gears more accurately reflects the relative activity of a species (Hubert 1996). Catch per unit effort can also be affected by the vertical distribution of fishes in the water column. Rinne et. al (1981) and Hover (1976) concluded that vertical distribution of most species in reservoirs is limited to water with dissolved oxygen levels >2 mg l^{-1} (i.e., zone of adequate oxygen; Rinne 1981). The lowest dissolved oxygen measured on the bottom of the Cumberland Pool was 2.56 mg l^{-1} (n = 33), but dissolved oxygen was usually between 6.18 and 7.26 mg l^{-1} (95% confidence interval). None of the nets were set in anoxic bottom water, so oxygen levels likely did not affect abundance estimates.

Within the zone of adequate oxygen, differences occur in their vertical distribution among species. Hover (1976) identified three basic patterns in the vertical distribution of fishes in the main pool of Lake Eufaula (Oklahoma, USA): (1) species that were bottom-oriented (common carp, *Cyprinus Carpio* and gizzard shad), (2) species that were surface-oriented (goldeye, *Hiodon alosoides*, shortnose gar, white crappie, white bass, and freshwater drum, *Aplodinotus grunniens*), and (3) species that were randomly distributed in the water column (channel catfish, *Ictalurus punctatus*). Based on these distribution patterns and the location of our sampling gear, we may have underestimated the abundance of surface-oriented species in deep water habitats (i.e. narrow

littoral, flooded highland, and open water). Randomly-oriented species were probably sampled proportionately, whereas bottom-oriented species were likely overestimated.

Changes in fish assemblages structure and fish-habitat associations

The overall structure of the Cumberland Pool fish assemblage did not change significantly between normal and flood water level (Table 2). However, between parallel habitat types at normal and flood water level, fish assemblage structure differed between the broad littoral and flooded lowland habitats (Table 2). Although we did not sample small fish or differentiate among solar seasons (which were correlated with water level fluctuation; Pearson r = -0.616), our results are similar to those of Gelwick & Matthews (1990); the greatest change in assemblage structure was between open and vegetated littoral zones. The difference in structure was caused mainly by a decline in goldeye CPUE (p = 0.014) and an increase in smallmouth buffalo CPUE (p = 0.001) in the transition from broad littoral to flooded lowland habitat. Assemblage structure was persistent among the other parallel habitat types with similar characteristics at different water levels (i.e. flooded timber and flooded lowland). Gelwick & Matthews (1990) also found that assemblage structure within wooded littoral zones (flooded timber) was persistent across seasons.

Several species demonstrated persistent habitat associations at both

normal and flood water levels. Blue catfish abundance was higher in open water than in narrow littoral habitat and this pattern persisted in the flooded habitats (Table 3). In nearshore habitats, white crappie were most abundant in both the narrow littoral and flooded highland habitats (Table 4). Greatest abundances of gizzard shad were in the broad littoral at normal water level and in the open water at both water levels (Table 3).

Some species were associated with specific habitats at normal water levels but not with their counterpart during flood levels, and visa versa. Goldeye abundance was highest in both nearshore habitats at normal water levels (Table 3). Goldeye were found to be surface-oriented by Hover (1976), so their strong association with shallow-water habitats may be a function of the sampling gear spanning the entire water column in these habitats. Common carp and shortnose gar were associated with the flooded lowland but not with its counterparts, the broad littoral and flooded timber (Tables 3 & 4).

Ecological associations

The most obvious explanation for the observed change in fish assemblage structure between the broad littoral and flooded lowland habitats was the spawning activity of phytophilic fishes. Fishes that are members of the phytophilic spawning guild (including common carp, *Ictiobus* species, *Lepisosteus* species, and white crappie; Balon 1975, Gelwicks 1995) respond

specifically to presence of flooded terrestrial vegetation or aquatic macrophytes. In general, these species were more abundant in the flooded lowland than in other habitats (Tables 3 & 4) with two exceptions (bigmouth buffalo *Ictiobus cyprinellus* and white crappie). Bigmouth buffalo and white crappie were most abundant in the flooded highland, where small patches of flooded lowland were available. The close proximity of white crappie to the narrow littoral habitat at normal water level could explain this pattern.

As might be expected, pelagophilic and litho-pelagophilic spawners (including freshwater drum, gizzard shad, and goldeye; Balon 1975; Gelwicks 1995) were less abundant in the flooded lowland than in the open water habitats (Table 3,4). However, only gizzard shad were significantly more abundant in the open water habitat during flood conditions.

Discussion

Seasonal flooding in vegetated lowlands has been correlated with increased growth, reproductive success, recruitment and production of many fish species (Cone et al. 1986, Ploskey 1986, Junk et al. 1989, Gelwicks 1995). Our analysis of fish-habitat associations did not distinguish between use of areas for feeding versus spawning. However, it is well known that common carp, smallmouth buffalo, and shortnose gar use flooded areas for foraging and as spawning habitat (Balon 1975, Cone et al. 1986, Ploskey 1986, Gelwicks 1995),

which explains their strong association with flooded lowland habitat in the Cumberland Pool. These species and other phytophilic spawners (bigmouth buffalo, longnose gar, spotted gar *Lepisosteus oculatus*, white crappie; Balon 1975, Gelwicks 1995) may depend on seasonal flooding to survive and proliferate in this floodplain lake.

Presumably during years when the floodplain is inundated in spring, common carp, smallmouth buffalo, shortnose gar, and other phytophilic spawners will contribute more recruits to the Cumberland Pool fish assemblage. Seasonal flooding in lowland, high order, riverine systems is prolonged and predictable, whereas it is short and unpredictable in low order streams (Junk et al. 1989, Bayley 1995, Sparks 1995). The lack of sufficient floodplain inundation could result in a shift in fish assemblage structure toward greater relative abundances of species that do not require flooded terrestrial vegetation for enhanced growth (i.e. gizzard shad, freshwater drum, blue catfish, striped bass; Cone et al. 1986, Rutherford et al. 1995) and reproduction (i.e. gizzard shad, freshwater drum, goldeye, river carpsucker, flathead catfish, channel catfish, white bass, striped bass; Balon 1975, Ploskey 1986, Gelwicks 1995). Lack of floodplain inundation could occur in drought years or it could be a regular occurrence in low order or heavily modified (impounded, channelized) streams (Junk et al. 1989, Bayley 1995, Rutherford et al. 1995, Sparks 1995).

At least two factors may confound the effects of floodplain inundation on fish populations. First, fish stocks do not always increase, even when a

successful spawn is achieved (Ploskey 1986); other factors such as predator-prey (Noble 1986) and competitive interactions (von Geldern 1971), and availability of suitable environmental conditions (Summerfelt 1975) play a role in limiting species abundance. Second, some species may not spawn or rear young successfully in the Cumberland Pool. Further, periodic immigration of species from the Washita River during flood pulses or from the feeder creeks may serve to sustain fish populations in the Cumberland Pool .

Although habitat-mediated biotic interactions could not be adequately evaluated in our study, they are important in determining recruitment dynamics of fishes (von Geldern 1971, Noble 1986). At flood water levels, small and juvenile fish reduce predation risk and competition by the increase in shallow, structurally complex habitats in lowland areas (Junk et al. 1989, Power et al. 1995). When the water recedes to normal levels, their vulnerability (and later recruitment) depends on habitat heterogeneity and characteristics of the existing aquatic community (Weaver et al. 1996). These ecological functions could alter predictions of fish assemblage change made solely on reproductive success.

If immigration from adjacent riverine systems during seasonal flooding was a significant factor influencing fish assemblage structure, we should have detected a difference in assemblage structure between water levels, but there was no difference. However, the short duration of our study may not have captured the significance of flooding in the Cumberland Pool except to document seasonal access to foraging and spawning habitat for fishes that depend on

floodplain vegetation. Seasonal flooding and immigration may only have a significant influence on fish assemblage structure after years of isolation from the river (Halyk & Balon 1983, Rodriguez & Lewis 1997).

The differences we found in fish-habitat associations between the broad littoral (open) and flooded lowland (vegetated) were similar to those observed by Gelwick & Matthews (1990) except for the rarity of large, adult centrarchids (*Micropterus salmoides* and *Lepomis macrochirus*) and the lack of small species (*Menidia beryllina, Cyprinella lutrensis, C. venusta, Pimephals vigilax*, and juvenile *Dorosoma* species) in our samples. In Lake Texoma, the abundance of small species was also a major differentiating factor between open and vegetated littoral zones (Gelwick & Matthews 1990). The rarity of large, adult centrarchids in the Cumberland Pool is related to differences in habitat characteristics between these two water bodies.

We failed to collect a single adult centrarchid other than white crappie and orangespotted sunfish while electrofishing in the narrow littoral habitat in summer 1996 (Chappell & Fisher, unpublished data). We did, however, collect substantial numbers of adult bluegill, *Lepomis macrochirus*, warmouth, *Lepomis gulosus*, redear sunfish *Lepomis microlophis*, and largemouth bass *Micropterus salmoides* in the lower reach of Big Sandy Creek which flows into the Cumberland Pool (Figure 1). Gelwick & Matthews (1990) also found greatest abundances of adult centrarchids along shady creek channels in summer. Access to calm, shaded waters in summer is likely an important factor limiting

the abundance of large, adult centrarchids in some reservoirs of the southern Great Plains.

Another important difference between our findings and those of Gelwick & Matthews (1990) was the spatial resolution of habitat types. Habitat types comparable in size to the broad littoral/flooded lowland (5,000-m perimeter) occur in riverine sections of reservoirs (Thornton 1990). Gelwick & Matthews (1990) sampled more lacustrine sections of Lake Texoma. These sections were most similar to the narrow littoral/flooded highland habitat types in the Cumberland Pool. Considering these differences, our data showed that large phytophilic algivore/benthivores (common carp, smallmouth buffalo) seem to prefer large areas of flooded vegetation, whereas a phytophilic piscivore/ insectivore (white crappie) seems to prefer patchy flooded vegetation.

A primary factor structuring the fish assemblage of the Cumberland Pool is low water clarity, which seems to have excluded many large centrarchid species from this floodplain lake. Turbidity (low water clarity) is an important factor in structuring lacustrine fish assemblages (Weaver et al. 1996; Rodriguez & Lewis 1997). The most common fishes in this floodplain lake (gizzard shad, white crappie, common carp, smallmouth buffalo, *Lepisosteus* species, and ictalurids) are all tolerant of low water clarity (Miller & Robison 1973).

Further research on the fish assemblage structure in floodplain lakes of reservoirs should focus on the functional response of ecological groups (i.e. vegetation using and/or vegetation spawning and all other fishes) to annual

differences in the spatial and temporal pattern of habitat availability in the ATTZ. Emphasis should be placed on the relative contribution (in terms of growth and survival) of those ecological groups to the fish assemblage under different flood conditions.

Acknowledgements

References cited

Balon, E.K. 1975. Reproductive guilds of fishes: a proposal and definition. J. Fish Res. Board Can. 32(6): 821-864.

Bayley, P.B. 1995. Understanding large river-floodplain ecosystems. Bioscience. 45(3): 153-158.

Cone, R.S., K. Barbour, M. Russell & S.K. Simonet. 1986. The effects of flooding on the growth rates of fishes in Lake Texoma. Proc. Oklahoma Acad. Sci. 66: 21-25.

Duncan, A. & J. Kubecka. 1995. Land/water ecotone effects in reservoirs on the fish fauna. Hydrobiologia. 303: 11-30.

Gelwicks, G.T. 1995. Fish movements between the lower Missouri River and a managed floodplain wetland in Missouri. M.S. thesis, University of Missouri - Columbia. 190 pp.

Gelwick, F.P. & W.J. Matthews. 1990. Temporal and spatial patterns in littoral-zone fish assemblages of a reservoir (Lake Texoma, Oklahoma-Texas (U.S.A.). Env. Biol. Fish. 27: 107-120.

Halyk, L.C., & E.K. Balon. 1983. Structure and ecological production of the fish taxocene of a small floodplain system. Canadian Journal of Zoology 61: 2446-2464.

Hover, J.H. 1976. Vertical distribution of fishes in the central pool of Eufaula Reservoir, Oklahoma. M.S. thesis, Oklahoma State University. 62 pp.

Hubert, W. 1996. Passive capture techniques. pp. 157-192. In: B.R. Murphy & D.W. Willis (ed.) Fisheries Techniques, 2nd Edition. American Fisheries

Society, Bethesda, Maryland.

Junk, W.J., P.B. Bayley, and R.E. Sparks. 1989. The flood pulse concept in river floodplain systems. pp. 110-127. In: D.P. Dodge (ed.) Proceedings of the International Large River Symposium. Can. Spec. Publ. Fish. Aquat. Sci. 106.

Matthews, W.J. 1988. North American prairie streams as systems for ecological study. J.N. Am. Benthol. Soc. 7(4): 387-409.

Miller, R.J. & H.W. Robison. 1973. The fishes of Oklahoma. Oklahoma State University Press, Stillwater. 246 pp.

Noble, R.L. 1986. Predator-Prey interactions in reservoir Communities. pp 137-143. In: G.E. Hall and M.J. Van Den Avyle (ed.) Reservoir Fisheries Management: Strategies for the 80's. Southern Division American Fisheries Society, Bethesda, Maryland.

Nybakken, J.W. 1993. Marine Biology, 3rd Edition. Harper Collins College Publishers. 462 pp.

Ploskey, G.R. 1986. Effects of water level changes on reservoir ecosystems, with implications for fisheries management. pp. 86-97. In: G.E. Hall & M.J. Van Den Avyle (ed.) Reservoir Fisheries Management: Strategies for the 80's. Southern Division American Fisheries Society, Bethesda, Maryland.

Power, M.E., A. Sun, G. Parker, W.E. Dietrich & J.T. Wootton. 1995. Hydraulic food-chain model. Bioscience 45(3): 159-167.

Rinne, J.N., W.L. Minckley & P.O. Bersell. 1981. Factors influencing fish

distribution in two desert reservoirs, Central Arizona. Hydrobiologia 80: 31-42.

Rodriguez, M.A. & W.M. Lewis, Jr. 1997. Structure of fish assemblages along environmental gradients in floodplain lakes of the Orinoco River. Ecological Mongraphs 61: 109-128.

Rutherford, D.A., W.E. Kelso, C.F. Bryan & G.C. Constant. 1995. Influence of physicochemical characteristics on annual growth increments of four fishes from the lower Mississippi River. Trans. Am. Fish. Soc. 124: 687-697.

Sly, P.G. & W.-D.N. Busch. 1992. A system for aquatic habitat classification oflakes. pp. 15-26. In: W.-D.N. Busch & P.G. Sly (ed.) The Development ofan Aquatic Habitat Classification System for Lakes. CRC Press, Inc. Boca Raton, Florida.

Sparks, R.E. 1995. Need for ecosystem managment of large rivers and their floodplains. Bioscience. 45(3): 168-182.

Stang, D.L. & W.A. Hubert. 1984. Spatial separation of fishes captured in passive gear in a turbid prairie lake. Env. Biol. Fish. 11: 309-314.

Summerfelt, R.C. 1975. Relationship between weather and year-class strength of largemouth bass. pp. 166-174. In: R.H. Stroud & H. Clepper (ed.) Black Bass Biology and Managment. Sport Fishing Institute, Washington.

Thorton, K.W. 1990. Sedimentary process. pp. 43-70. In: K.W. Thornton, B.L. Kimmel, and F.E. Payne (ed.) Reservoir Limnology: Ecological

Perspectives. John Wiley & Sons, Inc.

von Geldern, C.E., Jr. 1971. Abundance and distribution of fingerling largemouth

bass, Micropterus salmoides, as determined by electrofishing at Lake

Nacimiento, California. Calif. Fish & Game 57: 228-245.

Weaver, M.J., J.J. Magnuson & M.K. Clayton. 1996. Habitat heterogeneity and

fish community structure: inferences from north temperate lakes.

American Fisheries Society Symposium 16, Bethesda, Maryland. pp. 335-

346.

Wootton, R.J. 1990. Ecology of Teleost Fishes. Chapman & Hall, London.

404pp.

Table 1. Total catch from gill nets and trap nets for each analysis group. Relative abundance based on total catch of species in both gill nets and trap nets. [1] = mostly *Ictiobus bubalus*

Species	Normal water levels		Flood water levels		Overall relative abundance (%)
	Gill nets (n = 23)	Trap net (n = 45)	Gill nets (n = 26)	Trap net (n = 26)	
Dorosoma cepedianum	564	14	396	16	41.2
Pomoxis annularis	25	115	18	102	10.8
Lepisosteus platostomus	24	86	40	60	8.7
Ictiobus species.[1]	50	13	75	43	7.5
Lepisosteus osseus	29	12	90	7	5.7
Ictalurus furcatus	73	0	57	0	5.4
Morone chrysops	48	12	42	11	4.7
Carpiodes carpio	22	18	20	25	3.5
Ictalurus punctatus	43	0	16	0	2.5
Ictiobus cyprinellus	26	2	22	1	2.1
Morone saxatilis	12	0	29	0	1.7
Hiodon alosoides	24	0	10	0	1.4
Lepisosteus oculatus	5	9	5	15	1.4
Aplodinotus grunniens	21	3	7	1	1.3
Cyprinus carpio	6	1	14	9	1.2
Lepomis macrochirus	0	5	0	4	0.4
Pylodictis olivaris	2	0	2	0	0.2

Table 2. Spearman rank correlation coefficient (r_s) of species ranks for comparisons of assemblage structure among seasons (normal and flood). One-tailed student's t used to test for significance. [g] = gill net data, t = trap net data.

Comparison	r_s	P-value	Interpretation for assemblage structure
normal-flood (all habitats)[g,t]	0.723	< 0.001	not significantly different
broad littoral-flooded lowland [g,t]	0.204	0.340	significantly different
narrow littoral-flooded highland[g,t]	0.628	0.001	not significantly different
open water-open water (flood)[g]	0.639	0.010	not significantly different
flooded timber-flooded lowland[t]	0.745	0.021	not significantly different

Table 3. Mean gill net catch per unit effort (CPUE) for species by habitat type and water level. The p-values are from Kruskall-Wallis tests for differences in CPUE among habitat types. The * denotes significance at the α = 0.05 level. Means in the same row for each habitat type with the same letter are not significantly different (p < 0.05). NL = narrow littoral, BL = broad littoral, OW = open water, FH = flooded highland, and FL = flooded lowland. [1] = mostly *Ictiobus bubalus*

Species	Normal water levels				Flood water levels			
	NL (n = 6)	BL (n = 6)	OW (n = 11)	χ^2 P-value	FH (n = 9)	FL (n = 8)	OW (n = 9)	χ^2 P-value
Aplodinotus grunniens	0.400	0.200	0.771	0.082	0.136	0.150	0.214	0.570
Carpiodes carpio	1.197	1.796	0.135	0.073	1.331	0.449	0.415	0.994
Cyprinus carpio	0.400	0.400	0.129	0.599	0.000 [b]	1.836 [a]	0.036 [b]	0.001 *
Dorosoma cepedianum	5.179 [b]	23.015 [a]	18.836 [a]	0.024 *	4.864 [b]	6.336 [b]	14.493 [a]	0.014 *
Hiodon alosoides	1.379 [a,b]	2.000 [a]	0.336 [b]	0.043 *	0.271	0.150	0.314	0.664
Ictalurus furcatus	0.400 [b]	1.593 [a,b]	2.921 [a]	0.031 *	0.500 [b]	0.979 [a,b]	2.357 [a]	0.033 *
Ictalurus punctatus	3.964	2.393	0.421	0.132	1.293	0.229	0.186	0.830
Ictiobus cyprinellus	0.200	1.800	0.607	0.289	1.457	0.529	0.386	0.833
Ictiobus species[1]	1.586	1.200	1.657	0.474	0.893	5.922	1.214	0.057
Lepisosteus oculatus	0.112	0.000	0.007	0.179	0.052	0.038	0.000	0.188
Lepisosteus osseus	2.179	0.986	0.736	0.115	1.629	4.436	2.021	0.320
Lepisosteus platostomus	0.165	0.112	0.080	0.801	0.055 [c]	0.330 [a,b]	0.147 [b]	0.008 *
Morone chrysops	3.572	2.971	0.693	0.091	1.093	1.736	1.057	0.491
Morone saxatilis	0.593	1.593	0.000	0.158	1.264	0.600	0.521	0.641
Pomoxis annularis	0.028	0.055	0.136	0.126	0.070	0.090	0.060	0.772

Table 4. Mean trap net catch per unit effort (CPUE) for species by habitat type and water level. The p-values are from Kruskall-Wallis tests for differences in CPUE among habitat types. The * denotes significance at the α = 0.05 level. Means in the same row for each habitat type with the same letter are not significantly different (p < 0.05). NL = narrow littoral, BL = broad littoral, FT = flooded timber, FH = flooded highland, and FL = flooded lowland. [1] = mostly *Ictiobus bubalus*

Species	Normal water level				Flood water level		
	NL (n = 14)	BL (n = 16)	FT (n = 15)	χ^2 P-value	FH (n = 13)	FL (n = 13)	χ^2 P-value
Carpiodes carpio	0.500	0.188	0.500	0.417	0.462	1.462	0.193
Cyprinus carpio	0.000	0.188	0.000	0.147	0.077	0.615	0.956
Dorosoma cepedianum	0.077	0.235	0.600	0.090	0.308	0.923	0.511
Ictiobus species.[1]	0.077	0.235	0.533	0.049	0.308 [b]	3.000 [a]	0.003 *
Lepisosteus oculatus	0.071	0.063	0.438	0.192	0.154	1.000	0.526
Lepisosteus osseus	0.385	0.118	0.333	0.768	0.231	0.308	0.917
Lepisosteus platostomus	1.214	2.500	1.813	0.471	0.230	4.385	0.049
Morone chrysops	0.277	0.717	0.352	0.048	0.308	0.538	0.869
Pomoxis annularis	5.429 [a]	0.688 [b]	1.688 [b]	0.001 *	6.692 [a]	1.154 [b]	0.004 *

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Fig. 1. Habitat types in the Cumberland Pool at normal water levels.   The locationof the Cumberland Pool in the Tishomingo National Wildlife Refuge is shown along with its location in Oklahoma.

Fig. 2. Water regime in the Cumberland Pool over the study period. Cumberland Pool levels are in meters (m) above sea level and Washita River flow is in cubic meters persecond (cms).   The dotted line represents maximum normal water level.

Fig. 3. Habitat types at normal and flood water levels in the Cumberland Pool.
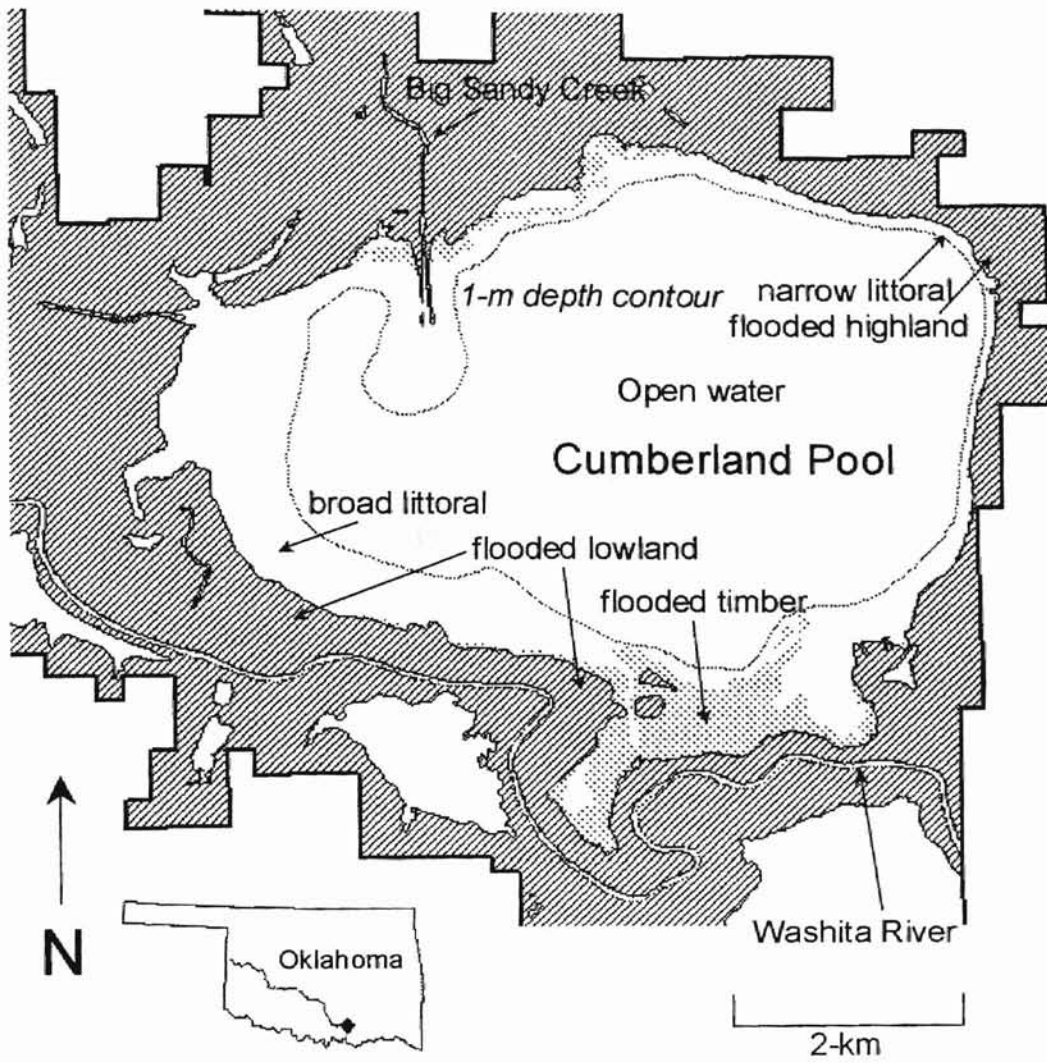
Figure 1

Figure 2

Figure 3

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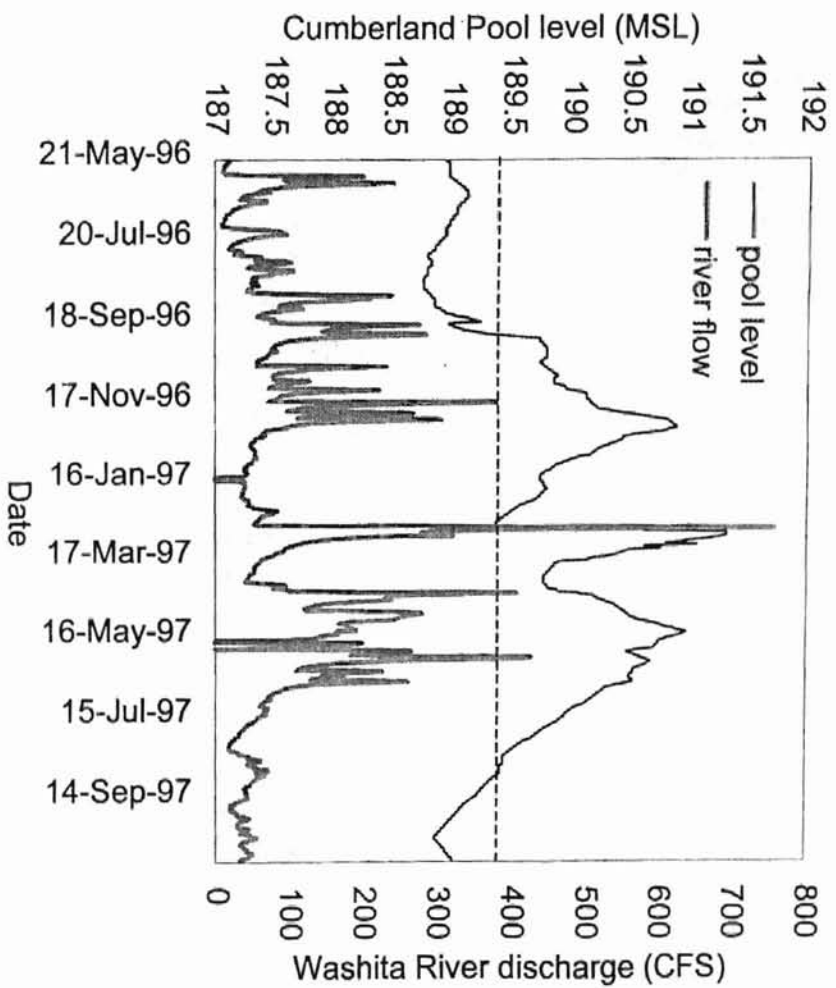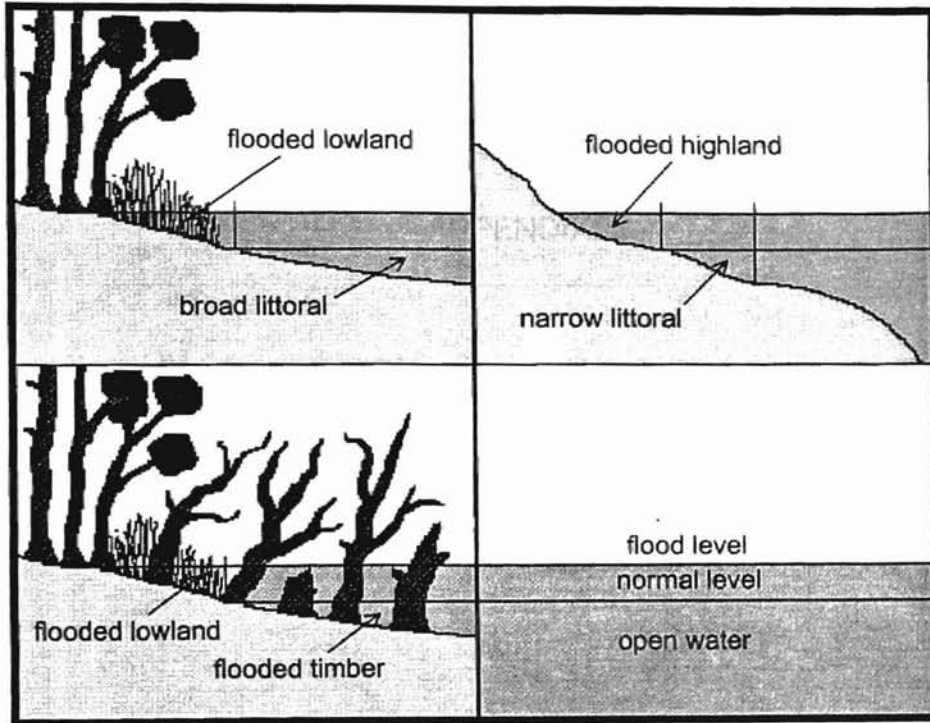
IV

APPENDIXES

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Appendix 1.--Total catch of species in experimental gill net samples by water body. Total sampling effort is given in Table 1 of chapter II and abbreviations for species and water bodies are in Tables 3 and 1 of chapter II.

Appendix 1.--Total catch of species in gill net samples by water body. Total sampling effort is provided in table 1 of chapter II and abbreviations for species and sites are in tables 3 and 1 of chapter II.

Species	Water bodies												
	BCG	BEL	BSL	CLP	GPP	LOL	MAP	MLE	MLW	PEN	RCA	RVR	TWP
BCF	0	4	0	130	3	0	0	1	0	1	1	0	0
BMB	0	9	0	48	4	0	5	1	3	0	0	4	0
CCF	1	2	7	59	1	4	0	0	2	9	9	11	3
CRP	1	0	1	20	3	2	0	1	0	4	2	0	1
FCF	0	1	0	4	0	0	0	0	0	0	0	0	0
FWD	0	1	0	28	0	0	0	0	0	0	0	0	0
GDE	0	0	0	34	0	0	0	1	0	0	0	0	0
GZS	13	8	10	960	17	10	0	10	4	19	34	16	30
ICT	0	5	11	125	8	1	1	1	1	5	19	1	1
LMB	0	0	1	0	0	0	0	0	0	0	0	0	0
LNG	0	0	0	119	0	0	0	0	0	2	0	0	0
RCS	0	1	5	42	1	0	0	0	4	2	1	0	0
SNG	0	1	1	64	0	0	0	0	0	0	0	1	0
SPG	1	0	0	10	1	0	0	0	0	3	5	0	6
STB	0	0	3	41	0	0	0	0	0	1	0	0	0
WHB	0	1	0	90	7	0	0	1	0	0	14	0	0
WHC	2	6	1	43	0	2	2	0	0	3	4	3	2
YBH	0	0	0	0	0	0	0	0	0	1	0	0	0

Appendix 1.--Total catch of species in gill net samples by water body. Total sampling effort is provided in table 1 of chapter II and abbreviations for species and sites are in tables 3 and 1 of chapter II.

Species	Water bodies												
	BCG	BEL	BSL	CLP	GPP	LOL	MAP	MLE	MLW	PEN	RCA	RVR	TWP
BCF	0	4	0	130	3	0	0	1	0	1	1	0	0
BMB	0	9	0	48	4	0	5	1	3	0	0	4	0
CCF	1	2	7	59	1	4	0	0	2	9	9	11	3
CRP	1	0	1	20	3	2	0	1	0	4	2	0	1
FCF	0	1	0	4	0	0	0	0	0	0	0	0	0
FWD	0	1	0	28	0	0	0	0	0	0	0	0	0
GDE	0	0	0	34	0	0	0	1	0	0	0	0	0
GZS	13	8	10	960	17	10	0	10	4	19	34	16	30
ICT	0	5	11	125	8	1	1	1	1	5	19	1	1
LMB	0	0	1	0	0	0	0	0	0	0	0	0	0
LNG	0	0	0	119	0	0	0	0	0	2	0	0	0
RCS	0	1	5	42	1	0	0	0	4	2	1	0	0
SNG	0	1	1	64	0	0	0	0	0	0	0	1	0
SPG	1	0	0	10	1	0	0	0	0	3	5	0	6
STB	0	0	3	41	0	0	0	0	0	1	0	0	0
WHB	0	1	0	90	7	0	0	1	0	0	14	0	0
WHC	2	6	1	43	0	2	2	0	0	3	4	3	2
YBH	0	0	0	0	0	0	0	0	0	1	0	0	0

Appendix 2.--Total catch of species in electrofishing samples by water body. Sampling effort is also given in Table 1 of chapter II and abbreviations for species and water bodies are in Tables 3 and 1 of chapter II.

Appendix 2.--Total catch of species in electrofishing samples by water body. Total sampling effort is provided in table 1 of chapter II and abbreviations for species and sites are in tables 3 and 1 of chapter II.

Species	Water bodies												
	BCG	BEL	BSL	CLP	DKP	GPP	LOL	MAP	MLW	PEN	RCA	RVR	TWP
BCF	0	0	0	0	0	0	0	0	0	0	0	0	0
BHM	1	0	2	2	0	9	2	0	0	1	0	0	5
BLG	47	14	38	0	10	2	24	40	84	44	22	166	17
BLP	2	0	1	0	0	1	1	0	14	19	0	25	5
BMB	0	2	0	1	0	0	1	2	1	0	0	4	0
BRS	0	0	0	0	0	0	0	2	0	0	0	0	0
BTS	0	0	6	10	0	0	0	0	0	5	1	0	0
CCF	2	1	1	6	0	1	1	0	0	1	2	0	0
CRP	0	0	1	0	2	0	2	1	0	3	1	0	1
FCF	0	1	0	1	0	0	0	0	0	0	1	0	0
FWD	3	0	7	14	0	4	0	0	0	4	48	0	2
GDR	0	0	0	0	1	0	0	0	0	0	0	0	0
GDS	0	0	0	0	0	0	0	1	0	0	0	1	2
GHS	0	0	0	8	0	4	0	0	0	1	1	0	0
GSF	0	0	1	0	0	0	0	0	0	0	0	2	0
GZS	16	123	34	43	0	34	8	4	3	66	220	28	103
ICT	6	21	13	11	0	4	17	13	8	9	4	5	0
LGS	9	4	9	0	0	0	0	1	16	14	7	8	0
LMB	3	1	5	2	1	1	4	9	4	3	7	14	2
LNG	0	0	1	1	0	0	0	1	0	3	1	0	0

Appendix 2.--Continued.

Species	Water bodies												
	BCG	BEL	BSL	CLP	DKP	GPP	LOL	MAP	MLW	PEN	RCA	RVR	TWP
MQF	0	1	0	0	1	2	0	0	2	1	0	1	0
OSS	12	0	2	5	0	3	0	0	24	39	0	0	4
RCS	0	7	3	7	0	0	0	1	0	34	3	0	1
RDS	0	0	2	0	0	0	0	0	0	5	0	0	0
RES	0	4	5	0	0	0	0	1	0	2	6	2	0
SLD	0	0	0	0	1	0	0	0	0	0	0	0	0
SNG	0	1	0	3	0	0	0	0	0	2	0	0	0
SPB	0	0	2	0	0	0	0	0	0	0	0	0	0
SPG	2	0	6	5	5	4	4	4	2	4	0	0	5
SPP	0	0	0	0	0	0	0	0	0	4	0	0	0
STB	0	0	0	0	0	0	0	0	0	0	0	0	0
TFS	0	0	6	57	0	0	0	0	0	0	0	0	0
TWS	4	26	40	168	0	3	2	2	0	4	53	54	14
WAM	0	15	4	0	27	0	6	13	18	20	10	16	1
WHB	0	3	2	24	0	14	0	0	0	2	4	5	0
WHC	10	11	2	15	1	3	5	1	20	9	3	6	8
Ameiurus	0	0	0	0	0	0	0	0	1	0	0	0	0
Atherinidae	4	26	40	168	0	3	2	4	0	4	53	100	14
Catostomidae	6	30	16	19	1	4	18	16	9	47	7	9	1
Cyprinella	0	0	8	10	0	0	0	0	0	10	1	0	0
Dorosoma	16	123	40	161	0	37	18	5	3	66	220	28	103

Appendix 2.--Concluded.

Species	Water bodies												
	BCG	BEL	BSL	CLP	DKP	GPP	LOL	MAP	MLW	PEN	RCA	RVR	TWP
Ictaluridae	2	2	1	7	0	1	1	1	0	3	4	0	0
Lepisosteus	2	1	7	9	5	4	4	6	2	10	1	2	5
Lepomis	68	105	70	6	37	6	30	60	142	129	243	200	24
Micropterus	3	1	7	2	1	1	4	11	4	5	7	14	2
Morone	0	3	2	24	0	14	0	0	0	2	4	5	0
Percina	2	7	2	0	0	1	1	1	14	19	3	25	5
Pimephales	1	2	17	3	0	9	2	0	0	1	52	8	5

Appendix 3.--Total catch of species in seine samples by water body. Total sampling effort is also given in Table 1 of chapter II and abbreviations for species and sites are in Tables 3 and 1 of chapter II.

Appendix 3.--Total catch of species in seine samples by water body. Total
sampling effort is provided in table 1 of chapter II and abbreviations for species
and sites are in tables 3 and 1 of chapter II.

Species	Water bodies				
	BSU	CLP	DKP	PEN	WAS
BCF	0	0	0	0	12
BHM	0	701	0	113	33
BLG	6	0	4	0	0
BLP	2	27	0	0	1
BNM	0	1	0	0	0
BTS	14	431	0	34	2
CCF	0	18	0	2	14
DKD	20	0	0	0	0
EMS	0	0	0	0	5
FCF	0	0	0	0	2
FHM	0	1	0	0	0
FWD	0	429	0	0	11
GDR	1	0	0	0	0
GDS	0	0	1	0	0
GHS	0	828	0	0	4
GSF	1	0	0	0	0
GZS	0	2,962	0	0	100
LGS	21	0	0	0	0
MMS	0	412	0	0	0
MQF	260	161	8	0	55
OSS	0	218	0	0	17
OTD	39	0	0	0	0
PLK	8	0	0	0	0
RCS	0	19	0	0	69
RDS	2	254	0	2	144
SDS	47	10	0	0	10
SLD	0	0	3	0	0
SMM	2	4	0	0	0
SNG	0	2	0	0	0
SPB	18	0	0	0	0
SPC	0	2	0	0	95
STR	48	10	0	0	0
SVC	0	0	0	0	1

Appendix 3.--Continued.

Species	Water bodies				
	BSU	CLP	DKP	PEN	WAS
TFS	0	1,233	0	0	231
TWS	0	34,446	0	4	445
WAM	1	0	4	0	0
WHB	0	83	0	1	0
WHC	0	10	0	0	0
YBH	2	0	0	0	0
Ameiurus	2	0	0	0	0
Atherinidae	0	34,446	0	4	445
Catostomid	1	20	0	2	133
Cyprinella	16	711	0	36	146
Dorosoma	0	4,546	0	0	345
Ictaluridae	0	22	0	2	48
Lepisosteus	0	2	0	0	0
Lepomis	29	218	8	21	17
Micropterus	107	7	0	1	2
Morone	0	83	0	1	0
Percina	22	27	0	0	1
Pimephales	0	1,027	0	113	33

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VITA

William Scott Chappell

Candidate for the Degree of Master of Science

Thesis: HABITAT ASSOCIATIONS OF FISH ASSEMBLAGES ON THE
TISHOMINGO NATIONAL WILDLIFE REFUGE

Major Field: Wildlife and Fisheries Ecology

Biography:

    Personal Data: Born in New Port News, Virginia, October 25, 1972, the
son of William L. and Sharon R. Chappell.

    Education: Graduated from Perquimans County High School, Hertford,
North Carolina, in June 1991; received Bachelor of Science
Degree in Wildlife and Fisheries Science from North Carolina State
University in December 1995; completed the requirements for the
Master of Science Degree at Oklahoma State University in
December 1998.

    Pofessional Experience: Field and lab assistant for Department of
Zoology faculty at North Carolina State University from May, 1994
till December, 1995. Graduate research assistant for the
Oklahoma Cooperative Fish and Wildlife Research Unit from
January, 1996 til July, 1998

    Organizational Memberships:
        American Fisheries Society, The Oklahoma Chapter of the
American Fisheries Soceity